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Field and Numerical Evaluation of Nitrogen Transport from Septic Systems in Surfical Aquifer Systems to Charlotte Harbor, Florida

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FIELD AND NUMERICAL EVALUATION OF NITROGEN TRANSPORT FROM SEPTIC SYSTEMS IN SURFICAL AQUIFER SYSTEMS TO CHARLOTTE HARBOR, FLORIDA

by

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Shallow water tables in coastal surficial aquifers limit effective treatment of septic effluent which can result in excess nutrient loading into nearby surface water bodies. Approximately 45,000 septic systems in Charlotte County, Florida transmit effluent into an under studied surficial aquifer and contribute to harmful algal blooms and outbreaks of E. coli. An undeveloped field site was characterized using standard hydrogeologic methods, including a one-year duration natural gradient tracer test, to obtain representative lithology of the sandy surficial aquifer and estimates of groundwater velocity, flow directions, effective porosity and dispersion. These data were used to support the development of a groundwater flow and nitrogen transport model of a nearby coastal subdivision connected to 2000 septic systems with high septic and canal density. Model results were used to assess the impacts of coastal ground water discharge in regions with high septic density near the coastline, and ground water – canal interaction and potential for rapid transport into Charlotte Harbor. Timescales associated with nitrogen removal by natural groundwater flow in the surficial aquifer following instantaneous septic to sewer conversion were on the order of 2-3 years for 50% reduction and 8-10 years for 90% reduction. Canals were found to significantly influence groundwater flow and rapidly convey nitrogen to Charlotte Harbor. Pre and post sewer conversion data on nitrate and total

nitrogen in shallow groundwater from a nearby field site was obtained post-model development and supports the timescales predicted by the numerical model.

ACKNOWLEDGMENTS

I would like to begin by expressing my gratitude to Dr. Matt Reeves for his guidance and patience during this process. Dr. Matt Reeves and I would also like to thank Charlotte County Utilities Director Craig Rudy for project funding, Ruta Vardys and Andreia Paulino of CCUD for field site and technical support, and developing volunteer sampling protocol; Elizabeth Staugler, University of Florida IAFS Extension for volunteer coordination and anonymous volunteers for tracer sampling; Sandra Lavoie and the East Port Laboratory for water chemistry analyses and analytical support; and the Enterprise Charlotte Economic Council and Western Michigan University for seed funding for the El Jobean study.

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INTRODUCTION

Eutrophication occurs when surface water is overly enriched in nutrients and is typically caused by the overabundance of phosphorus and/or nitrogen from agricultural runoff or livestock and human waste products (Schindler 1974; Nixon 1995; McIsaac et al. 2001; Conley et al. 2009). Excess nutrients create favorable conditions for plant, algae, and bacterial growth beyond the natural balance. A common indicator of eutrophication is excess cyanobacteria in freshwater systems and phytoplankton in marine environments. Accelerated growth of algae in nutrient-rich conditions degrade water quality through the uptake of dissolved oxygen and can lead to hypoxic conditions that adversely impact fish populations (Anderson et al. 2002; Conley et al. 2009; Hautier et al. 2009). Due to these factors, algal overgrowths are often described as Harmful Algal Blooms (HABs). *Karenia brevis*, a naturally occurring dinoflagellate in the Gulf of Mexico, has the potential to rapidly concentrate and form red tides, a subset of HABs where the combination of hypoxic conditions and the release brevetoxins can lead to significant fish kills (Poli et al. 1986; Landsburg 2002; Landsburg et al. 2009; Pierce and Henry 2008). Human health may be adversely impacted by red tides as near shore wave and wind action can facilitate the transfer of the neurotoxins to the airborne phase (Pierce 1986).

Phosphorus is typically not a limiting nutrient for eutrophication in Florida coastal waters due to natural enrichment of phosphorous in the soil, particularly along the southwest coast (Zhang et al. 2002). Shallow subsurface mining of phosphorite deposits within the Peace River watershed has occurred since 1883 (Mansfield 1942). Ground water dissolves and transports naturally occurring phosphorous in the soils to coastal waterways resulting in nitrogen serving as the limiting nutrient for phytoplankton growth. HAB frequency in southwest Florida has increased approximately 15-fold from a baseline period of 1954-1963 to 1994-2002 (Brand and Compton 2007). Many of these occurrences are located in the northern section of Charlotte Harbor, which is fed by the Peace River. Given the abundance of naturally occurring phosphorus, limiting nitrogen fluxes into coastal waters is a viable management tool to control HABs (Froelich et al. 1985; Vargo et al. 2004).

Anthropogenic sources of nitrogen, such as agricultural return flows, fertilizers, atmospheric deposition and effluent water disposal, are likely responsible for the increases in HABs and red tides (LaPointe and Clark 1992, Paul et al. 2000, Shaddox and Unruh 2018). Septic effluent is a poorly characterized, non-point source of nitrogen and may account for 3-7% of nitrogen loading to the Charlotte Harbor (MML 1997, Lapointe et al, 2015). Over 45,000 homes in Charlotte County are connected to septic systems, creating the potential for excess nitrogen loading into Charlotte Harbor. The vast majority (~95%) of these are conventional septic systems that capture outflowing waste in a tank where a baffle is used to separate solids (which settle to the bottom of the tank) from liquids (Tilly et al. 2014). Anaerobic conditions within the tank reduce solid volumes through biodegradation; liquid effluent flows out of the tank into a drain field with laterals comprised of perforated PVC pipes. Drain field laterals are installed in trenches filled with highly permeable media, such as gravel or pebbles, to facilitate infiltration (AGT 1998).

Treatment of waste by septic systems relies on naturally occurring geochemical reactions within the soil that are often facilitated by microbes (Cogger and Carlile 1984; Tilly et al. 2014). In a properly functioning system, nitrifying bacteria in the shallow drainfield convert ammonium (NH_4^+) in the presence of oxygen to nitrite (NO_2^-) and nitrate (NO_3^-). Nitrate may then be converted to N_2 gas under anaerobic conditions through denitrification; this results in a transfer of nitrogen to the atmosphere that reduces nitrogen loading to groundwater (Siler 1996; Tilly et al. 2014). The conversion of nitrate to N_2 gas is often limited by field conditions, with most conventional septic systems achieving only 20-25% nitrogen removal through denitrification (Costa et al. 2002).

Florida Administrate Code 64E-6.001 requires a minimum separation distance of 61 cm between the drain field and water table, although it is not uncommon for the water table to seasonally intersect land surface in some areas. Dramatic water table fluctuations between seasonal dry and wet periods, and close proximity to the land-water interface along coastlines and/or canal systems, often violate the minimum separation distance criterion. Shallow water table conditions render many of the older septic systems ineffective (Lambert and Burnett 2003; Meeroff 2008) as nitrogen-rich septic effluent is injected directly into surficial aquifers (Mallin 2013). Submarine groundwater discharge along coastal areas or to adjacent canals from these shallow aquifers serves as a primary mechanism of septic-derived nitrogen loading to Charlotte Harbor.

Several studies have investigated the role of nitrogen as the leading cause of the degradation of Charlotte Harbor water quality and have noted correlations between residential areas along the coast with high densities of septic systems and elevated nitrogen levels in nearby surface water bodies (LaPointe 1987; LaPointe and Clark 1992; LaPointe et al. 2004). These studies primarily rely on data obtained from surface water sampling and do not investigate specific mechanisms responsible for nitrogen loading and subsurface transport to coastal waters. LaPointe (2016) detected sucralose, an artificial sweetener, in canal waters as a proxy for septic discharge from shallow aquifers into surface water canals. The presence of sucralose found throughout the canal samples indicated relatively widespread discharge of septic effluent into the canal systems. Extensive usage of canals to lower water tables for home construction, coupled with a high density of septic systems, suggest that canal systems likely serve as a dominant transport pathway for nitrogen loading to Florida coastal waters. More importantly, canal systems are tidally influenced which may facilitate rapid nitrogen transport to Charlotte Harbor. The influence of tidal cycling on nitrogen flux to coastal waters has not been comprehensively investigated and remains relatively unknown.

Five dominant mechanisms are responsible for nitrogen loading to Charlotte Harbor: (1) streamflow from the Peace and Myakka rivers, (2) atmospheric deposition (3) saturation excess overland flow, (4) coastal groundwater discharge, and (5) groundwater discharge into canal systems. Together, these mechanisms account for 99% of estimated nitrogen fluxes to Charlotte Harbor (MML 1997; Badruzzaman et al. 2012). Agricultural applications of fertilizers and return flows have been identified as the dominant nitrogen sources to the Peace and Myakka rivers (McPherson et al. 1996; MML 1997). Mote Marine Laboratories (MML, 1997) estimates atmospheric deposition to contribute approximately 20% of total nitrogen into the harbor. The third mechanism, where the land table seasonally intersects the land surface, is limited to small drainages that convey only minor nitrogen contributions to the harbor but are responsible for numerous beach closures due to high total coliform levels (Lipp et al. 2001; LaPointe et al. 2016; CHNEP 2019). This study serves as one of the few groundwater-specific investigations into the fourth and fifth mechanisms, as very little is known about the surficial aquifer in the Charlotte Harbor region. This paper details two integrated projects to better characterize the transport of septic-derived nitrogen. The first is a reconnaissance groundwater study of the El Jobean area that involves hydrogeologic characterization of the surficial sandy aquifer, and application of a conservative bromide tracer to obtain realistic ranges of background groundwater velocity. These data are then used to support the development of a numerical model of the nearby Ackerman subdivision, an area with high septic and canal density, to simulate groundwater flow, groundwater-canal interaction, and decadal-scale nitrogen transport. The numerical model is motivated by the need to better understand impacts of septic-to-sewer conversion and provides estimates of timescales associated with the flushing of nitrogen from the shallow surficial aquifer after sewer conversion. Study findings are then placed in the context of broader implications for nitrogen transport from septic systems situated in coastal regions with and without canals.

