Estimation of Nitrogen Load from Septic Systems to Surface Waterbodies in Welaka Town, Putnam County, FL.

Manuscript Completed: August, 2014

Prepared by

Mohammad Sayemuzzaman and Ming Ye

Department of Scientific Computing, Florida State University, Tallahassee, FL 32306

Rick Hicks, FDEP Contract Manager

Prepared for

Florida Department of Environmental Protection

Tallahassee, FL

EXECUTIVE SUMMARY

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), was used in this study for Welaka town in Putnam County, FL. The purpose of this project is to estimate the reduction of nitrogen load from **125** removed septic systems to the St. Johns River and other surface waterbodies in the town. 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, including advection, dispersion, and denitrification. After preparing model input files 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 Darcy velocity, 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 were established using data downloaded from public-domain websites and provided by colleagues from FDEP.

Due to the lack of observations of hydraulic head and nitrogen concentration in the modeling area, model calibration was not conducted in this study. Site-specific data of DEM, waterbodies, septic locations, hydraulic conductivity, and porosity were used in the ArcNLET modeling. The ArcGIS layer of surface waterbodies was updated to include small waterbodies that are missing in the layer but reflected in the DEM layer. Literature-based parameter values were used for smoothing factor in the flow modeling, and for longitudinal dispersivity (α_L), transverse horizontal dispersivity (α_T), decay coefficient (k), inflow mass to groundwater (M_{in}), and source plane concentration (C_0) in the transport modeling. Due to uncertainty in the literature-based parameter values of dispersivities (α_L and α_T) and denitrification coefficient (k), four parameter sets of the parameters were considered. The parameter values are listed in Table ES-1 together with the literature-based values of the other three parameters. Cases 1 and 2 considered two sets of α_L and α_T values, and Cases 3 and 4 used a different k value. The justification of using these values is given in the text.

Table ES-1. Literature-based values of ArcNLET model parameters.

Parameter	Case 1	Case 2	Case 3	Case 4
Smoothing factor	60	60	60	60
M_{in} (g/d)	22.4	22.4	22.4	22.4
C_0 (mg/L)	40	40	40	40
α_L (m)	2.134	6	2.134	6
α_T (m)	0.0549	2.5	0.0549	2.5
k (d^{-1})	0.011	0.011	0.001	0.001

Figure ES-1 plots the simulated flow paths to the surface waterbodies. Waterbody 31 has the largest number (52) of contributing septic systems. The St. Johns River (waterbdy 57) has the second largest number (36) of contributing septic systems. Waterbody 58 has the third largest number (13) of contributing septic systems. It should be noted that only 29% of the septic systems contribute load to the St. Johns River, which indicates importance of considering spatial variability in the load reduction estimation so that overestimation can be avoided.

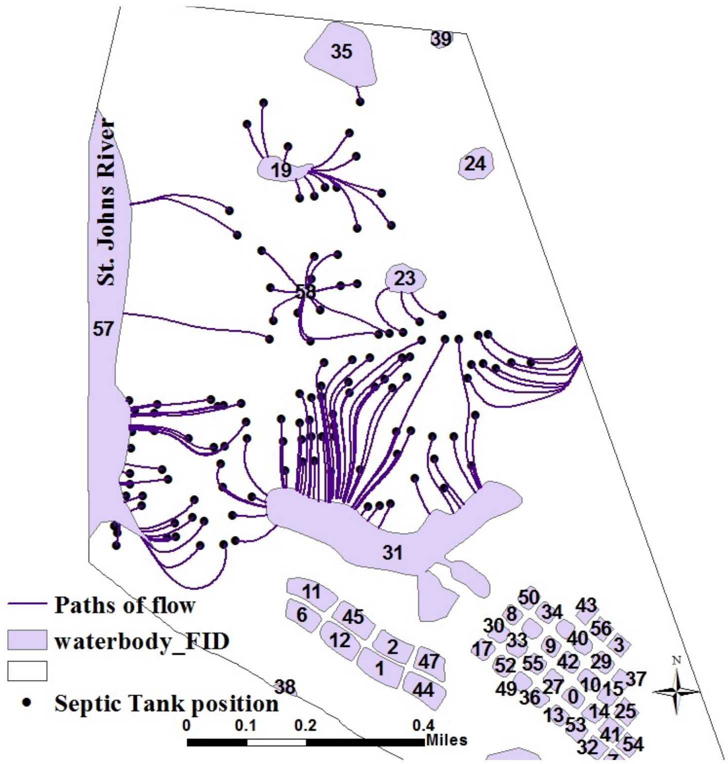

Figure ES-1.Waterbodies FID, Septic tank locations and the particle flow paths

Table ES-2 lists the ArcNLET-estimated total loads (in the units of g/d and lb/yr) for the 125 removed septic systems, load (g/d) per septic system, and the reduction ratios (%). The reduction ratio is the amount of denitrified nitrogen (i.e., the difference between the load to groundwater and the load to surface waterbodies) divided by the load to groundwater (i.e., the inflow mass, M_{in}). This table shows that the load estimates depends more on the denitrification coefficient than on the dispersivities. For example, Cases 1 and 3 have the same dispersivities (α_L = 2.134 m, α_T = 0.0549 m) but different denitrification (0.011 d^{-1} for Case 1 and 0.001 d^{-1} for Case 3). The two cases have dramatically different load estimates, with the estimate of Case 3 being 2.83 times as large as that of Case 1. The ratio is 2.77 between the estimates of Cases 4 and 2, when the dispersivities change to another set (α_L = 6 m and α_T = 2.5 m). Similarly, the reduction ratio changes from ~80% to ~50%, when the denitrification coefficient decreases from 0.011 d^{-1} to 0.001 d^{-1}. These reduction ratios are comparable with those reported in literature, as discussed in Ye and Sun (2013).

Figure ES-2 plots the estimated nitrogen loads to the six waterbodies (with FIDs of 57, 31, 19, 58, 35 and 23) that receive all the nitrogen loads from the septic systems. For all the four modeling cases, the St. Johns River (FID 57) receives the largest amount of load, although the river has the second largest number of contributing septic systems. Although waterbody 31 has the largest number of contributing septic systems, its load estimate is smaller than that of the St. Johns River, especially when the denitrification coefficient is small. This is not surprising, because the load estimates depend not only on the number of contributing septic systems but also on flow path and flow velocity. It was shown in Ye and Sun (2013) that longer flow path and small flow velocity always lead to large amount of denitrification and thus smaller load estimates. In this study, the length of flow paths is the determining factor, because the flow paths to waterbody 31 are longer than those to waterbody 57, as shown in Figure ES-1.

Table ES-2. Simulated nitrogen loads to surface waterbodies and nitrogen reduction ratios for 125 septic systems in Welaka town.

	Total load (g/d)	Total load (lb/yr)	Load per septic systems (g/d)	Reduction ratio (%)
Case 1	392	316	3.14	83
Case 2	404	325	3.23	82
Case 3	1108	892	8.86	52
Case 4	1120	901	8.96	51

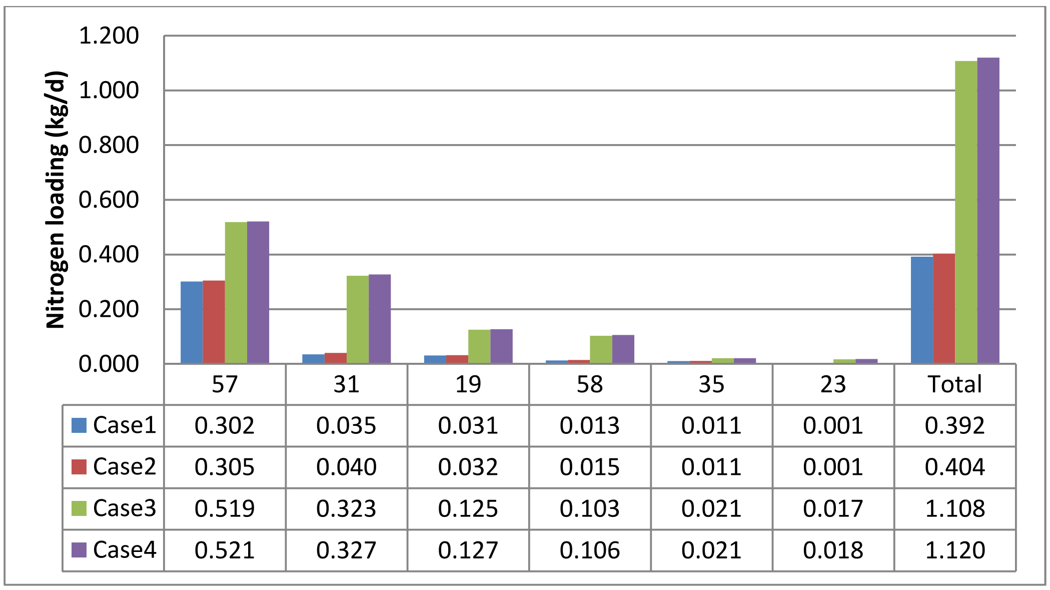

Figure ES-2. Nitrogen loading (kg/d) to the individual waterbodies for the four modeling cases.

