# Evaluating Pollutant Treatment Efficiency by Onsite Wastewater Treatment Systems in the North Carolina Piedmont

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## Executive Summary

The goal of this study was to characterize septic system performance at 3 sites in the Raleigh Belt geological setting. This project was an expansion project on the 2021 – 2022 NC Policy Collaboratory project that funded the instrumentation of 8 groundwater piezometers across 2 sites. In the current project, an additional site (Site 400) was instrumented with 3 piezometers and 3 additional piezometers were installed at Site 100. Sites 100 and 300 were adjacent to a stream. Thus, a total of 3 septic tanks, 14 piezometers, and 2 streams were monitored between the 3 sites. The current project supported 6 sampling events occurring approximately monthly from October 2022 – April 2023. During each sampling event, physicochemical parameters were collected in the field prior to sampling. Water samples were collected from each tank, piezometer, and stream to be analyzed for nitrogen, phosphorus, and *Escherichia coli* concentrations. Wastewater and groundwater samples were analyzed for ammonium, nitrate, total dissolved nitrogen (TDN), orthophosphate, total dissolved phosphorus (TDP), dissolved organic carbon (DOC), chloride (Cl), and *E. coli*. The same analytes were also measured for stream samples in addition to particulate nitrogen (PN), particulate phosphorus (PP), and total suspended solids (TSS). Nutrient and *E. coli* analytical processes were conducted at the Environmental Research Laboratory and Water Research Laboratory, respectively, at ECU. The concentrations of dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) were estimated by subtracting the concentration of dissolved inorganic species from TDN and TDP, respectively. Total nitrogen (TN) and total phosphorus (TP) concentrations were calculated by adding the concentrations of the total dissolved species with the particulate form. Septic system performance was assessed by proxy of concentration reduction of nutrient and bacteria and nutrient mass reduction in groundwater. Sub-watershed export of TN and TP masses and *E. coli* yields from the streams at Sites 100 and 300 were also estimated. Research highlights were included below to summarize the main trends observed. These highlights combined results from the current project with the 2021 – 2022 NC Policy Collaboratory to generate a more robust dataset. The appendix contains a summary of the major analyses only including data from the current project.

<u>Research Highlights</u>

- Wastewater contained the greatest median TDN, TDP, and *E. coli* concentrations of~~-~~, 62.15 mg $L^{-1}$, 7.23 mg $L^{-1}$, and 93,600 MPN 100 $mL^{-1}$, respectively.
- Groundwater near drainfields contained median TDN, TDP, and *E. coli* concentrations of 9.54 mg $L^{-1}$, 0.20 mg $L^{-1}$, and 1,426.4 MPN 100 $mL^{-1}$, respectively. This corresponded to a concentration reduction of 84.7%, 97.3%, and 98.5% for TDN, TDP, and *E. coli*, respectively, between tanks and drainfield groundwater.
- Downgradient groundwater (Site 100 only) contained a median concentration of 1.91 mg $L^{-1}$, 0.03 mg $L^{-1}$, and 45.2 MPN 100 $mL^{-1}$ for TDN, TDP, and *E. coli*,

respectively. This equated to a concentration reduction of 97.2%, 99.7%, and >99.9% for TDN, TDP, and *E. coli*, respectively, relative to wastewater.