EL JOBEAN

A reconnaissance field study was performed at a site located at 4399 Buckwheat Rd, El Jobean, FL. The site consists of an undeveloped 0.27 ha lot that trends northeast to southwest which is situated approximately 500 meters from the Charlotte Harbor coastline and is unaffected by canals (Figure 1). Neither septic systems nor structures are present, and the study location was intentionally selected to serve as an unimpacted location for obtaining estimates of natural background groundwater conditions unaffected by septic systems. Collected reconnaissance data from the site include subsurface lithology of the surficial aquifer system, hydraulic conductivity estimated from slug tests, water levels used to compute the hydraulic gradient, and general water chemistry. These data were then used in the experimental design of a one-year duration tracer test to determine tracer wellfield configuration and sampling frequency.

Figure 1. Map showing the locations of El Jobean and Ackerman relative to Charlotte Harbor and the Myakka and Peace rivers (Google Earth, 2020).

TRACER TEST DESIGN

A natural-gradient tracer test was performed to obtain accurate estimates of groundwater velocity from conservative tracer breakthroughs. Values of velocity obtained from tracer tests represent the effective integration of all hydraulic parameters governing groundwater movement and are superior to indirect Darcy estimates. Experimental tracer test design began with a series of laboratory permeameter tests performed on 50 soil samples manually collected from hand auger samples collected from Spring Lake, a proxy site located approximately 9 km from the study location. Rigid wall permeameters with a diameter of 5.1 cm were used to measure the flux of

water through the collected samples under constant head conditions (Fetter 2001). The samples were compacted in the permeameter columns to approximate natural field conditions using a slide hammer. Results of permeameter testing for the sandy surficial aquifer sediment yield an average hydraulic conductivity (K) value equal to
1.0×10^{-5} m/s.

Four separate sieve size analyses were performed to determine the distribution of particle and characteristic grain sizes of the aquifer sediment collected from the proxy site. Approximately 200 g of aquifer sediment was placed into a series of sieves and a mechanical shaker was used to partition the sample. The mass of each sieve was recorded before and after addition of the sediment, and sediment mass retained in each screen was normalized by total sample mass to calculate percent mass retained in each sieve. These data were plotted as a grain-size analysis curve (not shown) and d_{60}, the particle diameter corresponding to 60% of mass retained, ranged from 0.32 to 0.48 mm (Sterrett 2007). Uniformity coefficients ranging between 2.7 and 2.9 and characteristic grain sizes of the sediment indicate a well-sorted medium sand with some fines. All monitoring wells installed in the site utilized a 0.25 mm slot size.

Accurate characterization of groundwater flow directions and hydraulic gradient is essential for proper tracer test design in order to maximize tracer recovery and appropriately place wells, as the placement of monitoring wells at an angle to the dominant groundwater direction can lead to poor tracer mass recovery and inaccurate estimates of groundwater velocity and dispersion. Six 2.5 cm diameter PVC reconnaissance wells were installed using a 13 cm diameter hand auger. Four of the wells were installed on the outer boundaries (corners) of the site with the other two wells placed in the interior. Subsurface lithology was recorded during the installation of each well. Each reconnaissance well was developed using a low flow peristaltic pump for approximately ten minutes to ensure proper hydraulic connection to the shallow aquifer system. Water levels and water chemistry samples were collected 24 hours after well development for characterization of major cations and anions. The Charlotte County Utilities Department (CCUD) performed a high precision survey of the well field using a total station to ensure accurate delineation of flow directions. The hydraulic gradient within the field site is 0.0015 with a groundwater flow direction that is orthogonal to the coastline and trends approximately northeast to southwest. Falling head slug tests were performed on two, 5.1 cm reconnaissance wells located in the interior of the site to obtain in-situ estimates of hydraulic conductivity of the surficial aquifer. Slug test preparation began by uncapping of each of the test wells to allow for atmospheric pressure equilibrium to be reached; static water levels were recorded to ensure full recovery prior to each test. Each test began by pouring 11-15 L of water to fill the void space from the water table to the top of casing to generate the falling head slugs. Water level recovery was monitored using high-resolution, vented pressure transducers placed approximately 15 cm from the bottom of the well. Collected data were analyzed in AQTESOLV using the KGS solution which analyzes the full water-level response. Slug test estimated K is $1.2\times10^{-5} \pm 0.2\times10^{-5}$ m/s, consistent with the average of permeameter tests of the aquifer sediment from the Spring Lake proxy site (1.0×10^{-5} m/s). These hydraulic conductivity values are within the standard range of hydraulic conductivity for a well-sorted medium sand (Heath 1983).

Sodium bromide was selected for use as a conservative tracer, where bromide (Br^-) is the ion of interest. Proximity to the coastline indicated the possibility of higher background concentrations in groundwater, although analysis of water samples collected from the field site at the East Port Laboratory indicate background Br^- groundwater concentrations in the range of 2-5 mg/L. An analytical solution to the advection-dispersion equation (ADE) for an instantaneous tracer injection (Bear, 1972) was coded in Mathematica for use as a screening-level tool to determine total tracer mass and the placement and location of tracer wells. The Br^- tracer plume was simulated for a transport time of one year using a 2D Gaussian density (ADE solution) given values of tracer mass, average groundwater velocity, and scale-dependent longitudinal and transverse dispersion (Gelhar et al. 1992). The model dilutes concentrations in the vertical direction to account for smearing of the tracer according to a 1m estimate of natural water table fluctuation. An average groundwater velocity estimate of approximately 2 m/yr was used in the predictions, computed from Darcy's Law using a hydraulic gradient of 0.0015, hydraulic conductivity of 1.0×10^{-5} m/s, and an effective porosity of 0.25 based on literature values for medium sand. Values of longitudinal and transverse horizontal dispersivity of 0.2 m (10% of the estimated 2 m transport distance) and 0.01 m (10% longitudinal dispersivity), respectively, were input into the model. An iterative process was used to study the resultant spatial distribution of tracer concentrations for a given initial tracer mass to ensure that proper scaling of the applied tracer mass will lead to concentrations sufficiently above background Br^- levels. The analysis indicated that a tracer mass of 5 kg sodium bromide (equivalent to 3.8 kg Br^-) was ideal for the tracer test, and biweekly sampling of tracer wells would be suitable given the relatively slow velocity.

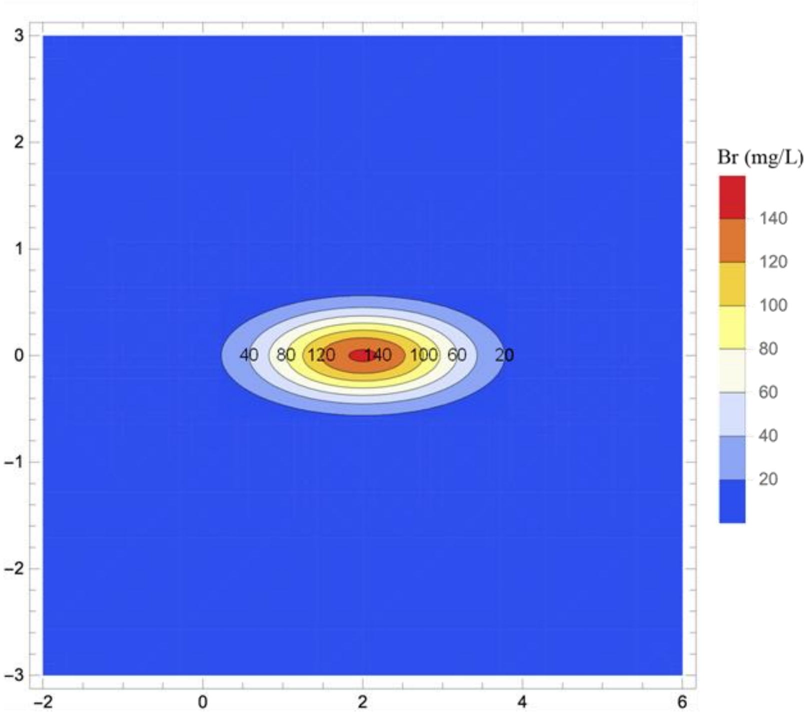

Figure 2. Predicted 2D bromide plume concentration (mg/L) profile after one-year of transport under natural gradient conditions.

Results from the analytical model suggest that at one year of elapsed time, the center of plume mass would travel approximately 2 m down gradient and spread nearly one meter in the transverse horizontal direction (Figure 2). Using this information, 9 monitoring wells were installed parallel to groundwater flow with two transects containing multiple wells located at 1 m and 2 m downgradient from the injection well to capture traverse horizontal spreading (Figure 3). The naming convention of the boreholes corresponds to the downgradient distance of each monitoring well from the tracer release well in feet. Boreholes were hand augered to depths ranging from 2 to 2.7 m and terminated at the contact with a laterally continuous clay unit with a thickness exceeding 3 m. Lithology was recorded as the bores were created. 2.5 cm-diameter wells ending in 1.5 m-long screens with 0.25 mm slots were installed by filling the annulus with sediment removed from the holes. All wells are screened across the entire saturated thickness of the shallow surficial aquifer. Similar to the shallow wells used to determine groundwater flow directions and hydraulic gradient, the tracer wells were surveyed by CCUD using a total station. Total longitudinal length of the tracer well field is 4.6 m. Well density in the first meter of the well field is higher to resolve early tracer migration.