The ArcNLET modeling of this study leads to the following major conclusions:

(1) Data and information needed to establish ArcNLET models for groundwater modeling and nitrogen load estimation are readily available in the Welaka town. The data are available in either the public domain (e.g., USGS websites) or FDEP database. Site-specific data include DEM, waterbodies, septic locations, hydraulic conductivity, and porosity. The values of smoothing factor, dispersivity, decay coefficient, inflow mass to groundwater, and source plane concentration are obtained from literature.

(2) Among the 125 removed septic systems, only 36 septic systems contribute nitrogen load to the St. Johns River. The rest of septic systems contribute nitrogen load to waterbodies that are not connected to the St. Johns River. As a result, not all the removed septic systems contributed to nitrogen load reduction in the TMDL practice. This suggests importance of considering spatial variability in environmental management of nitrogen contamination.

(3) For all the four modeling cases considered in this study, the load estimate to the St. Johns River (FID 57) is the largest, although the number of contributing septic systems to the river is not the largest. Although Waterbody 31 has the largest number of contributing septic systems, its load estimate is the second largest, because the flow paths associated with the waterbody are longer than those with the river.

(4) The denitrification coefficient is the most influential parameter to the load estimate. More effort should be spent to determine the appropriate value of the parameter for more accurate estimation of load reduction.

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1. INTRODUCTION

This report summarizes the modeling effort of the Florida State University (FSU) to support management of nutrient pollution of the Florida Department of Environmental Protection (FDEP) for the Welaka town. Location of the Welaka town and the septic system locations (provided by Richard Hicks, FDEP) are shown in Figure 1-1. The ArcGIS-Based Nitrate Load Estimation Toolkit (ArcNLET) (Rios et al. 2013a; Wang et al., 2013), developed by FSU, is used in this study to simulate groundwater flow and nitrate transport from septic systems in surficial aquifers. With the assumption that ammonium transports in the same way as nitrate, ArcNLET can be used to simulate nitrogen fate and transport. ArcNLET provides the following outputs:

(1) Approximation of water table shape in the modeling domain,
(2) Magnitude and direction of Darcy velocity at raster cells of the modeling domain,
(3) Nitrogen plumes from individual septic systems to nearby water bodies, and
(4) Nitrogen load estimates to the individual water bodies.

ArcNLET modeling requires the following ArcGIS raster and polygon files that are available in the public domain:

(1) Digital elevation model (DEM) of topography that is available at the USGS National Map Viewer and Download Framework (http://nationalmap.gov/viewer.html). The DEM data is smoothed to generate the shape of water table, based on the assumption that water table is a subdued replica of topography. The smoothed DEM is used to evaluate groundwater flow magnitude and direction by using the Darcy's law.
(2) Water body locations that are available from USGS National Hydrography Database (http://nhd.usgs.gov/). Flow paths evaluated from the Darcy velocity vector from septic systems are terminated at the water bodies.
(3) Locations of septic systems that are available in the database of Florida Department of Health. For this study, the locations are provided by Richard W. Hicks at FDEP.
(4) Hydraulic conductivity and porosity of the modeling domain are available from the Soil Survey Geographic Database (SSURGO).

The data above are site specific. ArcNLET modeling also involves the following parameters of groundwater flow and solute transport that are obtained from model calibration and/or literature:

(1) Dispersivities in the longitudinal and horizontal transverse directions,
(2) Inflow nitrogen mass to groundwater,
(3) Nitrogen concentration in the effluent entering groundwater, and
(4) Decay coefficient of denitrification.

Site-specific values of these parameters can be obtained by model calibration, in which the literature-based parameter values are adjusted to match simulations of hydraulic head and nitrogen concentration to corresponding field observations. However, due to the lack of field observations in this study, the model calibration was not performed. Instead, the ArcNLET simulation of this study is conducted using the parameter values obtained from literature and our pervious modeling experience, e.g., the Julington Creek neighborhood in the City of Jacksonville. Justification of using the parameter values is given below.

In the remainder of this report, the conceptual model of groundwater flow and nitrogen transport used in ArcNLET and its computational implementation are briefly described in Section 2. Sections 3 and 4 present the modeling practice and results for the Welaka town. The summary and conclusions of this study are discussed in Section 5.

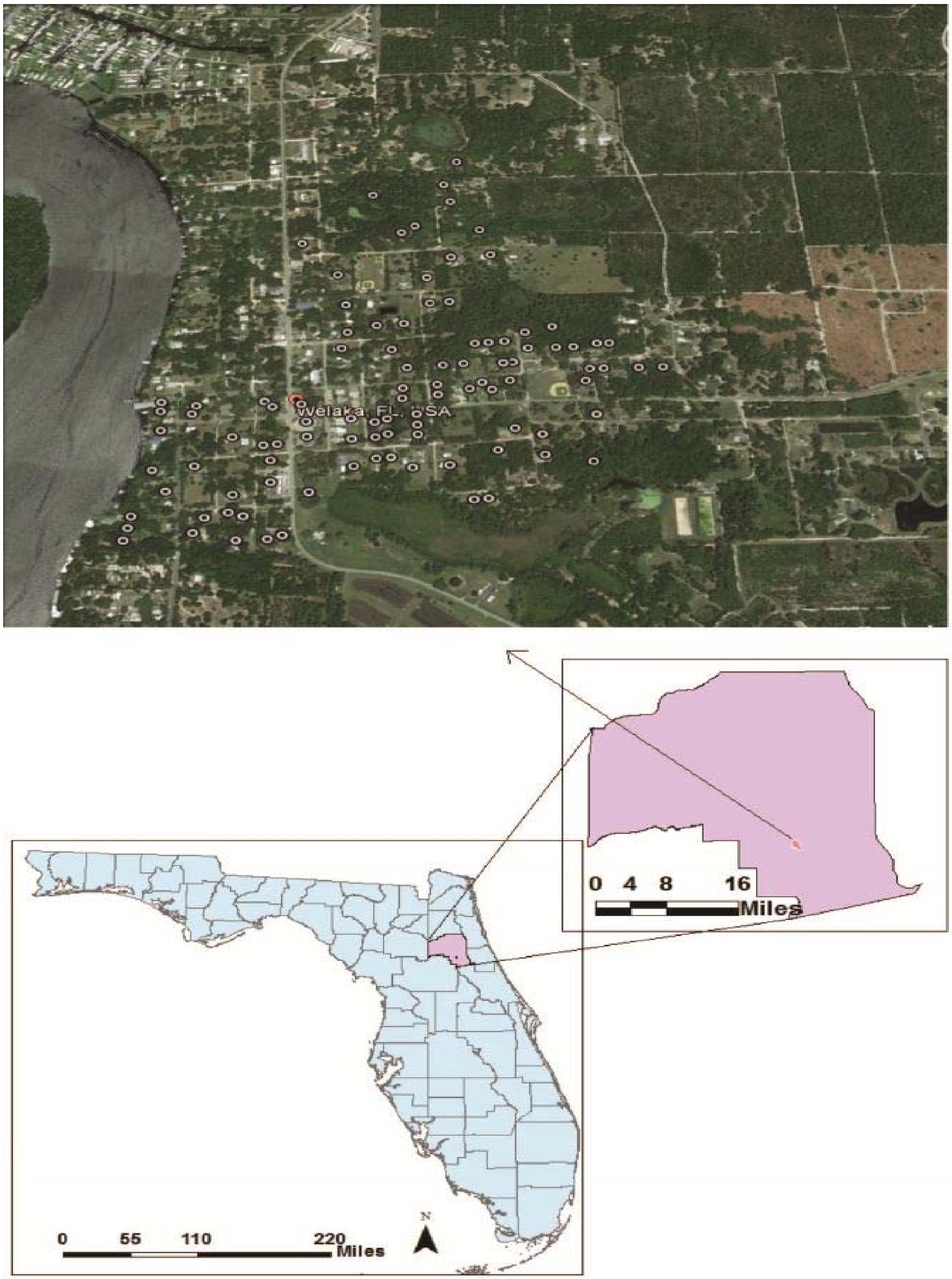


Figure 1-1. Location of Welaka town and septic system locations (dot circular in the upper map) in Putnam County.

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 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

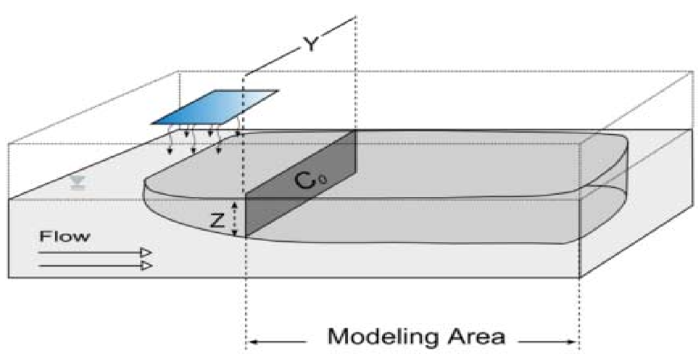


Figure 2-1. Conceptual model of nitrate transport in groundwater adapted from Aziz et al. (2000). The unsaturated zone is bounded by the rectangular box delineated by the dotted lines; the groundwater zone is bounded by the box delineated by the solid lines

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

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

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

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).

