Estimating the Influence of Onsite Wastewater Treatment Systems on Nutrient Loading to Falls Lake Watershed, North Carolina

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Executive Summary

This study approximated nutrient exports from onsite wastewater treatment systems (OWTS) in the Falls Lake watershed. Stream water quality data were used to evaluate the cumulative effects of onsite systems on adjacent surface water nutrient concentrations and loading. A literature review was conducted to document a range of past studies in Piedmont or similar settings that have directly measured nutrient exports from OWTS. A comparison of nutrient inputs to the soils and nutrient concentrations and exports from tributary streams helped to approximate the degree of nutrient attenuation from drainfields to surface waters. Stream water quality data during baseflow conditions were collected four times during this study between November 2019 and June 2020. The stream nutrient concentration data suggested that sub-watersheds that relied on OWTS were more likely to have elevated nutrient concentrations (particularly nitrogen) relative to sub-watersheds that relied predominantly on municipal wastewater treatment. A comparison of OWTS density and stream nutrient concentrations revealed a positive correlation between system density and nutrient concentrations. In addition to OWTS density, soil type and land-use were also shown to influence nutrient concentrations. Based on median nutrient loads from OWTS sub-watersheds (corrected by subtracting median loads from sewered sub-watersheds) divided by the number of sub-watershed residents, potential per capita onsite wastewater nutrient loads to streams were estimated. For nitrogen, it was estimated that between 39-100% of onsite wastewater nitrogen was attenuated and the median N transport to the streams was 1.07 kg-N/person/yr. The median estimate suggests that approximately 76% of the nitrogen from onsite systems is attenuated prior to stream discharge. For phosphate, there was greater attenuation between the onsite systems and the streams. It was estimated that the median per capita loading was 0.015 kg-PO_4-P/person/yr. This estimate suggests that approximately 99% mass attenuation occurs between the system and the streams, with a range of 68-100%. These estimates are based on loading measurements during baseflow conditions during the dormant season, however during a previous study a longer-term data set was collected between 2015-2019 for a sewered and an OWTS tributary in the Lick Creek watershed. This long-term paired watershed comparison for OWTS loading estimated that 0.94 kg-N/person/yr and 0.06 kg-PO_4-P/person/yr was transported from OWTS to the tributary. The estimate for N-mass attenuation was 79% and PO_4-P attenuation of 90%. Overall, these estimates were found to be comparable to other estimates found in the literature for Piedmont settings. To estimate loading across the entire Falls Lake watershed, data from previous studies were synthesized to approximate the influence of malfunctioning and alternative onsite systems. Earlier studies have suggested a 10% malfunction rate in North Carolina Piedmont settings. Field data from several recent studies suggest that malfunctioning systems may treat 26-85% of nutrient inputs. A first-cut approximation would be to consider those estimates for 10% of systems in the watershed and refine estimates as new data becomes available. In addition, nutrient exports from non-conventional OWTS, including sand filter and package plant systems, were estimated from several recent studies. Sand filters reduced nutrient concentrations by approximately 75%. Package treatment plants (extended aeration, advanced media filtration, and sequence batch reactors) reduced average TDN and PO4-P concentrations by 74% and 26%, respectively. These estimates are generally for system treatment and may not account for soil and aquifer treatment if the effluent is discharged to the land surface or subsurface. These initial estimates will be refined in an upcoming study which will include additional field data collection in the watershed. This information will support modeling efforts to approximate nutrient loading and evaluate the effects of onsite wastewater nutrient management on nutrient loading to the lake. Overall, these efforts to improve onsite wastewater nutrient loading estimates to Falls Lake will provide information that can assist with ongoing programs to reduce nutrient loading to the lake.

Introduction

Onsite wastewater treatment systems (OWTS) provide wastewater treatment to individual residences and businesses in rural and suburban areas across North Carolina. It is estimated that approximately 2 million systems currently exist across the state (modified from Pradhan et al. 2007). Numerous studies have shown that nutrient concentrations are typically elevated in residential wastewater and conventional septic systems are not designed for complete nutrient removal, therefore it is common for OWTS to leach nutrients to the groundwater system (Lusk et al. 2017). In hydrogeological settings where OWTS recharge the groundwater and the groundwater flowpaths provide groundwater baseflow to streams and direct groundwater inputs to lakes, OWTS have the potential to increase nutrient loading to surface waters. In the Falls Lake watershed, conventional OWTS (also known as septic systems) are the most common approach for wastewater management in rural and suburban communities that do not have access to municipal wastewater treatment. In an earlier study (Pradhan et al. 2007), it was estimated that approximately 75,000 people in the watershed rely on septic systems based on the 1990 Census data, however population has expanded rapidly in the watershed (e.g. Durham County increased from approximately 182,000 residents to 316,000 since 1990) since that estimate. NC DEQ (2009) estimated approximately 190,000 residents in the watershed in 2000 and based on the percentages that relied on septic systems (46%) in the watershed derived in the Pradhan et al. (2007) study, it is estimated that there are currently > 100,000 people reliant on septic systems for Falls Lake watershed. As each system has the potential to contribute nutrients to streams and ultimately Falls Lake, there is a growing need to quantify these non-point nutrient sources to the lake to evaluate their contributions to nutrient loading.

The effectiveness of nutrient treatment by OWTS is influenced by a number of variables including: soil type and hydrogeologic setting, waste characteristics and load, indoor water use, recent meteorological conditions, separation and setback distance, system density, presence of riparian buffers, system type and size, biomat formation, greywater, garbage disposals, system age, system maintenance, and system malfunctions (Beal et al. 2005, Humphrey et al. 2010, O'Driscoll et al. 2014, Withers et al. 2014, D'Amato et al. 2016, and Lusk et al. 2017). Based on the wide range of influences on nutrient treatment variability and nutrient transport to streams, detailed site-specific studies are called for to estimate nutrient loading from OWTS based on the local conditions.

Nutrient exports from onsite wastewater systems can vary by site due to the range of nutrient transport and attenuation mechanisms that can occur. Residential wastewater typically has elevated nitrogen and phosphorus concentrations (Lowe et al. 2009). Conventional OWTS generally utilize a septic tank, drainlines, and a drainfield underlain by a soil treatment unit (**Figure 1**). The tank is utilized for storage, anaerobic treatment and containment of solids, and drainlines convey the liquid effluent to a drainfield where additional treatment occurs in the unsaturated soils, also referred to as the soil treatment unit or STU. In the STU, wastewater contaminants such as nutrients and bacteria, can be reduced as they interact with the soils. The wastewater effluent then recharges the groundwater system underlying the drainfield. If treatment is inadequate, then the remaining contaminants in the effluent can be transported via groundwater to adjacent surface waters.

Nitrogen transformations can occur in the various components of the OWTS. Typically, there are anaerobic conditions in the septic tank and wastewater nitrogen is composed of organic N and ammonium nitrogen ($NH4^+$) (US EPA, 2002, Lusk et al. 2017). Organic N in wastewater in the septic tanks is usually mineralized to ammonium nitrogen and then, in the unsaturated drainfield soils (STU), the ammonium is transformed to nitrate via nitrification. In locations where clay soils are present, some of the ammonium may be adsorbed to soils. In addition, plant or microbial uptake can occur that will attenuate some of the nitrogen exports from the system. Typically, if unsaturated soils are present the majority of nitrogen leaching from the drainfield will be in the form of nitrate, which is mobile in groundwater and can leach

to local surface waters (Iverson et al. 2018). However, in biomats (a biologically active clogging zone that forms directly beneath the drain lines), soils, aquifer materials, riparian zones, and hyporheic zones (zone of surface water-groundwater interaction underlying streams), nitrate concentrations between the drainfield and adjacent surface waters can be reduced via denitrification (Humphrey et al. 2016a?). Denitrification can lead to nitrate concentration reductions in the groundwater or surface water as a result of the microbial conversion of nitrate to dinitrogen gas (N^2) that can be released to the atmosphere (**Figure 2**). Denitrification is favorable in settings with anaerobic conditions, where nitrate, organic carbon sources, and denitrifying microorganisms are present (Humphrey et al. 2016a).

Phosphorus in onsite wastewater can be transformed or retained via several mechanisms (**Figure 3**). Phosphorus concentrations in the septic tank can be reduced by solids settling and precipitation with iron (Lusk et al. 2017). After discharging from the tank, phosphorus concentrations in septic tank effluent can be reduced by a variety of processes in the STU and the surficial aquifer (**Figure 3**), including mineral precipitation, adsorption, and immobilization (Lusk et al. 2017, Humphrey et al. 2016b). Precipitation may occur when phosphate (PO_4) forms solid minerals with cations in the soil (Humphrey et al. 2016b). Adsorption occurs when phosphate binds to surface cations attached to soil particles. However, desorption and dissolution of phosphate precipitates can also occur making retention by subsurface materials not necessarily permanent (Lusk et al. 2017). Immobilization of phosphate may occur via uptake by riparian vegetation down-gradient from OWTS. The range of mechanisms mentioned have the potential to reduce phosphate exports from onsite systems. Recent work by Humphrey et al. (2016b) in the Falls Lake watershed showed that for conventional systems with 35 m setbacks to sampling wells, phosphate concentrations in groundwater were similar to background conditions. However, sand filter systems did contribute elevated phosphate concentrations to local creeks.

In addition to conventional OWTS (septic systems), several other types of OWTS exist in the Falls Lake watershed including sand filter systems, TS-II, and package plant systems in areas where access to municipal wastewater treatment is unavailable. These alternative OWTS can provide additional nutrient treatment based on their design and operation. Sand filter systems provide sand for filtration in areas where the water table may be shallow or low permeability soils or bedrock can prohibit vertical infiltration and treatment of the wastewater. In the Falls Lake watershed these systems exist in the Triassic Basin region of the watershed due to the presence of expansive clay soils (Humphrey et al. 2016a, 2016b). Typical components of sand filter systems include a septic tank, a sand filter bed, and an effluent discharge pipe (**Figure 1**). The wastewater effluent from the tank percolates through the sand filter and then drains to the effluent discharge pipe and discharge to the land surface or adjacent surface waters. Single-pass sand filter systems only filter the effluent through the sand filter a single time as the effluent percolates to the effluent discharge pipe. As an alternative, more advanced systems can improve treatment by recirculating the effluent through the sand filter. The filtered effluent can be cycled through the filter multiple times with every pass improving the effluent quality (recirculating systems). Relative to conventional systems, there is generally less information available on the treatment effectiveness of sand filter systems in Falls Lake watershed. However, recent work by Humphrey et al. (2016a, 2016b) documented that single-pass sand filter systems were less effective at nitrogen and phosphorus treatment than conventional systems. In addition, Iverson et al. (2018), included data on several surface water bodies in Durham Co. influenced by onsite wastewater nutrient transport from sand filter systems and Tetra Tech (2013a) provided effluent nutrient concentration data for a number of sand filter systems in Durham Co.

TS-II systems are advanced wastewater pretreatment systems that can treat wastewater to standards defined in 15A NCAC 18A .1970 (2011). These systems will treat effluent to have less than 10 mg/l of carbonaceous biochemical oxygen demand, total suspended solids, and ammonium nitrogen

and less than 1000 colonies of fecal coliform/100 ml. In addition, total nitrogen concentrations of either less than 20 mg/l or 60% of the influent concentrations. These systems included advanced media filtration systems such as AdvanTex (Orenco Systems, Inc.), sequencing fixed reactor systems such as FAST (BioMicrobics, Inc.), trickling filter systems such as SeptiTech STAAR™ Filter Systems (BioMicrobics, Inc.), and others (https://ehs.ncpublichealth.com/oswp/approvedproducts.htm).

For small communities, commercial developments, and individual properties, package treatment plants (PTPs) are often found in areas where insufficient disposal area exists for the load and access to a centralized municipal wastewater treatment plant is unavailable (US EPA 2000). These facilities are typically designed to treat flows ranging from 7500 to 1.9×10^6 LPD (2000−500,000 GPD) (US EPA 2000). PTP effluent is typically discharged onsite either via surface application or subsurface drainfields. Depending on the wastewater load and dispersal approach (surface vs. subsurface), these systems may be regulated by county health departments (for smaller systems and subsurface dispersal) or by NC DEQ (larger systems and surface application). State monitored systems can be found at: https://ncdenr.maps.arcgis.com/apps/webappviewer/index.html?id=87eddfd9d45a40a3999c3bba4d7e0eed

Several earlier studies (Tetra Tech 2013a, Hazen and Sawyer 2013) utilized literature review values to estimate the effects of onsite wastewater on nutrient loading to the lake. This current study builds on that earlier work and aims to refine those estimates based on recent field data and recently published research focused on Falls Lake and other southeastern Piedmont watersheds . The current study objectives were to collect and synthesize surface water, groundwater, and wastewater nutrient data to: 1) determine the influence of OWTS on nutrient concentrations and loading in the Falls Lake watershed and 2) to collaborate on model development and validate watershed models developed for the Upper Neuse River Basin Association (UNRBA).

Research Questions

1. Do onsite wastewater treatment systems cause elevated nutrient concentrations in streams draining to Falls Lake?
2. On a per capita basis, approximately how much nitrogen and phosphorus from onsite wastewater treatment systems drains to tributaries in the Falls Lake watershed?

Research Methods

The primary research questions were addressed using multiple lines of evidence obtained from a literature review, GIS analysis, and field monitoring of tributaries in the Falls Lake watershed. The literature review of onsite wastewater nutrient loading was focused on research conducted in the North Carolina Piedmont, and Piedmont regions in other southeastern states. Published studies that focused on wastewater and wastewater treatment in similar settings with clay to loam soils were also considered for conventional OWTS (septic systems). In addition, data and literature were reviewed for sand filter systems, TS-II systems, and PTPs.

Sampling locations were selected based on watershed and septic system characteristics of sub-watersheds draining to Falls Lake. Sub-watersheds were delineated using the USGS *StreamStats 4* (https://streamstats.usgs.gov/ss/) program. Land cover percentage for each sub-watershed were estimated based on a report compiled from *StreamStats 4* using the 2011 National Land Cover Database. Aerial photography was retrieved using NC OneMap (nconemap.gov), which is a publicly available database. Geological data of North Carolina were retrieved from the USGS (mrdata.usgs.gov), which were used to select sub-watersheds in differing geological settings in the Falls Lake Watershed. Geologic information was summarized by the dominant rock type based on USGS data. Dominant soil series and texture for

each sub-watershed were determined by integrating watersheds delineated from *StreamStats 4* into the USDA's Web Soil Survey (websoilsurvey.nrcs.usda.gov). Septic system data were acquired from Durham County and Wake County to generate a centroid point shapefile of parcels served by septic systems based on best available data. Septic system density was calculated by dividing the number of septic systems in each sub-watershed by the area of that sub-watershed (**Table 1**). Land cover data is included in **Table 2**.

The field monitoring component focused on evaluating nutrient concentrations and streamflow across a range of hydrogeological settings in the watershed (Carolina Terrane, Triassic Basin, and Falls Lake/Crabtree Terrane **(Figure 4)**. This field component was needed because models may not capture the nutrient attenuation between the drainfields and surface waters that can occur due to a variety of treatment processes in the soils, regolith, and aquifer materials along groundwater flowpaths to the adjacent surface water. Sub-watershed water quality data can help to show the cumulative effect of onsite systems on downstream water quality. At the onset of the study, site selection was conducted to identify small sub-watersheds (< 159 ha) that had a range of septic system densities (0.18-3 systems/ha; **Figure 5** and **Table 1**). Detailed maps with approximate septic system locations for each watershed are provided in **Appendix A** and site pictures are located in **Appendix B**. The highest septic system density was observed at the Jones Circle ditch site (Durham Co.), this sub-watershed relies on sand filter systems and the ditch drains shallow groundwater and surface water to Lick Creek (**Figure A-4**). In addition, 3 stream sites (Clover Hill, Ashburn, and Baileywick Park) in sub-watersheds that relied on municipal wastewater treatment plants were selected for comparison with streams draining watersheds that relied predominantly on OWTS. Although the sub-watershed for the stream at Baileywick Park had numerous septic systems, these were generally located near the uplands and the park provided an extensive buffer between OWTS properties and those served by municipal wastewater treatment. Sites were sampled during baseflow conditions on 11/15/2019, 12/18/2019, 2/10/2020, and 6/02/2020* (*due to lab closure associated with COVID-19, 6/02/2020 samples were collected and frozen for analyses when the lab reopens). At each site, direct readings were taken in the stream using a calibrated YSI multiprobe (model 556) to measure water temperature, specific conductance, pH, oxidation-reduction potential, and dissolved oxygen. Streamflow was estimated using a *Global Water* flow probe (model FP111) and channel dimensions were measured using a tape measure. Water samples were collected from streams and stored on ice, then delivered to the ECU Water Resources Lab on the same day. Samples were filtered within 24 hours and frozen for preservation prior to nutrient analyses at the ECU Environmental Research Lab (ERL). Water quality analyses at the ERL included NH_4, NO_3+NO_2, DKN, PO_4, TP, TDP, Cl, DOC, and TDN. Samples collected on 12/18/2019 were also sub-sampled in 50 ml plastic vials, filtered, then frozen and sent to UC-Davis Stable Isotope Facility for N-15 in NO_3 analyses. An additional round of isotopic samples were collected for isotopic analyses on June 2, 2020 and will be analyzed when the lab reopens after the COVID-19 closure.

To approximate the annual loading of onsite wastewater nutrients to the drainfield, several assumptions were necessary. It was assumed that recent average indoor water use estimates (DeOreo et al. 2016) approximated wastewater discharges to the subsurface. These estimates per capita (55.2 gallons/person/day or 209 liters/person/day) were scaled up to each individual sub-watershed based on the number of residents in the sub-watershed (estimated from residences and the Durham and Wake County average of 2.5 people per residence obtained from the US Census Bureau for 2019). Approximate nutrient loading to the soils was estimated using indoor water use estimates, wastewater nutrient concentrations were estimated from recent studies in the watershed (Humphrey et al. 2016 a,b) along with literature review values from a range of studies (**Tables 3 and 4**). Attenuation of nutrients between the septic tank and the soil treatment units (prior to groundwater recharge) was estimated using the

median literature review values found in field, lab, column, or numerical modeling studies in Piedmont or similar settings (**Tables 3 and 4**).

Nutrient exports from sub-watershed streams were estimated by multiplying the stream nutrient concentration by the stream discharge. To approximate onsite wastewater nutrient loading to the streams, median nutrient loads in sewered sub-watersheds were subtracted from the median loads for septic watersheds, this load represents the potential nutrient load contributed by onsite wastewater and can be used to approximate onsite wastewater nutrient loading to the streams. After estimating the nutrient exports at the sub-watershed outlet on the sampling date, the median export value (kg/yr) was divided by the number of septic systems in the watershed (loading per system) and then divided by 2.5 people per household (Durham/Wake County average) to quantify the per capita loading.

Initial Findings

Do onsite wastewater treatment systems cause elevated nutrient concentrations in streams draining to Falls Lake?

Streams draining septic sub-watersheds had elevated total dissolved nitrogen (TDN), phosphate (PO_4-P), and chloride (Cl) when compared to predominantly sewered sub-watersheds (**Figure 6**). These data suggest that nutrients and chloride from onsite wastewater is leaching to the shallow groundwater and a fraction of the inputs are translated to the local streams. As a whole the dataset suggests that streams that are draining sub-watersheds that rely predominantly on OWTS have a greater likelihood to have elevated nutrient concentrations. More detailed streamflow data is needed to evaluate if increased discharge occurs in these watersheds associated with recharge from OWTS, this factor may also influence nutrient loading and other studies have suggested that onsite wastewater inputs can increase local groundwater recharge and stream baseflow (Iverson et al. 2015; Hoghooghi et al. 2017).

Elevated TDN concentrations occurred for sub-watersheds with greater septic system densities **(Figure 7**), however the highest concentrations were found at the mid-level densities, related to high concentrations at the Passmore and Park Ridge sites. Patterns were not as clear for PO_4-P (**Figure 8**), possibly due to the lower mobility of PO_4-P in clay-rich soils and aquifer materials (Humphrey et al. 2016b). Chloride is typically elevated in wastewater and can serve as a conservative tracer. A comparison of chloride vs. septic system density suggested that sewered watersheds tended to have lower chloride concentrations (**Figure 9**). There were two dates (12/18/2019 and 2/10/2020) when the stream at the Ashburn (0.18 systems/ha) site had slightly elevated chloride concentrations, possibly because of the use of road salt during winter months. Nitrate was the dominant nitrogen form in streams and several sub-watersheds containing onsite systems were found to have elevated stream nitrate concentrations, with 3 sites (Jones-ditch, Passmore, and Park Ridge) that had stream nitrate-N concentrations greater than 10 mg/l (nitrate-N drinking water standard) on at least one occasion (**Figure 10**). In general, the sewered sub-watersheds and the sub-watershed with < 0.5 systems/ha (Rondelay, which also has a portion of the sub-watershed that is sewered) had the lowest stream nitrogen concentrations. Although other watershed nutrient sources may also contribute nitrogen, $^{15}N\text{-}NO_3$ data suggest that the nitrate sources are likely wastewater, because sewage and manure nitrate sources typically fall within the range of 7-20 $\delta^{15}N\text{-}NO_3$ (‰) (Silva et al. 2002, Xue et al. 2009) (**Figure 11**). For this study, all of the septic watersheds except Harold had > 7 $\delta^{15}N\text{-}NO_3$ (‰). These data suggest that wastewater is a source of some portion of the stream nitrate delivered to the OWTS sub-watersheds. Future work will include more N-15 sampling in these sub-watersheds. In general, sites with elevated total dissolved nitrogen, generally also had elevated PO_4-P concentrations. However, the Passmore and Park Ridge sites had highly elevated TDN but did not show similar elevated concentrations of PO_4-P (**Table 5**), these sites also appear to be affected by legacy

nutrients associated with historical agricultural land-use (cropland) directly upstream (**Table 2**) and lawn fertilizer, based on a review of the recent and historical aerial photos and land cover data.

Overall, the results indicate that onsite wastewater nutrients can be translated to streams, but the influence on stream nutrient concentrations may vary due to a range of factors including differences in wastewater loading, system density, variations in system type, soil, aquifer, and in-stream nutrient attenuation, the effectiveness of riparian buffers, hydrogeological setting, and stream-groundwater interactions. These and other factors may result in variability in onsite wastewater nutrient loading to streams. Other studies in Piedmont settings such as Atlanta, Georgia (**Figure 12**) (Hoghooghi et al. 2016) have shown strong correlations between septic system density and dissolved nitrogen concentrations in baseflow. These patterns suggest that septic system density can be used to identify sub-watersheds with greater likelihood of nutrient impairment. For the current study, septic system density explained 42% of the variability in stream nitrogen concentrations and for the earlier Lick Creek study it explained 46% of variability, in the Atlanta study they found that in high OWTS density watersheds OWTS density could explain 59% of the variability in stream nitrogen concentrations (Hoghooghi et al. 2016).

On a per capita basis, approximately how much nitrogen and phosphorus from onsite wastewater treatment systems drains to tributaries in the Falls Lake watershed?

Numerous studies have focused on international (Beal et al. 2005; Withers et al. 2014), national (McCray et al. 2005; McCray et al. 2009; Lowe et al. 2009; Lusk et al. 2017), and regional literature reviews (Pradhan et al. 2007; Hazen and Sawyer 2013; Tetra Tech 2013; and D'Amato et al. 2016) of onsite wastewater characteristics and nutrient attenuation by drainfield soils. More recently, several studies were conducted in the Falls Lake watershed that focused on onsite wastewater nutrient transport to Piedmont soils and streams (Humphrey et al. 2016a, 2016b; and Iverson et al. 2018). In addition, a number of recent studies have characterized onsite wastewater nutrient transport in Piedmont settings in Georgia (Bradshaw and Radcliffe, 2013; Radcliffe and Bradshaw, 2013; Hoghooghi et al. 2016). These studies were used in **Tables 3** and **4** to provide estimates of onsite wastewater nutrient loading and attenuation.