- Dilution can affect concentration reduction; thus, nutrient mass reductions were also estimated. Estimates of TDN mass reduction varied between sites. Mass reduction of TDN ranged from 38.3 – 96.6% in drainfield groundwater and 24.6 – 87.9% in downgradient groundwater (Site 100).
- Septic systems were more effective at removing TDP compared to TDN. Mass reduction of TDP ranged from 85.3 – 97.9% in drainfield groundwater and 58.2 – 98.5% in downgradient groundwater (Site 100).
- Treatment efficiency data suggest that nitrogen was the most mobile contaminant, whereas both phosphorus and *E. coli* were typically better treated across all sites.
- Concentration and mass reduction estimates typically were greatest at Site 400, followed by Site 100 and Site 300. Differences in septic system performance between sites was likely driven by periods of malfunction.
- Mixing models suggested that mass removal processes accounted for most of the nutrient treatment at Site 400, which did not show any signs of malfunction. At Sites 100 and 300, models suggested that mass removal processes still occurred, but dilution accounted for a larger percentage of concentration reduction. Dilution and dispersion processes can reduce pollutant concentrations, but they are unable to remove the mass of nutrients, thus do not reduce nutrient transport.
- Nutrient and *E. coli* concentrations were elevated in the stream at Site 300 relative to Site 100. The median concentration of TN, TP, and *E. coli* at Site 100 was 2.45 mg $L^{-1}$, 0.11 mg $L^{-1}$, and 272.3 MPN 100 $mL^{-1}$, respectively. At Site 300, median concentrations were 5.29 mg $L^{-1}$, 0.59 mg $L^{-1}$, and 1,986.3 MPN 100 $mL^{-1}$, respectively.
- Nutrient mass exports and *E. coli* loadings from the stream at Site 300 tended to be greater than Site 100. The stream at Site 100 transported a median of 39 g $day^{-1}$, 2 g $day^{-1}$, and 60,095 MPN $min^{-1}$ of TN, TP, and *E. coli*, respectively. The stream at Site 300 transported a median of 67 g $day^{-1}$, 14 g $day^{-1}$, and 116,141 MPN $min^{-1}$ of TN, TP, and *E. coli*, respectively.
- Background groundwater contained a median concentration of 0.95 mg $L^{-1}$, 0.01 mg $L^{-1}$, and 4.2 MPN 100 $mL^{-1}$ for TDN, TDP, and *E. coli*, respectively. Groundwater and/or streams at all the sites contained elevated nutrient and *E. coli* concentrations relative to background groundwater.

These results suggest that septic systems can be significant sources of nutrients and *E. coli* to water resources downgradient of drainfields, especially if the system is experiencing malfunction. Site 300 typically contained the greatest concentrations of pollutants in groundwater and surface water. The septic system at this site malfunctioned (effluent surfaced above drainfield and entered the tank's freeboard) during most sampling events. Elevated concentrations of nutrients and *E. coli* in groundwater also translated to increased surface water concentrations and export of these pollutants. Thus, septic systems that experience routine or intermittent malfunction can be potentially significant sources of nutrients to groundwater and surface water. Furthermore, surface water impairment is more likely in sub-watersheds that have higher densities of septic systems.

# 1 Study Background

## 1.1 North Carolina Septic System Usage

Septic systems are an important wastewater management strategy for North Carolina's citizens, especially for people residing in areas where municipal sewers are unavailable or cost prohibitive. The number of active septic systems in North Carolina is difficult to pinpoint since these data are not typically archived in digital repositories. Based on the 1990 US Census, it was estimated that there were approximately 1.4 million systems in use, and this number increased to more than 1.8 million by 2002 [1]. As of 2010, approximately 5 million residents were served by septic systems in North Carolina, which constituted about 50% of the population [2]. Today, there are approximately 2 million systems in use [3, 4]. Conventional style, or modified conventional, are the most used system due to cost and their simplicity in design and operation. These types of systems use gravity to distribute wastewater from the house into the tank, and eventually from the tank into a series of drainfield trenches (or a drainfield bed) via a distribution device. Septic tank effluent is stored in the drainfield until it can infiltrate into underlying and adjacent soil, which is where most of the treatment occurs [5, 6]. Separation distance is a key factor that can affect treatment and it is measured as the distance between the trench bottom and a soil wetness feature or saturated soils, such as a seasonal high water table or presence of groundwater. In North Carolina, conventional septic systems require a minimum of 45 cm (18 in) or 30 cm (12 in) of separation distance for Group I soils (sandy soils) or Group II – IV soils (loamy – clayey soils), respectively [7]. Violating the minimum separation distance requirement is considered a system malfunction and can result in hydraulic failure. Hydraulic failures may present themselves as ponded effluent (standing water above or near drainfields) or "backed-up" wastewater within the septic tank or household. Both the malfunction rate and severity of malfunction can be affected by several factors, which include soil characteristics, household wastewater loading rates, solid waste load to tank, frequency of maintenance, septic system design or installation, landscaping practices (e.g., building/driving over septic components, planting vegetation with extensive root systems near system), or some other unlisted factor. In the US, septic system malfunction rate has been estimated to be < 7 – 20% [5, 8], but some communities have reported up to 70% failure rates [9]. System malfunctions can translate into a myriad of socioeconomic and environmental damages. Among these environmental factors includes inhibition of biogeochemical processes that reduce or remove pollutants that may pose a risk to the environment or public.