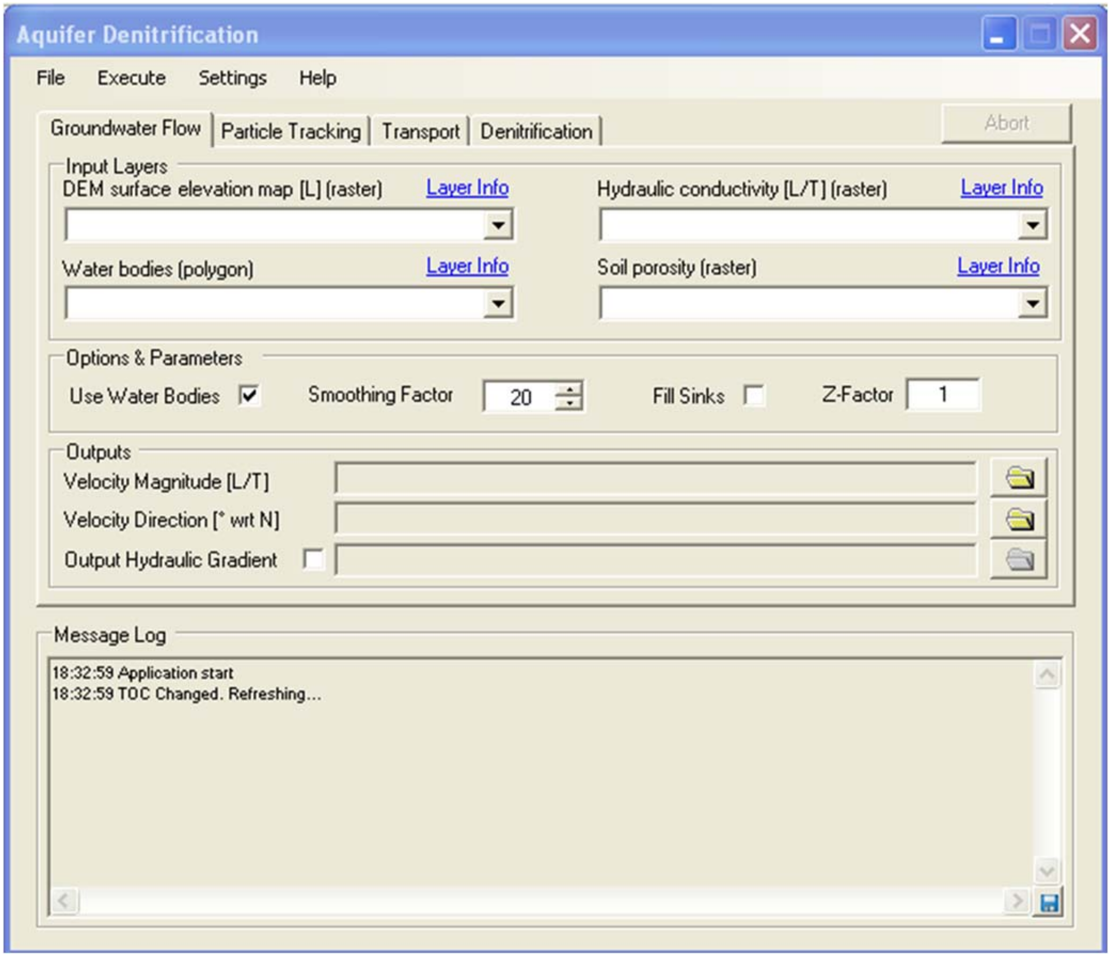


Figure 2-2. Main Graphic User Interface (GUI) of ArcNLET with four modules of Groundwater Flow, Particle Tracking, Transport, and Denitrification.

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.

3. ARCNLET MODELING FOR WELAKA TOWN

3.1. Modeling Area and ArcGIS Input Data

Welaka town is located in the Putnam County on the shore of the St. Johns River (Figure 3-1). Welaka town is situated 65 miles (104.6 km) southwest to Julington creek, where model calibration for ArcNLET parameters was conducted. Figure 3-1 shows the modeling area (in the shaded polygon) chosen to cover the removed septic systems and potential waterbodies to which the septic effluent may flow through the surficial aquifer. The modeling area covers all the septic tank locations and includes the waterbodies in the southeast corner that may potentially receive septic effluent through surficial aquifer.

Figure 3-1. Locations of removed septic systems (dot circles) and modeling area delineated in the shaded polygon.

Figure 3-2 shows the locations of the surface waterbodies downloaded from the NHD website (http://nhd.usgs.gov/). The swamps and marshes were merged into waterbodies so that they also receive nitrogen load from the septic systems. The merge is considered to be reasonable, because the swamps and marshes are of low elevation and should be discharge areas of groundwater flow.

Figure 3-2. Locations of waterbodies, swamps, and marshes highlighted in blue.

Figure 3-3 plots the DEM of the modeling area downloaded from the USGS National Map Viewer and Download Framework (http://nationalmap.gov/viewer.html). The DEM resolution is 3m × 3m, and it can capture small waterbodies, e.g., the pond marked by the red circle in Figure 3-3. The pond is shown in Google Earth (Figure 3-4) but not reflected in the ArcGIS layer of waterbodies (Figure 3-2). Therefore, the ArcGIS layer of waterbodies was updated manually accordingly to include the small pond as a polygon with FID-58 as shown in Figure 3-5. Similarly, the shape of the St. Johns River in the ArcGIS layer of waterbodies is also revised to include the tip shown in the red rectangular in Figure 3-3 of the DEM plot but missing in Figure 3-2 of the waterbodies layer.

After the update of waterbodies, three septic systems become located in waterbodies. Their locations were adjusted by examining the corresponding house location on Google Earth. For example, the septic system in the red circle shown in Figure 3-3 was located in the pond, but moved to the north, because the house using the septic system was located north to the pond. Therefore, Figure 3-3 shows the updated septic system locations.

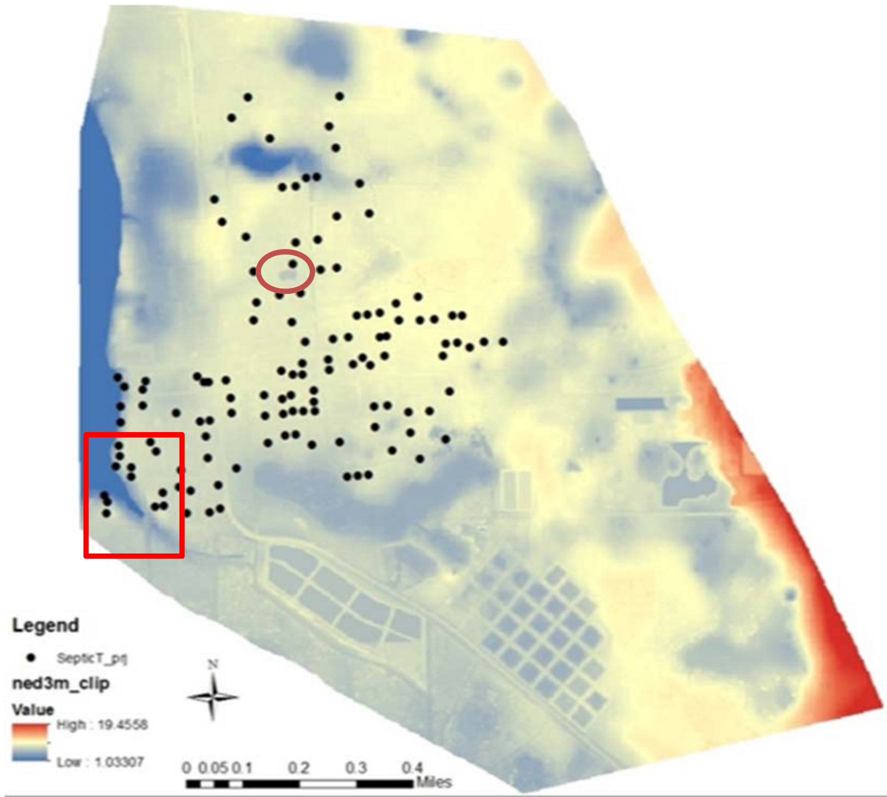


Figure 3–3. Plot of DEM maps and septic system locations. The red circle and rectangular highlight two areas where the ArcGIS layer of waterbodies (Figure 3-2) is updated to be consistent with the DEM data.