Onsite wastewater nutrient loading to the STU was approximated by utilizing water use, wastewater nutrient concentration, and nutrient attenuation factors from published studies, as described in the methods section. Additional attenuation along groundwater flowpaths was approximated from field studies by Humphrey et al. (2016a). **Tables 3** and **4** document the estimates for nitrogen and phosphorus exports to streams from conventional OWTS, respectively. **Table 5** documents estimates of potential onsite wastewater nutrients to streams based on the sub-watershed water quality and flow data collected during this study.

The nutrient concentration and stream discharge data collected during this study was used to quantify potential onsite wastewater nutrient contributions to streams. Median nitrogen loads from sewered sub-watersheds were used to estimate the potential nutrient exports not associated with onsite wastewater (potentially related to lawn fertilizer, soil organic matter, etc.). The median sewered sub-watershed loads of TDN (0.38 kg-N/ha/yr) and PO_4-P (0.11 kg-P/ha/yr) were subtracted from the median loads in the onsite wastewater treatment watersheds to provide an estimate of the potential OWTS contributions. These corrected loads were used to estimate the median baseflow nutrient exports at each site potentially contributed by OWTS. The potential nutrient exports associated with OWTS were quantified by dividing the potential onsite wastewater nutrient exports (median) in the streams by the number of people using OWTS in the sub-watersheds.

Per capita potential onsite wastewater nitrogen loading from sub-watersheds ranged from 0.07 kg-N/person/yr at Rondelay, to 2.7 kg-N/person/yr at Asbury (**Table 5**). The data for Passmore and Park Ridge sites had even greater nitrogen loads (30.2 and 7.7 kg-N/person/yr., respectively) but other data suggest that additional nitrogen sources are contributing to elevated loadings at these sites, therefore the calculated loadings do not likely represent strictly onsite wastewater nitrogen loading (further study is needed to evaluate where the elevated nitrogen inputs are occurring and their sources). For these two sites, the estimates based on literature values of inputs to the soils were assumed to better approximate the potential onsite wastewater nitrogen loadings at these sites (1.8 kg-N/person/yr; **Table 3**). These would be conservative estimates for these two sites, assuming minimal attenuation in the surficial aquifer. For all of the streams in the current study, the median per capita loading was estimated at 1.07 kg-N/person/yr. Based on estimates for loading to the soils (4.4 kg-N/person/yr) from literature values in Table 3, this suggests that a median 76% mass attenuation occurs between the drainfield and the stream with a range of 39-100%. Based on earlier studies in the NC Piedmont by Humphrey et al. (2016a) and Iverson et al. (2018), ranges from 63-92% were estimated in field studies with a median value of 77%. Other studies on larger watersheds in the region (Berkowitz et al. 2014) suggest greater attenuation (>95%), which may be explained in part by lower septic system densities in larger watersheds due to the greater likelihood of open space and other non-residential land-use and greater potential for in-stream attenuation and dilution by older groundwater as watershed-scale increases. In addition, the distance of systems to surface waters and presence of riparian buffers likely influences the nutrient attenuation.

Per capita potential onsite wastewater PO_4-P loading from sub-watersheds ranged from 0 at several sites to 0.19 kg-PO_4-P/person/yr at Asbury (**Table 5**). The median per capita loading was estimated at 0.015 kg-PO_4-P/person/yr. Based on estimates for loading to the soils (0.62 kg-PO_4-P/person/yr) from literature values in Table 4, this suggests that approximately 99% mass attenuation occurs between the drainfield and the stream with a range of 68-100%. Based on earlier studies in the NC Piedmont by Humphrey et al. (2016a) and Iverson et al. (2018), ranges from 92-98% were estimated in field studies with a median value of 94%.

When comparing sites based on loading per watershed area, several sites showed elevated nitrogen loading and these typically occurred for settings where the septic system density was > 1 system/ha (**Figure 13)**. However, it is important to mention that Macon and Brookfield sites had higher septic system densities but lower N- exports. As mentioned earlier, there are other variables besides septic system density that influence the potential for onsite wastewater nutrients to reach streams. For example, when per capita loading was compared between sub-watersheds that had a greater extent of sandy loam soils (30% or more of the sub-watershed area) vs. those sub-watersheds that had extensive clay/silt loam soils, the sites with sandier soils were more likely to have elevated nutrient exports, suggesting that nutrient attenuation is less effective in areas with sandy soils (**Figure 14**). However, some of the sandy loam surface soils, such as the Pacelot and Cecil series have a shallow sandy loam layer (~ 8 inches thick) that is frequently underlain by clay or clay loam subsoils. These clay-rich soils, when present at depth (where drainlines would typically be located), could be more effective at nutrient attenuation. However, when more permeable sandy soils overlay less-permeable clay-rich soils, these settings may also be vulnerable to lateral flows at the clay-sandy loam interface, which could result in transport of nutrients to adjacent surface waters. Although these data suggest that the presence of sandy soils may result in lower attenuation, more work is needed to quantify the variability of soil characteristics with depth and the influence on nutrient attenuation and stream-groundwater interactions in the Falls Lake watershed.

Current or prior land-use influences on nutrient loading were also noticeable, for example based on the 2011 data, crop cover showed an influence on the nutrient loading, particularly N-loading in sub-watersheds with > 10% cropland (**Table 2; Figure 15**). The Park Ridge and Passmore sites had the

highest nitrogen loads and also the highest percent of land-use in cropland in 2011. A review of past land-use data and aerial photos showed the presence of a large farm upstream of Passmore Court that was converted to residential development in the mid-2000s. In these sub-watersheds, much of the cropland and pastureland has been converted to residential development, these stream nitrogen data suggest there may be a legacy effect. In addition, a review of the aerial photography suggested that some of the residences may use lawn fertilizer which could also contribute to loading downstream. Due to the declines in nutrient concentrations between Passmore and Park Ridge, the data suggest the main nutrient sources are located upstream from the Passmore site. These data suggest that future work should consider the influence of watershed soil characteristics, current land-use, and prior land-use history to improve understanding of nutrient exports to streams in the Falls Lake watershed.

During a preceding study, streamflow and nutrient concentration data was collected at a sub-watershed with a high density of septic systems (HD1-1.86 systems/ha) and a sub-watershed that was sewered (**Figure 16**) in Durham County (Lick Creek watershed) from 2015-2019. This dataset provides a longer-term view of the potential transport of nutrients from OWTSs to tributaries of Falls Lake. The nitrogen and phosphate exports from the high density (HD) sub-watershed were compared to those from a sub-watershed that utilized a municipal wastewater treatment system (outside of the sub-watershed). These comparisons revealed that N-loading was significantly elevated for the sub-watershed with onsite wastewater treatment, suggesting that a fraction of the nutrients in wastewater were migrating to the stream (**Figure 16**). Similarly, PO_4-P exports from the onsite wastewater sub-watershed were elevated relative to the sewered watershed (**Figure 17**). These data suggest that nutrient exports from the onsite watershed were greater than those from the sewered watershed. The difference in exports can be used to approximate the nutrient exports from onsite sub-watersheds. The median difference between the onsite sub-watershed and the sewered watershed over the 2015-2019 monitoring period was 2.35 kg-N/ha/yr (TDN) and for 0.21 kg-P/ha/yr (PO_4-P). Based on Mann-Whitney tests the difference in exports between the sewered and septic sub-watersheds were significantly different at $p=0.002$ and 0.0001, respectively. Based on the number of systems and people in the sub-watershed, it was estimated that the per capita potential N-loading from onsite wastewater was 0.94 kg-N/person/yr and PO_4-P of 0.06 kg-PO_4-P/person/yr. The estimate for N-mass attenuation was 79% and PO_4-P attenuation of 90% based on these long-term data and this is within the range of the earlier reported studies mentioned above. This is likely the most reliable estimate because of the large number of loading data points across a range of hydrological conditions over time (n=52 per stream). These data also suggest that there is seasonal variability in attenuation, with generally greater attenuation and lower loading in the summer months and reduced attenuation during the late fall, winter, and early spring (dormant season). In addition, the greatest loads from the OWTS watershed often corresponded with higher flow events during the dormant season. These data suggest that more data is needed during storm runoff events, particularly during the dormant season to improve estimates of onsite wastewater-related nutrient loading.

Earlier work by Tetra Tech (2013a) provided estimates for functioning systems of 0.15 kg/person/yr for N and 0 for P. A recent study by D'Amato et al. (2016) based on literature review and modeling approaches for the Chesapeake Bay watershed estimate for N loading from OWTS to streams ranged from 0.8-2.1 kg-N/person/yr, however this approach did not account for riparian buffer or hyporheic zone attenuation. The most recent work focused on Falls Lake tributaries in Durham County used stream water and wastewater quality and loading data to estimate onsite wastewater nutrient loading at 0.6 kg-N/person/yr and 0.041 kg-PO_4-P/person/yr (Iverson et al. 2018).

This earlier study focused on the Lick Creek and Little Lick Creek watersheds in Durham Co., streamflow and stream nutrient concentration data were collected from 2015-2016. Those data are included for comparison with the data collected during the current study in **Table 5**. Although those

earlier data showed generally lower per capita N and P loads to the stream, the sewered and forested watersheds used for controls generally had higher nutrient concentrations than the control watersheds for the current study, therefore these estimates may be more conservative. However, this data set included a larger number of samples throughout the year, so may provide a better representation of annual loads. Future work will aim to continue sampling and expand to further locations to refine these estimates.

Our results based on sub-watershed stream data are in general agreement with these previously documented estimates (**Table 5**). From the current study, the data for PO_4-P suggest a greater input than 0 for sub-watersheds with greater septic system densities (>0.5 systems/ha) and recent work by Humphrey et al. (2018b) also suggests that there is evidence for some OWTS phosphate transport to streams in this region. Since the earlier estimates (Tetra Tech 2013a) were for functioning systems, the range of values in the current study can also indicate that not all conventional systems in the study sub-watersheds are properly functioning. In addition, as indicated between the upstream Passmore site and the downstream Park Ridge site, variable rates of in-stream nutrient attenuation can also contribute to differences in stream nutrient concentrations. These results (**Table 5**) and earlier work in these settings suggest that attenuation for PO_4-P is generally larger than for N (Humphrey et al. 2016a,b) **(Figures 18 and 19)**. Overall, the median estimates from this current study (1.07 kg-N/person/yr. and 0.015 kg-PO_4-P/person/yr.) fall within the range of reported values in similar settings.

Table 6 documents additional information for approximating exports from non-conventional OWTS, including sand filter, type II, and package plant systems. Although these systems may have greater loads, they are also fewer in number. Future work will aim to quantify the locations and types of these systems located in the Falls Lake watershed. Alternative OWTS reduced TDN and PO_4-P concentrations by 66% and 77%, respectively, on average. Two previously researched single pass sand filters (Humphrey et al. 2016a, 2016b) reduced TDN and PO_4-P concentrations by 81% and 90%, respectively, on average (**Table 6**). These systems exhibit variability in nutrient treatment efficiencies between systems. Site 1 had a greater treatment efficiency with mean effluent concentrations of 3.80 and 0.11 mg L^{-1} for TDN and PO_4-P, which was approximately 75% and 90% lower than sand filter effluent from Site 2 (TDN: 15.47 mg L^{-1}; PO_4-P: 1.07 mg L^{-1}). A recirculating, type II sand filter system was also sampled and found to reduce TDN and PO_4-P concentrations by 79% and 68%, respectively. PO_4-P treatment efficiency was lower than other septic systems; however, influent PO_4-P concentrations were approximately 50% lower than wastewater entering other system types, and sand filter effluent contained PO_4-P concentrations < 1 mg/L. TDN concentrations in effluent from this system were similar to the TDN mean from single pass sand filters (**Table 6**). An earlier study on package plants (Carteret Co., NC) found that plants reduced TDN and PO_4-P concentrations by 74% and 26%, respectively (Mahoney 2016). The sequencing batch reactor type package plants had the greatest nutrient reductions, whereas the Advantex systems were the lowest; however, the difference in nutrient treatment efficiencies between these two technologies was relatively small (TDN: 6%; PO_4-P: 15%; **Table 6**). NC DEQ provided effluent data for 4 package plants in the Falls Lake watershed, however since influent data was not collected the treatment efficiency could not be calculated. It should be noted that the mean effluent nitrogen concentration for the package plants in the Falls Lake watershed was substantially larger than for the Carteret Co. systems (24 mg/l vs. 12 mg/l, respectively), whereas the mean phosphorus concentration for the Falls Lake systems was lower relative to the Carteret Co. sites. Relative to other alternative OWTS, package plants contained similar TDN treatment efficiencies, but PO_4-P concentration treatment efficiencies were approximately 50% lower on average. These estimates are generally for system treatment and may not account for soil and aquifer treatment if the effluent is discharged to the land surface or subsurface.

In addition to the range of treatment based on system-type, there is also variability of treatment within systems due to system functionality. D'Amato (2014) and Tetra Tech (2013a) provided an overview of the challenges of accounting for nutrient transport from malfunctioning systems. Malfunctioning systems can exist across a spectrum, from untreated wastewater upwelling to the surface and receiving minimal treatment to a reduced treatment efficiency associated with reduced separation distance between the drainlines and groundwater table. Generally, the surface failures are more likely to be detected. There have been several studies in the North Carolina Piedmont region that document malfunctioning OWTS from 6-10% in the Jordan Lake watershed (Tetra Tech 2013b), and a broader study of 286 systems across the NC Piedmont where the failure rate was estimated at 9.8% (Uebler et al. 2016). A smaller scale study in Orange County, focused on Rogers – Eubanks Area , found that 27% of systems were malfunctioning (Orange County Health Department, 2010). Earlier work for Falls Lake assumed a 15% malfunction rate (NC DENR 2009). Although these studies generally document surface hydraulic failure, limited information is available on subsurface failure and inadequate nutrient treatment.

A conservative approach that will likely overestimate the impact of failing systems is to consider no soil treatment and estimate the load from systems by using the concentration of nutrients in the tank and multiplying that by the household indoor water use to estimate a load if all wastewater were directly discharged to the stream (4.4 kg-N/person/yr; 0.6 kg-P/person/yr). This may occur periodically during wet and cool periods but is less likely during warm summer months when the water table is generally deeper and vegetation is growing and taking up water and nutrients. At surface failing systems, since the upwelling wastewater effluent will typically have some interaction with the shallow soils and vegetation, it is likely that some level of nutrient attenuation occurs as that waste migrates to the stream. Treatment may vary based on the distance to the stream, water use and flow rate/residence time, the capability for downgradient soils to allow re-infiltration, soil moisture, stormflow vs. baseflow conditions, site slope, the amount of evapotranspiration, downgradient potential for denitrification, the presence of downgradient riparian buffers, and other factors that influence surface runoff and nutrient retention. Based on the Uebler et al. (2016) study, a 10% malfunction rate could be considered for an initial estimate.

During an earlier study on Lick Creek, a malfunctioning system was found to treat ~ 30% of the phosphate and 26% of the nitrogen (**Table 6**). That system was experiencing a hydraulic malfunction because effluent was flowing across the land surface and spilling into a creek. Samples were collected from the initial point of surfacing and compared to samples collected at the creek to determine concentration reductions. At a different site within the same watershed, groundwater discharge at a seep down-gradient from a conventional-style system was sampled and compared to nutrient concentrations in wastewater from the septic tank serving that property. There was an 85% reduction in the concentration of TDN, and a 75% reduction in the concentration of PO_4-P in water sampled from the seep in comparison to wastewater in the tank. Most of the TDN was comprised of NH_4^+, followed by DON, and NO_3^-, indicating that conditions necessary for complete nitrification were not observed beneath the system, likely due to an elevated water table. In an earlier study focused on seep inputs to streams, a system in Craven County was found to have wastewater from an upgradient system upwelling to a seep which flowed to a local creek. At the seep site, a 79% reduction in TDN was observed between the wastewater tank and the seep, whereas for groundwater that flowed in the subsurface (as measured in riparian buffer wells) was shown to have a 93% TDN reduction (O'Driscoll et al. 2019). At this site there was approximately a 14% decline in nitrogen treatment associated with the wastewater-affected groundwater upwelling to the surface. These examples provide some insight into the potential nutrient contributions to surface waters from onsite systems experiencing hydraulic malfunction and subsurface treatment failure. A third type of failure also observed in the Lick Creek watershed was discharge of grey water

from homes to adjacent waterways. Samples from a flowing grey water pipe revealed TDN and PO_4-P concentrations of 4.89 mg L^{-1} and 1.22 mg L^{-1}, respectively. These examples provide some evidence of the nutrient contributions from failing onsite systems to surface waters.

Connecting grey water pipes to existing septic tanks and drainfields may help reduce nutrient loads to surface waters, but will increase hydraulic loads to the systems, eventually resulting in a need to upgrade the existing drainfields. Installation of additional or replacement drainfield trenches may help improve nutrient treatment at sites with failing systems. For example, a replacement drainfield was installed at a site with a hydraulic failure in the Lick Creek watershed. Lysimeters were installed below the new drainfield trenches. Soil water samples collected from the lysimeters showed DIN and PO_4 concentrations were 87% and 99.6% lower respectively than wastewater concentrations (Humphrey et al., 2019) (**Table 6**). For that site, there was available space to accommodate new drainfield trenches. Some properties experiencing malfunction may not have available space or soil resources for conventional-style repairs, and thus may require more expensive advanced pretreatment systems. Future work is needed to better quantify the number of malfunctioning systems in the watershed and the influence of failing systems on nutrient transport to surface waters to improve these estimates.

Management Implications

Nutrient inputs to Falls Lake may be elevated in some sub-watersheds due to OWTS. Based on water quality data collected during this study and previous studies, conventional OWTS likely contribute between 0.1 and 2.7 kg-N per capita/yr and between 0 and 0.2 kg-P per capita/yr for functioning systems. Since nutrient treatment by OWTS can vary, it is also important to evaluate the likelihood of system malfunctions and their effects on nutrient exports.

The relationships between septic system density, soil type, and stream nutrient concentrations suggest **(e.g. Figure 20)** that a GIS approach may help to screen stream sites that are more likely to have elevated nutrient concentrations associated with OWTS. Since the Passmore and Park Ridge sites had significantly elevated relative nutrient loads, these sites were looked at in-depth and appear to also be affected by legacy agricultural nutrients and current lawn fertilization. These data suggest that legacy agricultural land-use data in the watershed may help screen streams that can have elevated N. Future work should attempt to further identify the nitrogen sources contributing to the elevated nitrogen loads along this stream. If the primary sources are upstream of Passmore, the stream reach between the Passmore and Park Ridge sites may be ideal to study in-stream nitrogen attenuation. When the Passmore and Park Ridge sites are excluded from the dataset, there is a stronger correlation with septic system density and nutrient concentrations. Multiple regression models were developed that used septic system density and % sandy loam soil sub-watershed data to predict stream nitrogen and PO4-P concentrations (**Figure 21**) (excluding those two outliers that appear affected by legacy agricultural nutrients and possibly lawn fertilizer). These relationships suggest that the sub-watersheds with the greatest potential for elevated nutrient exports are those with a high septic system density and sandy loam soils. However, since the sandy loam soils in these watersheds may overlie clay-rich soils, more work is needed to evaluate if this relationship is valid throughout the watershed.

Median TDN Conc.(mg/l) = -3.14 + 2.047 Septic System Density + 0.0276 % Sand; (R^2= 63.6)

Median PO4-P Conc. (mg/l)= -0.140 + 0.1019 Septic System Density + 0.001701 % Sand; (R^2= 53.4)

In addition to GIS screening (sandy loam soils, septic system density, current or historical agricultural land use, etc.) for sites that may have a greater likelihood of having elevated nutrient

concentrations, the data suggested that specific conductivity in streams may be helpful for identifying tributaries with elevated nitrate concentrations. Positive correlations were found between stream nitrate concentrations and specific conductivity (**Figure 22**), which suggests that specific conductivity may be used as a screening tool to identify elevated stream nitrate concentrations. Future work will aim to monitor stream stage, flow, nitrate concentrations, and specific conductivity during and after storm events to improve the loading estimates and enhance understanding of onsite wastewater nutrient transport to the streams in the Falls Lake watershed. The nutrient loading estimates in this study for most sites were based on grab sampling typically during baseflow conditions. Although funding was only available for a limited number of sampling events, a longer-term site (HD1) provided a more robust estimate of onsite wastewater nutrient loading. Future work will aim to refine estimates with more frequent baseflow sampling and high frequency data collection with water quality sensors equipped with data loggers. In addition, chloride and additional tracers, including fluorescence will be utilized to verify the transport of wastewater to the streams. Preliminary fluorescence data was analyzed for a subset of the OWTS sites to represent each geological setting under study (Passmore- Falls Lake and Crabtree Terrane; Jones and Jones Ditch- Triassic Basin; Harold- Carolina Terrane). Initial analysis indicates the presence of wastewater at Passmore, Jones, Jones Ditch, and Harold (**Figure 23**). Jones ditch site, which had the highest density of systems (sand filter systems), elevated nitrate, chloride, and N-15, also had the highest fluorescence intensity. Future efforts will utilize fluorescence data to evaluate the spatial and temporal variability of wastewater inputs in the watershed.

Overall, the data and literature review suggested that approximately 74-100 % of nitrogen and 90-100 % of PO4 are attenuated between the sites with conventional OWTS and streams. This level of treatment can be similar to what would be expected at a conventional municipal wastewater treatment plant (Iverson et al. 2015). In general, in sub-watersheds served by OWTS at low densities (< 0.5 systems/ha) it is less likely to observe a measurable increase in stream nutrient concentrations during baseflow conditions. At low levels of septic system density (<0.5 systems/ha) the nutrient concentrations in streams in watersheds utilizing onsite wastewater systems may be similar to undeveloped watersheds (Iverson et al. 2018). However, when system density is greater than 1 system/ha (Hoghooghi et al. 2016, Iverson et al. 2018) increased stream nutrient concentrations become more likely, particularly in settings with sandier soils where nitrate is more readily transported to streams. Future work will aim to identify tributaries in Falls Lake watershed with elevated nutrient concentrations associated with onsite wastewater and develop best management approaches to aid in nutrient attenuation.

COVID-19 Updates

Due to the COVID-19 situation the isotopic data analysis for the June sampling event has not yet been completed due to lab closure at UC-Davis and nutrient analyses for the June sampling have not yet been completed due to the closure of the Environmental Research Lab at ECU. Once the labs reopen and the samples are analyzed, results will be updated to include the additional data.

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Isotope Facility provided N-15 analysis. Lori Farley and Bobby Lee Vaughan helped with sample collection. The ECU Water Resources Center provided equipment and logistical support.

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FIGURES

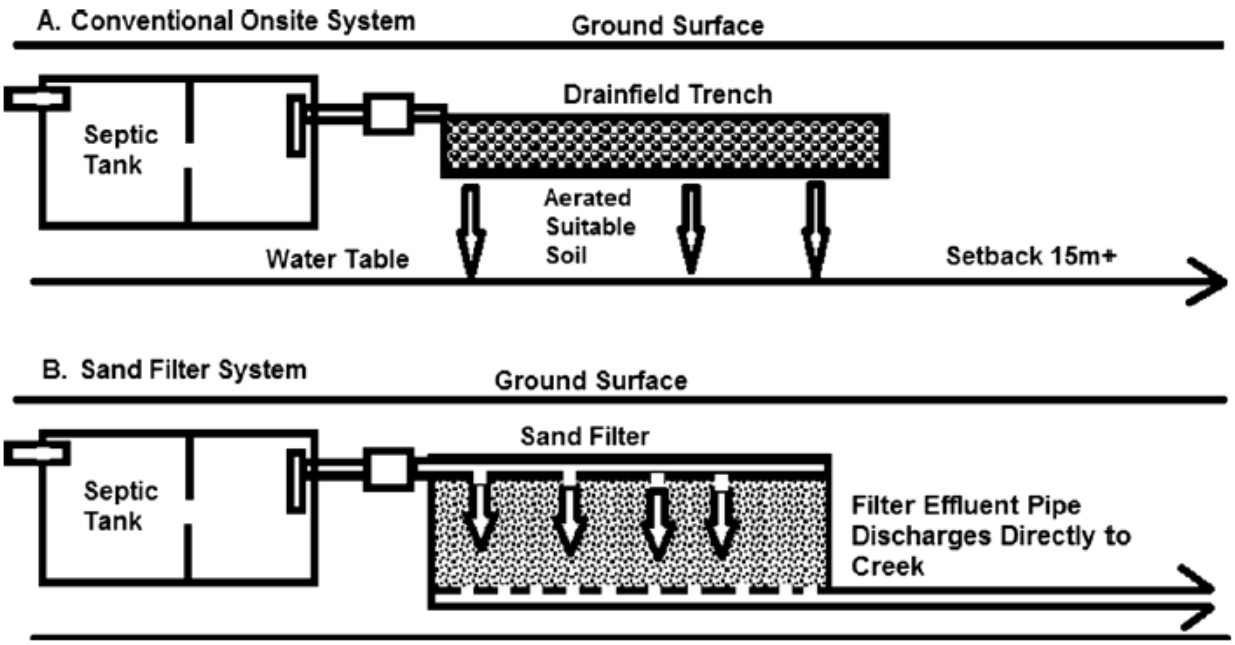


Figure 1. Comparison of a conventional onsite wastewater treatment system (A.) and a sand filter system (B.) (Humphrey et al. 2016a).

