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

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EXECUTIVE SUMMARY

This study is to estimate nitrogen loads from removed septic systems to surface water bodies in the City of Port St. Lucie, City of Stuart, and Martin County located in the St. Lucie River and Estuary Basin. The load estimates can be used to calculate credit for septic tank phase out projects in support of the on-going Basin Management Action Plan (BMAP). The ArcGIS-based Nitrate Load Estimation Toolkit (ArcNLET), which was developed for the Florida Department of Environmental Protection (FDEP) by the Florida State University (FSU), is used for the nitrogen load estimation. While ArcNLET is based on a simplified model of groundwater flow and nitrogen transport, the model considers heterogeneous hydraulic conductivity and porosity as well as spatial variability of septic system locations, surface water bodies, and distances between septic systems and surface water bodies. ArcNLET also considers key mechanisms controlling nitrogen transport, i.e., advection, dispersion, and denitrification. After preparing model input files (e.g., raster file of hydraulic conductivity) in the ArcGIS format, setting up an ArcNLET model run is easy through a graphic user interface. The modeling results are readily available for post-processing and visualization within ArcGIS. The modeling results include groundwater flow paths from septic systems to surface water bodies, spatial distribution of nitrogen plumes, and nitrogen load estimates to individual surface water bodies; these results can be used directly for environmental management and regulation of nitrogen pollution.

The ArcNLET flow and transport models of this study are established using data downloaded from public-domain websites (e.g., the website of U.S. Geological Survey (USGS) for Digital Elevation Model (DEM) and National Hydrography Database (NHD)) and data provided by colleagues from FDEP and the cities and county (e.g., ArcGIS files of canals in Port St. County and septic system locations in the modeling areas). The flow model is calibrated by adjusting the smoothing factor (a model parameter) to match the shapes of smoothed DEM and water table. The transport model is calibrated by adjusting transport parameters (i.e., source plane concentration, C_0, longitudinal dispersivity, α_L, horizontal transverse dispersivity, α_{TH}, and first-order decay coefficient of denitrification, k) to match simulated and observed nitrogen concentrations (the limited observations are historical data compiled from USGS websites). The calibrated parameter values are listed for in Table ES-1.

Table ES-1. Calibrated values of ArcNLET model parameters for all the sites. Calibration of transport parameters for the City of Stuart and calibration of smoothing factor for Martin County are not conducted due to lack of data. The calibrated transport parameters for Martin County are used for the City of Stuart, and the calibrated smoothing factor of the City of Port St. Lucie and City of Stuart Cities is used for Martin County.

Parameter	City of Port St. Lucie	City of Stuart	Martin County
Smoothing factor	40	40	-
C_0 (mg/L)	40	-	40
α_L (m)	60	-	35
α_{TH} (m)	1.6	-	1.1
k (d^{-1})	0.0011	-	0.001

The calibrated ArcNLET models are used to simulate nitrogen plumes and estimate nitrogen load from the removed septic systems to surface water bodies. The simulated nitrogen plumes and load estimates exhibit substantial spatial variability, which manifests the importance of considering spatial variability in the load estimation. In the City of Port St. Lucie, the canals are critical to control groundwater flow paths and loads, because groundwater from most septic systems discharges to the canals instead of to the St. Lucie River, as shown in Figure ES-1. Since the canals are distributed over the entire modeling area, effective management of nitrogen pollution should be conducted over the entire modeling area. Figure ES-2 shows that the load estimates are strongly correlated with nitrogen concentrations in surface water quality data, suggesting that septic load is a significant factor for water quality deterioration. In the City of Stuart and Martin County, because the areas with removed septic systems are of a smaller scale, it happens often that majority of the load is to one or two surface water bodies. For example, at Seagate Harbor of Martin County, 99% of the load is to water body 17, as shown in Figure ES-3.

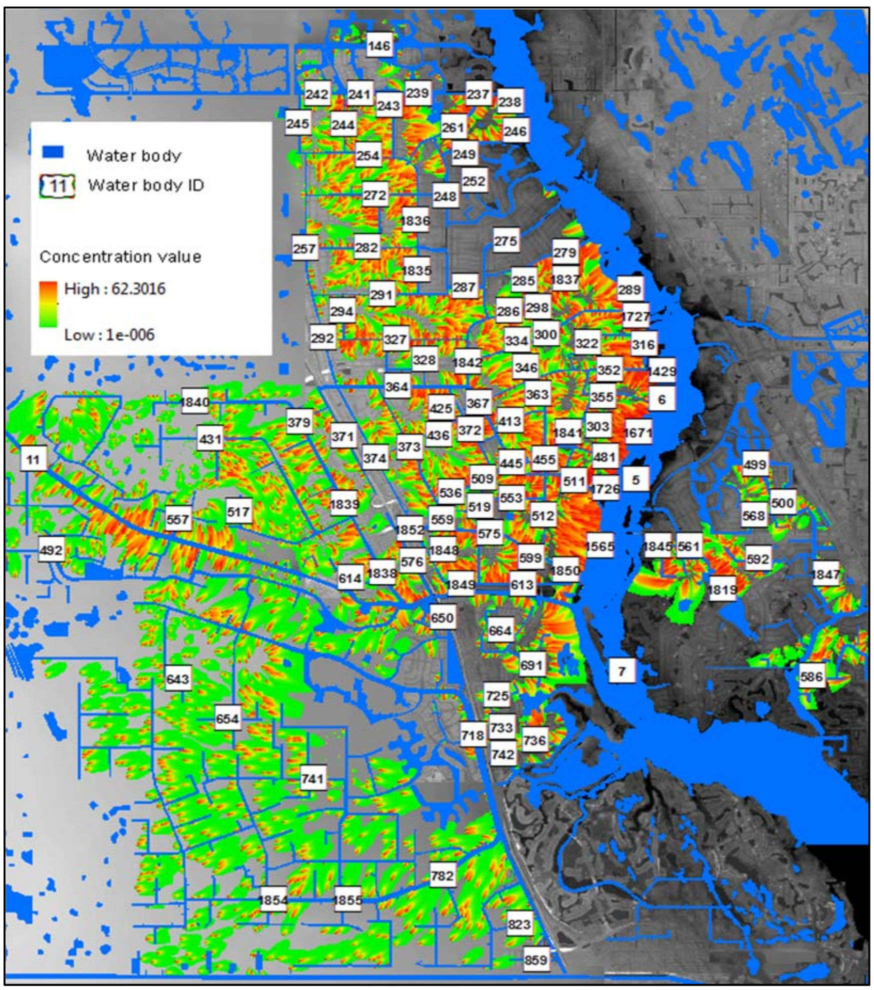

Figure ES-1. Simulated nitrogen plumes from removed septic systems in the City of Port St. Lucie. The FIDs of water bodies with estimated load larger than 0.05 kg/d are labeled. The loads are mainly to the canals that are distributed over the entire modeling area.

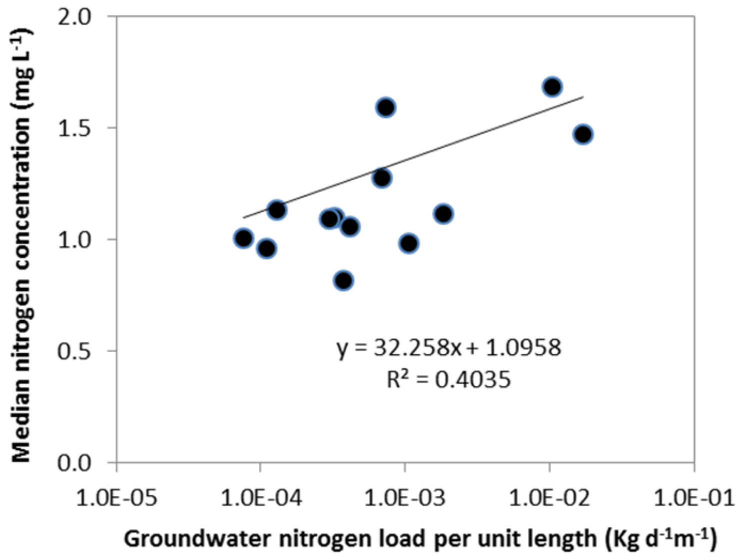

Figure ES-2. Median surface water nitrogen concentrations and estimated groundwater nitrogen load per unit length along the fourteen surface water bodies where the surface water nitrogen concentrations were measured. The x-axis is in the logarithm scale. The correlation between the two variables suggests a direct impact of septic load to surface water quality

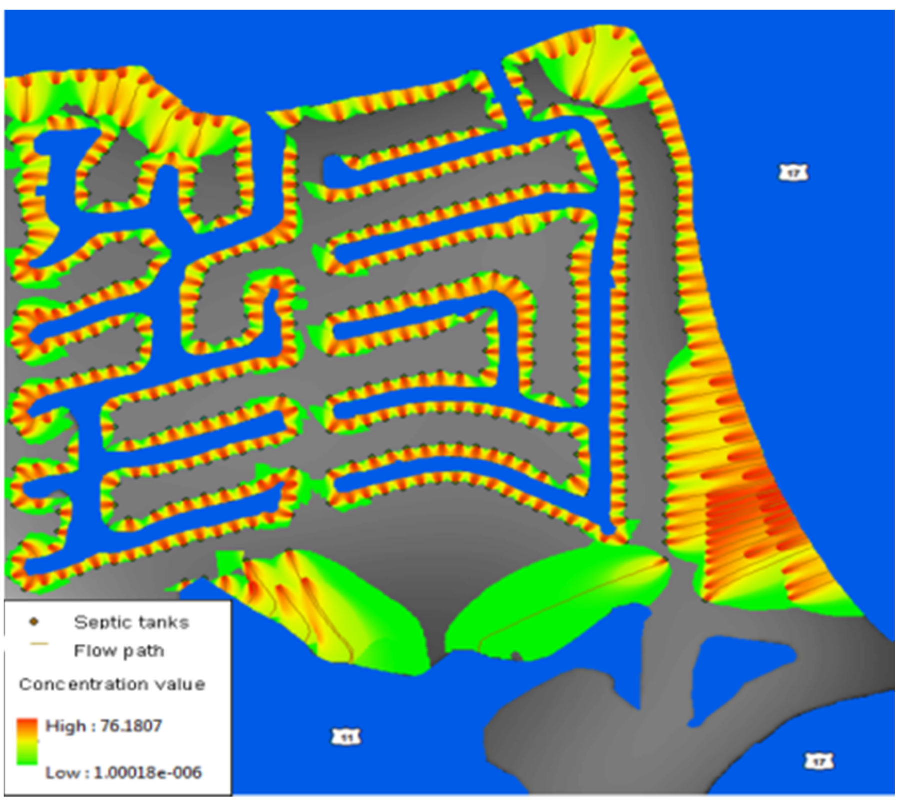

Figure ES-3. Simulated flow paths and nitrogen plumes from the removed septic systems in Seagate Harbor. The flow paths are the shortest among the seven modeling sites.

Table ES-2. ArcNLET estimated total load, number of removed septic systems, load per septic system, and nitrogen reduction ratio per septic system at the City of Port St. Lucie, City of Stuart, and five sites of Martin County.

	City of Port St. Lucie	City of Stuart	Martin County				
			North River Shores	Seagate Harbor	Banner Lake	Rio	Hobe Sound
Total Load (kg/d)	42.48	1.665	8.346	9.255	0.856	0.317	0.346
Number of Septic Systems	5592	146	411	451	105	66	51
Load per Septic System (g/d)	7.60	11.40	20.31	20.52	8.15	4.80	6.78
Nitrogen Reduction Ratio (%)	67.0	50.4	11.7	10.8	64.6	79.1	70.5

Table ES-2 lists the ArcNLET estimated load from all the septic systems and the load per septic system to the surface water bodies. It is found in this study that the amount of load is controlled by the following physical factors: length of flow paths, flow velocity, and drainage condition. Figure ES-4 shows that the load estimate decreases with the mean length of flow paths; the two largest loads per septic system are for North River Shores and Seagate Harbor where the flow paths are the shortest (see Figure ES-3 for Seagate Harbor). This is reasonable because longer flow paths result in more denitrification and thus smaller load estimate. In line with this, larger flow velocity corresponds to shorter travel time and thus smaller amount of denitrification and larger amount of load, as shown in Figure ES-5. Figures ES-4 and ES-5 indicate that the setback distance should be determined not only by the distance between septic systems to surface water bodies but also by groundwater flow conditions (the distance probably plays a more important role here). The groundwater flow conditions are closely related to soil drainage conditions at the modeling sites. Figure ES-6 shows that, in the Port St. Lucie site, the load estimate increases when the drainage condition changes from very poorly drained to excessively drained.

Given that the input load from each septic system to groundwater is 23 g/d, the nitrogen reduction ratios are calculated as (load to groundwater – load to surface water) / (load to groundwater). The ratios listed in Table ES-2 are comparable with literature data, i.e., 70.0% in Roeder (2008), 57.1% in Vaiela et al. (1997), and 65 – 85% in Meile et al. (2010).

The ArcNLET estimated load per septic system is smaller than that of 31 g/d obtained using a method considered by Martin County (Dianne Hughes, 2013, Personal Communication). While the load of 31g/d is close to the average load of 32.9 g/d to an individual septic system (4.8 kg/yr (from a review article of Valiela et al. (1997)) × 2.5 people/house in St. Lucie and Martin Counties), it ignores nitrogen loss in septic systems, drain fields, and during transport in aquifers. For example, a report of MACTEC (2007) for the Wekiva study conducted by the Florida Department of Health suggests that about 70% nitrogen is lost in septic tanks and leaching fields. Therefore, the loads estimated by Martin County are larger than those of this study obtained using ArcNLET that considered the loss in septic tanks, leaching fields, and aquifers.

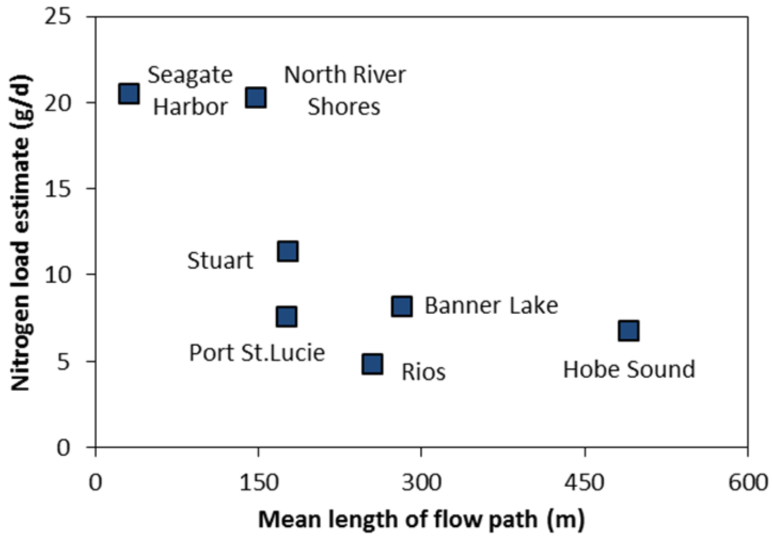

Figure ES-4. Variation of nitrogen load estimate per septic systems with mean lengths of flow paths in the seven sites of this study.

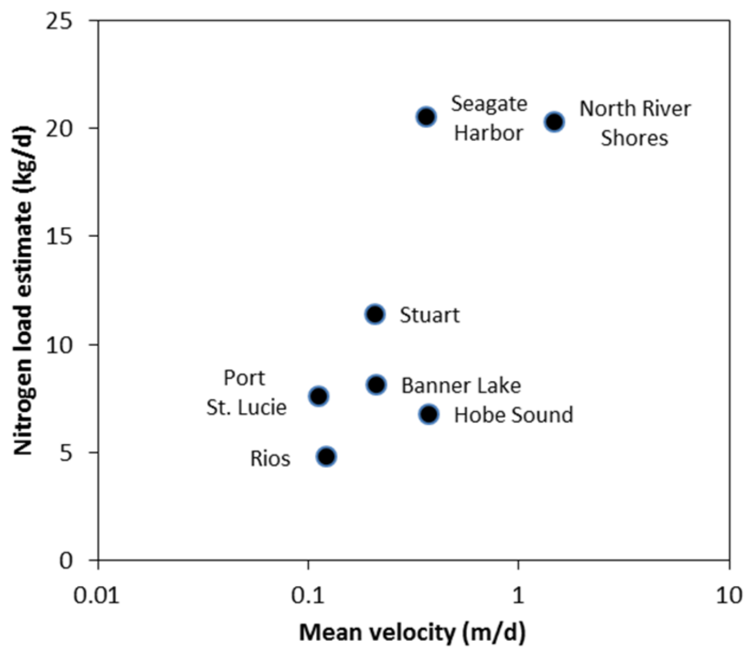

Figure ES-5. Variation of nitrogen load estimate per septic systems with mean velocity in the seven sites of this study.

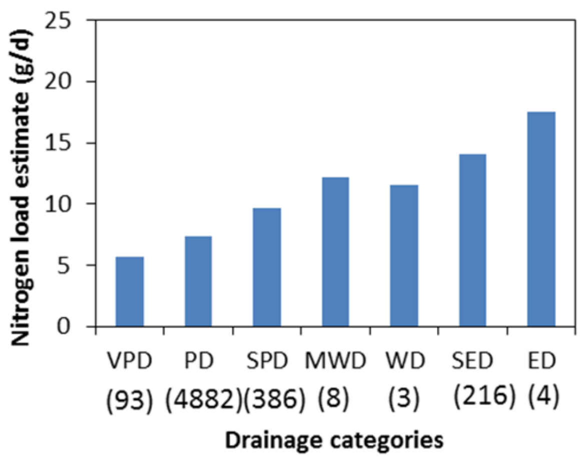

Figure ES-6. Variation of nitrogen load estimate per septic systems with drainage conditions of the soil zones where septic systems are located at the Port St. Lucie site. Abbreviations of the drainage conditions are as follows: excessively drained (ED), somewhat excessively drained (SED), well drained (WD), moderately well drained (MWD), somewhat poorly drained (SPD), poorly drained (PD), and very poorly drained (VPD). The number of septic systems corresponding to each drainage condition is given in the parentheses.

The load estimates of this study are discussed in the BMAP context to evaluate significance of the removal to BMAP nitrogen pollution management. This is done by calculating the percentages of nitrogen load from the removed septic systems to the BMAP estimated total load given in the draft BMAP (2013); several assumptions are invoked in the calculation as listed in the report. As shown in Table ES-3, the largest percentage is 31.20% for the North Fork sub-basin, followed by the second largest of 22.87% for the Basin 4-5-6. These numbers appear to be reasonable, considering the absolutely large number of septic systems in North Fork and the relatively large numbers of septic systems in Basin 4-5-6. The percentages are negligible for the C-23 and C44/S-135 sub-basins, which is not unreasonable because of the small number septic systems in the two sub-basins. The percentage is 10.33% on average for South Fork, which seems to be reasonable given the number of septic systems in the sub-basin. Note that a recent study (USEPA, 2013) indicates that septic systems contribute approximated 5% of the total nitrogen load in the Chesapeake Bay watershed. Although the South Coastal sub-basin is not included in the draft BMAP (2013), the ArcNLET modeling suggests that the load from septic systems is expected to be significant in this sub-basin.

Table ES-3 also lists the percentages of the estimated load from the removed septic systems to the BMAP required load reduction. A scenario analysis is further conducted to estimate the amount of nitrogen load reduction when functioning septic systems are further removed in the St. Lucie River and Estuary Basin. The percentages of load reduction (due to the actual and hypothetical removal) to the BMAP required load reduction are also listed in Table ES-3. The results suggest that the actual and hypothetical removal are worthy for the North Fork and Basin 4-5-6 sub-basins, because the actual and hypothetical removal together can achieve more than 80% of the required nitrogen load reduction. While the effort of septic removal is also worthy for the South Fork sub-basin, the effort of removing septic systems does not help reduce nitrogen load for the C-24, C-23 and C-44/S-135 sub-basins. These exercises may be helpful for using ArcNLET to facilitate nitrogen pollution management.

Table ES-3. In a scenario analysis that all septic systems are removed, percentages of nitrogen load from removed and functioning septic systems to the BMAP estimated total load and percentages of load reduction to the BMAP required load reduction.

	Basin 4-5-6	**C-23**	**C-24**	**C-44/ S-153**	**North Fork**	**South Fork**
Percentage of nitrogen load from septic systems to BMAP estimated load	22.87%	0.03%	1.66%	0.00%	31.20%	10.33%
Percentage of load reduction of removed septic systems to BMAP required reduction	33.67%	0.05%	1.71%	0.00%	17.02%	1.35%
Percentage of load reduction to BMAP required reduction	81.02%	0.06%	3.25%	0.00%	85.75%	25.76%

The ArcNLET modeling results are subject to the following limitations:

(1) The simplified flow and transport models of ArcNLET may not be able to sufficiently simulate groundwater flow and nitrogen reactive transport in the St. Lucie River and Estuary Basin. For example, impacts of hurricane and salt water intrusion are not considered in the ArcNLET modeling. Nitrogen load due to storm events is not considered in the ArcNLET modeling either.

(2) While ammonium and TKN concentrations are higher than NO_x (including both nitrite and nitrate) concentrations at the modeling areas, the current version of ArcNLET does not explicitly consider transport of ammonium and organic nitrogen. Instead, ArcNLET assumes that their transport is the same as the nitrate transport so that ArcNLET (developed for nitrate transport modeling) can be used to simulate the concentrations of total nitrogen (TN) or dissolved inorganic nitrogen (DIN). The calibration target is the concentrations of total nitrogen (TN) or dissolved inorganic nitrogen (DIN) (depending on availability of TKN concentrations), not nitrate concentrations. This may overestimate ammonium and/or organic nitrogen concentrations due to disregarding adsorption, nitrification, and other possible reactions.

(3) For the groundwater flow and transport processes considered in ArcNLET, they may not be fully characterized by the limited calibrated data available to this study. As a result, there may be a large number of parameter sets that can simulate equally well the observed hydraulic heads and nitrogen concentrations. In other words, parametric uncertainty is substantial in this modeling study. As a result, the load estimates are also uncertain.

(4) In the scenario analysis in which functioning septic systems are removed, the ArcNLET estimated load per septic system obtained for the septic system removal

areas is extrapolated to the entire sub-basins. The extrapolation may give inaccurate results.

While resolving the first problem is beyond the scope of ArcNLET that is developed as a groundwater model, the second problem can be resolved by adding a module of ammonium transport within ArcNLET, which is on-going. Resolving the fourth problem is theoretically straightforward by conducting the model calibration and model simulation for the functioning septic systems, which however is beyond the scope of this study.

To address the third problem of uncertainty above, a Monte Carlo (MC) simulation is conducted using the recently developed MC function of ArcNLET (Rios et al., 2012b). Instead of providing a single deterministic load estimate obtained from the calibrated model, the MC simulation gives multiple values of nitrogen concentration at user specified locations (monitoring points) and nitrogen load estimate to the water bodies involved in the modeling. These values represent ArcNLET predictive uncertainty due to parametric uncertainty. The MC simulation is conducted for three sites in Martin County: the calibration site where ArcNLET is calibrated against nitrogen concentration at a monitoring well, Seagate Harbor where the load estimate is high, and Hobe Sound where the load estimate is low. Figure ES-7 plots the histogram of simulated nitrogen concentration at the monitoring well of the calibration site. The histogram indicates that, with the parameter distributions considered in this study, it significantly more likely for the model to simulate low concentration values than to high values at the monitoring point. This is consistent with the low nitrogen concentration of 0.29 mg/L observed at the monitoring well, suggesting that the calibrated model is likely to reflect nitrogen transport at the calibration site. Figure ES-8 plots the relation between the load estimate and the simulated concentration at the monitoring well. The overall positive correlation indicates that larger nitrogen concentration corresponds to larger load. In the context of site monitoring, if higher concentrations are continuously observed at the monitoring well, the load estimate should be larger than the deterministic estimate. The same relation is also observed for the Seagate Harbor and Hobe Sound Site. However, at Seagate Harbor, the increase of load estimate from the deterministic estimate is limited because the deterministic estimate is already relatively large. At Hobe Sound, while the increase of load estimate can be substantial relatively to the deterministic estimate, the maximum load estimate obtained from the MC simulation is still smaller than that of the other sites. In this sense, having more monitoring data does not necessarily lead to substantial increase of load estimate. It is also possible that collecting more data leads to decrease of the load estimate. For example, if observed nitrogen concentrations are smaller than the deterministic simulation of the calibration model, the corresponding load estimate may be smaller than the deterministic estimate, as shown in Figure ES-8.

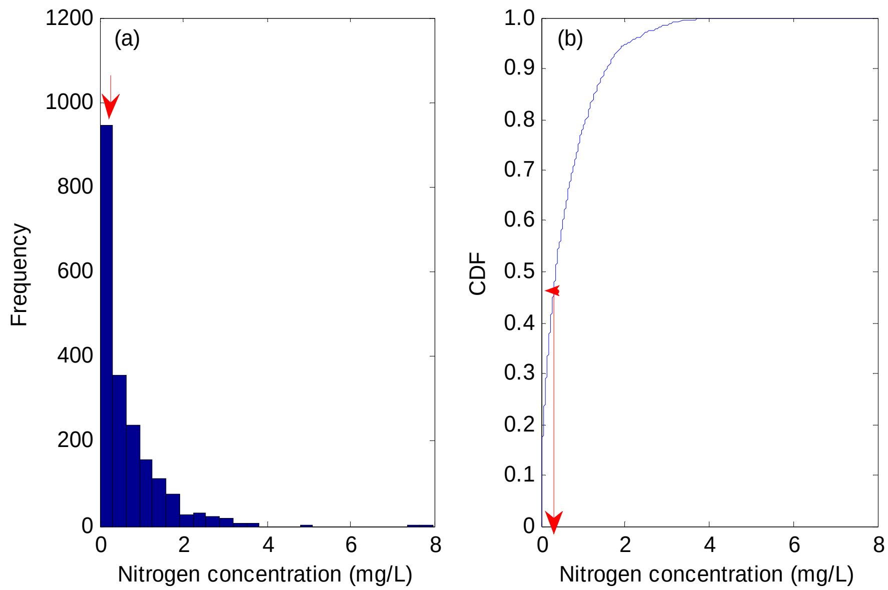

Figure ES-7. (a) Histograms and (b) cumulative distribution function (CDF) of 2,000 realizations of simulated nitrogen concentration at the monitoring well of the calibration site in Martin County.

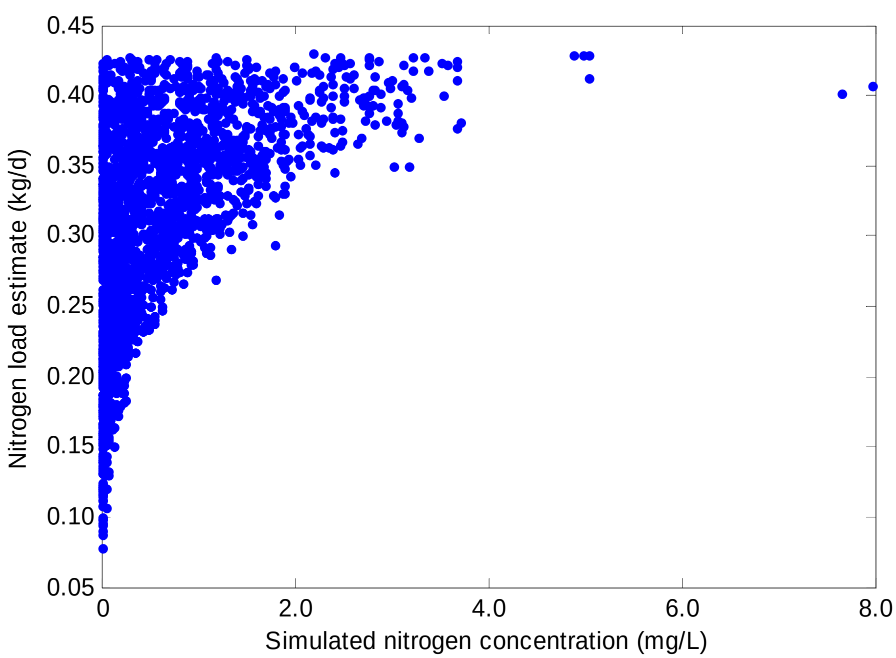

Figure ES-8. Relation between nitrogen load estimate and concentration simulation over the 2,000 MC realizations at the monitoring well of the calibration site in Martin County.

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1. INTRODUCTION

This study is aimed at supporting management of coastal nitrogen pollution, which is considered as the most prevalent and challenging pollution problem currently facing U.S. coastal waters (National Research Council, 2000). The pollution has resulted in serious environmental, ecological, economical, and human health problems, such as groundwater contamination, methemoglobinemia (also known as blue baby syndrome), eutrophication, fish kills, harmful algal growth, and some shellfish poisoning (Walters, 1983; National Research Council, 2000; Howarth, 2008). In the continental United States, moderate to severe degradation of water quality due to nitrogen and phosphorus pollution has been reported in more than 60 percent of coastal rivers and bays (Howarth et al., 2000). Among the various nitrogen sources (e.g., atmospheric deposition, fertilizer use, and wastewater discharge), an important source of nitrogen in the environment, especially in highly populated coastal areas, is due to wastewater treatment using onsite sewage treatment and disposal systems (OSTDS) (a.k.a., septic systems). According to U.S. Environmental Protection Agency (USEPA, 2003), approximately 25% of the homes and 33% of new developments in the United States utilize septic systems; in the state of Florida, nearly one-third of households use septic systems (Ursin and Roeder 2008). In the Chesapeake Bay watershed that includes multiple states, 25% of the population utilizes septic systems (Maizel et al., 1997). Valiela and Costa (1988) showed that, in Buttermilk Bay, MA, 40% of nitrogen and phosphorous entering watershed was from septic systems and 80% of the nitrogen was transported through groundwater. In the Waquoit Bay, MA, 48% nitrogen load was from septic systems (Valiela et al., 1997). In the study of Kroeger et al. (2006) at Cape Cod, wastewater from septic systems is considered as the principal source of nitrogen entering estuaries from urbanized or suburbanized watersheds. For the Chesapeake Bay watershed of larger scale, a recent study (USEPA, 2013) indicates that septic systems contribute approximated 8.3 million pounds to the Bay, about 5% of the total nitrogen load. While this is not the largest source of nitrogen pollution to the Bay, it is important to reduce the load from septic systems in the effort to improve water quality. Given the trends in population growth, nitrogen loads from septic systems are expected to increase. Therefore, sustainable decision-making and management of nitrogen pollution due to septic systems are urgently needed.

The site of interest to this study is the St. Lucy River and Estuary Basin (Figure 1-1) located in Martin, St. Lucie, and Okeechobee Counties in southeast Florida. It is a major tributary to the Southern Indian River Lagoon, where the ecological and biological integrity has deteriorated in the last several decades due to the decline in water quality caused in part by nitrogen pollution (Sigua et al., 2000). Nitrogen load from septic systems is an important reason to nitrogen pollution. A study of Sigua and Tweedale (2003) showed that, at the Indian River Lagoon, the load from groundwater seepage is 84,920 kg/year, about 8% to the overall nitrogen loading, the second largest source after the agricultural/urban runoff which contributes 79% of the load. In the groundwater load, a large portion is expected to be from septic systems, especially in areas with high population density such as the City of Port St. Lucie, the City of Stuart, and Martin County. In the St. Lucie River and Estuary Basin, nitrogen pollution from septic systems is evidenced by elevated concentrations of fecal coliform bacteria (Lapointe et al., 2012). Belanger et al. (2007) pointed out that the high seepage rate may imply high load of nitrogen and other pollutants to the Indian River Lagoon; at the St. Lucie Estuary, field data suggest high nitrogen load from groundwater is partly attributed to the septic systems.

Figure 1-1. Location of St. Lucie River and Estuary in Florida, USA.

The Florida Department of Environmental Protection has adopted Total Maximum Daily Loads to reduce the watershed nutrient inputs to the St. Lucie River and Estuary. In support of the TMDL implementation, the Basin Management Action Plan (BMAP, 2013) has been developed. The BMAP includes various management means, and one of them is to convert septic into sewage. In the City of Port St. Lucie, the City of Stuart, and Martin County (Figure 1-1), sewer line has been built, and new developments and existing houses with failed septic systems are required to use sewage. However, nitrogen load reduction due to the septic system removal is still unknown. Quantifying the reduction is important for planning and controlling of wastewater nitrogen loading, as well as to nitrogen trading and offset program described in U.S. EPA (2013), which can help minimize overall cost of TMDL implementation.

There have been no well-accepted techniques to estimate nitrogen loads from septic systems (removed or functioning). In certain methods, load estimates are given from interpolation and extrapolation of field measurements, such as in the studies of Reay (2004) at Chesapeake Bay and Sigua and Tweedale (2004) at Indian River Lagoon. While modeling approaches are more popular and practical, the models of load estimation range from simple arithmetic calculation based on empirical rules to sophisticated evaluation based on state-of-the-art understanding of physical, chemical, and biological processes and their interactions involved in nitrogen transport. In the Nitrogen Load Model (NLM) of Valiela et al. (1997, 2000), the load from septic systems is evaluated in a simple way as: nitrogen released per person per year × people/house × number of houses × 60% not lost in septic tanks and leaching fields × 66% not lost in plumes × 65% not lost in aquifer. The coefficients were derived from literature, limited filed data, and/or best engineering judgment. This NLM-kind modeling method has been used widely. For example, Vadeboncoeur et al. (2010), Giordano et al. (2011), and Kinney and Valiela (2011) estimated nitrogen load to Narragansett Bay, Virginia Lagoon, and Great South Bay, respectively, by multiplying literature-derived load coefficients to various nitrogen sources. More such models can be found in the NLOAD software developed by Bowen et al. (2007), an interactive, web-based modeling tool for nitrogen management in estuaries While the NLM model yields satisfactory results in comparison with field measurements (Valiela et al., 2000), justification of using the coefficients for new sites remains challenging, especially when there is no measurement of nitrogen loads to evaluate accuracy of the coefficients. In addition, the coefficients do not explicitly consider spatial variability of nitrogen concentrations and cannot reflect heterogeneity of hydrogeologicl properties (e.g., hydraulic conductivity) and distance of septic systems to surface water bodies. The variability cannot be ignored. For example, Sigua and Tweedale (2004) and Lapointe et al. (2012) showed that spatial variability in nitrogen concentration is large in the St. Lucie River and Estuary Basin. For example, Sigua and Tweedale (2004) reported that the nitrogen concentration in the central Indian River Lagoon can be twice as large as that in the northern Lagoon. As a result, the estimated nitrogen loads using the NLM-type methods may not be sufficient for providing site-specific information for effective management of nitrogen pollution and implementation of TMDL.

In the wide spectrum of nitrate models, one extreme is to consider, to the extent possible, all biohydrogeochemical processes involved in nitrate/nitrogen fate and transport. For example, Maggi et al. (2008) developed a numerical code, TOUGHREACT-N, one of the most sophisticated code to date, for simulating coupled processes of advective and diffusive nutrient transport, multiple microbial biomass dynamics, and equilibrium and kinetic chemical reactions in soil and groundwater. While this model has the potential of being applied for coastal areas, the applications have not been reported in literature. Instead, simplified models have been used. Meile et al. (2010) and Porubsky et al. (2011) developed a two-dimensional model that numerically solves the advection-dispersion equation coupled with the reaction network encompassing reactions of sorption-desorption, nitrification, denitrification, and dissimilatory nitrate reduction to ammonium. A similar simplified model can be found in Spiteri et al. (2008) with focus on modeling biogeochemical processes for simulating nutrients in submarine groundwater discharge. These complex models can handle spatial variability of hydrogeologic properties and distance of septic systems to surface water bodies, and may yield results that can potentially agree well with field observations. However their complexity may be a hurdle for general users to set up the models; a trained professional is always required to operate the models and interpret modeling results for decision-makers of environmental management. In addition, to utilize sophisticated functions of the models, a large amount of model input and calibration data as well as long execution time may be needed, which may not be available or affordable in practice. For many management projects of nitrate transport modeling and load estimation, including those related to environmental regulation such as total maximum daily load (TMDL), it may not be practical to use the complex models. Therefore, alternative modeling methods are needed.

Models based on Geographic Information System (GIS) have gained popularity in environmental modeling (National Research Council, 2010). GIS-based screening models are one of the six primary types of models to facilitate modeling assessment and decision-making associated with pollutants from septic systems (McCray et al., 2009). GIS is an efficient tool to integrate regional/local spatial characteristics (e.g., digital hydrologic and topographic data) of a system. GIS-based models here are not GIS tools simply for preparing model input files and post-processing output files of other modeling programs. Instead, the GIS-based models provide a modeling environment for simulating quantities of interest and for analyzing and visualizing modeling results by non-technical citizens. A number of GIS-based modeling software have been developed, including PRO-GRADE (Lin et al. 2009) and uWATER-PA (Yang and Lin 2011; Rios et al., 2011a) for groundwater problems, TNT2 (Beaujouan et al. 2002) and GWLF (Romshoo and Muslim, 2011) for nitrogen transport and load estimation, and WARMF (Herr et al. 2001) for calculation of TMDLs for most conventional pollutants at the watershed scale. The GIS-based models and software always have a simple conceptual model and use computationally inexpensive methods that can be implemented within GIS. For example, Becker and Jiang (2007) developed a GIS-based groundwater contaminant transport model using the analytic element method. Focusing on nitrate pollution from agricultural sources, Schilling and Wolter (2007) developed a GIS-based model to estimate groundwater travel time using DEM data. The GIS-based models can be viewed as a compromise between the rule-based models (e.g., NLM) and the sophisticated numerical models (e.g. TOUGHREACT-N), with the attractive features of incorporating site-specific information (e.g., DEM), considering spatial variability of hydrogeologic properties, and simulating hydrogeochemical processes involved in nitrogen transport. There are other reasons that render GIS-based models suitable to environmental management. For example, skills required for applying GIS-based models are widely available; the models are easy to set up and computationally efficient to execute; the modeling process is transparent; the modeling results are readily available to interpret and visualize within GIS; and the results are quantitative and can be used directly for environmental management and regulation.

The ArcGIS-Based Nitrate Load Estimation Toolkit (ArcNLET, Rios et al. 2013a; Wang et al., 2013) is used in this study to simulate nitrate transport in surficial aquifers due to septic systems and to estimate corresponding nitrate load to surface water bodies. ArcNLET considers advection, hydrodynamic dispersion, and denitrification processes involved in nitrate transport. It also incorporates heterogeneous hydraulic parameters that vary by raster elements and considers spatial variation of locations and loads from individual septic systems. In addition, as shown in Wang et al. (2013), ArcNLET is able to simulate field observations at specific sites. However, similar to other GIS-based models, ArcNLET relies on a simple conceptual model involving a number of assumptions and simplifications, and uses an analytical equation to describe nitrogen transport. Therefore, ArcNLET should be used as a screening model to provide quick estimates, especially when field data are insufficient to calibrate parameters that are needed to simulate groundwater flow and nitrogen transport.

ArcNLET is used in this study to simulate nitrate load from removed septic systems at seven sites in the City of Port St. Lucie, the City of Stuart, and Martin County to surface water bodies in the St. Lucie River and Estuary Basin. The load estimates can be used directly to support the on-going TMDL implementation in the coastal estuary. ArcNLET is used to simulate the loads from the individual sites. For the sites (e.g., the City of Port St. Lucie and Martin County) where monitoring data of hydraulic head and nitrogen concentration are available, model calibration is conducted in the trial-and-error manner to estimate values of model parameters such as dispersivity and denitrification coefficient. When monitoring data are not available (e.g., in the City of Stuart), the model parameter values from the calibrated sites are used. The simulated results are evaluated by comparing them with literature data and those obtained using other methods. The results are also discussed in the context of Basin Management Action Plan (BMAP) to evaluate the relative contribution of the septic-related load to the total nitrogen load; a scenario analysis is conducted to estimate the amount of nitrogen load reduction if functioning septic systems are converted to sewer.

Measured nitrogen concentrations in the St. Lucie River and Estuary Basin are different from those in Jacksonville, FL., in that concentrations of ammonimum and TKN are higher than those of nitrate, which may be attributed to incomplete nitrification process in the vadose zone. This poses a challenge to ArcNLET modeling, since it is developed for nitrate transport modeling. While developing a new ArcNLET version to simulate ammonium transport is warranted in a future study, it is not attempted in this study. Instead, the calibration target is the concentration of total nitrogen or dissolved inorganic nitrogen (DIN) (including ammonium, nitrite, and nitrate), depending on availability of ammonium or TKN concentrations. This is tantamount to assuming that the transport mechanisums of ammonimum and/or organic nitrogen are the same as those of nitrate, which diregards the processes of nitrification, adsorption, and other reactions. Their effects on accuracy of the load estimates have not been investated but are not expected to be significant.

The estimates of nitrogen loading are inherently uncertain, because of the lack of information and data to fully characterize and simulate hydrology and hydrogeology of the modeling sites and the biogeochemical processes involved in nitrogen transport. Valiela et al. (1997) estimated that the uncertainty of model simulated load is approximately 37-38% due to uncertainty in the coefficients used in the NLM modeling. In this study, there is no measurement of hydraulic and transport parameters (e.g., hydraulic conductivity and dispersivity). Although model calibration is conducted to estimate the parameter values, the monitoring data is too scarce to yield parameter estimates with small uncertainty. While uncertainty quantification has been conducted for decades in groundwater modeling, few attempts have been made to quantify uncertainty in nitrogen load estimates (Collins et al., 2000). Recently, a new function of Monte Carlo (MC) simulation is developed for ArcNLET to conduct uncertainty quantification (Rios et al., 2013b). The MC simulation addresses uncertainty in model parameters and gives the distribution and statistics of load estimates (instead of a single value), which can be used to facilitate decision-making to better define and defend the best management action for managing nitrogen pollution. As shown in the report, the uncertainty is large. Reducing the uncertainty requires collecting more data of hydrogeologic properties and system states such as hydraulic heads and nitrogen concentrations. The MC simulation can help evaluate to what extent the load estimate can increase or decrease, depending on the magnitude of observed concentration at hypothetical monitoring points.

In the remainder of this report, the conceptual model groundwater flow and nitrogen transport used in ArcNLET and its computational implementation are briefly described in Section 2. Section 3 presents the hydrologic and hydrogeologic conditions of the modeling areas as well as the historical monitoring data compiled in this study for model calibration. The results of model calibration and load estimation are given in Section 4, followed by uncertainty analysis in Section 5. The summary and conclusions of this study are discussed in Section 6.