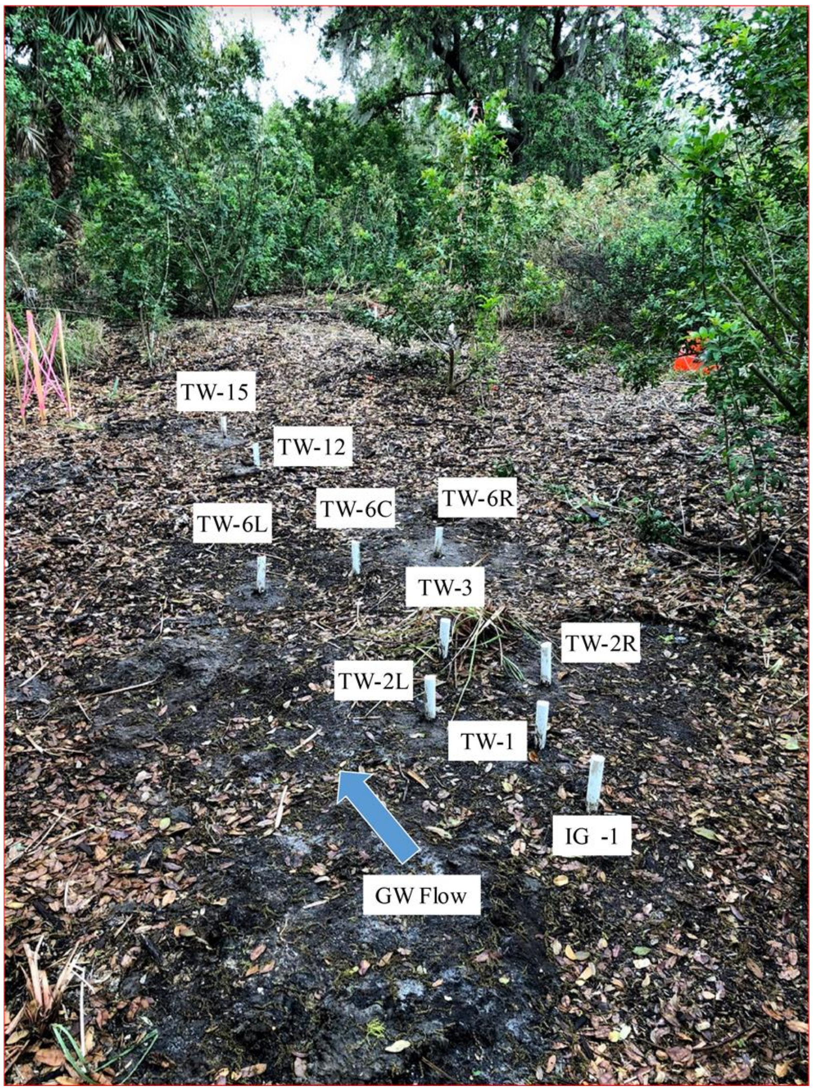

Figure 3. Tracer well field located at the El Jobean site. The center transect is directly parallel to the direction of groundwater flow.

TRACER TEST RESULTS

A total of 5 kg sodium bromide dissolved in 19 liters of distilled water was injected via peristaltic pump on March 7, 2018 at an approximate rate of 0.4 L/min into the tracer injection well. A standard operating procedure for tracer sampling was created in partnership with the CCUD and University of Florida Institute of Food and Agriculture (IFAS) Extension. Wells were sampled biweekly by a network of volunteers from the IFAS Extension until 11/14/2019. The water samples were transported and analyzed for bromide at the East Port Laboratory. The duration of the tracer test allowed for detailed study of tracer migration under seasonal wet and dry conditions. Temporal breakthroughs were numerically fit to a 1D form of the ADE for an instantaneous slug injection of a conservative solute (Bear, 1972):

$$C(x,t) = \frac{\gamma}{\sqrt{4\pi\alpha Vt}} exp\left[\frac{x - Vt}{4\alpha Vt}\right]$$

where velocity (v) [L/T], dispersivity (α) [L], time (t) [T], and gamma (γ) [dimensionless] scale breakthrough mass according to monitoring well distance (x) [L] downgradient from the injection well. To fit the ADE curve to the observed data, the center of plume mass was first visually matched to the peak of the data by adjusting velocity, and the non-linear GRG solver in Microsoft Excel was then used to numerically compute best-fit estimates of gamma and dispersivity. Breakthroughs for all tracer wells with sufficient data were analyzed, although wells TW-6R, TW-6C and TW-6L provided longer transport distances (~2 m) and subsequently better data for computation of velocity and dispersivity (Figure 4). ADE curve fits to these wells indicate groundwater velocity ranges from 3.6 to 4.6 m/yr and horizontal dispersivity ranges from 4 to 10% of transport distance. These velocity values are approximately twice as high as the initial tracer estimates and can be obtained using Darcy's law with field-estimated hydraulic conductivity and hydraulic gradient if effective porosity is reduced to 12-16%.

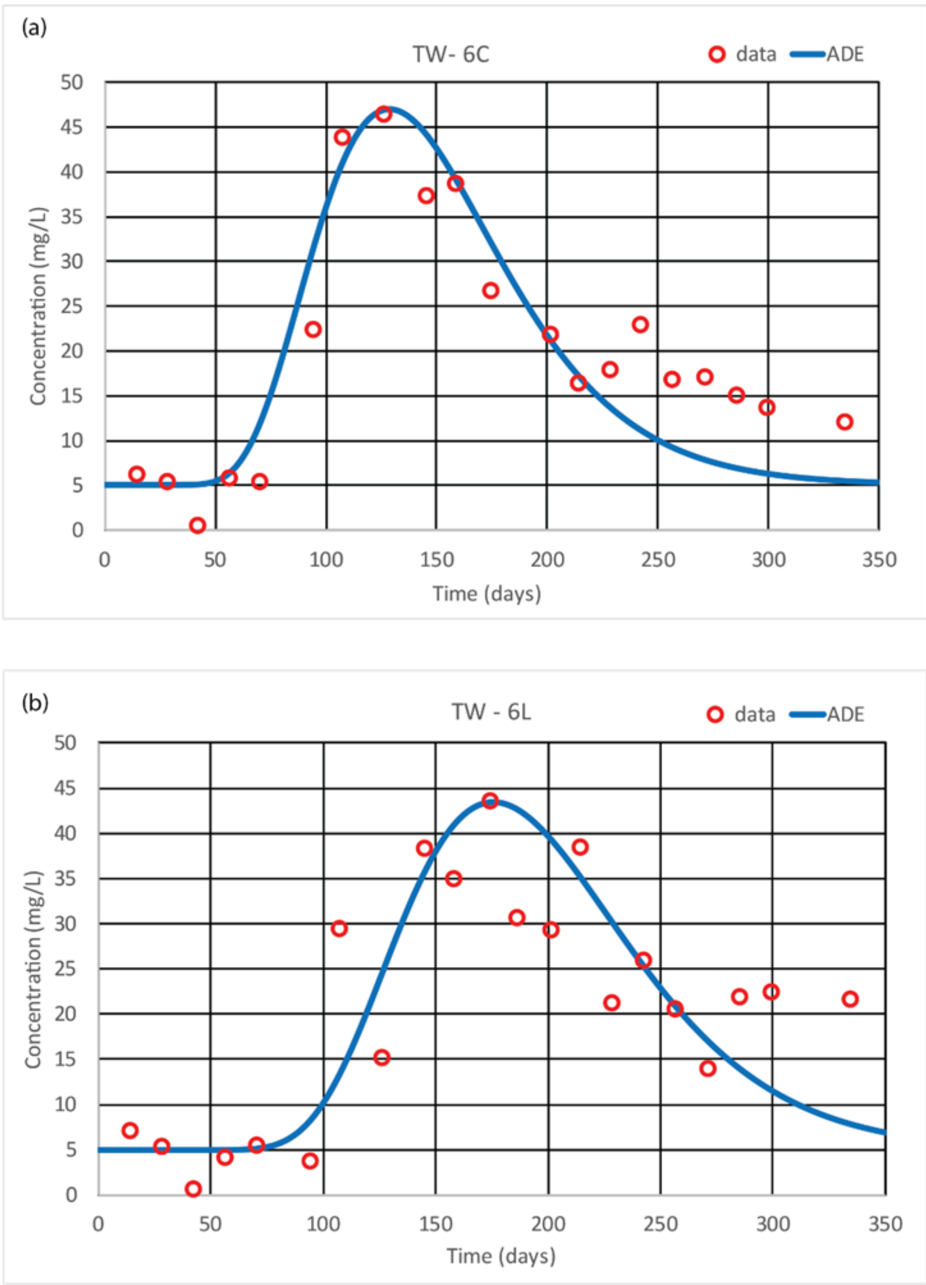

Figure 4. Breakthrough data for TW-6C (A) and TW-6L (B) with best-fit ADE solutions.

ACKERMAN

The Ackerman subdivision is located along the coast of Charlotte Harbor approximately 3.5 km west of the El Jobean tracer test (Figure 1) and contains 2000 homes connected to septic systems. Due to its proximity to the coastline and high canal density, the CCUD is planning on converting all homes in the Ackerman subdivision to sewer to reduce the loading of septic-derived nitrogen to Charlotte Harbor. Charlotte County requested a study of this area to provide likely trends and timescales associated with the flushing of nitrogen by natural groundwater flow processes through the shallow surficial aquifer and into Charlotte Harbor after septic to sewer conversion, and to enhance the current knowledge of impacts of: (1) coastal groundwater discharge in regions with high septic density near the coastline, and (2) ground water – canal interaction and the potential for rapid nitrogen transport into Charlotte Harbor. The numerical model was developed using Visual Modflow Flex 6.1 (VMF) which integrates MODFLOW 2005 (Harbaugh 2005), MODPATH Version 7 (Pollock 2016), and MT3DMS (Bedkar et al. 2016). The flow and transport models are supported by field data collected from the El Jobean site, including hydraulic gradient, tracer, slug, and permeameter tests, and hand auger observations of subsurface lithology at the Ackerman site. Additional data include estimates of monthly precipitation and evapotranspiration, and septic system effluent volumes and nitrogen concentrations. The flow and transport model development, calibration, and nitrogen transport results are provided in detail below.