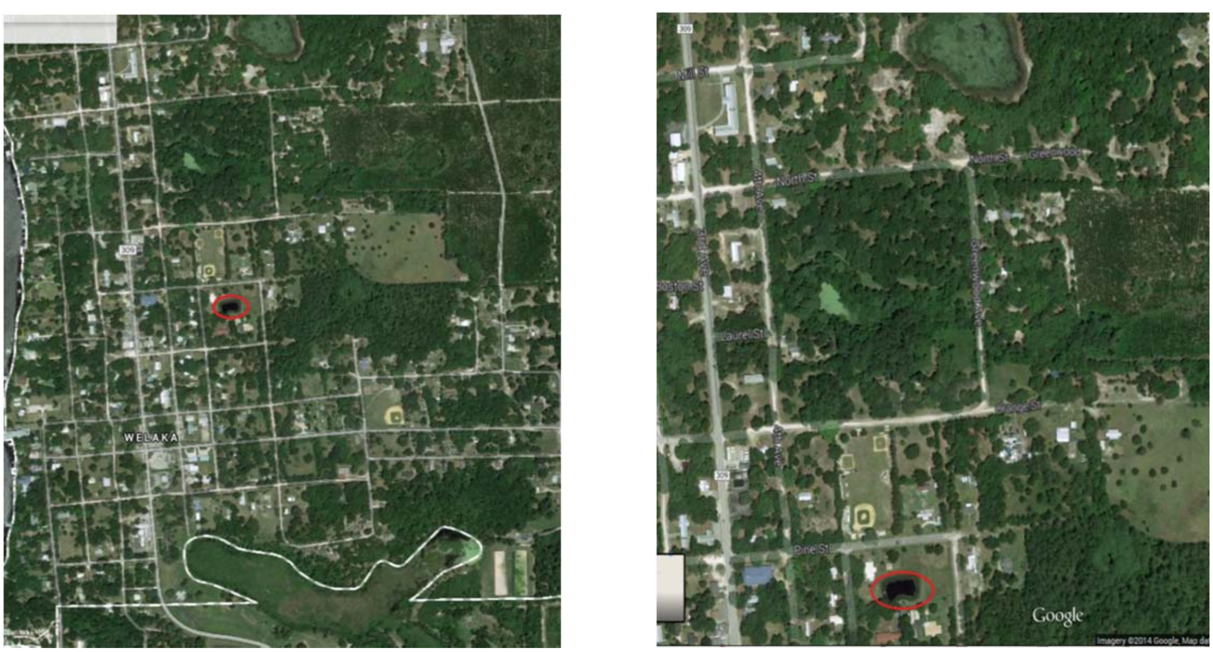


Figure 3–4. Google earth picture shows a small pond in red circle that is missing in the ArcGIS layer of waterbodies.

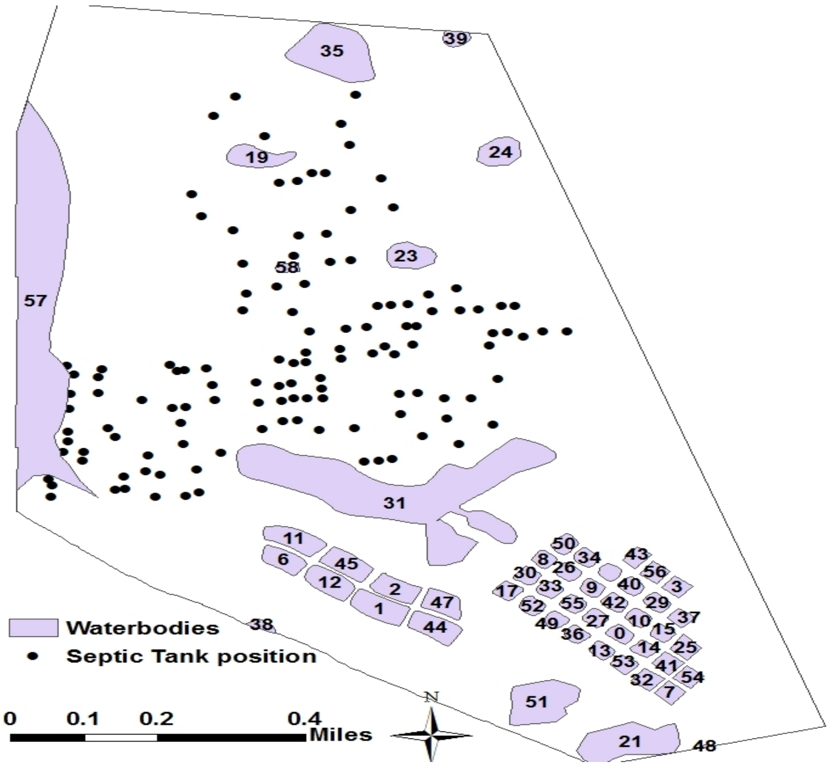


Figure 3–5. Updated waterbodies map based on DEM and Google Earth. The waterbodies are labeled by their FIDs, and the added pond has the new FID of 58.

3.2. Groundwater Flow Parameters

For the groundwater flow modeling, the hydraulic conductivity values are obtained directly from the Soil Survey Geographic database (SSURGO) database, i.e., the representative values contained in the database. Since the database does not include porosity data, the data was evaluated as $1 - (D_B/D_P)$, where D_B and D_p are bulk density and particle density, respectively; bulk density was obtained directly from the SSURGO database and particle density was assumed to be 2.65 gcm^{-3} that is commonly used in soil physics. The SSURGO database contains data at the horizon levels; the data were aggregated to the component and subsequently unit levels, because the hydraulic conductivity and porosity at the soil units are used in ArcNLET modeling. The aggregation was completed by following the procedure described in Wang et al. (2011). Figures 3-6 and 3-7 plot the heterogeneous hydraulic conductivity and porosity fields, respectively, for the soil units in the modeling area. Figure 3-6 shows that the hydraulic conductivity values are high near waterbodies; the values are assigned to be zero at waterbodies. For porosity, there are two zones with unreasonable high values larger than 0.5. This is attributed to data error in SSURGO. The error is not corrected because these zones do not affect groundwater and transport, as shown in Section 4.

The smoothing factor value needed for the flow modeling was taken as 60, which is used in our previous modeling at the Orange County (Ye, 2014). This value is smaller than the value of 150 used in the Julington Creek modeling. It would be ideal to have head observations for model calibration so that site-specific value of the smoothing factor can be determined.

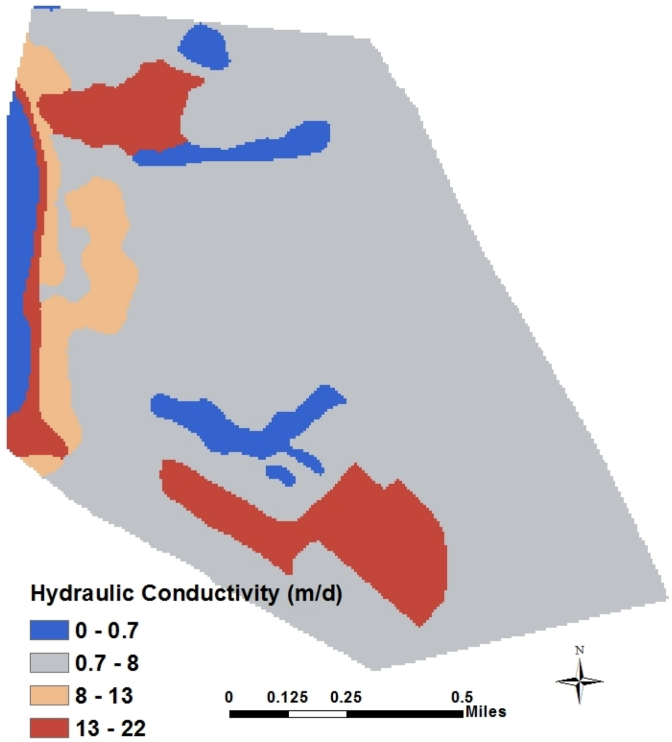


Figure 3–6. Heterogeneous hydraulic conductivity (m/d) of soil units.

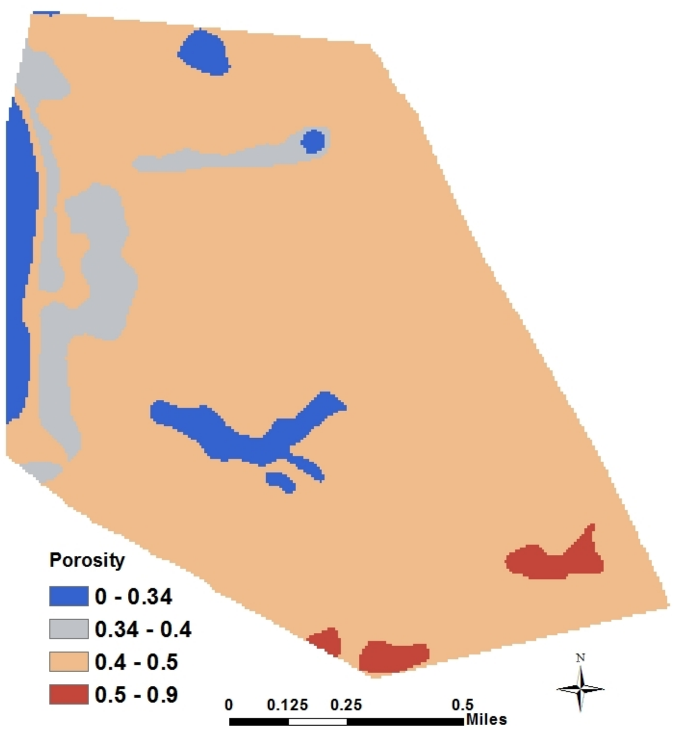


Figure 3–7. Heterogeneous porosity ([-]) of soil units.

3.3. Nitrogen Transport Parameters

For the longitudinal and transverse dispersivities, Davis (2000) used 7 ft. (2.134m) and 0.18 ft. (0.0549m), respectively, for solute transport modeling in the Naval Air Station located in the City of Jacksonville. In our previous modeling for the Julington Creek neighborhood (also located in the City of Jacksonville), the two parameters were calibrated against field observations of nitrogen concentrations, and the calibrated values are 6m and 2.5m for longitudinal and transverse dispersivity, respectively. Considering the scaling effect of dispersivity (i.e., dispersivity increases with spatial scale), the first set of 2.134m and 0.0549m seems more reasonable, because the modeling are of this study is a small. However, due to parametric uncertainty and given that the two sites are close (within the 65miles radius) to our modeling area, we adopted the two sets of parameter values in our study.

