Environ Earth Sci (2017)76:32
DOI 10.1007/s12665-016-6358-y

ORIGINAL ARTICLE

Numerical estimation of nitrogen load from septic systems to surface water bodies in St. Lucie River and Estuary Basin, Florida

Ming Ye[1] · Huaiwei Sun[2,1] · Katie Hallas[3]

Received: 4 August 2014 / Accepted: 8 July 2015

Abstract Nitrogen pollution is one of the most prevalent and challenging environmental problems worldwide. Wastewater treatment using onsite sewage treatment and disposal systems (aka septic systems) is one source of nitrogen sources in the environment. This study presents a numerical study for estimating nitrogen load from septic systems to surface water bodies in the St. Lucie River and Estuary Basin, Florida, USA. The load estimation is conducted using an ArcGIS-based nitrogen load estimation toolkit, a screening-level modeling software that considers key mechanisms (i.e., advection, dispersion, and denitrification) controlling nitrogen transport as well as spatial variability of model parameters (e.g., hydraulic conductivity, porosity, septic system locations, and surface water bodies). The simulated nitrogen plumes and load estimates demonstrate the importance of considering spatial variability in the load estimation for nitrogen pollution management. The load estimates are strongly correlated with nitrogen concentrations in surface water quality data, suggesting that septic systems are an important factor for water quality deterioration in the St. Lucie River. The estimates can be used directly to support Basin Management Action Plan (BMAP). Evaluating the load estimates in the BMAP context indicates that nitrogen load from septic systems is a significant nitrogen source in the St. Lucie River and Estuary Basin. It is found in this study that the load estimates depend on lengths of flow paths, groundwater flow velocities, and soil drainage conditions. These findings are useful for nitrogen pollution management in similar areas facing nitrogen pollution problems.

Keywords TMDL · BMAP · Screening-level model · GIS-based modeling · Nitrogen reactive transport · Drainage conditions

✉ Ming Ye
mye@fsu.edu

Huaiwei Sun
huaiweisun@whu.edu.cn

[1] Department of Scientific Computing, Florida State University, Tallahassee, FL, USA

[2] School of Hydropower and Information Engineering, Huazhong University of Science and Technology, Wuhan, China

[3] Florida Department of Agriculture and Consumer Services, Tallahassee, FL, USA

Introduction

Nitrogen pollution is one of the most prevalent and challenging environmental problems currently facing US coastal waters (USNRC 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 (USNRC 2000; Howarth 2008). In the continental USA, moderate to severe degradation of water quality due to nitrogen and phosphorus pollution has been reported in more than 60% 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 wastewater treatment using onsite sewage treatment and disposal systems (OSTDS) (aka septic systems). In Buttermilk Bay, MA, 40% of nitrogen and phosphorous entering watershed was from septic systems,

Published online: 28 December 2016

and 80% of the nitrogen was transported through groundwater (Valiela and Costa 1988). In Waquoit Bay, MA, 48% of the 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. In the watershed scale at the Chesapeake Bay watershed, USEPA (2013) indicates that septic systems contribute approximately 8.3 million pounds to the Bay, about 5% of the total nitrogen load. The percentage is significantly larger in Florida. Badruzzaman et al. (2012) estimated that nitrogen loading from septic systems to the Florida environment is 2.4–4.9 $\times$ 10^{10} g-N/year, which is smaller than the estimate of 1.4 $\times$ 10^{11} g-N/year from fertilizer application but larger than the estimate of 1.2 $\times$ 10^{8}–2.6 $\times$ 10^{10} g-N/year from reclaimed water and 5.9–9.4 $\times$ 10^{9} g-N/year from atmospheric deposition. The large load is not surprising, given that nearly a third of households use septic systems in Florida (Ursin and Roeder 2008). The field study at Florida Keys, FL, by Lapointe et al. (1990) found that groundwater concentration of dissolved inorganic nitrogen can be enriched an average of 400-fold due to septic systems. Given the trends in population growth, nitrogen loads from septic systems are expected to increase. Therefore, sustainable management of nitrogen pollution due to septic systems is urgently needed, and it requires a quantitative capability for estimating nitrogen loads.

Various modeling approaches have been developed for estimating nitrogen loads from septic systems, and they 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 involved in nitrogen reactive transport. An example of the simple approaches is the nitrogen load model (NLM) of Valiela et al. (1997, 2000), in which nitrogen load is evaluated as follows: nitrogen released per person per year $\times$ people/house $\times$ number of houses $\times$ 60% not lost in septic tanks and leaching fields $\times$ 66% not lost in plumes $\times$ 65% not lost in aquifer. The percentages were based on literature values, field data, and best engineering judgment. Other modeling methods similar to NLM have been used widely in the studies of, e.g., Vadeboncoeur et al. (2010) at Narragansett Bay, Giordano et al. (2011) at Virginia Lagoon, and Kinney and Valiela (2011) at Great South Bay. A collection of such methods can be found in the NLOAD software developed by Bowen et al. (2007), which is an interactive, web-based modeling tool for nitrogen management in estuaries. However, the NLM methods do not explicitly consider spatial variability of nitrogen concentrations and loads. The spatial variability is important and needs to be considered in nitrogen pollution management (Lapointe et al. 2012). For example, Sigua and Tweedale (2003) reported that nitrogen concentrations in the central Indian River Lagoon can be twice as large as those in the northern Lagoon. Therefore, more sophisticated methods are needed to simulate spatial variability of nitrogen concentrations and loads.

Complex computer codes of groundwater reactive transport modeling can handle not only spatial variability but also coupled bio-hydro-geochemical processes that determine nitrogen fate and transport. Spiteri et al. (2008) developed a numerical model with focus on simulating biogeochemical processes occurring in submarine groundwater discharge. Meile et al. (2010) and Porubsky et al. (2011) developed two-dimensional models that solve the advection–dispersion equation coupled with the reaction network encompassing reactions of sorption–desorption, nitrification, denitrification, and dissimilatory nitrate reduction to ammonium. TOUGHREACT-N, a numerical code developed by Maggi et al. (2008), is probably 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. However, the complex models are mainly for fundamental research and are of limited use in environmental management for several reasons. First, the complexity of the models 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 data for model input and calibration as well as long execution time are needed, which, however, is not always available and/or unaffordable in practice. For many management projects of nitrogen transport modeling and load estimation, including those related to environmental regulation such as total maximum daily load (TMDL), more practical approaches are needed for estimating nitrogen load from septic systems.

This study uses the ArcGIS-based Nitrate Load Estimation Toolkit (ArcNLET) developed by Rios et al. (2013a) to simulate nitrogen transport from septic systems in surficial aquifers and to estimate corresponding nitrate load to surface water bodies. In comparison with the simple models such as NLM, ArcNLET considers the advection, hydrodynamic dispersion, and denitrification processes involved in nitrate transport. It also incorporates heterogeneity of hydraulic parameters and spatial variation of septic locations and surface water bodies. As shown in Wang et al. (2013a), ArcNLET is able to simulate spatial variability of nitrogen concentrations and loads at the neighborhood scale. Recently, Monte Carlo function was added in ArcNLET to quantify uncertainty in load estimates (Rios et al. 2013b; Ye et al. 2014). However, since ArcNLET relies on a simple conceptual model (described below) and uses an analytical equation to describe nitrogen transport, ArcNLET is a screening-level modeling tool that can be used to provide quick estimates, especially when data and knowledge are limited to set up a complicated model.