MODEL DEVELOPMENT AND CALIBRATION

The site is located between the freshwater outlets of the Myakka and Peace rivers. The MODFLOW model domain adheres to natural hydrologic boundaries and encompasses the entire peninsula landform bounded by canals on the east, west, and south; the northern extend of the model is constrained at Edgewater Drive (Figure 1). The canals are hydraulically connected to Charlotte Harbor and facilitate drainage of the shallow groundwater system. Both the shallow groundwater salinity measured at El Jobean (average 3 parts per thousand) in this study and salinity measured in the Ackerman canals (average 7.6 parts per thousand) by LaPointe et al. (2016) indicate brackish waters with relatively minor salinity differences that are approximately 10% of the contrast between freshwater and salt water. Given these minor differences, density driven effects on fluid flow were not simulated in the model. Canals bounding the model domain are represented as constant head boundaries set to an elevation of mean sea level.

A finite-difference model grid with horizontal cell dimensions of 40 m on a side was found to best discretize canal geometry (Figure 5). Variable cell thickness was used to capture the distribution of land surface elevations from a 10 m resolution digital elevation map. These elevations were converted to gridded points, kriged using VMF, and then assigned to individual model cells. All homes in the Ackerman area are connected to city water and the subsurface geology at the site is unknown beyond the sparse hand augering performed in this study to map water table elevations in support of flow model calibration and regional geologic interpretations. Subsurface textures encountered during augering mostly included sandy sediment (similar to El

Jobean) with a higher content of limestone in gravel and cobble size fractions. The occurrence of rock fragments increased with depth and made hand augering significantly more challenging. Discussions with local utility operators with experience in this area and visual inspection of canal dredged sediment further confirmed the ubiquitous limestone cobbles. Based on regional interpretations by Wolansky (1983), the surficial aquifer system at Ackerman consists of undifferentiated fine to medium light grey quartz sands with some interbedded clay lenses and is underlain by a continuous confining unit of regional extent. Torres et al. (2001) reports the same regional lithologic sequence and notes the Upper Hawthorn Formation (shallow near Ackerman) consisting of sand, with shell beds and limestone clasts and a thick clay unit near the top, approximately 7.6 m below land surface. Consistent with regional interpretations and visual interpretations indicating a lack of a clay confining unit in the canals and canal excavated sediment, the lower model domain is set is set to 5 m below sea level to correspond to the presence of a confining unit located several meters below the bottom of the canals.

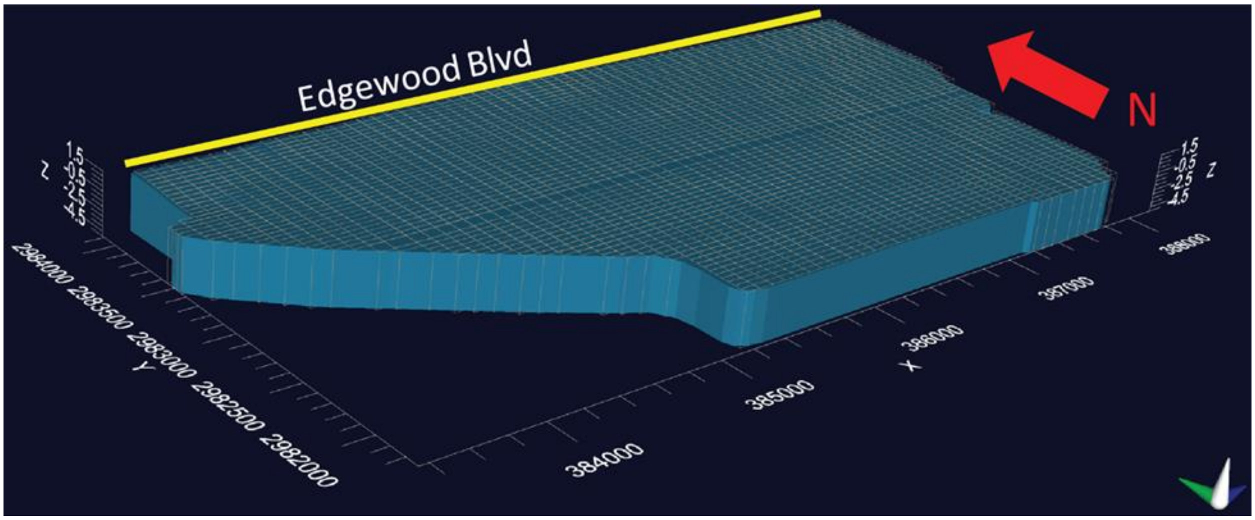

Figure 5. Model grid domain showing surface topography and variable cell thickness as a function of easting and northing and elevation. All values are in meters.

Texturally distinct sediment layers were not observed during either the augering or inspection of the canals and canal excavated sediment. Groundwater flow through the sandy surficial aquifer is simulated in the model using a single layer that extends from land surface to the clay confining unit. Hydraulic properties are assigned to the model according to two distinct zones representing the sandy surficial aquifer and canal system. Using a georeferenced Google Earth image of the Ackerman area, a shapefile was created in ArcMap to map the canal boundaries. This shapefile was then imported into VMF and superimposed on the model grid for delineation of grid cells representing the canals. The sandy surficial aquifer was assigned a best-fit K value of 6.5×10^{-4} m/s determined during calibration, and a K value of 10 m/s was assigned to the canals to establish an approximate five order of magnitude contrast between the canals and surrounding aquifer (Figure 6). This level of contrast ensures that the canals serve as highly preferential flow features within the groundwater flow system while maintaining numerical stability (Reeves et al. 2014). Aquifer recharge is approximated by the net difference between annual precipitation data retrieved from the Punta Gorda County Airport NOAA weather station and annual evapotranspiration computed by the Southwest Florida Water Management District (SWFWMD). The SWFWMD data are limited to 1997 to 2005 and restricted our analysis to this time period. The average of net differences in annual precipitation and ET for these data is 178 mm/yr, approximately 10% of precipitation. This value is uniformly applied to the top of all active cells within the model domain as a constant recharge flux boundary.

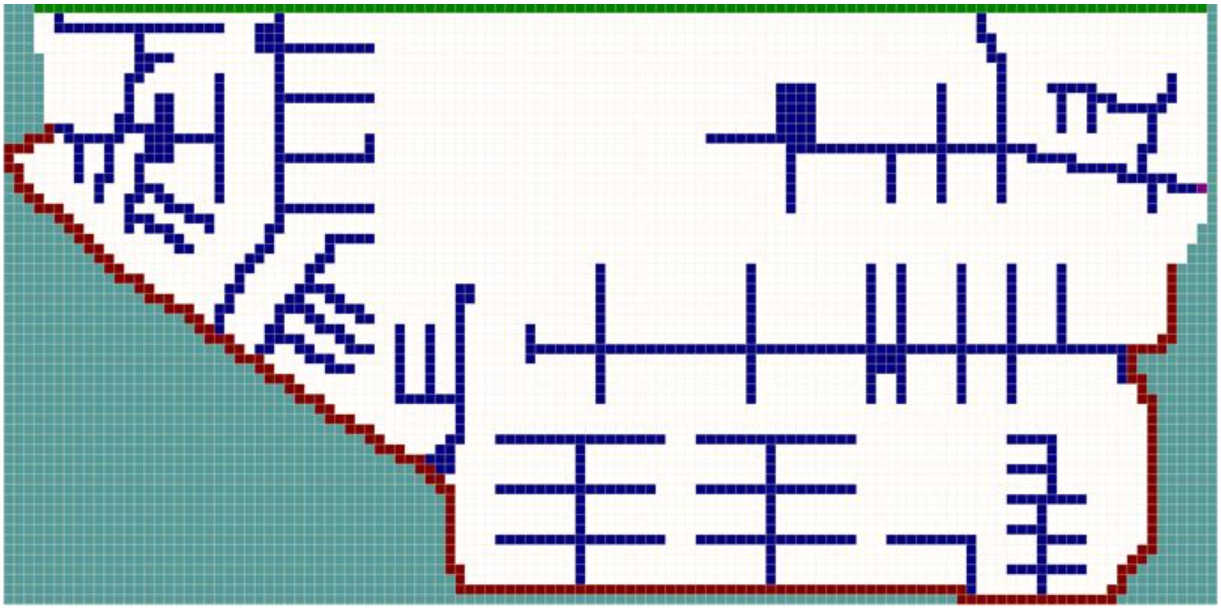

Figure 6. Model grid showing canal systems and boundary conditions. Green cells denote the northern constant head boundary, with red cells denoting the southern head boundary. White cells represent surficial aquifer cells with the canals shown in blue; inactive cells are aquamarine.

Proper assignment of the northern head boundary and calibration of the model necessitated the collection of water levels. In November of 2019, a water table survey was performed by hand augering until the water table was encountered and recording depths to water from land surface using an e-tape. Surface elevation for each of the auger holes was surveyed by the CCUD using a Trimble R10 rover with a reported elevational accuracy of ± 5 mm. The water level measurements were concentrated along four transects: E-W on Edgewood Dr, N-S on Collingswood and Midway Blvd, and along the southern canals, with 6 measurements were taken in the interior. The water levels along the northern transect yielded head values in the range of 0.5 to 0.7 m above sea mean level. The contour patterns show groundwater flow directions consistent with landform geometry and topography, and a water table high in the area near the center of the domain lacking canals (Figure 7). Head values assigned to the northern boundary are spatially variable to capture this pattern but are held constant over time. Of the 17 observation points, 12 were used in the model calibration. The 5 discarded points had erroneously low water table elevations likely caused by the presence of fine-grained sediment at the top of the saturated zone and insufficient time for water levels to equilibrate in the borehole prior to measurement. The parameter estimator PEST (Doherty 2015) was used to calibrate the model to the measured head data with best fit *K* values for the sandy aquifer and canals, respectively. The calibrated model has a root mean squared error (RMSE) equal to 0.035 m (3.5 cm), and when normalized by a 0.75 m head drop across the model domain, results in a model error of 6%.