## 1.2 Environmental and Public Health Implications from Septic Systems

Septic systems are designed to collect, treat, and dispose of wastewater via infiltration of effluent into soil. Domestic wastewater contains elevated concentrations of nitrogen, phosphorus, pathogens, emerging contaminants, and other pollutants. Wastewater typically contains concentrations of total dissolved nitrogen (TDN) that are variable, ranging from approximately 26 – 95 mg $L^{-1}$ [5, 6, 10-12]. Total dissolved phosphorus (TDP) concentrations in domestic wastewater typically range from 0.14 – 32.49 mg $L^{-1}$ [13-15]. Concentrations of *Escherichia coli* (*E. coli*) exhibit the most variability in domestic wastewater, typically ranging from approximately 150 to 32,000,000 MPN 100 $mL^{-1}$ [16-19]. Most of the treatment in a septic system occurs within the soil adjacent and beneath drainfields. Thus, systems should be designed to maximize contact between the soil surface and septic tank effluent, thereby facilitating nutrient transformations and pathogen removal. O'Driscoll et al. [20] found that nitrogen and phosphate treatment efficiencies ranged from 74 – 100% and 90 – 100%, respectively, between the septic tank and adjacent streams. *E. coli* treatment is also highly effective if adequate separation distance exists between the drainfield trench bottom and water table. Humphrey et al. [19] found that >99% of *E. coli* was removed between the septic tank and groundwater adjacent to a stream. Despite high treatment efficiencies for nutrients and *E. coli*, past studies have found elevated concentrations of nutrients and/or pathogens in water resources downgradient from septic systems [11, 14, 20-32], especially in areas with a high density of systems [20, 23, 29, 30]. Most of these studies focused on coastal plain settings where soils are sandier, although recently more work evaluating nutrient and/or pathogen inputs from septic systems has been conducted in Piedmont settings [20, 23, 26, 28-31]. Many of these studies conducted in North Carolina's Piedmont have occurred in Triassic Basin geological settings, which tend to contain finer textured soils relative to soils found in Carolina Slate or Raleigh Belt geology. More information is needed to evaluate attenuation of nutrients and *E. coli* by septic systems located in Raleigh Belt geology where soils tend to have higher hydraulic conductivities, which may influence pollutant transport. These data are important to further understanding of lot-scale treatment of common pollutants in septic systems, which can help constrain model estimates of nutrient loading to important water resources, such as Falls Lake, Jordan Lake, and other surface waters with similar geological, soil, and weather characteristics.

### 1.3 Study Goals and Objectives and Updated Scope of Work

The goal of this study was to quantify pollutant treatment by septic systems located in the Raleigh Belt geological setting. This study expands upon a previous study funded by the 2021 – 2022 NC Policy Collaboratory by leveraging and adding to existing site infrastructure to generate a longer-term dataset of system performance. The objectives were to: 1) quantify nitrogen, phosphorus, and *E. coli* concentrations in groundwater and streams downgradient of septic systems; 2) evaluate septic system performance by proxy of pollutant concentration and/or mass reductions in groundwater; and 3) estimate nutrient mass exports and *E. coli* loadings in streams adjacent to septic systems.

## 2 Methods

2.1 Study Area

Researchers from ECU collaborated with Wake County Environmental Services to identify potential study sites. These sites were then verified to ensure that they were within either Carolina Slate or Raleigh Belt geology via the state geological map. These geological settings tend to be comprised of metamorphic or igneous rocks [33]. After receiving a list of potential volunteers, property owners were contacted to determine if they would be interested in volunteering access to their system. An interested landowner of a mobile home community located in the Lake Benson Watershed was identified. This location offered numerous septic systems to study that were in similar geology as the eastern regions of the Falls Lake Watershed (Fig. 1). This community is approximately 5 miles upstream of Lake Benson Park. The Park is an important recreational resource for the region, which includes the Town of Garner and City of Raleigh. There are several amenities at the park that include a dog park, two playgrounds, a picnic area, walking trails, and the park hosts numerous community events throughout the year. During the warm season (mid-March through October), the Park also offers boat rentals and accommodates anglers. The Lake is known to have issues with eutrophication in the late Summer or early Fall. The Google Maps page for Lake Benson Park contains publicly shared photos by park visitors that show several algal bloom events occurring at the Lake from 2019 – 2022 (Appendix A). The recreational importance of Lake Benson paired with its seasonal eutrophication make it an ideal candidate to improve understanding of nutrient and pathogen delivery to its tributaries.