2. SIMPLIFIED CONCEPTUAL MODEL OF ArcNLET

ArcNLET is based on a simplified conceptual model of groundwater flow and nitrate transport. The model has three sub-models: groundwater flow model, nitrate transport model, and nitrate load estimation model. The results from the flow model are used by the transport model, whose results are in turn utilized by the nitrate load estimation model. By invoking assumptions and simplifications to the system being modeled, computational cost is significantly reduced, which enables ArcNLET to provide quick estimates of nitrate loads from septic systems to surface water bodies. The three submodels are briefly described here; more details of them can be found in Rios (2010) and Rios et al. (2013a). Ammonium is not explicitly simulated in ArcNLET. Instead, it is assumed in this study that ammonium transport is the same as nitrate transport so that ArcNLET can simulate nitrogen transport and estimate nitrogen load, not merely nitrate load, from septic systems to surface water bodies. This assumption however may overestimate nitrogen loads.

The groundwater flow model of ArcNLET is simplified by assuming that the water table is a subdued replica of the topography in the surficial aquifer. According to Haitjema and Mitchell-Bruker (2005), the assumption is valid if

$$\frac{RL^2}{mKHd} > 1\,, \tag{1}$$

where R [m/day] is recharge, L [m] is average distance between surface waters, m is a dimensionless factor accounting for the aquifer geometry, and is between 8 and 16 for aquifers that are strip-like or circular in shape, K [m/day] is hydraulic conductivity, H [m] is average aquifer thickness, and d [m] is the maximum distance between the average water level in surface water bodies and the elevation of the terrain. The criterion, as a rule of thumb, can be met in shallow aquifers in flat or gently rolling terrain. Based on the assumption, the shape of water table can be obtained by smoothing land surface topography given by DEM of the study area. In ArcNLET, the smoothing is accomplished using moving-window average via a 7 × 7 averaging window. The smoothing process needs to be repeated for multiple times, depending on discrepancy between the shapes of topography and water table. The number of the smoothing process, called smoothing factor, is specified by ArcNLET users as an input parameter of ArcNLET. This parameter needs to be calibrated against measured hydraulic heads in the study area, as explained in detail in Section 4.

With the assumption that smoothed DEM has the same shape (not the same elevation) of water table, hydraulic gradients can be estimated from the smoothed DEM. Subsequently, groundwater seepage velocity, v, can be obtained by applying Darcy's Law

$$\begin{aligned} v_x &= -\frac{K}{\phi}\frac{\partial h}{\partial x} \approx -\frac{K}{\phi}\frac{\partial z}{\partial x} \\ v_y &= -\frac{K}{\phi}\frac{\partial h}{\partial y} \approx -\frac{K}{\phi}\frac{\partial z}{\partial y} \end{aligned}\,, \tag{2}$$

where K is hydraulic conductivity [LT^{-1}], ϕ is porosity, h is hydraulic head, and hydraulic gradient ($\partial h/\partial x$ and $\partial h/\partial y$) is approximated by the gradient of the smoothed topography ($\partial z/\partial x$ and $\partial z/\partial y$). Implementing the groundwater flow model in the GIS environment yields the magnitude and direction of the flow velocity for every discrete cell of the modeling domain, which are used to estimate flow paths originating from individual septic systems and ending in surface water bodies. The calculation considers spatial variability of hydraulic conductivity, porosity, hydraulic head, and septic system locations. Because hydraulic gradients and water bodies are not hydraulically linked in the model, ArcNLET users need to evaluate whether the resulting shape of the water table is consistent with the drainage network associated the water bodies. The values of hydraulic conductivity and conductivity can be obtained from field measurements, literature data, and/or by calibration against measurements of hydraulic head and groundwater velocity.

Additional assumptions and approximations of the flow model are made as follows: (1) the Dupuit-Forchheimer assumption is used so that the vertical flow can be ignored and only two-dimensional (2-D) isotropic horizontal flow is simulated; (2) the steady-state flow condition is assumed, since this software is used for the purpose of long-term environmental planning; (3) the surficial aquifer does not include karsts or conduits so that Darcy's Law can be used; (4) mounding on water table due to recharge from septic systems and rainfall is not explicitly considered (but assumed to be reflected by the steady-state water table); (5) the flow field is obtained from the water table without explicit consideration of a water balance; (6) groundwater recharge from the estuary is disregarded. While these assumptions may not be ideal, especially the assumption of steady-state, they are needed to make model complexity compatible with available data and information and to make the model run efficient in the GIS modeling environment.

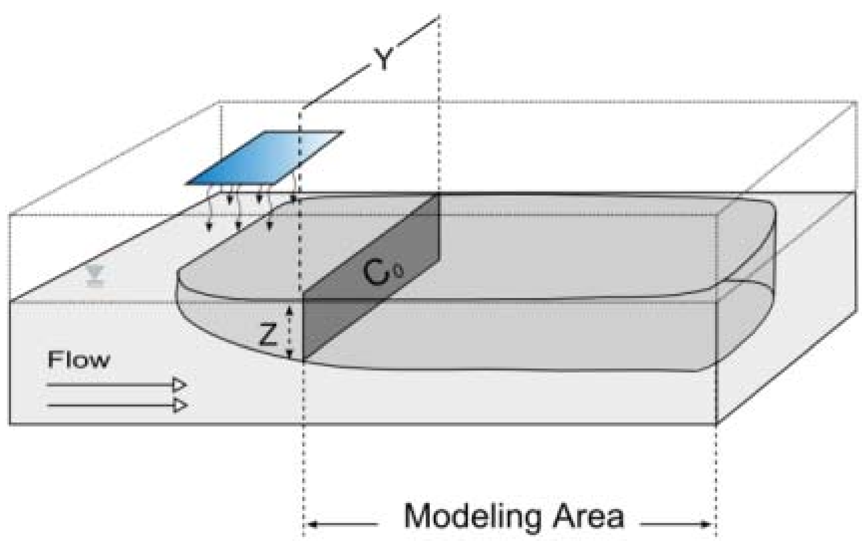

Figure 2-1. Conceptual model of nitrate transport in groundwater adapted from Aziz et al. (2000). The unsaturated zone is bounded by the rectangular box delineated by the dotted lines; the groundwater zone is bounded by the box delineated by the solid lines

Figure 2-1 shows the conceptual model of nitrate transport in ArcNLET, which is similar to that of BIOSCREEN (Newell et al. 1996) and BIOCHLOR (Aziz et al. 2000) developed by the U.S. EPA. In the conceptual model, nitrate enters the groundwater zone with a uniform and steady flow in the direction indicated. The Y−Z plane in Figure 2-1 is considered as a source plane (with a constant concentration C_0 [ML^{-3}]) through which nitrate enters the groundwater system. Two-dimensional (2-D) nitrate transport in groundwater is described using the advection-dispersion equation

$$\frac{\partial C}{\partial t} = D_x \frac{\partial^2 C}{\partial x^2} + D_y \frac{\partial^2 C}{\partial y^2} - v\frac{\partial C}{\partial x} - kC \tag{3}$$

where C is the nitrate concentration [M/L^3], t is time [T], D_x and D_y are the dispersion coefficients in the x and y directions, respectively [L^{2}T^{-1}], v is the constant seepage velocity in the longitudinal direction [L], and k is the first-order decay coefficient [T^{-1}]. This equation assumes homogeneity of parameters (e.g., dispersion coefficient) and uniform flow in the longitudinal direction. The last term in Eq. 3 is to simulate the denitrification, in which nitrate is transformed into nitrogen gas through a series of biogeochemical reactions. Following McCray et al. (2005) and Heinen (2006), the denitrification process is modeled using first-order kinetics and included as the decay term, which can also be used to take into account other loss processes. The steady-state form, semi-analytical solution of Eq. 3 is derived based on that of West et al. (2007), which is of 3-D, steady-state form and similar to the work of Domenico (1987). The analytical solution used in this study is (Rios, 2010; Rios et al., 2013a)

$$\begin{aligned} C(x,y) &= \frac{C_0}{2} F_1(x) F_2(y,x) \\ F_1 &= \exp\left[\frac{x}{2\alpha_x}\left(1-\sqrt{1+\frac{4k\alpha_x}{v}}\right)\right] \\ F_2 &= erf\left(\frac{y+Y/2}{2\sqrt{\alpha_y x}}\right) - erf\left(\frac{y-Y/2}{2\sqrt{\alpha_y x}}\right) \end{aligned} \tag{4}$$

where α_x and α_y [L] are longitudinal and horizontal transverse dispersivity, respectively, Y [L] is the width of the source plane, and C_0 [M/L^3] is the constant source concentration at the source plane. A review of analytical solutions of this kind and errors due to assumptions involved in their derivation is provided by Srinivasan et al. (2007).

The 2-D concentration plume is extended downwards to the depth Z of the source plane (Figure 2-1); the pseudo three-dimensional (3-D) plume is the basis for estimating the amount of nitrate that enters into groundwater and loads to surface water bodies. While each individual septic system has its own source concentration, C_0, drainfield width, Y, and average plume thickness, Z, the information and data of these variables are always unavailable in a management project. Therefore, constant values are used for all septic systems in this study. ArcNLET allows using different C_0 values for different septic systems, if the data are available. Despite of the constant values used for all the septic systems, each individual septic system has its own concentration plume, because flow velocity varies between the septic systems. Since the flow velocity estimated in the groundwater flow model is not uniform but varies in space, in order to use the analytical solution with uniform velocity, the harmonic mean of velocity (averaged along the flow path of a plume) is used for evaluating each individual plume. The plumes either end at surface water bodies or are truncated at a threshold concentration value (usually very small, e.g., 10^{-6}). After the plumes for all septic systems are estimated, by virtue of linearity of the advection-dispersion equation with respect to concentration, the individual plumes are added together to obtain the spatial distribution of nitrate concentration in the modeling domain. The superposition however may result in higher and shallower concentrations than exist in the field unless the averaging depth is deep enough.

The nitrate load estimation model evaluates the amount of nitrate loaded to target surface water bodies. For the steady-state model, this is done using the mass balance equation, $M_{out} = M_{in} - M_{dn}$, where M_{out} [MT^{-1}] is mass load rate to surface water bodies, M_{in} [MT^{-1}] is mass inflow rate from septic systems to groundwater, and M_{dn} [MT^{-1}] is mass removal rate due to denitrification. The mass inflow rate, M_{in}, consists of inflow due to advection and dispersion, and is evaluated via

$$M_{in} = YZ\phi\left(vC_0 - \alpha_x v \left.\frac{\partial C}{\partial x}\right|_{x=0} \right) = YZ\phi v C_0 \frac{1 + \sqrt{1 + \frac{4ka_x}{v}}}{2} \quad . \tag{5}$$

The derivative, $\partial C/\partial x$, used for calculating the dispersive flux is evaluated using an analytical expression based on the analytical expression of concentration in equation (4). When the mass inflow rate is known, it can be specified within ArcNLET. Otherwise, the mass inflow rate is calculated by specifying the Z value. The mass removal rate due to denitrification, M_{dn}, is estimated via

$$M_{dn} = \sum_i kC_i V_i \phi_i \ , \tag{6}$$

where C_i and V_i are concentration and volume of the i-th cell of the modeling domain, and kC_i is denitrification rate assuming that denitrification is the first-order kinetic reaction (Heinen 2006). If a plume does not reach any surface water bodies, the corresponding nitrate load is theoretically zero.

The simplified groundwater flow and nitrate transport model is implemented as an extension of ArcGIS using the Visual Basic .NET programming language. In keeping with the object oriented paradigm, the code project is structured in a modular fashion. Development of the graphical user interface (GUI) elements is separated from that of the model elements; further modularization is kept within the development of GUI and model sub-modules. The main panel of the model GUI is shown in Figure 2-2; there are four tabs, each of which represents a separate modeling component. For example, the tab of Groundwater Flow is for estimating magnitude and direction of groundwater flow velocity, and the tab of Particle Tracking for estimating flow path from each septic system. Each tab is designed to be a self-contained module and can be executed individually within ArcGIS. Five ArcGIS layers are needed for running ArcNLET. They are DEM, hydraulic conductivity, and porosity in raster form, septic system locations in point form, and surface water bodies in polygon form. These ArcGIS files need to be prepared outside ArcNLET. The output files are also ArcGIS layers that can be readily post-processed and visualized within ArcGIS. More details of the software development, including verification and validation, are described in Rios (2010) and Rios et al. (2013a).

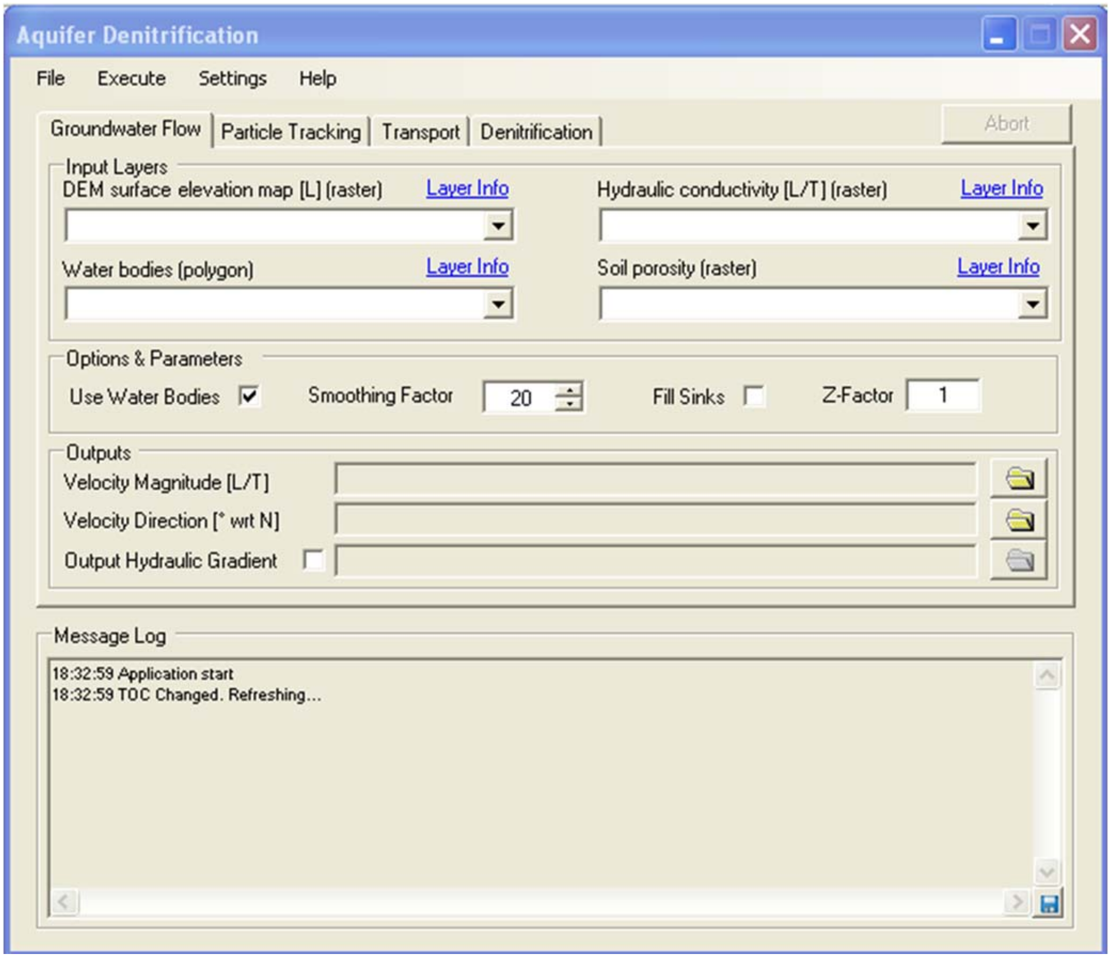

Figure 2-2. Main Graphic User Interface (GUI) of ArcNLET with four modules of Groundwater Flow, Particle Tracking, Transport, and Denitrification

Before ArcNLET is used for estimating nitrogen load to surface water bodies, calibrating the model parameters is always needed to match model simulations to field observations. However, in many projects of nitrogen pollution management, field observations are scarce. This is the reason of developing ArcNLET whose complexity is compatible with available data. As shown in the next section, observation data is extremely limited in the modeling area, and the conceptual model based on the limited data should be simple. On the other hand, the calibrated ArcNLET model is able to reasonably match field observations, as shown in Section 4.

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3. ANALYSIS OF MONITORING DATA

This section analyzes the monitoring data compiled in this study to qualitatively evaluate potential impacts of septic systems on water quality (of surface water and groundwater) and to understand groundwater flow and nitrogen transport in the modeling areas. While there is a monitoring network of surface water quality, groundwater monitoring data is extremely limited in the modeling areas. This restricts development of complex conceptual models of groundwater flow and nitrogen transport, and leads to uncertainty in calibration of ArcNLET in the next section. Most of the data used in this section are downloaded from the DBHYDRO database managed by the South Florida Water Management District (SFWMD) and the Hydrologic Information System managed by The Consortium of Universities for the Advancement of Hydrologic Science, Inc. (CUAHSI). Monitoring data from technical reports of Steve Krupa at SFWMD and Thomas Belanger at Florida Institute of Technology are also used to understand groundwater and surface water interaction and nitrogen transport from septic system in the Indian River Lagoon area.

3.1. Hydrogeology and Hydrology in the Modeling Areas

The surficial aquifer in the St. Lucie River and Estuary Basin is unconfined and separated from the underlaid confined Floridan aquifer by the relatively impermeable Hawthorn Formation. The surficial aquifer consists of Tamiami and Anastasia formations (Toth, 1987), which are primarily composed of well permeable sand and beds or lenses of more permeable limestone, sandstone, and shell (Lichtler, 1960). A lithological study of Miller (1979) indicated that the surficial aquifer in Martin and St. Lucie Counties extends from the water table to about 46 ~ 61 m (150 ~ 200 feet) below land surface. There are three layers in the surficial aquifer. At the top of the surficial aquifer is a medium to fine grained layer extending to a depth of approximately 9 to 12 m (30 to 40 ft.) below land surface. The second is a thin clay layer only several meters thick. The bottom layer is composed of sand and shell and has a thickness of about 30 to 37 m (100 to 120 feet). Although there is lack of data of nitrogen plume thickness, nitrogen transport is expected to occur in the top layer.

Figures 3-1 plots the monthly and annual average precipitation at weather station SVWX near the modeling areas, whose location is shown in Figure 3-2. The precipitation data are downloaded from DBHYDRO. The observation period is from 05/14/1997 to 07/14/2013; observations in 2005 are missing due to hurricanes in 2005. Figure 3-1a shows that a wet season can be delineated from June to October; the smallest (35.8 mm) and largest (184.4 mm) monthly precipitation are in January and August, respectively. Figure 3-2b shows that the lowest and highest annual precipitations are 611.9 mm in 2002 and 1576.1 mm in 2001. While hurricanes and high precipitation have impacts on nutrient and microbial pollution at the study area Lapointe et al. (2012), the impacts are not considered in this study.

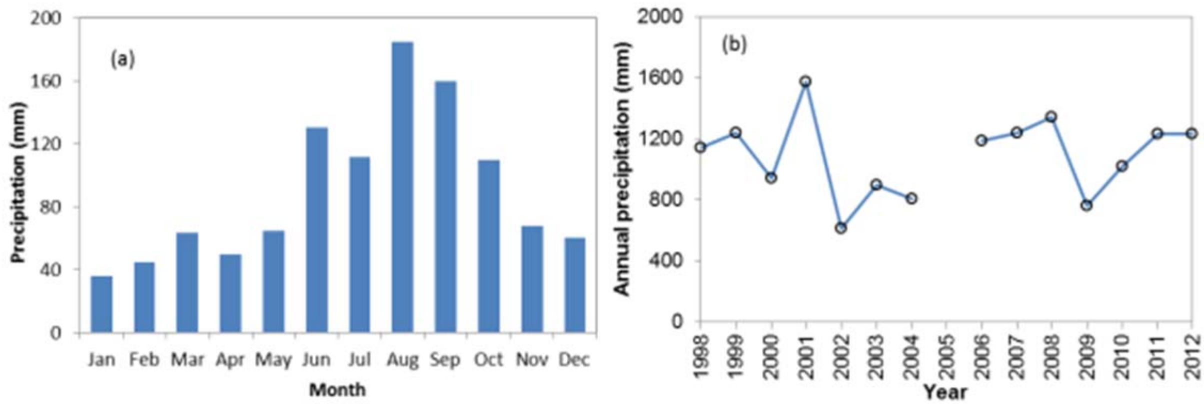

Figure 3-1. (a) Monthly and (b) annual average precipitation from 1998 to 2012 at weather station SVWX, whose location is shown in Figure 3-2.

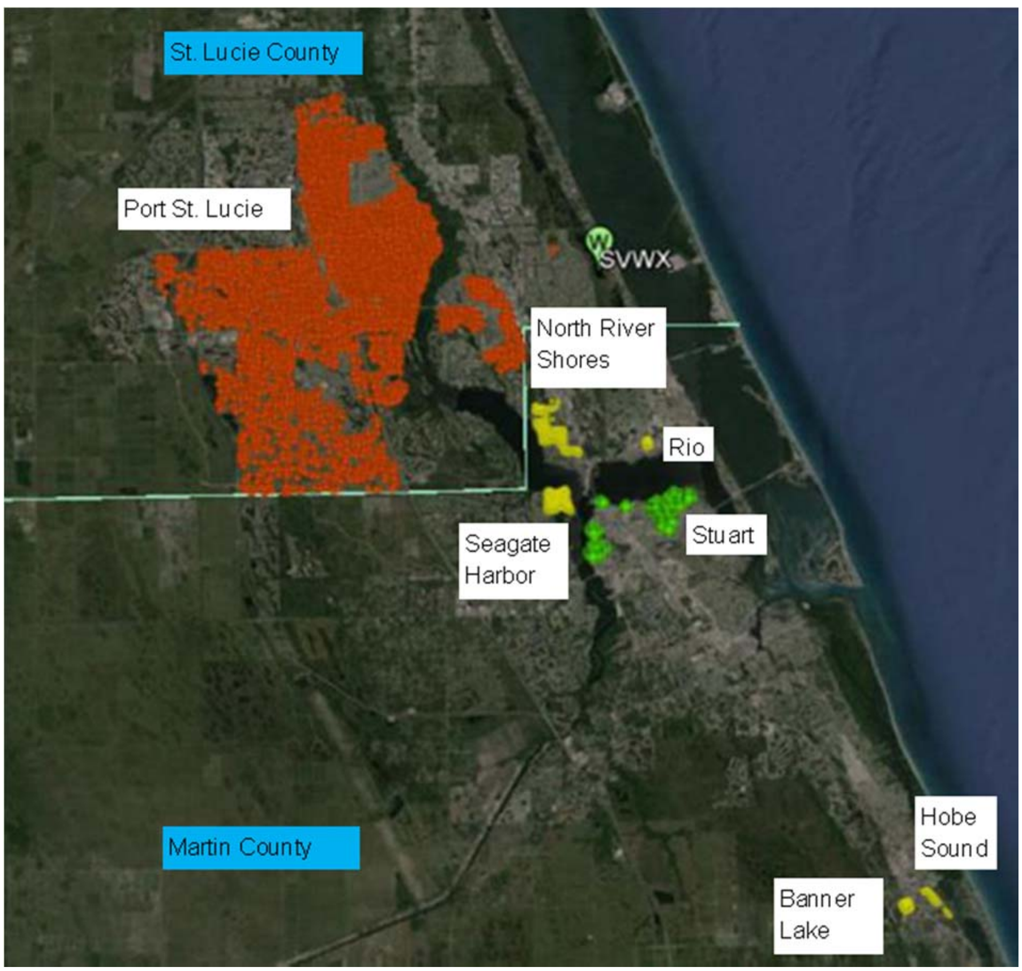

Figure 3-2. Locations of 5,601 removed septic systems in the City of Port St. Lucie (red), 146 in the City of Stuart (blue), and 1,087 at five sites in Martine County (yellow). Location of the weather station (SVWX) with precipitation data is also shown.

3.2. Removed Septic Tanks

This study aims at conducting ArcNLET modeling to estimate nitrogen loads from removed septic systems to surface water bodies. Figure 3-2 shows locations of the removed septic tanks, among which 5,601 are located in the City of Port St. Lucie, 1,087 in Martin County, and 146 in the City of Stuart. The locations are provided by Dale Majewski from the City of Port St. Lucie, Dianne K. Hughes from Martin County, and William Griffin from the City of Stuart. It should be noted that the locations are not exact, but approximated by the geometric center of each parcel. Comparing Figure 3-2 with Figure 3-3 (that plots the sub-basins of the St. Lucie River and Estuary Basin) shows that the removed septic systems are located in the following sub-basins: Basins 4-5-6, North Fork, South Fork, and South Coastal. Therefore, this study should be of direct to use to the on-going TMDL implementation, although the South Coastal sub-basin is not considered in the current BMAP (2013).

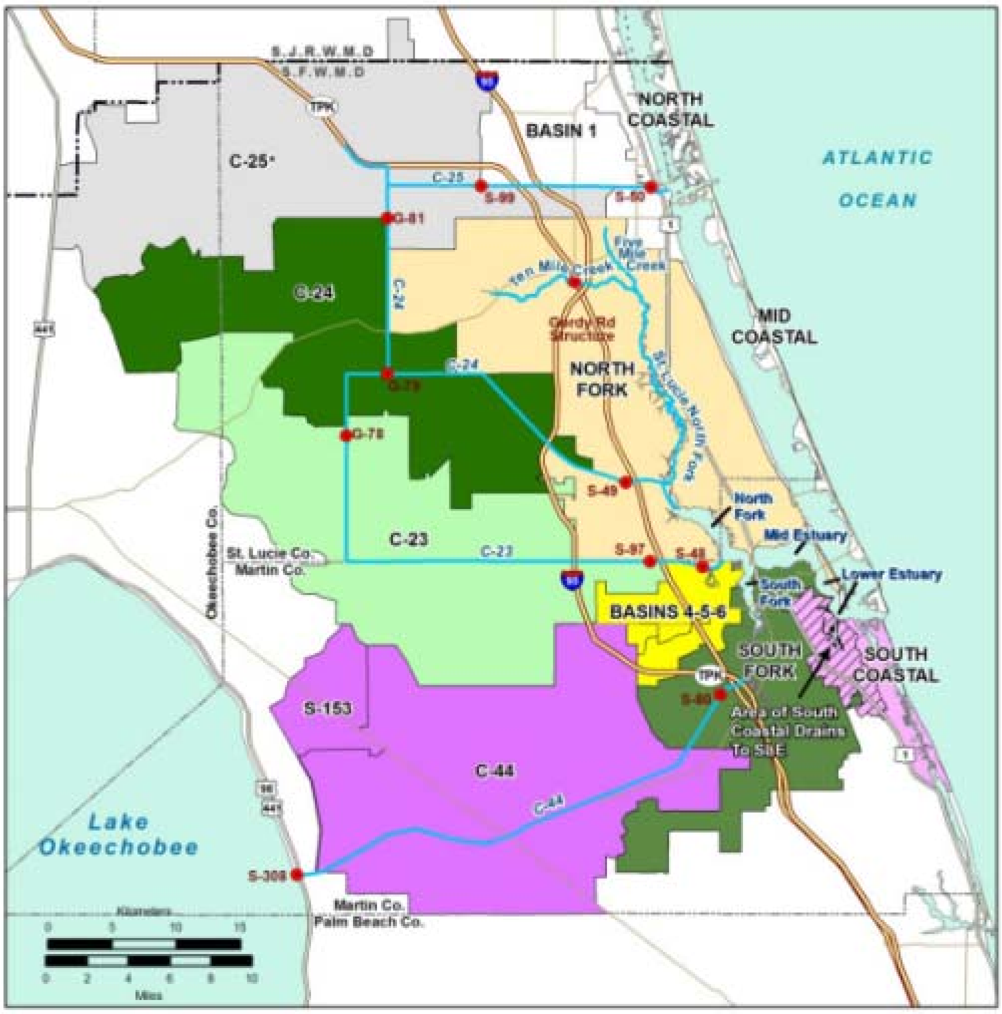

Figure 3-3. Sub-basins of the St. Lucie River and Estuary, adapted from SFWMD (2012).

3.3. Surface Water Quality

A network of surface water quality exists in the St. Lucie Estuary (BMAP, 2013). Dale Majewski from the City of Port St Lucie provided the surface water quality data measured at 21 stations. The data from fourteen stations (Figure 3-4) located in the septic tank removal area are analyzed here. Figure 3-5 plots time series of concentrations (mg/l) of total nitrogen, NO_x (nitrate and nitrite), and TKN at the stations. The sampling frequency is 1-2 times per year from 2004 to 2012 (BMAP, 2013). The figure also shows the TN target of 0.72 mg/L annual TMDL water quality specified in the BMAP of St. Lucie River and Estuary Basin (BMAP, 2012, 2013). At all the stations, the TN concentrations are higher than the TMDL target for most of the monitoring period. This is mainly caused by the high TKN concentrations, which are significantly higher than NO_x concentrations. This figure indicates that management actions are needed to meet the TMDL target.

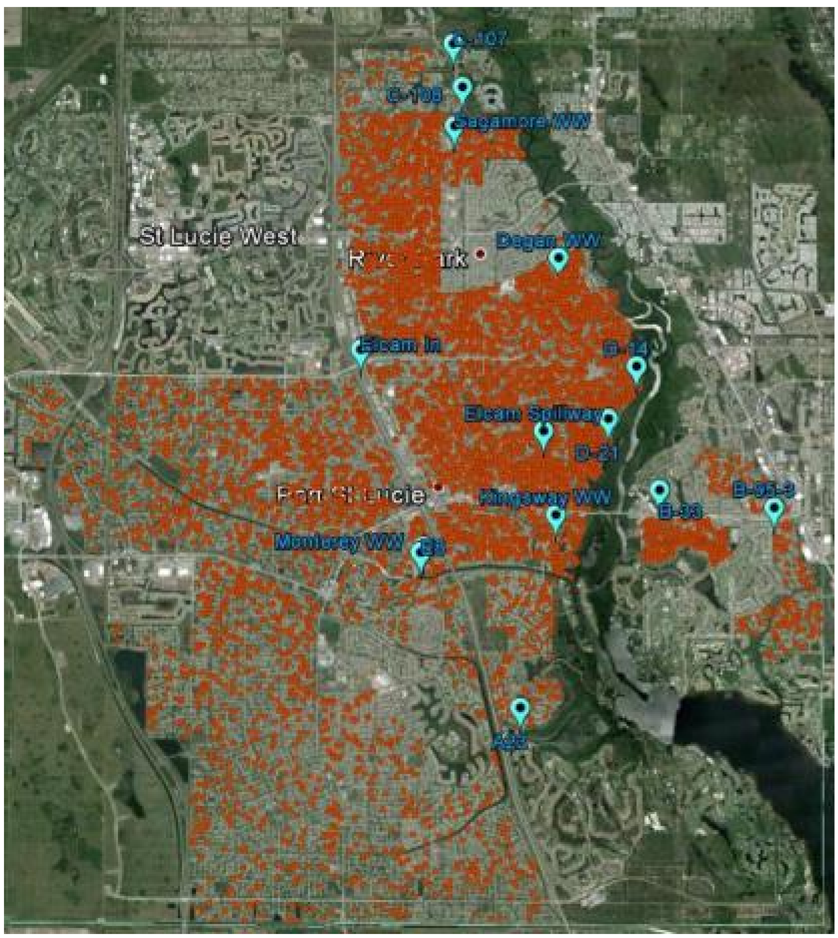

Figure 3-4. Locations of monitoring stations of surface water quality in the City of Port St. Lucie. Station names are labeled in blue.

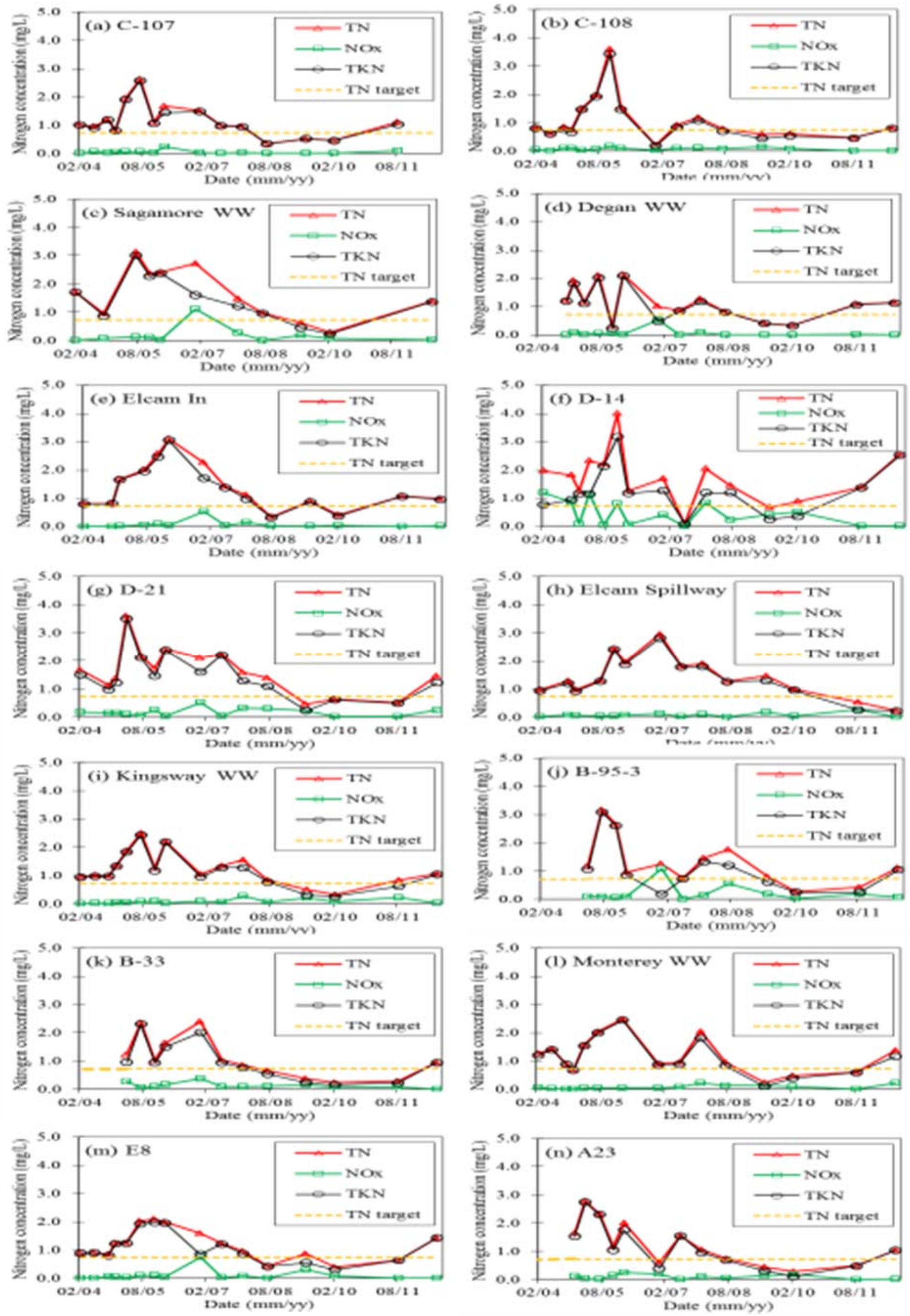

Figure 3-5. Time series of the concentrations (mg/l) of total nitrogen (TN), NOx (nitrite and nitrate), and TKN at monitoring stations of the City of Port St. Lucie. The horizontal line indicates the annual TMDL water quality TN target specified in the BMAP for the St. Lucie River and Estuary Basin (BMAP, 2013).

Temporal variation of the concentrations is observed in Figure 3-5. At most of the stations, peak TN concentrations occurred in 2005 to 2006. Specifically speaking, the peak concentrations occurred in July 2005 at five stations (C-107, Sagamore WW, D-21, Kingsway WW, and B-95-3), in November 2005 at three stations (C-108, D-14, and E8), and in February 2006 at three stations (Degan WW, Elcam In, and Monterey WW). The peak concentrations may be attributed to hurricanes during 2005 that produced record rainfall and large-scale stormwater runoff into the St. Lucie Estuary (Lapointe et al., 2012). The recent TN concentrations in 2011 and 2012 are still higher than the TN target, suggesting water quality deterioration without the hurricane impacts.

Figure 3-6 plots boxplots of the TN concentrations at the fourteen stations. In the boxplots, the central mark of box is the median, and the edges of the box are the 25th and 75th percentiles. The boxplots indicate spatial variation in the concentrations. Among the stations, the median values are the lowest at stations C-107 and C-108 located upstream of the St. Lucie river, but increase at the downstream stations. This may be partly attributed to load from septic systems. In particular, stations Sagamore WW, D-14, D-21, and ELCAM Spillway with the highest median TN concentrations are located in the areas with high septic density (Figure 3-4). The spatial variation indicates the impact of septic systems on surface water quality in the highly populated area. A more quantitative discussion is given in Section 4, after the load estimates are evaluated using the calibrated ArcNLET model.

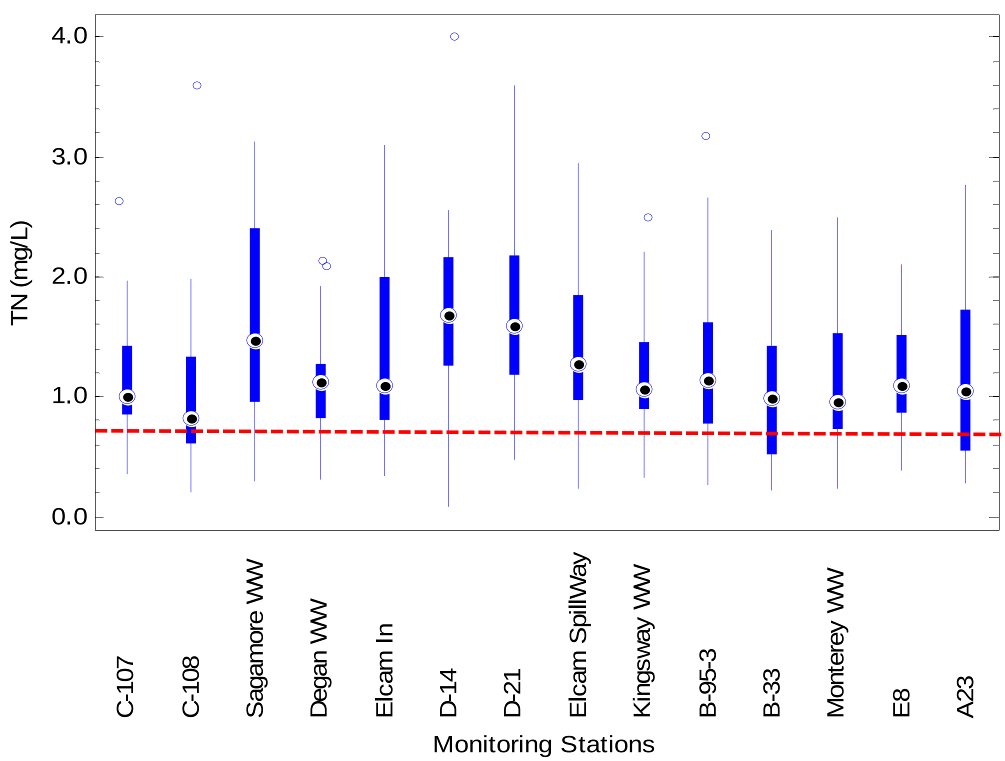

Figure 3-6. Boxplots of TN concentrations at the monitoring stations. The horizontal line indicates the TMDL TN target of 0.72 mg/L.

Figure 3-7 plots the boxplots of NO_x concentrations at the monitoring stations. Most of the concentrations are smaller than the NO_x limit of 0.35 mg/L. This limit value, established by EPA for springs to prevent excess algal growth (Kaufman et al., 2010), is used by BMAP (2012) as the TMDL target to achieve reductions in nutrients in the Santa Fe River, FL. The two highest median NO_x concentrations occur at stations D-14 and D-21. While the reason remains unknown, it is noted that D-14 and D-21 are located in the areas with high septic density, implying that wastewater from septic systems may contribute to the high concentrations of NO_x.

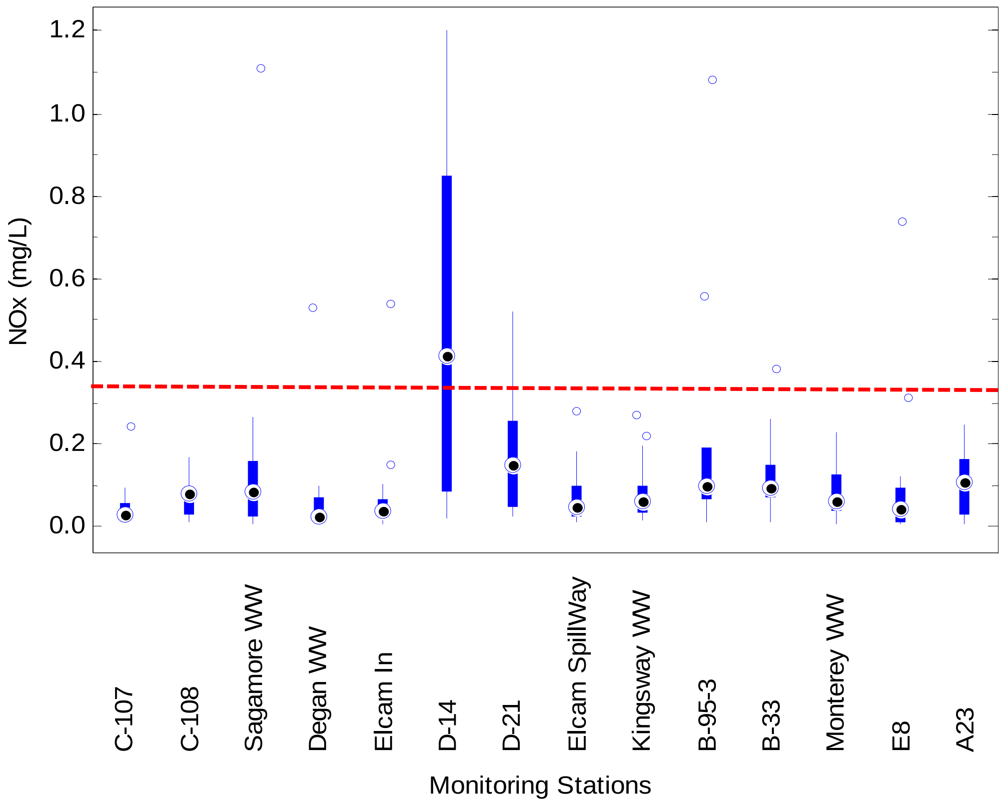

Figure 3-7. Boxplots of NOx concentrations at the monitoring stations. The horizontal line indicates the limit of NOx as 0.35 mg/L for Florida springs (Kaufman et al., 2010).