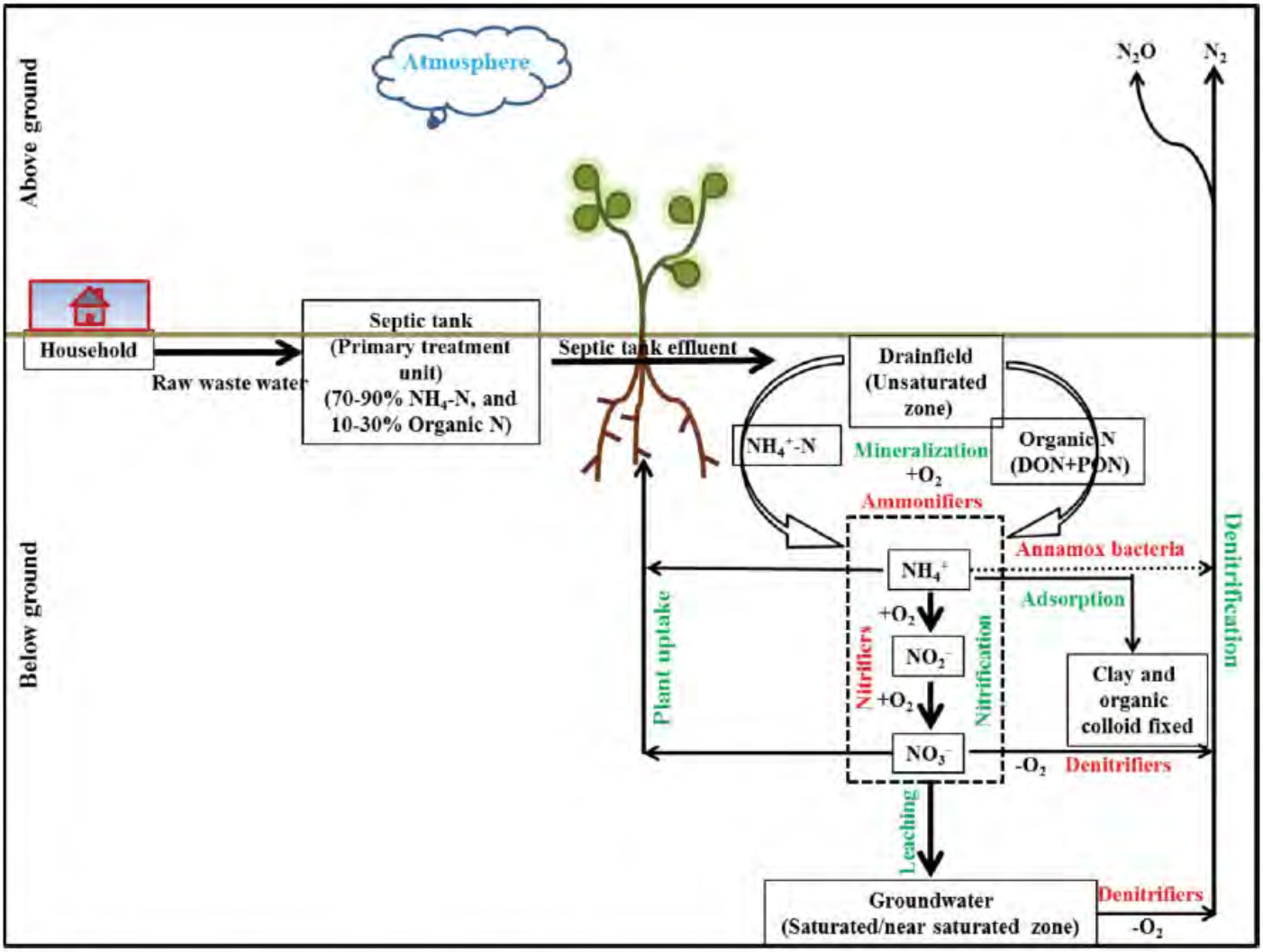


Figure 2. Nitrogen transformations and retention that occur in onsite wastewater treatment systems and the adjacent surficial aquifer (modified from Lusk et al. 2017).

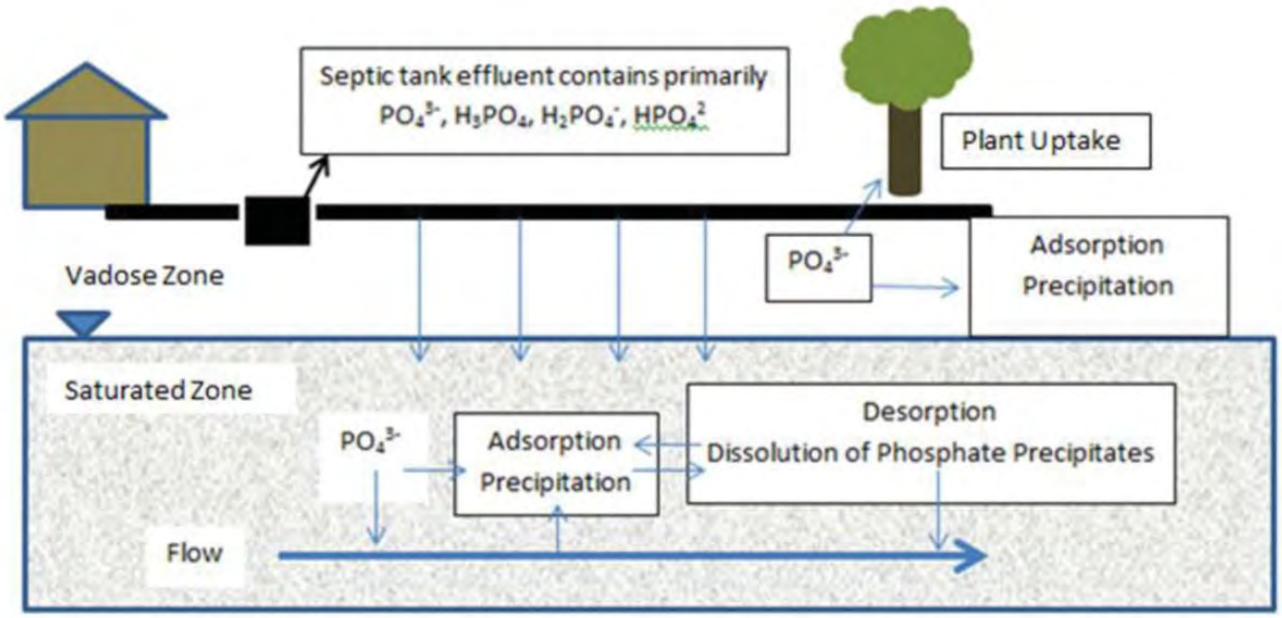


Figure 3. Phosphorus transformations and retention that occur in onsite wastewater treatment systems and the adjacent surficial aquifer (modified from Lusk et al. 2017).

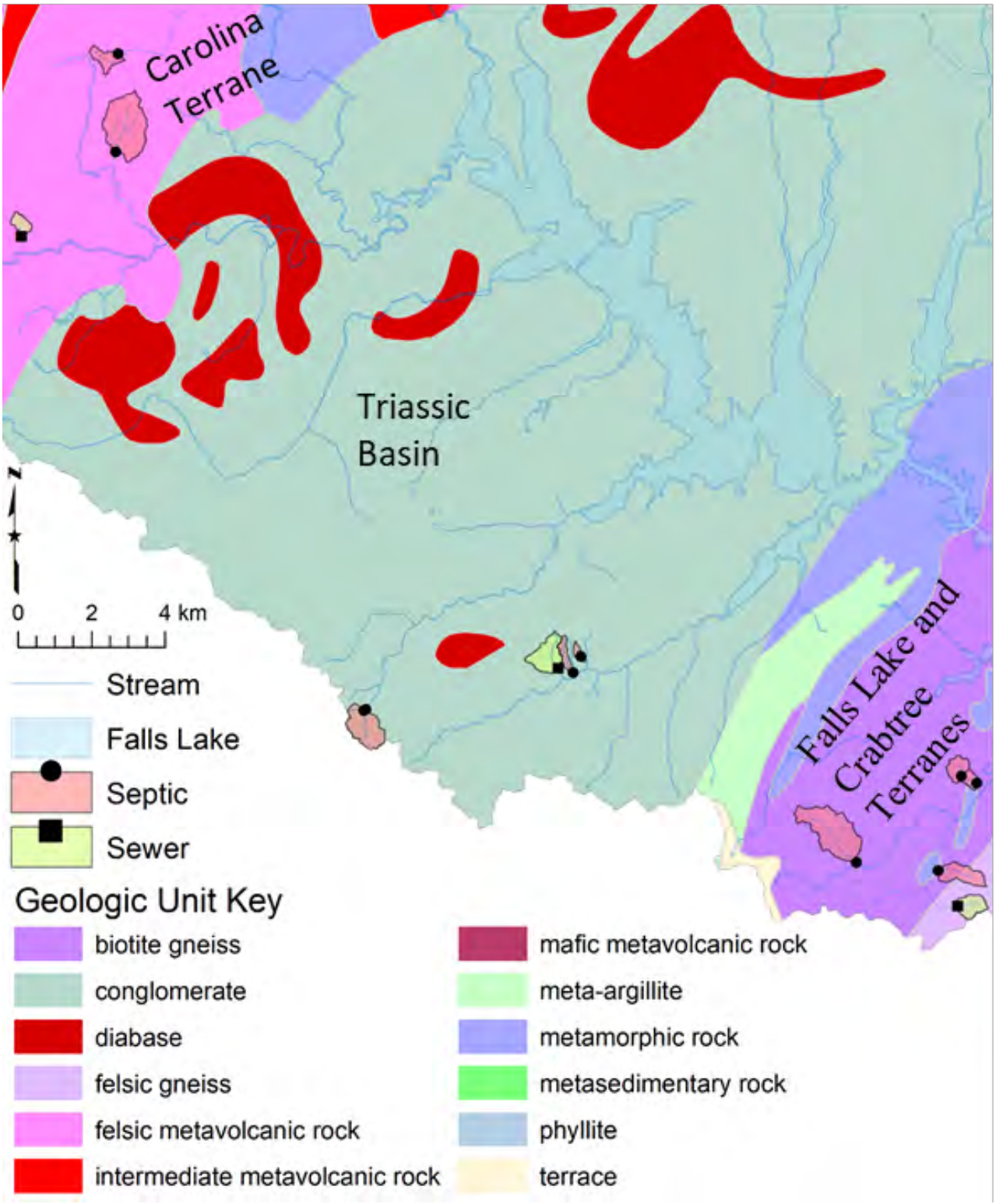


Figure 4. Geologic setting for sampling sites. Northern sites are located in Carolina Terrane (Durham County), central sites are in the Triassic Basin (Durham County), and southern sites are in the Falls Lake and Crabtree Terranes. Details on site soils are provided in **Table 1**.

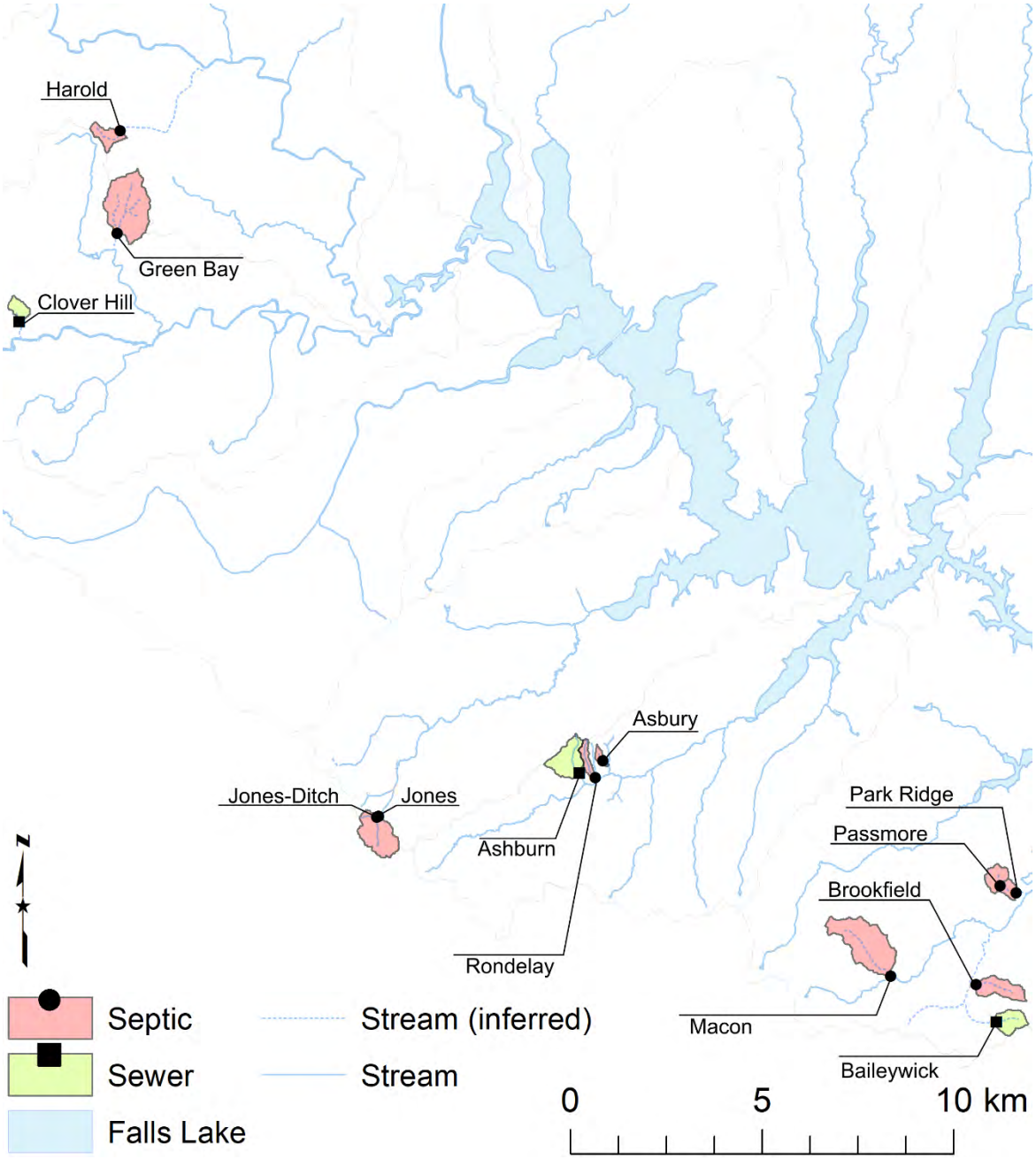


Figure 5. Locations of onsite wastewater treatment system sub-watersheds (OWTS or septic) and sewered (municipal wastewater treatment) sub-watersheds in Wake and Durham Counties, NC. Site information is provided in **Table 1** and individual site maps are included in **Appendix C**.

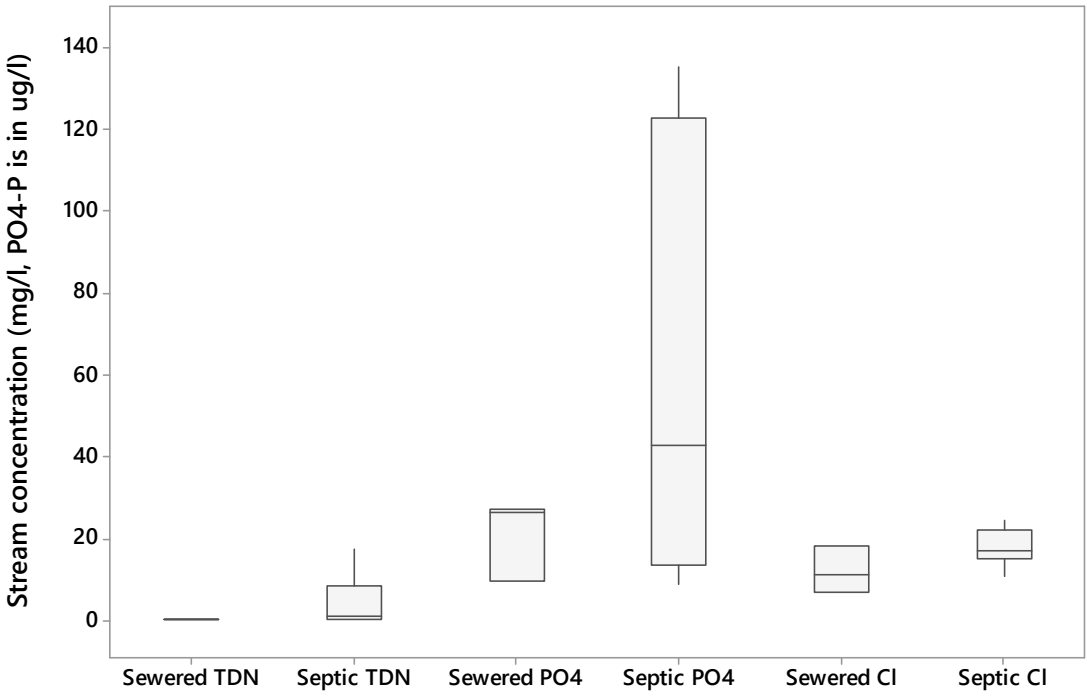


Figure 6. Comparison of Total Dissolved Nitrogen (TDN), Phosphate (PO4-P), and Chloride (Cl) in streams draining sewered vs. OWTS (septic) watersheds. Note that PO4-P concentrations are in ug/l. Data included are from Nov., Dec. 2019 and Feb. 2020 baseflow sampling events.

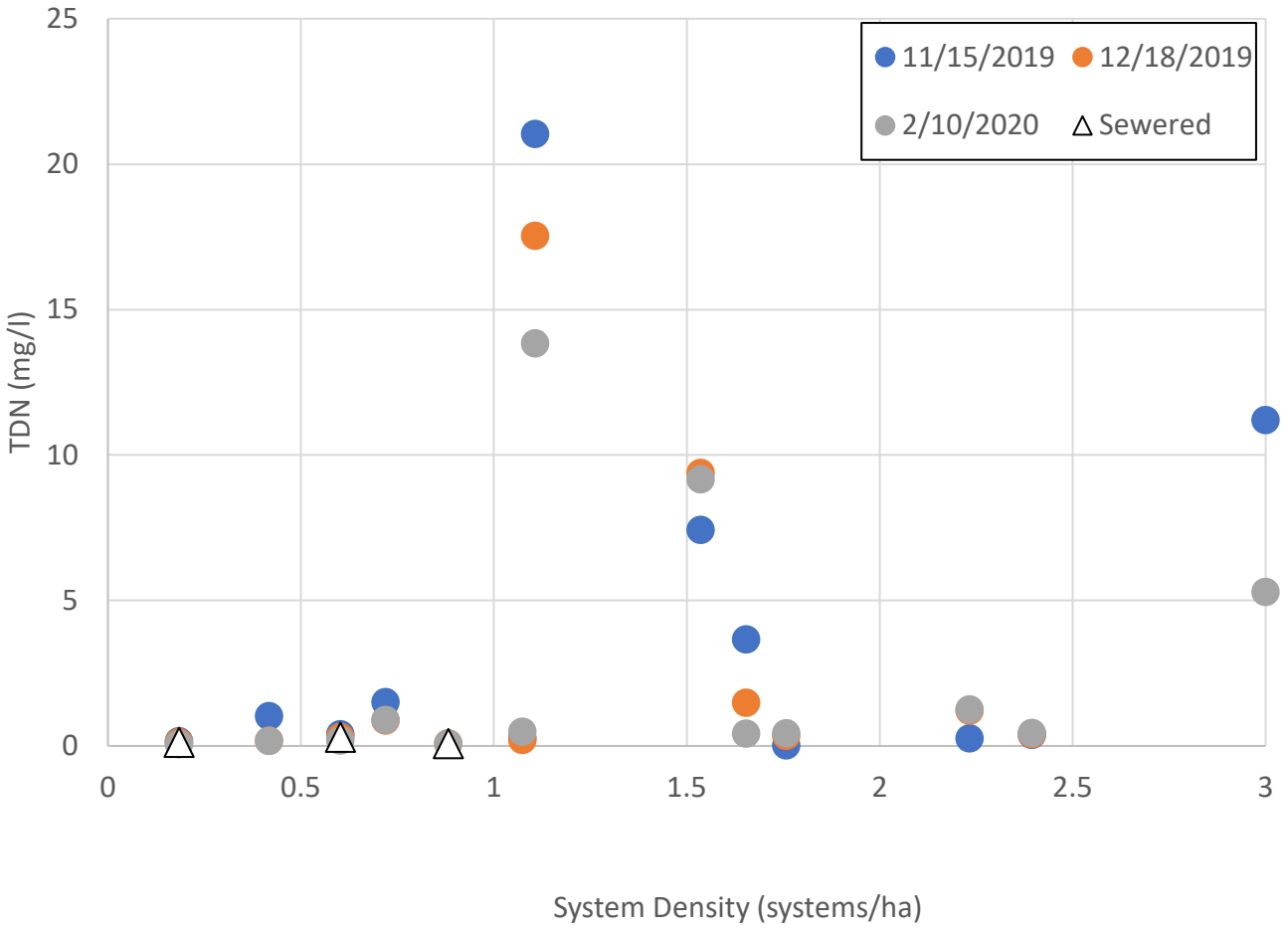


Figure 7. Comparison of septic density vs. total dissolved nitrogen concentrations. Sewered sub-watersheds are indicated by triangles (median concentration data for sewered data is indicated by the triangle, and individual dates are shown for those by system density (Ashburn=0.18 systems/ha; Clover Hill=0.6 systems/ha; Baileywick =0.88 systems/ha). Data included are from Nov., Dec. 2019 and Feb. 2020 baseflow sampling events.