The decay coefficient of denitrification is also subject to substantial uncertainty. McCray et al. (2005) reported the range of 0.004 – 2.27 d^{-1}, and Rios et al. (2013b) used the range of 0.004 – 1.08 d^{-1} in the recently developed ArcNLET Monte Carlo simulation package. Our previous model calibration at Julington Creek yielded a value of 0.015 d^{-1}. According to Anderson's (1998), denitrification rate is positively correlated with particulate organic carbon (POC) content; the higher POC content typically suggests a higher denitrification rate. Based on this, we estimate the coefficient value by comparing the organic carbon content at the modeling area with that at Julington Creek.

Figures 3-8 and 3-9 plot the soil organic matter content of the soil units at Julington Creek and the modeling area of this study, respectively, using the data viewer tool available at the soil survey website, http://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm. For a more quantitative comparison, Figure 3-10 plots the histograms of the soil organic matter content for the two sites. The figure shows that the soil organic matter content at the Julington Creek area is larger than that of the Welaka town. This is supported by the spatially weighted average of the organic matter content, which is 2.90% at Julington Creek and 1.58% at Welaka. Assuming a linear relation between the soil organic matter and the denitrification coefficient, the estimated coefficient for Welaka is $1.58/2.9 \times 0.015 = 0.008$ d^{-1}. We took the value of 0.011 d^{-1}, because it was obtained in our previous study at St. Lucie River and Estuary Basin where model calibration was conducted (Ye and Sun, 2013). A significantly smaller value of 0.001 d^{-1} was also used to investigate impacts of the coefficient on the load estimation. A larger load was obtained for the smaller coefficient, as shown in Section 4.

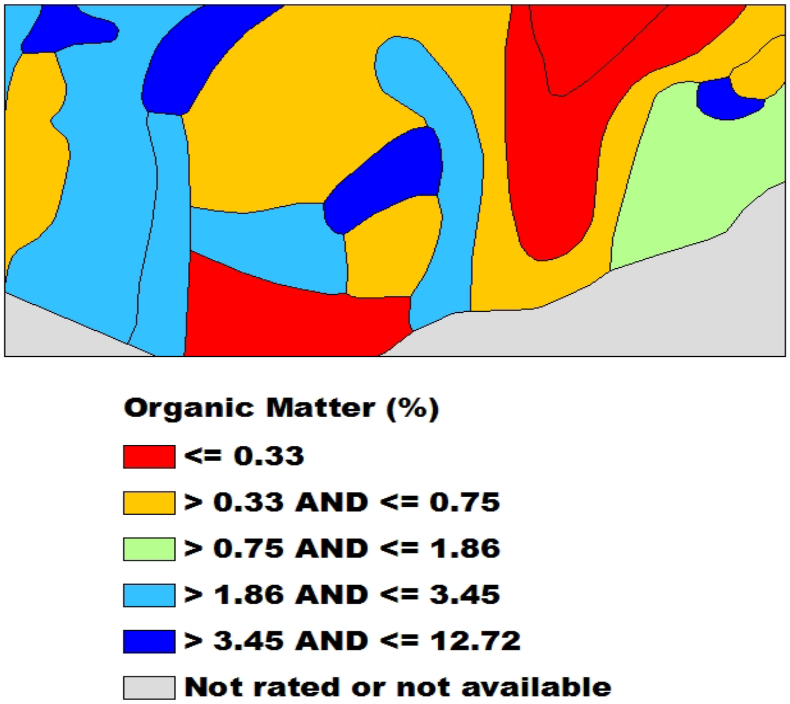


Figure 3–8. Soil organic matter contents at Julington Creek and its vicinity.

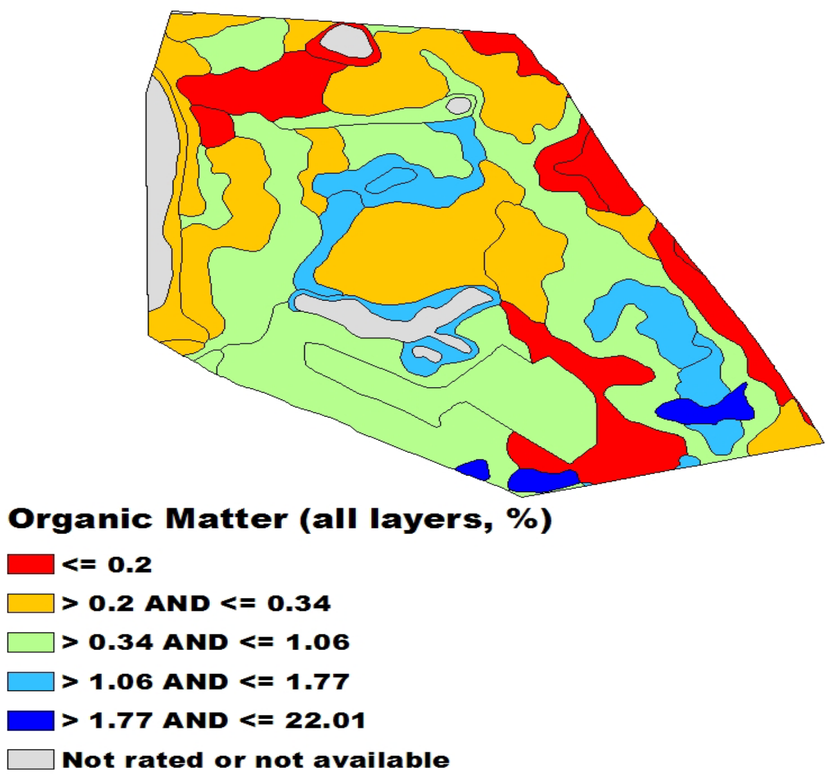
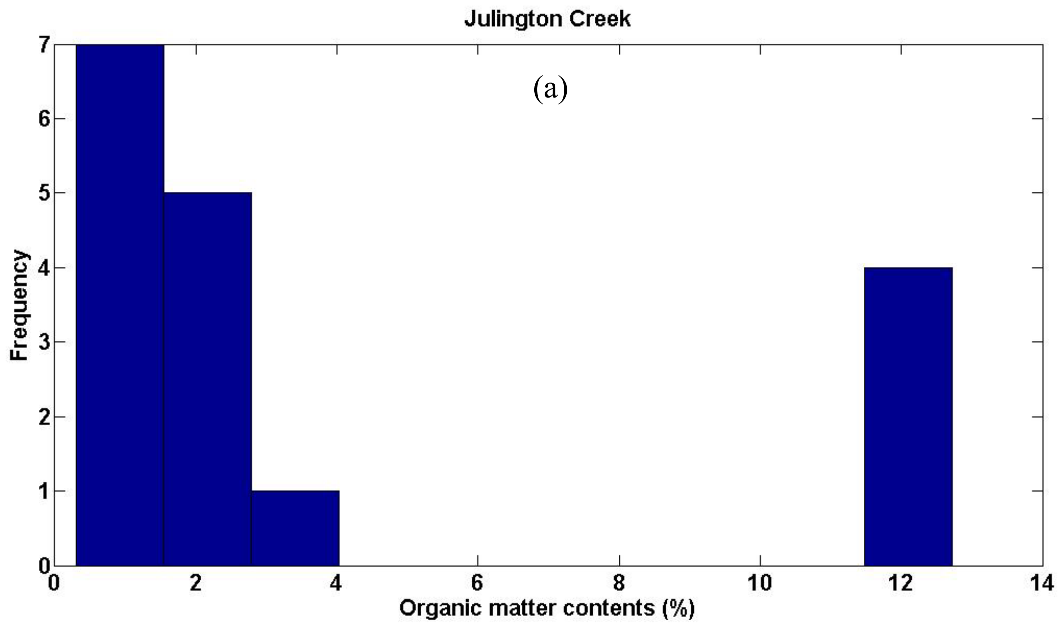


Figure 3–9. Soil organic matter contents in the Welaka town modeling area.