3.4. Groundwater and Surface Water Interaction

There is no existing monitoring network of groundwater level and quality, and monitoring data of groundwater nitrogen concentrations is extremely scarce in the modeling areas. Belanger et al. (2004) conducted a series of field measurements to study submarine groundwater discharge. While none of the stations are in the modeling area, monitoring data from the stations are used to understand groundwater flow and nitrogen transport on river shores. The data were gathered during the period from March 2002 to November 2004 at six stations located in the Indian River Lagoon (Figure 3-8). Near the individual stations, monitoring wells and piezometers are constructed at three different depths: shallow (~30 ft.), intermediate (~60 ft.), and deep (~100 ft.); only shallow and intermediate wells are available at sites IRMN, SLHR, and SLLT). Observations of water table from the shallow wells (30ft.) are more relevant to this study and used to interpret the seepage from groundwater to surface water.

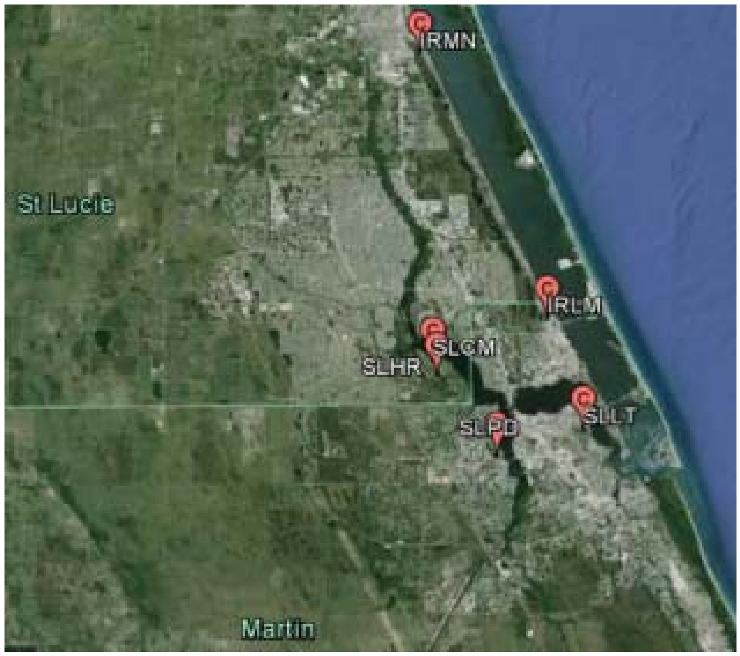

Figure 3-8. Locations of monitoring stations in the seepage meter program for St. Lucie River and Indian River Lagoon (Belanger et al., 2004)

Figure 3-9 plots time series of river stages and groundwater levels at the shallow wells (30 ft.). Generally speaking, groundwater levels are higher than river stages, and groundwater discharges to surface water. The highest water table is at stations SLCM and SLHR. However, at station SLLT, groundwater levels were constantly lower than river stages from July 2002 to August 2004, indicating groundwater is recharged by the river. While the exact reason is unknown, this may be attributed to groundwater pumping, tidal effect, and/or wind effect at the station. Groundwater recharge occurred for a short period at stations IRLM and SLPD. Therefore, it is reasonable to assume that groundwater discharges to the river in the modeling area.

Groundwater recharge from the river is also confirmed by chloride concentrations in groundwater and surface water. As chloride concentration is significantly higher in surface water than in groundwater, groundwater chloride concentration should be elevated if groundwater recharge from river occurs. This is observed in Figure 3-10 for station SLLT, where groundwater chloride increases about three orders of magnitude. It is also observed from Figure 3-10 that surface water chloride concentration is higher at stations SLLT, IRMN, and IRLM that are close to the ocean than at stations, SLCM, SLHR, and SLPD that are far away from the ocean. It suggests that saltwater intrusion may play a certain role in groundwater solute transport, which however is not considered in this study.

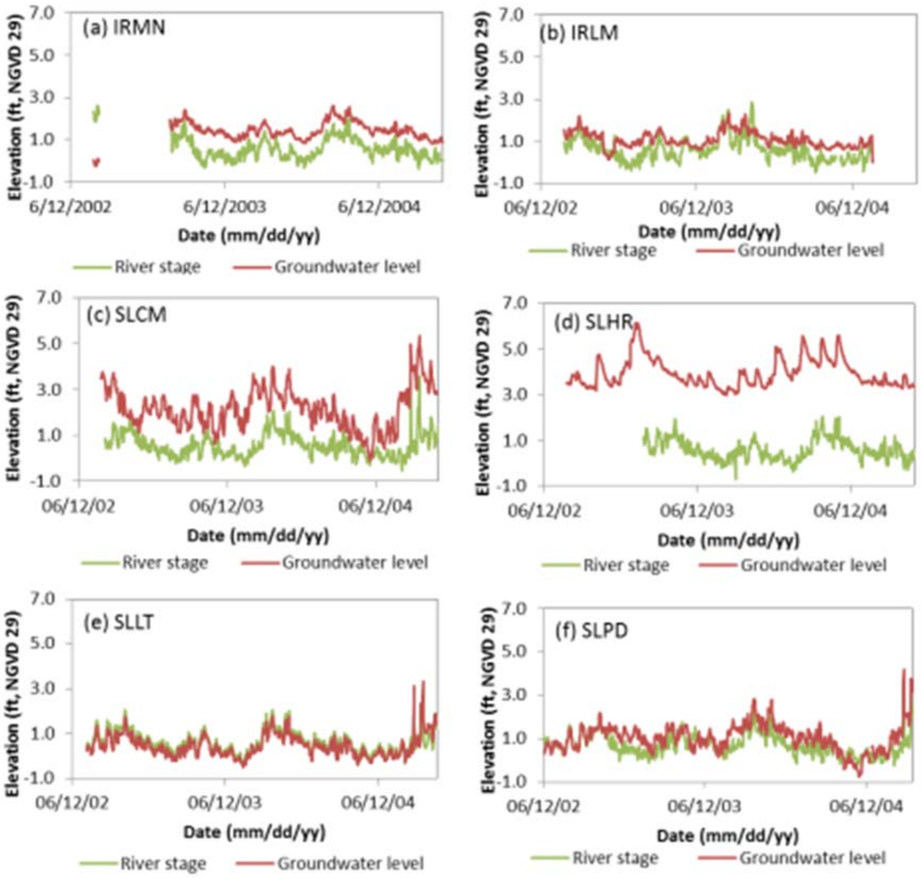

Figure 3-9. Time series of river stages and groundwater levels at the six stations shown in Figure 3-8.

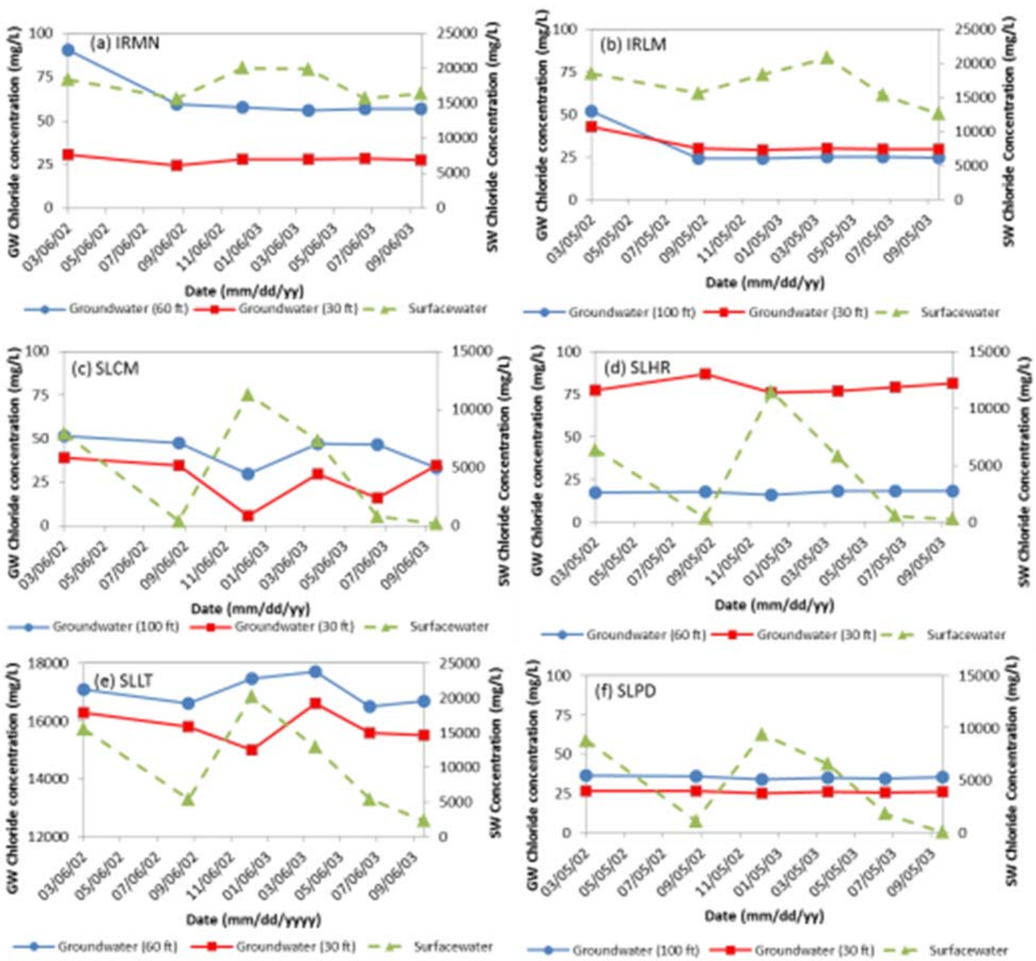

Figure 3-10. Chloride concentrations in groundwater and surface water at the six stations shown in Figure 3-8.

3.5. Groundwater Data Selected to Calibrate ArcNLET Flow Model

Observations of groundwater levels are needed to calibrate the ArcNLET flow model by adjusting the smoothing factor so that the shape of smoothed DEM can match the shape of water table. Given that the flow model is a steady-state one, average (over time) water levels are used as the calibration target. A total of seven monitoring wells are identified for the City of Port St. Lucie, and six wells for Sutart City. However, there is no monitoring well available in Martin County. As shown in Section 4, the calibrated smoothing factors of the City of Port St. Lucie and the City of Stuart are the same, and it is used for the ArcNLET modeling in Martin Conty.

Figure 3-11 shows locations of the seven wells in the City of Port St. Lucie. Data of well STL-270 was gathered from the DBHYDRO, and data of the other six wells are gathered from USGS database (http://waterdata.usgs.gov/nwis). These wells are selected because they are located in the septic tank removal area and reflect site-specific hydrogeologic conditions. In addition, the wells are distributed over the domain and can refelect spatial variability of the hydrogeologic conditions. While the data are relatively old, mesured in the peirod of 1988 – 1995, a study below shows that mean hydraulic head only changes slightly in the modeling areas. Time series of the water level are plotted in Figure 3-12. Despite of moderate fluctuation, there is no apparent trend in the observations. Therefore, it is reasonable to use the average water level as the calibration target and to conduct steady-state ArcNLET modeling. Among the wells, water level in wells PG-23 and STL-270 are the lowest, less than 10 feet (Figure 3-12d and 3-12f). Water levels in well STL-272, STL-214, and STL-273 are the highest, about 20 feet. The spatial trend of water level is consistent with that of land surface elevation (descresing from west to east), suggesting that it is reasonable to assume that water table is a subdued replica of topography.

Figure 3-13 plots locations of six wells in the City of Stuart. Data of well M-1153 are gathered from DBHYDRO, and data of the other five wells are gathered from the USGS database (http://waterdata.usgs.gov/nwis). Several wells located outside of the septic tank removal area are chosen for calibration so that the calibrated model can be used for other areas in the city. This manefests flexibility of the modeling approach. For example, when more septic systems are removed in the City of Stuart, the calibrated flow model can be used directlly for ArcNLET flow modeling. Time series of observed groundwater levels are plotted in Figure 3-14. Similar to Figure 3-12, this figure does not show apparent temporal trend, suggesting the average value can be used as the calibration target. Groundwater level is the highest in M-1153 located in the inland but low in the wells close to the river, which is consistent with the spatial trend of topography.

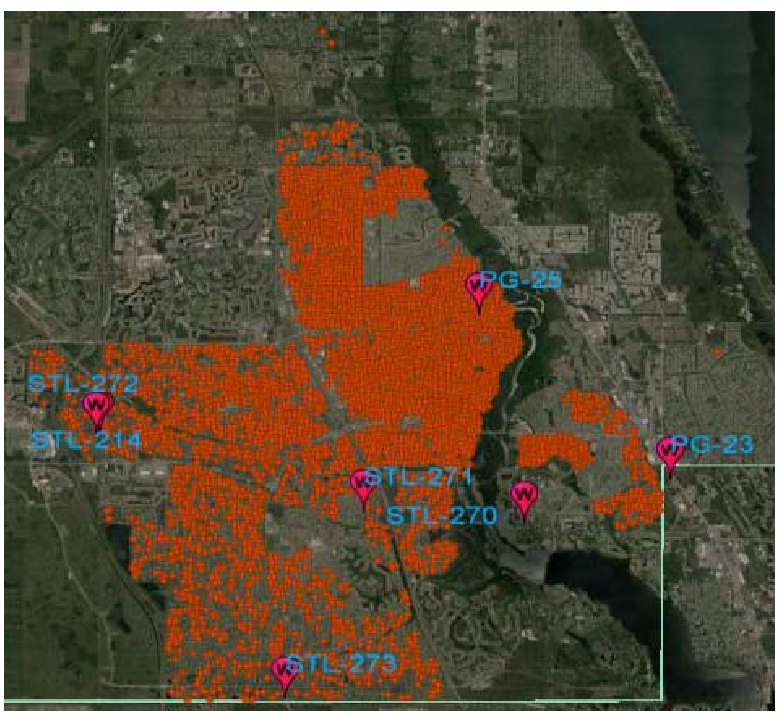

Figure 3-11. Location of monitoring wells of water level selected for calibrating ArcNLET flow model in the City of Port St. Lucie.

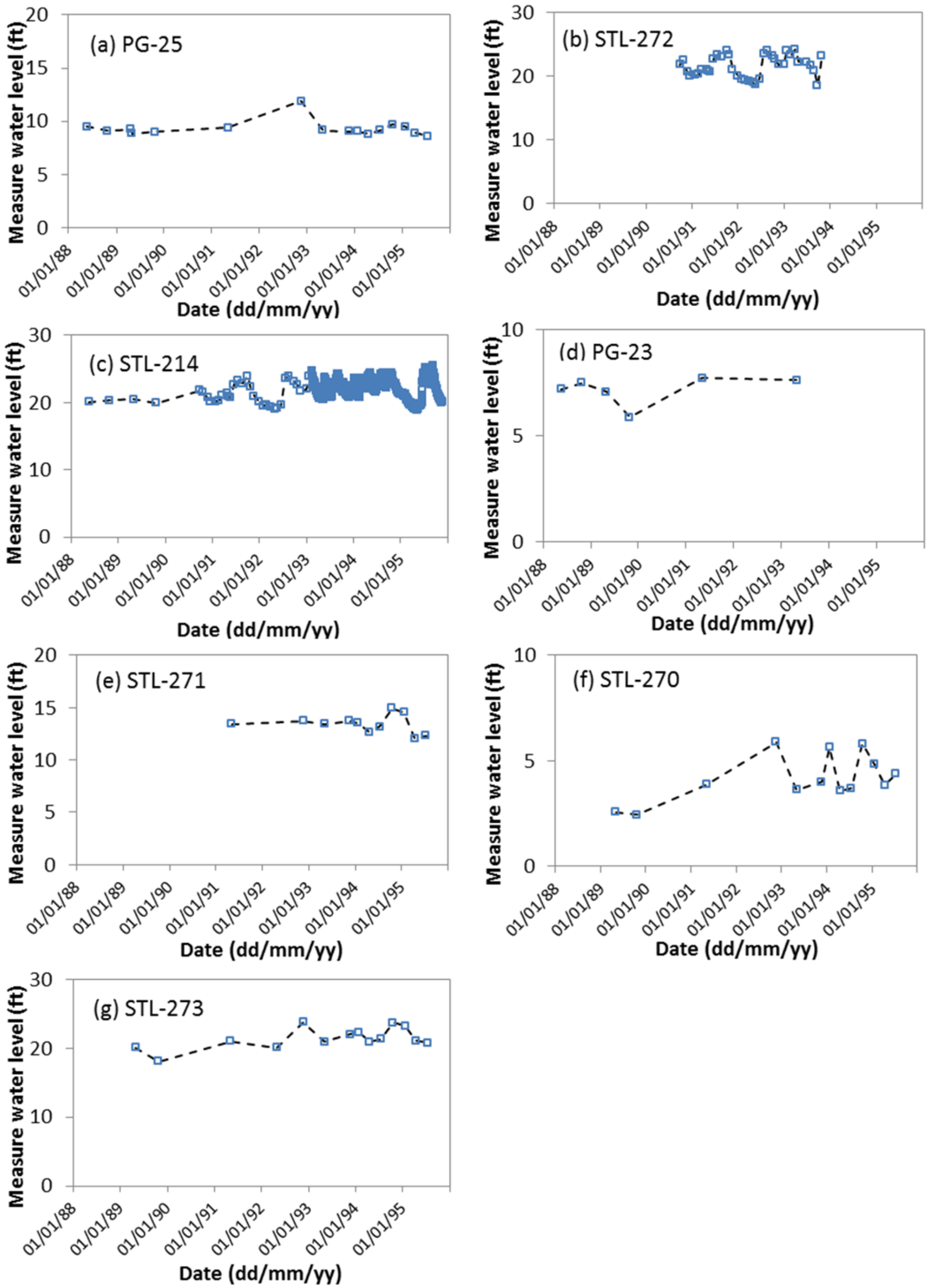

Figure 3-12. Time series of groundwater levels in seven wells selected for calibrating ArcNLET flow model in the City of Port St. Lucie.

Figure 3-13. Location of monitoring wells of water level selected for calibrating ArcNLET flow model in the City of Stuart. Removed septic systems (green points) and functioning septicsystems (yellow points) are also shown.

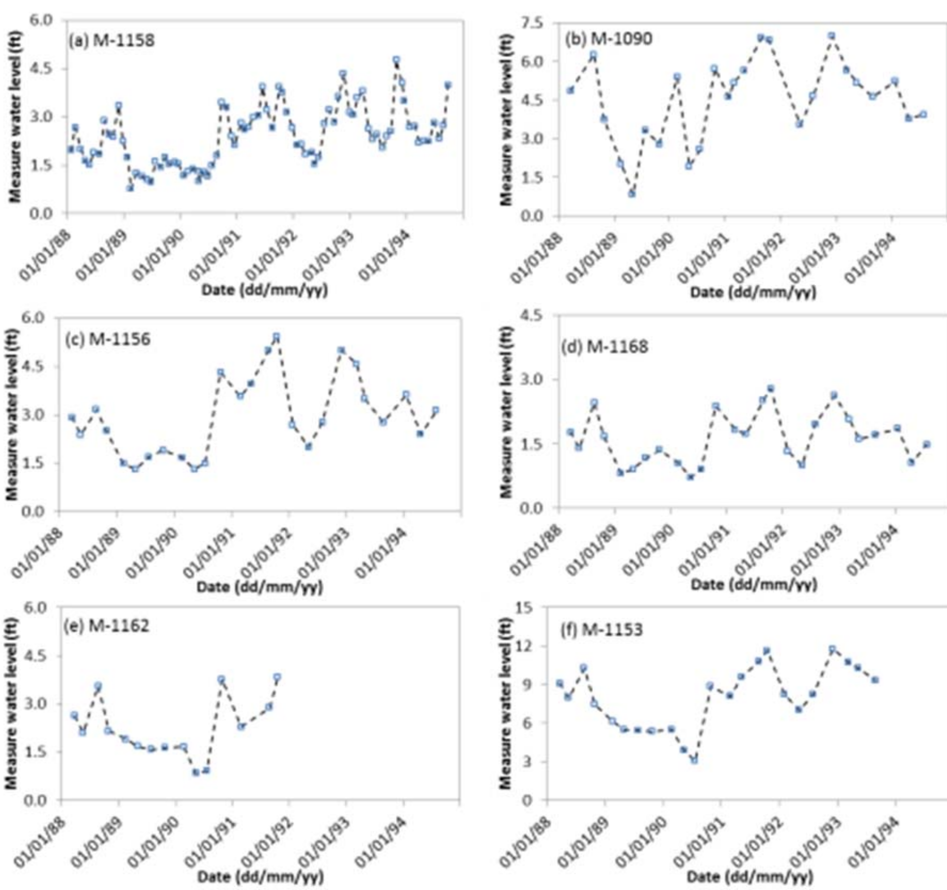

Figure 3-14. Time series of groundwater levels in seven wells selected for calibrating ArcNLET flow model in the City of Stuart.

3.6. Analysis of Trend in Groundwater Levels

As pointed in Section 3.5, the calibration data of groundwater level were measured in the period of 1988 – 1995. An analysis is conducted to investigate whether there is a trend in groundwater level. Six wells with daily monitoring data are selected for this analysis. As shown in Figure 3-15, these wells are outside the modeling area but are still within St. Lucie and Martin counties. Therefore, the results of the trend analysis are expected to be applicable to the calibration data discussed above. Well M-1004 has the longest record of groundwater level observations. The time series of this well (Figure 3-16) shows that the mean values for the period of 1988-1995 is about 0.303 feet (0.1 m) larger than the mean of 1988-2013. The small difference suggests that it is acceptable to use the calibrated data discussed above to calibrate the ArcNLET flow model.

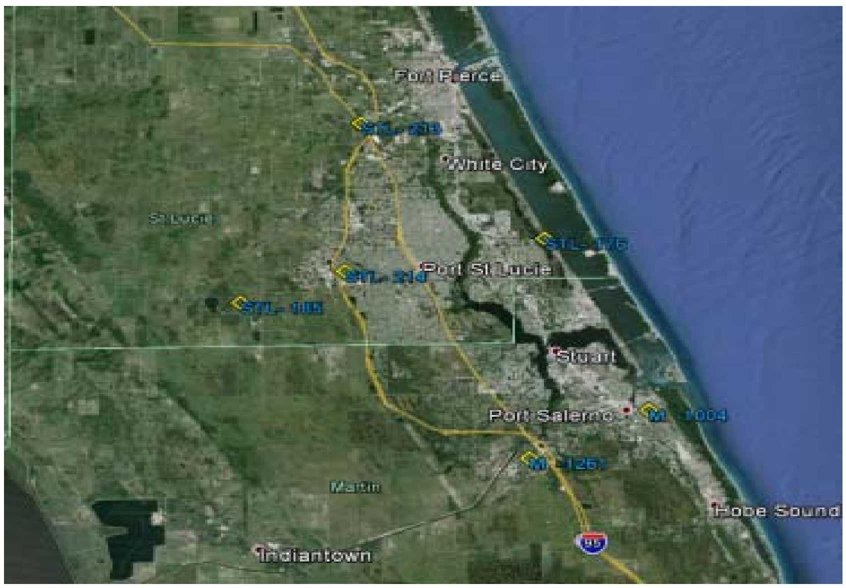

Figure 3-15. Location of monitoring wells of daily groundwater levels in St. Lucie and Martin Counties.

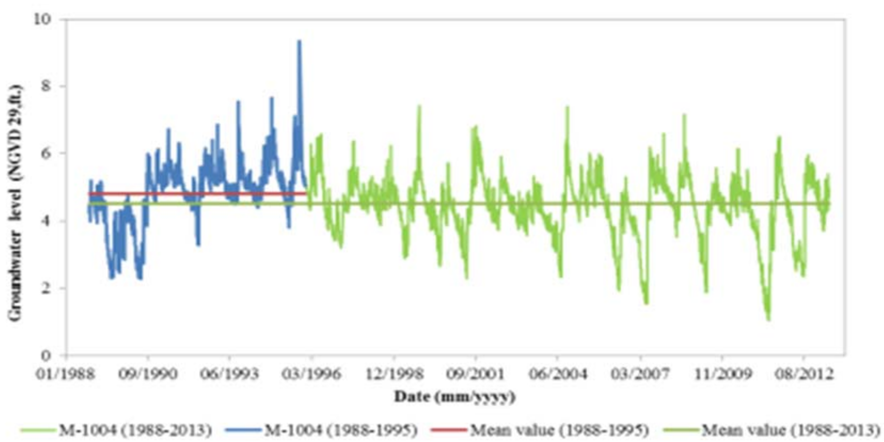

Figure 3-16. Time series of daily groundwater level observations in well M-1004 from 1988 to 2013.

To further analyze the long-term trend in groundwater level, the nonparametric Mann-Kendall test (Helsel and Hirsch, 2002) is conducted to test the null hypothesis of no change. The results of the Mann-Kendall test listed in Table 3-1 show that the null hypothesis is rejected except for well STL-214. In other words, except at well STL-214, there is a trend of changes in groundwater levels. However, as shown in Figure 3-17, the change is small, because the slopes of the fitted trend are close to zero for all the six wells. This further confirms that it is acceptable to use the calibrated data to calibrate the ArcNLET flow model and that it is reasonable to use the mean values as the calibration target.

Table 3-1. Mann-Kendall Tau values at selected wells for groundwater levels

	STL-176	**STL-214**	**STL-185**	**STL-213**	**M-1004**	**M-1261**
tau value	-0.3654	-0.0654	-0.3272	-0.1753	-0.1522	-0.1985
Significance level	p<0.001	0.1331	p<0.001	p<0.001	p<0.001	p<0.001

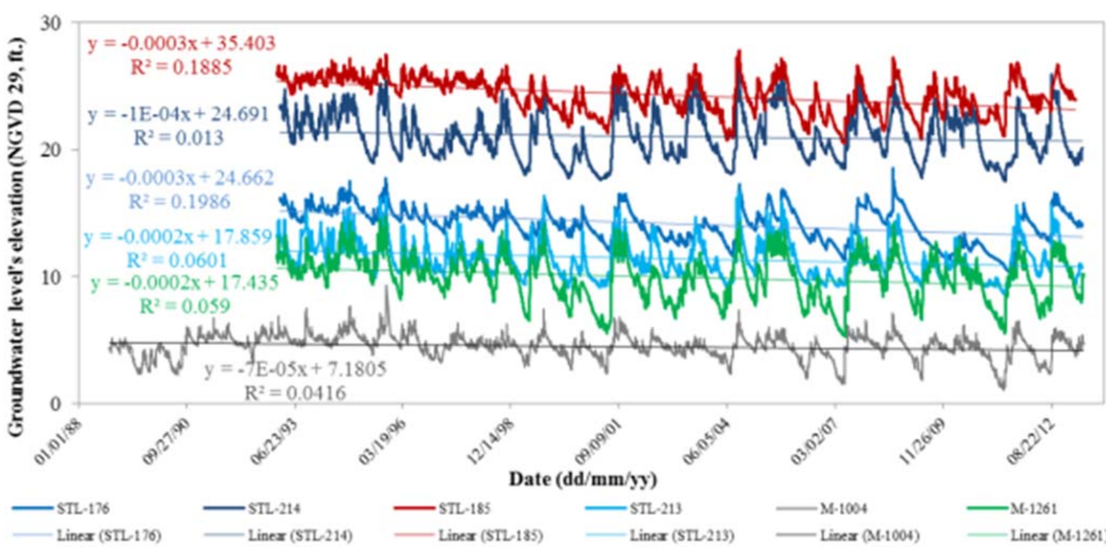

Figure 3-17. Time series of daily groundwater levels at six monitoring wells

3.7. Analysis of Nitrogen Concentration Data

Observations of nitrogen concentration are extremely scarce. Data are compiled from the following four sources: (1) USGS database through the CUASHI HIS data portal, (2) DBHYDRO database of SFWMD, (3) the study of Belanger et al. (2004) for groundwater seepage and nitrogen load in southern Indian River Lagoon, and (4) the study of Belanger et al. (2009) for nitrogen load from septic systems in northern Indian River Lagoon. While the latter two data sources include more recent data, they are far from the modeling areas, especially the study of Belanger et al. (2009) in the northern Indian River Lagoon. Therefore, the data from the latter two sources are used only for qualitative understanding of nitrogen transport, but not for model calibration.

Locations of the six monitoring wells used in the study of Belanger et al. (2004) are shown in Figure 3-8. Time series of ammonium and NO_x concentrations are plotted in Figure 3-18. The NO_x concentrations are significantly smaller than ammonium concentrations, regardless of the sampling depths. It indicates incomplete nitrification process, which is not surprising because all the wells are located on river shores with a shallow water table.

The study of Belanger et al. (2009) in the northern Indian River Lagoon is specifically to study importance of septic systems to contaminant load. Three residential houses were selected, and a sampling network was designed for the possible nitrogen plumes. Figure 3-19 shows locations of the three sites and the general sampling scheme; Figure 3-20 plots the boxplots of the measured nitrogen concentrations at three field sampling events on 01/02/2009, 03/23/2009, and 06/23/2009. Figure 3-20 shows that, while ammonium concentrations are higher than NO_x concentrations at the Huy and Lounibos residence, NO_x concentrations are higher than ammonium concentrations at the Grimes apartment building. Therefore, it should be cautious to draw a general conclusion that NO_x concentrations are low in the coastal areas.

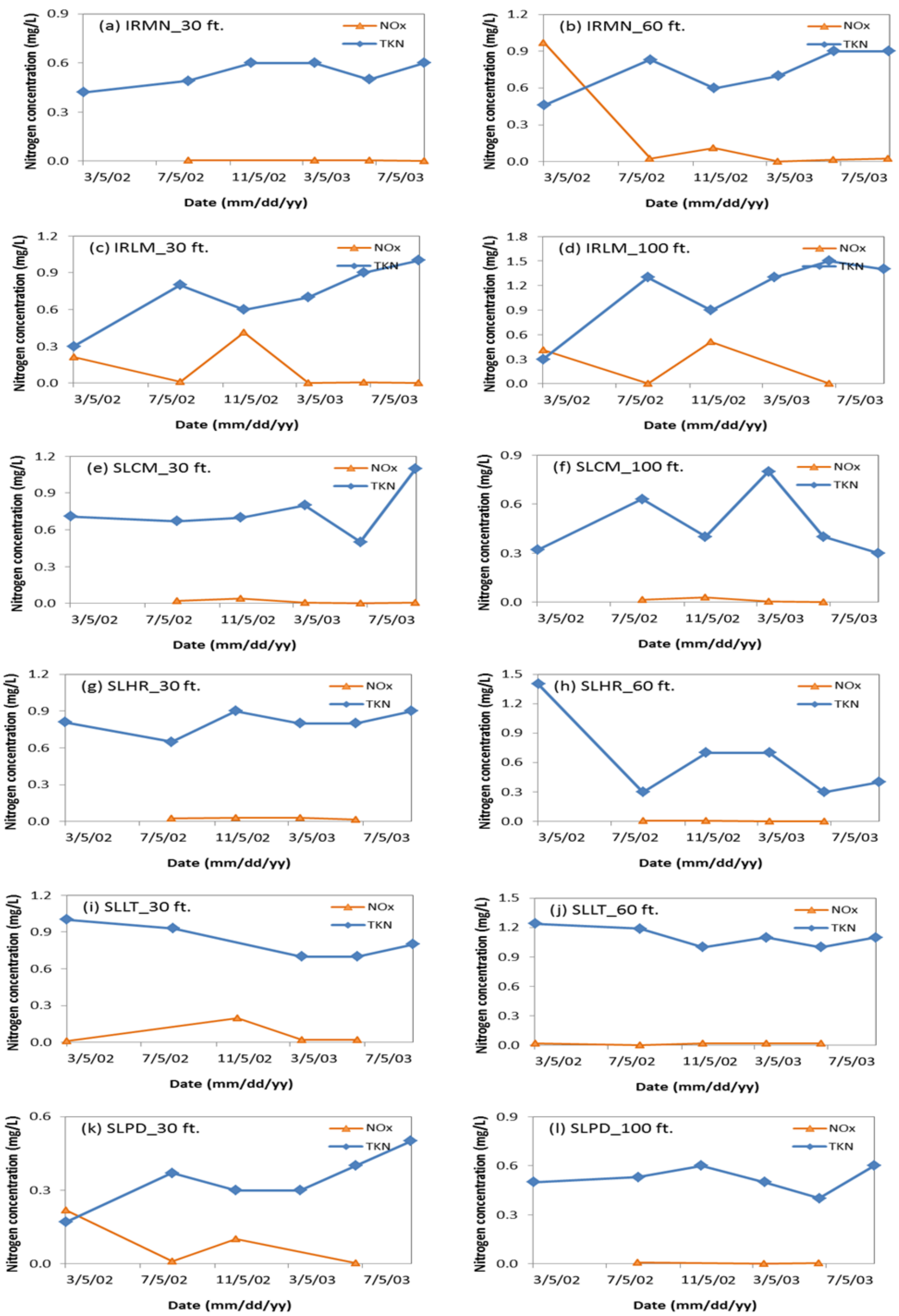

Figure 3-18. Time series of ammonium and NOx concentrations in six wells, whose locations are shown in Figure 3-8.

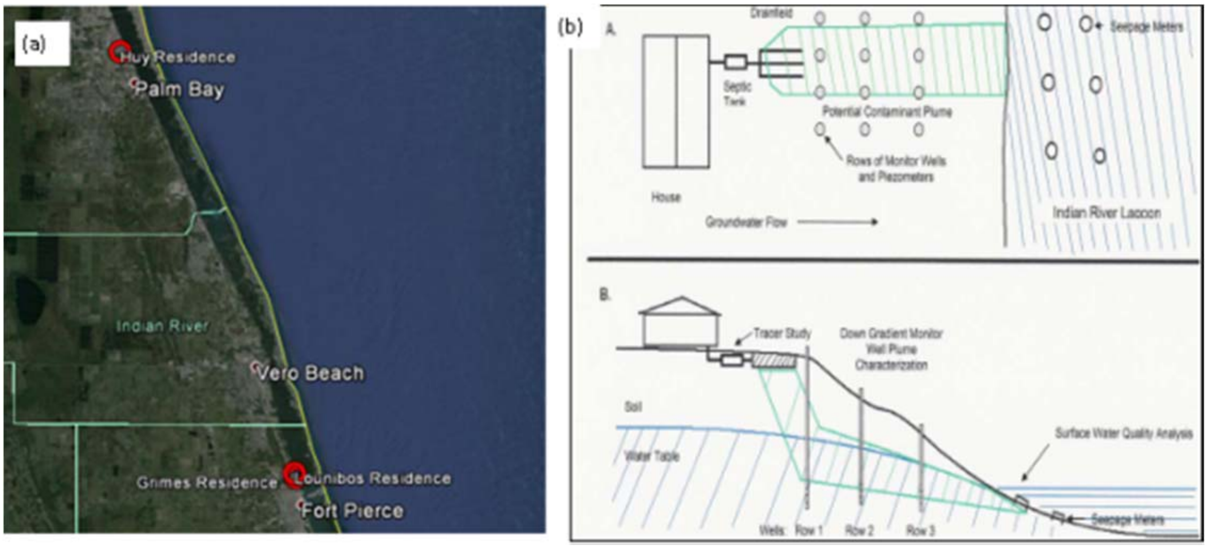

Figure 3-19. Locations of the three residential sites and general sampling scheme (adapted from Belanger et al., 2009).

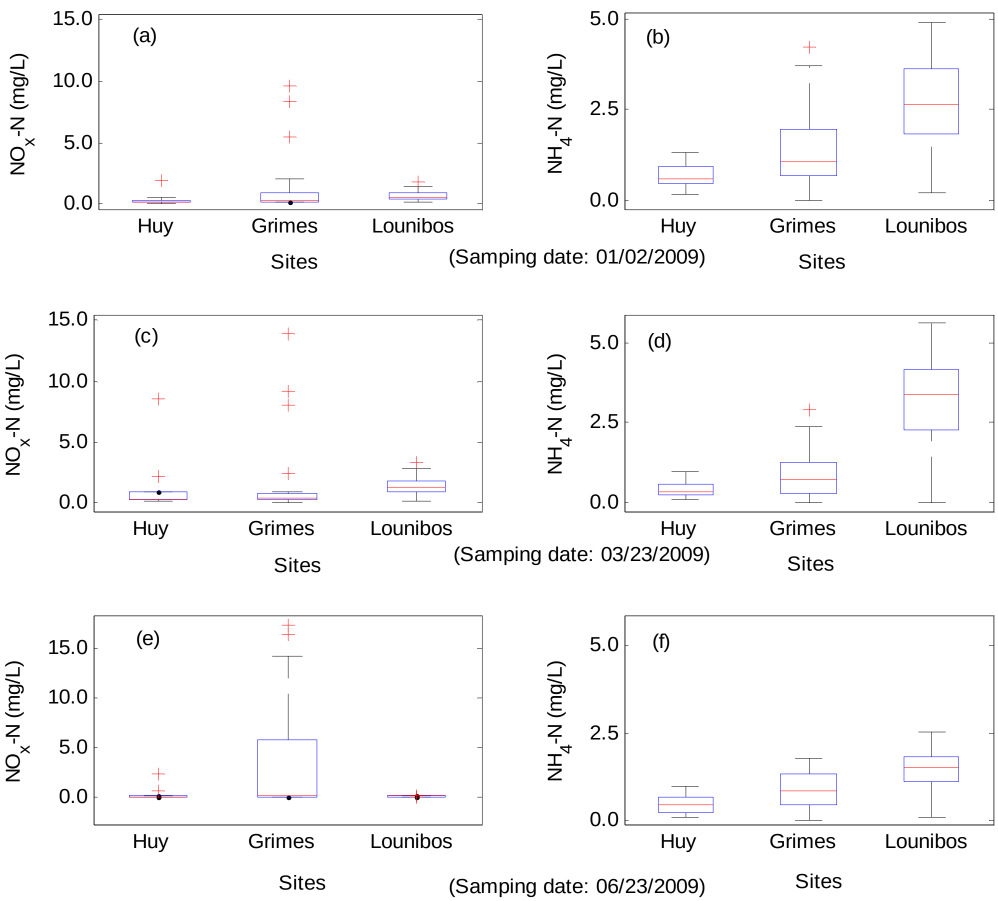

Figure 3-20. Boxplots of NH4 and NOx concentrations measured at three residential houses (Huy, Grimes, and Lounibos) on 01/02/2009, 03/2/2009, and 06/23/2009.

Figure 3-21 shows locations of monitoring wells from which nitrogen data are compiled from the USGS and DBHYDRO database. While the wells are located in or close to the modeling areas, measurements of nitrogen concentration are limited, with more data at USGS wells, PG-23, PG-25, and PG-30.At each of the four wells, four sets of measurements of ammonium and NO_x concentrations from 1976 to 1977 are available. These data show that ammonium concentrations are higher than NO_x concentrations (Figure 3-22).

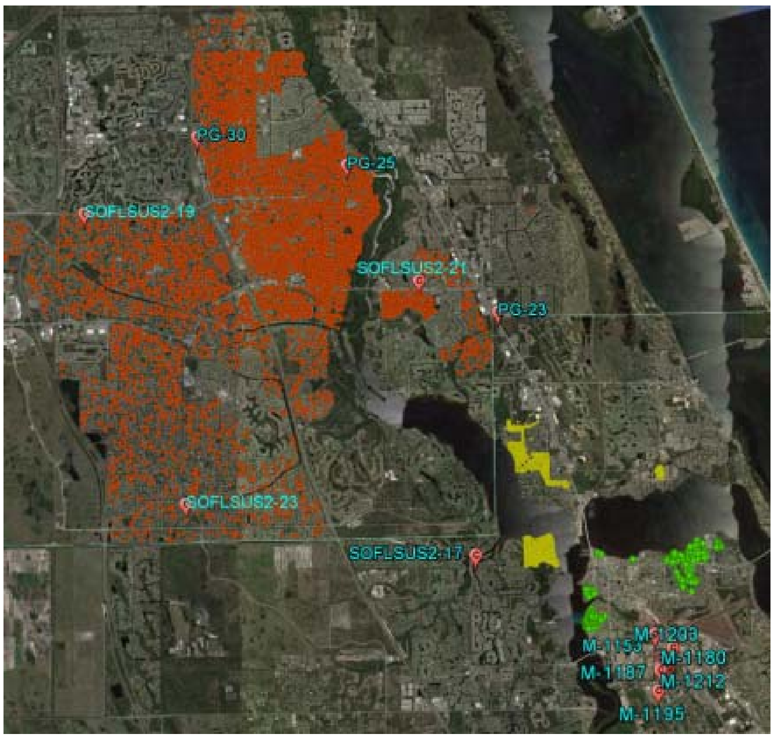

Figure 3-21. Location of monitoring wells with groundwater quality data compiled from USGS and DBHYDRO databases.

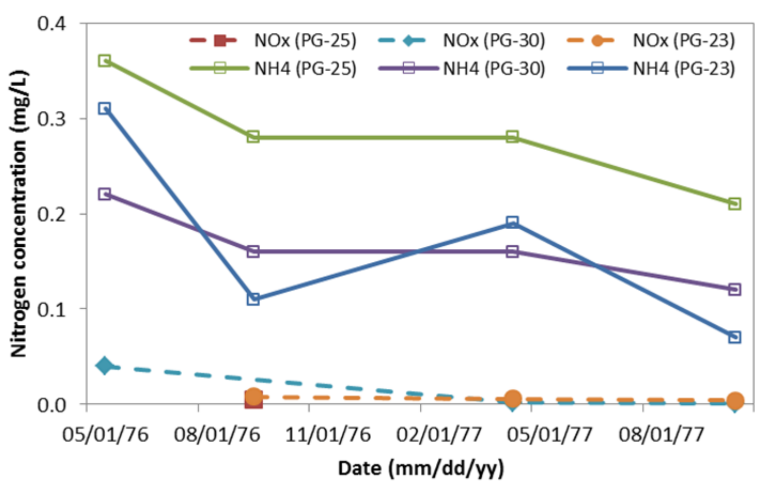

Figure 3-22. Nitrogen concentrations at three USGS wells (PG-23, PG-25, and PG-30) measured in 1976-1977.