ArcNLET has the features of GIS-based screening models, 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). The GIS-based models are easy to use for preparing model input files and for post-processing model output files. They also provide a modeling environment for simulating quantities of interest and for analyzing model results by non-technical citizens. A number of GIS-based programs 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, WARMF (Herr et al. 2001) and WQM-TMDL-N (Yang et al. 2014) for TMDL calculation. The GIS-based models are particularly useful for environmental management due to the following reasons: (1) the skills required for applying GIS-based models are widely available; (2) the models are easy to set up and computationally efficient to execute; (3) the modeling results are readily available to interpret and visualize within GIS; (4) the results are quantitative and can be used directly for environmental management and regulation; (5) the modeling process is transparent for public scrutiny of stakeholders. For these reasons, GIS-based models have gained popularity in environmental modeling and management (USNRC 2010).

The site of interest to this study is the St. Lucie River and Estuary Basin (Fig. 1) located in Martin, St. Lucie, and Okeechobee Counties in southeast Florida. It is a major tributary to the Southern Indian River Lagoon, where ecological and biological conditions have deteriorated in the last several decades due in part to nitrogen pollution (Sigua et al. 2000). According to Ye and Sun (2013), monitoring data of surface water quality in the St. Lucie River indicated that, for most monitoring periods, total nitrogen (TN) concentrations at all monitoring stations are higher than the annual TMDL target of 0.72 mg/L specified in the Basin Management Action Plan (BMAP 2013). This may be attributed partly to nitrogen load from septic systems. A study of Sigua and Tweedale (2003) showed that, at the Indian River Lagoon, the nitrogen load from groundwater seepage is 84,920 kg/year, the second largest source after agricultural/urban runoff. In the groundwater load, a large portion is expected to be from septic systems, especially in areas with high population density, because of elevated concentrations of fecal coliform bacteria (Lapointe et al. 2012). However, the nitrogen loads from septic systems have not been included in the Basin Management Action Plan (BMAP 2013). The BMAP includes various management means, and one of them is to replace existing septic systems and connect them to a central sewer line. In the study area, sewer lines have been built, and new houses and existing houses with failed septic systems are required to be hooked to the sewer lines. However, nitrogen load reduction due to the septic system removal is still unknown. ArcNLET is used in this study to estimate the load reduction at seven sites in the St. Lucie River and Estuary Basin. The load estimates can be used directly for planning and controlling wastewater nitrogen loading, as well as for nitrogen trading and offset program described in US EPA (2013), which can help minimize overall cost of TMDL implementation. Since available data and information of groundwater flow and nitrogen transport in the St. Lucie River and Estuary Basin are scarce, it is more suitable to develop a simple nitrogen model than to develop a complicated one.

In the remainder of this paper, the conceptual model groundwater flow and nitrogen transport used in ArcNLET and its computational implementation are briefly described in "Conceptual and mathematical models of ArcNLET" section. The "ArcNLET modeling for the St. Lucie River and Estuary Basin, Florida, USA" section presents the ArcNLET modeling procedure and modeling results for nitrogen load estimation. Discussion of the results is given in "Discussion of load estimates in BMAP context" section, followed by a summary of the "Conclusions" section.

Conceptual and mathematical models of ArcNLET

The conceptual and mathematical models of ArcNLET are described here briefly to make the paper self-contained. More details of the models are referred to Rios (2010) and Rios et al. (2011b, 2013a). The conceptual model of ArcNLET has three sub-models: groundwater flow, nitrate, and nitrate load estimation. In the groundwater flow model, because of the assumption that water table is a subdued replica of topography in surficial aquifers, the shape of water table is obtained by smoothing land surface topography (given in the DEM format), not by solving a partial differential equation of groundwater flow. The smoothing operation is done by applying a smoothing filter to the DEM, and the details are referred to (Rios et al. 2011a, b, p.30). The number of the smoothing operation, called smoothing factor, is specified by ArcNLET users as an input parameter. If measurements of hydraulic heads are available, the smoothing parameter can be obtained by calibrating ArcNLET-simulated water table shape against

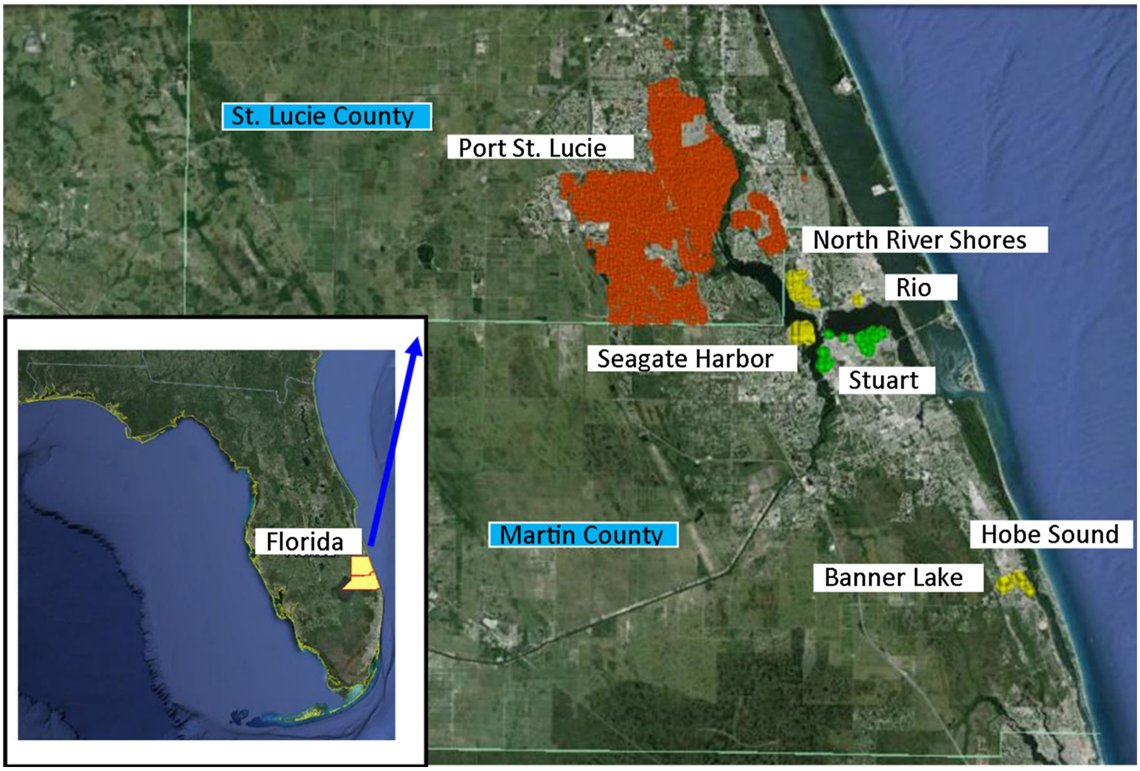

Fig. 1 Location of seven modeling sites in the city of Port St. Lucie, the city of Stuart, and Martin County in the St. Lucie River and Estuary in Florida, USA

the real shape based on field measurements. After the smoothing process, hydraulic gradients are estimated from the smoothed DEM. By invoking the Dupuit–Forchheimer assumption, the vertical flow can be ignored and only two-dimensional (2-D) isotropic horizontal flow is simulated. The groundwater seepage velocity, v, is evaluated via Darcy's law,

$$v_x = -\frac{K}{\phi}\frac{\partial h}{\partial x} \approx -\frac{K}{\phi}\frac{\partial z}{\partial x}, \quad v_y = -\frac{K}{\phi}\frac{\partial h}{\partial y} \approx -\frac{K}{\phi}\frac{\partial z}{\partial y} \tag{1}$$

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 $\partial z/\partial y$). Running the groundwater flow model yields the magnitude and direction of flow velocity for every discrete cell of the modeling domain, and they are used to estimate flow paths originating from individual septic systems and ending in surface water bodies. The calculation considers heterogeneity of hydraulic conductivity and porosity as well as spatial variability in the locations of septic systems and surface water bodies. The values of hydraulic conductivity and porosity can be obtained from field measurements, literature data, and/or by calibration against measurements of hydraulic head and groundwater velocity.