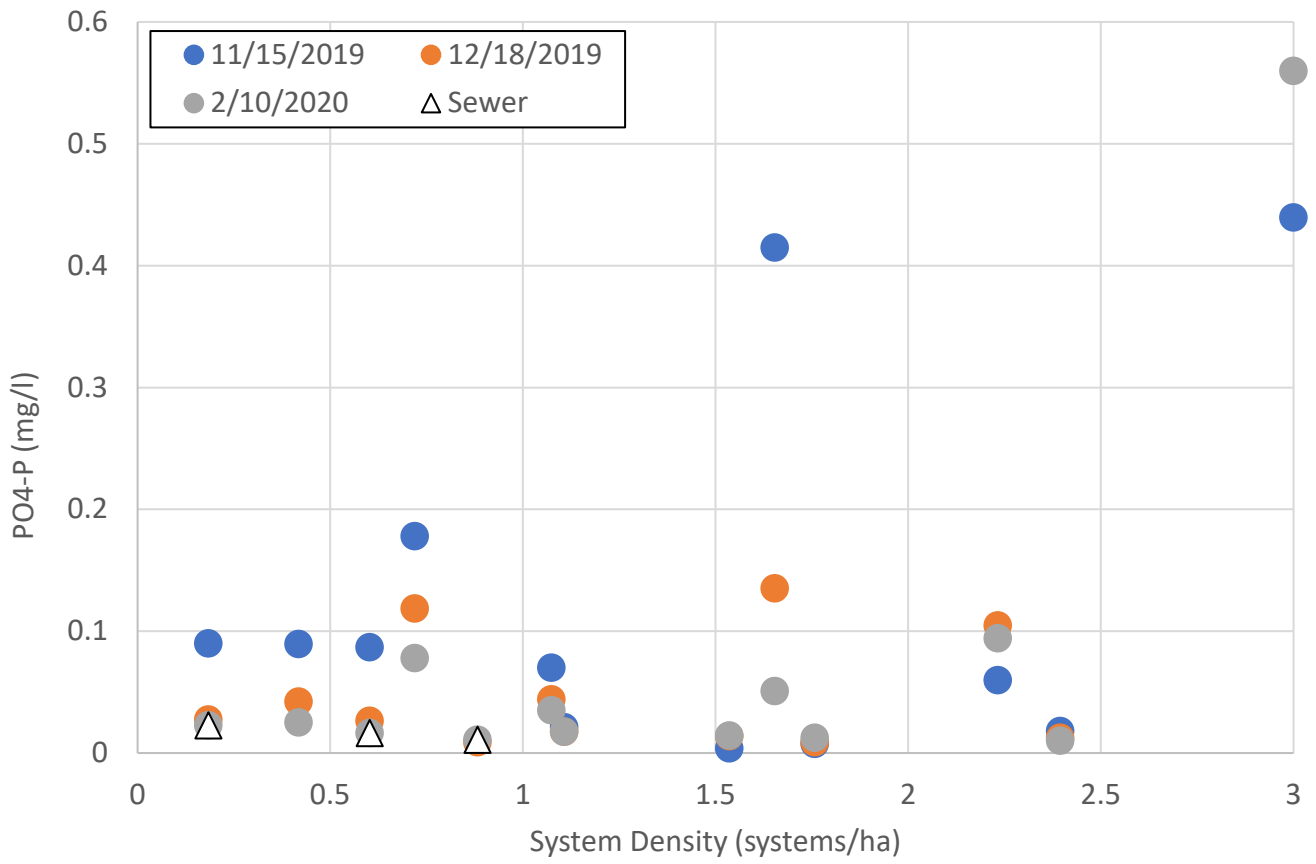


Figure 8. Comparison of septic density vs. PO4-P concentrations. Sewered sub-watersheds are indicated by triangles (median concentration data for sewered data is indicated by the triangle, and individual dates are shown for those by system density (Ashburn=0.18 systems/ha; Clover Hill=0.6 systems/ha; Baileywick =0.88 systems/ha). Data included are from Nov., Dec. 2019 and Feb. 2020 baseflow sampling events.

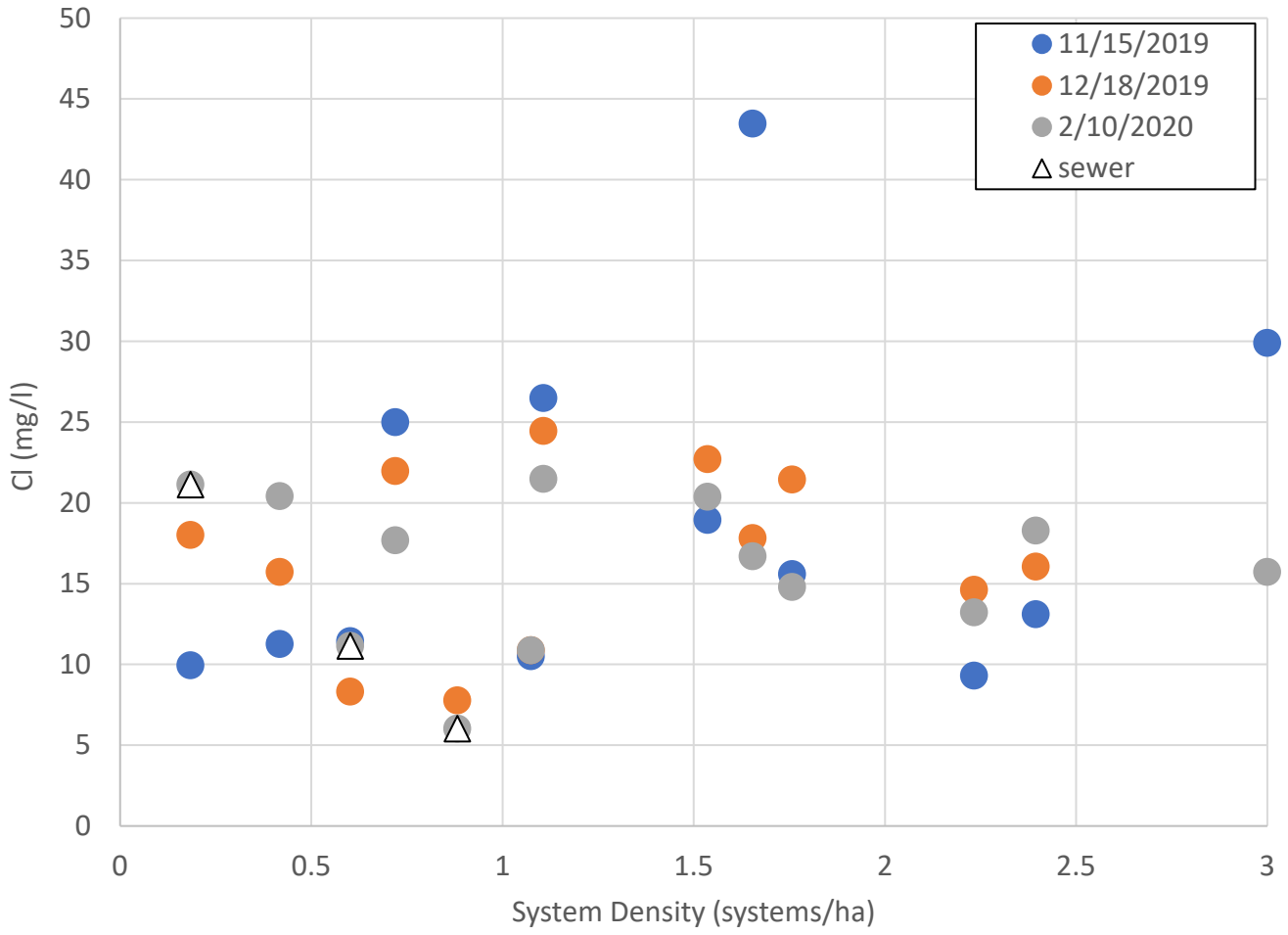


Figure 9. Comparison of septic density vs. Cl concentrations. Sewered sub-watersheds are indicated by triangles. Data included are from Nov., Dec. 2019 and Feb. 2020 baseflow sampling events. (median concentration data for sewered data is indicated by the triangle, and individual dates are shown for those by system density (Ashburn=0.18 systems/ha; Clover Hill=0.6 systems/ha; Baileywick =0.88 systems/ha).

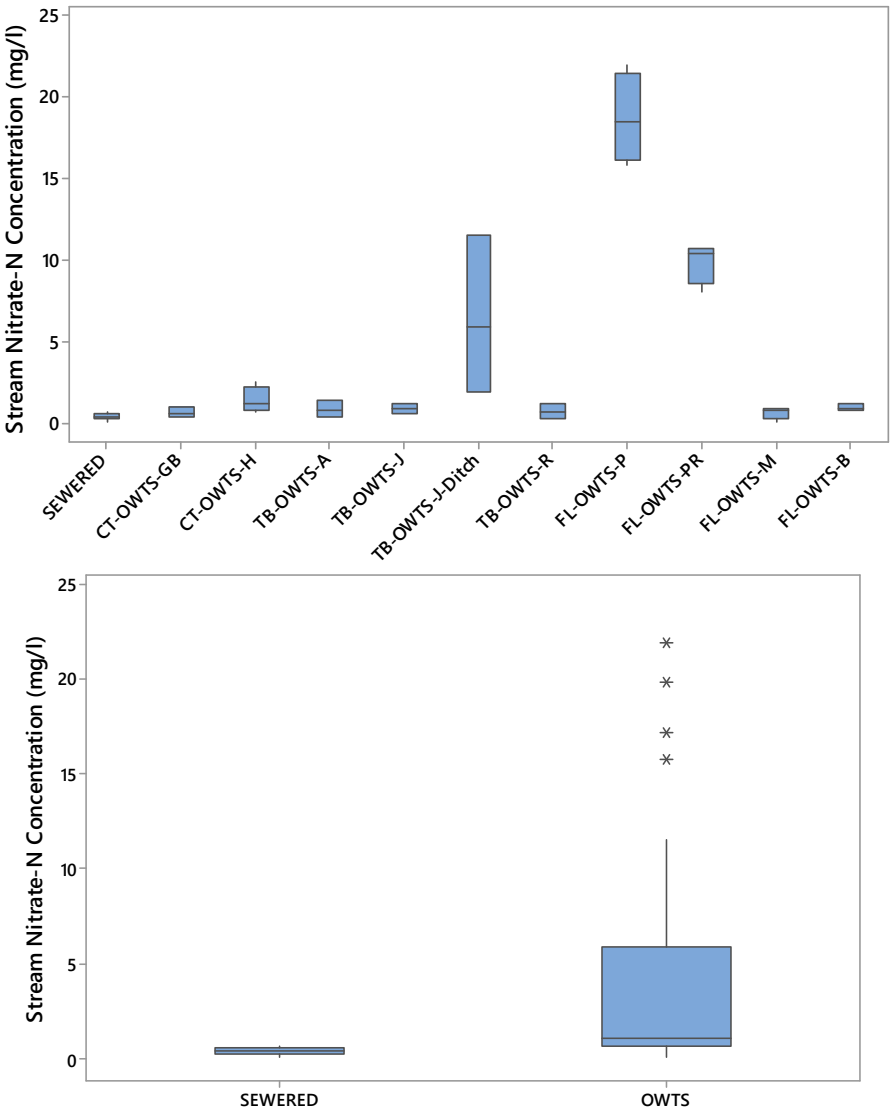


Figure 10A. Nitrate-N stream concentrations at individual sites for sampling dates in November, December 2019 and February 2020. Sewered sites are sub-watersheds that are sewered and utilize municipal wastewater treatment plants outside of the sub-watersheds to treat and dispose of their wastewater. OWTS sites are sub-watersheds that use onsite wastewater treatment systems (OWTS). Site abbreviations CT (Carolina Terrane), TB (Triassic Basin), and FL (Falls Lake and Crabtree Terrane) indicate the geological setting. Typically, the sewered watersheds had lower stream nitrate concentrations during baseflow conditions. For the OWTS sub-watersheds, most sites showed slightly elevated nitrate concentrations relative to the sewered sub-watershed in that geologic setting. Several sites had stream nitrate-N concentrations that were greater than 10 mg/l on at least one occasion (TB-OWTS-J-Ditch (Jones Circle, Durham Co.); FL-OWTS-P (Passmore, Wake Co.); and FL-OWTS-PR (Park Ridge, Wake Co.). Data included are from Nov., Dec. 2019 and Feb. 2020 baseflow sampling events. **Figure 10B**. A comparison of nitrate-N stream concentrations for sewered vs. OWTS sub-watersheds revealed that OWTS sites generally had elevated nitrate concentrations. Median nitrate concentration for OWTS watersheds was 1.05 mg/l vs 0.41 mg/l for the sewered sub-watersheds and a Mann-Whitney revealed the difference in distribution was significant at p=0.001. These data suggest that nitrate concentrations during baseflow conditions are approximately 2.5 times larger for OWTS sub-watersheds when compared to sewered sub-watersheds. However, it appears 2 outliers (Passmore and Park Ridge are also influenced by legacy agricultural nutrients, when these sites are excluded the comparison is still significantly different at p=0.001 and the median OWTS nitrate-N concentration declines to 0.92, 2.24 times larger.

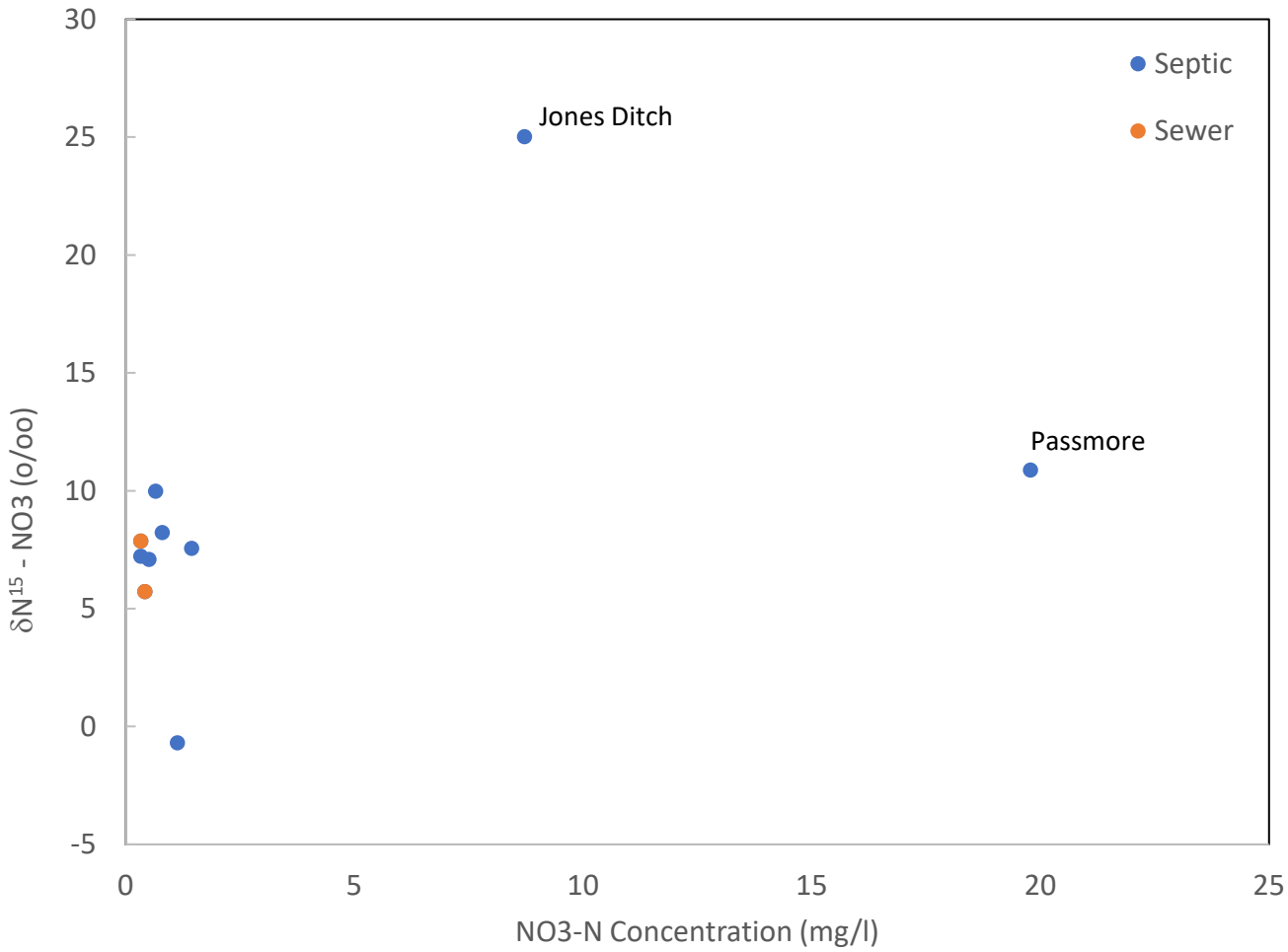
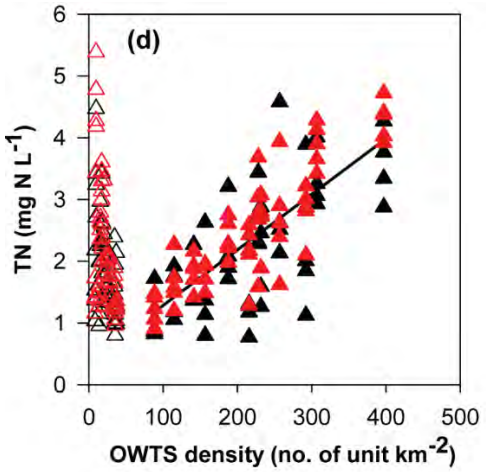


Figure 11. Comparison of nitrate concentrations and δN^{15} - NO3 (o/oo) for stream samples collected on 12/18/2019. Wastewater and manure δN^{15} - NO3 (o/oo) ranges from prior studies (Silva et al. 2002, Xue et al. 2009) are typically between 7-20 δN^{15} - NO3 (o/oo) The data from the current study suggest that wastewater is a contributing source to stream nitrate in these sub-watersheds with OWTS.



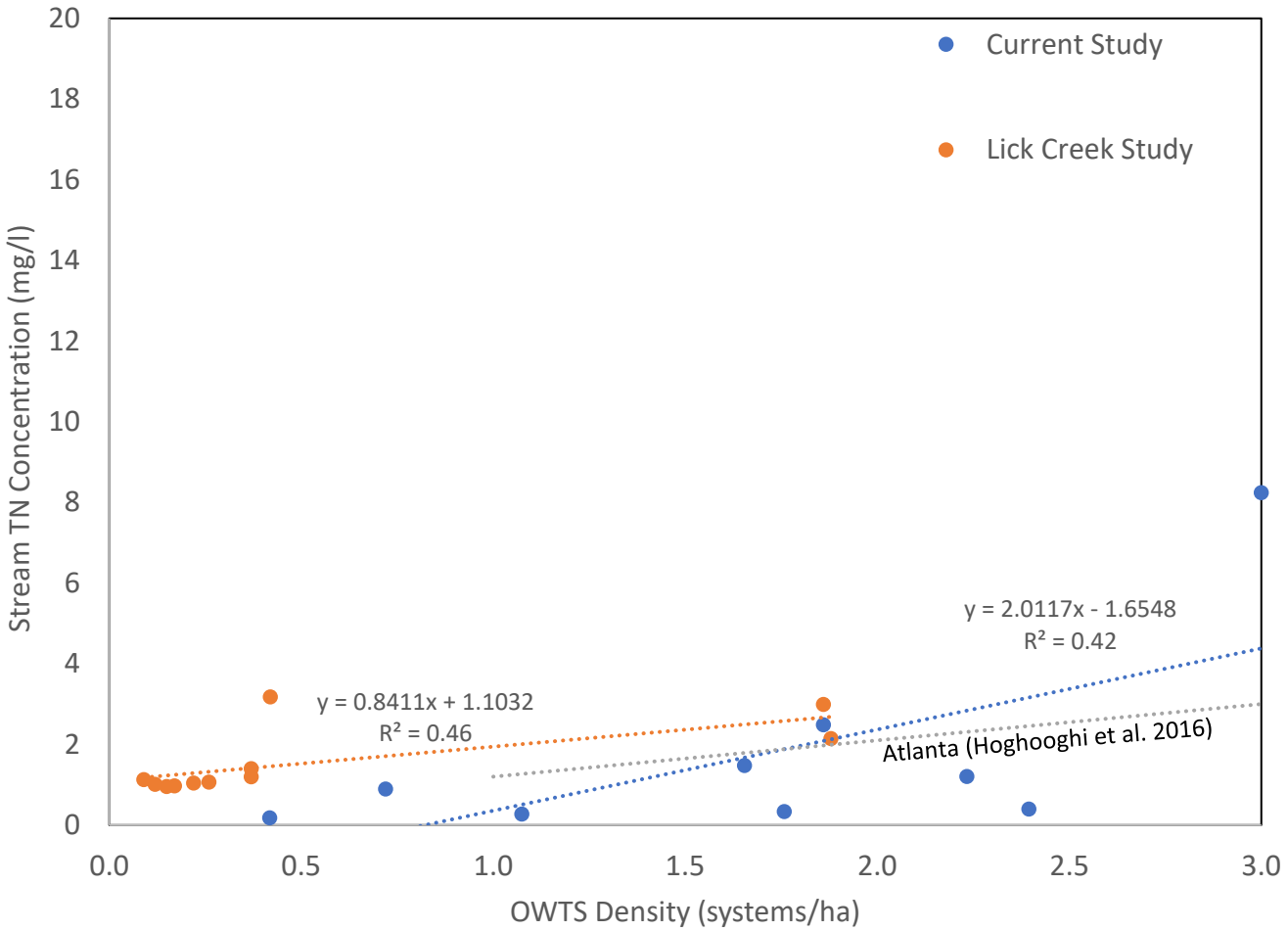


Figure 12A. A comparison of TDN vs onsite wastewater treatment system density for a study in the Atlanta, GA region (modified from Hoghooghi et al. 2016). **12B.** A comparison of TDN vs onsite wastewater treatment system density for the current study (note the outliers affected by agricultural legacy nutrients, Passmore and Park Ridge, were excluded), and a former study on Lick Creek (2015-2016). The trendline from Atlanta streams by Hoghooghi et al. (2016) is included for reference.

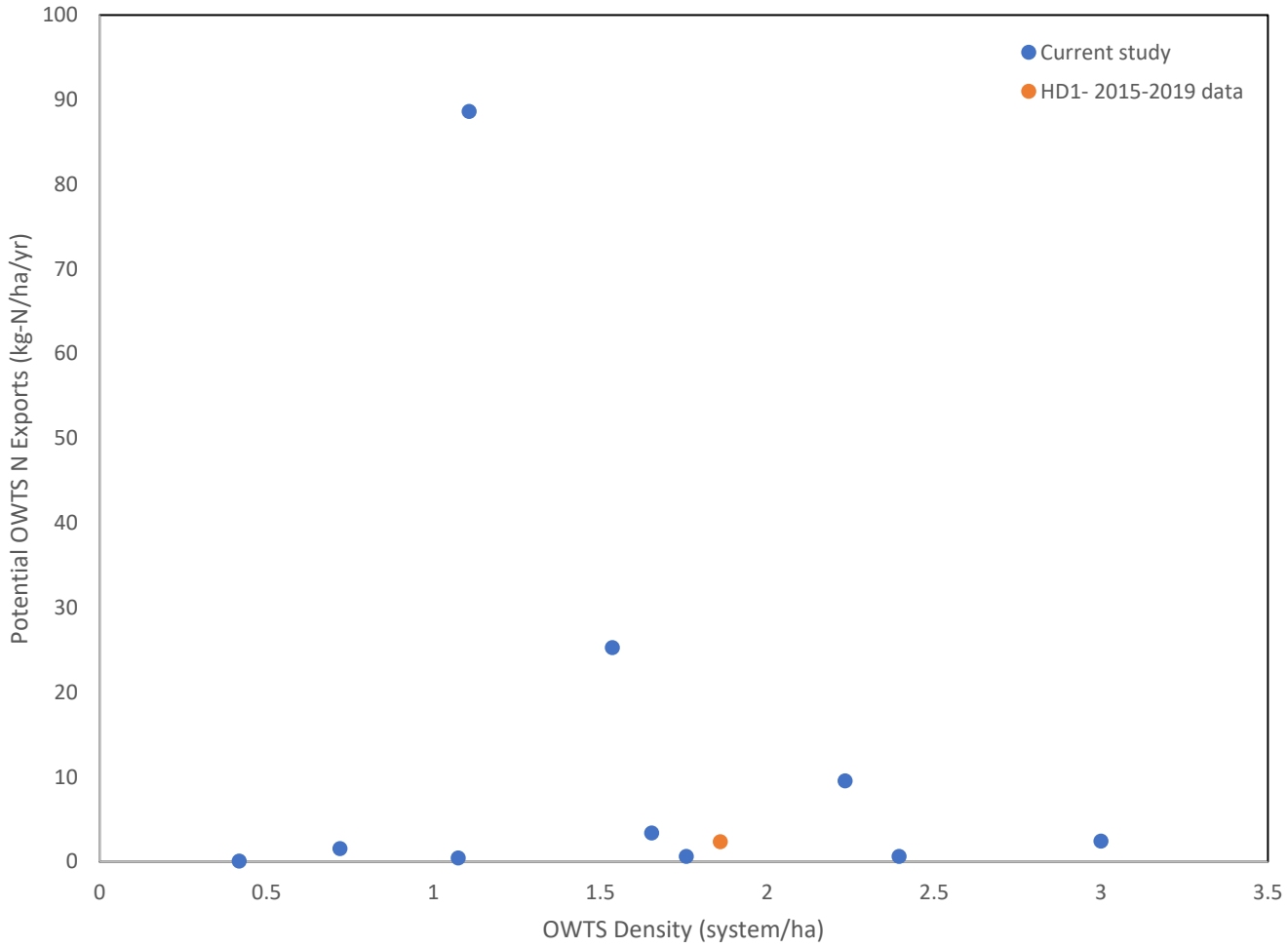
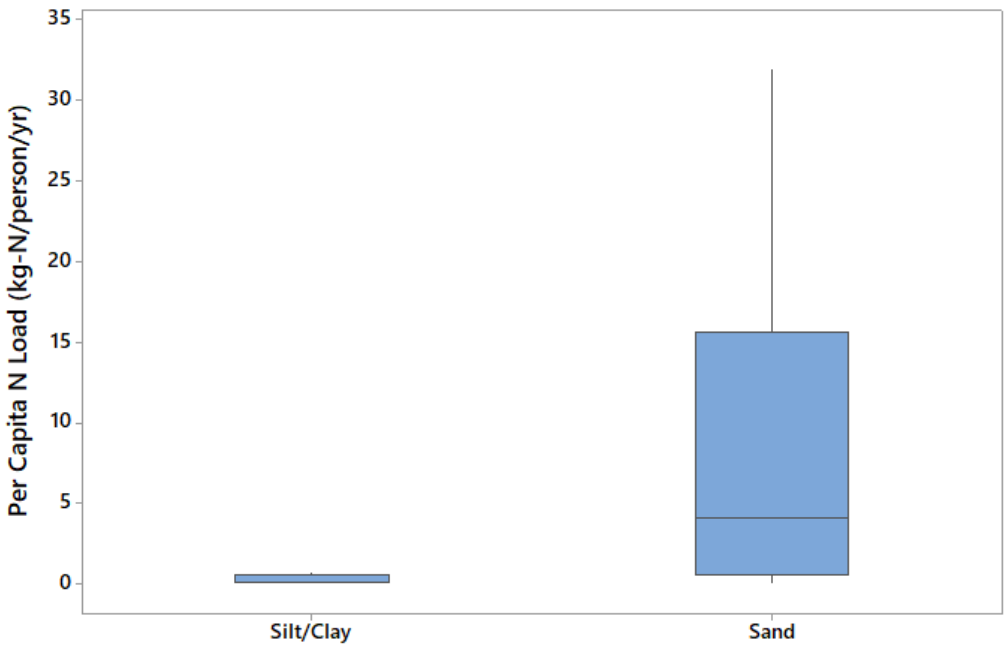


Figure 13. A comparison of the potential OWTS N exports (kg-N/ha/yr) vs OWTS density (systems/ha). The HD1 site is the long-term site that includes monthly data from 2015-2019, representing a longer-term estimate (corrected by subtracting the loading from a paired sewered watershed). Note the elevated loading at Passmore (88kg-N/ha/yr) and Park Ridge (25 kg-N/ha/yr) is presumably related to agricultural legacy nutrients and fertilizer.