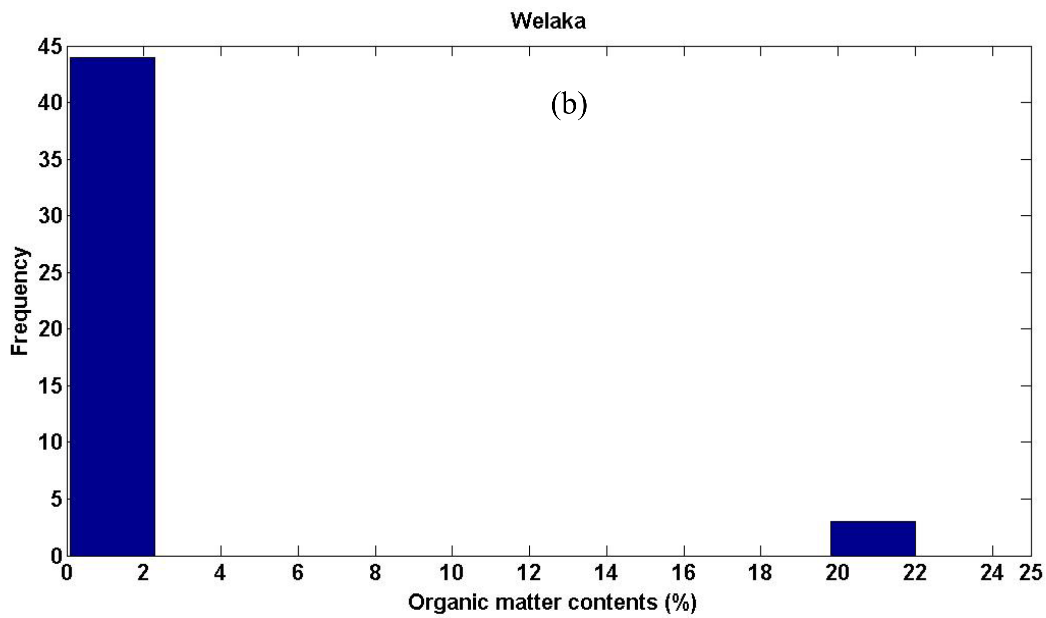


Figure 3-10. Histograms of soil organic matter contents at (a) Julington Creek area and (b) Welaka town

3.4. Nitrogen Load to Groundwater

Nitrogen load from a septic system to groundwater, which is the inflow mass M_{in} (g/d) of equation 5, is an important parameter 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 was used, and the value of inflow mass was approximated as nitrogen release per person per day × people/household × 0.7 (the amount of nitrogen not lost in septic tanks). Instead of using national average, we used the average total nitrogen (TN) load of 4.61kg per person per year, i.e., 12.63g per person per day, provided According by Xueqing Gao at FDEP (personal communication, November 2013) for a ArcNLET modeling project for a site in the Orange County (Ye, 2014). The U.S. Census Bureau website (http://quickfacts.census.gov/qfd/states/12/12095.html) reported that there are 2.53 persons per household in Putnam County. It is a general consensus that about 30% of nitrogen is removed in Septic tank (e.g., MACTEC, 2007). Therefore, the input mass from septic to groundwater is 22.37 g/d, which is slightly larger than 21.7 g/d reported in the Wekiva study (Roeder, 2008). This value is used for all the individual septic systems in the modeling sites.

Another quantity needed to use equation (5) for evaluating the inflow mass is the source plane concentration (C_0). It ranges between 0 mg/L and 80 mg/L (McCray et al., 1995), and the average of 40 mg/L was used in this study.

As a summary, Table 3-1 lists the literature-based values of the three parameters (smoothing factor, M_{in}, and C_0) common and the three parameters (α_L, α_T, and k) specific to the four modeling cases.

Table 3-1. Literature-based values of ArcNLET model parameters.

Parameter	**Case 1**	**Case 2**	**Case 3**	**Case 4**
Smoothing factor	60	60	60	60
M_{in} (g/d)	22.4	22.4	22.4	22.4
C_0 (mg/L)	40	40	40	40
α_L (m)	2.134	6	2.134	6
α_T (m)	0.0549	2.5	0.0549	2.5
k (d^{-1})	0.011	0.011	0.001	0.001

4. RESULTS AND DISCUSSION

4.1. Results of Groundwater Flow Modeling

With the data and information described in Section 3, ArcNLET groundwater flow module and particle tracking module were executed. The flow model provides magnitude and direction of seepage velocity (Darcy velocity divided by porosity) at the individual raster cells in the modeling domain. They are used in the particle tracking module to evaluate flow paths from individual septic systems to receiving waterbodies of particles placed at the septic system locations. Figure 4-1 plots the simulated flow paths. The surface waterbodies in the southeast corner of the modeling area do not have septic effluent, which is intercepted by the waterbody with FID of 31. This waterbody also has the largest number (52) of contributing septic systems. The St. Johns River (waterbdy 57) has the second largest number (36) of contributing septic systems. Waterbody 58 (the added pond) has the third largest number (13) of contributing septic systems. It should be noted that only 29% of the 125 removed septic systems contribute load to the St. Johns River, which indicates importance of considering spatial variability in the load reduction estimation so that overestimation can be avoided.

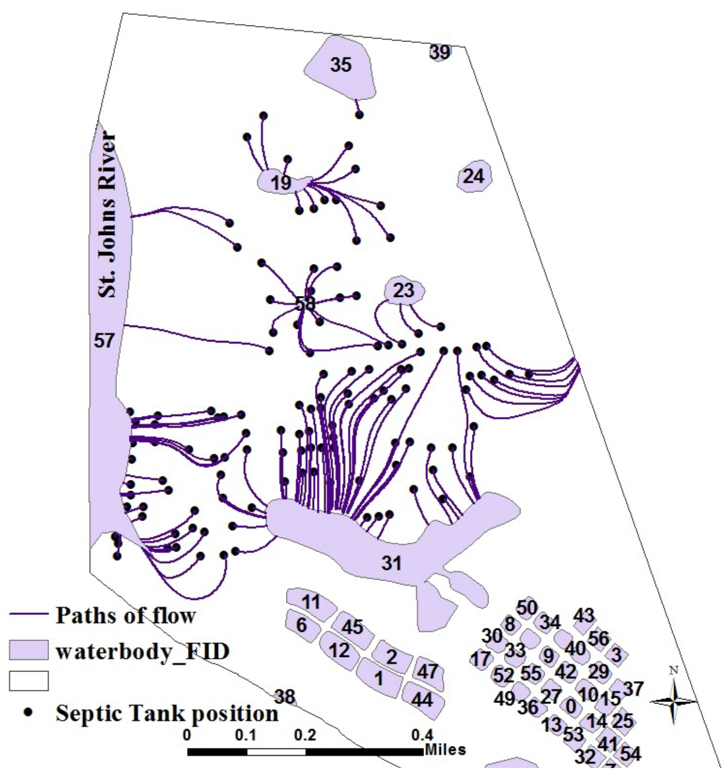


Figure 4–1. Simulated flow paths from septic systems to surface waterbodies.

4.2. Nitrogen Transport Modeling Results

Using the nitrogen transport parameter values discussed in Section 3, ArcNLET transport module was executed. The simulated nitrogen plumes are plotted in Figure 4-2 to Figure 4-5 for the four cases. Table 4-1 – Table 4-4 list the nitrogen load estimates to the receiving waterbodies and reduction ratios for the four cases with different values of longitudinal dispersivity (α_L), horiontal transverse dispersivity (α_T), and denitrification coefficient (k) listed in Table 3-1. Comparing Figure 4-2 of Case 1 with Figure 4-3 of Case 2 shows that, the simulated plumes become fatter when horizontal transverse dispersivity increases. However, since the same denitrification coefficient is used for the two cases, the load estimates of the two cases are similar, as shown in Tables 4-1 and 4-2. When the denitrification coefficient decreass from 0.011 d^{-1} in Cases 1 and 2 to 0.001 d^{-1} in Cases 3 and 4, the load estimates increase significantly. This is reasonable because it was found in our previous sensitivity analysis that denitrification coefficient is the most influential parameter to load estimate (Wang et al., 2012).

For the convenience of comparing the load estimates of the four modeling cases, Table 4-5 lists the total loads (in the units of g/d and lb/yr), load (g/d) per septic system to the surface water bodies, and the reduction ratios (%) of the four parametric cases. This table shows again that the load estimates depends more on the denitrification coefficient than on the dispersivities. For example, Cases 1 and 3 have the same dispersivities (α_L = 2.134 m, α_T = 0.0549 m) but different denitrification (0.011 d^{-1} for Case 1 and 0.001 d^{-1} for Case 3). The two cases have dramatically different load estimates, with the estimate of Case 3 being 2.83 times as large as that of Case 1. The ratio is 2.77 between the estimates of Cases 4 and 2, when the dispersivities change to another set (α_L = 6 m and α_T = 2.5 m). Similarly, the reduction ratio changes from ~80% to ~50%, when the denitrification coefficient decreases from 0.011 d^{-1} to 0.001 d^{-1}. These reduction ratios are comparable with those reported in literature, as discussed in Ye and Sun (2013).

Figure 4-6 plots the estimated nitrogen loads to the six waterbodies (with FIDs of 57, 31, 19, 58, 35 and 23) that receive all the nitrogen loads from the septic systems. For all the four modeling cases, the St. Johns River (FID 57) receives the largest amount of load, although the river has the second largest number of contributing septic systems. Although waterbody 31 has the largest number of contributing septic systems, its load estimate is smaller than that of the St. Johns River, especially when the denitrification coefficient is small. This is not surprising, because the load estimates depend not only on the number of contributing septic systems but also on flow path and flow velocity. It was shown in Ye and Sun (2013) that longer flow path and small flow velocity always lead to large amount of denitrification and thus smaller load estimates. In this study, the length of flow paths is the determining factor, because the flow paths to waterbody 31 are longer than those to waterbody 57, as shown in Figure 4-1.

Table 4-1. Simulated nitrogen loads to surface waterbodies and groundwater (inflow mass) for Case 1. Locations of the waterbodies are shown in Figure 4-2.