The transport model shown in Fig. 2 is similar to that used in USEPA models BIOSCREEN (Newell et al. 1996) and BIOCHLOR (Aziz et al. 2000). In the conceptual model, nitrogen enters groundwater from the source plane (i.e., the Y–Z plane shown in Fig. 2) with a constant concentration C_0 [ML^{-3}]. The 2-D nitrogen transport in groundwater is described by using the advection–dispersion equation,

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

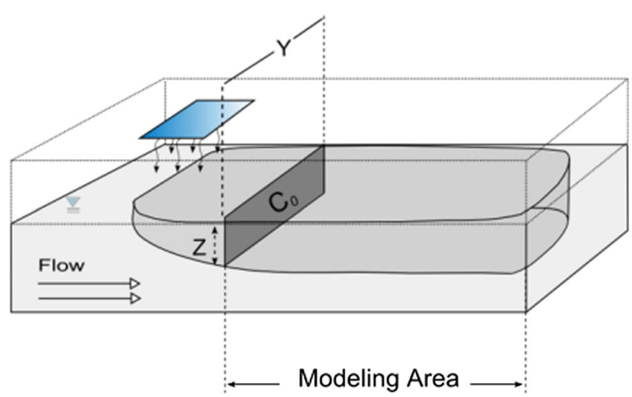

Fig. 2 Conceptual model of nitrogen transport in groundwater adapted from Aziz et al. (2000)

where C is nitrogen concentration [M/L^3], t is time [T], $D_x = \alpha_x v$ and $D_y = \alpha_y v$ are the dispersion coefficients in the x and y directions, respectively [L^{2}T^{-1}], α_x and α_y are the dispersivities, respectively [L], v is the constant seepage velocity in the longitudinal direction [L], and k is the first-order decay coefficient [T^{-1}]. Homogeneous parameters (e.g., dispersion coefficient) and uniform flow in the longitudinal direction are assumed in Eq. (2). Following McCray et al. (2005) and Heinen (2006), the first-order kinetic of denitrification process is used, and it corresponds to the kC term in Eq. (2); in the denitrification process, nitrate is transformed into nitrogen gas through a series of biogeochemical reactions. By assuming that ammonium and nitrate have the same transport mechanism, ArcNLET simulates nitrogen transport and estimates nitrogen load from septic systems to surface water bodies. Based on Domenico (1987) and West et al. (2007), Rios et al. (2013a) derived the steady-state analytical solution used in ArcNLET. Different septic systems have different concentration plumes because flow velocity varies between the septic systems. The plumes may or may not reach surface water bodies, depending on the magnitude of denitrification. 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 plumes are written to output files that are ArcGIS layers and can be readily post-processed and visualized within ArcGIS.

In the conceptual model of nitrate load estimation, the 2-D concentration plume is extended downward to the depth Z [L] of the source plane to create the pseudo-three-dimensional plume shown in Fig. 2. The model first evaluates the inflow mass rate,M_{in} [MT^{-1}], to groundwater, consisting of inflow mass due to advection and dispersion. For the steady-state model, the outflow mass rate, M_{out}, to surface water bodies is evaluated by using the mass balance equation, $M_{out} = M_{in} - M_{dn}$, where M_{dn} [MT^{-1}] is mass removal rate due to denitrification evaluated for each plume if the plumes reach surface water bodies.

ArcNLET modeling for the St. Lucie River and Estuary Basin, Florida, USA

This study uses ArcNLET to estimate nitrogen loads (as a TMDL credit) from a large number of removed septic systems to surface water bodies. Figure 1 shows the locations (approximated by the geometric center of each parcel) of the removed septic tanks, among which 5601 (red) are located in the City of Port St. Lucie, 1087 (yellow) in Martin County, and 146 (green) in the City of Stuart. This section starts with a discussion of the data needed to set up the ArcNLET model and to calibrate the model. Subsequently, the results of model calibration and the load estimate are given. More details of the data, modeling process, and analysis of model results are referred to Ye and Sun (2013).

Data sources

An ArcNLET run requires five ArcGIS layers prepared outside ArcNLET. They are (1) DEM, hydraulic conductivity, and porosity in raster form, (2) septic system locations in point form, and (3) surface water bodies in polygon form. Except the ArcGIS layer of septic tank locations that is managed by county and state environmental agencies, other ArcGIS files are readily available from national databases available in the public domain. The layers of DEM and surface water bodies can be downloaded directly from the US Geological Survey (USGS) website of national map, available at http://nationalmap.gov/ (accessed as of July 2014). The layers of hydraulic conductivity and porosity can be produced by following the procedure described in Wang et al. (2011) using the Soil Survey Geographic (SSURGO) database available at http://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm (accessed as of July 2014) managed by the US Department of Agriculture (USDA). It is usually necessary to incorporate into the ArcGIS layers site-specific data that are not available in the national database. In this study, a canal layer for the City of Port St. Lucie is merged into the water body layer. In addition, the LiDAR DEM with a resolution of 10×10 feet^2 is downloaded from the South Florida Water Management District (SFWMD) GIS Data Catalog available at http://www.floridadisaster.org/gis/lidar/ (accessed as of July 2014) for the coastal part of St. Lucie and Martin Counties.

Calibrating the ArcNLET model parameters is always needed to match model simulations to field observations of hydraulic head and nitrogen concentration. Given that the flow model is steady state, average (over time) water levels are used as the calibration target. Using the data portals of DBHYDRO managed by SFWMD and Hydrologic Information System (HIS) managed by the Consortium of Universities for the Advancement of Hydrologic Science, Inc. (CUAHSI), a total of seven monitoring wells are identified for the modeling area in the City of Port St. Lucie. Similarly, six wells are identified in the City of Stuart. There is no monitoring well available in the modeling area in Martin County. Although the data were collected in the period of 1988–1995, their mean values are similar to those of long-term monitoring data in the modeling area (Ye and Sun 2013).

Observations of nitrogen concentration are extremely scarce in the modeling area, and the data quality is also low as indicated in the databases that contain the data. Five monitoring wells are selected, among which four wells are located in the City of Port St. Lucie and one well in Martin County. The data quality is also low. The calibration target is the average (over time) concentration of TN or (DIN), depending on availability of ammonium or TKN concentrations. At the monitoring wells, nitrate (NO_x^-) concentrations are significantly smaller than ammonium NH_4^+ concentrations, suggesting an incomplete nitrification process due to shallow water table (Wang et al. 2013b). However, this may not a general case, as indicated by the study of Belanger et al. (2009) in the northern Indian River Lagoon. More data are needed to better understand nitrogen reactive transport in the modeling area.