Figure 3-23. Locations of USGS monitoring wells (left: PG-23, middle: PG-25, right: PG-30) and septic systems surrounding the wells.

For the measurements of nitrogen concentration at the three USGS wells (PG-23, PG-25, and PG-30), only those at well PG-25 are selected for model calibration, after examining the spatial relation between the wells and septic systems. As shown in Figure 3-23, there are no residential houses around well PG-23, and the residential houses around well PG-30 are on sewer. Only well PG-25 is surrounded by houses that are still on septic systems or were converted to sewer in last several years. While the concentrations at well PG-25 were measured in 1976-1977, they are comparable with recent data from USGS wells SOFLSUS2-17, SOFLSUS2-19, SOFLSUS2-21, and SOFLSUS2-23. Figure 3-24 shows that ammonium concentrations at the four USGS wells are between 0.2 and 0.4 mg/l, comparable with those from well PG-25 (Figure 3-22). Therefore, the data from well PG-25 are used for calibrating ArcNLET transport model. Note that well SOFLSUS2-17 is located in Martin County and thus used to calibrate the ArcNLET transport model in Martin County.

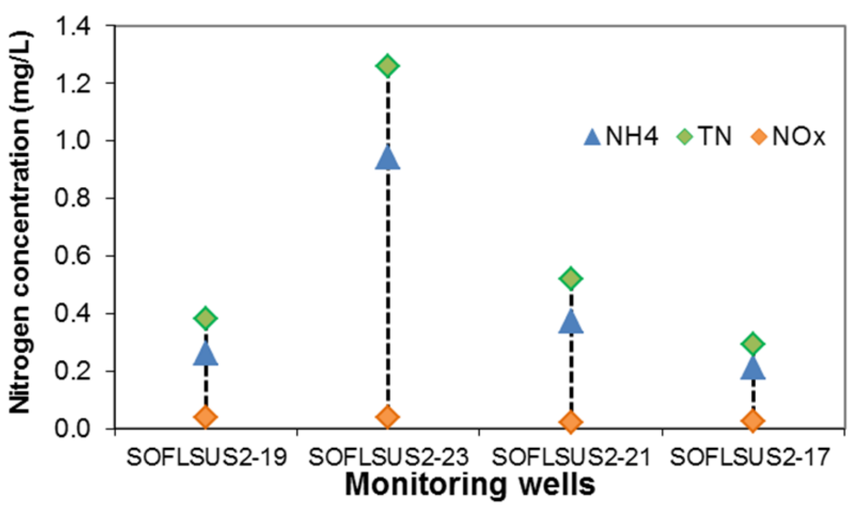

Figure 3-24. Nitrogen concentrations in four monitoring wells (measured in 10/2008 or 11/2008).

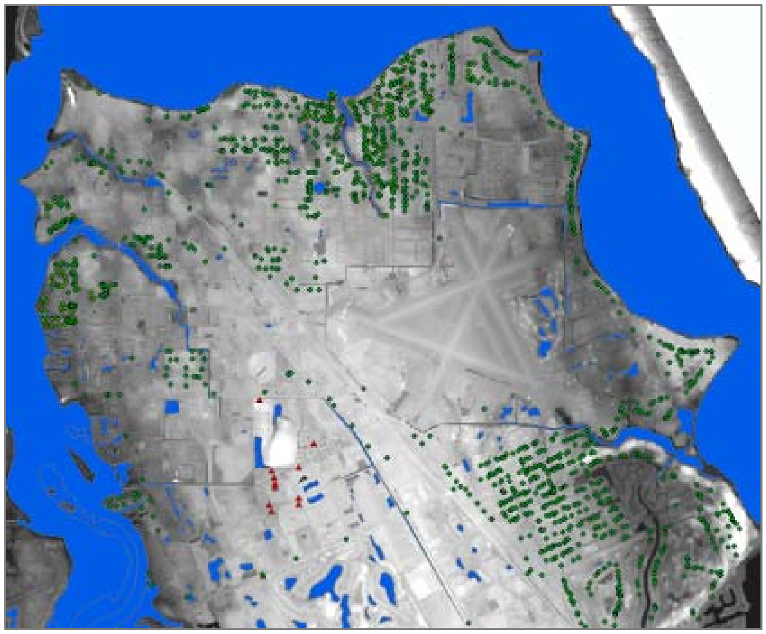

Figure 3-25. Locations of monitoring wells (red triangles) with nitrogen concentration data compiled from DBHYDRO. Septic systems in this area are denoted by green diamonds.

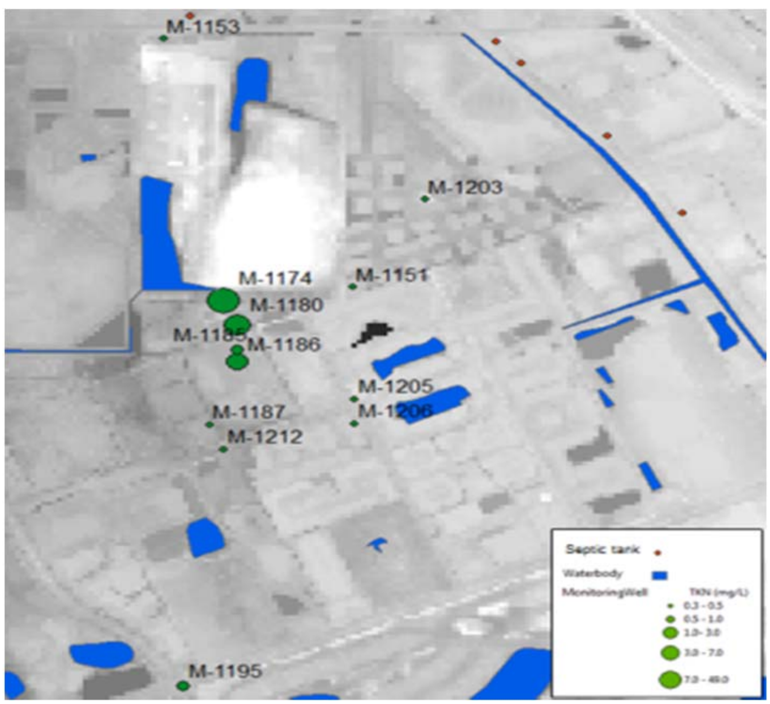

Figure 3-26. Spatial distribution of TKN concentration measured in the period of 1986-1987 in the City of Stuart.

For the City of Stuart, twelve wells with nitrogen concentration data are found in DBHYDRO (Figure 3-25), and the data indicate that TKN concentrations are higher than NO_x concentrations. However, no wells are used for calibration for two reasons. First, except for well M-1153, the other eleven wells are not located in the area with removed septic systems. The other reason is that the TKN data from the several wells are suspicious. As shown in Figure 3-26, the TKN concentrations at M-1174 are as high as 49 mg/L. This value is too high and cannot be explained by septic systems, recalling the concnetrations of Belanger et al. (2009) specifically for septic systems (Figure 3-20). Because of this high value, TKN concentrations at the wells south to well M-1174 are also high. As a result, these wells are not used for calibrating the ArcNLET transport model. Instead, ArcNLET modeling for the City of Stuart is conducted using the parameters calibrated against data from well SOFLSUS2-17 located in Martin County but close to the City of Stuart.

Figure 3-27 shows locations of the five monitoring wells selected for calibrating the ArcNLET transport model. Among these wells, wells SOFLSUS2-19, SOFLSUS2-21, SOFLSUS2-23, and PG-25 are used to calibrate the ArcNLET model in the City of Port St. Lucie, and well SOFLSUS2-17 for Martin County. Table 3-2 lists nitrogen concentrations at the selected wells. The calibration target is the concentration of TN or (DIN), depending on availability of ammonium or TKN concentrations. The values at well PG-25 are the average of the four measurements plotted in Figure 3-22.

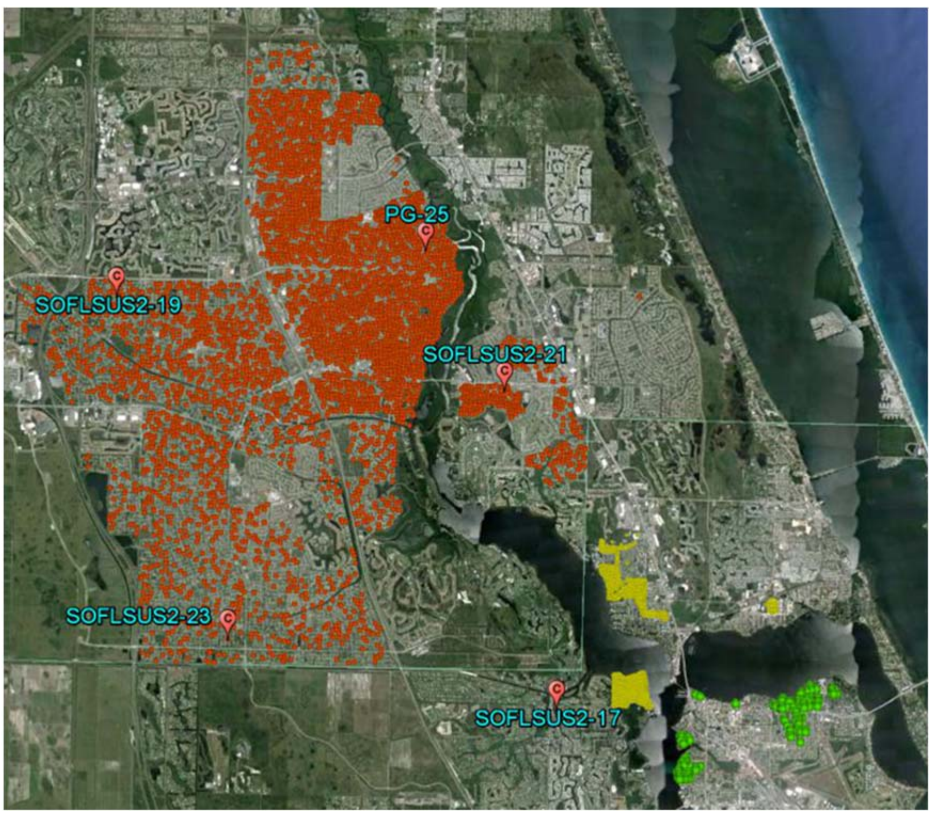

Figure 3-27. Locations of monitoring wells with nitrogen concentrations used for calibrating ArcNLET transport model.

Table 3-2. Nitrogen concentrations (mg/l) at monitoring wells selected for calibrating the ArcNLET transport model. Dissolved inorganic nitrogen (DIN) concentration at well PG-25 and total nitrogen (TN) concentration at the other four wells are used as the calibration target.

Area	**Wells**	**Data source**	**NO_x**	**NH4**	**TN/DIN**
City of Port St. Lucie	SOFLSUS2-19	USGS	0.040	0.220	0.380
	SOFLSUS2-21	USGS	0.021	0.349	0.520
	SOFLSUS2-23	USGS	0.040	0.900	1.260
	PG-25	USGS	0.005	0.283	0.288
Martin County	SOFLSUS2-17	USGS	0.002	0.210	0.290

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4. ARCNLET MODEL SETUP, CALIBRATION, AND LOAD ESTIMATION

This section starts with a brief description of the data needed for ArcNLET modeling in Section 4.1, followed by identification of calibrated parameters of the ArcNLET model in Section 4.2. Model calibration results for the City of Port St. Lucie and Martin County are given in Sections 4.3. The load estimates obtained using the calibrated model for the City of Port St. Lucie, City of Stuart, and Martin County are discussed in Section 4.4, and the load estimates are evaluated in Section 4.5. At last, the estimates are discussed in the BMAP context in Section 4.6 to provide an example of using the ArcNLET load estimates in support of BMAP and TMDL implementation.

4.1. Data Needed for ArcNLET Modeling

Data needed for ArcNLET modeling include DEM, locations of removed septic systems, locations of surface water bodies, and values and spatial distributions of hydraulic conductivity and porosity. These data are collected with assistance of colleagues at FDEP, the City of Port St. Lucie, the City of Stuart, and Martin County. The data need to be prepared in the ArcGIS data format so that they can be used directly by ArcNLET. The basic procedure of preparing the data is given below; more details of the procedure are referred to ArcNLET user's manual (Rios et al., 2011a) and application manual (Wang et al., 2011). The site-specific transport parameter values (e.g., source plane concentration, dispersivities, and first-order decay coefficient of denitrification) are not available but obtained by model calibration as described in Section 4.2.

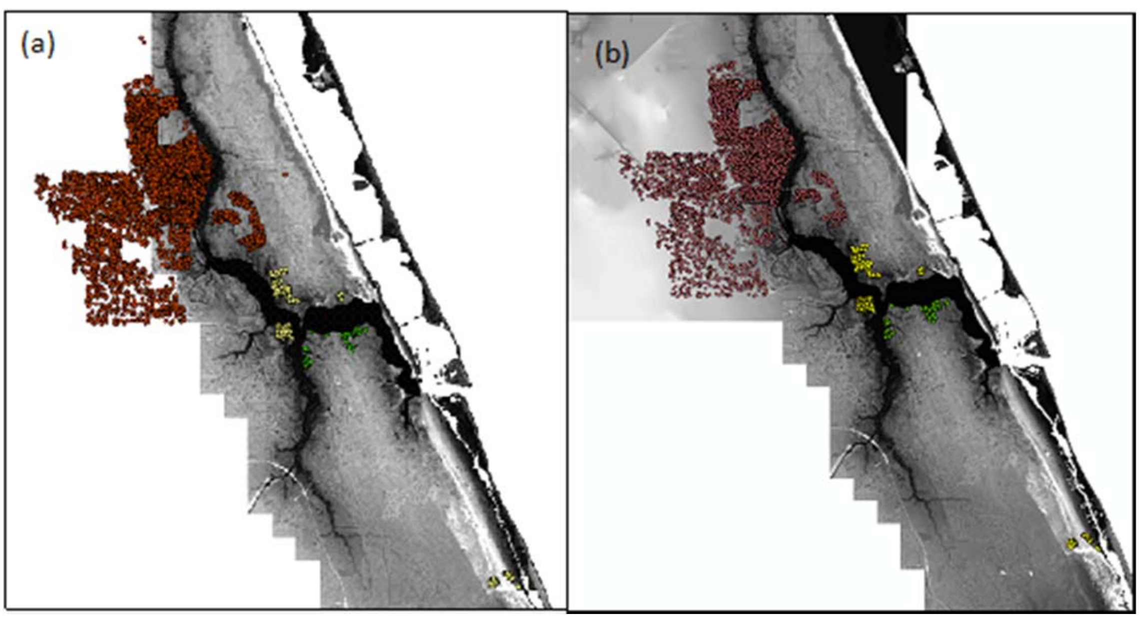

Figure 4-1. (a) LiDAR DEM with resolution of 10×10 feet2 for the coastal areas of St. Lucie and Martine Counties and (b) final DEM of resolution of 10×10 m2 after merging the LiDAR DEM and regular DEM with resolution of 10×10 m2. Locations of removed septic systems in the City of Port St. Lucie (red), the City of Stuart (green), and Martin County (yellow) are also shown.

The DEM data used in this study includes LiDAR DEM and regular DEM. The LiDAR DEM with resolution of 10 × 10 $feet^2$ is downloaded from SFWMD's GIS Data Catalog (http://www.floridadisaster.org/gis/lidar/) for the coastal part of St. Lucie and Martin Counties, as shown in Figure 4-1a. For the rest of the modeling area where the LiDAR DEM is unavailable, regular DEM with the resolution of 10 × 10 m^2 is downloaded from the USGS's National Elevation Dataset (http://ned.usgs.gov/). By applying the mosaic function, the LiDAR DEM and the regular DEM are merged together with the resolution of 10 × 10 m^2. The final DEM used in this study is shown in Figure 4-1b. It should be noted that the low resolution of the regular DEM affects accuracy of the numerical simulation, because it cannot reveal spatial variability at the scale smaller than 10 ×10 m^2, the DEM resolution.

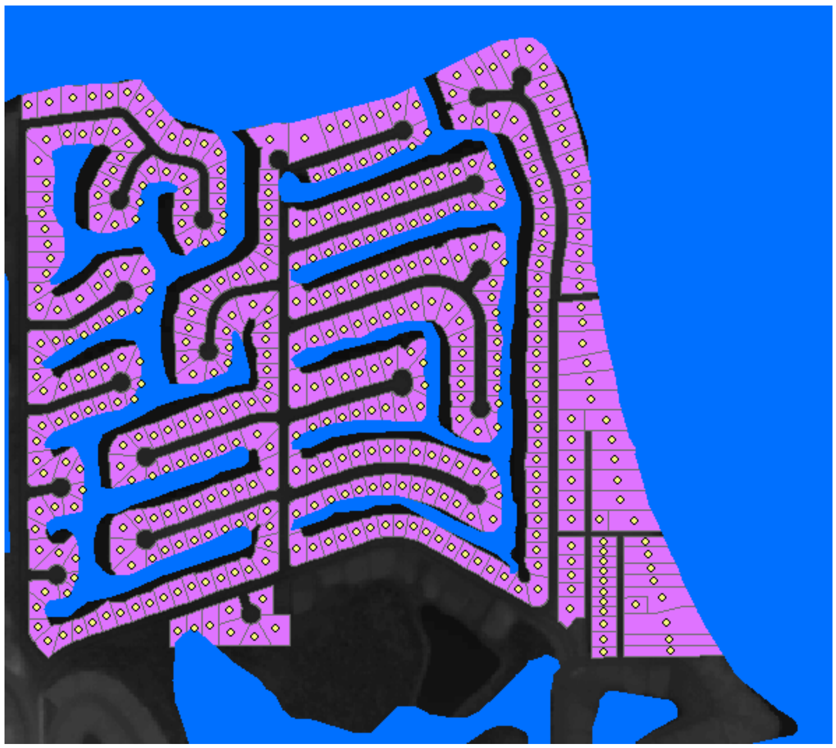

Figure 4-2. Locations of parcel polygons and assumed locations of removed septic systems at Seagate Harbor of Martin County. Water bodies are in blue and DEM are in gray.

The locations of removed and functioning septic systems are provided by Dale Majewski in the City of Port St. Lucie, William Griffin in the City of Stuart, and Dianne Hughes in Martin County. The location files are of different data formats and thus processed as follows:

(1) For the City of Port St. Lucie, a parcel polygon file is provided by Dale Majewski for both removed and functioning septic systems. The septic system locations are assumed to be at the geometric center of the parcel polygons.

(2) For the City of Stuart, a parcel polygon file for removed septic systems is provided by William Griffin, and it is processed in the manner same as that for the City of Port St. Lucie.

(3) For Martin County, a parcel polygon file of removed septic systems is provided by Dianne Hughes, and it is processed in the manner same as that for the City of Port St. Lucie.

(4) William Griffin from the City of Stuart also provides a point file for locations of functioning septic systems in the entire state (adapted from the Department of Health). This file is useful to the model calibration and a scenario analysis described in Section 4.5.

Figure 4-2 demonstrates the parcel polygons and assumed locations of removed septic systems in Seagate Harbor of Martin County. The locations of all the removed septic systems are plotted in Figure 3-2 of Section 3.2. During the model calibration, the locations of removed and functioning septic systems are used in different ways, depending on when the calibration data were measured at the monitoring wells listed in Table 3-2. Given that nitrogen concentrations were measured in 2008 from wells SOFLSUS2-19, SOFLSUS2-21, and SOFLSUS2-23 in the City of Port St. Lucie and from well SOFLSUS2-17 in Martin County, and further considering that septic system removal occurred since 2000 in the City Port St. Lucie and since 2008 in the City of Stuart, only functioning septic systems are used for the calibration to simulate the measured nitrogen concentrations. Since nitrogen concentrations at well PG-25 (in the City of Port St. Lucie) were measured during 1976-1977, both removal and functioning septic systems are used for model calibration.

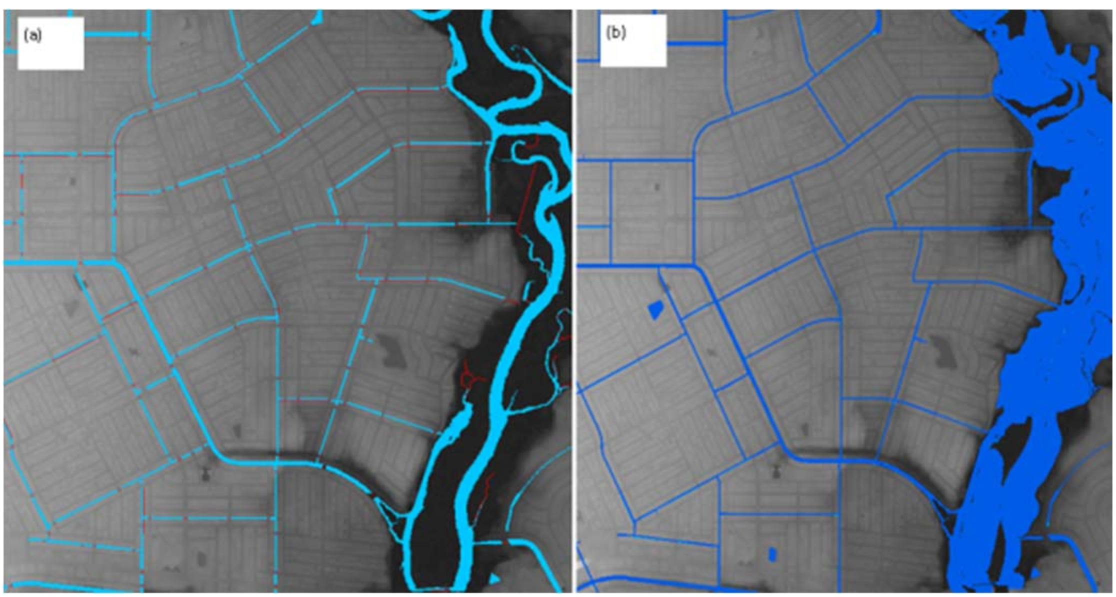

Figure 4-3. ArcGIS files of (a) canals and (b) merged surface water bodies in the City of Port St. Lucie. The flow lines (red in figure a) from the USGS NHD database are used to revise the canal data to make the canals continuous.

The locations of surface water bodies are obtained from two sources: (1) the USGS National Hydrography Dataset (NHD, http://nhd.usgs.gov/) for all the modeling areas, and (2) the canal data of the City of Port St. Lucie provided by Marcy Policastro. As shown in Figure 4-3a, the canals are not continuous (due to overlaid objectives such as bridges) and thus modified using the flowline layer of the NHD data to make the canals continuous. Note that the flow lines in the NHD database are not useful to ArcNLET modeling, because the lines do not have width. The NHD data and revised canal data are merged to form the final ArcGIS layer of surface water bodies, which is shown in Figure 4-3b for the City of Port St. Lucie as an example.

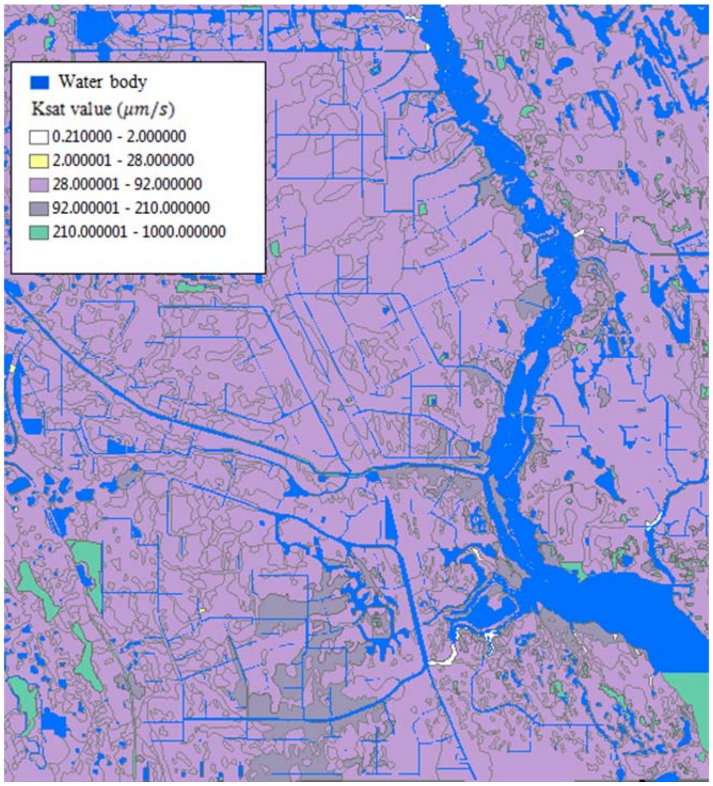

Figure 4-4. Spatial distribution of hydraulic conductivity generated from the SSURGO soil database for the City of Port St. Lucie.

Three steps are needed to prepare hydraulic conductivity data for ArcNLET modeling. The first step is to download the soil data for St. Lucie and Martin Counties from the SSURGO soil database (http://soils.usda.gov/survey/geography/ssurgo). Subsequently, the polygons of heterogeneous hydraulic conductivity is generated by aggregating soil data from horizons to components and then to units. Afterward, by using the "polygon to raster" function of ArcGIS, the polygon file of hydraulic conductivity is converted into a raster file used by ArcNLET. The details of these operations are referred to Wang et al. (2011). The resulting raster file of hydraulic conductivity data for the City of Port St. Lucie is shown in Figure 4-4 as an example.

Since the SSURGO soil databases of St. Lucie and Martin counties do not include porosity data, a literature value of 0.37 for the Indian River Lagoon (Smith et al., 2008) is used. The porosity is assumed to be a constant at all the modeling sites.

4.2. Calibrated and Uncalibrated ArcNLET Model Parameters

Following Wang et al. (2011, 2013), calibration of ArcNLET model is conducted by manually adjusting model parameters to match smoothed DEM values to observed hydraulic heads for the ArcNLET flow model and to match simulated nitrogen concentrations to observed concentrations for the ArcNLET transport model. The details of the calibration data of hydraulic head and nitrate concentration are given in Section 3.

The calibrated ArcNLET parameters are the smoothing factor, source plan concentration (C_0), longitudinal dispersivity (α_L), horizontal transverse dispersivity (α_{TH}), and first-order decay coefficient of denitrification (k). Table 4-1 lists the initial values and ranges of the parameters obtained from literature and our previous experience of using ArcNLET. The reasons of selecting the initial value and the ranges are as follows:

(1) **Smoothing factor**. This parameter is specific to ArcNLET. It is used to smooth DEM to generate the shape of the water table. During the calibration, this parameter value is adjusted so that a 1:1 slope between observed hydraulic heads and corresponding smoothed DEMs can be obtained. A larger value of smoothing factor results in a smoother shape of the water table and thus smaller hydraulic gradient. The parameter is site specific and strongly correlated to site topography. A fine resolution of topography requires a large value of smoothing factor. Rios (2010) tested several smoothing factors for a groundwater model developed for the U.S Naval Air Station (NAS) in Jacksonville. The groundwater model was calibrated by Davis et al. (1996) and the calibrated water table was used as the reference to evaluate the best smoothing factor. It was found that a value of 50 yielded a good approximation to the water table (Rios, 2010). Therefore, the initial value is set as the average value of 50. The smoothing factor is assumed to follow uniform distribution and the range is set as 20 ~ 80 empirically.

(2) **Source plane concentration** (C_0). Nitrogen enters groundwater through the source plan shown in Figure 2-1. As shown in equation (4), the simulated nitrogen concentration is linearly proportional to C_0. There is no field measurement of this parameter value in the modeling areas. A review article of McCray et al. (2005) suggested a range of 25~80 mg/L. Valiela et al. (1997) gave the value of 44 mg/L. A value of total nitrogen concentration of 39 mg/L at the edge of the drainfield was assumed for conventional septic systems (U.S. EPA, 2013). For conventional septic tanks in Florida, an average concentration of 45 mg/L was estimated in Wakulla (Harden et al., 2010) and 35 mg/L of total nitrogen was estimated in Santa Fe River Basin (BMAP, 2012). The value of 45 mg/L is used as the initial value.

(3) **Longitudinal dispersivity (α_L)**. There is no field measurement of this parameter, and literature values are used. While Dann (1996) used the range of 1.2 ~ 7.5 m for α_L in Pinellas County in Florida, Merritt (1996) gave a calibrated value of 76 m for the Biscayne aquifer in Florida. In Gelhar et al. (1992), a range between 6 and 170 m was reported. Therefore, a range of 1 ~100m is set for α_L in this study, and the initial value is 10 m, which was obtained in the ArcNLET modeling by Wang et al. (2013).

(4) **Horizontal transverse dispersivity (α_{TH})**. While the ratio between α_L and α_{TH} is 10 as a rule of thumb, the ratio varies at different sites (Rehfeldt, et al, 1992). For example Merritt (1996) calibrated the two parameters at the Biscayne aquifer and obtained 76 m and 0.03 m for α_L and α_{TH}, respectively. Ratios between 5:1 and 100:1 have been reported in literature (Delgado, 2007). Therefore, α_{TH} is taken as 1/5 ~ 1/100 of α_L. The initial value is set as 1m, 1/10 of the initial value of α_L.

(5) **First-order decay coefficient of denitrification (k)**. There is no site-specific measurement of this parameter in the modeling areas. Tesoriero and Puckett (2011) gave a range of 5.4×10^{-5} ~ 7.3×10^{-4} d^{-1} for shallow aquifers. It is significantly smaller than the range of 0.004 ~ 2.27 d^{-1} given in McCray et al. (2005). Since the soil organic carbon in the SSURGO database is low (2.014% and 1.129% for Port St. Lucie and Stuart, respectively), it is reasonable to set a small range for the parameter. According to the model calibration of Wang et al. (2012), the maximum value of 0.015 d^{-1} is used, and range is 5.4×10^{-5} ~ 0.015 d^{-1}. The initial value is taken as 0.005 d^{-1}, which was obtained in the ArcNLET modeling by Wang et al. (2013).

The initial values are not critical, as the parameter values are adjusted during the calibration. The ranges can be adjusted if more data and information bring more information of the parameters.

Table 4-1. Ranges, initial values, and calibrated values of ArcNLET model parameters for all the sites. Calibration of transport parameters for the City of Stuart and Calibration of smoothing factor for Martin County is not conducted due to lack of data. The calibrated transport parameters for Martin County are used for the City of Stuart, and the calibrated smoothing factor of Port St. Lucie and Stuart Cities are used for Martin County.

Parameter	Range	Initial Value	The City of Port St. Lucie	The City of Stuart	Martin County
Smoothing factor	20 to 80	50	40	40	-
C_0 (mg/L)	25 ~ 80	45	40	-	40
α_L (m)	1 ~ 100	10	60	-	35
α_{TH} (m)	1/5 ~ 1/100 of α_L	1	1.6	-	1.1
k (d^{-1})	5.4×10^{-5} ~ 0.015	0.005	0.0011	-	0.001

The un-calibrated parameters are hydraulic conductivity, porosity, and the inflow mass (M_{in} (g/d) of equation 5) from a septic system to groundwater. In ArcNLET modeling, there are two equivalent ways of handling the inflow mass: (1) to fix M_{in} and evaluate Z using equation 5 (Z is the source plan height needed for calculating the mass of denitrification and load), and (2) to fix Z and calculate M_{in} using equation 5. In this study, the first option is used, and the value of inflow mass is approximated as nitrogen release per person per day (4.8 kg/yr, a review value in Table 5 from Valiela et al. (1997)) × people/house (2.5 on average according recent census data) × 0.7 (not lost in septic tanks and leach fields, according to a report of MACTEC, 2007). The number of 2.5 people per house is the average of 2.4 and 2.6 people per house in Martin and St. Lucie counties (census data available at http://quickfacts.census.gov/qfd/states/12/12111.html). Therefore, the input mass from septic to groundwater is 8.4 kg/yr, i.e., 23.0 g/d. This estimate is larger than the most literature values listed in Table 4-2 but close to 21.7 g/d reported in the Wekiva study (Roeder, 2008). It is also close to the value of 22.4 g/d calculated according to Anderson (2006) as 11.2 g/person/d × 2.5 persons/household × 80% not lost in failed septic systems. This value is used for all the individual septic systems in the modeling sites. Site-specific M_{in} should be used if it can be estimated using site-specific data and information. While the M_{in} value is not calibrated, its estimate is subject to large uncertainty. In the study of Geay (2004), this value ranges from 4.2 to 10.7 kg/yr, because nitrogen reduction rate in septic tanks and drainfields is an uncertain variable. Although the value of 30% (MACTEC, 2007) obtained from the Wekiva study of the Florida Department of Health is used here, the rate is between 25% and 50% in the three sites of the Wekiva study (Roeder, 2008).

Table 4-2. Daily nitrogen loadings per household (g/d) from septic system to groundwater reported in literature and in this study.

Reference	Site Location	Daily N load per household
Koppelman, 1978[a]	Long Island, NY	15.8
Gold et al., 1990[a]	Kingston, RI	21.9
Weiskel and Howes 1991[a]	Buzzards Bay, MA	11.0-18.5
Maizel et al., 1997[a]	Chesapeake Bay	16.4-24.0
Valiela et al., 1997[a]	Waquoit Bay, MA	19.9
Reay, 2004[a]	Coastal Plain, VA	16.4-19.9
Roeder, 2008	Wekiva, FL	21.7
This study	St. Lucie Estuary, FL	23.0

Note: [a] Calculated by multiplying 2.5 persons/houshold to daily nitrogen loadings per person converted from annual loading per person listed in Table 3 of Reay (2004).

4.3. Calibration of ArcNLET Flow and Transport Models

The ArcNLET flow model is calibrated by adjusting the smoothing factor to match the shape of smoothed DEM and the water table. The calibration results are plotted in Figures 4-5a for the City of Port St. Lucie and in Figure 4-5b for the City of Stuart. The locations of monitoring wells for the water table are shown in Figure 3-11 for the City of Port St. Lucie and Figure 3-13 for the City of Stuart; discussion of the calibration data is given in Section 3.5. For the calibration at the two sites, the starting value of smoothing factor is 60, and the calibrated value is 40 (Table 4-1). As shown in Figure 4-5, the slopes of the linear regression lines between smoothed DEM and water table are close to 1 (0.931 for the City of Port St. Lucie and 1.051 for the City of Stuart), and the R^2 values of the linear regression are significant (0.968 for the City of Port St. Lucie and 0.759 for the City of Stuart). While the calibrated value of 40 is smaller than other calibrated values in previous ArcNLET modeling (e.g., 50 for U.S. Naval Air Station, 60 for Eggleston Height, and 100 for Julington Creek in Jacksonville), the smaller value appears to be reasonable because the topography in the coastal areas of Port St. Lucie and Stuart is flatter than that in Jacksonville. The same value of the calibrated smoothing factor in Port St. Lucie and Stuart is not surprising, because the two areas are close to each other. The smoothing factor of 40 is also used for Martin County, where the flow model calibration is not conducted due to the lack of calibration data.

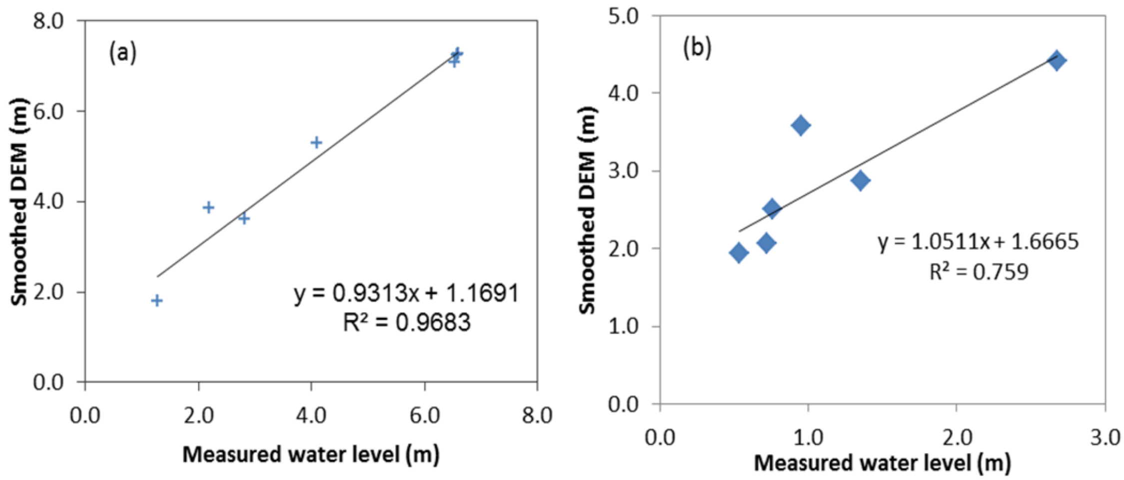

Figure 4-5. Smoothed DEM and measured water level at (a) the City of Port St. Lucie and (b) the City of Stuart. The smoothing factor is 40 for the two areas.

The ArcNLET transport model is calibrated by adjusting the values of source plane concentration, longitudinal dispersivity, horizontal transverse dispersivity, and first-order decay coefficient of denitrification. The septic tank files used for the calibration are prepared in the way described in Section 4.1. The calibrated parameter values for the City of Port St. Lucie and Martin County are listed in Table 4-1. The calibration is not conducted for the City of Stuart due to the lack of data. Table 4-1 shows that the calibrated values of longitudinal dispersivity are different for the two sites, and so are the calibrated values of horizontal transverse dispersivity. The difference may result in different spatial distributions of simulated plumes. Generally speaking, large values of longitudinal dispersivity correspond to long plumes, and large values of horizontal transverse dispersivity to wide plumes. The calibrated values of the first-order decay coefficient of denitrification are similar for the two sites. They are larger than the literature values of $5.4\times10^{-5} \sim 7.3\times10^{-4}$ d^{-1} in Tesoriero and Puckett (2011) but smaller than $0.004 \sim 2.27$ d^{-1} reported in McCray et al. (2005).

Figure 4-6a plots the simulated and measured nitrogen concentrations at the five monitoring wells listed in Table 3-2 for the City of Port St. Lucie and Martin County. Figure 4-6b is plotted in the same manner except that wells PG-25 and SOFLSUS2-21 are excluded from the plot. The reasons of excluding the two wells are given below. For the three remaining wells, the simulated and measured TN concentrations fall on the line with slope close to 1, suggesting a satisfactory model fit.

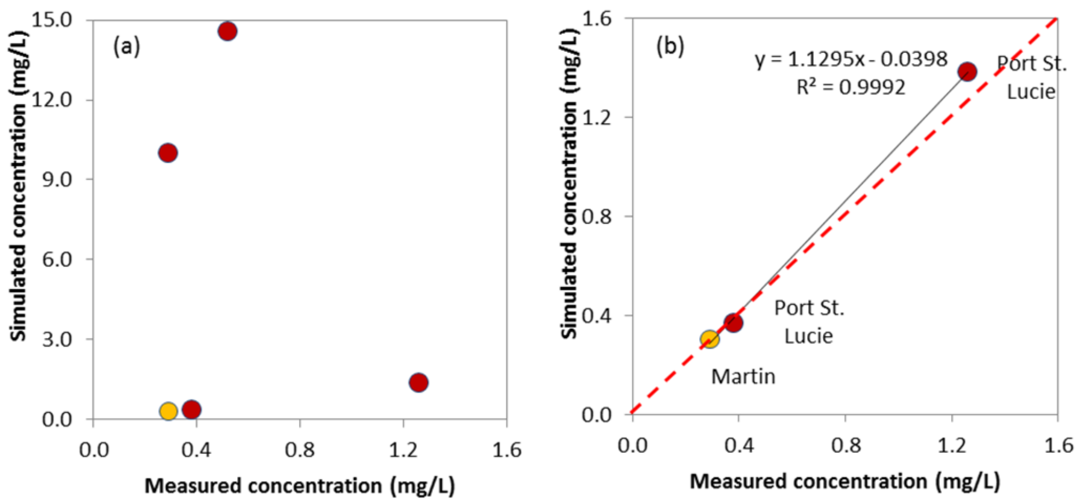

Figure 4-6. Simulated and measured concentrations (a) at all wells listed in Table 3-2 for the City of Port St. Lucie (red) and Martin County (yellow), and (b) excluding wells PG-25 and SOFLSUS2-23 in the City of Port St. Lucie. The 1:1 line is also shown.

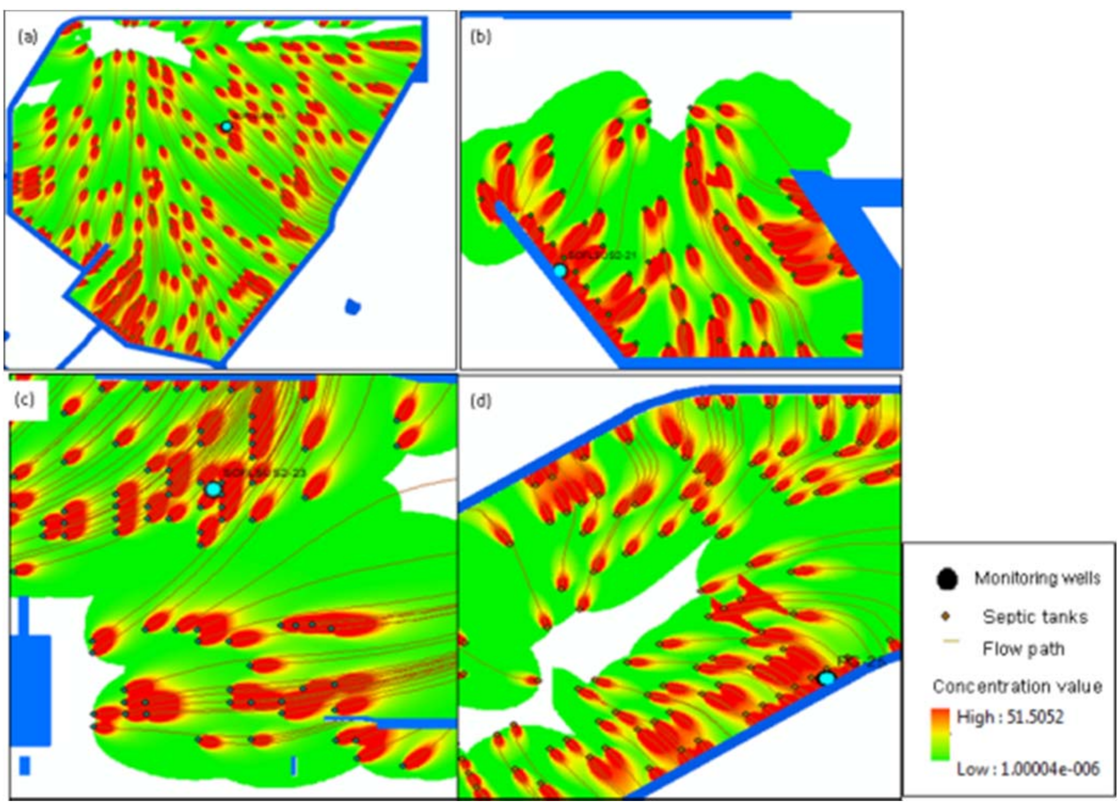

Figure 4-7. Simulated flow paths and plumes by the calibrated ArcNLET model in the City of Port St. Lucie around wells (a) SOFLSUS2-19, (b) SOFLSUS2-21, (c) SOFLSUS2-23, and (d) PG-25. Surface water bodies are plotted in blue.