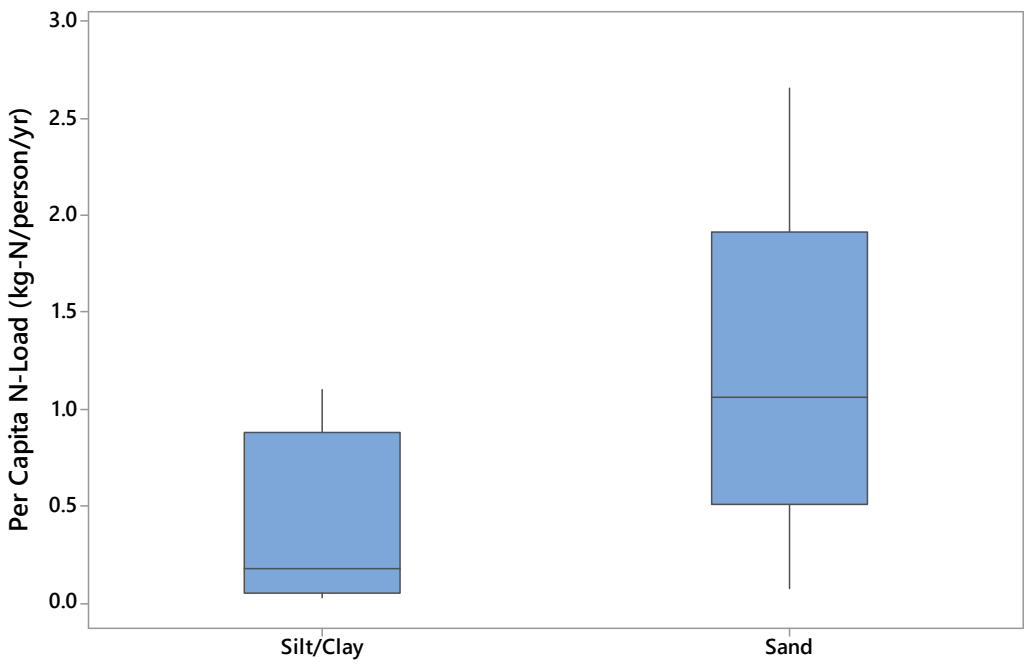


Figure 14A. Comparison of potential per capita N-loading from onsite systems in watersheds that have > 30% sandy loam soils in the sub-watershed vs. sub-watersheds with 70% or more clay/silt loam soils. **14B.** Same comparison, excluding the outliers likely influenced by legacy agricultural and fertilizer-related nutrients (Passmore and Park Ridge).

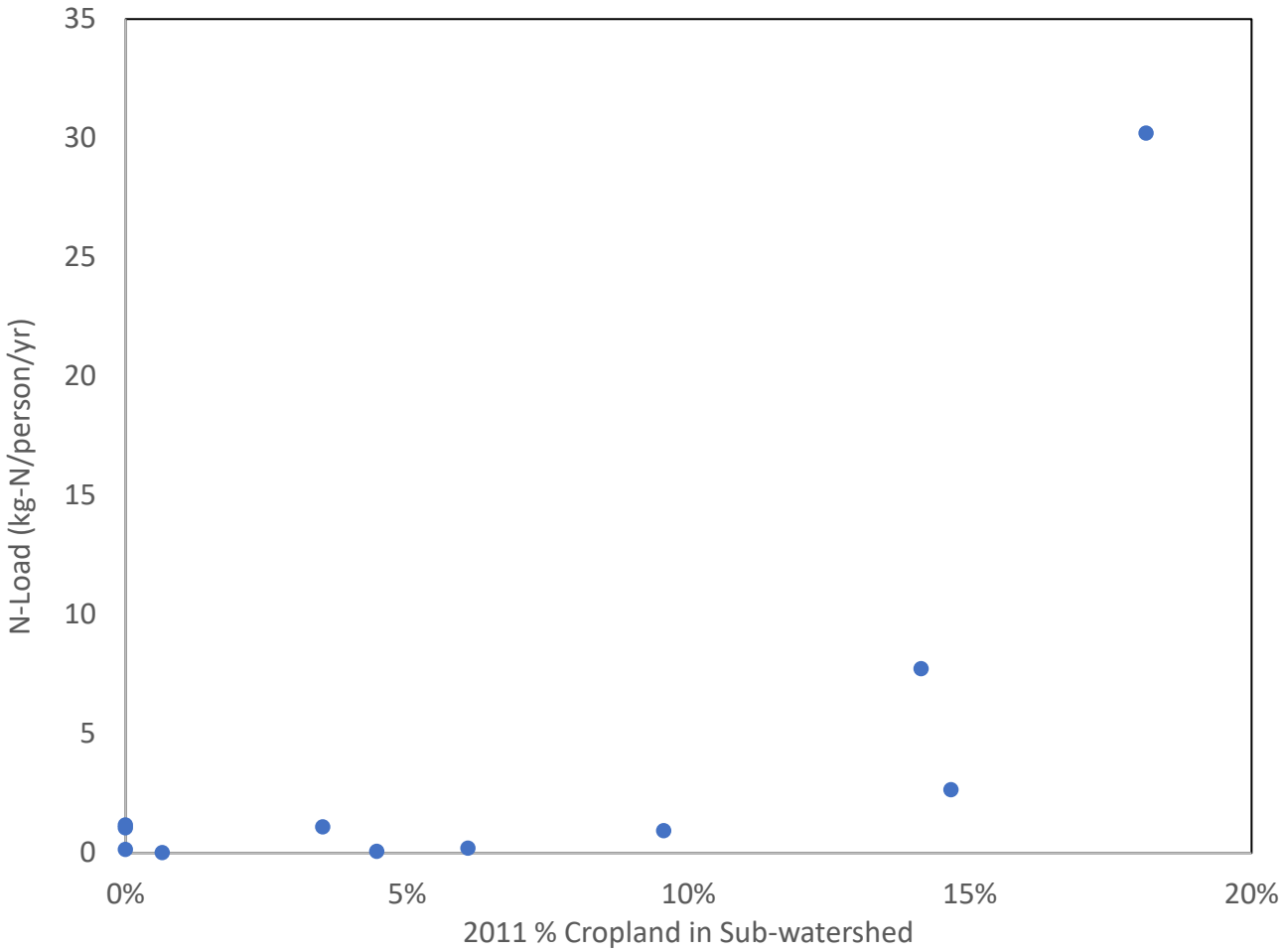


Figure 15. Comparison of potential per capita N-loading from onsite systems with the % cropland in the sub-watershed from the 2011 National Land Cover dataset.

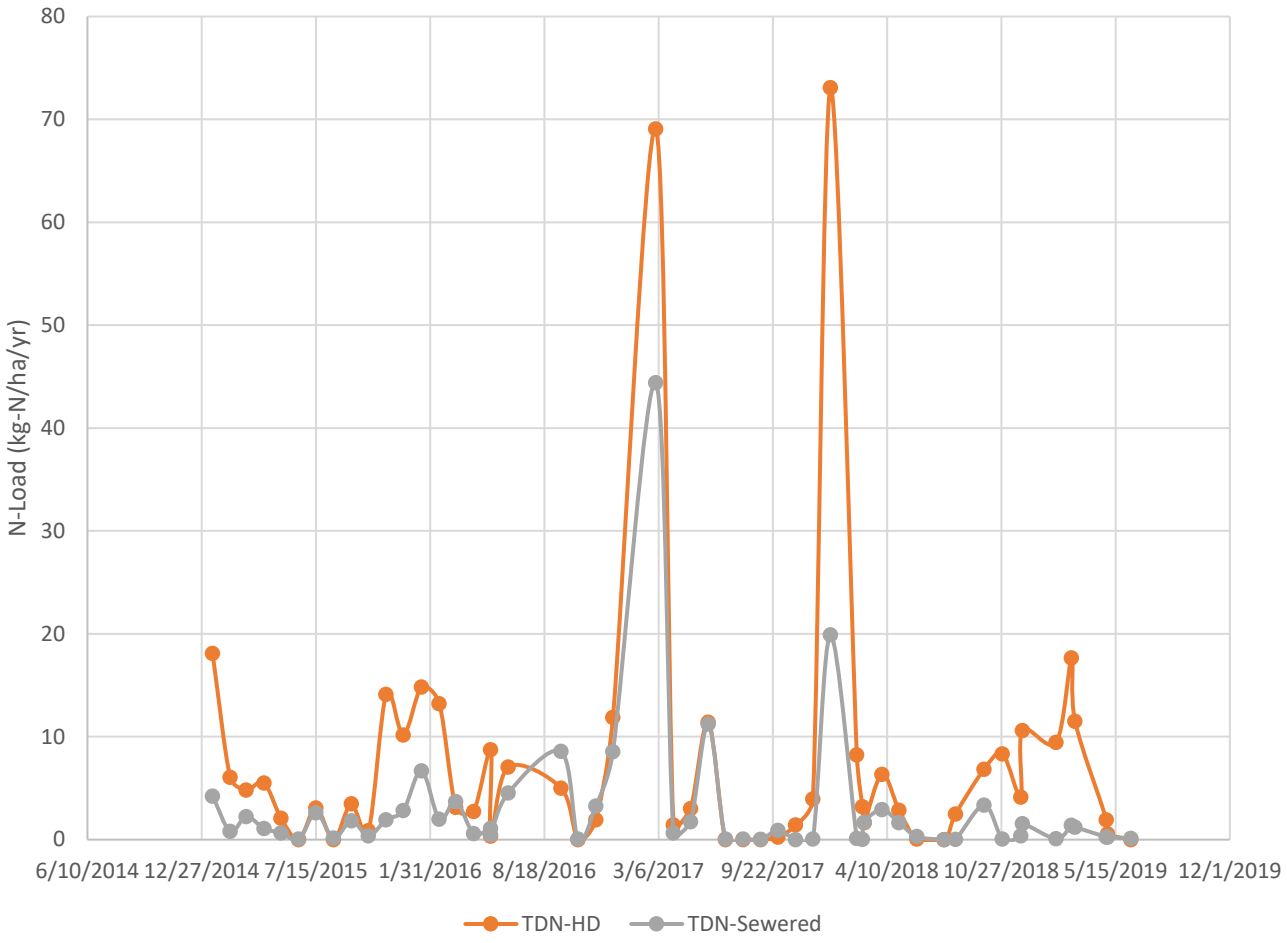


Figure 16. A comparison of TDN exports at a stream in a sub-watershed with a high density of onsite systems and a sewered sub-watershed. The onsite sub-watershed had significantly elevated TDN exports (median exports were 2.35 kg-N/ha/yr greater, p= 0.002) relative to the sewered sub-watershed based on concentration and flow data collected from 2015-2019.

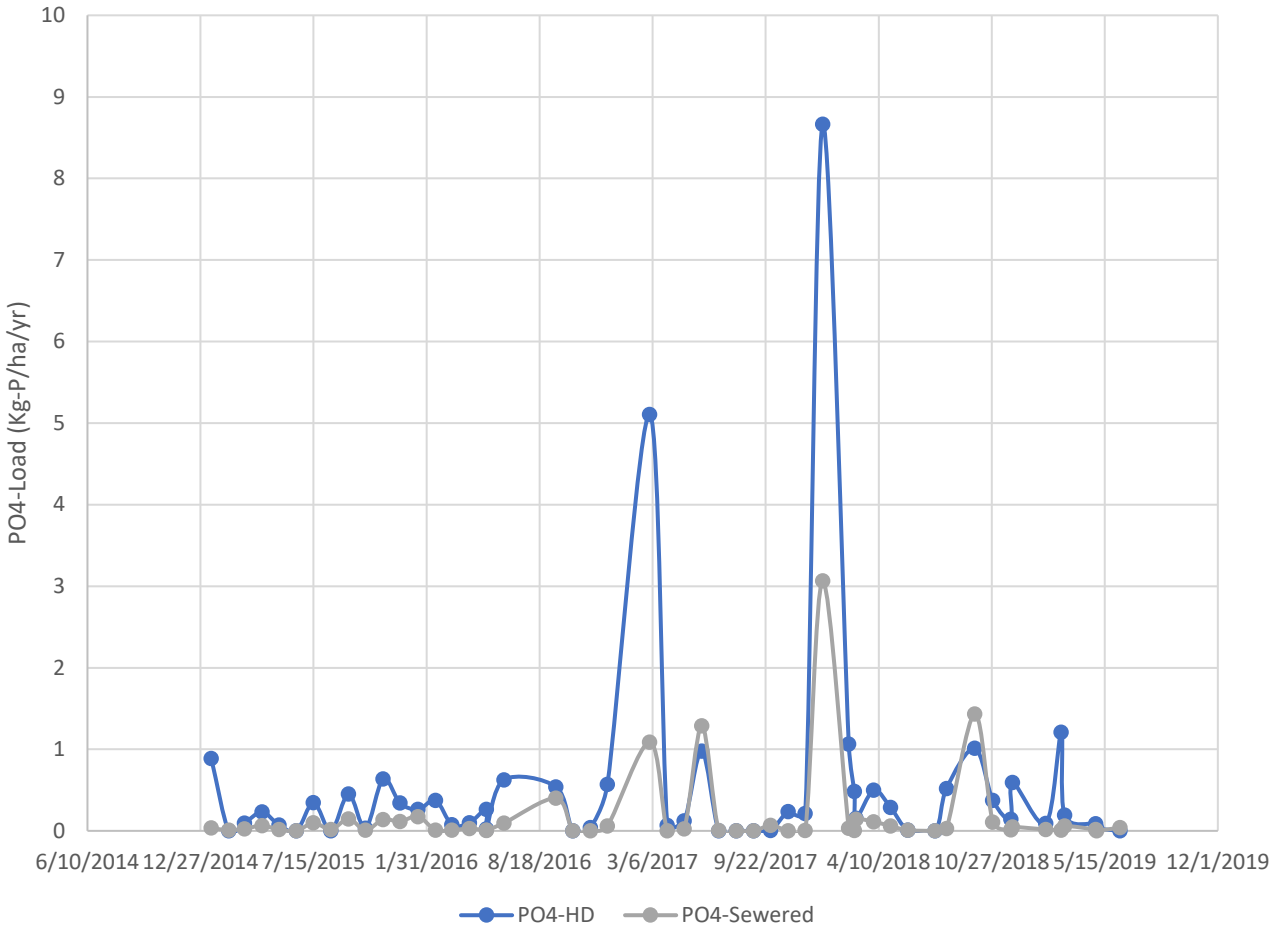


Figure 17. A comparison of PO4-P exports at a stream in a sub-watershed with a high density of onsite systems and a sewered sub-watershed. The onsite sub-watershed had significantly elevated PO4-P exports (median exports were 0.21 kg-N/ha/yr greater, p= 0.0001) relative to the sewered sub-watershed based on concentration and flow data collected from 2015-2019.

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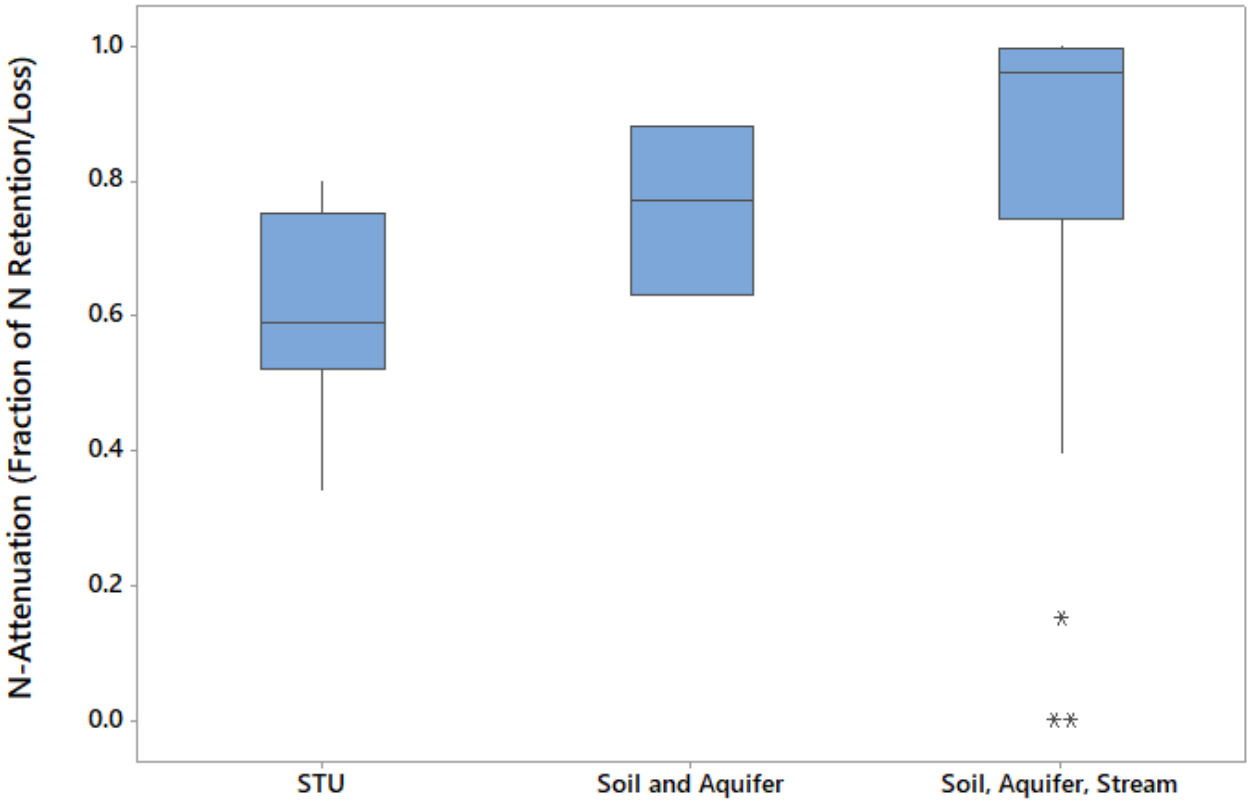

Figure 18. N-attenuation factors estimated from literature on Piedmont soils and surficial aquifer data. Attenuation between the tank and stream (soil, aquifer, stream attenuation) is based on the surface water data collected during this study and the earlier surface water data. Data and references are found in **Table 3**.

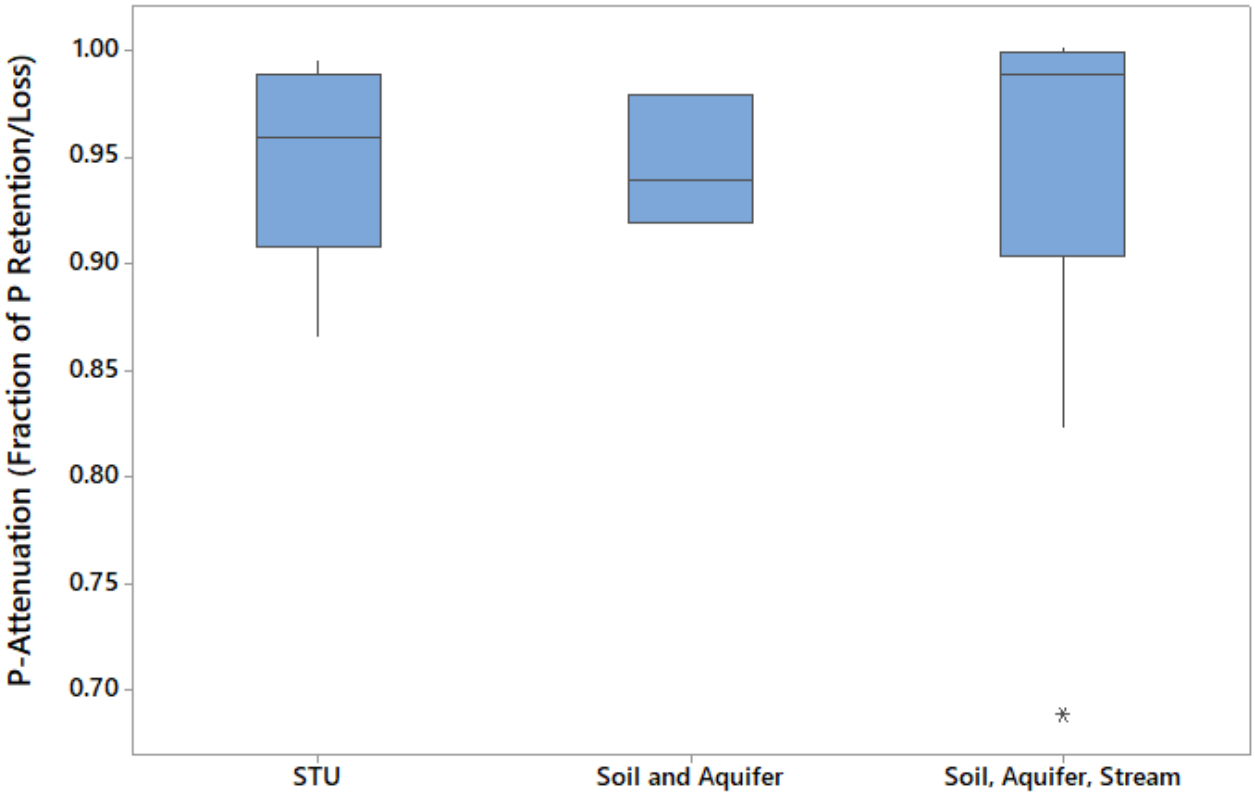

Figure 19. PO4-P-attenuation factors estimated from literature on Piedmont soils (STU estimates also include Total P) and surficial aquifer data. Attenuation between the tank and stream (soil, aquifer, stream attenuation) is based on the surface water data collected during this study and the earlier surface water data. Data and references are found in **Table 4**.