ArcNLET model calibration

Following Wang et al. (2013a), calibration of the 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 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 1 lists the initial and calibrated values of the parameters as well as the parameter ranges reported in literature. Because there are no water level data in Martin County, the smoothing factor is not calibrated. Instead, the calibrated smoothing factor of the City of Port St. Lucie and the City of Stuart is used for the five sites in Martin County (Fig. 1). Similarly, the calibrated transport parameter values of Martin County are used in the ArcNLET modeling for the City of Stuart. The values of hydraulic conductivity and porosity are taken from the SSURGO database and not calibrated. The inflow mass [M_{in} (g/day)] from a septic system to groundwater is approximated as follows: nitrogen release per person per day (4.8 kg/yr used in 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 estimate is 23.0 g/day (8.4 kg/year) and similar to 21.7 g/day reported in the Wekiva study in Florida (Roeder 2008). It is also close to the value of 22.4 g/day calculated by Anderson (2006). This value is used for all the septic systems in the modeling sites. It should be noted that, when information is available to estimate inflow mass for each individual septic system, ArcNLET allows assigning different values of inflow mass to different septic systems to take into account of spatial variability of the inflow mass.

Figure 3 plots the measured water levels and smoothed DEM at the wells in the City of Port St. Lucie and the City of Stuart. The smoothed DEM is obtained using the calibrated smoothing factor of 40, i.e., the smoothing through spatial average is performed 40 times for the modeling domain. The figure shows that the smoothed DEM agree well with the observed water level in that the slope of the linear regression is close to 1 (0.931 for the City of Port St. Lucie and 1.051 for the City of Stuart) and that the linear correlation coefficient, R^2, is statistically meaningful (0.968 for the City of Port St. Lucie and 0.759 for the City of Stuart). The slope of regression close to 1 is critical, since it ensures that the shape of the smoothed DEM mimics the shape of water table. The smoothed DEM is higher than the water level, and the distance between them is the intercept of the linear regression equations shown in Fig. 3. It is not surprising that the calibrated smoothing factor is the same at the two sites close to each other (Fig. 1). The smoothing factor of 40 was also used for Martin County, where the flow model calibration was not conducted due to the lack of water level measurements.

The calibrated transport parameter values are similar for the City of Port St. Lucie and Martin County, except for longitudinal and horizontal transverse dispersivity, as shown in Table 1. Transport model calibration was not conducted for the City of Stuart due to the lack of concentration data. Figure 4 illustrates the flow paths and

Table 1 Ranges, initial values, and calibrated values of ArcNLET model parameters for all the sites

Parameter	Range	Initial value	The City of Port St. Lucie	The City of Stuart	Martin County
Smoothing factor	20–80	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 (day^{-1})	5.4×10^{-5}–0.015	0.005	0.001	–	0.001

The parameter ranges are given in Ye et al. (2014) and Rios et al. (2013b) based on a literature review

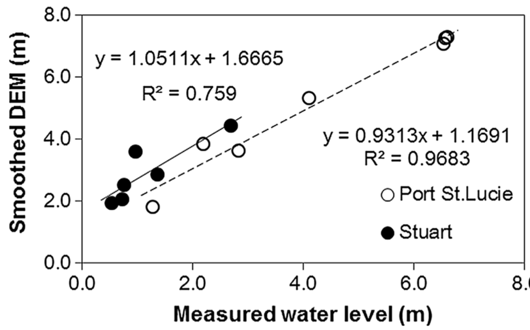

Fig. 3 Smoothed DEM and measured water level at the city of Port St. Lucie and the city of Stuart

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. The figures illustrate the flow pattern of groundwater discharge to surface water bodies and the spatial distribution of the nitrogen plumes. The simulated plume lengths range between 15 and 60 m, which agree with the plume lengths reported in literature. The field data of Belanger et al. (2009) indicate that, for residential houses near water bodies, plume migration distances range between 10 and 30 m. Robertson et al. (1991) reported that plumes from septic systems may extend up to 130 m. Figure 5 shows the comparison of simulated and measured nitrogen concentrations at the five monitoring wells. The match between the simulations and observations is acceptable at three wells (Fig. 5b) but not at the other two wells (PG-25 and SOFLSUS2-21), where the measured concentrations are significantly smaller than the simulated ones. The difference may be attributed to various reasons such as low quality of the measurements, locations of the septic tanks, and biogeochemical processes not considered in the model. While a better match can be obtained at the two wells by further adjusting model parameters (e.g., increasing the values of decay coefficient of denitrification and/or horizontal transverse dispersivity), it is not implemented because the amount of nitrogen reduction ratio in the City of Port St. Lucie is comparable with the literature data, as discussed below.

Load estimation and result evaluation

The calibrated parameters listed in Table 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. Figure 6 illustrates simulated nitrogen plumes from the removed septic systems to surface water bodies. ArcNLET

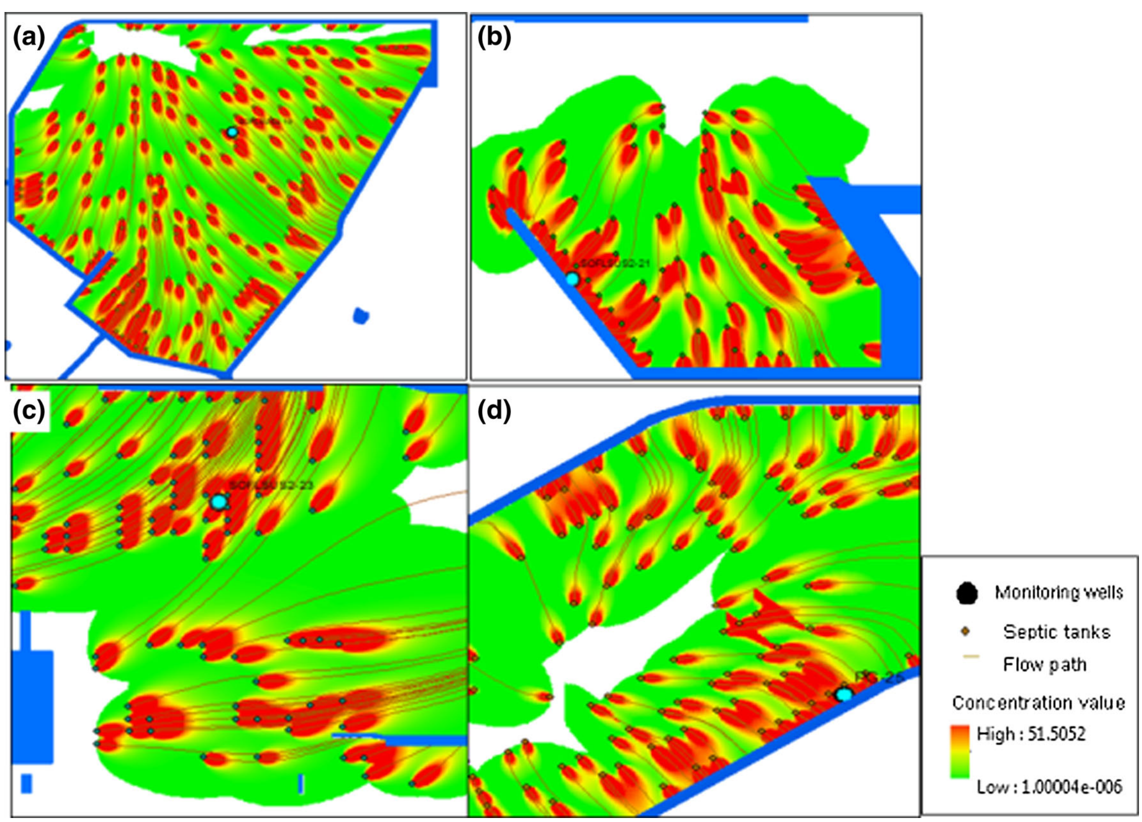

Fig. 4 Simulated flow paths and plumes by the calibrated ArcNLET model in the city of Port St. Lucie around monitoring wells **a** SOFLSUS2-19, **b** SOFLSUS2-21, **c** SOFLSUS2-23, and **d** PG-25. Surface water bodies are plotted in *blue*