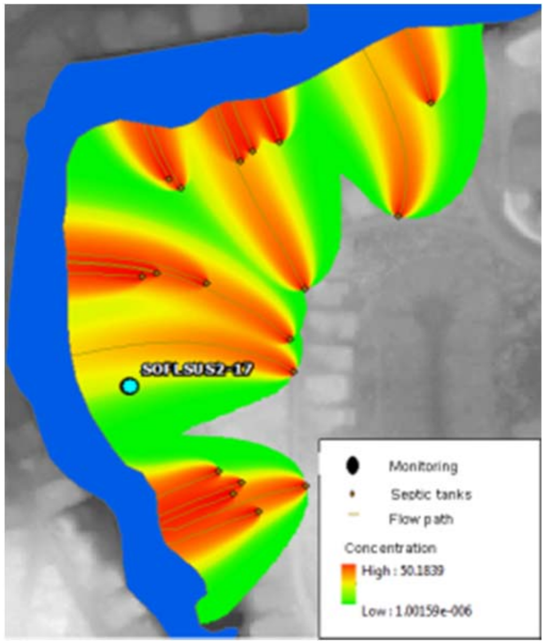

Figure 4-8. Simulated flow paths and plumes by the calibrated ArcNLET model in Martin County around well SOFLSUS2-17 (highlight in figure). Surface water bodies are plotted in blue.

Figure 4-7 plots the flow paths and nitrogen plumes simulated by the calibrated ArcNLET model in the areas close to the four monitoring wells (SOFLSUS2-19, SOFLSUS2-21, SOFLSUS2-23, and PG-25) in the City of Port St. Lucie. Figures 4-8 is plotted in the same manner for the monitoring well in Martin County. The figures manifest the flow pattern of groundwater discharge to surface water bodies and the spatial distribution of the nitrogen plumes. The simulated plume lengths range between 15m (near water bodies) and 60m (far from water bodies). Robertson et al. (1991) reported that that plumes from septic systems may extend to 130 m. Belanger et al. (2009) reported that, for residential houses near water bodies, plume migration distances range between 10m and 30m. As a result, the ArcNLET-simulated plumes are considered to be reasonable.

To understand why the low nitrogen concentrations at monitoring wells SOFLSUS2-21 and PG-25 (0.520 mg/L and 0.288 mg/L, respectively) are not simulated, Figure 4-9 plots the simulated flow paths and plumes at the close vicinity of the four wells. The figure shows that, different from wells SOFLSUS2-19 and SOFLSUS2-23 located between the plumes (Figures 4-9a, c), wells SOFLSUS2-21 and PG-25 are close to the center of plumes where nitrogen concentrations are high. The simulated nitrogen concentrations at wells SOFLSUS2-21 and PG-25 can be reduced by various means, such as increasing the value of the decay coefficient of denitrification or increasing the horizontal transverse dispersivity. These however are not implemented because the amount of nitrogen reduction ratio in the City of Port St. Lucie is comparable with the literature data, as discussed in Section 4.5.

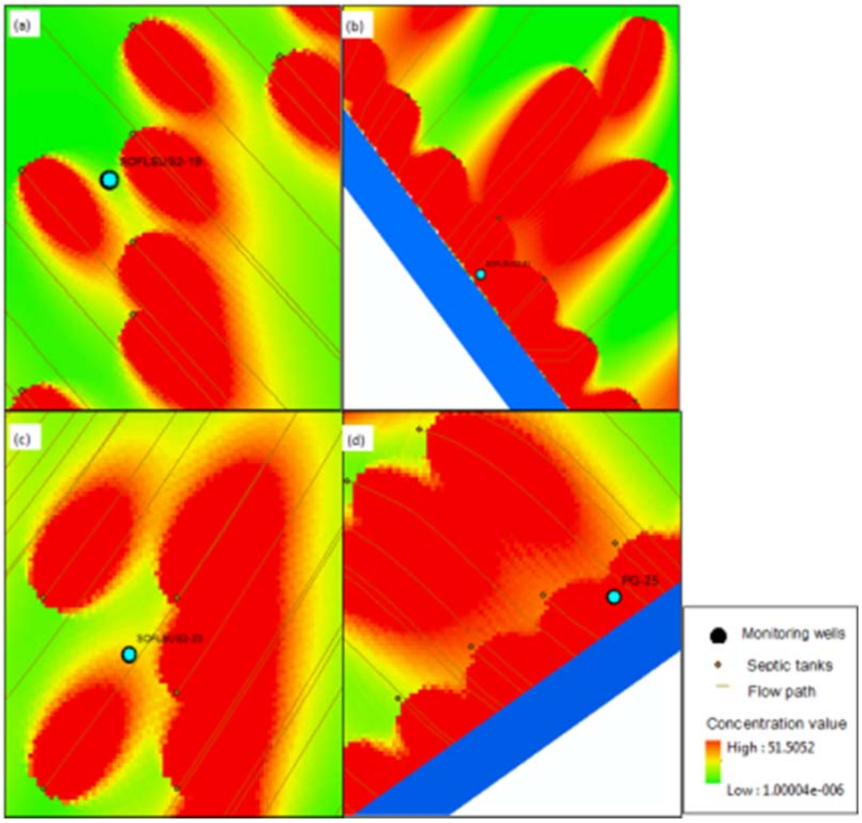

Figure 4-9. Simulated flow paths and plumes at the close vicinity of monitoring wells (a) SOFLSUS2-19, (b) SOFLSUS2-21, (c) SOFLSUS2-23, and (d) PG-25. Surface water bodies are plotted in blue.

4.4. Nitrogen Load Estimation

The calibrated parameters listed in Table 4-1 are used to estimate nitrogen load from the removed septic systems in the City of Port St. Lucie, the City of Stuart, and Martin County. The estimation for the City of Stuart is based on the calibrated parameters of Martin County, since model calibration is not conducted for the City of Stuart.

4.4.1. Nitrogen Load Estimation for the City of Port St. Lucie

Figure 4-10 plots the simulated flow paths from the removed septic systems to surface water bodies. The canals (Figure 4-3) are important to determine groundwater flow direction, because groundwater discharges to nearby canals, not directly to the St. Lucie River. This is a unique feature in the City of Port St. Lucie, as there are not canals in the modeling areas in the City of Stuart and Martin County.

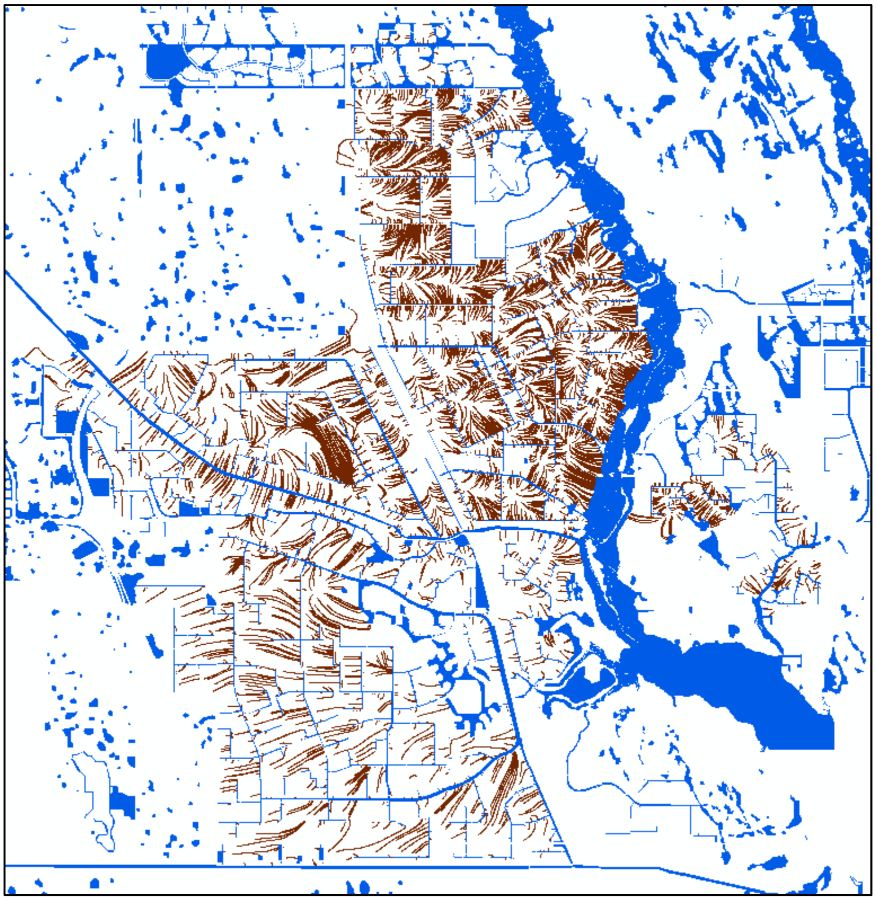

Figure 4-10. Simulated flow paths from removed septic systems to surface water bodies in the City of Port St. Lucie.

Figure 4-11 depicts the simulated nitrogen plumes from the removed septic systems. The estimated load from 5,592 septic systems (9 septic systems are not considered because they are located within water bodies) in the entire modeling area is 42.48 kg/day. The load estimates and the number of contributing septic systems to the individual water bodies are listed in Appendix A. The nitrogen load per septic system is 7.60 g/d. The removal ratio due to denitrification is 67.0%, given that the daily nitrogen load from each septic system is 23.0 g/d (Table 4-2). This ratio is comparable to those reported in the literature, and more discussion is given in Section 4.5.

Among the estimated nitrogen loads, the largest one is 1.375 kg/d to C-24 canal (FID 11) shown in Figure 4-11. The canal is the longest water body in the NHD database, and it receives nitrogen load from 202 removed septic systems. In the NHD database, the North Fork St. Lucie River is separated into eleven water bodies (Figure 4-12), and the estimated load to each segment of the river is listed in Table 4-3. The total load to North Fork St Lucie River is 2.8 kg/d, more than twice as large as that to C-24 canal. However, the load to the river is only 6.7% of the total load in the City of Port St. Lucie, and the number of contributing septic systems is only 5.7% of the total number of the removed septic systems.

The majority of the load is to the canals. The detailed load estimates for the individual water bodies makes it possible for environmental managers to determine areas with high priority of septic removal in planning of future removal.

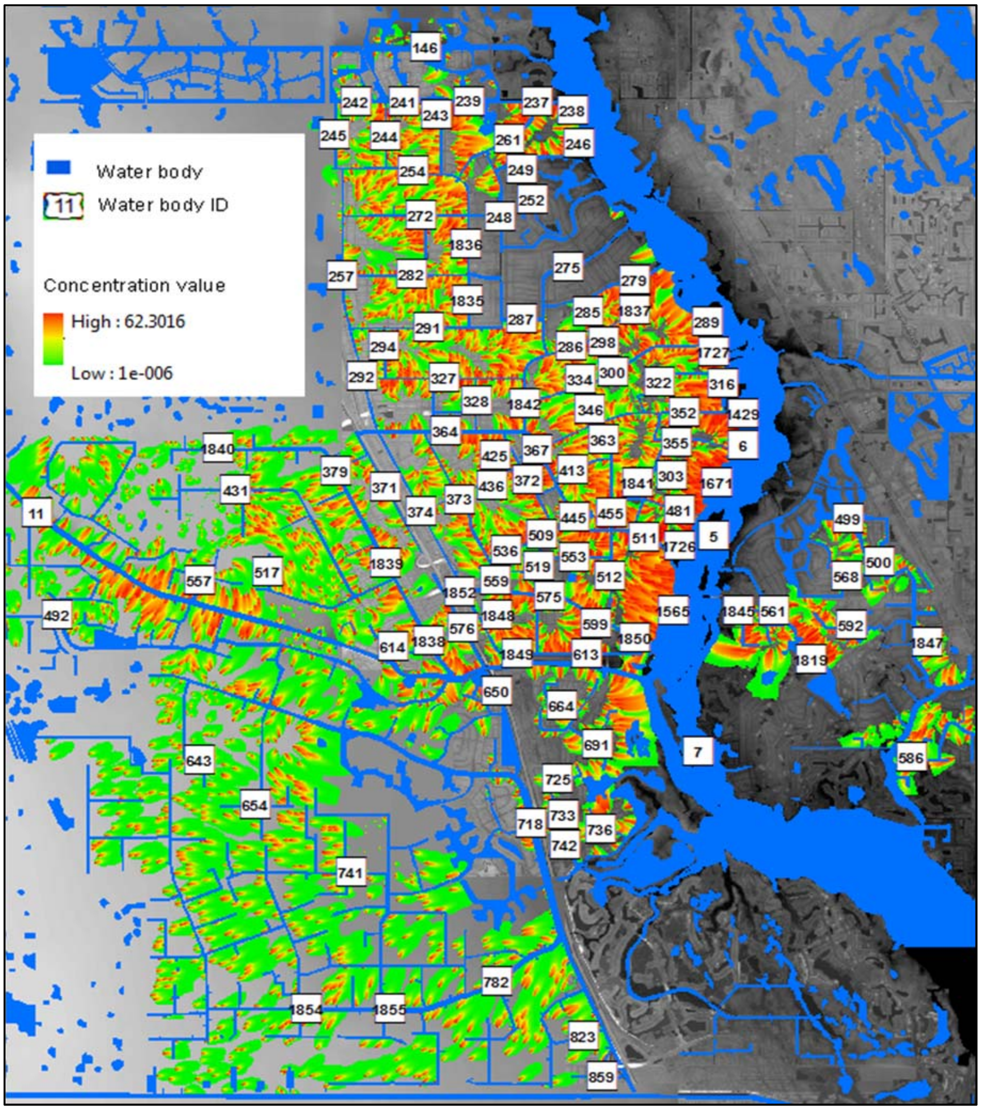

Figure 4-11. Simulated nitrogen plumes from removed septic systems in the City of Port St. Lucie. The FIDs of water bodies with the estimated load larger than 0.05 kg/d are labeled.

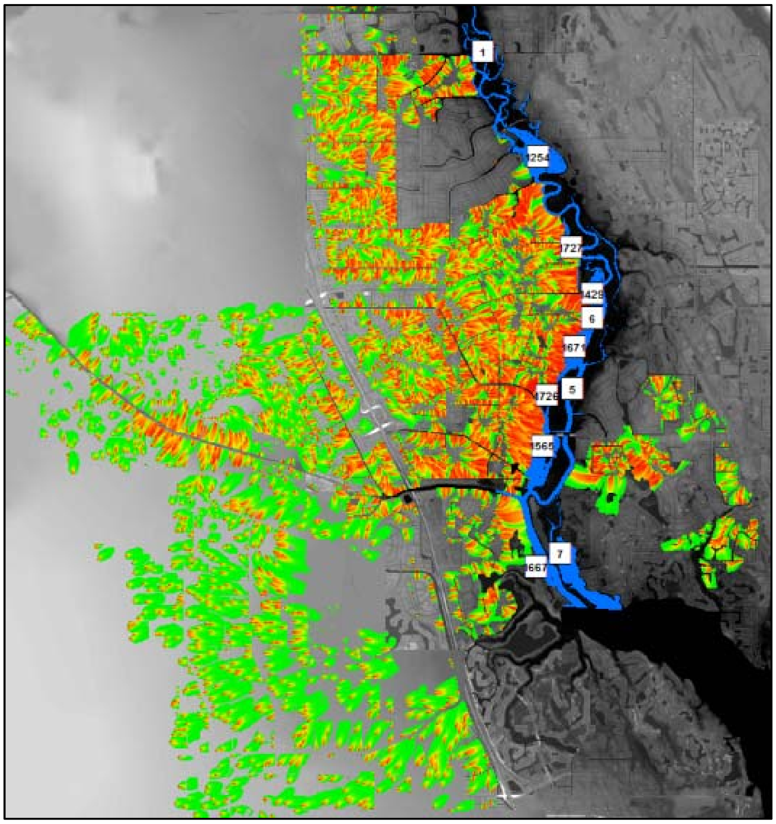

Figure 4-12. FIDs of the water bodies along the North Fork St. Lucie River.

Table 4-3. ArcNLET estimated nitrogen load and numbers of contributing septic systems to individual segments of North Fork St Lucie River. The sum of the load estimates is 6.7% of the total load, and the sum of the contributing septic systems is 5.7% of the total number of removed septic systems in The City of Port St. Lucie.

Water Body FID	Water Body Category	Nitrogen Load to Water bodies (kg/d)	Number of Contributing Septic Systems
1	River	0.024	2
5	River	0.113	22
6	River	0.086	5
7	River	0.249	27
1254	swamp	0.002	1
1429	swamp	0.296	19
1565	swamp	0.470	85
1667	swamp	0.001	1
1671	swamp	0.522	61
1726	River	0.694	62
1727	River	0.384	33
Total		2.842	318
Percentage		6.7%	5.7%

The estimated nitrogen loads are well correlated to water quality of surface water bodies. For example, near stations Sagamore WW, D-14, D-21, and ELCAM Spillway (Figure 3-4), where median values of nitrogen concentrations in surface water are high (Figure 3-5), the simulated nitrogen concentrations are also high, as shown in Figure 4-11. Table 4-4 lists the median nitrogen concentration at the fourteen monitoring stations and the groundwater nitrogen load per unit length along the water bodies corresponding to the monitoring stations. Figure 4-13 shows a linear relation between the median values of surface water nitrogen concentration and logarithm of the nitrogen loads to the surface water bodies. It suggests that nitrogen load from septic systems to surface water bodies is one of the reasons for deterioration of surface water quality. This analysis also illustrates that the spatial variability revealed in the ArcNLET modeling results is useful to nitrogen contaminant management.

Table 4-4. Median nitrogen concentration of surface water and estimated groundwater nitrogen load to the water bodies corresponding to monitoring stations of surface water.

Monitoring Station	Surface Water nitrogen Concentrations (mgL^{-1})	FID of Water Bodies Corresponding to Stations	Length of Water Bodies (m)	Nitrogen Load to Water Bodies ($kg\ d^{-1}$)	Nitrogen loading per unit length ($kg\ d^{-1}\ m^{-1}$)
C-107	1.010	146	1761.0	0.133	7.58E-05
C-108	0.821	239	794.0	0.294	3.71E-04
Sagamore WW	1.476	261	53.0	0.895	1.69E-02
Degan WW	1.119	286	469.0	0.861	1.84E-03
Elcam In	1.098	364	1730.5	0.564	3.26E-04
D-14	1.689	355	53.0	0.549	1.04E-02
D-21	1.594	481	401.5	0.296	7.37E-04
ELCAM SpillWay	1.279	445	1520.0	1.049	6.90E-04
Kingsway WW	1.060	575	2289.0	0.949	4.15E-04
B-95-3	1.135	1847	3448.5	0.445	1.29E-04
B-33	0.983	561	614.0	0.651	1.06E-03
Monterey WW	0.960	1838	2151.5	0.235	1.09E-04
E8	1.095	492	1341.0	0.397	2.96E-04
A23	1.050	753	608.0	-	-

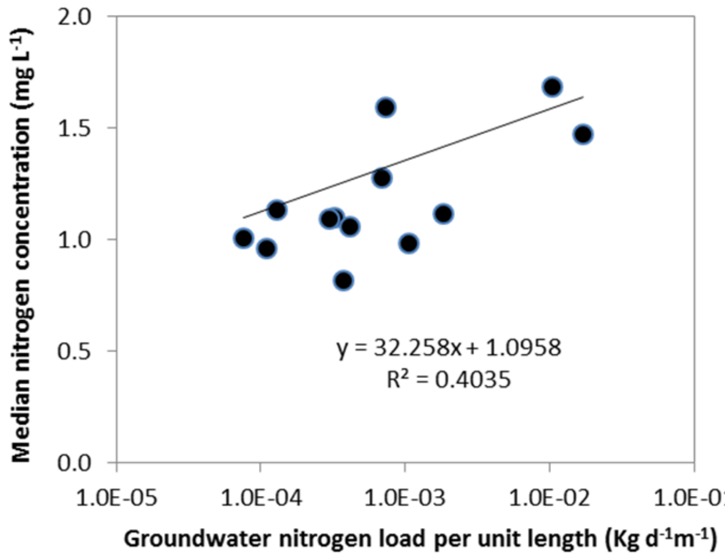

Figure 4-13. Median nitrogen concentration at fourteen monitoring stations and groundwater nitrogen load per unit length along the water bodies corresponding to the monitoring stations. The x-axis is in the logarithm scale.

The water bodies with the top ten largest load estimates are labeled with their FIDs in Figure 4-14, and the estimated loads and numbers of contributing septic systems are listed in Table 4-5. The sum of the load estimates is 24.3% of the total load, and the sum of the contributing septic systems is 20.5% of the total number of removed septic systems. These percentages suggest that reducing nitrogen load to a small number of surface water bodies cannot help effectively reducing nitrogen load the entire modeling area. In addition, Table 4-5 shows that, while the largest load estimate corresponds to the largest number of contributing septic systems, a large number of contributing systems does not necessarily lead to large nitrogen load. For example, the load from 112 septic systems to water body 352 is smaller than the load from 88 septic systems to water body 445. This is not surprising, because the load estimation depends on groundwater flow, solute transport, denitrification, and travel distance and time from septic systems. In other words, the load estimation should not be based on the number of septic systems but needs to consider spatial variability of flow and transport conditions in the modeling areas.

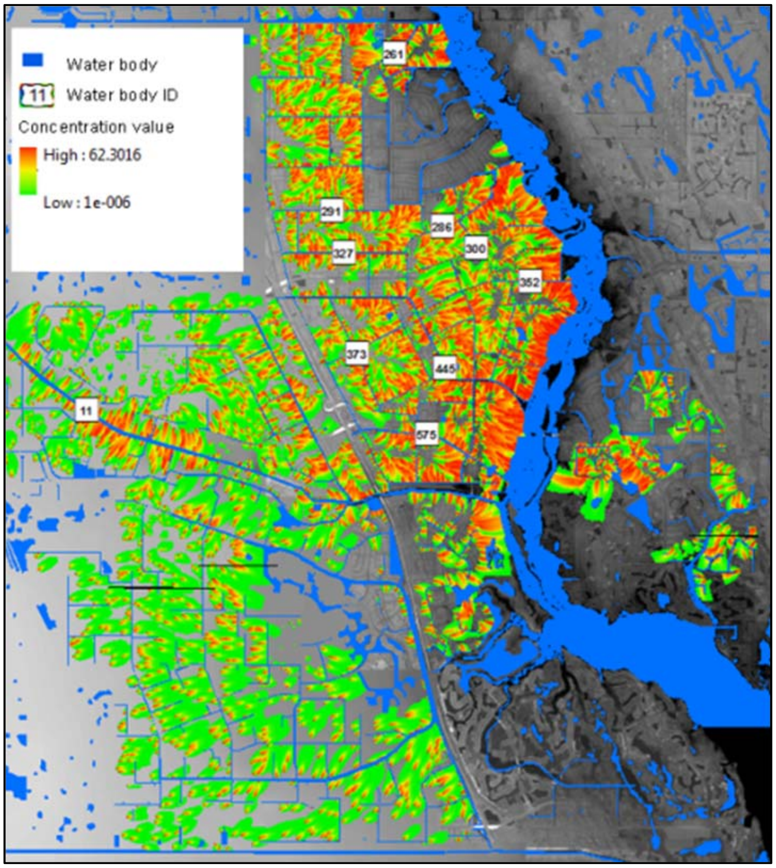

Figure 4-14. Simulated nitrogen plumes from removed septic tanks in the City of Port St. Lucie. The water bodies with the top 10 largest loads are labeled.

Table 4-5. FIDs and numbers of contributing septic systems of the ten largest load estimates for the City of Port St. Lucie. The sum of the load estimates is 24.3% of the total load, and the sum of the contributing septic systems is 20.5% of the total number of removed septic systems in the City of Port St. Lucie.

Water Body FID	Nitrogen load (kg/d)	Number of contributing septic systems
11 (C-24 Canal)	1.375	202
327	1.283	162
445	1.049	88
352	1.047	112
300	1.036	108
575	0.949	88
373	0.948	100
261	0.895	80
291	0.874	92
286	0.861	115
Total	10.317	1147
Percentage	24.3%	20.5%

4.4.2. Nitrogen Load Estimation for the City of Stuart

Since model calibration is not conducted for the City of Stuart due to the lack of data, the nitrogen load is estimated using the transport parameters calibrated against the well in Martin County but close to the City of Stuart. The simulated flow paths and nitrogen plumes are shown in Figure 4-15, and the estimated loads to the individual water bodies are listed in Table 4-6. The largest load of 1.152 kg/d is from 91 septic systems to water body 109 (the middle estuary); the second largest load of 0.373 kg/d is from 30 septic systems to water body 11. The first and second largest load corresponds to 91.6% of the total load of 1.664 kg/d. This is different from the situation in the City of Port St. Lucie where there are no water bodies receiving majority of load.

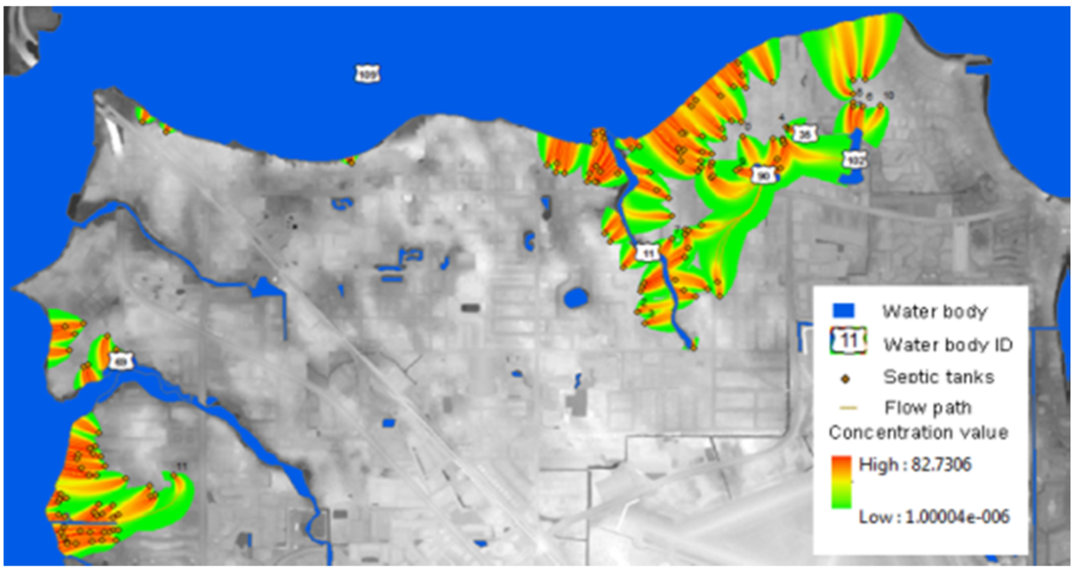

Figure 4-15. Simulated flow paths and plumes from the removed septic systems in the City of Stuart.

Table 4-6. ArcNLET estimated nitrogen load and numbers of contributing septic systems to individual water bodies in the City of Stuart.

Water Body FID	109	11	90	102	69	35	**Sum**
Nitrogen Load (kg/d)	1.152	0.373	0.062	0.034	0.033	0.010	**1.664**
Number of Contributing Septic Systems	91	30	17	5	2	1	**146**

4.4.3. Nitrogen Loading Estimation for Martin County

There are five modeling sites in Martin County: North River Shores, Seagate Harbor, Banner Lake, Rio, and Hobe Sound. The load estimation for the five sites is conducted using the same calibrated parameter values listed in Table 4-1.

Figure 4-16 plots the ArcNLET simulated flow paths and nitrogen plumes from 411 septic systems (one septic tank is not considered because it is located in surface water bodies) in **North River Shores**. The estimated nitrogen load into the surface water bodies is 8.346 kg/d. For the individual surface water bodies, the estimated nitrogen load and number of contributing septic systems are listed in Table 4-7. Water body 45 (North Fork St Lucie River) receives 6.530 kg/d nitrogen load (78.2% of the total load) from 321 septic systems. Since the removed septic systems are close to the water bodies, the simulated plume lengths are short, and the amount of nitrogen loss due to denitrification is small. Impacts of the length of flow paths are discussed in Section 4.5.

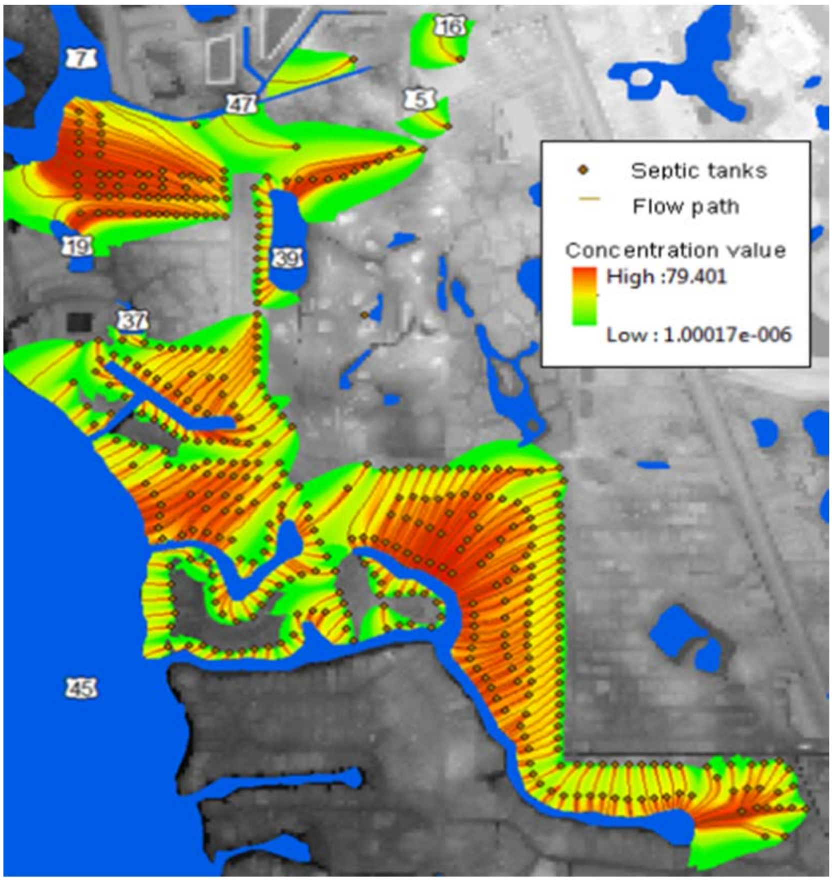

Figure 4-16. Simulated flow paths and nitrogen plumes from the removed septic systems in North River Shores.

Table 4-7. ArcNLET-estimated nitrogen load and number of contributing septic systems to individual surface water bodies in North River Shores.

Water Body FID	45	7	39	19	37	47	5	16	**Sum**
Nitrogen Load (kg/d)	6.530	0.950	0.482	0.222	0.067	0.056	0.021	0.019	**8.346**
Number of Contributing Septic Systems	321	48	23	11	3	3	1	1	**411**

Figure 4-17 plots the ArcNLET simulated flow paths and nitrogen plumes from 453 septic systems (2 septic tanks are not considered because they are located in surface water bodies) at **Seagate Harbor**. The estimated nitrogen load into the surface water bodies is 9.255 kg/d. For the individual surface water bodies, the estimated nitrogen load and number of contributing septic systems are listed in Table 4-8. Among the 451 septic systems, nitrogen from 443 septic systems is loaded to water body 17. The simulated plume lengths at Seagate Harbor are the shortest and the nitrogen removal ratio is also the smallest among the five sites.

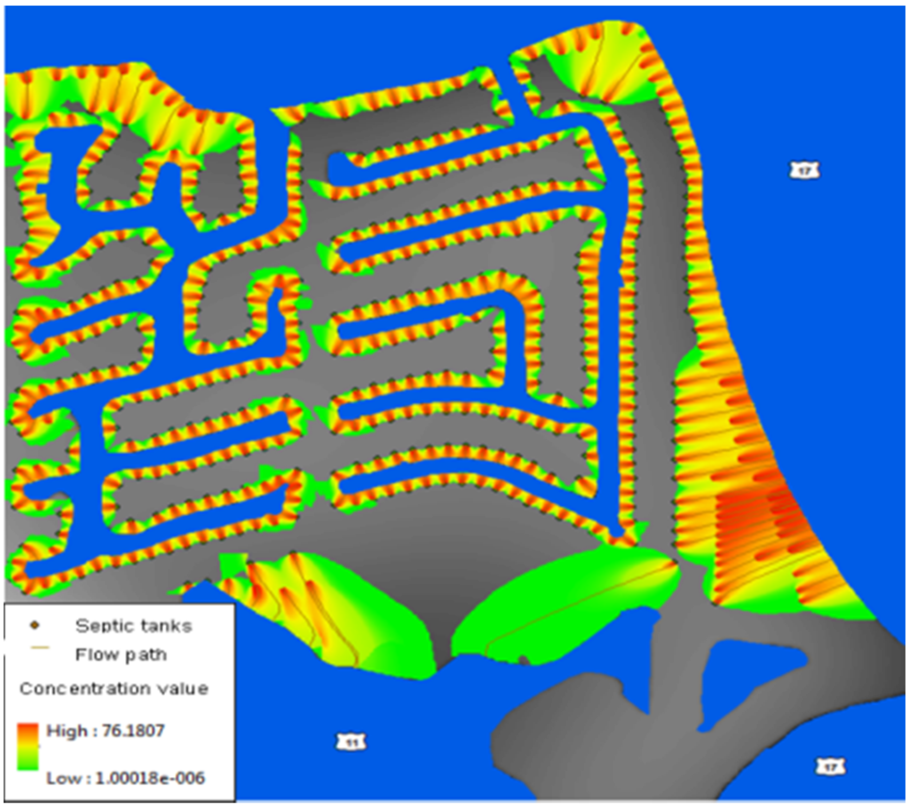

Figure 4-17. Simulated flow paths and nitrogen plumes from the removed septic systems in Seagate Harbor.

Table 4-8. ArcNLET-estimated nitrogen load and number of contributing septic systems to individual surface water bodies in Seagate Harbor.

Water Body FID	17	11	**Sum**
Nitrogen Load (kg/d)	9.183	0.072	**9.255**
Number of Contributing Septic Systems	443	8	**451**

Figure 4-18 plots the ArcNLET simulated flow paths and nitrogen plumes from 105 septic systems in **Banner Lake**. The estimated nitrogen load into the surface water bodies is 0.856 kg/d. For the individual surface water bodies, the estimated nitrogen load and number of contributing septic systems are listed in Table 4-9. The Banner Lake (with the water body with FID of 3) receives 0.429 kg/d nitrogen load (about 50% of the total load) from 33 septic systems. The plume lengths are comparable with those in the City of Port St. Lucie and the City of Stuart, and so is the nitrogen removal ratio due to denitrification, as discussed in Section 4.5.

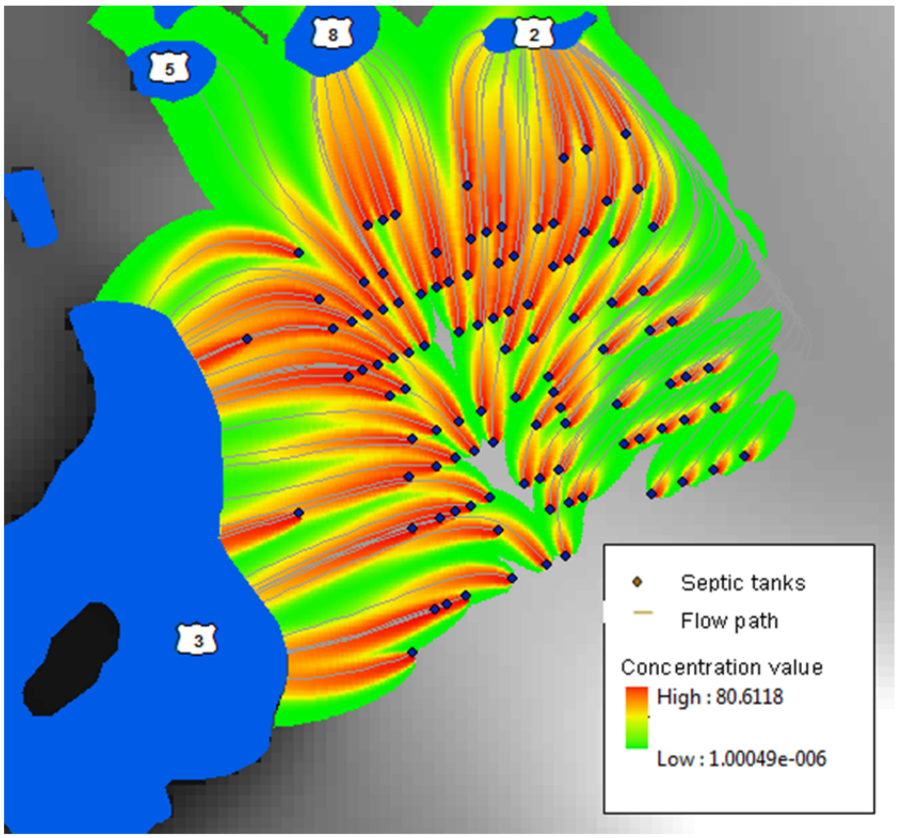

Figure 4-18. Simulated flow paths and nitrogen plumes from the removed septic systems in Banner Lake.

Table 4-9. ArcNLET-estimated nitrogen load and number of contributing septic systems to individual surface water bodies in Banner Lake.

Water Body FID	3	2	8	5	**Sum**
Nitrogen Load (kg/d)	0.429	0.291	0.119	0.017	**0.856**
Number of Contributing Septic Systems	33	59	11	2	**105**

Figure 4-19 plots the ArcNLET-simulated flow paths and nitrogen plumes from 66 septic systems in **Rio**. The estimated nitrogen load into the surface water bodies is 0.317 kg/d. For the individual surface water bodies, the estimated nitrogen load and number of contributing septic systems are listed in Table 4-10. The small water bodies with FIDs of 9 and 13 receive the first and second largest nitrogen load, because they are located down-gradient of the flow paths from 36 septic systems. The plume lengths are comparable with those in Port St. Lucie and Stuart Cities, and so is the nitrogen removal ratio due to denitrification.

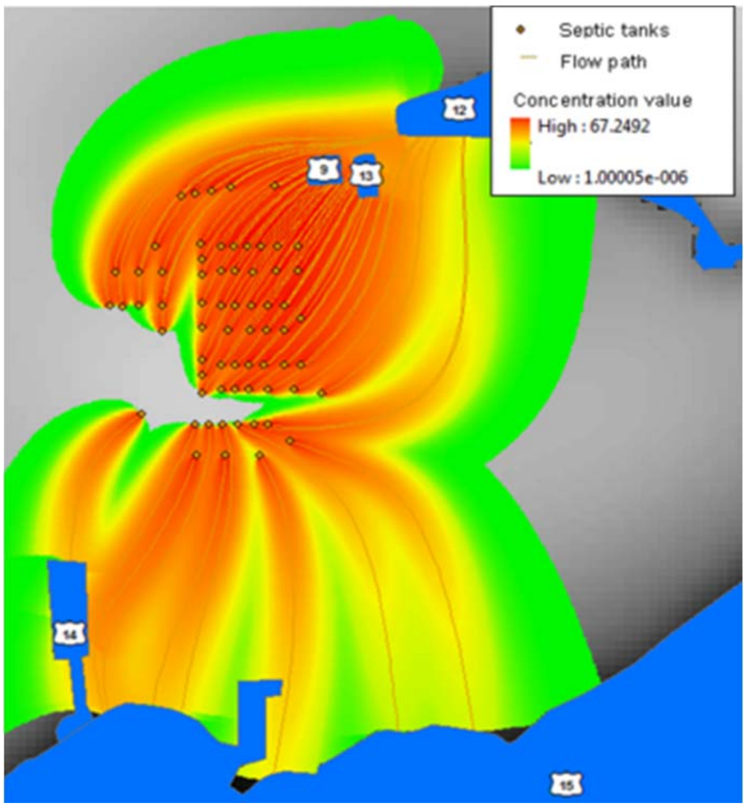

Figure 4-19. Simulated flow paths and nitrogen plumes from the removed septic systems in Rio.

Table 4-10. ArcNLET-estimated nitrogen load and number of contributing septic systems to individual surface water bodies in Rio.

Water Body FID	9	13	15	12	14	**Sum**
Nitrogen Load (kg/d)	0.137	0.101	0.034	0.030	0.015	**0.317**
Number of Contributing Septic Systems	19	17	7	21	2	**66**

Figure 4-20 plots the ArcNLET simulated flow paths and nitrogen plumes from 51 septic systems in **Hobe Sound**. The estimated nitrogen load into the surface water bodies is 0.346 kg/d. For the individual surface water bodies, the estimated nitrogen load and number of contributing septic systems are listed in Table 4-11. The flow directions from the septic systems are uniformly to the surface water bodies. The water bodies with FIDs of 7 – 9 are part of the Indian River Lagoon. Water body 7 receives the largest nitrogen loading from 36 septic systems. The nitrogen load per septic system to water body 9 (14.3 g/d) is larger than that to water body 7 (6.6 g/d), because of the shorter distance between the septic systems to water body 9. The plume lengths are comparable with those in Port St. Lucie and Stuart Cities, and so is the nitrogen removal ratio due to denitrification.