The mobile home community was selected based on geological setting, the high number and density of septic systems within the community, and the proximity of the community to Lake Benson. The study site is located west of Garner, NC near a geological contact between Cambrian/Late Proterozoic lineated felsic mica gneiss, injected gneiss, and biotite gneiss [33]. Geospatial data for North Carolina geology [34] was integrated into ArcGIS Pro (ESRI) to evaluate site-specific geology. These data indicated that most of the community is underlain by Cambrian/Late Proterozoic aged biotite gneiss and mica schist. A small area in the northwest corner of the parcel is underlain by Cambrian/Late Proterozoic felsic gneiss and mica schist. This community can serve up to 220 single-wide trailers and has an area of 29.2 ha. Generally, each septic system in the area serves 2 homes. The septic system density in this community was estimated to be approximately 3.7 systems ha$^{-1}$ assuming that each site is occupied and there is 1 system per 2 homes (220 / 2 = 110 systems). During the 2021 – 2022 study, 3 sites served by septic systems were selected in the North Carolina Piedmont within Raleigh Belt geology (Fig. 1). An additional site (Site 400) was identified and included for the 2022 – 2023 study. The studied septic systems are conventional, gravel bed systems that serve 1 or 2 single-wide trailers and located approximately 15 m (50 ft) to 45 m (~150 ft) upgradient from a stream.