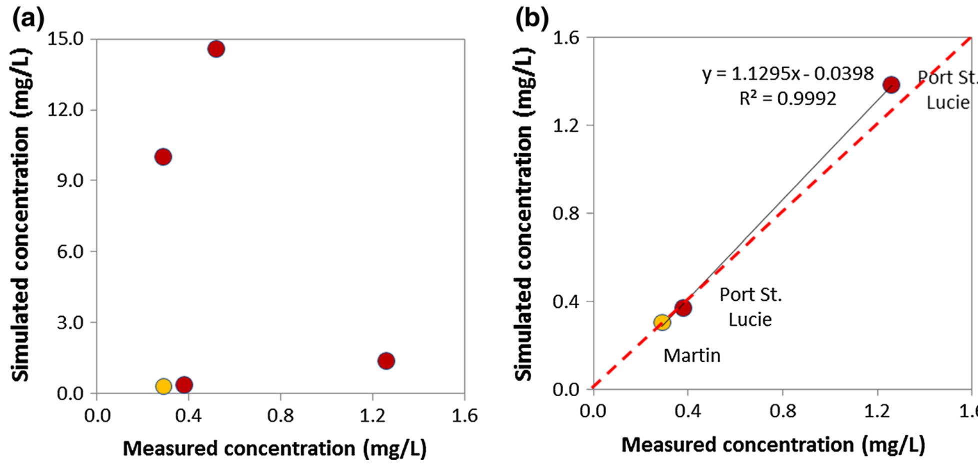

Fig. 5 Measured and simulated concentrations **a** at all five monitoring wells in the city of Port St. Lucie (*red*) and Martin County (*yellow*) and **b** at the three monitoring wells excluding wells PG-25 and SOFLSUS2-23 in the city of Port St. Lucie. The 1:1 *line* is also shown in **b**

estimates nitrogen loads to the individual water bodies labeled by their FIDs in Fig. 6. The canals are important to the load estimation, because groundwater far away from the St. Lucie River is discharged to canals, not directly to the river. Among the estimated nitrogen loads (42.48 kg/day from the 5601 septic systems), the largest one is 1.375 kg/day from 202 septic systems to C-24 canal (FID 11), the longest water body in the NHD database (the river is separated into multiple smaller water bodies in the database). It is useful to consider spatial variability of the load estimates for nitrogen pollution management (e.g., to determine areas with high priority for septic removal), because a large number of septic systems do not necessarily lead to a large load estimate. For example, the load of 1.05 kg/day from 112 septic systems to water body 352 is similar to 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, as shown in "Controlling factors of load estimates" section. In other words, the load estimation should not be based only on the number of septic systems but needs to consider spatial variability of flow and transport conditions in the modeling areas.

Due to the lack of field-scale load measurements, the load estimates are evaluated indirectly by comparing the estimates with literature data of nitrogen reduction ratio per septic system due to denitrification. As shown in Table 2, the reduction ratio is calculated by dividing the amount of nitrogen reduction due to denitrification (i.e., the difference between the load to groundwater and the load to surface water bodies) by the load to groundwater, i.e., $M_{dn}/M_{in} \times 100\%$. The ratios of this study are comparable with the literature data, except for the ratios at North River Shores and Seagate Harbor (Fig. 1). The reason is that septic systems at the two sites are close to water bodies, and more details are given below together with the controlling factors of the load estimates.

Another way of evaluating the ArcNLET-estimated nitrogen load estimates is to relate them with water quality of surface water monitoring, because high load estimates should correspond to high nitrogen concentrations in surface water bodies. Concentrations (mg/L) of total nitrogen, NO_x (nitrate and nitrite), and TKN were sampled at fourteen monitoring stations 1–2 times per year from 2004 to 2012 (BMAP 2013). At all the stations, the TN concentrations were higher than the TMDL target of 0.72 mg/L for most of the monitoring period. Figure 7 plots the median nitrogen concentration at the fourteen monitoring stations and the groundwater nitrogen loads per unit length along the water bodies corresponding to the monitoring stations. The linear relation between the median concentrations and logarithm of the load estimates suggests that nitrogen loads from septic systems to surface water bodies could be 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.

Controlling factors of load estimates

It is found in this study that the load estimates depend on the following three physical factors: length of flow paths, flow velocity, and drainage condition. Figure 8a plots the load estimates with the mean lengths of flow paths at the seven modeling sites (for each septic system, the length of flow path is from the septic system to receiving surface water body). The figure shows that shorter flow paths generally correspond to larger load estimates, because of less amount of denitrification. The two largest loads per septic systems are for North River Shores and Seagate Harbor where the flow paths are the shortest; Fig. 9 illustrates the short flow paths and simulated nitrogen plumes at Seagate Harbor, where septic systems are close to water bodies (mainly canals). A similar relation between mean length of flow path and nitrogen load was also reported in Meile et al. (2010). It is therefore important to consider the