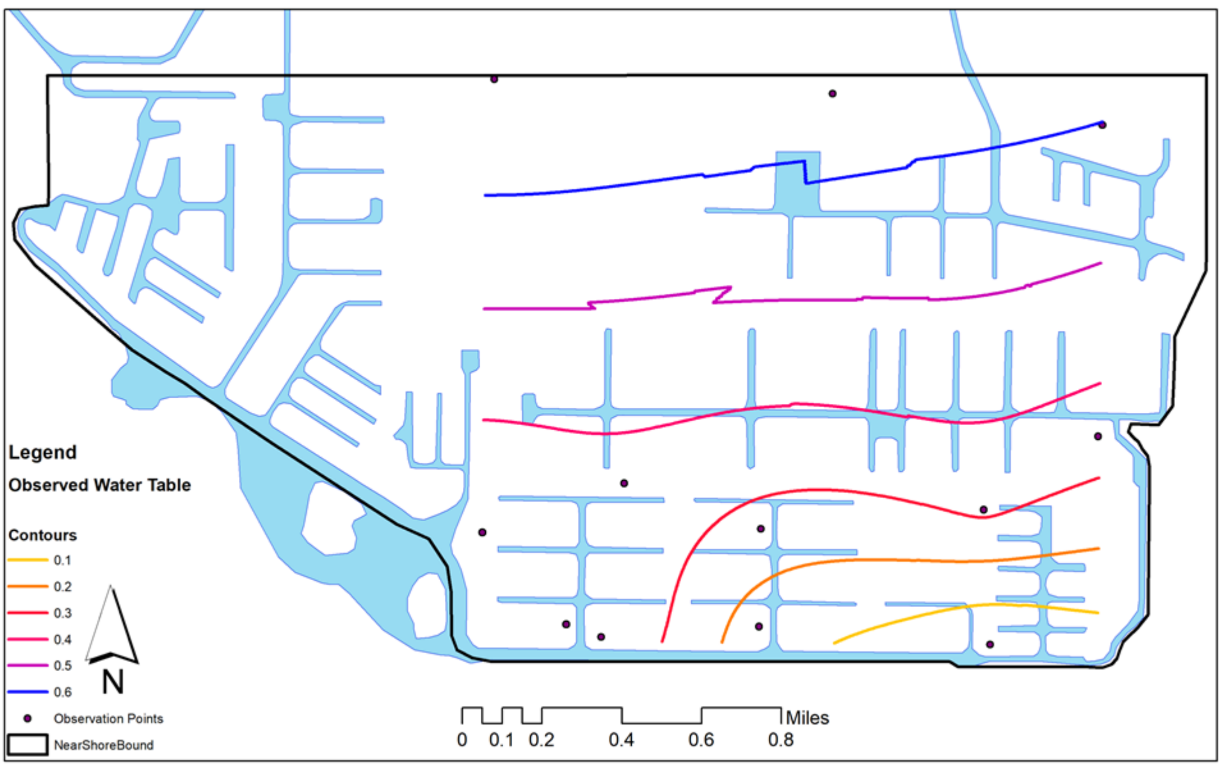

Figure 7. Map of water table contours created from field measurements.

Steady-state head contours of the calibrated model along with path lines are shown in Figure 8. Consistent with the water table map (Figure 7) general groundwater flow follows the geometry and topography of the landform with flow directions from north to south, southeast and southwest. Observed mounding in the north central area where canals are lacking is reproduced in the model. MODPATH particles (white dots) placed in the approximate location of the sewer conversion zones with pathways mapped through the flow system (white lines). The pathways show the concentration and preferred migration of particles through the canal systems.

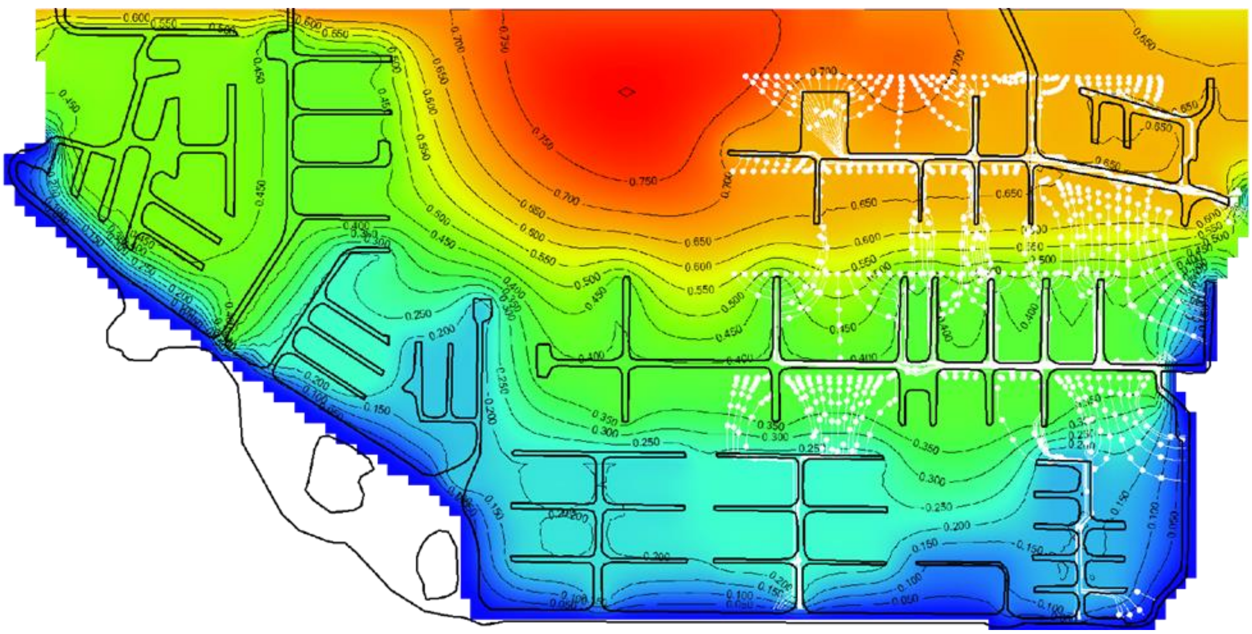

Figure 8. Steady-state head profile of base case model along with canal shapefile and MODPATH generated path lines shown as white dots.

SEPTIC NITROGEN TRANSPORT

Nitrogen and effluent volumes at the 23 O'Hara lift station were monitored by CCUD over an approximate 12-month period: 10/06/2016 to 09/25/2017. The lift station receives effluent from 733 households via low pressure sewer tanks that function analogous to septic tanks: the solids settle and only the liquid effluent leaves the tank. Thus, the lift station effluent serves as an ideal proxy for septic fluid and nitrogen contributions. The primary difference between the two systems is effluent in the low-pressure sewer is conveyed to a collection station for treatment, whereas septic tank effluent is gravity fed to the drain field for treatment in the soil. Fluid samples were collected weekly to quantify the mass loading of total nitrogen to the lift station. Records of water usage were then used by CCUD to account for variability in home occupancy throughout the year, resulting in an average household contribution of 11 kg/yr of total nitrogen and 160 m^3/yr of effluent.

Fluid and nitrogen loading within the Ackerman model is assigned according to five conversion zones outlined in the CCUD sewer conversion plan (Figure 9). The fluid and nitrogen mass fluxes from the 23 O'Hare lift station are scaled to the number of homes within each zone, and MT3DMS is used to model contaminant input as a constant flux (m/d) of water into the aquifer with a constant nitrogen concentration (mg/L) (Table 1). All nitrogen applied to the subsurface is assumed nitrate and non-sorptive (Almasri and Kaluarachchi 2007; Bhatnagar et al. 2010). Nitrogen transformations by various process were not simulated. The El Jobean tracer test data were used to assign an effective porosity of 14% to the surficial aquifer, longitudinal dispersivity of 0.4 m (10% of the cell size) and a transverse horizontal dispersivity of 0.04 m. The base case model applies a constant recharge rate of 178 mm/yr and represents canals as high K features.