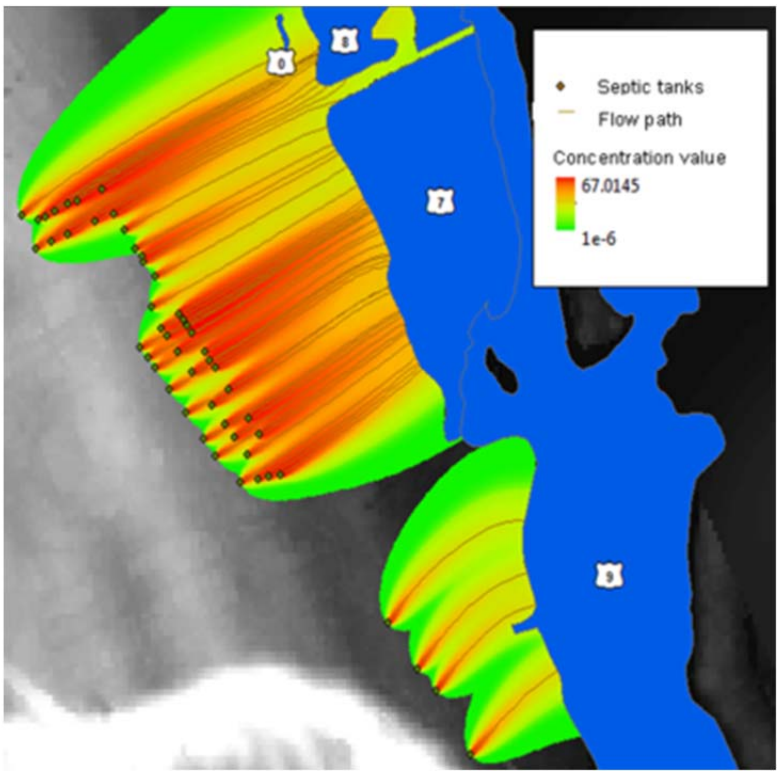

Figure 4-20. Simulated flow paths and nitrogen plumes from the removed septic systems in Hobe Sound.

Table 4-11. ArcNLET-estimated nitrogen load and number of contributing septic systems to individual surface water bodies in Hobe Sound.

Water Body FID	7	9	8	0	**Sum**
Nitrogen Load (kg/d)	0.239	0.057	0.038	0.011	**0.346**
Number of Contributing Septic Systems	36	4	9	2	**51**

4.5. Evaluation of the ArcNLET Estimated Nitrogen Loads

The ideal way to evaluate the load estimates is to compare them with the corresponding field measurements. This however is impossible due to lack of field-scale load measurements. Instead, the load estimates are evaluated by identifying controlling factors of the load estimates, comparing the estimates with literature data of annual load per hectare and nitrogen reduction ratio and with the load estimates of another method used by Martin County. For the convenience of discussion, the total loada, numera of septic tanks, and loada per septic system for the seven modeling sites are listed in Table 4-12.

Table 4-12. ArcNLET estimated total load, number of removed septic systems, and load per septic system.

	Port St. Lucie	Stuart	North River Shores	Seagate Harbor	Banner Lake	Rio	Hobe Sound
Total Load (kg/d)	42.48	1.665	8.346	9.255	0.856	0.317	0.346
Number of Septic Systems	5592	146	411	451	105	66	51
Load per Septic System (g/d)	7.60	11.40	20.31	20.52	8.15	4.80	6.78

4.5.1. Controlling factors of load estimate

It is found in this study that the amount of load estimate is controlled by the following physical factors: length of flow paths, flow velocity, and drainage condition. The length of flow paths is important, because longer flow paths result in more denitrification and thus smaller load estimate. Figure 4-21 plots the load estimates with the mean lengths of flow paths at the seven modeling sites of this study (for each septic system, the length of flow path is from the septic system to receiving surface water body). The two largest loads per septic systems are for North River Shores and Seagate Harbor where the flow paths are the shortest (Figures 4-16 and 4-17). The relation between mean length of flow path and nitrogen load is also reported in Meile et al. (2010). Therefore, in the management of nitrogen pollution, it is important to consider spatial variability of the distance between septic systems and surface water bodies as implemented in ArcNLET.

Figure 4-22 plots variation of the load estimate with the mean velocity at the seven modeling site. It shows that the load estimate increases with the mean velocity. This is reasonable, since larger flow velocity corresponds to shorter travel time and thus smaller amount of denitrification and larger amount of load. Figures 4-21 and 4-22 indicate that the setback distance in nitrogen pollution management should be determined not only by the distance between septic systems to surface water bodies but also by groundwater flow conditions. However, in comparison with the mean length of flow paths, the flow conditions may play a secondary role, since variation of the mean velocity is small, as shown in Figure 4-22.

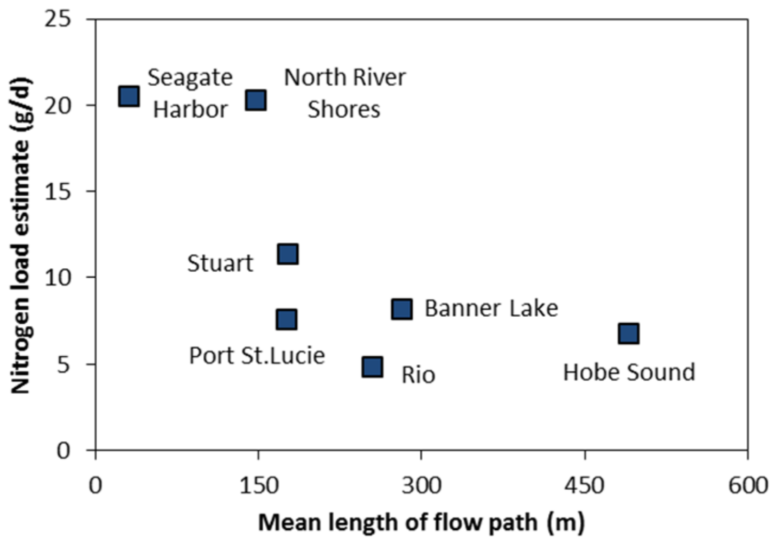

Figure 4-21. Variation of nitrogen load estimate per septic systems with mean lengths of flow paths in the seven sites of this study.

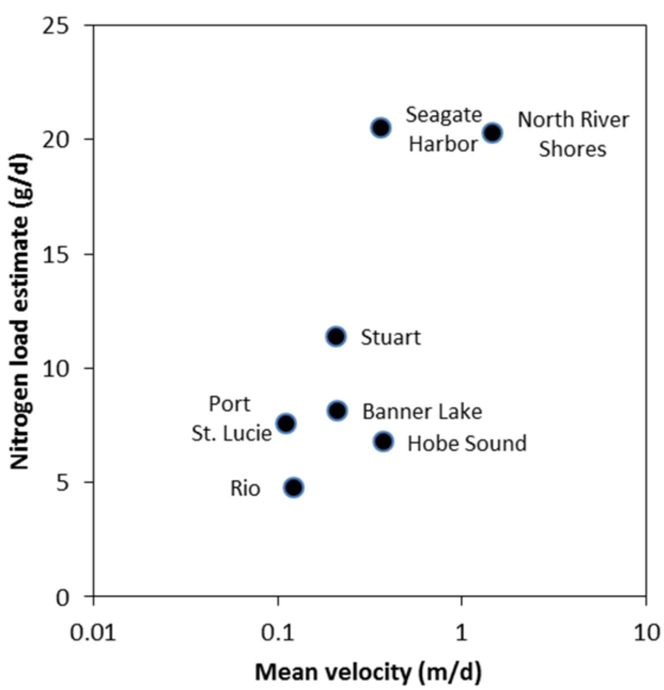

Figure 4-22. Variation of nitrogen load estimate per septic systems with mean velocity in the seven sites of this study.

Figure 4-23 plots the load estimate and soil drainage conditions in the Port St. Lucie site. Figure 4-24 does the same for a total of five sites, excluding Banner Lake and Hobe Sound sites where there is only one drainage category (excessively drainage). The drainage conditions are classified in the SSURGO database into seven categories: excessively drained

(ED), somewhat excessively drained (SED), well drained (WD), moderately well drained (MWD), somewhat poorly drained (SPD), poorly drained (PD), and very poorly drained (VPD). For each drainage category, the number of septic systems is counted and the load from the septic systems is calculated, which is used for plotting Figures 4-23 and 4-24. Figure 4-23 shows that the load estimate increases when the drainage condition changes from very poorly drained to excessively drained. This is not surprising because nitrogen transport is faster in well-drained soil is faster than in poorly drained soil. Figure 4-24(a) shows the groundwater velocity increases when the drainage condition changes from very poorly drained to excessively drained. However, the relation between the load estimates and soil drainage conditions is not observed in the other four sites plotted in Figures 4-24(b)-(e). A possible reason is that, at the four sites, the removed septic systems are located in areas smaller than that in the Port St. Lucie site. In addition, the numbers of removed septic systems are also smaller than those in the Port St. Lucie site. As a result, the relation at the four sites is less statistically meaningful than that at the Port St. Lucie site.

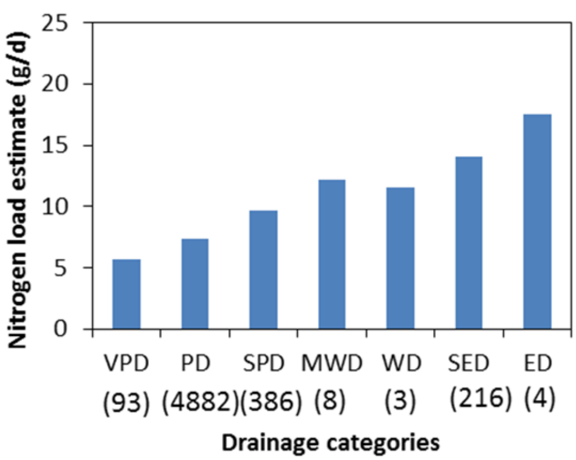

Figure 4-23. Variation of nitrogen load estimate per septic systems with drainage conditions of the soil zones where septic systems are located at the Port St. Lucie site. Abbreviations of the drainage conditions are as follows: excessively drained (ED), somewhat excessively drained (SED), well drained (WD), moderately well drained (MWD), somewhat poorly drained (SPD), poorly drained (PD), and very poorly drained (VPD). The number of septic systems corresponding to each drainage condition is given in the parentheses

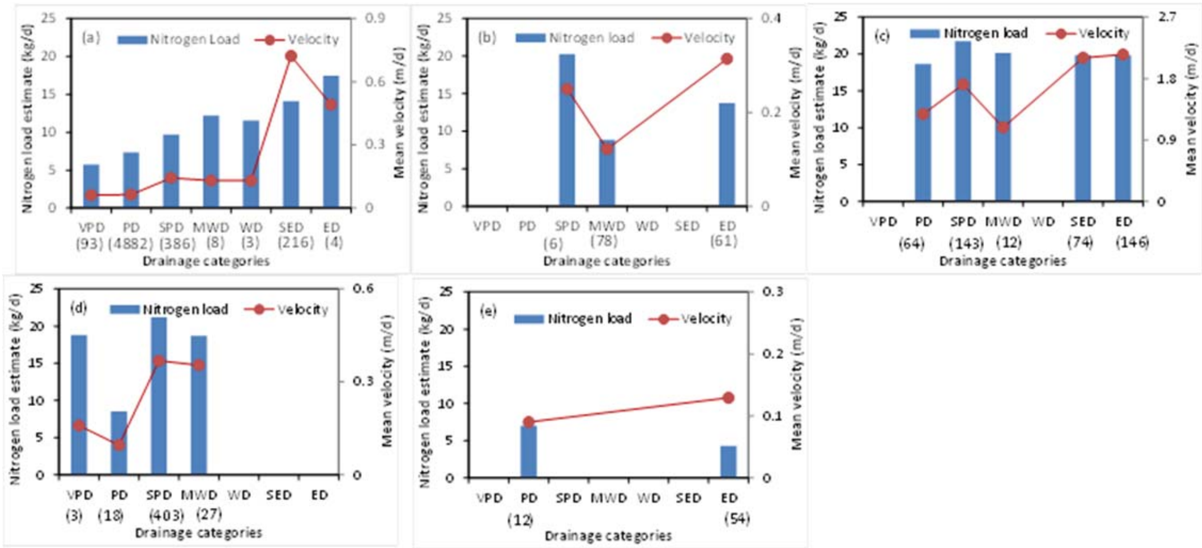

Figure 4-24. Variation of nitrogen load estimate per septic systems with drainage conditions of the soil zones where septic systems are located at (a) Port St. Lucie, (b) Stuart, (c) North River Shores, (d) Seagate Harbor, and (e) Rio.

4.5.2. Comparison with literature data

The ArcNLET load estimates are also evaluated by comparing them with literature data of annual load per hectare and nitrogen reduction ratio. According to the online information available at http://www.city-data.com/housing/houses-Port-St.-Lucie-Florida.html, house density is 1.89 homes/ha in the City of Port St. Lucie. Using this data together with average load of 23 g/d per septic system estimated in this study, the annual load is 15.9 kg/ha. It is larger than the equivalent annual load in Chesapeake Bay watershed, for which Reay (2004) gave an estimate of annual load of 19 kg/ha for house density of 2.5 homes/ha.

Table 4-13 lists the nitrogen reduction ratios per septic system reported in three references; the ratio corresponding to Valiela et al. (1997) is calculated by first estimating the load to surface water bodies using the method of Valiela et al. (1997) (the detailed calculation is given in the note of Table 4-13). The ratios of this study have a large range but are comparable with the literature data, especially with that of Roeder (2008) obtained in the Wekiva Study.

Table 4-13. Daily nitrogen load to groundwater and surface water bodies and nitrogen reduction ratio per septic system from the literature and this study.

Reference	Site Location	Daily nitrogen loads per septic system (g/d)	Daily nitrogen loadings to surface water per septic system (g/d)	Nitrogen reduction ratio
Roeder (2008)	Wekiva Study Area, FL	21.7		70.0% [a]
Valiela et al. (1997)	Waquoit Bay, MA	23	9.87 [b]	57.1%
Meile et al. (2010)	McIntosh County, GA			65-85 % [c]
This study	Port St. Lucie, FL	23	7.60	67.0%
	Stuart, FL	23	11.4	50.4%
	North River Shores, FL	23	20.3	11.7%
	Seagate Harbor, FL	23	20.5	10.8%
	Banner Lake, FL	23	8.15	64.6%
	Rio, FL	23	4.80	79.1%
	Hobe Sound, FL	23	6.78	70.5%

Note: [a] This ratio is for the removal before nitrogen input enters groundwater or a river, i.e., in vadose zone and aquifer.

[b] Nitrogen loading is calculated by using equations: nitrogen released per person per year (4.8kg) × people/house (2.5) × 70% not lost in septic tanks and leaching fields × 66% not lost in plumes × 65% not lost in aquifer

[c] Three different nitrogen reduction ratios are 65%, 69%, and 85% for distance with 15 m, 30 m, and 58 m, respectively.

4.5.3. Comparison with a method used by Martin County

According to Dianne Hughes in Martin County (2013, Personal Communication), the county used another method to estimate nitrogen load from septic systems to surface water bodies. The estimate is based on assumed 250 gallons per day per septic tank and measured influent nitrogen concentration at wastewater treatment plants. The estimated loads for the five sites in Martin County are included in an EXCEL file provided by Dianna Hughes, and they are converted to the unit of g/d and listed in the third column of Table 4-14. Dividing the load estimate by the number of septic systems in the EXCEL file (copied to the second column of Table 4-14) gives the estimated load per septic system of 31 g/d, given in the fourth column of Table 4-14. This number is larger than the input load of 23 g/d into groundwater used in this study, because the measured effluent nitrogen concentration at wastewater treatment plants does not consider nitrogen loss in septic tanks and leaking fields. If the 30% loss (Valiela et al., 1997) is considered, the nitrogen load to groundwater is 22 g/d, close to 23 g/d used in this study. The load to surface water bodies (the sixth column of Table 4-14) estimated in this study is smaller than the load to groundwater due to denitrification. As shown in Table 4-14, the reduction is smaller at North River Shores and Seagate Harbor than at the other three sites, due to shorter flow paths at North River Shores and Seagate Harbor, as discussed above.

Table 4-14. Estimated nitrogen load using the method considered by Martin County and ArcNLET in this study. Data in the second and third columns are provided by Dianne Hughes from Martin County.

Site	Number of septic systems	Method used by Martin County		ArcNLET	
		Total Load (g/d)	Load per septic system (g/d)	Load to groundwater per septic system (g/d)	Load to surface water bodies per septic system (g/d)
North River Shores	435 single and multi-family residential units	13646.3	31	23	20.3
Seagate Harbor	450 single and multi-family residential units	14117.1	31	23	20.5
Banner Lake	116 single family residential units	3638.9	31	23	8.15
Rio	68 single family residential and 3 commercial units	2133.3	31	23	4.80
Hobe Sound	49 single family residential and commercial units	1536.7	31	23	6.78

4.6. Discussion in the BMAP Context

The ArcNLET estimated nitrogen loads are discussed in the BMAP context in two ways. First, the estimated annual loads per hectare are compared with the starting loads from agriculture, natural lands, and all entities (excluding natural lands) listed in BMAP (2013). This comparison suggests significance of septic systems to nitrogen load relative to other nitrogen sources. The comparison for other entities (e.g., MS4) can be conducted in a similar manner but is not performed in this study. In addition, a scenario analysis is conducted to estimate the amount of nitrogen load reduction when functioning septic systems are further removed in the St. Lucie River and Estuary Basin. Percentages of the amount of reduction (due to the actual removed septic systems and to the actual and hypothetical removal) to the total amount of reduction given in BMAP are calculated for the six sub-basins. These exercises may be helpful to facilitate nitrogen management using ArcNLET. However, it should be noted that these exercises are based on extrapolation of ArcNLET-estimated load for the septic removal areas to the entire sub-basin, which may give inaccurate results, as discussed in detail below.

Since the BMAP loads are evaluated for the sub-basins, it is necessary to locating to which sub-basin the removed septic systems belong. Figure 4-25 shows the boundaries of the sub-basins and the locations of the removed septic systems (the sub-basin boundaries are provided by Katie Hallas at FDEP). Figure 4-25 shows that the removed septic systems in the City of Port St. Lucie, North River Shores of Martin County, and Rio in Martin County belong to the North Fork sub-basin, those in the City of Stuart to the South Fork, and those in Seagate Harbor to the Basin 4-5-6. The two sites of Banner Lake and Hobe Sound are located in South Coastal sub-basin, which is not considered in the current BMAP (2013).

Annual nitrogen loads (in the unit of kg/ha) from septic systems in the three sub-basins (North Fork, South Fork, and Basin 4-5-6) are estimated by multiplying the daily load (g/d) per septic system to the house density of 1.89 homes/ha in the City of Port St. Lucie. The daily load per septic system is listed in Table 4-12, and the annual load per hectare is given in the fourth column of Table 4-15. This calculation is based on the following two assumptions: (1) the house density of all the sites is the same as that of the City of Port St. Lucie, and (2) the load estimates for the removed septic systems are representative at the sub-basin scale. While the first assumption may be reasonable, the second assumption is skeptical because the removed septic systems are only a small portion of the septic systems of the sub-basins. The error corresponding to the second assumption is only qualitatively discussed but not quantified in this study.

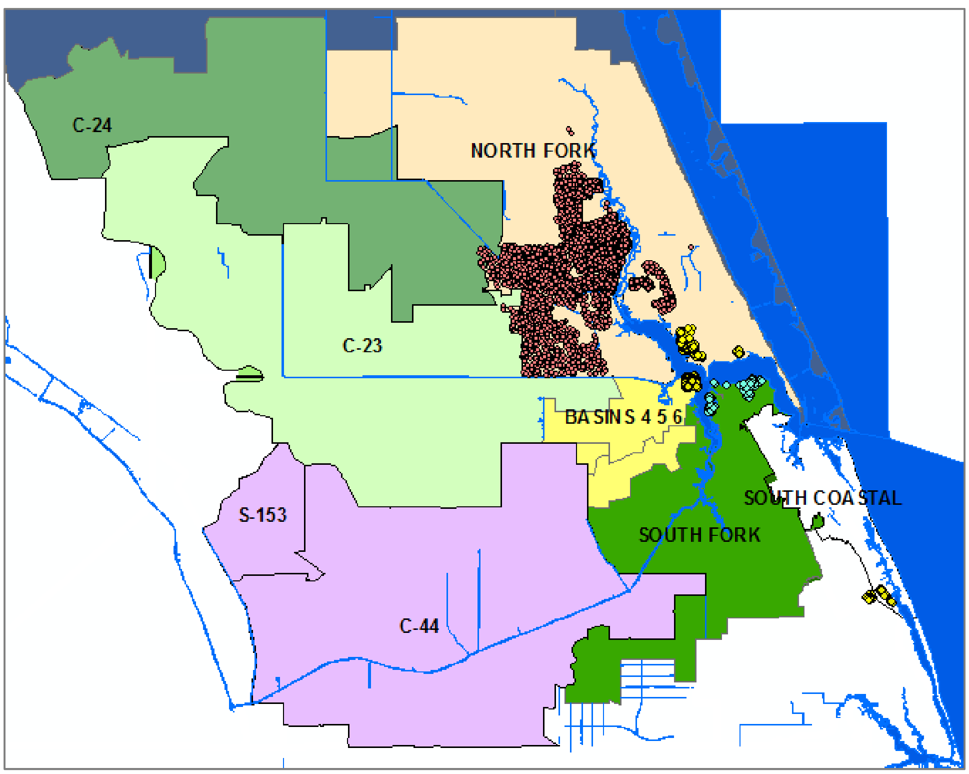

Figure 4-25. Boundaries of sub-basins of the St. Lucie River and Estuary Basin and locations of removed septic systems in the City of Port St. Lucie (red), the City of Stuart (green), and Martin County (yellow).

The annual loads (lbs/acre) from agriculture and natural lands, listed in Table 4-15, are calculated via dividing the loads listed in Table 6 of BMAP (2013) by the corresponding areas listed in Table 8 of BMAP (2013). The results are then converted to the unit of kg/ha and compared with those of ArcNLET. For the City of Port St. Lucie and Rio, the load estimate from septic systems is smaller than that from agriculture but larger than that from natural lands. For the City of Stuart, North River Shores, and Seagate Harbor, the load estimate from septic systems is larger than that from agriculture. This however may be inaccurate, because the removed septic systems are close to water bodies and their loads are larger than those from septic systems far away from the water bodies. It is particularly the case in North River Shores and Seagate Harbor where the flow paths are significantly short, as discussed in Section 4-4.

Table 4-15. Annual nitrogen load (kg/ha) from agriculture and natural lands estimated by ArcNLET and in BMAP (2013).

Sites	Corresponding sub-basins	ArcNLET		BMAP (2013)			
		Daily load per septic system (g/d)	Annual load (kg/ha)	Annual load from agriculture		Annual load from natural lands	
				lbs/acre	kg/ha	lbs/acre	kg/ha
Port St. Lucie	North Fork	7.60	5.24	6.14	6.89	1.31	1.47
Stuart	South Fork	11.40	7.86	6.94	7.79	1.42	1.60
North River Shores	North Fork	20.31	14.01	6.14	6.89	1.31	1.47
Seagate Harbor	Basin 4-5-6	20.52	14.15	6.97	7.83	1.93	2.17
Banner Lake	-	8.15	5.62				
Rio	North Fork	4.80	3.31	6.14	6.89	1.31	1.47
Hobe Sound	-	6.78	4.68				

Table 4-16 is similar to Table 4-15 but for the load from all BMAP entities excluding natural lands; the values in the unit of lbs/acre are copied from Table 11 of BMAP (2013). Spatial variability of the load estimates is observed. For Port St. Lucie, North River Shores, and Rio that are all located in North Fork sub-basin, the load from septic systems in Port St. Lucie is close to that of the sub-basin, the load in North River Shores is larger, and the load in Rio is smaller. Since the modeling site of Port St. Lucie is the largest among the seven modeling sites, the results of Port St. Lucie are more meaningful when they are extrapolated to the sub-basin scale.

Table 4-16. Annual nitrogen load (kg/ha) from all entities of BMAP (excluding natural lands) estimated by ArcNLET and in BMAP (2013).

Sites	Corresponding sub-basins	ArcNLET		BMAP (2013)	
		Daily load per septic system (g/d)	Annual load (kg/ha)	Annual load from all BMAP entities (excluding natural lands)	
				lbs/acre	kg/ha
Port St. Lucie	North Fork	7.60	5.24	4.67	5.25
Stuart	South Fork	11.40	7.86	6.25	7.02
North River Shores	North Fork	20.31	14.01	4.67	5.25
Seagate Harbor	Basin 4-5-6	20.52	14.15	5.83	6.55
Banner Lake	-	8.15	5.62		
Rio	North Fork	4.80	3.31	4.67	5.25
Hobe Sound	-	6.78	4.68		

Significance of the nitrogen load reduction (due to the actual removal of septic systems) to BMAP nitrogen pollution management is evaluated by calculating the percentages of the nitrogen load from the removed septic systems to the BMAP estimated total load given in the draft BMAP (2013) and the percentages of the nitrogen load from the removed septic systems to the BMAP required load reduction. A scenario analysis is conducted to evaluate the hypothetical amount of nitrogen load reduction when all septic systems (including those functioning) are removed. The percentage of the load reduction of removing all septic systems to the BMAP required reduction is also evaluated, which may help the management of nitrogen pollution to meet TMDL requirements. These results are listed in Table 4-17 for the sub-basins, and an example calculation is given in Table 4-18 for North Fork sub-basin.

The most important quantity of the analysis is the loads from removed and functioning septic system. Figure 4-26 plots the locations of functioning septic systems provided by Dale Majewski in the City of Port St. Lucie, David Duncan in the City of Stuart, and Dianne Hughes in Martin County. Based on this figure and Figure 4-25 for the removed septic systems, the numbers of removed and functioning septic system in each sub-basin can be estimated by using the overlay analysis function of ArcGIS. These numbers are listed in the second and third rows of Table 4-17. While the loads from the removed septic systems are evaluated in this study, the loads from the functioning septic systems are extrapolated by multiplying the number of functioning septic systems to the load per septic system obtained in this study. The load per septic system for Port St. Lucie is used for the extrapolation, because the results of Port St. Lucie are more representative than those of the other sites considering its large area and large number of septic systems. However, impacts of the canals on the load estimate in Port St. Lucie may be a concern and should be investigated in future study. The estimated loads from removed and functioning septic systems are listed in rows 4 – 5 of Table 4-17, and an example calculation for North Fork sub-basin is given in Steps 1 – 5 of Table 4-18.

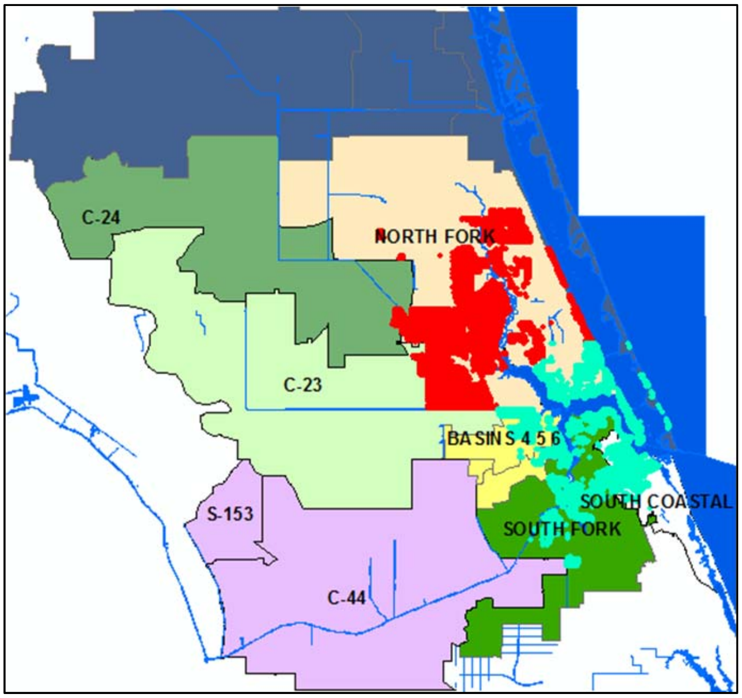

Figure 4-26. Boundaries of sub-basins of the St. Lucie River and Estuary Basin and locations of functioning septic systems. The locations were provided by Dale Majewski in the City of Port St. Lucie, David Duncan in the City of Stuart, and Dianne Hughes in Martin County.

When septic systems are considered in BMAP, the nitrogen loads estimated in BMAP (2013) needs to be updated to include the septic loads. The BMAP estimated loads (in the unit of lbs/yr) listed in Table 6 of BMAP (2013) are copied in row seven of Table 4-17 and converted to the unit of kg/yr in row 8 of Table 4-17. The updated loads with the septic loads are given in row 9 of Table 4-17 following Step 6 in Table 4-18 of the example calculation for North Fork sub-basin.

The percentages of nitrogen loads to the updated total loads for the sub-basins are listed in row 10 of Table 4-17. A large variability is observed. The largest percentage is 31.20% for the North Fork sub-basin, followed by the second largest of 22.87% for the Basin 4-5-6. These numbers appear to be reasonable, considering the absolutely large number of septic systems in North Fork and the relatively large numbers of septic systems in Basin 4-5-6 (Figure 4-26). The percentages are negligible for the C-23 and C24 sub-basins, which is also not unreasonable because of the small number septic systems in the two sub-basins (Figure 4-26). The percentage is 10.33% for South Fork, which seems to be reasonable given the number of septic systems in the sub-basin. Note that a recent study (USEPA, 2013) indicates that septic systems contribute approximately 5% of the total nitrogen load in the Chesapeake Bay watershed. The load percentage is 0% for the C-44/S-153 sub-basin, because it does not contain septic systems (removed or functioning). Although the South Coastal sub-basin is not included in BMAP (2013), the ArcNLET modeling suggests that the load from septic systems is expected to be significant in this sub-basin.

Table 4-17. Results of the scenario analysis that all septic systems are removed and the comparison with those of BMAP (2013).

		Basin 4-5-6	**C-23**	**C-24**	**C-44/ S-153**	**North Fork**	**South Fork**
Load from septic systems	Number of removed septic systems	453	1	173		5,905	146
	Number of functioning septic systems	1,713	5	878		26,524	3,964
	Load from removed septic systems (kg/yr)	3,378	53	2,713		18,216	608
	Load from functioning septic systems (kg/yr)	4,752	14	2,436		73,578	10,996
	Load from all septic systems (kg/yr)	8,130	67	5,149		91,794	11,604
BMAP starting load without septic systems (lbs/yr)		60,314	498,874	670,326	533,437	445,238	221,643
BMAP starting load without septic systems (kg/yr)		27,415	226,761	304,694	242,471	202,381	100,747
Updated load with septic systems (kg/yr)		35,545	226,828	309,843	242,471	294,175	112,351
Percentage of nitrogen load from septic systems		22.87%	0.03%	1.66%	0.00%	31.20%	10.33%
BMAP load from natural lands in BMAP (lbs/yr)		15,128	14,991	24,792	49,942	43,326	26,980
BMAP load from natural lands in BMAP (kg/yr)		6,876	6,814	11,269	22,701	19,694	12,264
BMAP required reduction (%)		35%	52%	53%	47%	39%	45%
Percentage of septic load reduction from actual septic system removal to BMAP required reduction		33.67%	0.05%	1.71%	0.00%	17.02%	1.35%
Percentage of septic load reduction from actual and hypothetical removal to BMAP required reduction		81.02%	0.06%	3.25%	0.00%	85.75%	25.76%

To evaluate the percentages of load reduction from the hypothetical removal of septic systems to the BMAP required load reduction, it is necessary to calculate the amount of BMAP required load reduction. An example calculation of this amount for North Fork sub-basin is given in Steps 8-9 of Table 4-18. Following BMAP (2013), the loads from natural lands are excluded from the calculation, and the percentages of required reduction (listed in Table 11 of BMAP (2013) and copied row 13 of Table 4-17) are used in the calculation. The final results are listed in the last row of Table 4-17. These results suggest that the hypothetical removal of functioning septic systems is absolutely worthy for the North Fork and Basin 4-5-6 sub-basins, because the removal can achieve more than 80% of the required nitrogen load reduction. This may be also true for the South Fork sub-basin because the percentages are about 25%. However, for the C-23 and C-44/S-135 sub-basins, the effort of removing functioning septic systems does not help reduce nitrogen load.

While these observations do not appear to be unreasonable, they are based on the assumption that the load per septic system obtained in Port St. Lucie for the actual removed septic systems is representative for the sub-basins. Ideally, the model calibration conducted in this study for the septic removal areas should be conducted for the sub-basins, which will improve accuracy of the loads of the sub-basins. Although this can be completed in the same calibration procedure described in Section 4.3, it is beyond the scope of this study.

Table 4-18. Example calculation for the results of the North Fork sub-basin listed in Table 4-17.

Step	Items	Calculation	Notes
1	Number of removed septic systems	5,905 = 5,427 (Port St. Lucie) + 66 (Rio) + 412 (North River Shores)	5427 septic systems removed from the City of Port St. Lucie, 66 from Rio, and 412 from North River Shores
2	Number of functioning septic systems	26,524	Obtained by using overlay analysis function of ArcGIS
3	Nitrogen load from removed septic systems (kg/yr)	18,216 kg/yr = 5,427×7.60g/d + 317g/d (Rio) +8,346g/d (North River Shores)) × 365d/1000g	7.60 g/d is the estimated load per septic system in the City of Port St. Lucie.
4	Nitrogen load from functioning septic systems (kg/yr)	73,578 kg/yr = (26,524 × 7.60g/d) × 365d/1000g	Assume that the load per septic system in the City of Port St. Lucie is representative, which may not be correct
5	Nitrogen load from all septic systems (kg/yr)	91,794 kg/yr = 18,216 kg/yr + 73,578 kg/yr	Summation of the loads of Steps 3 and 4. Left and right sides are not exactly equal because of rounding.
6	Nitrogen total load considering septic systems (kg/yr)	294,175 kg = 91,794 kg/yr + 202,381 kg/yr	202,381 kg/yr is equivalent to 445,238 lbs/yr listed in BMAP (2013).
7	Percentage of nitrogen load from septic systems	31.20 % = 91,794/294,175 × 100%	Ratio between the loads of Steps 5 and 6
8	BMAP required nitrogen load reduction (kg/yr)	107,048 kg/yr = (294,175kg/yr – 43,326lbs/yr/2.2lbs/kg) × 0.39	43,326lb/yr is the BMAP load from natural lands, and 0.39 is the BMAP required reduction in North Fork.
9	Percentage of load reduction from septic removal	85.75% = 91,794/107,048 × 100%	Ratio between the loads of Steps 5 and 8. Left and right sides are not exactly equal because of rounding.

5. QUANTIFICATION OF UNCERTAINTY IN ARCNLET LOAD ESTIMATE

Uncertainty quantification for implementing TMDL program has recently received more attention, and the trend is to conduct uncertainty quantification during the TMDL assessment phase and implementation phase (Shirmohammadi et al., 2006). In 2013, a function of Monte Carlo (MC) simulation for uncertainty quantification was developed for ArcNLET (Rios et al., 2013b), which makes ArcNLET a unique software for both deterministic and stochastic modeling. To distinguish the load estimates of this and the last section, the load estimates of the last section are referred to as deterministic estimates, while those of this section as random estimates.

In this section, the software was used to quantify uncertainty in ArcNLET estimated nitrogen load for the following three sites in Martin County:

(1) The site in Martin County where calibration is conducted, which is referred to as the calibration site hereinafter. This site is selected because it has a monitoring well (SOFLSUS2-17) and the observed nitrogen concentration at the well can be used as a yard stick to evaluate reasonableness of the MC simulation results by comparing the observed value with the MC simulate nitrogen concentrations. There are nineteen septic systems in the calibration site.

(2) The Seagate Harbor site with 453 septic systems. The site is selected because the deterministic load estimate per septic system at the site is the highest among all the modeling sites (Table 4-14). The MC simulation of this study may help evaluate whether it is possible to obtain a load estimate larger than the deterministic one.

(3) The Hobe Sound site with 51 septic systems. The site is selected because its deterministic load is relatively low (Table 4-14) and its mean length of flow paths is the longest (Figure 4-21) among all the modeling sites. MC simulation of this study may help evaluate whether it is possible to obtain a load estimate larger than the deterministic one.

For each of the three sites, the probability density functions (PDFs) and cumulative distribution functions (CDFs) of the random load estimate are presented. Based on them, the probabilities of the deterministic load estimates are evaluated and used to evaluate how likely larger or lower estimates can be obtained when more monitoring data of nitrogen concentrations become available. It however should be noted that obtaining more concentration measurement do not necessarily lead to higher load estimates, depending on the values of future concentration measurements, as shown below.

5.1. Brief Introduction of MC Simulation Function of ArcNLET

The ArcNLET estimated nitrogen loads are inherently uncertain due to lack of data (e.g., measurements of model parameters and observations of state variables such as hydraulic head and nitrogen concentration) to constrain the modeling systems and knowledge to adequately describe the bio-hydro-geo-chemical processes controlling nitrogen reactive transport. It is helpful to quantify the uncertainty before the estimates are used for environmental management and planning. This motivated development of MC function of ArcNLET to facilitate the uncertainty quantification. The MC simulation is set up and executed via a graphical user interface (GUI), created as an extension to ArcGIS, and accessed as a tool on the toolbar of the main ArcMap window. In order to facilitate the user interaction, a point and click approach is used. With basic understanding of MC simulations described below, an ArcNLET user can quickly set up the MC simulation and process the MC results. More details of using the MC function of ArcNLET are referred to the user manual (Rios et al., 2013b).

Generally speaking, MC simulation is to propagate uncertainty in model parameters (e.g., hydraulic conductivity) to model outputs (e.g., ArcNLET load estimate). The uncertainty propagation involves two steps. The first is to generate samples of random parameters according to their probability distributions, and the other step is to run the model (ArcNLET) for each set of the parameter samples, which is automated in ArcNLET through the GUI. The MC simulation of ArcNLET addresses uncertainty in the following seven model parameters: (1) smoothing factor, (2) longitudinal dispersivity, (3) horizontal transverse dispersivity, (4) first-order denitrification coefficient, (5) hydraulic conductivity, (6) porosity, and (7) source plane concentration. Before running the MC simulation, the users need to determine which parameters are random. The MC function of ArcNLET is designed to be flexible to consider a single or multiple random parameters. For the random parameters, the users need to characterize the random parameters by specifying their probabilistic distributions. ArcNLET includes four commonly used distributions: uniform, triangular, normal, and lognormal. The Latin Hypercube Sampling (LHS) method is used for generating samples of the random parameters.

In earth and environmental science and engineering, with respect to their spatial variation, random parameters can be categorized into two classes: randomly homogenous and randomly heterogeneous. A randomly homogeneous parameter is homogeneous in space but its value is random. A randomly heterogeneous parameter is more complicated, because its parameter value varies in space and, at each location, its value is random. While all model parameters vary in space in reality, certain parameters have significantly smaller variability than other parameters and can be viewed as spatially homogeneous. In ArcNLET, the randomly homogeneous parameters include (1) smoothing factor, (2) longitudinal dispersivity, (3) horizontal transverse dispersivity, (4) first-order denitrification coefficient, and (5) source plane concentration. Following the common practice of groundwater solute transport modeling, the horizontal transverse dispersivity is assumed to be proportional to the longitudinal dispersivity. In other words, multiplying the longitudinal dispersivity to a multiplier (less than one) gives the horizontal transverse dispersivity. The multiplier is not random but specified by ArcNLET users.

For randomly heterogeneous parameters, generating their random fields is difficult due to lack of data to characterize spatial continuity of the random parameters. To resolve the problem of data scarcity and maintain certain level of accuracy for uncertainty quantification, ArcNLET uses the concept of randomly zonal heterogeneous parameters, a compromise between the randomly homogeneous and heterogeneous parameters. Taking hydraulic conductivity as an example, it is delineated into soil zones as in the SSURGO database, and the delineation is deterministic. Within each soil zone, hydraulic conductivity is randomly homogeneous. In ArcNLET, such parameters are hydraulic conductivity and porosity. For convenience of discussion, these parameters are still referred to as randomly heterogeneous parameters.

While in most of ArcNLET modeling situations, the source plane concentration is treated as a randomly homogeneous parameter, ArcNLET considers another situation in which the source plane concentration varies randomly in space in each realization. More specifically speaking, in each MC realization, the source plane concentrations are different for different septic systems. A user can utilize this feature by selecting the source plane concentration as a randomly heterogeneous parameter. However, this feature is not practical because it is difficult, if not impossible, to collect information of the source plane concentration for individual septic systems. This feature is not used in this study for another reason that the MC results are consistent with the deterministic results, which are obtained using a constant source plane concentration over the entire modeling area.

The outputs of the MC simulation consists of multiple values of nitrogen concentrations at user specified locations (monitoring points) and nitrogen loads at all water bodies involved in the modeling. These values represent ArcNLET predictive uncertainty due to parametric uncertainty. The predictive uncertainty of the variables can be quantified by estimating their distributions and/or statistics (e.g., mean and variance) in post-processing within or outside of ArcNLET. The distributions are more informative than the statistics and can be used to assess risk, i.e., the probability that the load exceeds a specific threshold or performance measure target value. The probability distributions provide more valuable information than a deterministic simulation to decision/policy makers for making science-informed decisions.

5.2. MC Simulations for the Three Sites in Martin County

Table 5-1 lists the distributions of the randomly homogeneous variables: smoothing factor, longitudinal dispersivity, first-order decay coefficient of denitrification, and source plane concentration, which are obtained from literature (Rios et al., 2013b). The defining statistics of the distributions listed in Table 5-1 are identical to the ranges of these parameters listed in Table 4-1. Figure 5-1 plots the histograms of 2,000 samples generated for each of the four random parameters, and illustrates the shape of the distributions for readers who are not familiar with probabilities. The ratio between longitudinal and transverse horizontal dispersivity is set as 10:1. The distribution of hydraulic conductivity is assumed to be triangular, whose two ends are the low and high values and the mode is the representative value contained in the SSURGO database. Since the SSURGO soil databases of St. Lucie and Martin counties do not include porosity data, a literature value of 0.37 for the Indian River Lagoon (Smith et al., 2008) is used. The porosity is assumed to be a deterministic variable and constant over the modeling sites. For the three sites, the only difference is the distributions for hydraulic conductivity, and the distributions are given below.