Waterbody FID	Load (g/d)	Load (lb/yr)	Inflow Mass (g/d)	Reduction Ratio (%)
57	302	243	805	63
31	35	28	828	96
19	31	25	268	88
58	13	10	291	96
35	11	9	22	52
23	1	1	67	99
Total	392	316	2282	83

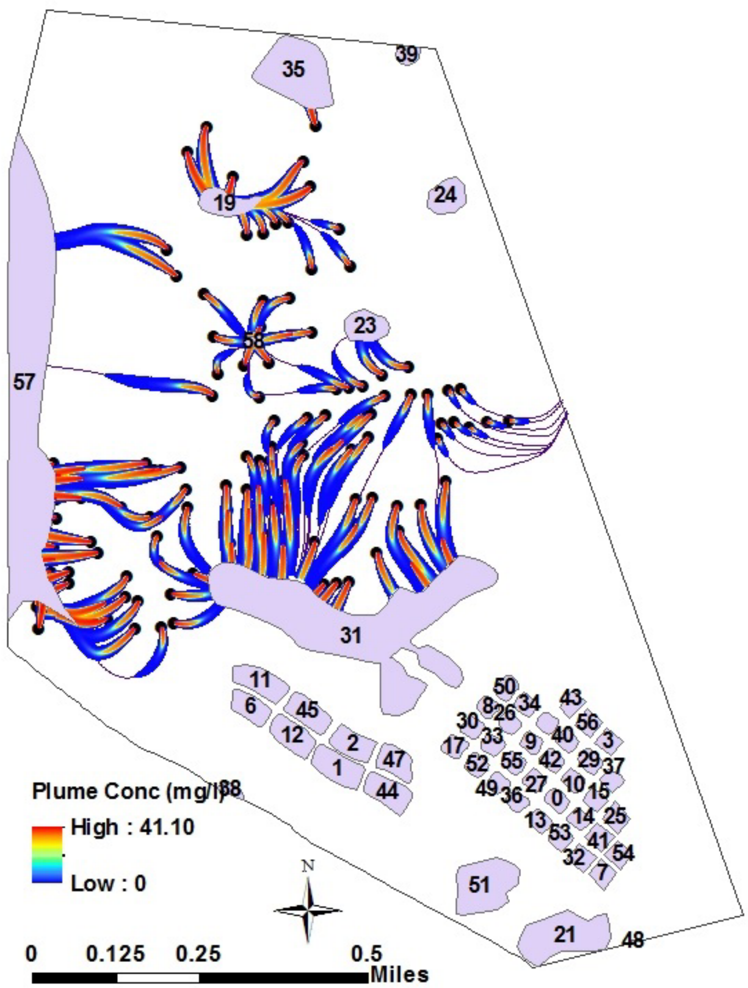


Figure 4-2. Simulated flow paths from septic systems to surface waterbodies. Paramter values specific to Case 1 are α_L = 2.134 m, α_T = 0.0549 m, and k = 0.011 d^{-1}.

Table 4-2. Simulated nitrogen loads to surface waterbodies and groundwater (inflow mass) for Case 2. Locations of the waterbodies are shown in Figure 4-3.

Waterbody FID	Load (g/d)	Load (lb/yr)	Inflow Mass (g/d)	Reduction Ratio (%)
57	305	245	805	62
31	40	32	828	95
19	32	26	268	88
58	15	12	291	95
35	11	9	22	50
23	1	1	67	99
Total	404	325	2282	82

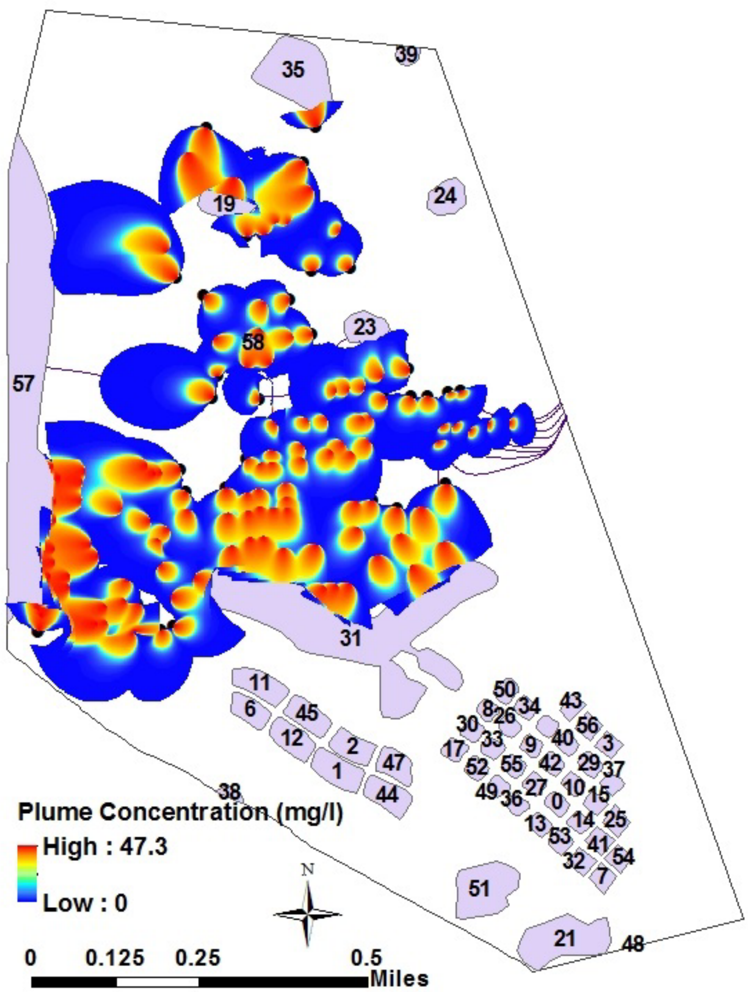


Figure 4-3. Simulated flow paths from septic systems to surface waterbodies. Parameter values specific to Case 2 are $\alpha_L = 6$ m, $\alpha_T = 2.5$ m, and $k = 0.011$ d^{-1}.

Table 4-3. Simulated nitrogen loads to surface waterbodies and groundwater (inflow mass) for Case 3. Locations of the waterbodies are shown in Figure 4-4.

Waterbody FID	Load (g/d)	Load (lb/yr)	Inflow Mass (g/d)	Reduction Ratio (%)
57	519	417	805	36
31	233	188	828	61
19	125	101	268	53
58	103	83	291	65
35	21	17	22	7
23	17	14	67	74
Total	1108	892	2282	51

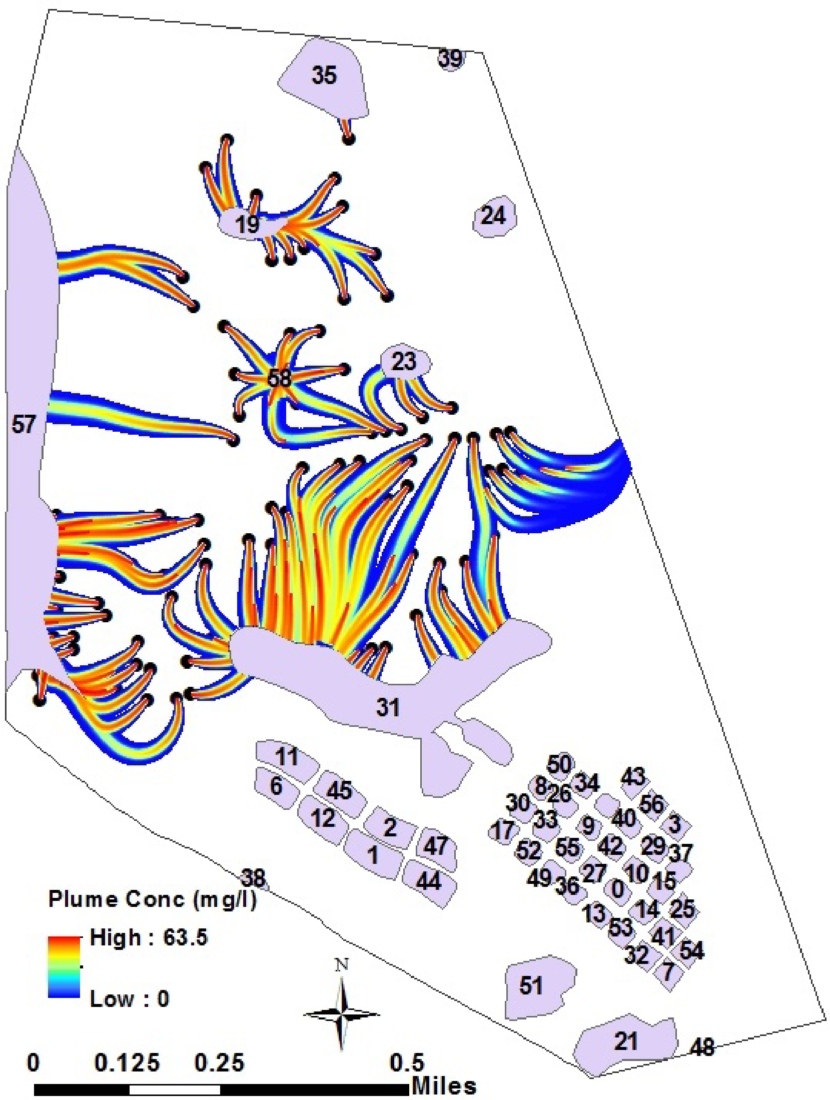


Figure 4-4. Simulated flow paths from septic systems to surface waterbodies. Parameter values specific to Case 3 are $\alpha_L = 2.134$ m, $\alpha_T = 0.0549$ m, and $k = 0.001$ d^{-1}.