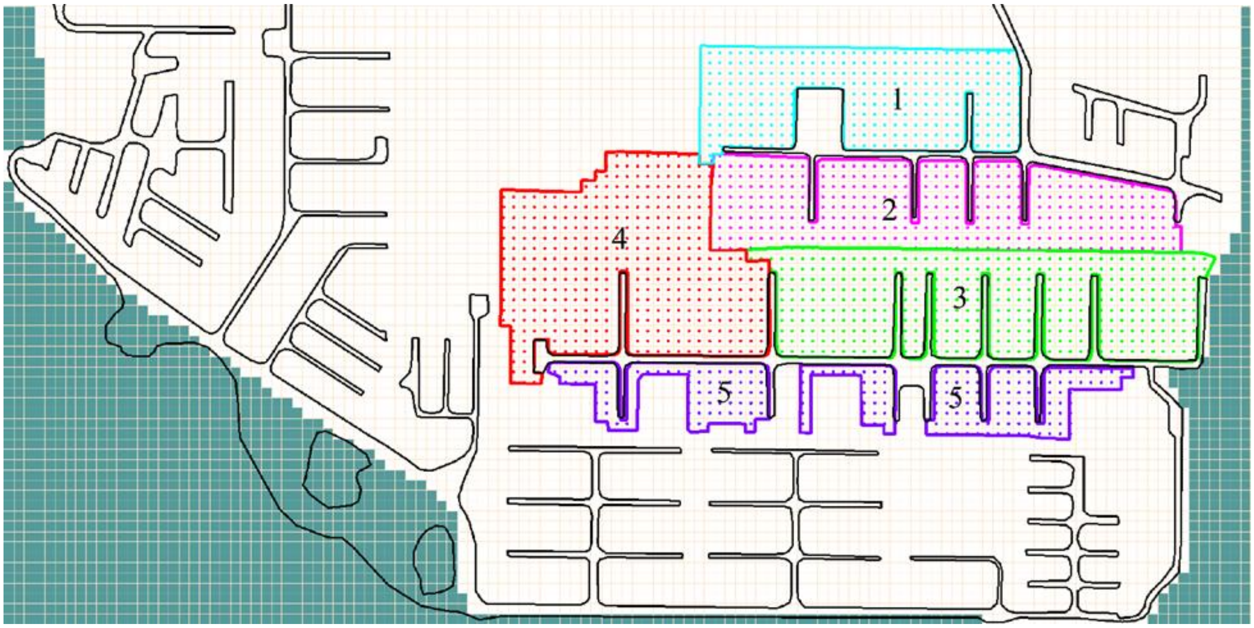

Figure 9. The Charlotte County Utilities District septic to sewer conversion plan zones mapped onto the Ackerman model domain.

Table 1. Septic loading rates and concentrations for each of the conversion zones outlined in the CCUD master plan.

Septic Conversion Zone	Flux Applied (m/day)	Nitrogen Concentration (mg/l)
1	2.1×10^{-4}	67
2	2.5×10^{-4}	67
3	2.3×10^{-4}	67
4	2.0×10^{-4}	67
5	2.4×10^{-4}	67

A 40-year model spin-up period is used to approximate long-term nitrogen loading and nitrate accumulation in the study area prior to septic conversion (Figure 10). Simulated nitrogen loading begins on January 1, 1980. After 30 years of septic loading into the system, nitrogen mass reaches an approximate steady-state plateau facilitated by constant rates of net recharge and applied septic effluent. This leads to an initial condition where the nitrogen mass applied to the surficial aquifer is equal to the amount of nitrogen leaving the aquifer with a large continuous nitrate plume extending from the Ackerman subdivision to the downgradient model boundary. An instantaneous and complete sewer conversion is initiated on January 1, 2020, ceasing all septic contributions of nitrogen and water to the aquifer (Figure 10). Sharp declines in nitrogen concentrations occur over time in the simulated surficial aquifer after sewer conversion. The timescales associated with the decline in nitrogen mass in the surficial aquifer is the focus of the model results.

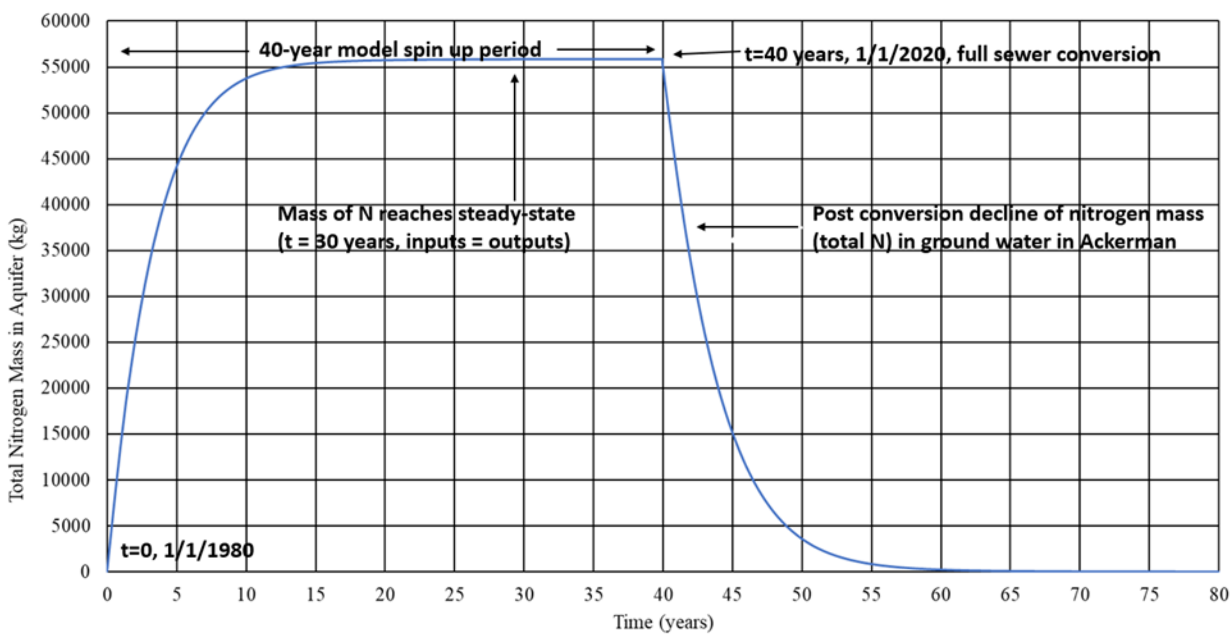

Figure 10. Simulated trends in total nitrogen mass within the surficial aquifer.

A total of 6 scenarios were used to account for uncertainty in recharge (and subsequent volumetric flow through the surficial aquifer) and provide a range of timescales associated with nitrogen flushing in the surficial aquifer after sewer conversion. Fluxes of 75% (R75), 100% (R100), and 125% (R125) of the base case recharge rate of 178 mm/yr were applied to the model. Each of these recharge scenarios were simulated with and without canal features to better understand the impact of canal features on nitrogen transport (Figure 11). Transport times are quantified using t_{50} and t_{10} values which represent the time after sewer conversion for nitrogen concentrations to decline to 50% and 10% of the original nitrogen mass remaining in the aquifer, respectively (Table 2). The models containing canals generated t_{50} values ranging from 2.4 (R125) to 3.2 (R75) yrs, with a base case (R100) estimate of 2.8 yrs (Table 2). Timescales associated with t_{10} are closer to a decade and range from 7.6 to 9.9 yrs. As expected, models without canals resulted in slower t_{50} and t_{10} values with these timescales ranging from 3.6 (R125) to 5.4 (R75) yrs and 11.6 (R125) to 16.6 (R75) yrs, respectively.

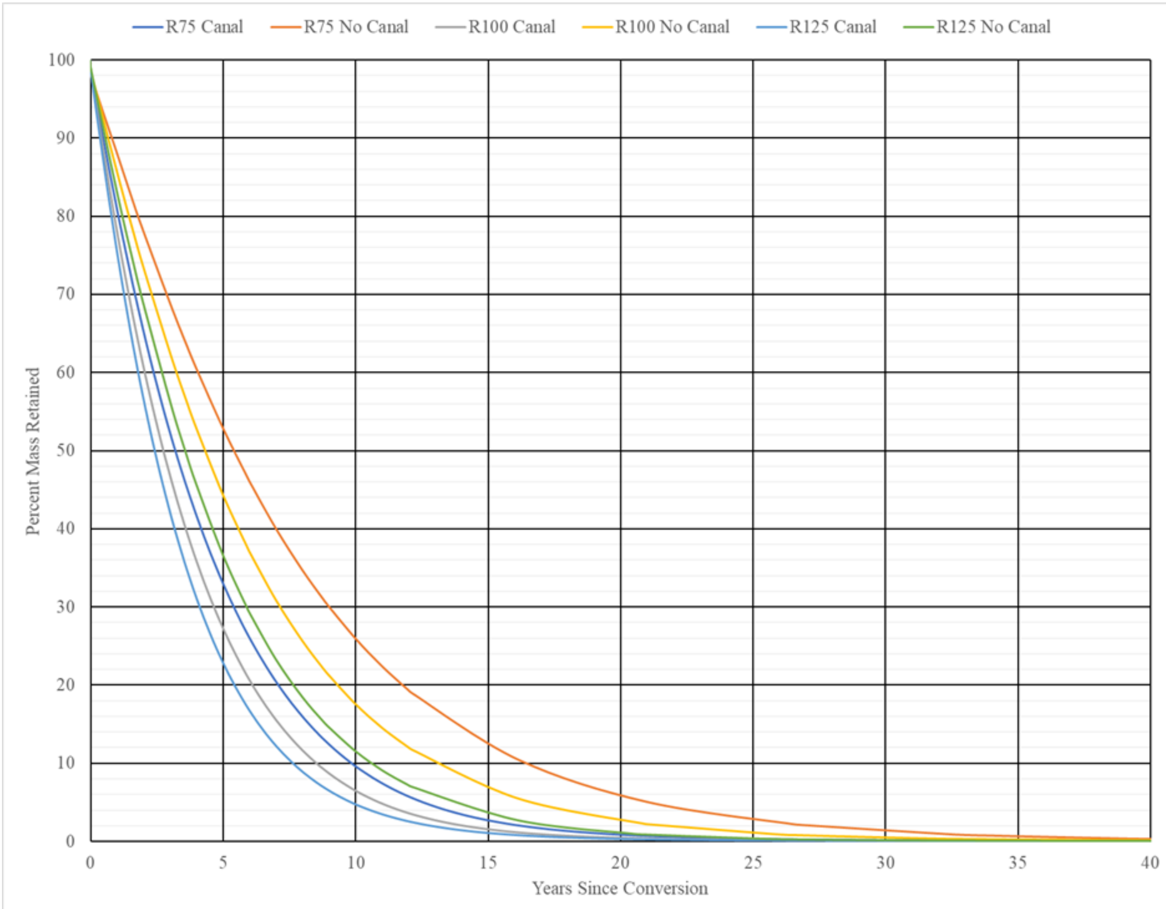

Figure 11. Graph of percent total nitrogen retained in the surficial aquifer post sewer conversion for all 6 model scenarios.