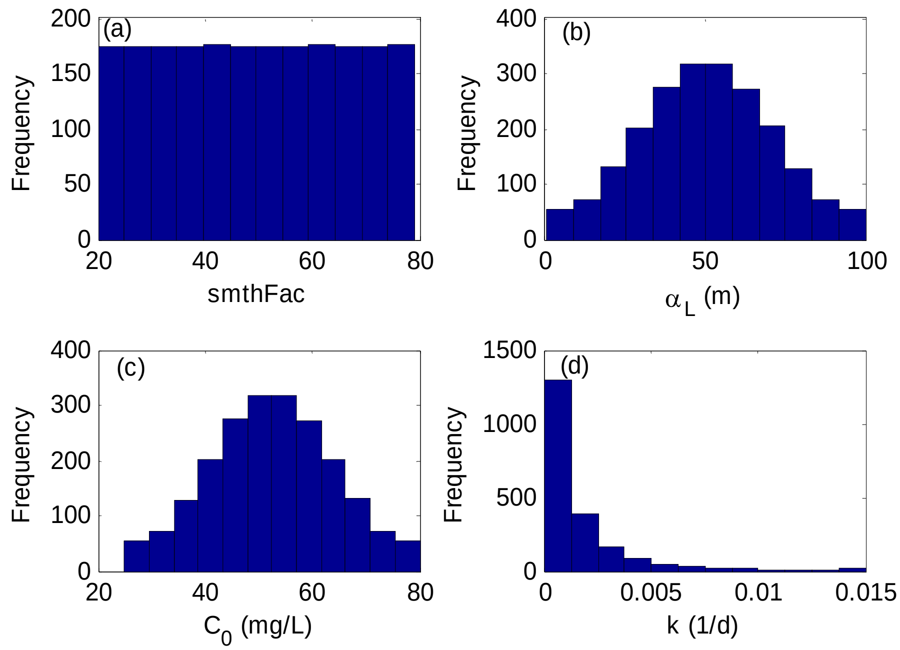

Figure 5-1. Histograms of 2,000 random samples generated for (a) smoothing factor (smthFac) following uniform distribution), (b) longitudinal dispersivity (α_L) following normal distribution, (c) source plane concentration (C_0) following normal distribution), and (d) first-order denitrification coefficient (k) following lognormal distribution.

Table 5-1. Probability distributions and their defining statistics for the spatially homogeneous parameters common to all the three sites where MC simulation is conducted.

Parameter	Distribution	Minimum	Mode	Maximum
Smoothing Factor	Uniform	20	N/A	80
Longitudinal Dispersivity	Normal	1	N/A	100
Source Plane Concentration	Normal	25	N/A	80
Decay Coefficient	Lognormal	5.4E-5	N/A	0.015

5.2.1. MC Simulation for the calibration site

As shown in Figure 5-2, there are three soil zones (with FIDs of 5, 8, and 9) in the calibration sites. It is assumed that hydraulic conductivity of each zone follows triangular distribution and its defining statistics (min, max, and mode) are listed in Table 5-2. Figure 5-3 shows the histograms of 2,000 random samples of the hydraulic conductivities. When generating the samples of the random parameters, it is assumed that all the parameters are statistically uncorrelated, because there is no data to evaluate the parameter correlation. This is confirmed by the negligible linear correlation coefficients calculated from the 2,000 samples (results not shown).

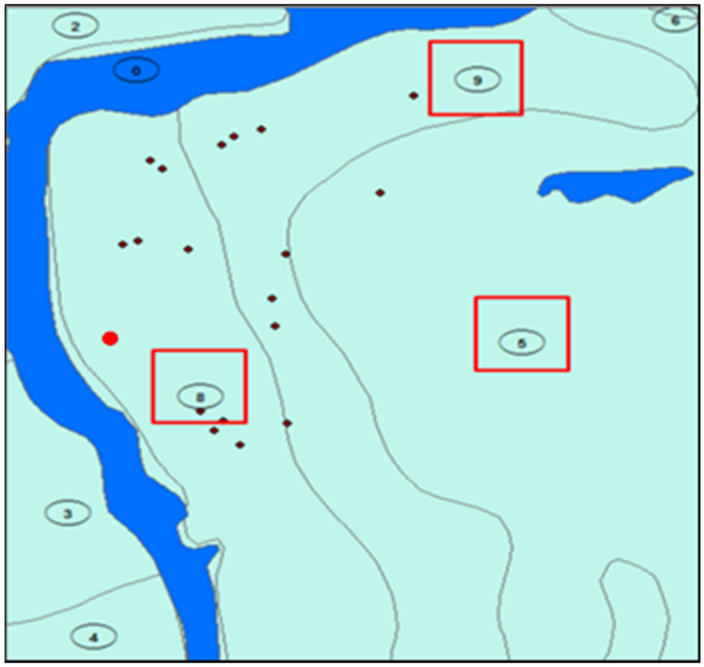

Figure 5-2. Boundaries of three soil zones (with FIDs 5, 8, and 9) in the calibration site. Location of the monitoring well is shown as the red point.

Table 5-2. Defining statistics of the triangular distributions of hydraulic conductivity at the calibration site.

Soil Zone FID	Minimum	Mode	Maximum
5	3.629	7.949	12.18
8	12.18	18.14	24.36
9	12.18	18.14	24.36

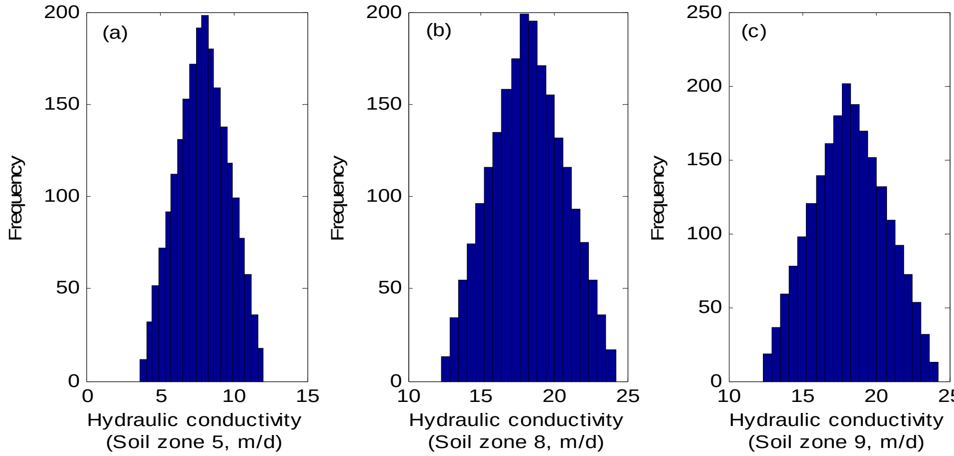

Figure 5-3. Histograms of 2,000 samples generated for hydraulic conductivity of (a) soil zone 5, (b) soil zone 8, and (c) soil zone 9. Triangular distribution is assumed for the hydraulic conductivity.

One monitoring point is set at the monitoring well (SOFLSUS2-17). Convergence of the MC simulation is examined by plotting the running mean and variance of concentrations at the monitoring point. As shown in Figure 5-4, the mean and variance values converge after 1,400 realizations, suggesting that 2,000 samples are sufficient to evaluate the ensemble statistics as well as the PDFs and CDFs of the quantities of interest.

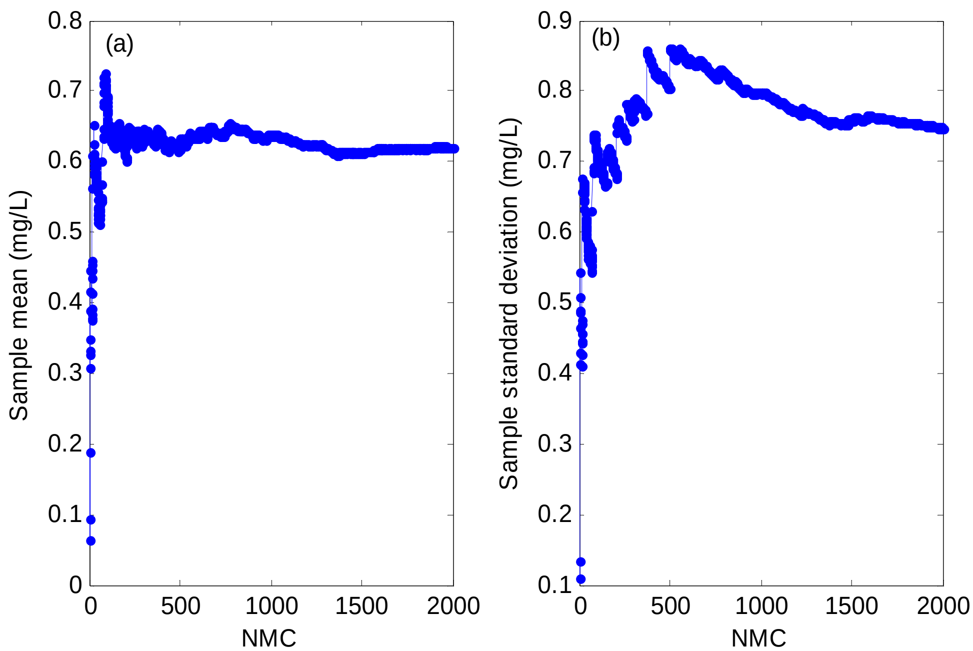

Figure 5-4. Sample (a) mean and (b) standard deviation of simulated nitrogen concentration versus number of MC (NMC) simulations at the monitoring point.

Based on the 2,000 realizations of ArcNLET runs, Figure 5-5 plots the histogram and CDF of the simulated nitrogen concentration at the monitoring well. Figure 5-5(a) of the histogram shows that, roughly speaking, the concentration follows a lognormal distribution, which is attributed to the lognormal distribution of the first-order decay coefficient of denitrification (Figure 5-1(d)), the most influential parameter to nitrogen concentration (Wang et al., 2012). The histogram indicates that, with the parameter distributions listed in Tables 5-1 and 5-2, it is significantly more likely for the model to simulate low concentration values than to high values. This is consistent with the low nitrogen concentration of 0.29 mg/L observed at the monitoring well, suggesting that the calibrated model is likely to reflect nitrogen transport at the calibration site. The simulated value of the calibrated model is 0.30 mg/L, and the corresponding distribution function value is 0.45 (Figure 5-5(b)), indicating that there is 55% probability that the simulate nitrogen concentration can be higher. How likely that the load estimate can be higher is discussed below after examining uncertainty of the load estimate.

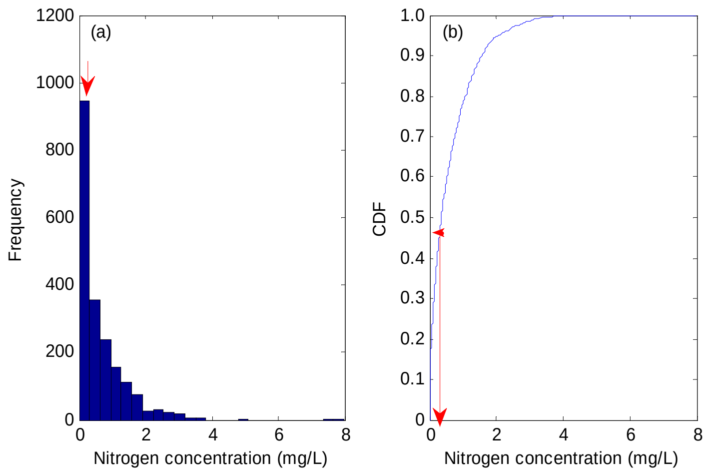

Figure 5-5. (a) Histograms and (b) cumulative distribution function (CDF) of 2,000 realizations of simulated nitrogen concentration at the monitoring point.

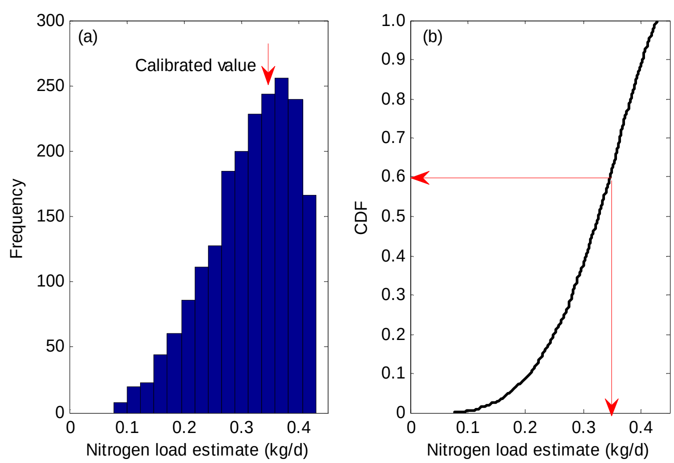

Figure 5-6. (a) Histograms and (b) cumulative distribution function (CDF) of 2,000 realizations of nitrogen load estimates in the calibration area.

Figure 5-6 plots the histogram and CDF of the 2,000 realizations of simulated nitrogen load. Figure 5-6(a) of the histogram shows that the load estimate varies between 0.077 and 0.429 kg/d. The deterministic load estimate of 0.350 kg/d given by the calibrated model has the second highest frequency; the highest frequency is for the simulated load of 0.359-0.383 kg/d. Based on the CDF in Figure 5-6(b), the 95% confidence interval is 0.144 to 0.421 kg/d, calculated as the 2.5th to 97.5th percentile of load estimate. The 95% confidence interval includes the deterministic load estimate of 0.350 kg/d. As shown in Figure 5-6(b), the cumulative distribution function value of the deterministic load estimate is 0.6, indicating that there is 40% probability that larger load estimate can be simulated.

To answer the question that how the load estimate changes when more observations of nitrogen concentration are available at the monitoring point, Figure 5-7 plots the relation between the load estimate and the simulated concentration at the monitoring well. The overall positive correlation indicates that larger nitrogen concentration corresponds to larger load. In the context of site monitoring, if higher concentrations are continuously observed at the monitoring well, the load estimate should be larger than the deterministic estimate of 0.35 kg/d. However, larger load estimate may be still possible for low concentration, because uncertainty in the load estimate increases when the simulated concentration decreases. The uncertainty can be reduced by collecting more field observations (e.g., continuous monitoring at the well), as more monitoring data can remove the realizations that cannot simulate the monitoring data. In other words, more data can better characterize the model parameters, especially the first-order decay coefficient as shown below.

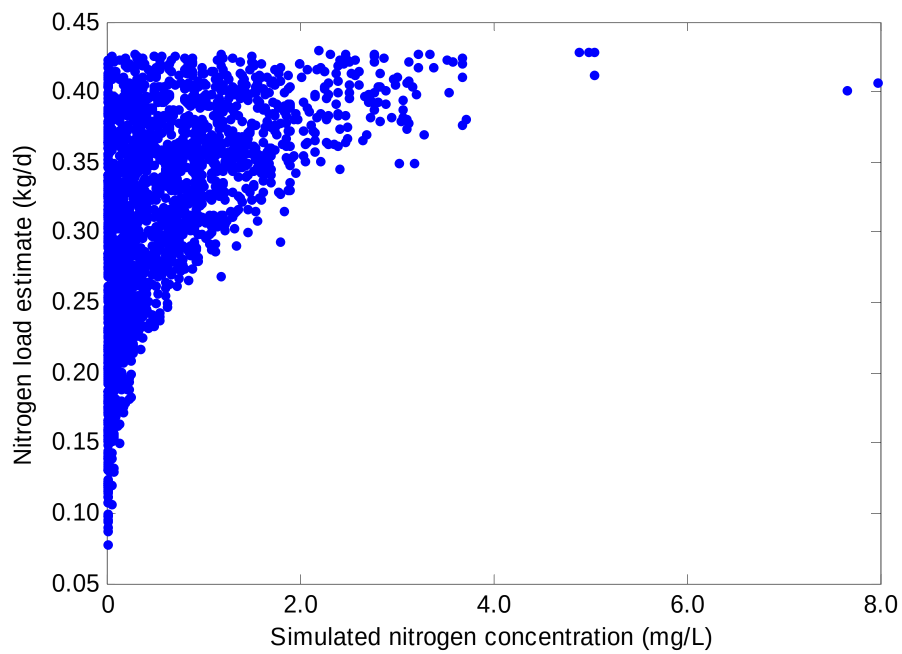

Figure 5-7. Relation between nitrogen load estimate and concentration simulation at the monitoring well point over the 2,000 MC realizations.

The relation between the random parameters and the simulated nitrogen concentration and load estimate is examined using scatter plots. Figure 5-8 is the scatter plots for the first-order decay coefficient of denitrification, the most influential parameter. The figure shows that the load estimate and simulated concentration decrease exponentially with the decay coefficient. While the uncertainty in the load estimate increases with the decay coefficient, the uncertainty in the simulated concentration decreases. This is physically reasonable. Considering that the load from an individual septic system to groundwater is a constant, when the decay coefficient is small, the amount of denitrification is small, which leads to small uncertainty in the load estimate. On the other hand, when the decay coefficient is small, the simulated concentration at the monitoring well can vary dramatically, depending on the combination of the values of the other parameters. For example, the concentration is higher, if the monitoring well is located at the center of the plume along the flow path than at the edge of the plume; the location of flow path is determined by the smoothing factor. When the decay coefficient is high, the simulate concentration is low due to denitrification, regardless of the other parameters. The magnitude of the decay coefficient can be better determined when more data are available.

There is no apparent relation between the other parameters and the load estimate and simulated nitrogen concentration, as shown in Figures 5-9 – 5-13. It suggests that the nitrogen load and concentration are not determined solely by one of the parameters. The joint effects of the parameters on nitrogen load and concentration can be quantified using a global sensitivity analysis method (e.g., the Sobol' method), which however is beyond the scope of this study. Uncertainty caused by these parameters can also be reduced when more observations of nitrogen concentrations are available. The scatter plots of the other two sites are not shown because they are similar to those of the calibration site.

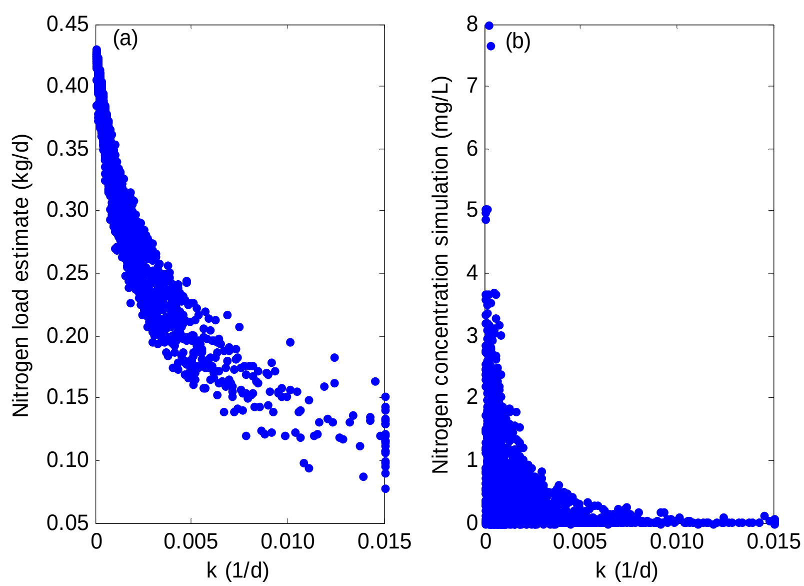

Figure 5-8. Scatterplots for the first-order decay coefficient of denitrification (*k*) and (a) nitrogen load estimate and (b) nitrogen concentration simulation.

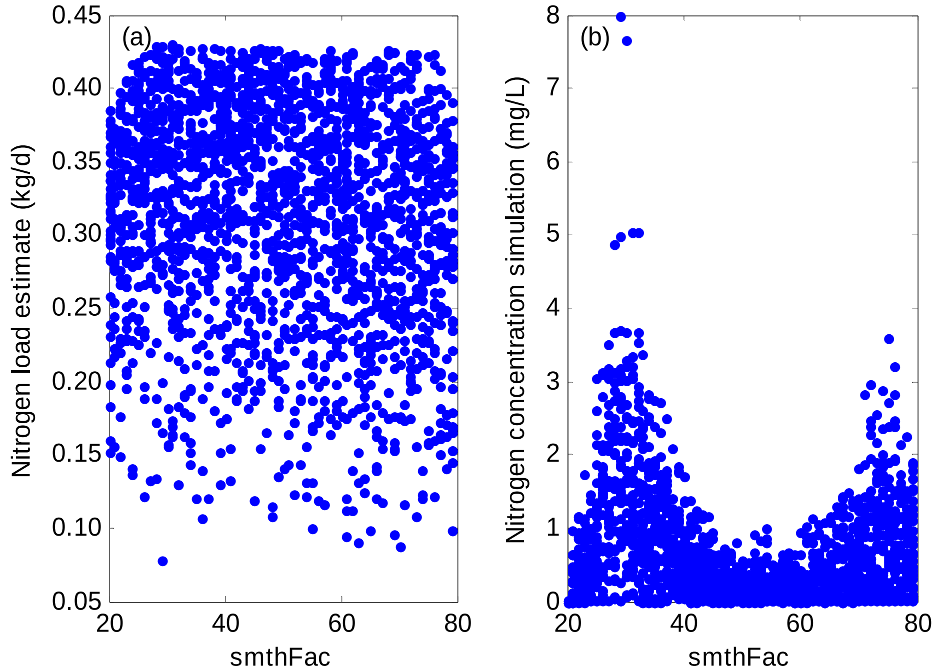

Figure 5-9. Scatterplots for smoothing factor (smthFac) and (a) nitrogen load estimate and (b) nitrogen concentration simulation.

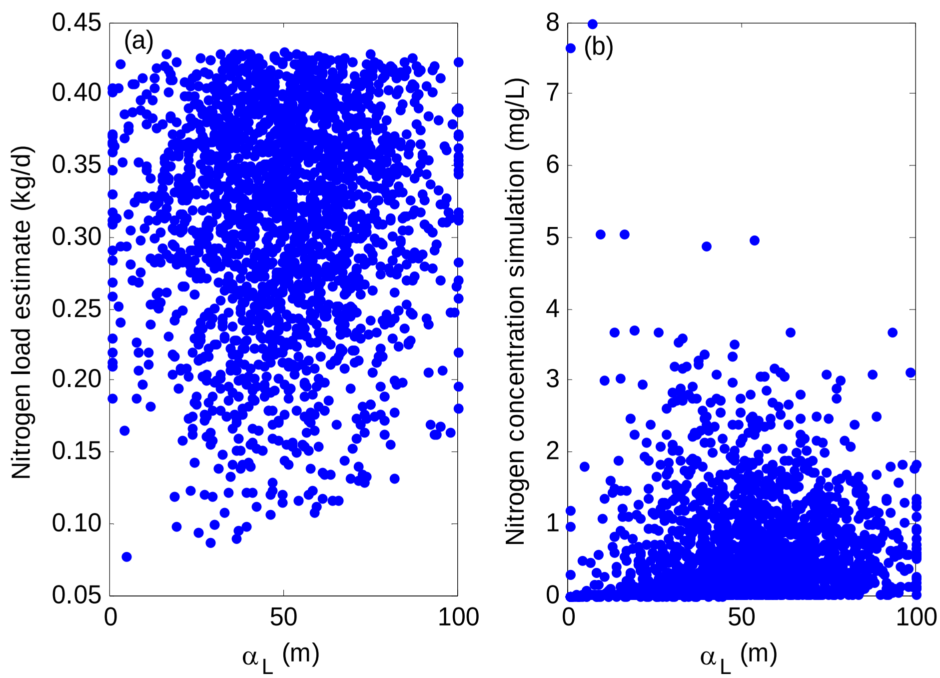

Figure 5-10. Scatterplots for longitudinal dispersivity ($\boldsymbol{\alpha_L}$) and (a) nitrogen load estimate and (b) nitrogen concentration simulation.

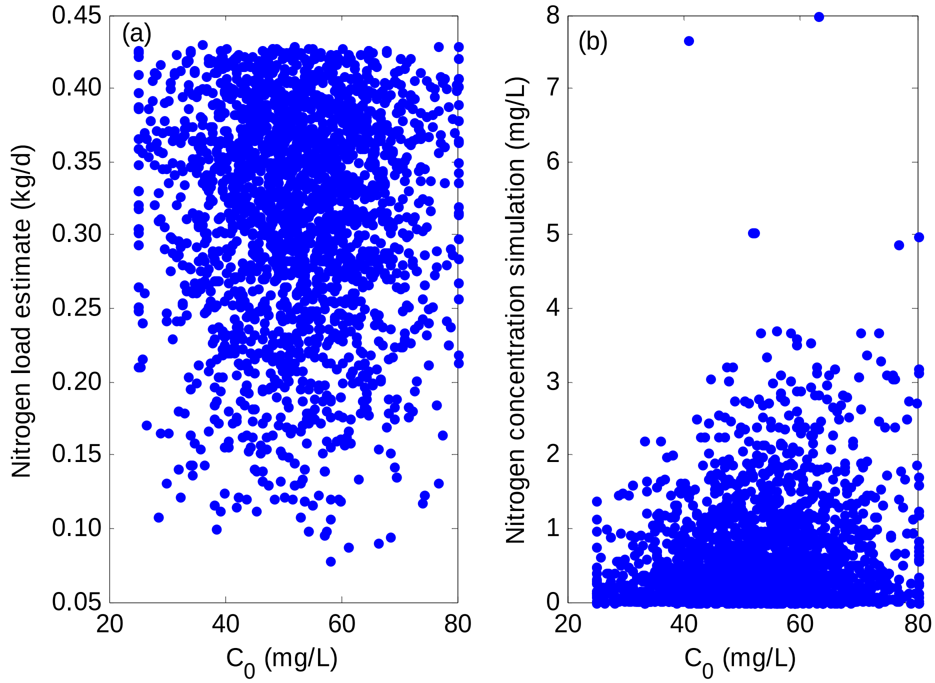

Figure 5-11. Scatterplots for source plane concentration (C_0) and (a) nitrogen load estimate and (b) nitrogen concentration simulation.

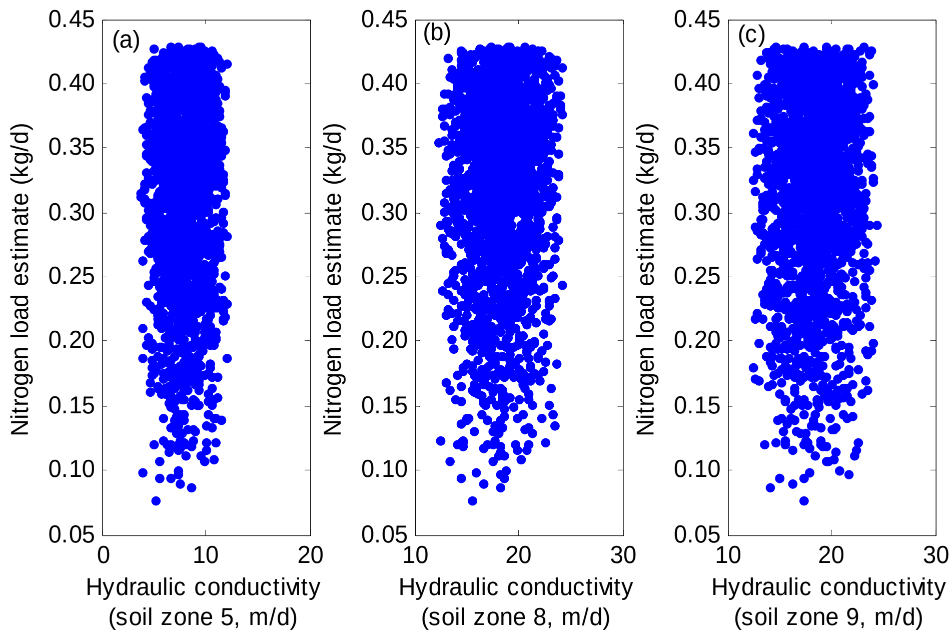

Figure 5-12. Scatterplots for nitrogen load estimate and hydraulic conductivity of (a) soil zone 5, (b) soil zone 8, and (c) soil zone 9.

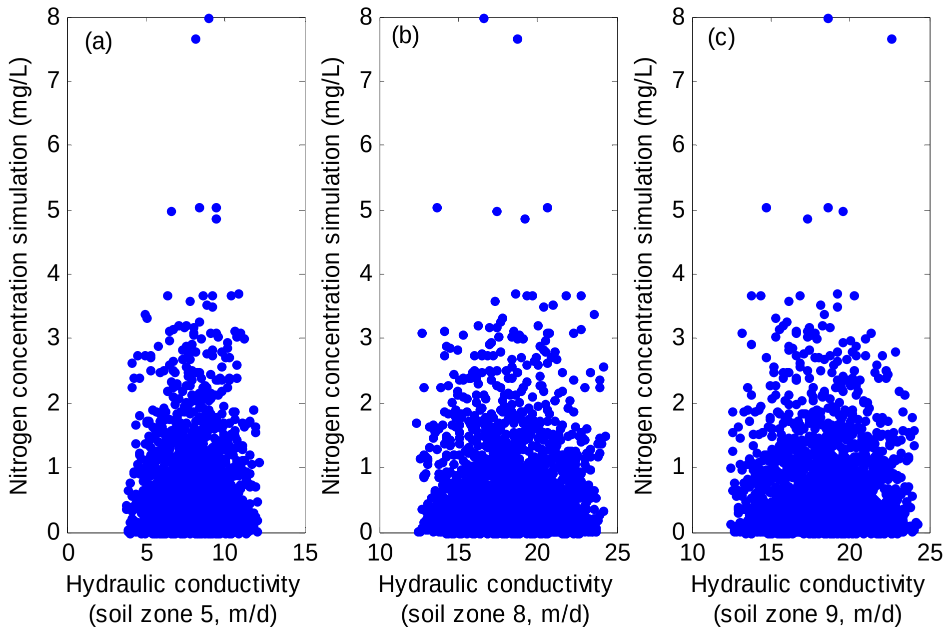

Figure 5-13. Scatterplots for nitrogen concentration simulation and hydraulic conductivity of (a) soil zone 5, (b) soil zone 8, and (c) soil zone 9.

5.2.2. MC Simulation for the Seagate Harbor site

Figure 5-14 shows the spatial distribution of soil zones in the Seagate Harbor site. Most of the septic systems are located in the soil zone 0. The defining statistics (minimum, maximum, and mode) of the triangle distributions of hydraulic conductivity of these soil zones are listed in Table 5-3. The distributions and their defining statistics for smoothing factor, longitudinal dispersivity, source plane concentration, and decay coefficient are given in Table 5-1. A total of 2,200 realizations of the random parameters are generated. Their histograms are not shown, since they are similar to those shown in Figures 5-1 and 5-3. The random samples are uncorrelated, as confirmed by negligible linear correlation coefficients (results not shown).

As shown in Figure 5-15, a total of sixteen monitoring points are placed in the area with high concentrations simulated by the calibrated model. The monitoring points are 10 m (one cell size) away from water body 17, which, as shown in Table 4-8, receives significantly more nitrogen load than water body 11, the other water body in the modeling area. The convergence diagnosis shown in Figures 5-16 and 5-17 indicates that the reliable sample statistics are obtained by running the 2,200 model simulations.

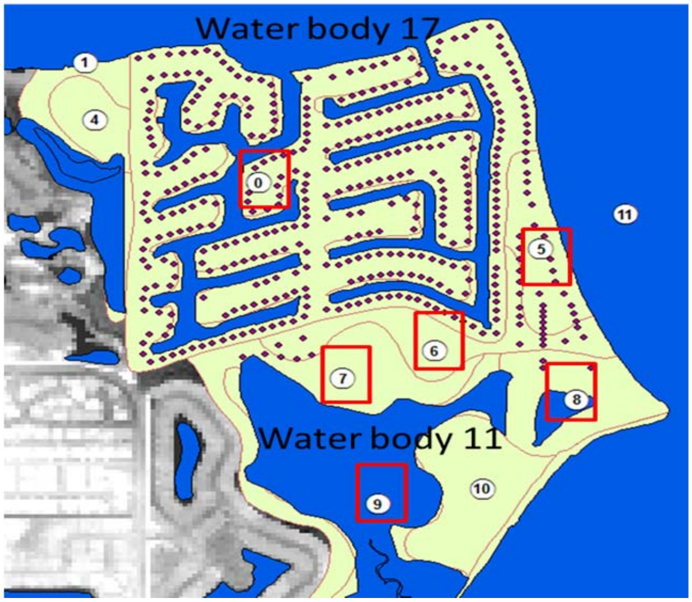

Figure 5-14. Boundaries of three soil zones in the Seagate Harbor site.

Table 5-3. Defining statistics (minimum, mode, and maximum) of the of triangle distributions of hydraulic conductivity of the soil zones at the Seagate Harbor site.

Soil Zone FID	Minimum	Mode	Maximum
0	3.629	7.949	12.18
5	12.18	18.14	24.36
6	3.629	7.949	12.18
7	3.629	7.949	12.18
8	3.629	7.949	12.18
9	3.629	7.949	12.18

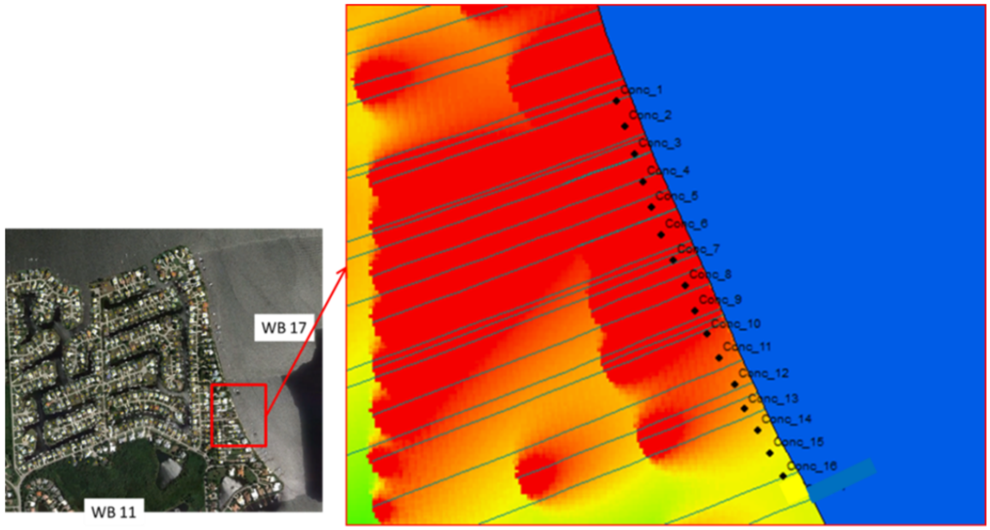

Figure 5-15. Locations of monitoring points placed at Seagate Harbor. The flow paths and plumes are generated by the calibrated model.

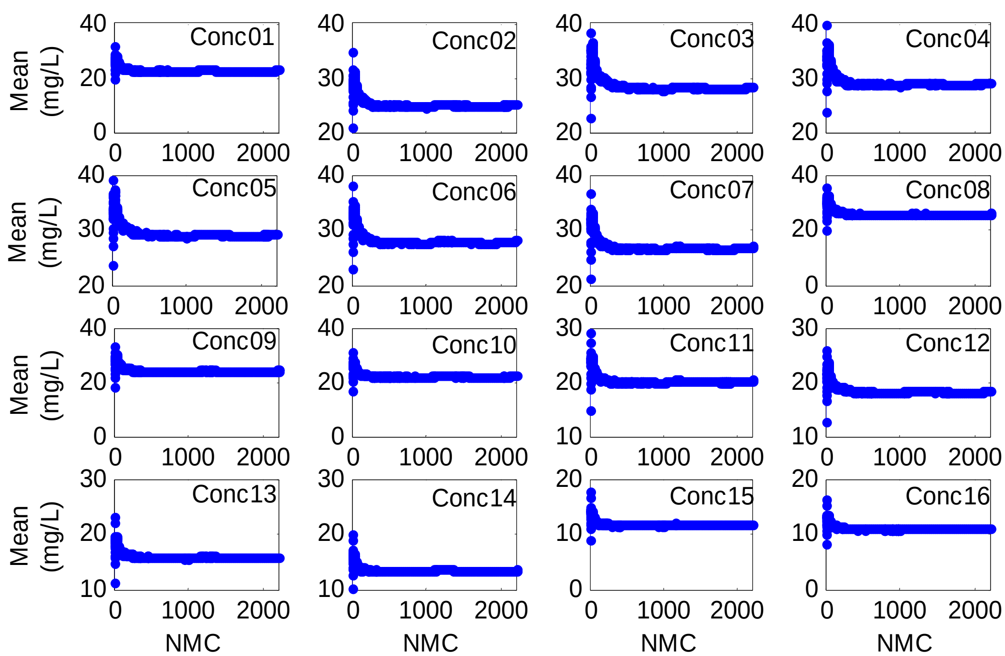

Figure 5-16. Sample mean of simulated nitrogen concentration versus number of MC (NMC) simulations at the monitoring point.

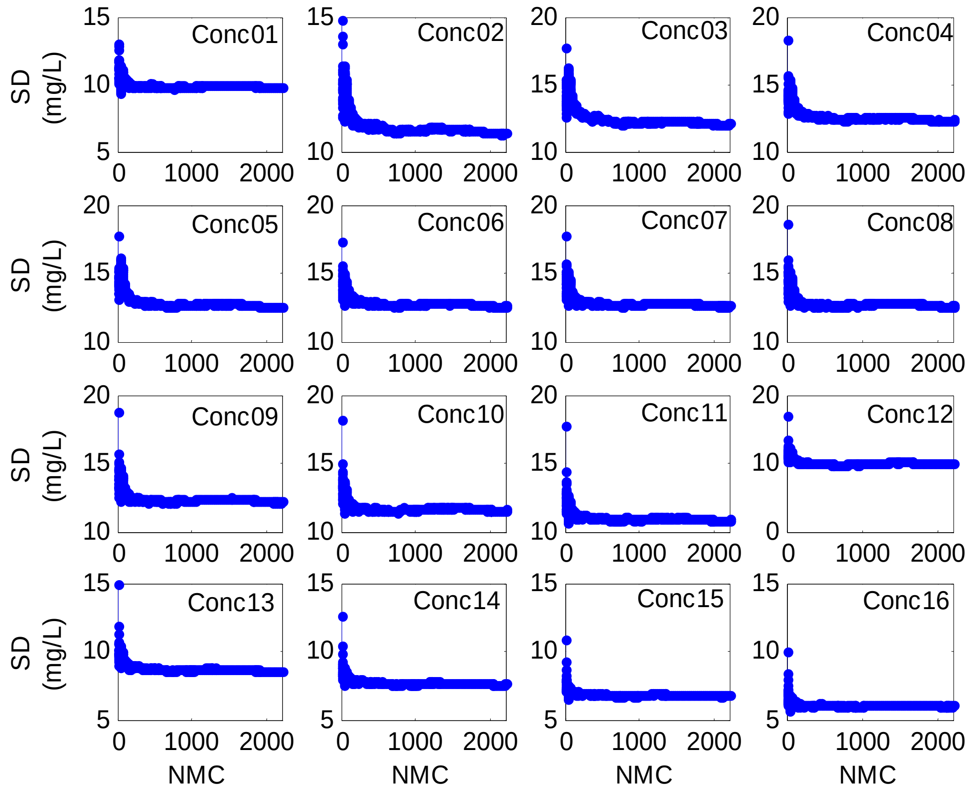

Figure 5-17. Sample standard deviation (SD) of simulated nitrogen concentration versus number of MC (NMC) simulations at the monitoring point.

Figure 5-18 shows the histograms of the 2,200 realizations of nitrogen load estimate to water bodies 17 and 11 and the both. While the deterministic estimates (9.183 kg/d to water body 17, 0.072 kg/d to water body 11, and 9.255 kg/d to the both) given by the calibrated ArcNLET are not the largest ones among the 2,200 realizations, they are relatively large with high frequency. The CDF plots in Figure 5-19 show that the deterministic load estimates to water bodies 17 and 11 and the both correspond to the cumulative distribution function values of 0.40, 0.55, and 0.45, respectively. In other words, there are 60%, 45%, and 55% probability, respectively, that the load estimates can be larger than those given by the calibrated model. However, due to the sharp CDFs shown in Figures 5-19(a) and (c), the magnitude of load increase is not large. For example, the maximum loads to water body 17 and the both water bodies are 10.154 and 10.311 kg/d, less than 1.0 kg/d more than the corresponding deterministic load estimates. In other words, increase of load estimates is not significant at the Seagate Harbor site, which is not surprising because the load estimate at the site is already high.

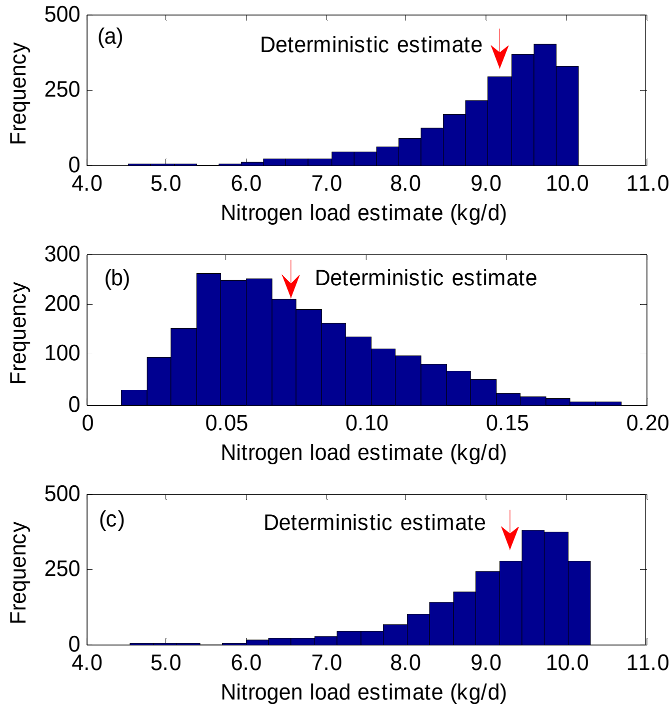

Figure 5-18. Histograms of 2,200 realizations of nitrogen load estimates for (a) water body 17, (b) water body 11, and (c) the two water bodies in the Seagate Harbor site.