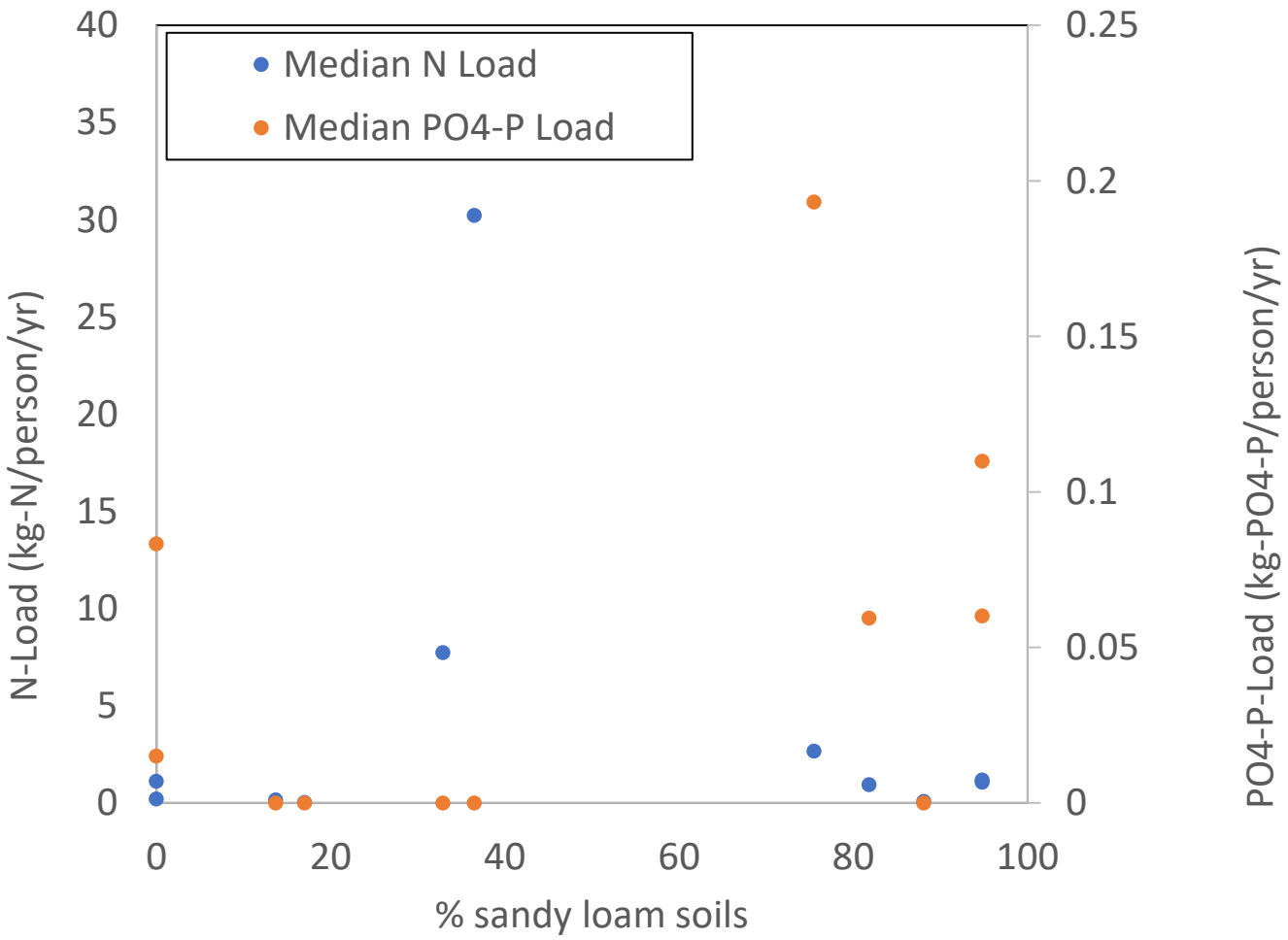

Figure 20. Comparison of median nutrient loads at sub-watershed outlets and % sandy loam soils.

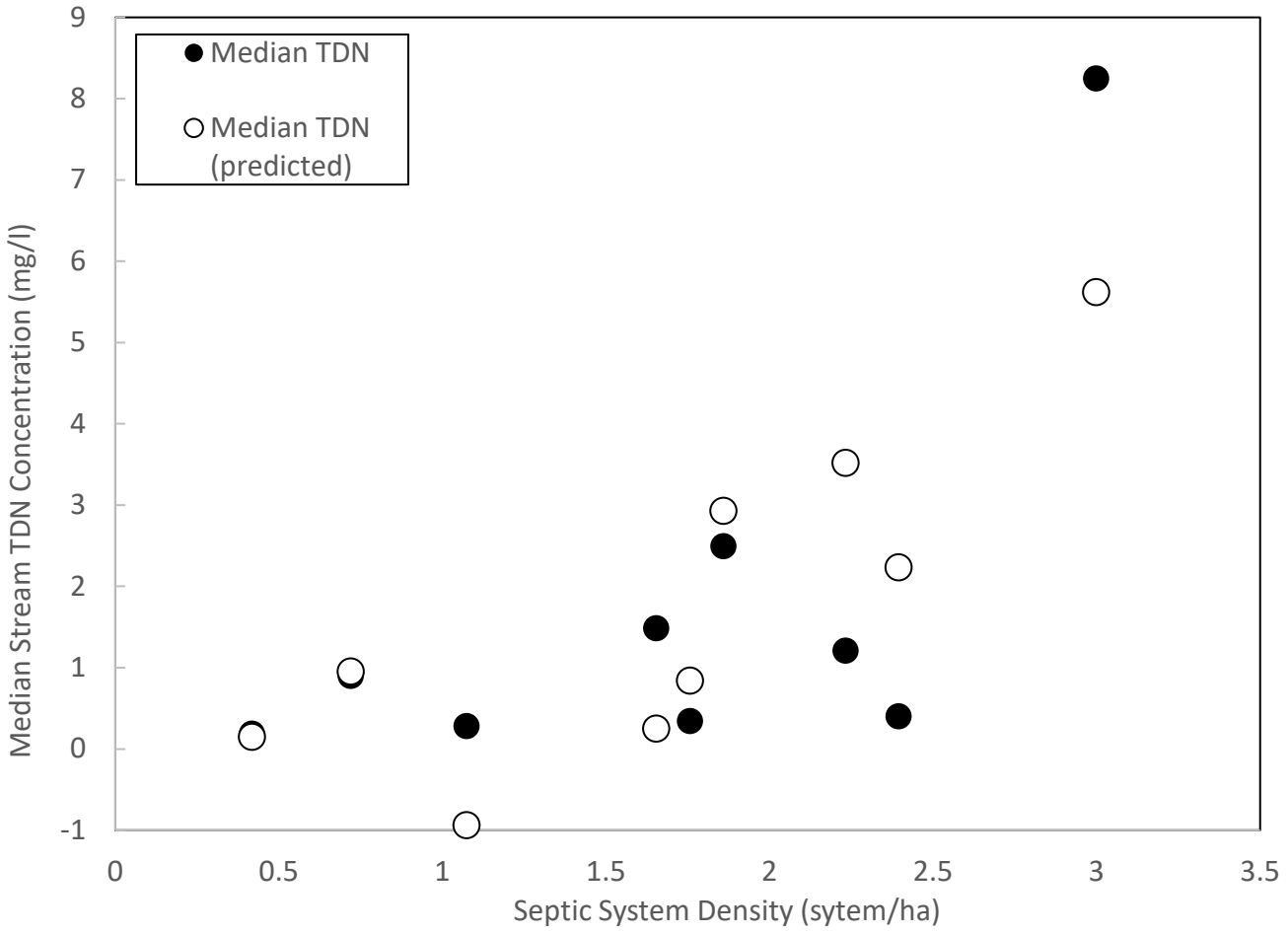

Figure 21. Septic system density and % sandy loam soil data were used to estimate stream nitrogen concentrations from the following equation: Median TDN Conc.(mg/l) = -3.14 + 2.047 (Septic System Density) + 0.0276 (% Sand) (R^2= 63.6). Note the sub-watersheds believed to be affected by legacy agricultural nutrients (and possibly lawn fertilizer) (Passmore and Park Ridge) were excluded.

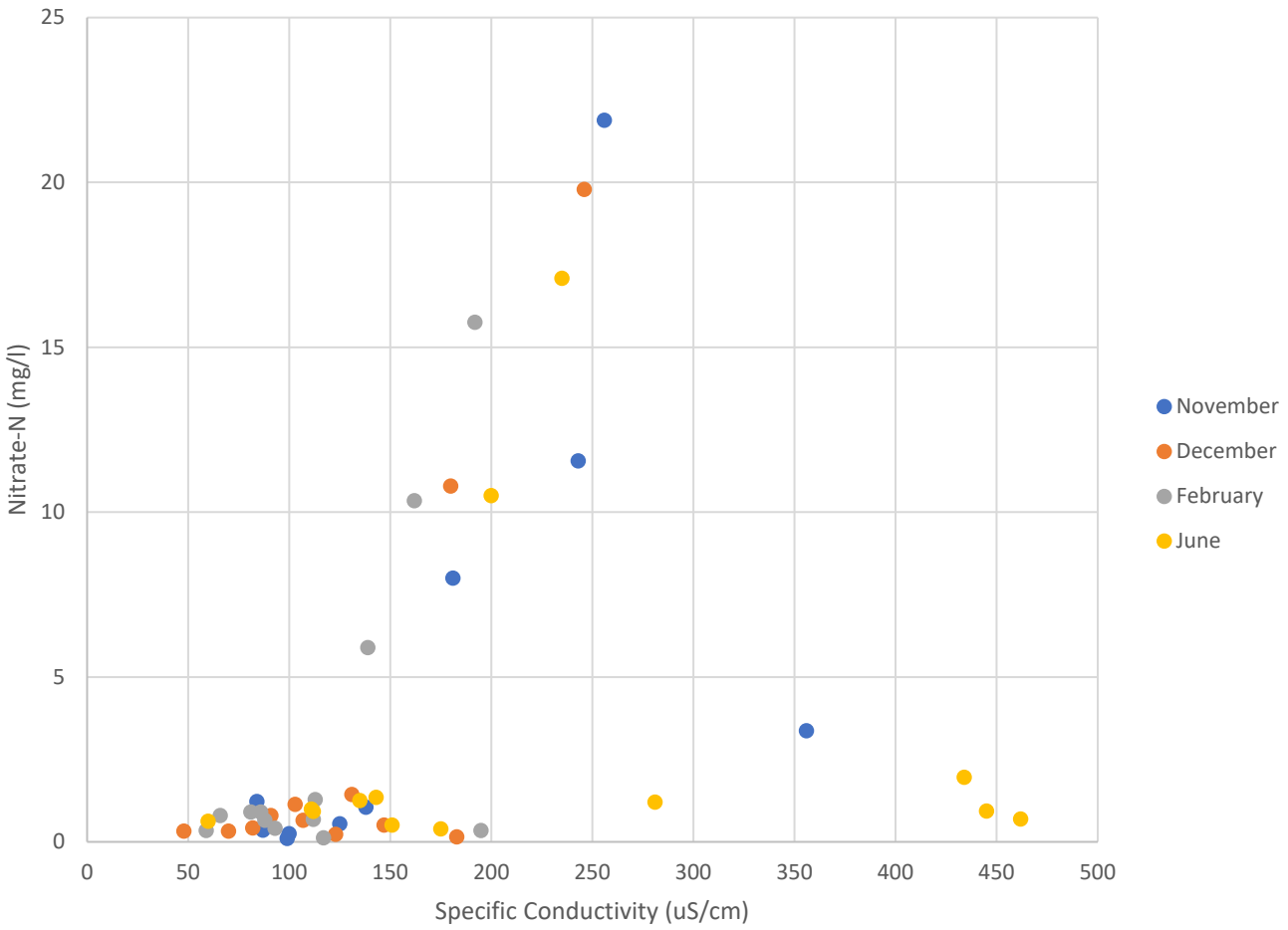

Figure 22A. Specific conductivity vs stream nitrate concentrations in sub-watersheds. A general positive correlation existed between specific conductivity and nitrate, however during June at the Rondelay, Jones, and Ashburn (Triassic Basin sites), elevated specific conductivity did not relate to elevated nitrate concentrations, suggesting that the relationship may change during the growing season possibly due to biological assimilation and/or denitrification.

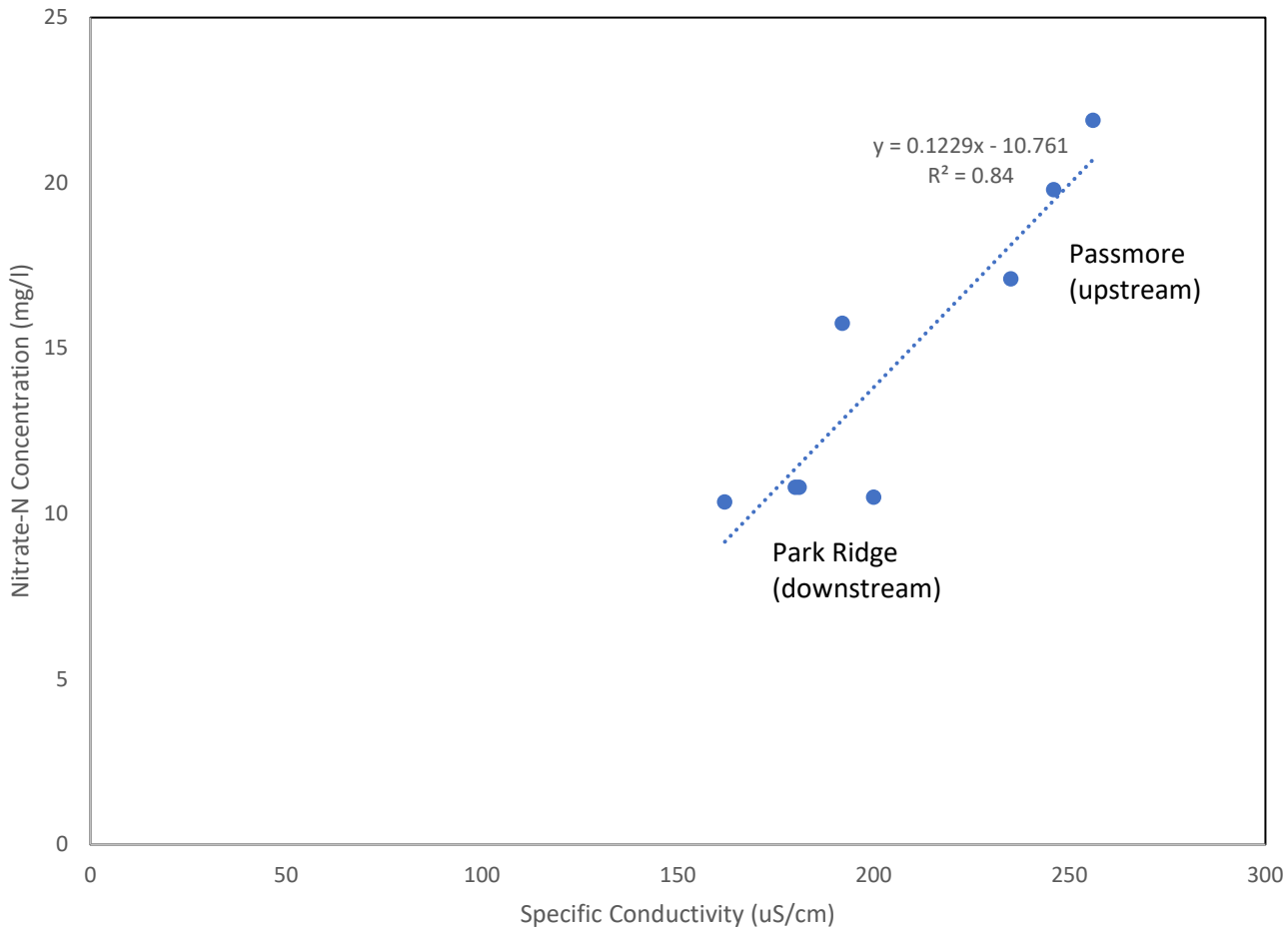

Figure 22B. Specific conductivity vs stream nitrate concentrations patterns at specific sub-watersheds suggests that at individual stream sites, a relationship between specific conductivity and nitrate concentrations over time may be used to evaluate nitrate concentration changes over time and space. This example is for the Passmore and Park Ridge sites.

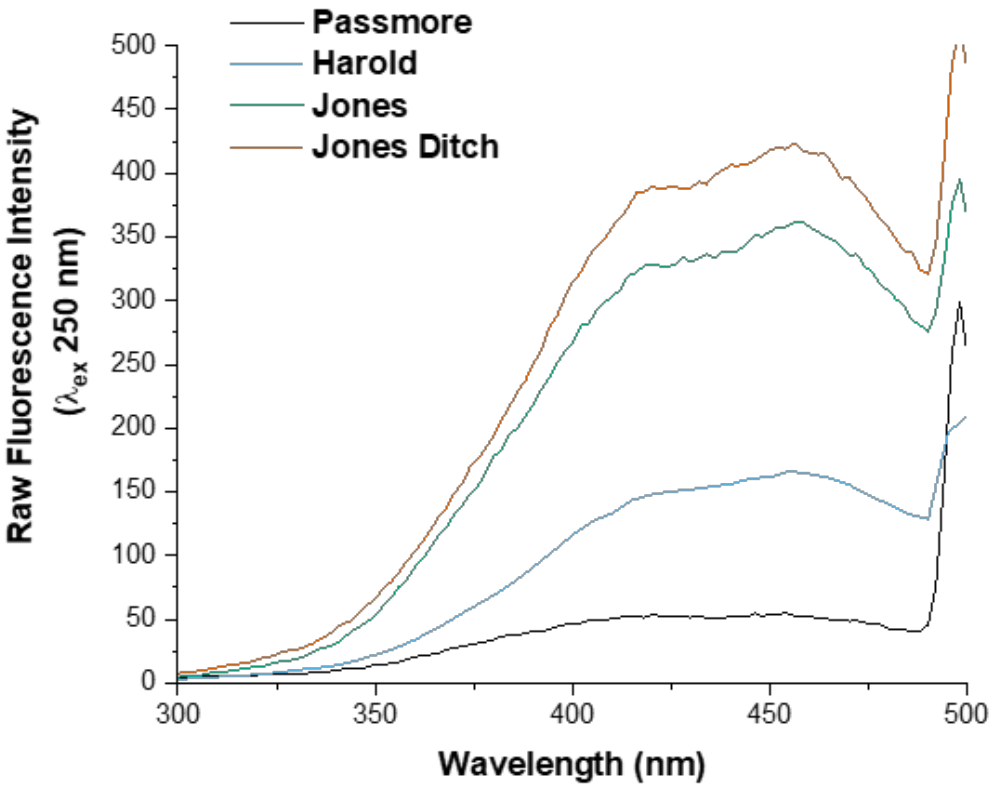

Figure 23. Raw fluorescence intensity (excitation at 250 nm) for the December 18, 2020 sampling event. Emission at ~422 nm was considered a spectral feature of fluorescent whitening agents in a region of the spectrum free from interference from naturally occurring humic acid (Boving, T., Meritt, D. L., Boothroyd, J. C., (2004). Fingerprinting sources of bacterial input into small residential watersheds: fate of fluorescent whitening agents. *Env Geol 46*, 228-232).

TABLES

Sub-Watershed	Geologic Unit	Dominant Soil Series	Parcels on Sewer (#)	Parcels on Septic (#)	Watershed Area (ha)	Septic System Density (# ha^{-1})
Ashburn	Triassic conglomerate	White Store, sandy loam	281	12	65.00	0.18
Asbury (sub)		Creedmoor, sandy loam	0	14	6.44	2.17
Asbury (full)		Creedmoor, sandy loam	0	19	8.51	2.23
Rondelay		White Store, sandy loam	42	8	19.13	0.42
Jones		White Store, sandy loam	0	60	83.30	0.72
Macon	Cambrian/Late Proterozoic biotite gneiss	Cecil, urban land complex	0	268	152.44	1.76
Passmore		Pacolet, urban land complex	0	33	29.80	1.11
Parkridge	Cambrian/Late Proterozoic biotite gneiss, Paleozoic/Late Proterozoic metamorphic rock	Pacolet, urban land complex	0	71	46.21	1.54
Baileywick	Cambrian/Late Proterozoic felsic gneiss	Cecil, urban land complex	6	34	38.52	0.88
Brookfield	Cambrian/Late Proterozoic felsic gneiss, Cambrian/Late Proterozoic biotite gneiss	Cecil, urban land complex	0	126	52.60	2.40
Green Bay	Cambrian/Late Proterozoic felsic metavolcanic rock	Herndon, silt loam	137	171	159.14	1.07
Harold		Herndon, silt loam	0	53	32.03	1.65
Clover Hill		Herndon, silt loam	72	13	21.57	0.60

Table 1. Sites selected for stream water quality monitoring. Bold and gray highlighted sites are predominantly served by municipal wastewater treatment plants and/or have the majority of their septic systems located in the upland areas away from the stream sampling point and are considered control watersheds that have minimal nutrient inputs associated with onsite wastewater treatment systems. Individual sub-watershed maps with approximate locations of onsite wastewater treatment systems are presented in Appendix C.* Jones Circle contained a ditch/small tributary that was also sampled and drained a neighborhood with sand filter systems. ** although Baileywick sub-watershed (**Fig. A-2**) had 34 septic systems, it was considered a sewered watershed because the sewered properties were adjacent to the stream and a large park resulted in the area near the stream to be buffered from septic systems. The other sewered watersheds were Ashburn and Clover Hill. Although Rondelay had numerous sites on sewer, since the septic systems were located close to the stream and to the monitoring point, this was considered a low-density septic watershed.

	2011 North Carolina Land Cover Assessment									
Watershed	Barren	Cropland	Urban	Forest	Grass	Impervious Area	Shrubland	Open Water	Wetland	% sandy loam soils
Ashburn	2.56%	7.57%	63.80%	18.29%	7.82%	17.90%	0.00%	0.00%	0.00%	86
Asbury (sub)	0.00%	8.55%	38.60%	42.93%	9.93%	4.23%	0.00%	0.00%	0.00%	76
Asbury (full)	0.00%	14.66%	29.60%	50.73%	4.40%	3.13%	0.00%	0.00%	0.59%	76
Rondelay	0.00%	4.46%	63.30%	25.66%	4.24%	12.60%	2.32%	0.00%	0.00%	88
Jones	0.00%	0.00%	52.40%	44.85%	2.10%	6.95%	0.68%	0.00%	0.00%	95
Macon	0.00%	0.00%	50.60%	47.84%	0.00%	3.32%	0.00%	0.00%	1.53%	14
Passmore	0.00%	18.13%	35.40%	42.87%	0.00%	5.23%	0.00%	3.62%	0.00%	37
Parkridge	0.00%	14.13%	53.40%	30.08%	0.00%	6.26%	0.00%	2.42%	0.00%	33
Baileywick	1.53%	0.00%	54.30%	33.11%	11.06%	9.76%	0.00%	0.00%	0.00%	15
Brookfield	0.00%	0.65%	43.10%	52.70%	0.80%	3.25%	0.00%	2.74%	0.00%	17
Green Bay	0.00%	6.08%	34.50%	52.50%	6.11%	3.95%	0.28%	0.52%	0.00%	0
Harold	0.00%	3.50%	29.10%	61.86%	5.56%	1.29%	0.00%	0.00%	0.00%	0
Clover Hill	0.00%	1.55%	92.90%	5.54%	0.00%	7.58%	0.00%	0.00%	0.00%	0

Table 2. Land cover data for sub-watersheds selected for study from the 2011 NC Land Cover Assessment.