Table 2. Values of t_{50} and t_{10} for the six modeled scenarios.

	Canal		**Non-Canal**	
Recharge Scenario	t50	t10	t50	t10
R75	3.2	9.9	5.4	16.6
R100	2.8	8.5	4.3	13.2
R125	2.4	7.6	3.6	11.6

DISCUSSION

The El Jobean study provided reliable estimates of hydraulic conductivity, hydraulic gradient, effective porosity, dispersivity, and background velocity for areas unimpacted by canals and septic systems. These data ensured realistic parameterization of the Ackerman model. Water level contours indicate that groundwater flow is orthogonal to the coast roughly 500 m southwest of the site. Subsurface lithology consisted of layers of medium sand with varying organic content and underlain by a thick continuous clay unit with a thickness of at least 3 m. The area experiences an average of approximately 1370 mm of rainfall annually that resulted in dramatic water table fluctuations during the year on the order of 1 m. The one-year tracer test duration captured the influences of both wet and dry periods on tracer transport. Sampling after intense rainfall periods indicated significant dilution effects on bromide tracer concentrations during the wet period.

The Ackerman model allowed for detailed transport study of septic derived nitrogen in a near coastal environment with high canal and septic density. Application of hydraulic and transport properties estimated from El Jobean was relatively straightforward, except for aquifer hydraulic conductivity which is approximately one a half orders of magnitude greater than in-situ estimates at El Jobean. We attribute these differences to a combination of a textural transition from a sandy aquifer at El Jobean to a sandy aquifer with gravel and cobbles at Ackerman and scale effects. In addition to the textural differences between Ackerman and El Jobean, permeameter and slug tests provide small-scale estimates of K and do not take into account documented scale effects on hydraulic conductivity that arise from interconnected high K pathways that are detected at larger scales of measurement (Bradbury and Muldoon 1990; Rovey and Cherkauer 1995; Rovey 1997; Makuch and Cherkauer 1998; Makuch et al. 1999).

Values of hydraulic conductivity used in a model reflect the complex interplay between boundary conditions and recharge. In the Ackerman model, these include a tapered constant head boundary ranging from 0.75 to 0.60 m consistent with field data, and application of constant head boundaries set to sea level reflect the connection of the canal system to the northern portion of Charlotte Harbor. A constant recharge flux of 178 mm/yr was applied to the model based on the average net difference between precipitation and evapotranspiration over a 9-year period. These boundary configurations and applied fluxes serve as realistic constraints on the overall hydraulic conductivity of the surficial aquifer. The canals are surface water features approximated as high *K* features in the model. Even though hydraulic conductivity of the canals approaches infinity, contrasts of 5 orders of magnitude provides more than sufficient contrast to appropriately capture the hydraulic function of the canals on the groundwater flow system. This can be observed in the steady-state head distributions where the canals flatten the hydraulic gradient by effectively draining the shallow aquifer (Figure 8) and decrease the natural hydraulic gradient of 0.0015 measured at El Jobean to 0.00031 at Ackerman. A groundwater mound in the north central portion of the model forms in the only region not intersected by the canal system. This mound is consistent with the natural system and observed in field data collected by CCUD.

Lift station data collected from low pressure sewer tanks served as an ideal proxy for volumes and nitrogen concentrations in septic effluent. Net values per household were upscaled to a total of 5 sewer conversion zones in Ackerman (Figure 9). A model spin-up period with constant background recharge, septic effluent volumes, and nitrogen mass loading were used to approximate past background conditions (Figure 10). A constant nitrogen mass in the surficial aquifer was reached after 30 years with a plume emanating from the septic systems to the harbor (Figures 10 and 11). The septic to sewer conversion was initiated instantaneously and resulted in sharp declines in nitrogen mass in the shallow surficial aquifer. The plume sharply follows the canal system that effectively conveys nitrogen to the south and east of the model, as also indicated by the simulated pathlines shown in Figure 8. Results of canal scenarios show 50% reduction of nitrogen mass in 2.4-3.2 years and 90% reduction in 7.9-9.9 years given uncertainty in recharge. Exclusion of canals in the base model increased t_{50} and t_{10} timescales by 55%, emphasizing the impact of canals on transport times.

The model incorporated many simplifying assumptions and boundary conditions for investigating nitrogen transport in the Ackerman area, including steady-state recharge, inclusion of canals as high K porous media, instantaneous septic to sewer conversion, and no tidal cycling or processes affecting nitrogen transformation. These generalizations were useful for assessing nitrogen transport in areas with high septic and canal density and predict timescales associated with declines in nitrogen after sewer conversion. The steady-state recharge conditions allowed for a smooth spin-up period with the surficial aquifer reaching a constant nitrogen mass after 30 years. In reality, climate is non-stationery and variability in net recharge and ET will lead to perturbations in the simulated trends and non-steady state groundwater flow conditions and nitrogen mass transport. These perturbations will naturally lead to some variability and differences in nitrogen transport rates but are not expected to dramatically change the overall study findings and outcomes that indicate relatively fast reduction in nitrogen concentrations after septic to sewer conversion and the role of the canals as fast transport pathways for nitrogen transport to Charlotte Harbor.

An independent field study was performed on a small plot of coastal land, located approximately 7.1 km east of Ackerman, by the environmental firm Tetra Tech. The area converted contained 42 septic systems within an area of 4.25 ha located immediately on the Charlotte Harbor coast. Study results were shared by CCUD after the Ackerman model was completed and provided a rare opportunity to assess model performance and study findings. Total nitrogen, total phosphorus, and fecal coliforms were sampled in monitoring wells downgradient of a septic tank. After establishing a nitrogen and phosphorus baseline, houses in the area were converted to sewer and the monitoring well was sampled quarterly. Groundwater nitrogen concentrations pre-construction averaged 27 mg/l with concentrations ranging from 13 to 43 mg/l. These concentrations are consistent overall with the simulated plume concentrations, with the exception that the model evenly mixes the applied nitrogen over the entire aquifer thickness leading to more dilute concentrations. The nitrogen concentrations are stratified and accumulate in the upper portion of the surficial aquifer leading to higher concentrations. Timescales of t_{50} were achieved in the monitoring well after approximately 15 months, which is reasonably close to the 2-3-year prediction by the model. It is worth noting that the t_{50} values simulated by the model approximate the timescale at which nitrogen concentrations within the entire surficial aquifer are decreased in half. Visual inspection of plume concentrations after sewer conversion indicate that some regions of the model reach t_{50} in dramatically less time, while t_{50} for other regions is significantly higher. The primary difference is caused by the relative proximity of the septic systems to the nearest canal.

The simulation of tidal fluctuations requires a complex set of boundary conditions that was not possible in the model. Tidal fluctuations are likely an important rapid transport mechanism for septic derived nitrogen, particularly for septic systems located in close proximity to canals. At high tide, water levels in the canals exceed groundwater elevations and suppress groundwater discharge due to differences in hydrostatic head. This likely creates a mixing zone in the surficial aquifer surrounding the canals where harbor water mixes with shallow groundwater. This region is expected to have geochemical differences in salinity, pH, and redox conditions that may influence nitrogen transformations. As the tide subsides, water levels in the canal will become lower than shallow groundwater creating an enhanced hydraulic gradient between the shallow groundwater and canals. As discharge occurs across the groundwater-canal interface, any dissolved nitrogen, some of which may be in the ammonium from for septic systems that are located in very close proximity to the canals, will migrate into the canal. Once in the canal, nitrogen will naturally be transported to the harbor in the outgoing tidal water. A series of two high and low tides occurs approximately every 25 hours, and thus, the residence time of nitrogen in canal waters may be on the scale of several hours to a day. A future study using field and geochemical methods to better understand canal-shallow groundwater interaction and tidal cycling on nitrogen transport is currently in the planning stages.

Many coastal communities in Florida and elsewhere are experiencing similar issues with aging septic systems (Lapointe and Clark 1992; Bowen and Valiela 2004; LaPointe et al. 2004). Results of this study can be extrapolated to other coastal communities with and without canal systems to aid homeowners and legislators with policy decisions concerning septic systems. Both

Tomasko et al. (2001) and LaPointe et al. (2004) conclude that septic systems contribute a significant portion of excess nitrogen into coastal waters causing eutrophication. For many areas. the primary form of constraining nitrogen fluxes to coastal waters is converting septic systems to municipal sewers. This process is both costly and time consuming. In the case of Charlotte County, approximately 800 homes can be converted to sewer per year, resulting in over three decades for full conversion of all septic. Our study results indicate that sewer conversion plans should prioritize areas concentrated along the coast with high septic and canal density.

CONCLUSION

Two integrated projects were used to study nitrogen transport from shallow groundwater into Charlotte Harbor, FL. The first study at El Jobean provided reliable observations of subsurface lithology and estimates of hydraulic conductivity, hydraulic gradient and groundwater flow directions, velocity, dispersivity, and water chemistry that were used in the development of the conceptual and numerical model of the Ackerman subdivision. The Ackerman model has high septic and canal density, and model generated timescales associated with the flushing of nitrogen in the shallow surficial aquifer were used to assess the impacts of septic to sewer conversion. Steady-state head profiles, flow path lines, and timescales of nitrogen flushing for models incorporating varying recharge and canal and no canal scenarios were used to comprehensively investigate the influence of canals and recharge on nitrogen loading from Ackerman to Charlotte Harbor. The base case model indicated t_{50} and t_{10} values of 2.8 and 8.5 years, indicating the time for 50% and 10% of the original mass, respectively, to exit the aquifer and enter the harbor. Recharge variability was used to provide uncertainty bounds in t_{50} of 2.4 to 3.2 yrs., and t_{10} of

7.6 to 9.9 yrs. Excluding canals in the base model increased t_{50} and t_{10} timescales by 55%. The estimated times for the base case model are in good agreement with an independent field study by Tetra Tech of a nearby coastal area showing t_{50} values on the scale of 15 months and nitrate plume concentrations of 15-40 mg/L consistent with the numerical model. These data were provided after model development and provided a rare opportunity for validation of the model results. Future work is needed to further investigate fast transport mechanisms associated with canal-groundwater interaction and tidal cycling. Planning for a study to address these features is underway and will include a combination of geochemical, isotopic, and physical measurements.