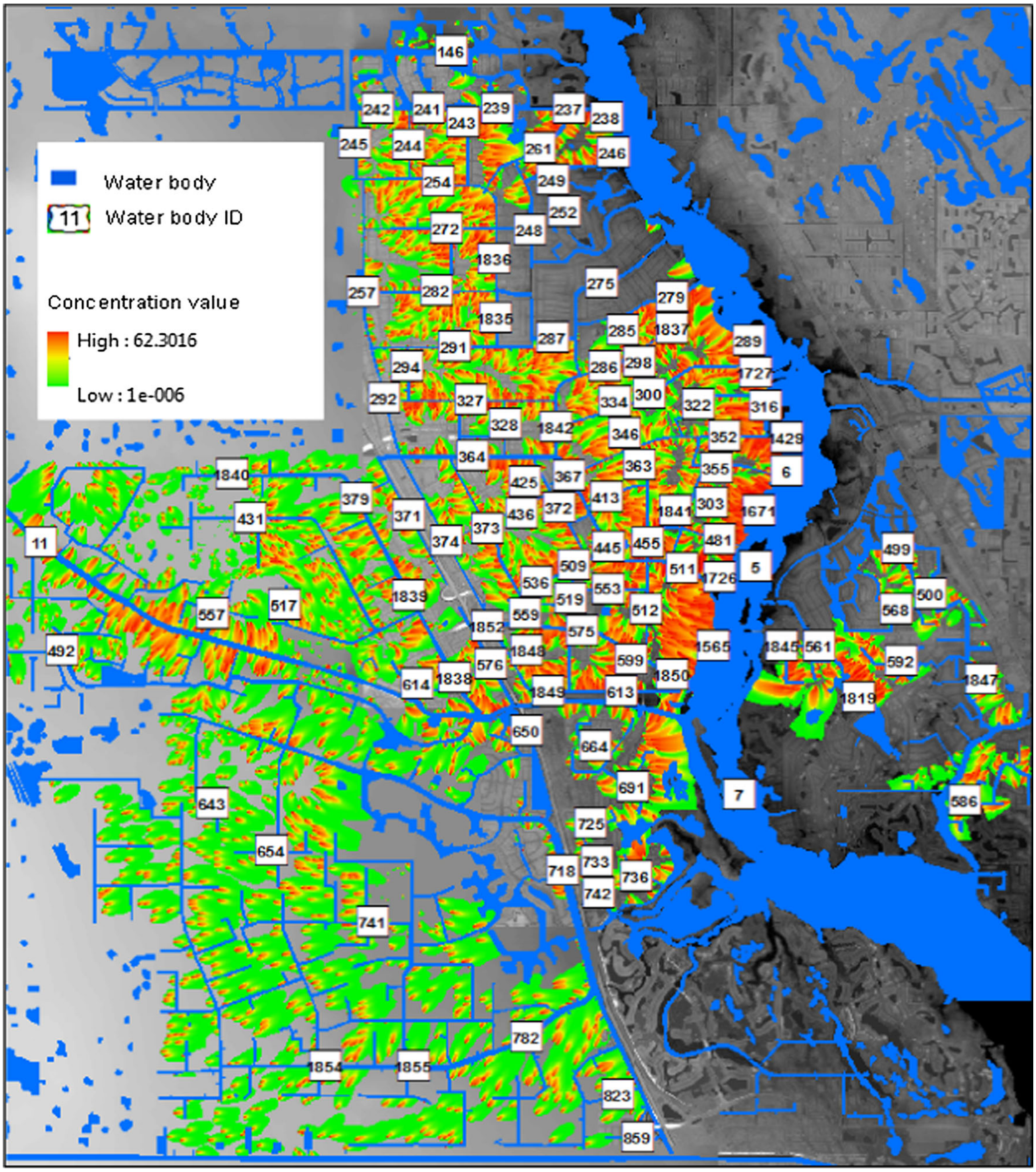

Fig. 6 Simulated nitrogen plumes from removed septic systems in the city of Port St. Lucie. The FIDs of water bodies with the estimated load larger than 0.05 kg/day are labeled. Water bodies are plotted in *blue*

distances between septic systems and surface water bodies in the management of nitrogen pollution by establishing appropriate setback distances.

Figure 8b plots variation of the load estimate with the mean velocity at the seven modeling sites. It shows that the load estimate generally 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. The two plots in Fig. 8 suggest 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. The flow condition is closely related to the drainage conditions, which 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). Figure 10 plots the load estimate per septic system and soil drainage conditions in the City of Port St. Lucie. The figure shows that the load estimate increases when the drainage condition changes from very

Table 2 Estimated nitrogen load to groundwater and surface water bodies and nitrogen reduction ratio per septic system from the literature and this study

Reference	Site location	Nitrogen loads per septic system (g/day)	Nitrogen loadings to surface water per septic system (g/day)	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

[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 (4.8 kg) × 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, 30, and 58 m, respectively

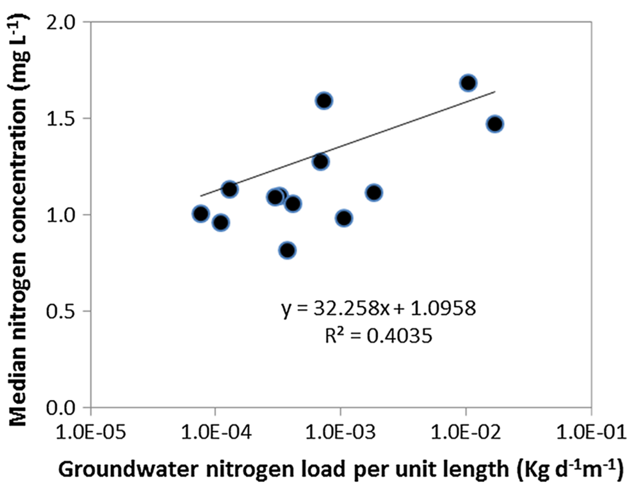

Fig. 7 Median nitrogen concentrations at fourteen surface water monitoring stations and groundwater nitrogen loads per unit length along the water bodies corresponding to the monitoring stations. The *x*-axis is in the logarithm scale

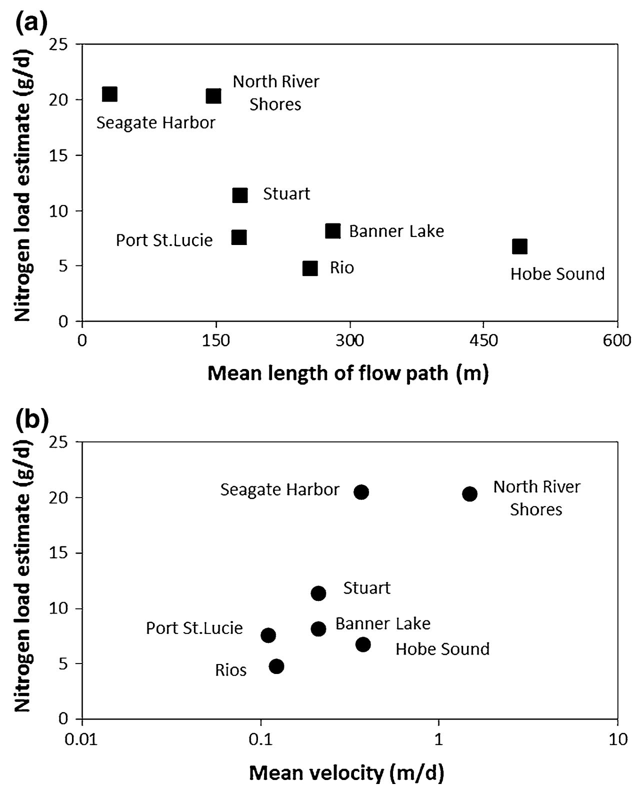

Fig. 8 Variation of nitrogen load estimate per septic systems with **a** mean lengths of flow paths and **b** mean velocity in the seven sites of this study

poorly drained to excessively drained. This is not surprising, because nitrogen transport is faster in well-drained soils than in poorly drained soils. It is thus important to consider heterogeneity in hydraulic conductivity and porosity as implemented in ArcNLET.

Discussion of load estimates in BMAP Context

The ArcNLET-estimated nitrogen loads are discussed in the BMAP context by comparing the estimated annual loads per hectare with the loads from agriculture, natural lands, and all entities (excluding natural lands) listed in BMAP (2013). The St. Lucie River and Estuary Basin is separated into several sub-basins. The City of Port St. Lucie, North River Shores, and Rio are located in the North

Fig. 9 Simulated nitrogen plumes from the removed septic systems in Seagate Harbor, where the nitrogen reduction rate is the largest among the six modeling sites due to short distance between septic systems and water bodies (plotted in *blue*)

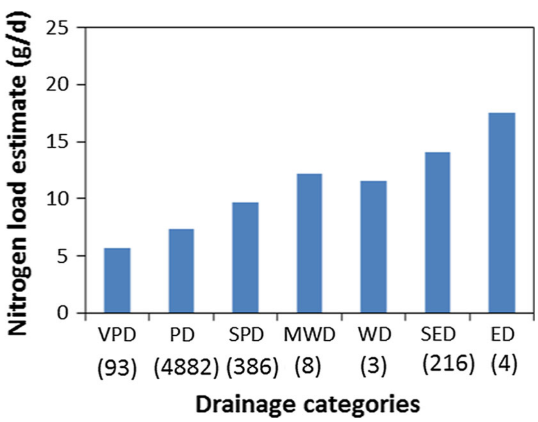

Fig. 10 Variation of nitrogen load estimate per septic systems with drainage conditions of the soil zones where septic systems are located at the city of Port St. Lucie site. Abbreviations of the drainage conditions are as follows: *ED* excessively drained, *SED* somewhat excessively drained, *WD* well drained, *MWD* moderately well drained, *SPD* somewhat poorly drained, *PD* poorly drained, *VPD* very poorly drained. The number of septic systems corresponding to each drainage condition is given in the parentheses