Wastewater	TDN (mg/l)	Location	Soil Type	Reference	Approach
	55.00	Wake Co., NC		Humphrey et al. 2016a	annual mean
	54.00	Durham Co., NC		Humphrey et al. 2016a	annual mean
	61.00	US (Literature review)		Hazen and Sawyer 2013	* mean (range- 24-237)
	51.00	US (Literature review)		EPA-2002	*mid-range
	60.00	US (Literature review)		McCray et al. 2005	* mid-range
	60.00	US (Literature review)		Lowe et al. 2009	
Estimate for Falls Lake Watershed	57.50	**Median**			
Water Use					
	2.37 people /home	Durham Co.		US Census, 2019	
	2.62 people/home	Wake Co.		US Census, 2019	
	2.5 people/home	Durham/Wake Co. Average			
	138 gallons per household/day	National Average		DeOreo et al. 2016	
Per capita water use	55.2 gallons/person/day	Durham/Wake Co. Average		DeOreo et al. 2016	
	209 liters/person/day	Durham/Wake Co. Average			
Per capita nitrogen load to soil	12017.50	mg/person/day			
	0.01	kg/person/day			
	4.39	kg/person/yr			
	10.97	kg/household/yr			
N- mass reduction in soil treatment unit	0.74	Wake Co., NC	Clay loam	Humphrey et al. 2016a	conventional system
	0.47	Durham Co., NC	Silt loam	Humphrey et al. 2016a	conventional system
	0.61	Griffin, GA	Sandy clay loam-clay	Bradshaw and Radcliffe, 2013	conventional system - research facility
	0.34	Model (60 cm dtw)	Loam	D'Amato et al. 2016	conventional system- design hydraulic loading rate
	0.54	Model (60 cm dtw)	Clay	D'Amato et al. 2016	conventional system- design hydraulic loading rate
	0.59	Model (60 cm dtw)	Loam	D'Amato et al. 2016	conventional system- 1/2 design hydraulic loading rate
	0.80	Model (60 cm dtw)	Clay	D'Amato et al. 2016	conventional system- 1/2 design hydraulic loading rate
	0.52	Model (78 cm dtw)	Loam	Radcliffe and Bradshaw 2013	conventional system - Hydrus model- 2yr simulation
	0.75	Model (78 cm dtw)	Clay	Radcliffe and Bradshaw 2013	conventional system - Hydrus model- 2yr simulation
	0.80	Model (100 cm dtw)	Clay loam	McCray et al. 2009	field data and model (low hydraulic loading rate-0.46 cm/d)
	0.55	Model (100 cm dtw)	Clay loam	McCray et al. 2009	field data and model (high hydraulic loading rate-2.16 cm/d)
	0.59	**Median**			
Median N-Mass Load to Surficial Aquifer	1.80	per capita (kg/person/yr)			
	4.50	per household (kg/residence/yr)			
N-Mass Reduction from drainfield to stream					
	0.77	Wake Co., NC	Clay loam	Humphrey et al. 2016	conventional system
	0.63	Durham Co., NC	Silt loam	Humphrey et al. 2016	conventional system
	0.88	Durham Co., NC	Watershed-scale	Iverson et al. 2018	conventional system
	0.77	**Median**			
Median N-Mass Loading to Stream	1.01	per capita (kg/person/yr)			
	2.52	per household (kg/residence/yr)			
Just Reduction in Aquifer to Stream	0.03	Wake Co., NC	Clay loam	Humphrey et al. 2016	conventional system
	0.15	Durham Co., NC	Silt loam	Humphrey et al. 2016	conventional system

Table 3. Wastewater Nitrogen Loading Estimates for Conventional Onsite Systems

Wastewater	PO4-P or TP (mg/l)	Location	Soil Type	Reference	Approach
PO4-P	6.06	Wake Co., NC		Humphrey et al. 2016	annual mean
PO4-P	7.78	Durham Co., NC		Humphrey et al. 2016	annual mean
PO4-P	5.66	Durham Co., NC		Humphrey et al. 2016	annual mean
PO4-P	5.68	Durham Co., NC		Humphrey et al. 2016	annual mean
PO4-P	9.00	US (Literature review)		McCray et al. 2005	* mean
TP	8.60	US (Literature review)		EPA-2002	*mid-range
TP	12.00	US (Literature review)		Hazen and Sawyer 2013	* mean
TP	9.80	US (Literature review)		Lowe et al. 2009	
Estimate for Falls Lake Watershed	**8.19**	**Median**			
Water Use					
	2.37 people /home	Durham Co.		US Census, 2019	
	2.62 people/home	Wake Co.		US Census, 2019	
	2.5 people/home	Durham/Wake Co. Average			
	138 gallons per household/day	National Average		DeOreo et al. 2016	
Per capita water use	**55.2 gallons/person/day**	Durham/Wake Co. Average		DeOreo et al. 2016	
	209 liters/day	Durham/Wake Co. Average			
Per capita phosphorus load to soil	1711.71	mg/person/day			
	0.00	kg/person/day			
	0.62	kg/person/yr			
	1.56	kg/household/yr			
P- mass reduction in soil treatment unit	1.00	Wake Co., NC	Clay loam	Humphrey et al. 2016	conventional system
	0.98	Durham Co., NC	Silt loam	Humphrey et al. 2016	conventional system
	0.87	Soil column	Clay Loam	Karathanasis et al 2006	column
	0.96	Soil column	Silty Clay	Karathanasis et al 2006	column
	0.95	Soil column	Silty Clay	Karathanasis et al 2006	column
	0.97	**Median**			
Median P-Mass Loading to Surficial Aquifer	**0.02**	per capita (kg/person/yr)			
	0.04	per household (kg/residence/yr)			
P-Mass Reduction from drainfield to stream					
	0.98	Wake Co., NC	Clay loam	Humphrey et al. 2016	conventional system
	0.94	Durham Co., NC	Silt loam	Humphrey et al. 2016	conventional system
	0.92	Durham Co., NC	Watershed-scale	Iverson et al. 2018	conventional systems
	0.94	**Median**			
Median P-Mass Loading to Stream	**0.04**	per capita (kg/person/yr)			
	0.09	per household (kg/residence/yr)			

Table 4. Wastewater Phosphorus Loading Estimates for Conventional Onsite Systems

Watershed	OWTS (#)	WS Area (ha)	OWTS Density (# ha^{-1})	Med. TDN (mg/l)	Med. PO4-P (mg/l)	Med. Q l/yr	Med. OWTS N (kg/person/yr)	Med. OWTS PO4-P (kg/person/yr)	% N reduction	% P reduction
Asbury	19	8.51	2.23	1.21	0.09	75904881	2.65	0.19	39.38	68.85
Rondelay	8	19.13	0.42	0.18	0.04	33040948	0.08	0.00	98.22	100.21
Jones	60	83.30	0.72	0.89	0.12	169669735	1.07	0.11	75.67	82.27
Jones Ditch	24	8.00	3.00	8.24	0.50	17849376	1.18	0.06	73.14	90.31
HD1*	28	15.10	1.86	2.49	0.12	34631579	0.94	0.06	78.54	90.42
Macon	268	152.44	1.76	0.34	0.01	475521757	0.15	0.00	96.56	100.00
Passmore**	33	29.80	1.11	17.53	0.02	151809763	30.2 (<1.8)	0.00	0.00	100.00
Parkridge**	71	46.21	1.54	9.16	0.01	129484798	7.7 (<1.8)	0.00	0.00	100.00
Brookfield	126	52.60	2.40	0.40	0.01	125019805	0.03	0.00	99.39	100.00
Green Bay	171	159.14	1.07	0.28	0.04	509009205	0.21	0.02	95.26	97.57
Harold	53	32.03	1.65	1.48	0.13	80369874	1.11	0.08	74.75	86.57
2015-16 (LC)										
HD1	28	15.08	1.86	2.99	0.11	26871612	0.71	0.03	83.71	95.45
HD2	16	8.51	1.88	2.15	0.06	26789958	1.00	0.03	77.06	94.97
SF1	35	83.30	0.42	3.18	0.27	42685333	0.02	0.09	99.57	85.25
SF2	15	40.01	0.37	1.40	0.05	30753007	0.00	0.00	100.00	100.00
SF3	74	334.83	0.22	1.04	0.03	71497897	0.00	0.00	100.00	100.00
LD1	20	220.90	0.09	1.13	0.01	353599294	0.00	0.00	100.00	100.00
LD2	141	1178.54	0.12	1.01	0.02	1705627335	0.10	0.01	97.61	98.94
LD3	594	2282.83	0.26	1.07	0.02	4402483122	0.07	0.01	98.42	98.86
LD4	31	183.60	0.17	0.97	0.02	133949791	0.00	0.00	100.00	100.00
LD5	125	834.71	0.15	0.96	0.02	634029009	0.00	0.00	100.00	100.00
LD6	7	19.13	0.37	1.20	0.02	40194472	3.72	0.04	15.17	93.70

Table 5. Potential onsite wastewater nutrient loading estimates for conventional onsite systems based on sub-watershed stream nutrient concentration and flow data (subtracted median loads from sewered/control watersheds to help isolate the load that could potentially be attributed to onsite systems). The upper portion of the table features estimates based on stream data collected during the current study and the lower portion represents data from an earlier study in Durham Co. in the Lick Creek (LC) watershed from 2015-2016 (Documented in Iverson et al. 2018). *HD1 included in the upper portion of the table shows results for a long-term monitored site from 2015-2019. The Passmore and Park Ridge sites appeared to be affected by additional nutrient sources (agricultural and lawn fertilizer). Based on wastewater input data, the estimates of >7 kg-N/person/yr at these sites would be too large (even if no treatment occurred in the soils), therefore an additional conservative estimate (1.8 kg-N/person/yr) is included based on literature values (Table 3).

System Type	Reference	Wastewater TDN (mg L^{-1})	Wastewater PO_4-P (mg L^{-1})	Effluent/Overflow[a] TDN (mg L^{-1})	Effluent/Overflow[a] PO_4-P (mg L^{-1})	Treatment Efficiency TDN (%)	Treatment Efficiency PO_4-P (%)
Sand Filter Systems							
Single Pass Sand Filter, Site 1[b]	Humphrey et al. (2016a, 2016b)	52.39	5.66	3.80	0.11	93 (50)	98 (83)
Single Pass Sand Filter, Site 2	Humphrey et al. (2016a, 2016b)	51.49	5.68	15.47	1.07	70.0	81.2
Single Pass Sand Filter Mean	Humphrey et al. (2016a, 2016b)	**51.94**	**5.67**	**9.64**	**0.59**	**81.0**	**90.0**
Recirculating Sand Filter, Type II	Unpublished data (Lick Ck)	45.94	2.76	9.55	0.89	79.2	67.7
Sand Filter Systems Mean		**49.94**	**4.70**	**9.61**	**0.69**	**74.6**	**74.5**
Malfunctioning Conventional Systems							
Malfunctioning Pump to Conventional Septic System	Unpublished data (Lick Ck)	65.67	4.98	48.91	3.45	25.5	30.7
Seep that Discharges to Tributary of Lick Ck	Unpublished data (Lick Ck)	95.39	7.46	14.22	1.86	85.0	75.0
Seep that Discharges to Tributary (Craven Co.)	O'Driscoll et al. 2019	57.30	n/a	12.30	n/a	79	n/a
Repaired Conventional System[c]	Humphrey et al. 2019	34.88	1.08	4.66	0.00	87	99.6
Gray Water Pipe Discharges to Tributary of Lick Ck	Unpublished data (Lick Ck)	n/a	n/a	4.89	1.22	n/a	n/a
Package Plants	Mahoney (2016)						
Extended Aeration 1	Mahoney (2016)	30.60	2.70	7.00	3.20	77	-19
Extended Aeration 2	Mahoney (2016)	55.10	3.90	24.70	2.70	55	31
Extended Aeration 3	Mahoney (2016)	61.30	4.40	7.00	2.20	89	50
Advantex 1	Mahoney (2016)	38.60	4.00	16.10	2.80	58	30
Advantex 2	Mahoney (2016)	49.40	5.00	7.10	4.30	86	14
Sequencing Batch Reactor 1	Mahoney (2016)	55.30	8.10	12.40	6.40	78	21
Sequencing Batch Reactor 2	Mahoney (2016)	49.20	5.80	10.70	2.70	78	53
Package Plants (Carteret Co.) Mean		**48.50**	**4.84**	**12.14**	**3.47**	**74**	**26**
Orange Co.	NC DEQ 2018[d]	n/a	n/a	20.05	2.85	n/a	n/a
Wake Co. (1)	NC DEQ 2018[d]	n/a	n/a	30.41	0.6	n/a	n/a
Wake Co. (2)	NC DEQ 2018[d]	n/a	n/a	42	1.3	n/a	n/a
Durham Co.	NC DEQ 2018[d]	n/a	n/a	5.25	0.67	n/a	n/a
Package Plants (Falls Lake) Mean				**24.43**	**1.34**		

[a]= Overflow represents the concentration of nutrients in wastewater that surfaced from a malfunctioning septic system.

[b]= Treatment efficiency in parenthesis are load reductions that were estimated using mixing models.

[c]= Effluent was collected as soil water from suction lysimeters at 25-55 cm below the trench.

[d]= Median nutrient concentrations from 4 PTPs (0.006-0.25 MGD) in Falls Lake watershed (2014-2018; draining to Lower Barton Ck, Upper Barton Ck, Stony Ck, and Panther Ck).

Table 6. Sand Filter, Malfunctioning System, and Package Plant Attenuation and Loading Estimates

Appendices

Appendix A- Septic system density map and individual maps for sub-watersheds

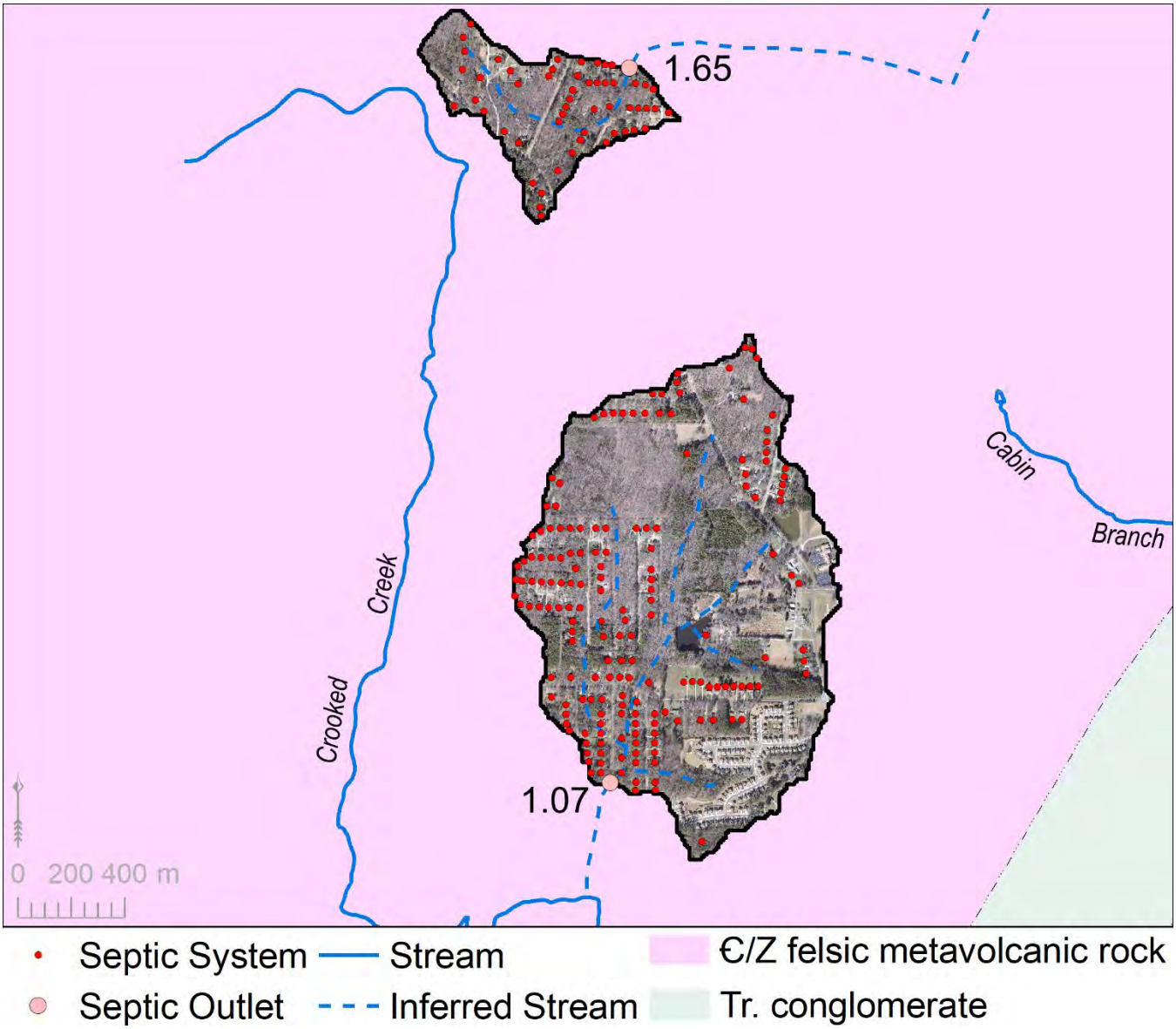

Figure A-1. Sites in North Durham, Durham Co. NC. Northernmost site is Harold Dr. and southernmost site is Green Bay Dr.

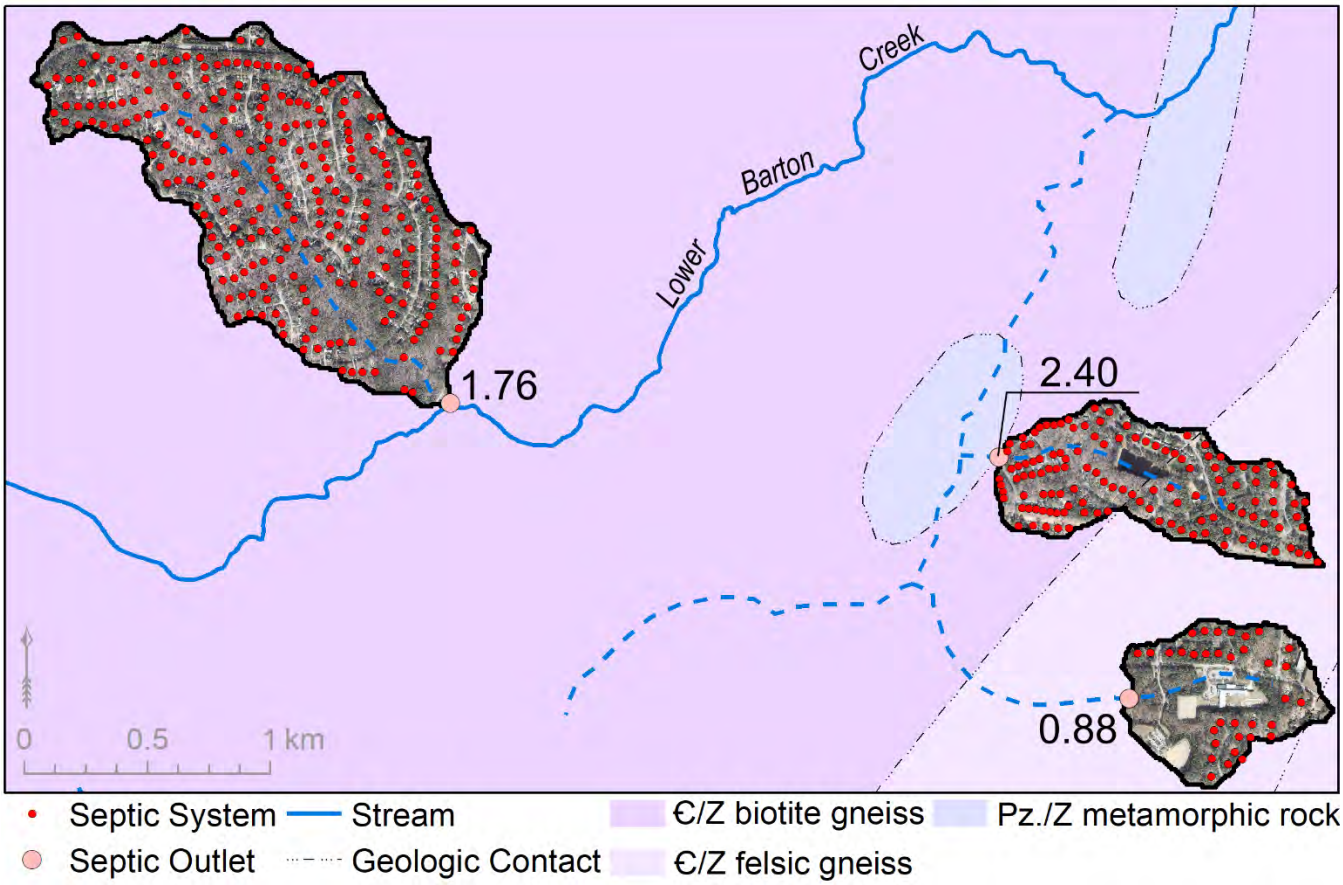

Figure A-2. Sites in Wake Co. cluster tributaries draining to Lower Barton Creek. Northernmost site is Macon Rd. and southernmost sites are Brookfield (2.4 systems/ha) and Baileywick Park.

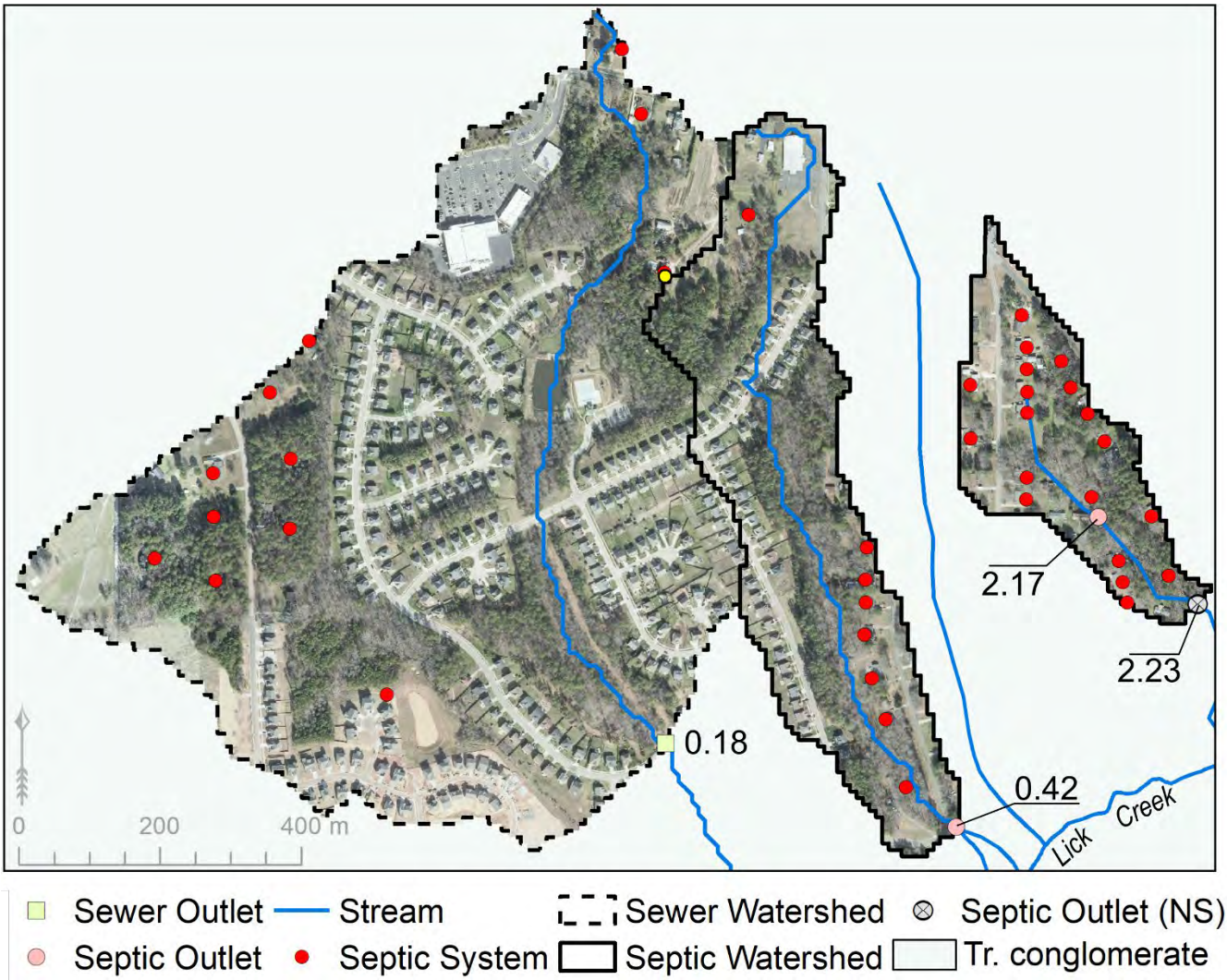

Figure A-3. Sites in Durham Co. cluster tributaries draining to Lick Creek. Westernmost site is Ashburn Rd. and the adjacent sub-watershed is Rondelay (0.42 systems/ha). The easternmost sub-watershed is Asbury Rd. HD1 site monitored for long-term nutrient exports is located on the tributary between the Rondelay Rd. and Asbury Rd. sites. The small yellow circle indicates a site that utilized a sand filter system.