APPENDIX

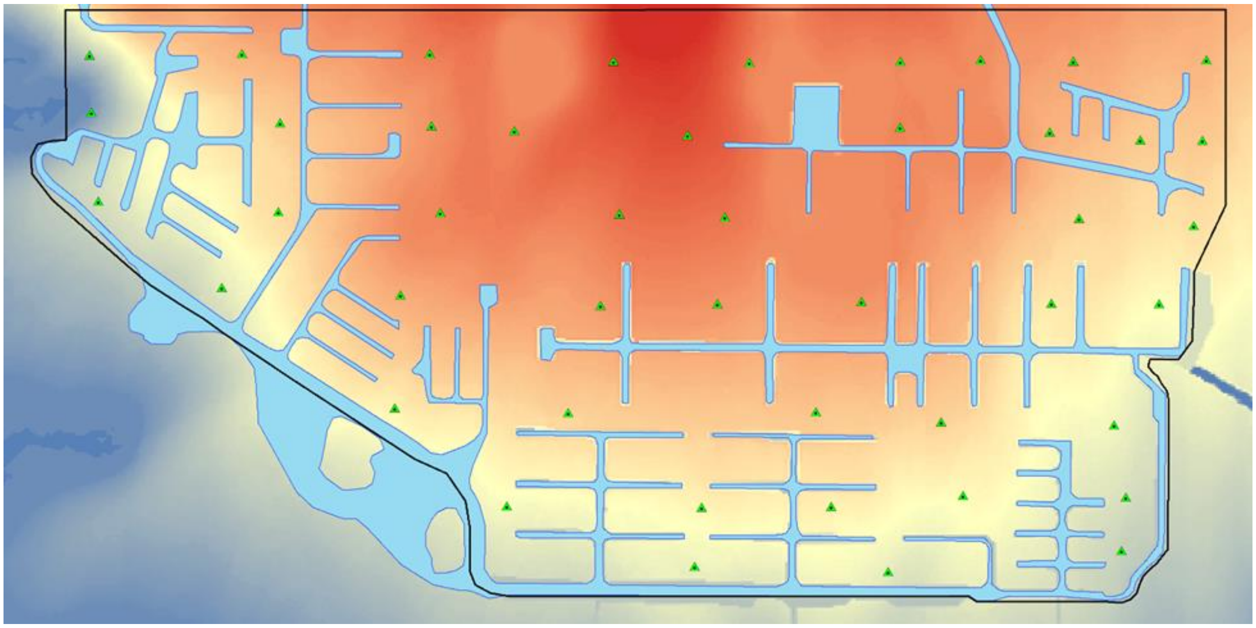

Figure 1 – Distribution of pilot points (green triangles) used during model calibration of the aquifer using PEST tool. The points were placed evenly over the model area.

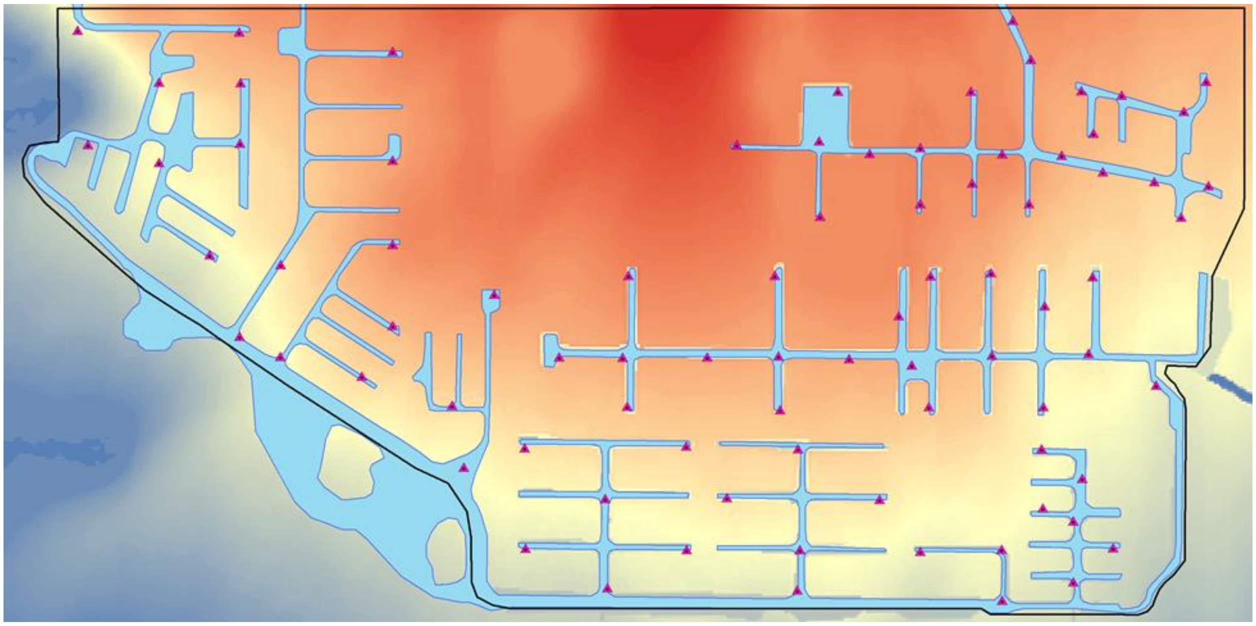

Figure 2 – Distribution of pilot points (pink triangles) used during model calibration of the canals using the PEST tool.

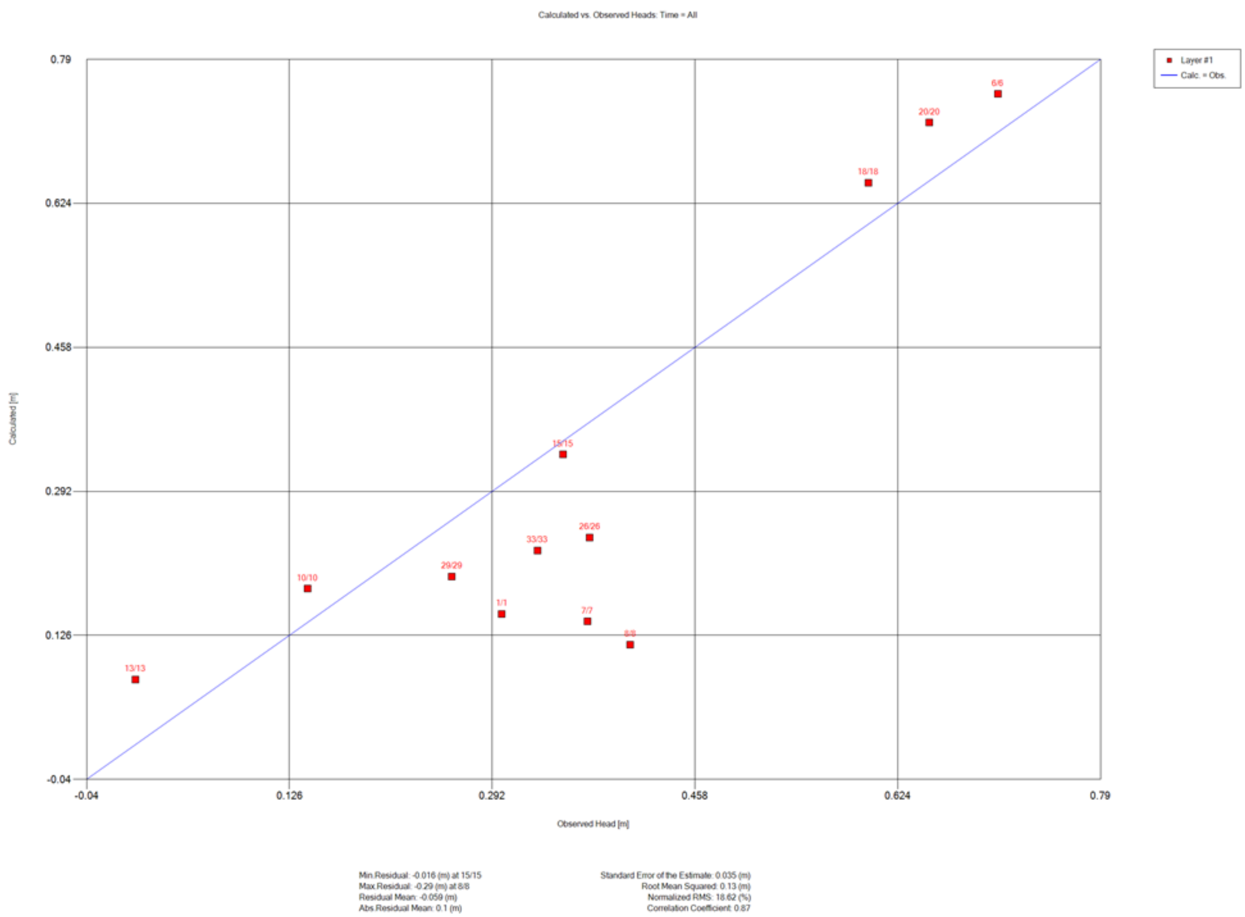

Figure 3 – A graph generated by Visual Modflow of the observed head values vs. the calculated head values.

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