Fork sub-basin, the City of Stuart in the South Fork sub-basin, and Seagate Harbor is located in Basin 4–5–6. Banner Lake and Hobe Sound of the Martin County are located in South Coastal sub-basin, which is not considered in the current BMAP (2013). Annual nitrogen loads (kg/ha) from septic systems in the three sub-basins were estimated by multiplying the daily load (g/day) per septic system by the house density of 1.89 homes/ha in the City of Port St. Lucie. This is equivalent to extrapolating the load estimate from the neighborhood scale to the sub-basin scale. The estimates are listed in Table 3, which also lists the BMPA (2013) load estimates from all BMAP entities excluding natural lands where no management is needed. It should be noted that nitrogen loads from septic systems were not considered in the current BMAP (2013). The ArcNLET-estimated loads in the City of Port St. Lucie and the City of Stuart are similar to those of BMAP (2013), suggesting that it is important to include the septic loads in BMAP practice. In the other three areas listed in Table 3, the load estimates are significantly different from the BMAP estimates, and this may be attributed to the extrapolation from the neighborhood to the sub-basin scale. Nevertheless, the results for the City of Port St. Lucie are expected to be meaningful, because of the large modeling domain and the large number of septic systems considered in the modeling (Fig. 1). The comparison in Table 3 suggests significance of septic systems to nitrogen load relative to other nitrogen sources (e.g., agriculture and natural lands) included in BMAP.

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. Because data and information needed to establish ArcNLET models for nitrogen load estimation are readily available in public-domain databases, ArcNLET models can be set up easily for supporting ongoing TMDL and BMAP practice for environmental management and protection.
2. After calibrating the ArcNLET flow and transport models, model simulations can reasonably match corresponding field observations of hydraulic head and nitrogen concentration. However, due to the lack of concentration data, the calibrated transport parameters are subject to substantial uncertainty, and quantifying the uncertainty is necessary in a future study to support science-informed decision making.
3. ArcNLET-estimated nitrogen loads in the modeling sites vary substantially in space, and it is necessary to consider the spatial variability for management of nitrogen pollution such as determining areas with high priority of converting septic systems to central sewage. As shown in Fig. 7, the load estimates are correlated with nitrogen concentrations in surface water quality data, suggesting that septic load could be a significant

Table 3 Annual nitrogen load (kg/ha) estimated by ArcNLET and BMAP (2013) from all entities of BMAP (excluding natural lands)

Sites	Corresponding sub-basins	ArcNLET	BMAP (2013)
Port St. Lucie	North Fork	5.24	5.25
North River Shores	North Fork	14.01	
Rio	North Fork	3.31	
Stuart	South Fork	7.86	7.02
Seagate Harbor	Basin 4–5–6	14.15	6.55

factor for water quality deterioration in the modeling areas.

4. The ArcNLET-estimated nitrogen loads are comparable with literature data in terms of nitrogen reduction ratios, i.e., the ratio between removed nitrogen in aquifers and input nitrogen to aquifers. The ratios appear to be 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 drainage condition improves.
5. In the BMAP context, the ArcNLET-estimated annual loads per area are comparable with the BMAP estimates for all entities (excluding natural lands). It is found that the load from septic systems in Port St. Lucie is close to that of the sub-basin, suggesting that it is important to consider septic nitrogen load for future BMAP practice.

Since the monitoring data of hydraulic head and nitrogen concentration are limited and outdated in the modeling areas, more observations (especially those of nitrogen concentration) are needed to better understand the current groundwater flow and nitrogen transport. In addition to data quantity, data quality is also important for improving the understanding. It is expected that the conclusions of this study on the relation between the load estimates and the flow paths and velocity will still hold when more observations become available.

Acknowledgements The ArcNLET software package, example data, and user's manual are available for free download from http://people.sc.fsu.edu/~mye/ArcNLET/index.html. This research was supported in part by National Natural Science Foundation of China grant 51629901, Collaborative Research Fund of the State Key Laboratory of Water Resources and Hydropower Engineering Science of the Wuhan University, and contract WM956 with the Florida Department Environmental Protection. We thank Dale Majewski, Dianne K. Hughes, and William Griffin for helping collecting data. Any opinions, findings, and conclusions or recommendations expressed in this material are those of the author(s) and do not necessarily reflect the views of the Florida Department of Environmental Protection.

References

Anderson D (2006) A review of nitrogen loading and treatment performance recommendations for onsite wastewater treatment systems (OWTS) in the Wekiva Study Area. Hazen and Sawyer, P.C., London

Aziz CE, Newell CJ, Gonzales JR, Haas P, Clement TP, Sun Y (2000) BIOCHLOR natural attenuation decision support system user's manual version 1.0. U.S. Environmental Protection Agency, Washington, DC

Badruzzaman M, Pinzon J, Oppenheimer J, Jacangelo JG (2012) Source of nutrients impacting surface waters in Florida: a review. J Environ Manag 109:80–92

Beaujouan V, Durand P, Ruiz L, Aurousseau P, Cotteret G (2002) A hydrological model dedicated to topography-based simulation of nitrogen transfer and transformation: rationale and application to the geomorphology-denitrification relationship. Hydrol Process 16(2):493–507

Belanger TV, Price TL, McGinnis DI (2009) The importance of OSTDS contaminant loading to the IRL, final project report for contract 25141 of St. Johns River Water Management District, Melbourne, FL

BMAP (2013) Basin management action plan for the implementation of total maximum daily loads for nutrients and dissolved oxygen by the Florida Department of Environmental Protection in the St. Lucie River and Estuary Basin, developed by the St. Lucie River and Estuary Basin Technical Stakeholders

Bowen JL, Ramstack JM, Mazzilli S, Valiela I (2007) NLOAD: an interactive, web-based modeling tool for nitrogen management in estuaries. Ecol Appl 17:S17–S30

Domenico PA (1987) An analytical model for multidimensional transport of a decaying contaminant species. J Hydrol 91:49–58

Giordano JCP, Brush MJ, Anderson IC (2011) Quantifying annual nitrogen loads to Virginia's coastal lagoons: sources and water quality response. Estuaries Coasts 34:297–309. doi:10.1007/s12237-010-9345-7

Heinen M (2006) Simplified denitrification models: overview and properties. Geoderma 133(3–4):444–463

Herr J, Weintraub L, Chen CW (2001) User's guide to WARMF, Documentation of graphical user interface

Howarth RW, Anderson D, Cloern J, Elfring C, Hopkinson C, Lapointe B, Malone T, Marcus N, McGlathery K, Sharpley A, Walker D (2000) Nutrient pollution of coastal rivers, bays, and seas. Issues Ecol 7:1–15

Howath RW (2008) Coastal nitrogen pollution: a review of sources and trends globally and regionally. Harmful Algea 8:14–20

Kinney EL, Valiela I (2011) Nitrogen loading to Great South Bay: land use, sources, retention, and transport from land to bay. J Coastal Res 27(4):672–686

Kroeger KD, Cole ML, York JK, Valiela I (2006) Nitrogen loads to estuaries from waste water plumes: modeling and isotopic approaches. Ground Water 44(2):188–200

Lapointe BE, O'Connell JD, Garrett GS (1990) Nutrient coupling between on-site sewage disposal systems, groundwaters, and nearshore surface waters of the Florida keys. Biogeochemistry 10(3):289–307

Lapointe BE, Herren LW, Bedford BJ (2012) Effects of Hurricanes, Land Use, and Water Management on Nutrient and Microbial Pollution: St. Lucie Estuary, Southeast Florida. J. Coast. Res. 28(6):1345–1361

Lin YF, Wang JH, Valocchi AJ (2009) PRO-GRADE: GIS toolkits for groundwater recharge and discharge estimation. Ground Water 47(1):122–128