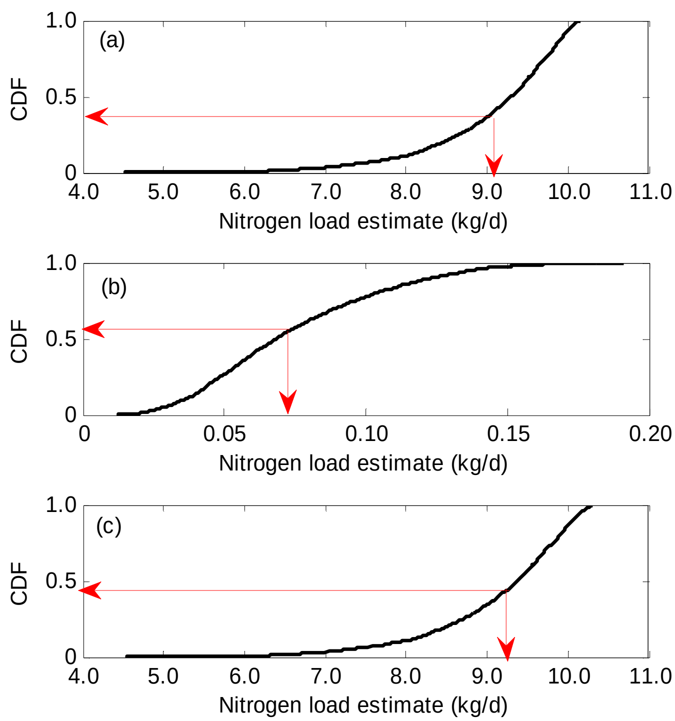

Figure 5-19. Cumulative distribution functions (CDFs) of 2,200 realizations of nitrogen load estimates for (a) water body 17, (b) water body 11, and (c) the two water bodies in the Seagate Harbor site.

Figure 5-20 shows the relation between the nitrogen load (to water bodies 17 and 11) and simulated nitrogen concentration at four monitoring points whose locations are shown in Figure 5-15. The positive correlation shown in Figure 5-20 indicates that, if nitrogen concentrations at the monitoring points are higher than those simulated by the calibrated model (marked by the arrows in Figure 5-20), the load estimate will be larger than that given by the calibrated model. The positive correlation is relatively weaker at monitoring point 16 than at the other three monitoring points, which maybe because the monitoring point that is located at the edge of the modeling domain. Nevertheless, the positive correlation revealed in Figure 5-20 is stronger than that in Figure 5-7 for the calibration site, suggesting that having more observations at Seagate Harbor can reduce more uncertainty than at the calibration site.

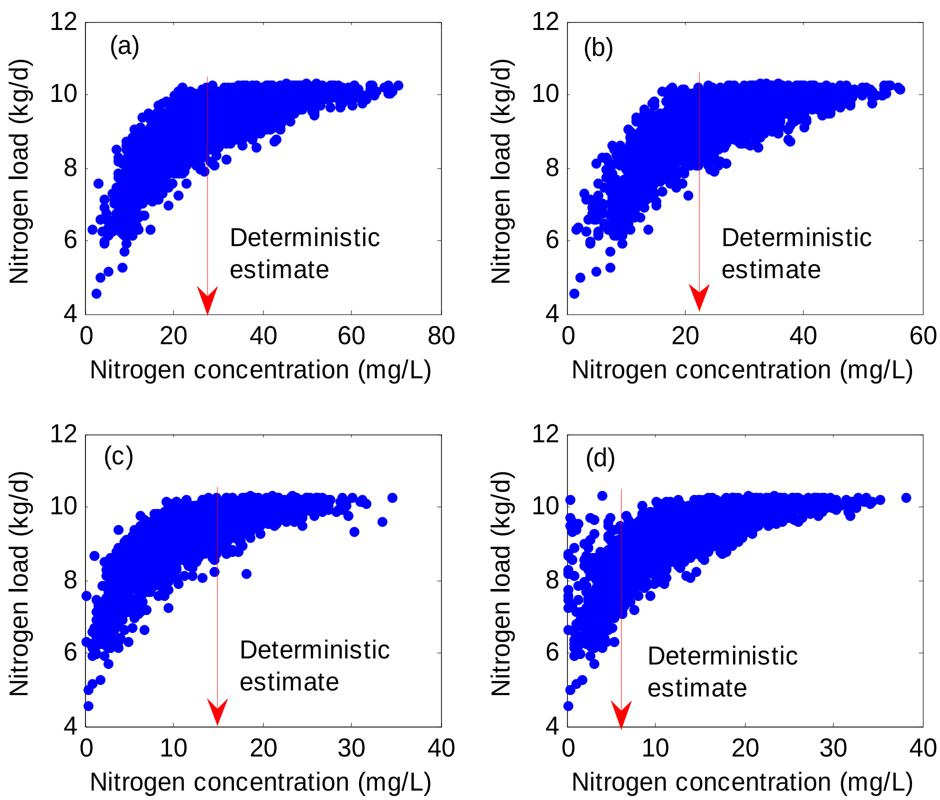

Figure 5-20. Relation between nitrogen load estimate and concentration simulation at monitoring point (a) Conc04, (b) Conc08, (c) Conc12, and (d) Conc16 over the 2,000 MC realizations.

5.2.3. MC Simulation for the Hobe Sound site

Figure 5-21 shows the spatial distribution of the three soil zones in the Hobe Sound site. All the septic systems are located in soil zone 22. The defining statistics (minimum, maximum, and mode) of the triangle distributions of hydraulic conductivity of these soil zones are listed in Table 5-4. The distributions and their defining statistics for parameters smoothing factor, longitudinal dispersivity, source plane concentration, and decay coefficient are the same as those listed in Table 5-1. A total of 2,000 realizations of the random parameters are generated.

Their histograms are not shown, since they are similar to those shown in Figures 5-1 and 5-3. The random samples are uncorrelated, as confirmed by negligible linear correlation coefficients (results not shown) similar to those listed in Table 5-2. As shown in Figure 5-21, a total of twenty-two monitoring points are placed in the area along the shore. The convergence diagnosis shown in Figures 5-22 and 5-23 indicates that the reliable sample statistics are obtained by running the 2,000 model simulations.

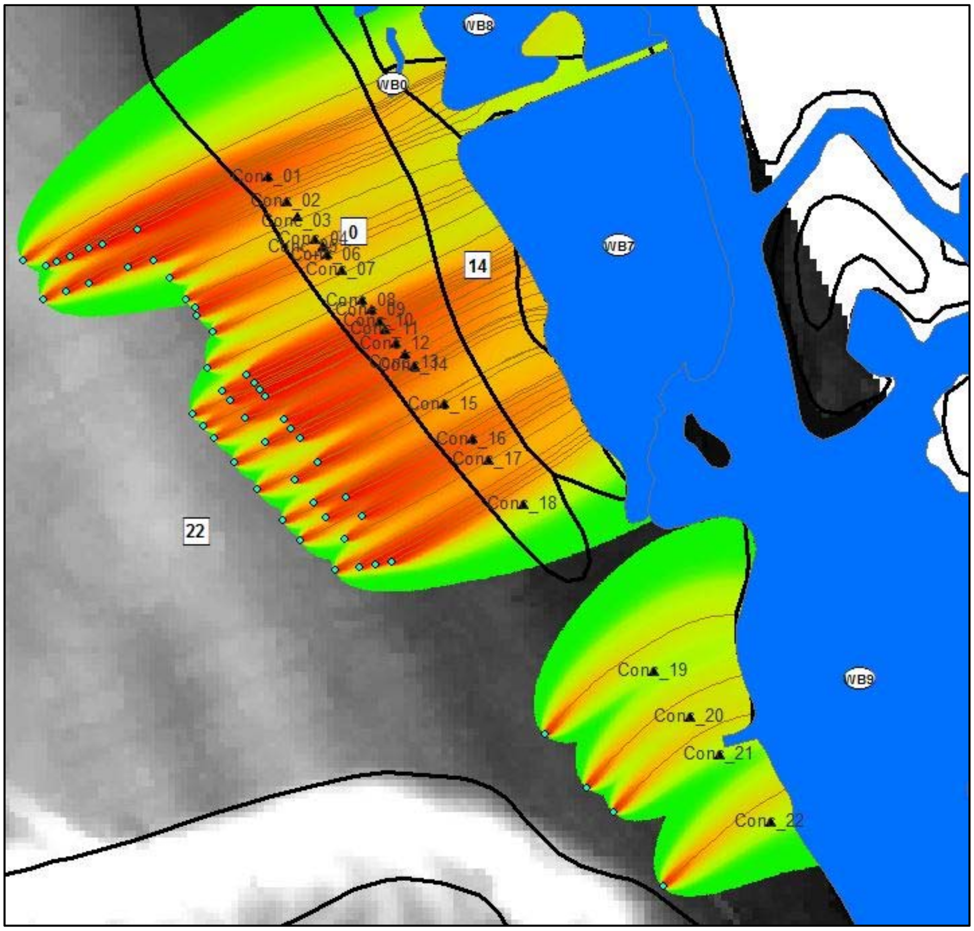

Figure 5-21. Boundaries of soil zones and locations of monitoring points at the Hobe Sounds site.

Table 5-4. Defining statistics (minimum, mode, and maximum) of the of triangle distributions of hydraulic conductivity of the soil zones at the Hobe Sound site.

Soil Zone FID	Minimum	Mode	Maximum
0	12.18	18.14	24.36
14	3.629	7.949	12.18
22	12.18	18.14	24.36

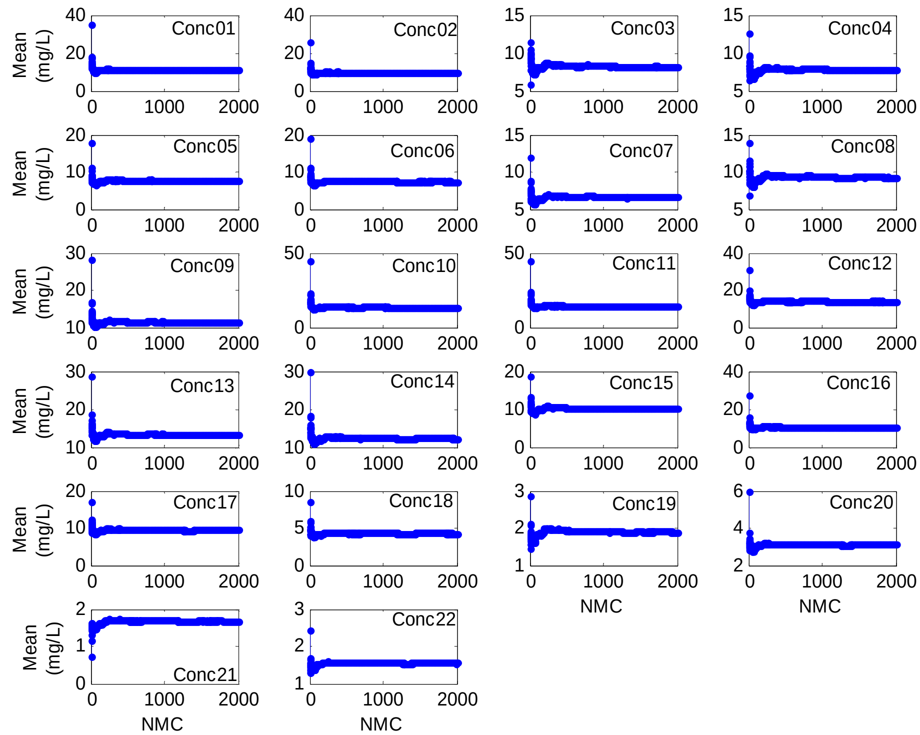

Figure 5-22. Sample mean of simulated nitrogen concentration versus number of MC (NMC) simulations at the monitoring points.

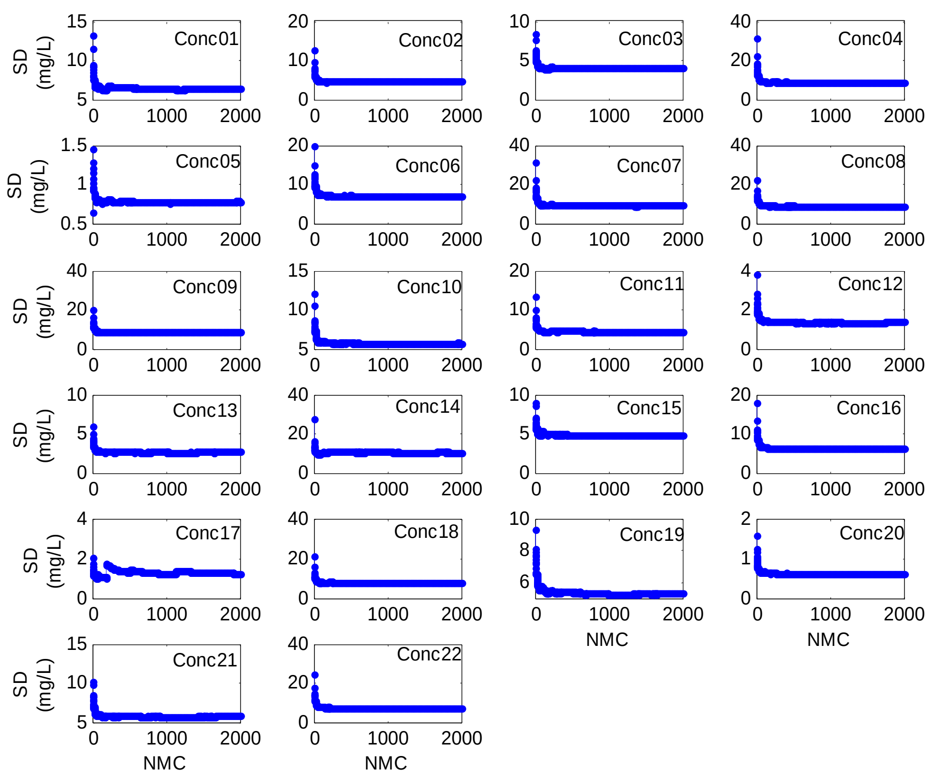

Figure 5-23. Sample standard deviation of simulated nitrogen concentration versus number of MC (NMC) simulations at the monitoring points.

Figures 5-24 and 5-25 show the histograms of the 2,000 realizations of nitrogen load estimate to the individual water bodies and to all the water bodies, respectively. Figures 5-26 and 5-27 do the same for the CDFs. Generally speaking, the load to water body 7 is the largest. As shown in the figures, it is likely that the load estimates can be larger than the deterministic values. While the maximum load estimate of 1.1 kg/d is significantly larger than the deterministic load estimate of 0.3 kg/d, it is still significantly smaller than those of Seagate Harbor and North River Shores. Therefore, it may not be necessary to spend resources of monitoring at the Hobe Sound site.

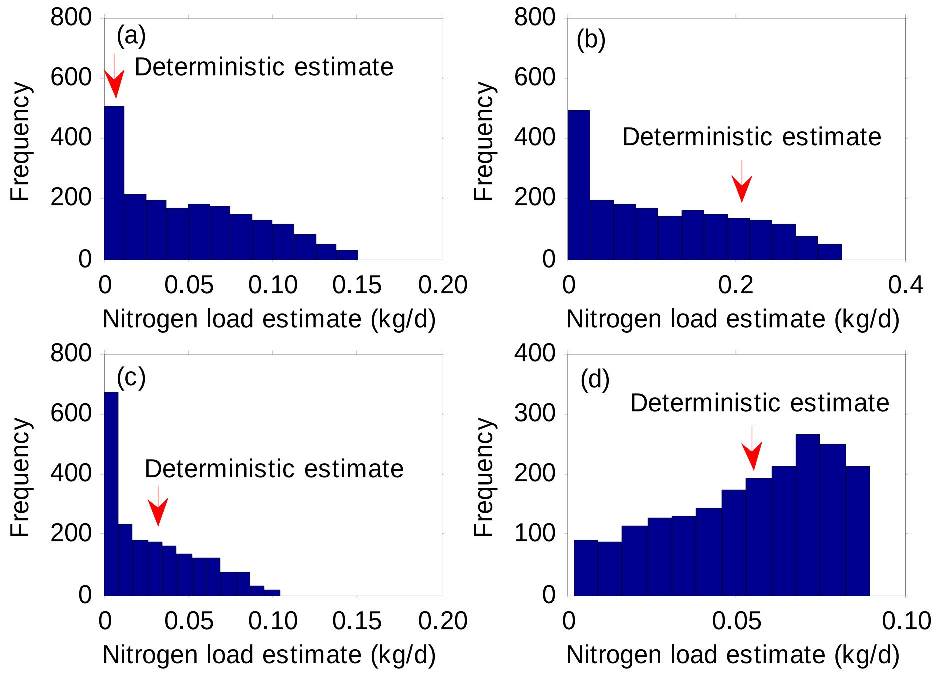

Figure 5-24. Histograms of 2,000 realizations of nitrogen load estimates for (a) water body 0, (b) water body 7, (c) water body 8, and (c) water body 9 in the Hobe Sound site.

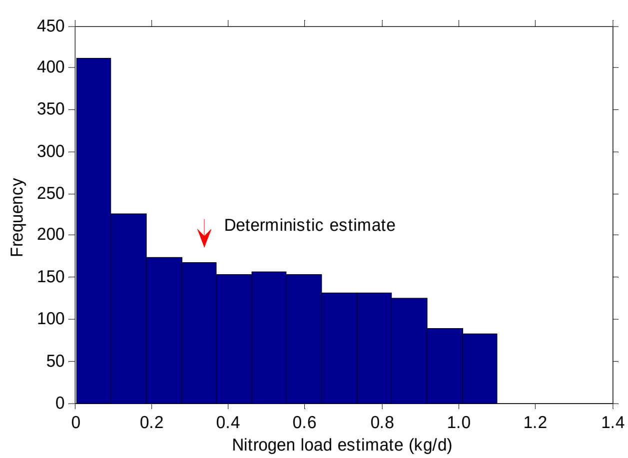

Figure 5-25. Histograms of 2,000 realizations of nitrogen load estimate to all the water bodies in the Hobe Sound site.

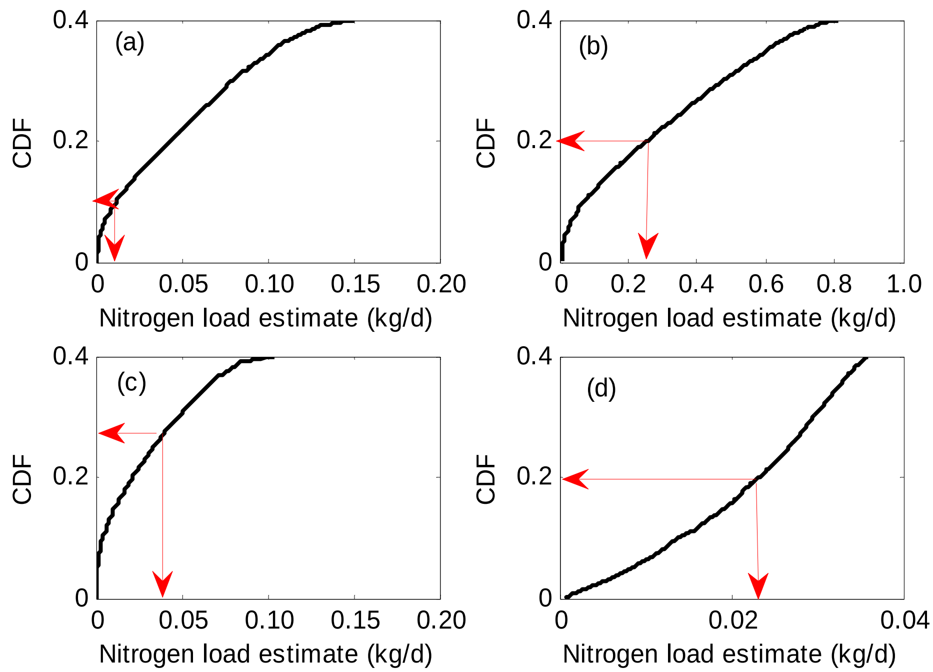

Figure 5-26. Cumulative distribution functions (CDFs) of 2,000 realizations of nitrogen load estimates for (a) water body 0, (b) water body 7, (c) water body 8, and (d) water body 9 in the Hobe Sound site.

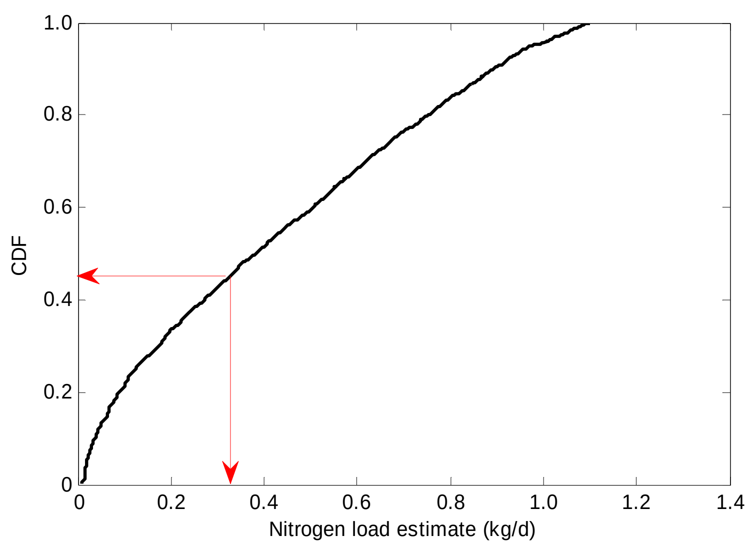

Figure 5-27. Cumulative distribution functions (CDFs) of 2,000 realizations of nitrogen load estimates to all the water bodies in the Hobe Sound site.

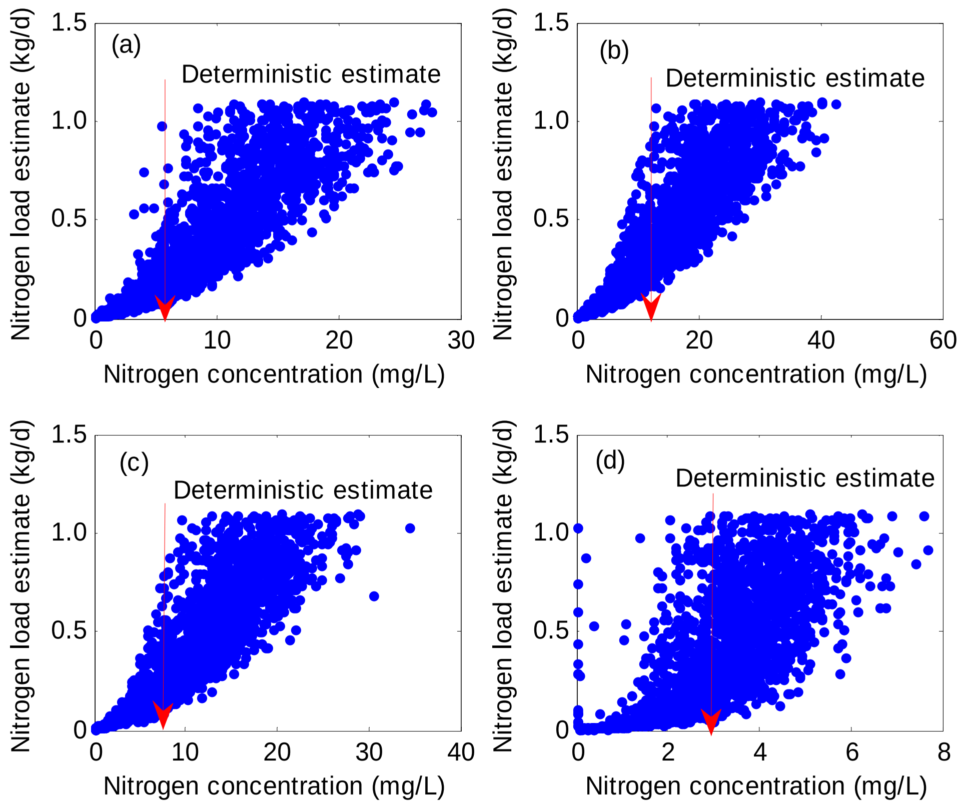

Figure 5-28. Relation between nitrogen load estimate and concentration simulation at the four monitoring points (a) Conc04, (b) Conc08, (c) Conc12, and (d) Conc16 over the 2,000 MC realizations.

Figure 5-28 shows the relation between the nitrogen load and simulated nitrogen concentration at four monitoring points whose locations are shown in Figure 5-21. In comparison with Figures 5-7 and 5-20, Figure 5-28 shows a different pattern that uncertainty in the load estimate increases with nitrogen concentration. This is not surprising, because the deterministic concentration simulations and load estimate are relatively low. From the viewpoint of uncertainty reduction, it is worthy obtaining more observations. However, this effort is questioned because the absolute amount of increase is small as discussed above.

6. CONCLUSIONS

This study uses ArcNLET to estimate nitrogen load from removed septic systems to surface water bodies in the City of Port St. Lucie, the City of Stuart, and Martin County located in the St. Lucie River and Estuary Basin. This study leads to the following major conclusions:

(1) Data and information needed to establish ArcNLET models for nitrogen load estimation are readily available in the modeling areas. The data of DEM, surface water bodies, and hydraulic conductivity can be downloaded from public-domain databases. FDEP and environmental agencies of the cities and counties have site-specific data and information such as locations of canals and septic systems. Therefore, ArcNLET models can be set up easily for other areas of interest to support the on-going BMAP.

(2) Although there is no groundwater monitoring network, historical data are available from public-domain databases (e.g., DBHYDRO and USGS websites). The data compiled in this study indicate that ammonium/TKN concentrations are significant and in general higher than NO_x concentrations in the study areas. However, the available data are limited and outdated. While the observations of water table may represent the groundwater flow system, nitrogen concentrations may not reflect the current system of nitrogen transport, which leads to limited understanding of nitrogen transport at the modeling sites.

(3) After calibrating the ArcNLET flow and transport models, model simulations can reasonably match corresponding field observations. The calibrated smoothing factor for the City of Port St. Lucie and the City of Stuart are the same; the calibrated transport parameters for the City of Port St. Lucie and Martin County are similar. However, the calibrated transport parameters are subject to substantial uncertainty due to the lack of concentration data. The uncertainty is quantified using MC simulation, and the conclusions of the MC simulation are given below.

(4) ArcNLET estimated nitrogen loads in the modeling sites vary substantially in space, and the spatial variability is useful to management of nitrogen pollution. In the City of Port St. Lucie, the canals are critical to control groundwater flow paths and nitrogen loads, and the largest load among all the surface water bodies is that to C-24 canal. Because the loads are distributed relatively uniformly over the City of Port St. Lucie, effective management of nitrogen pollution should be conducted over the entire modeling area. The load estimates are strongly correlated with nitrogen concentrations in surface water quality data, suggesting that septic load is a significant factor for water quality deterioration. In the City of Stuart and Martin County, because the areas of septic system removal are of small scale, it happens often that majority of the load is to one or two surface water bodies. At North River Shores and Seagate Harbor of Martin County, the flow paths are substantially shorter, and the load per septic system is significantly larger than those at the other sites.

(5) The ArcNLET estimated nitrogen loads are comparable with literature data of loads per area and nitrogen reduction ratios, i.e., the ratio between removed nitrogen in aquifers and input nitrogen to aquifers. The ratios are closely related to the length flow paths; the smallest reduction ratios are for North River Shores and Seagate Harbor where the flow paths are the shortest. The load estimate is also related to

groundwater velocity and soil drainage conditions. Generally speaking, the load estimate increases with the velocity and also increases when the drainage condition changes from poorly to excessively drained. The ArcNLET estimated load per septic system is smaller than that obtained using a method considered by Martin County (Dianne Hughes, 2013, Personal Communication). This is not surprising, because the Martin County method does not consider nitrogen loss in septic systems, drain fields, and aquifers.

(6) In the BMAP context, the ArcNLET estimated annual loads per area are compared with the starting loads from agriculture and natural lands listed in the draft BMAP for the St. Lucie River and Estuary Basin. For the City of Port St. Lucie and Rio Site in Martin County, the load estimate from septic systems is smaller than that from agriculture but larger than that from natural lands. For the City of Stuart and North River Shores and Seagate Harbor in Martin County, the load estimate from septic system is larger than that from agriculture. This however may not be accurate, because the removed septic systems are close to water bodies and may leads to an overestimate of the loads. When comparing the ArcNLET estimates with the BMAP estimates for all entities (excluding natural lands), it is found that, among Port St. Lucie, North River Shores, and Rio that are all located in North Fork sub-basin, the load from septic systems in Port St. Lucie is close to that of the sub-basin, the load in North River Shores is larger, and the load in Rio is smaller. Since the modeling site of Port St. Lucie is the largest among the seven sites, the results of Port St. Lucie are more meaningful than those of the other six sites.

(7) In the scenario analysis that all septic systems (removed and functioning) in the St. Lucie River and Estuary Basin are removed, the amount of nitrogen load reduction is the largest for the North Fork sub-basin, followed by the second largest one for the Basin 4-5-6. This appears to be reasonable, considering the absolutely large number of septic systems in North Fork and the relatively large numbers of septic systems in Basin 4-5-6. The results suggest that the hypothetical removal of functioning septic system is worthy for the North Fork and Basin 4-5-6 sub-basins. This may be also the case for the South Fork sub-basin, as well as for the South Coastal sub-basin which however is not included in the draft BMAP. However, for C-23 and C-44/S-135 sub-basins, the effort of removing functioning septic systems does not help reduce nitrogen load, because of the negligible amount of load reduction. This scenario analysis may be helpful for using ArcNLET to facilitate nitrogen management.

(8) The MC simulation is conducted for three sites in Martin County: the calibration site where ArcNLET is calibrated against nitrogen concentration at a monitoring well, Seagate Harbor where the load estimate is high, and Hobe Sound where the load estimate is low. For the calibration site, the histogram of ArcNLET simulated nitrogen concentration indicates that, with the parameter distributions considered in this study, it significantly more likely for the model to simulate low concentration values than to high values at the monitoring well. This is consistent with the low nitrogen concentration of 0.29 mg/L observed at the monitoring well, suggesting that the calibrated model is likely to reflect nitrogen transport at the calibration site.

(9) The overall positive correlation between the load estimate and the simulated concentration at the three sites of MC simulation indicates that larger nitrogen

concentration corresponds to larger load. In the context of site monitoring, if higher concentrations are continuously observed at the monitoring well, the load estimate should be larger than the deterministic estimate given by the calibrated model. However, at Seagate Harbor, the increase of load estimate from the deterministic estimate is limited because the deterministic estimate is already relatively large. At Hobe Sound, while the increase of load estimate can be substantial relatively to the deterministic estimate, the maximum load estimate obtained from the MC simulation is still smaller than that of the other sites. In this sense, having more monitoring data does not necessarily lead to substantial increase of load estimate. It is also possible that collecting more data leads to decrease of the load estimate. For example, if observed nitrogen concentrations are smaller than the deterministic simulation of the calibration model, the corresponding load estimate may be smaller than the deterministic estimate.

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APPENDIX A: NITROGEN LOAD TO SURFACE WATER BODIES IN THE CITY OF PORT ST. LUCIE

The modeling area in the City of Port St. Lucie includes a total of 336 surface water bodies. The FIDs of the water bodies are shown in Figure A-1, and the load to the individual water bodies and number of contributing septic systems are listed in Table A-1.

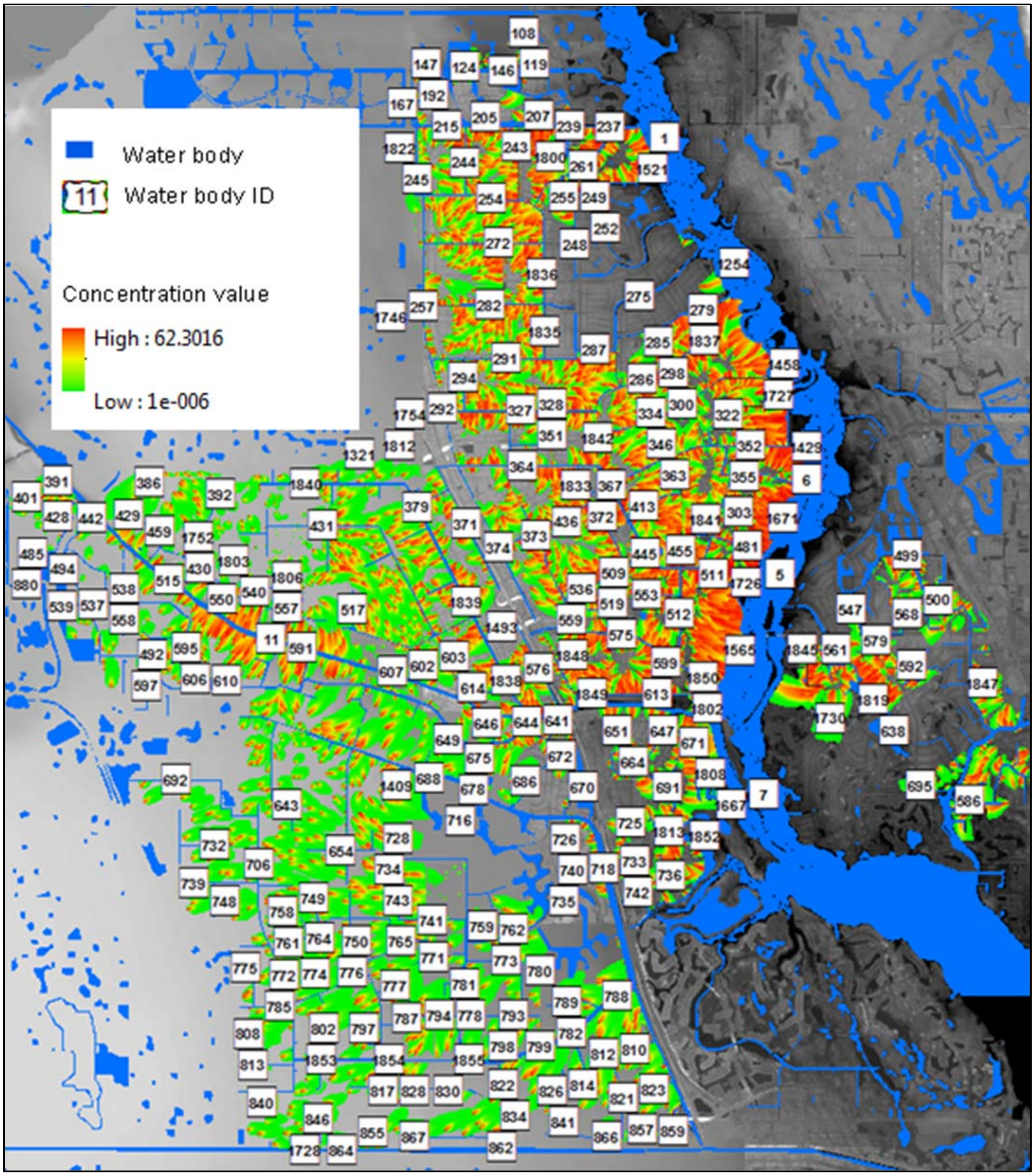

Figure A-1.Plots of the water bodies received nitrogen in the City of Port St. Lucie.

Table A-1. ArcNLET estimated nitrogen load (g/d) and number of contributing septic systems to individual surface water bodies (indexed by FID) in the City of Port St. Lucie.

Water Body FID	Nitrogen load (g/d)	Number of Contributing Septic Systems
11 (C-24 Canal)	1374.88	202
327	1282.72	162
445	1049.46	88
352	1046.85	112
300	1035.53	108
575	949.25	88
373	947.90	100
261	894.52	80
291	873.89	92
286	860.81	115
272	790.79	91
512	754.42	66
367	745.03	64
254	702.97	116
1726 (North Fork St Lucie River)	694.27	62
282	686.21	120
561	651.23	51
613	631.19	45
553	592.42	52
1837	584.09	57
1845	581.90	43
1839	573.17	70
322	569.33	41
511	567.89	46
364	563.85	57
586	555.77	58
355	548.70	38
1671	521.54	61
249	493.61	33
1565	470.45	85
1847	445.27	38
592	443.28	45
243	435.98	65
1841	423.18	49
517	423.13	139
500	418.25	30
492 (E-8)	396.97	119
279	393.26	35
1727 (North Fork St Lucie River)	384.29	33
372	379.34	59

Water Body FID	Nitrogen load (g/d)	Number of Contributing Septic Systems
1848	356.23	21
81	340.10	31
273	334.19	46
1835	305.24	38
536	296.93	34
1429	296.07	19
481	295.86	19
239	294.27	33
473	285.77	19
241	278.92	22
363	270.70	53
509	269.03	33
298	268.51	29
294	262.69	30
1836	255.09	30
7 (North Fork St Lucie River)	249.30	27
718	235.17	20
1838	235.10	31
1849	225.89	31
425	224.01	30
316	216.74	27
252	214.53	13
436	211.49	21
455	211.20	27
599	206.14	19
654	198.76	96
244	192.51	35
431	190.05	89
237	187.66	12
379	186.06	64
285	184.08	21
248	179.13	17
275	175.76	17
1850	173.18	12
499	168.89	13
371	166.17	34
257	164.06	19
289	160.41	14
246	147.51	14
242	145.28	12
1842	142.63	14
643	140.40	50
725	137.26	9

Water Body FID	Nitrogen load (g/d)	Number of Contributing Septic Systems
146	133.43	8
287	128.66	28
238	127.15	7
782	126.58	51
1852	117.67	11
559	116.36	11
557	114.69	23
5 (North Fork St Lucie River)	113.36	22
303	112.64	26
467	104.68	19
292	100.83	7
564	98.60	11
614	96.92	11
519	95.98	9
245	90.58	14
413	86.90	12
465	86.74	10
6 (North Fork St Lucie River)	86.14	5
736	85.49	8
742	84.27	6
398	81.23	5
741	78.58	51
691	76.46	8
403	74.34	11
1819	72.96	36
1840	71.44	39
416	70.87	21
576	70.59	15
329	66.43	6
823	64.20	7
859	61.52	4
374	60.62	12
568	59.97	11
733	59.15	4
451	58.91	3
328	58.86	6
334	56.99	8
1855	55.02	7
346	54.78	11
650	54.07	4
126	52.28	3
664	51.03	4
1854	50.84	14

Water Body FID	Nitrogen load (g/d)	Number of Contributing Septic Systems
750	48.78	35
603	47.87	8
1458	46.12	3
663	45.90	4
1813	45.75	10
505	44.77	4
644	43.49	3
651	41.66	4
1808	41.13	8
400	40.24	10
495	39.70	2
662	37.32	3
332	36.88	2
430	36.53	27
205	36.36	2
468	36.29	5
1802	35.70	2
706	35.21	32
647	34.77	2
540	34.47	9
504	32.70	2
726	32.46	2
459	31.98	5
422	31.03	2
779	30.48	8
668	29.95	2
646	29.52	4
703	28.72	2
147	27.89	2
734	26.91	12
201	25.90	2
671	25.38	3
602	25.29	4
830	25.20	3
1 (North Fork St Lucie River)	23.91	2
765	23.18	5
255	22.30	2
386	21.67	7
124	21.64	3
814	21.63	3
755	21.58	1
670	21.52	1
837	21.10	1

Water Body FID	Nitrogen load (g/d)	Number of Contributing Septic Systems
537	20.93	12
119	20.60	2
350	20.53	1
862	20.26	2
810	19.93	1
507	19.88	1
192	19.82	3
502	19.47	1
740	19.30	1
296	18.67	2
794	18.62	14
775	18.49	5
825	18.40	1
618	18.28	1
944	18.26	4
376	17.46	1
839	16.92	3
838	16.71	1
158	16.61	2
485	16.53	3
751	16.48	1
826	16.32	5
787	16.14	2
681	15.57	2
773	15.50	8
207	14.86	2
130	14.79	1
108	14.68	1
797	14.44	4
689	14.29	1
161	14.27	1
429	14.19	5
579	14.11	2
816	14.10	3
359	13.96	1
793	13.80	4
812	13.42	4
834	13.28	9
758	13.19	6
660	13.10	2
165	13.06	1
692	11.55	9
822	11.14	6

Water Body FID	Nitrogen load (g/d)	Number of Contributing Septic Systems
351	11.10	2
113	11.09	2
391	10.13	2
785	10.06	12
680	10.04	1
762	9.96	3
550	9.80	3
641	9.68	1
724	9.26	2
715	9.10	1
649	9.10	13
127	9.02	1
295	8.98	1
1853	8.91	4
595	8.76	3
846	8.50	6
789	8.35	5
578	8.06	6
771	7.56	9
421	7.44	2
761	7.43	5
687	7.31	3
866	7.20	1
841	6.68	1
685	6.47	1
739	6.41	5
772	6.17	6
817	5.95	16
1834	5.73	2
428	5.66	2
1728	5.10	4
638	5.00	1
808	4.69	4
539	4.66	5
471	4.51	1
392	4.24	5
565	4.00	1
607	3.84	1
695	3.84	1
749	3.73	9
728	3.63	11
855	3.60	1
1833	3.47	2

Water Body FID	Nitrogen load (g/d)	Number of Contributing Septic Systems
558	3.47	2
813	3.43	1
743	3.39	13
735	3.01	12
1493	2.93	1
167	2.60	1
610	2.52	1
491	2.41	1
1254	2.34	1
672	2.13	2
1521	2.10	4
679	2.06	1
798	2.00	3
769	1.97	1
799	1.97	1
847	1.95	2
515	1.89	1
780	1.72	2
450	1.68	1
474	1.62	1
690	1.61	1
1800	1.56	3
732	1.52	2
764	1.40	4
880	1.32	7
676	1.32	1
777	1.21	1
759	1.19	3
781	1.18	4
774	1.17	3
1409	1.12	1
215	0.99	1
1799	0.96	2
1730	0.94	4
688	0.92	8
748	0.91	2
840	0.89	1
778	0.88	4
538	0.74	1
686	0.74	2
678	0.74	2
1667	0.72	1
1807	0.72	6

Water Body FID	Nitrogen load (g/d)	Number of Contributing Septic Systems
867	0.64	1
1822	0.62	1
597	0.61	1
606	0.57	1
864	0.55	1
40	0.53	1
857	0.53	1
675	0.52	1
802	0.52	2
821	0.48	3
828	0.46	1
683	0.45	1
591	0.41	1
790	0.41	1
442	0.40	1
849	0.40	1
623	0.36	1
417	0.35	1
776	0.34	2
326	0.33	1
547	0.28	1
667	0.28	1
358	0.20	1
608	0	1
788	0	1
1752	0	11
1803	0	1
525	0	1
1321	0	1
1754	0	7
878	0	1
401	0	1
494	0	1
1806	0	7
716	0	2
1812	0	2
1746	0	1
508	0	1