Table 4-4. Simulated nitrogen loads to surface waterbodies and groundwater (inflow mass) for Case 4. Locations of the waterbodies are shown in Figure 4-5.

Waterbody FID	Load (g/d)	Load (lb/yr)	Inflow Mass (g/d)	Reduction Ratio (%)
57	521	419	805	35
31	237	191	828	60
19	127	102	268	53
58	106	85	291	64
35	21	17	22	7
23	18	14	67	73
Total	1120	901	2282	51

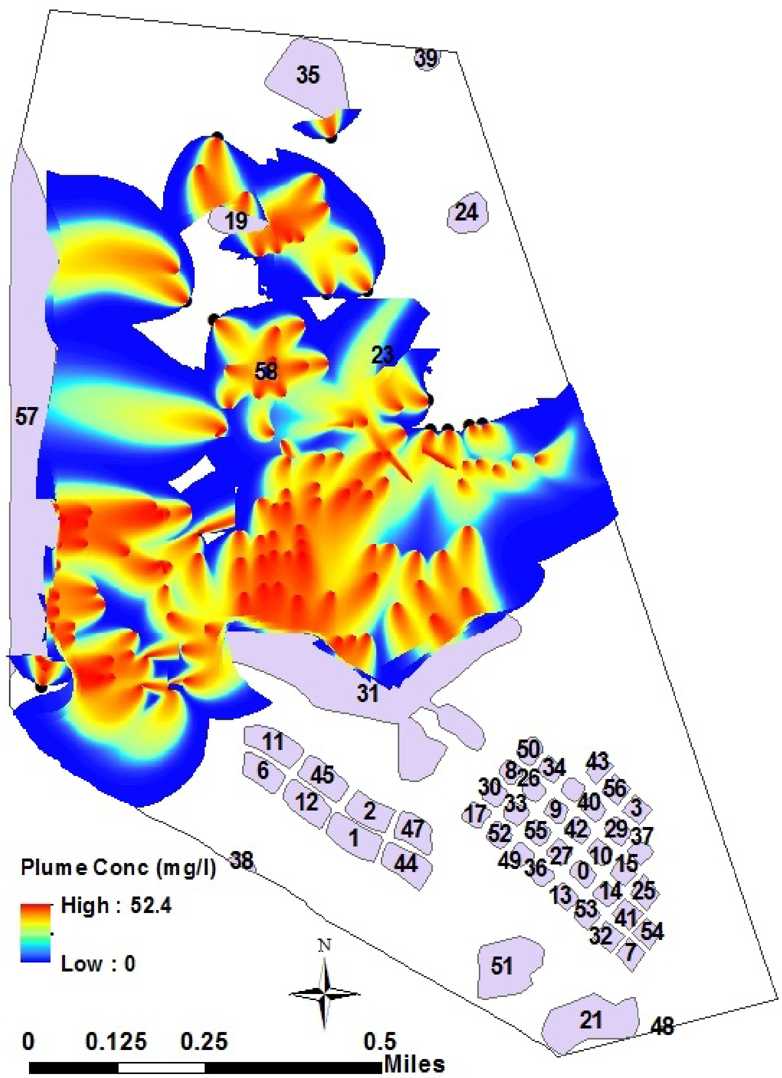


Figure 4-5. Simulated flow paths from septic systems to surface waterbodies. Parameter values specific to Case 4 are $\alpha_L = 6$ m, $\alpha_T = 2.5$ m, and $k = 0.001$ d^{-1}.

Table 4-5. Simulated nitrogen loads to surface waterbodies and nitrogen reduction ratios for the four cases

	Total load (g/d)	**Total load (lb/yr)**	**Load per septic systems (g/d)**	**Reduction ratio (%)**
Case 1	392	316	3.14	83
Case 2	404	325	3.23	82
Case 3	1108	892	8.86	52
Case 4	1120	901	8.96	51

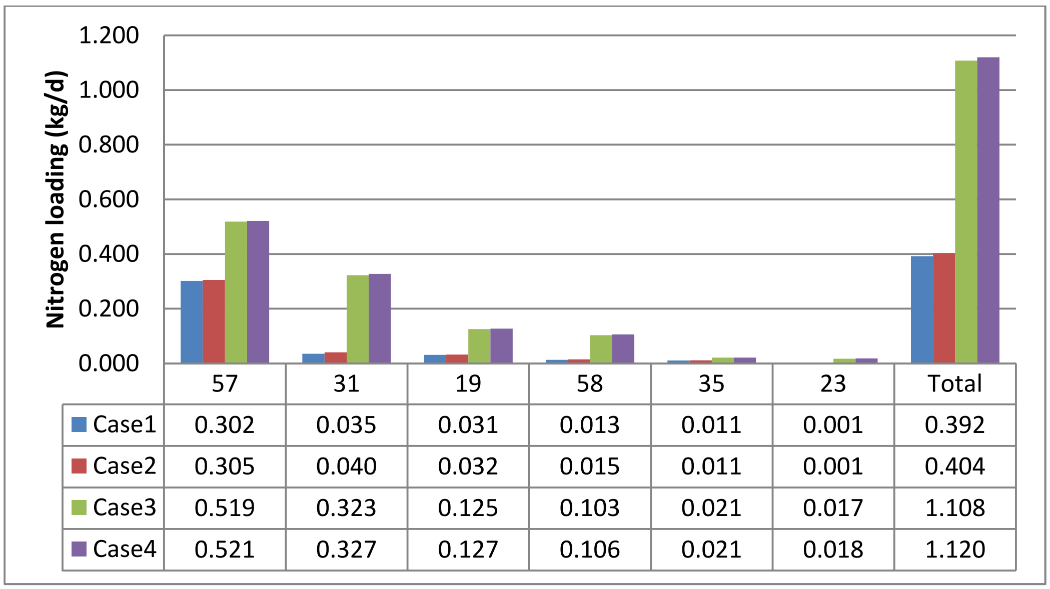


Figure 4-6. Nitrogen loading (kg/d) to the individual waterbodies for the four modeling cases.

5. CONCLUSION

The ArcNLET modeling of this study leads to the following major conclusions:

(5) Data and information needed to establish ArcNLET models for groundwater modeling and nitrogen load estimation are readily available in the Welaka town. The data are available in either the public domain (e.g., USGS websites) or FDEP database. Site-specific data include DEM, waterbodies, septic locations, hydraulic conductivity, and porosity. The values of smoothing factor, dispersivity, decay coefficient, inflow mass to groundwater, and source plane concentration are obtained from literature.

(6) Among the 125 removed septic systems, only 36 septic systems contribute nitrogen load to the St. Johns River. The rest of septic systems contribute nitrogen load to waterbodies that are not connected to the St. Johns River. As a result, not all the removed septic systems contributed to nitrogen load reduction in the TMDL practice. This suggests importance of considering spatial variability in environmental management of nitrogen contamination.

(7) For all the four modeling cases considered in this study, the load estimate to the St. Johns River (FID 57) is the largest, although the number of contributing septic systems (36 nos.) to the river is not the largest. Waterbody 31 has the largest number (52) of contributing septic systems, its load estimate is the second largest, because the flow paths associated with the waterbody are longer than those with the river.

(8) The denitrification coefficient is the most influential parameter to the load estimate. More effort should be spent to determine the appropriate value of the parameter for more accurate estimation of load reduction.

Appendix A lists the input and output files of this modeling project. The readers who are interested in repeating the modeling results may contact Professor Ming Ye (mye@fsu.edu) to request these files.

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APPENDIX A

The following files were used in the ArcNLET modeling. The readers who may repeat the ArcNLET modeling using the input files listed below, and their modeling results should be the same as the output files listed below.

List of Input Files:

1. NED-DEM-3m×3m (NED 3m_Clip)
2. Waterbody-NHD-River, Lakes, Ponds, Swamps and Marshes (waterbody_clipped.shp)
3. Septic tank location file (SepticT_prj.shp)
4. Heterogeneous Porosity (porosity2)
5. Heterogeneous hydraulic conductivity (hydr_cond1)

List of Output Files:

1. Smoothed DEM of the first round of smoothing with the smoothing factor of 60: (tmp_smoothedDEM635417071394261169.img).
2. DEM file for the waterbodies: (WB_maskRaster)
3. Smoothed DEM after including the waterbodies elevation and having of the second round of smoothing with the smoothing factor of 10: (tmp_smoothedDEM635417074562592387.img)
4. Smoothed DEM after including the waterbodies elevation and having of the third round of smoothing with the smoothing factor of 5: (tmp_smoothedDEM635417086069920569.img)
5. Seepage velocity: velocity magnitude (vel_mag4.img) and velocity direction (vel_dir4.img)
6. Simulated flow path: Particle paths (part_Path4Y)
7. Simulated plume for Case 1: Geodatabase files for Info and raster files for plume distribution (Case1_Plume_info and Case1_Plume)
8. Simulated plume for Case 2: Geodatabase files for Info and raster files for plume distribution (Case2_Plume_info and Case2_Plume)
9. Simulated plume for Case 3: Geodatabase files for Info and raster files for plume distribution (Case3_Plume_info and Case3_Plume)
10. Simulated plume for Case 4: Geodatabase files for Info and raster files for plume distribution (Case4_Plume_info and Case4_Plume)