MACTEC (2007) Phase I report Wekiva River Basin nitrate sourcing study. Newberry, FL: MACTEC. http://www.dep.state.fl.us/water/wekiva/docs/phase-i-final-report.pdf

Maggi F, Gu C, Riley WJ, Hornberger GM, Venterea RT, Xu T, Spycher N, Steefel C, Miller NL, Oldenburg CM (2008) A mechanistic treatment of the dominant soil nitrogen cycling processes: model development, testing, and application. J Geophys Res. doi:10.1029/2007JG000578

McCray JE, Kirkland SL, Siegrist RL, Thyne GD (2005) Model parameters for simulating fate and transport of on-site wastewater nutrients. Ground Water 43(4):628–639

McCray JE, Geza M, Murray KF, Poeter EP, Morgan D (2009) Modeling onsite wastewater systems at the watershed scale: a user's guide. Water Environ Res Found, Alexandria

Meile C, Porubsky WP, Walker RL, Payne K (2010) Natural attenuation of nitrogen loading from septic effluents: spatial and environmental controls. Water Res 4(4):1399–1408

Newell CJ, McLeod KR, Gonzales JR (1996) BIOSCREEN, natural attenuation decision support system, user's manual, version 1.3. EPA/600/R-96-087. U.S. Environmental Protection Agency, Washington, DC

Porubsky WP, Joye SB, Moore WS, Tuncay K, Meile C (2011) Field measurements and modeling of groundwater flow and biogeochemistry at Moses Hammock, a backbarrier island on the Georgia coast. Biogeochemistry 104(1–3):69–90. doi:10.1007/s10533-010-9484-8

Rios JF (2010) A GIS-based model for estimating nitrate fate and transport from septic systems in surficial aquifers. Master thesis, Florida State University, Tallahassee, FL

Rios JF, Ye M, Wang L (2011a) uWATER-PA: ubiquitous WebGIS analysis toolkit for extensive resources—pumping assessment. Ground Water 49(6):776–780. doi:10.1111/j.1745-6584.2011.00872.x

Rios JF, Ye M, Wang L, Lee P (2011b) ArcNLET: an ArcGIS-based nitrate load estimation toolkit, user's manual, Florida State University, Tallahassee, FL. http://people.sc.fsu.edu/~mye/ArcNLET/users_manual.pdf

Rios JF, Ye M, Wang L, Lee PZ, Davis H, Hicks RW (2013a) ArcNLET: a GIS-based software to simulate groundwater nitrate load from septic systems to surface water bodies. Comput Geosci 52:108–116. doi:10.1016/j.cageo.2012.10.003

Rios JF, Ye M, Sun H (2013b) AcrNLET 2.0: new Arc-NLET function of Monte Carlo simulation for uncertainty quantification, user's manual, Department of Scientific Computing, Florida State University, Tallahassee, FL. http://people.sc.fsu.edu/~mye/ArcNLET/ArcNLET2.0_manual.pdf. Accessed Apr 2015

Robertson WD, Cherry JA, Sudicky EA (1991) Ground-water contamination from two small septic systems on sand aquifers. Ground Water 29:82–92. doi:10.1111/j.1745-6584.1991.tb00500.x

Roeder E (2008) Revised estimates of nitrogen inputs and nitrogen loads in the Wekiva Study Area. http://www.dep.state.fl.us/water/wekiva/docs/doh-wekiva-estimate-final2008.pdf

Romshoo SA, Muslim M (2011) Geospatial modeling for assessing the nutrient load of a Himalayan lake. Environ Earth Sci. doi:10.1007/s12665-011-0944-9

Sigua GC, Tweedale WA (2003) Watershed scale assessment of nitrogen and phosphorus loadings in the Indian River Lagoon basin, Florida. J Environ Manag 67:363–372

Sigua GC, Steward JS, Tweedale WA (2000) Water quality monitoring and biological integrity assessment in the Indian River Lagoon, Florida: status, trends, and loadings (1988–1994). J Environ Manag 25(2):199–209

Spiteri C, Slomp CP, Tuncay K, Meile C (2008) Modeling biogeochemical processes in subterranean estuaries: effect of flow dynamics and redox conditions on submarine groundwater discharge of nutrients. Water Resour Res 44:W02430. doi:10.1029/2007WR006071

U.S. EPA (2013) A model program for onsite management in the Chesapeake Bay Watershed, Office of Wastewater Management, U.S. Environmental Protection Agency

Ursin EL, Roeder E (2008) An assessment of nitrogen contribution from onsite wastewater treatment systems (OWTS) in the Wekiva study area of central Florida. In: National onsite wastewater recycling association (NOWRA) Nitrogen Symposium. Florida Department of Health, Tallahassee

U.S. NRC (2000) Clean coastal waters: understanding and reducing the effects of nutrient pollution. National Research Council, National Academies Press, Washington, DC

U.S. NRC (2010) Understanding the changing planet strategic directions for the geographical sciences. National Academies Press, Washington, DC

Vadeboncoeur MA, Hamburg SP, Pryor D (2010) Modeled nitrogen loading to Narragansett Bay: 1850 to 2015. Estuaries Coasts 33:1113–1127. doi:10.1007/s12237-010-9320-3

Valiela I, Costa JE (1988) Eutrophication of Buttermilk Bay, a Cape Cod coastal embayment: concentrations of nutrients and watershed nutrient budgets. Environ Manag 12(4):539–553

Valiela I, Collins G, Kremer J, Lajtha K, Geist M, Seely B, Brawley J, Sham CH (1997) Nitrogen loading from coastal watersheds to receiving estuaries: new method and application. Ecol Appl 7(2):358–380

Valiela I, Geist M, McClelland J, Tomasky G (2000) Nitrogen loading from watersheds to estuaries: verification of the Waquoit Bay nitrogen loading model. Biogeochemistry 49(3):277–293

Wang L, Ye M, Rios JF, Lee P (2011) ArcNLET: an ArcGIS-based nitrate load estimation toolkit, application manual, Florida State University, Tallahassee, FL. http://people.sc.fsu.edu/~mye/ArcNLET/application_manual.pdf

Wang L, Ye M, Rios JF, Fernandes R, Lee PZ, Hicks RW (2013a) Estimation of nitrate load from septic systems to surface water bodies using an ArcGIS-based software. Environ Earth Sci. doi:10.1007/s12665-013-2283-5

Wang L, Ye M, Lee PZ, Hicks RW (2013b) Support of sustainable management of nitrogen contamination due to septic systems using numerical modeling methods. Environ Syst Decis 33:237–250. doi:10.1007/s10669-013-9445-6

West MR, Kueper BH, Ungs MJ (2007) On the use and error of approximation in the Domenico (1987) solution. Ground Water 45(2):126–135

Yang YCE, Lin YFF (2011) A new GIScience application for visualized natural resources management and decision support. Trans GIS 15(s1):109–124

Yang G, Best EPH, Whiteaker T, Teklitz A, Yeghiazarian L (2014) A screening-level modeling approach to estimate nitrogen loading and standard exceedance risk, with application to the Tippecanoe River watershed, Indiana. J Environ Manag 135:1–10

Ye M, Sun H (2013) Estimation of nitrogen load from removed septic systems to surface water bodies in the city of Port St. Lucie, the city of Stuart, and Martin County, Department of Scientific Computing, Florida State University, Tallahassee, FL. http://people.scs.fsu.edu/~mingye/FDEP/PortStLucieModeling.pdf, accessed as of April, 2015

Ye M, Rios JF, Shi L (2014) A new ArcGIS-based software of uncertainty analysis for nitrate load estimation. Groundwater 52(5):649–650. doi:10.1111/gwat.12228