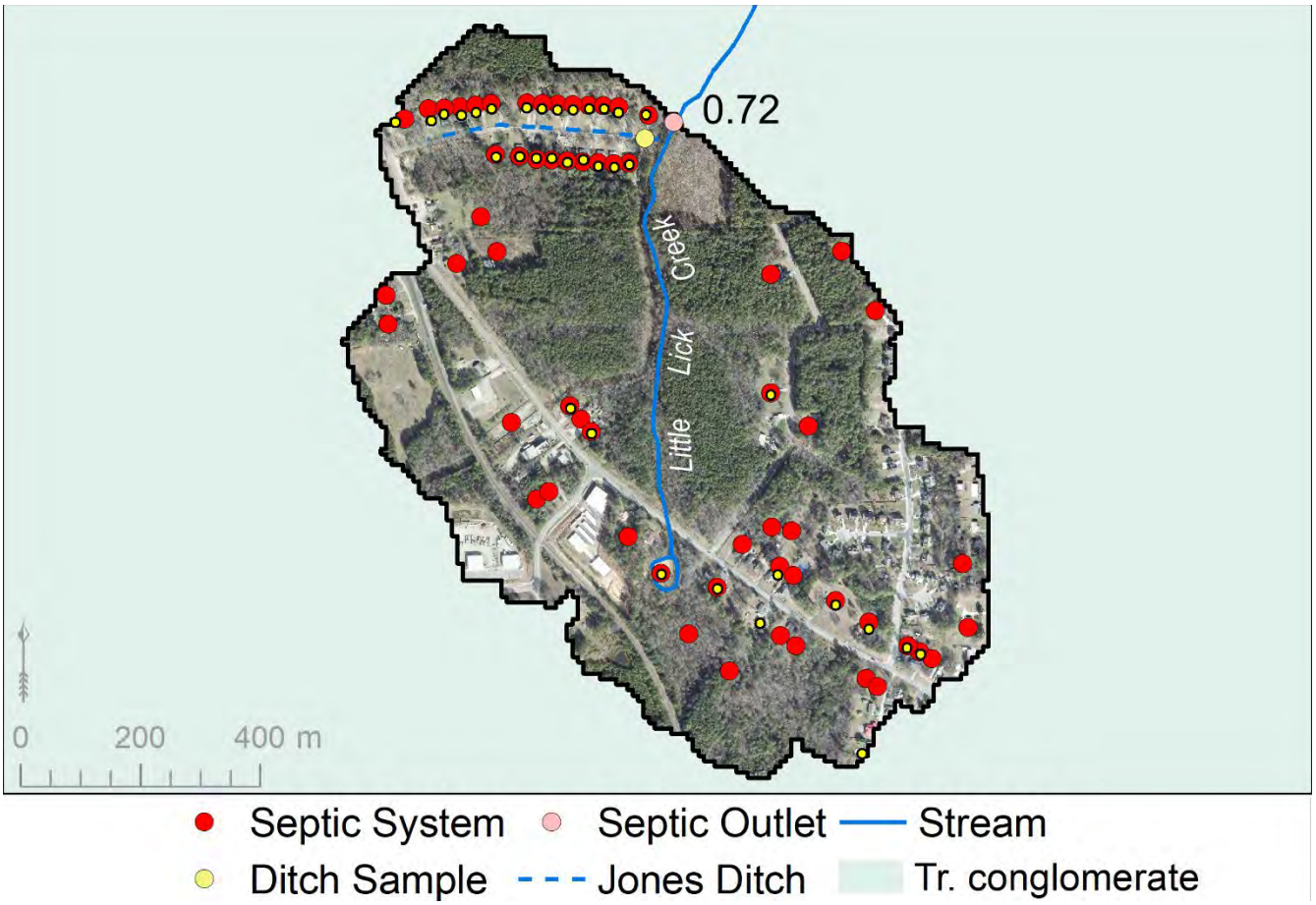

Figure A-4. Jones Circle Sites in Durham Co. along Little Lick Creek. Samples were collected along Little Lick Creek and from a ditch along Jones Circle that flowed into the Creek. The small yellow circles indicate sites that utilized sand filter systems.

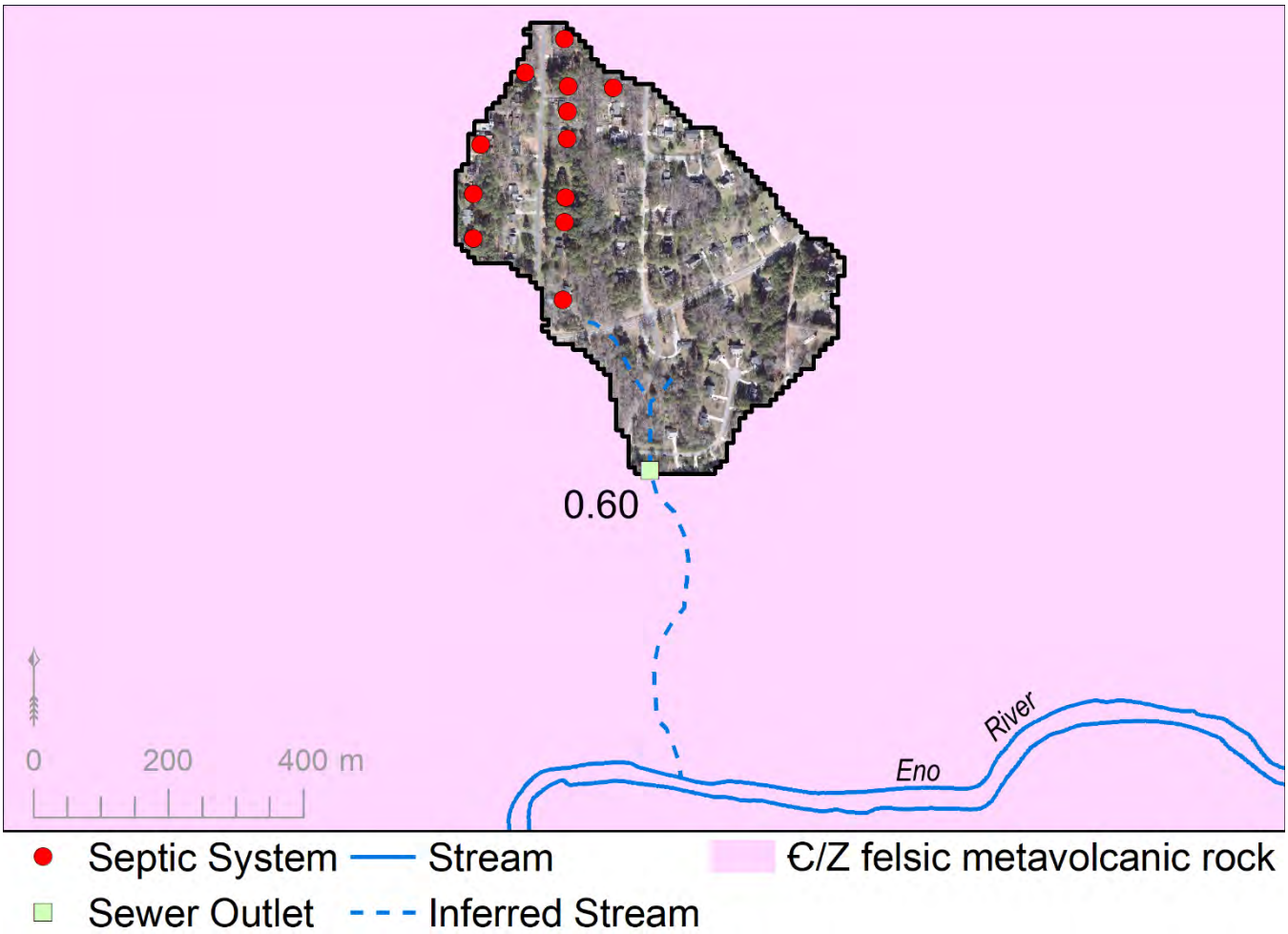

Figure A-5. Clover Hill Rd. site in Durham Co.

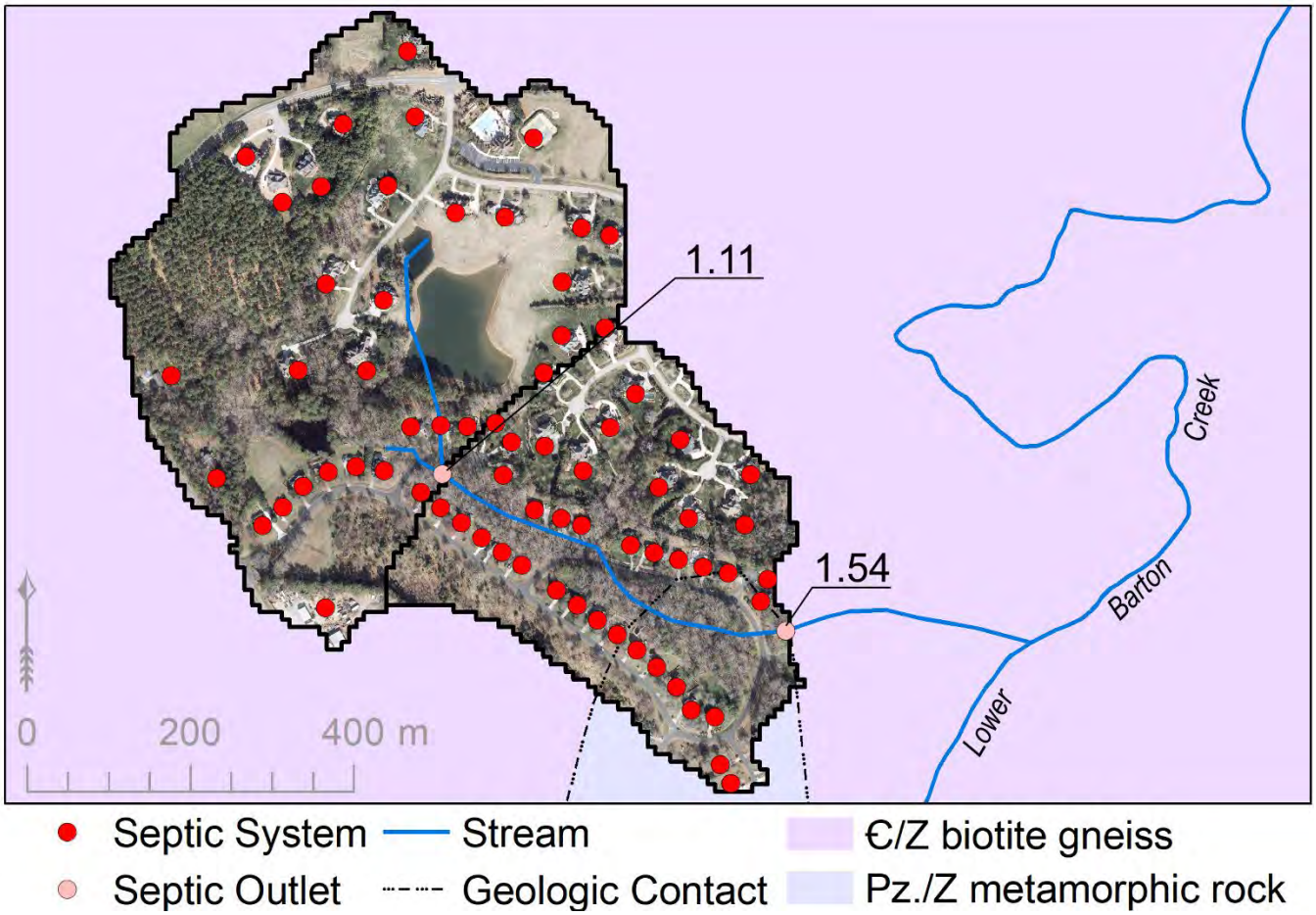

Figure A-6. Passmore (upstream) and Park Ridge Rd. (downstream) sites along tributary draining to Lower Barton Creek in Wake County, NC.

Appendix B- Site Photos

Passmore

Jones

Jones- ditch

Macon

Park Ridge

Harold

Green Bay

Clover Hill

Brookfield

Baileywick

Ashburn

Rondelay

Asbury

Appendix C- Water Quality Data

Date	Site	Baileywick	Brookfield	Passmore	Park Ridge	Macon Rd	Asbury	Rondelay	Ashburn	Jones	Jones Ditch	Harold	Green Bay	Clover Hill
11/15/2019	Gage (ft)							0.61						
11/15/2019	Flow (cfs)		0.02	0.09	0.17	0.12	0.12	0.01	1.44	0.24	0.00	0.06	0.54	0.12
11/15/2019	Temp - C		9.61	11.96	10.50	7.18	8.62	9.20	8.16	7.54	9.62	9.34	8.15	10.92
11/15/2019	Conductivity uS		88.00	256.00	181.00	99.00	87.00	84.00	100.00	138.00	243.00	356.00	87.00	125.00
11/15/2019	DO %		88.50	68.10	78.30	90.80	90.70	66.90	86.30	79.50	62.40	69.30	88.50	84.10
11/15/2019	DO mg/l		10.07	7.39	8.73	10.97	10.59	7.69	10.16	9.51	7.09	7.62	10.49	9.25
11/15/2019	pH		7.01	6.56	6.86	7.19	7.02	6.41	7.18	7.15	6.70	6.90	7.10	6.66
11/15/2019	ORP (mV)		159.50	179.90	144.30	78.10	205.10	214.90	187.40	192.60	203.30	208.30	206.20	216.90
11/15/2019	NO3-N mg/l (Lachat)		1.10	21.70	10.40	0.26	0.70	1.95	0.54	1.32	11.60	3.37	0.69	0.92
11/15/2019	NaCl-Cl mg/L (Lachat)		14.00	25.00	21.00	16.00	9.00	12.00	10.00	22.00	25.00	52.00	13.00	13.00
11/15/2019	NH4-N mg/l		0.02	0.03	0.02	0.01	0.01	0.01	0.03	0.85	0.14	1.81	0.05	0.06
11/15/2019	NO3-N mg/l		0.76	21.89	8.00	0.11	0.36	1.23	0.25	1.05	11.55	2.54	0.40	0.55
11/15/2019	DKN mg/l		0.00	0.11	0.12	0.00	0.23	0.41	0.31	1.30	1.06	2.16	0.24	0.26
11/15/2019	PO4-P mg/l		0.02	0.02	0.00	0.01	0.06	0.09	0.09	0.18	0.44	0.41	0.07	0.09
11/15/2019	TDP mg/l		0.02	0.03	0.03	0.02	0.09	0.10	0.10	0.20	0.46	0.44	0.08	0.10
11/15/2019	Cl mg/l		13.11	26.48	18.94	15.59	9.30	11.26	9.94	24.99	29.90	43.48	10.51	11.45
11/15/2019	DOC mg/l		2.72	2.18	2.62	3.70	7.92	10.37	6.36	11.25	11.60	8.20	5.66	5.82
11/15/2019	TDN mg/l		0.38	21.04	7.42	0.00	0.25	1.02	0.16	1.51	11.20	3.66	0.28	0.39
11/15/2019	DON (DKN-NH4) mg/l		0.00	0.09	0.09	0.00	0.21	0.40	0.28	0.46	0.92	0.34	0.19	0.20
12/18/2019	Gage (ft)							0.78						
12/18/2019	Flow (cfs)	0.10	0.08	0.16	0.12	0.53	0.05	0.05	0.50	0.14	0.01	0.18	0.60	0.03
12/18/2019	Temp - C	11.09	9.32	11.35	9.82	8.45	9.48	8.55	9.09	7.60	8.19	8.92	9.14	9.58
12/18/2019	Conductivity uS	48.00	91.00	246.00	180.00	107.00	131.00	123.00	183.00	147.00	171.00	103.00	70.00	82.00
12/18/2019	DO %	89.00	107.90	97.00	108.80	117.80	103.80	84.50	98.70	101.20	62.60	80.20	107.20	84.00
12/18/2019	DO mg/l	9.82	12.36	10.45	12.35	13.80	11.91	9.89	11.37	12.14	7.40	9.33	12.35	9.61
12/18/2019	pH	5.62	6.56	6.62	7.05	7.17	7.33	6.94	7.18	7.87	6.81	7.23	6.80	6.79
12/18/2019	ORP (mV)	221.90	186.60	151.20	157.60	143.50	176.60	182.90	204.30	217.30	87.30	175.30	170.40	154.40
12/18/2019	NH4-N mg/l	0.02	0.03	0.02	0.02	0.01	0.17	0.02	0.03	0.61	*	0.80	0.05	0.03
12/18/2019	NO3-N mg/l	0.33	0.80	19.79	10.80	0.66	1.44	0.23	0.16	0.52	*	1.14	0.33	0.43
12/18/2019	DKN mg/l	0.05	0.00	0.03	0.02	0.04	0.69	0.41	0.41	1.19	*	1.31	0.29	0.53
12/18/2019	PO4-P mg/l	0.01	0.01	0.02	0.01	0.01	0.10	0.04	0.03	0.12	*	0.13	0.04	0.03
12/18/2019	TP mg/l	0.02	0.02	0.02	0.03	0.02	0.18	0.08	0.11	0.22	*	0.22	0.07	0.06
12/18/2019	TDP mg/l	0.02	0.02	0.02	0.01	0.02	0.14	0.07	0.07	0.17	*	0.17	0.06	0.05
12/18/2019	Cl mg/l	7.77	16.05	24.45	22.70	21.45	14.62	15.73	18.02	21.96	*	17.82	10.88	8.32
12/18/2019	DOC mg/l	3.77	2.94	2.03	2.13	4.97	8.56	10.25	7.40	12.60	*	9.42	8.81	9.24
12/18/2019	TDN mg/l	0.08	0.40	17.53	9.38	0.34	1.21	0.18	0.12	0.87	*	1.48	0.20	0.30
12/18/2019	DON (DKN-NH4) mg/l	0.04	0.00	0.01	0.00	0.02	0.52	0.39	0.39	0.58	*	0.50	0.23	0.50
2/10/2020	Gage (ft)							0.96						
2/10/2020	Flow (cfs)	0.20	0.20	0.34	0.40	0.72	0.16	0.12	0.49	0.48	0.01	0.12	0.80	0.08
2/10/2020	Temp - C	10.90	8.88	11.53	9.93	8.14	10.65	9.63	10.24	7.61	9.34	9.51	10.55	9.50
2/10/2020	Conductivity uS	59.00	86.00	192.00	162.00	81.00	113.00	195.00	117.00	112.00	139.00	88.00	66.00	93.00
2/10/2020	DO %	83.50	99.30	94.50	103.00	107.70	98.70	98.70	100.40	99.50	81.50	92.60	105.10	78.10
2/10/2020	DO mg/l	9.22	11.49	10.28	11.62	12.69	10.97	11.20	11.23	11.83	9.34	10.56	11.70	8.91
2/10/2020	pH	5.15	6.32	6.43	6.44	6.41	6.63	6.94	6.86	6.84	6.22	6.56	6.47	6.54
2/10/2020	ORP (mV)	279.70	223.70	224.00	196.10	147.30	167.00	165.00	167.90	156.90	174.60	182.80	177.30	166.20
2/10/2020	NH4-N mg/l	0.02	0.03	0.03	0.03	0.02	0.46	0.05	0.03	0.61	0.04	0.11	0.04	0.03
2/10/2020	NO3-N mg/l	0.35	0.80	15.76	10.35	0.91	1.28	0.35	0.13	0.69	5.89	0.66	0.81	0.41
2/10/2020	DKN mg/l	0.00	0.00	0.21	0.02	0.03	0.73	0.39	0.26	0.98	0.85	0.21	0.14	0.14
2/10/2020	PO4-P mg/l	0.01	0.01	0.02	0.01	0.01	0.09	0.02	0.02	0.08	0.56	0.05	0.03	0.02
2/10/2020	TP mg/l	0.01	0.02	0.06	0.02	0.01	0.17	0.06	0.06	0.16	0.75	0.10	0.05	0.04
2/10/2020	TDP mg/l	0.02	0.02	0.03	0.02	0.02	0.13	0.06	0.05	0.13	0.62	0.09	0.05	0.04
2/10/2020	Cl mg/l	6.04	16.05	21.50	20.38	14.80	13.23	20.41	21.13	17.68	15.72	16.70	10.87	11.14
2/10/2020	DOC mg/l	4.39	2.94	3.12	2.32	3.63	7.55	7.37	7.35	10.94	9.78	4.89	5.59	5.07
2/10/2020	TDN mg/l	0.08	0.40	13.83	9.16	0.43	1.25	0.17	0.05	0.89	5.29	0.42	0.48	0.17
2/10/2020	DON (DKN-NH4) mg/l	0.00	0.00	0.18	0.00	0.01	0.27	0.34	0.23	0.37	0.81	0.09	0.10	0.11
6/2/2020	Gage (ft)							0.70						
6/2/2020	Flow (cfs)	0.06	0.21	0.18	0.12	0.54	0.05	0.02	0.12	0.10	0.02	0.04	0.45	0.03
6/2/2020	Temp - C	16.91	16.73	17.80	16.52	16.80	18.60	19.93	18.57	16.77	21.63	17.97	18.83	17.22
6/2/2020	Conductivity uS	60.00	135.00	235.00	200.00	112.00	175.00	445.00	462.00	281.00	434.00	143.00	111.00	151.00
6/2/2020	DO %	72.70	87.90	78.10	84.40	94.00	87.70	104.30	72.40	43.10	34.10	76.00	90.50	62.10
6/2/2020	DO mg/l	7.03	8.54	7.50	8.25	9.10	8.20	9.48	6.76	4.15	3.00	7.18	8.41	5.96
6/2/2020	pH	5.03	5.71	5.78	5.89	6.04	6.27	6.90	6.88	6.59	6.51	6.42	6.77	6.60
6/2/2020	ORP (mV)	212.00	164.30	228.00	178.50	159.50	172.00	168.00	189.80	281.00	215.00	125.50	102.70	694.00
6/2/2020	NO3-N (Lachat) mg/l	0.63	1.26	17.10	10.50	0.92	0.40	0.94	0.70	1.21	1.96	1.35	1.00	0.51
6/2/2020	NaCl-Cl (Lachat) mg/L	8.00	16.00	18.00	17.00	13.00	30.00	30.00	20.00	28.00	39.00	15.00	10.00	13.00
6/2/2020	NH4-N mg/l	*	*	*	*	*	*	*	*	*	*	*	*	*
6/2/2020	NO3-N mg/l	*	*	*	*	*	*	*	*	*	*	*	*	*
6/2/2020	DKN mg/l	*	*	*	*	*	*	*	*	*	*	*	*	*
6/2/2020	PO4-P mg/l	*	*	*	*	*	*	*	*	*	*	*	*	*
6/2/2020	TP mg/l	*	*	*	*	*	*	*	*	*	*	*	*	*
6/2/2020	TDP mg/l	*	*	*	*	*	*	*	*	*	*	*	*	*
6/2/2020	Cl mg/l	*	*	*	*	*	*	*	*	*	*	*	*	*
6/2/2020	DOC mg/l	*	*	*	*	*	*	*	*	*	*	*	*	*
6/2/2020	TDN mg/l	*	*	*	*	*	*	*	*	*	*	*	*	*
6/2/2020	DON (DKN-NH4) mg/l	*	*	*	*	*	*	*	*	*	*	*	*	*

* samples stored and frozen for future analyses when lab reopens