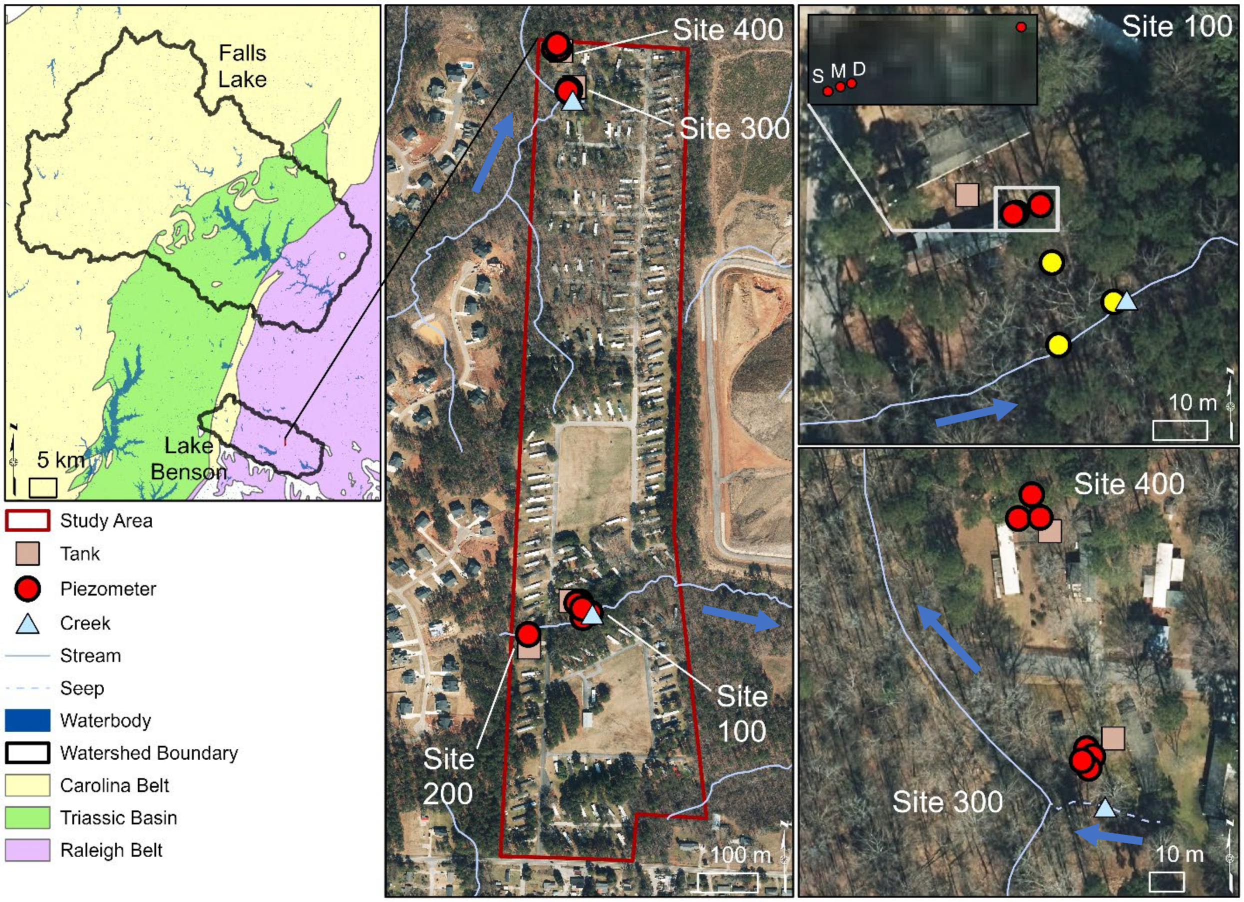


**Figure 1.** Map of the 4 sites monitored from 2021 – 2023 for the 2021 – 2022 and 2022 – 2023 NC Policy Collaboratory studies. The inset at Site 100 depicts a nest of piezometers installed near the drainfield where S= shallow, M= medium, and D= deep. Red circles denote drainfield piezometers and yellow circles denote downgradient piezometers (Site 100 only). Blue arrows indicate the direction of stream flow.

2.2 Site Instrumentation and Soils Data

After the sites were identified, each site was surveyed to identify septic system components (e.g., location and areal footprint of septic tanks and drainfields). Risers on septic tanks allowed for easy identification and access for sampling. The areal extent of the drainfield was located and flagged by using a tile probe rod. Piezometer locations were identified between, adjacent to, or downgradient of drainfield trenches. Boreholes were augered (Fig. 2A & 2B) and the depth to soil wetness indicators were noted (e.g., common low chroma colors, increased soil moisture content, water table). During piezometer installation, a soil profile was constructed to assess soil characteristics and soil wetness indicators (Fig. 2C). Once the seasonal high-water table was established, an additional 0.3 – 0.9 m (1 – 3 ft) was augered to reduce likelihood that the piezometer would be dry. After reaching the desired depth, a piezometer was constructed by cementing a section of schedule-40, solid PVC pipe to a section of schedule-40, screened PVC approximately 0.3 – 0.9 m (1 – 3 ft) in length, which was then inserted into the borehole. Each piezometer had a diameter of 3.18- or 5.08-cm (1.25- or 2-in) and total depths ranged from 0.9 – 2.7 m (2.9 – 8.7 ft). The annular space between the pipe and borehole was filled with sand until the space around the screened interval was filled. Bentonite was poured above the sand layer to prevent contamination of the surficial aquifer by surface water pollutants migrating vertically down the side of the piezometer. Valve boxes were secured around the top of the piezometer and installed just below the ground surface (Fig. 2D). Identification numbers were written on each piezometer cap and valve box lid. A total of 15 piezometers were installed across the 4 sites. Site 100 had 4 piezometers installed near the drainfield including a nest of 3 piezometers installed at different depths. There were also 3 downgradient piezometers installed at Site 100 with 1 approximately 15 m downgradient from the system and another 2 approximately 30 m downgradient from the system adjacent to a stream. Site 200 contained 1 piezometer installed near the drainfield. Site 300 contained 4 piezometers installed immediately adjacent to the drainfield. Site 400 contained 3 piezometers installed near the drainfield. A background piezometer was also installed at the Booth Training Center (< 3.2 km [2 mi] away) to assess water quality in groundwater not affected by wastewater (Appendix B).

Soil data was compiled using the USDA [35] Web Soil Survey for each site and summarized in Table 1. Sites 100 and 200 were in the Pacolet soil series. The typical soil profile from this series is well drained with deeper depths to water (> 203 cm). Site 300 contained a mixture of the Chewacla and Wehadkee soil series. The typical soil profile from this series is characterized by poorly to somewhat poorly drained and shallower water tables (within 30 – 61 cm). Site 400 consisted of Altavista soil. The typical profile from this series is moderately well drained with water table depths at 45 – 76 cm. The drainage class and depth to water data are summarized by the USDA [35] based on the typical soil profile from that series. Thus, actual drainage classes and depth to water may vary at individual sites. During installation, soil textures ranged from sandy loams to sandy clays, which are Group II to Group IV soils. Thus, the vertical separation distance between the trench bottom and seasonal high-water table should be at least 30 cm (15A NCAC 18A .1955(m)). These textures were similar to the typical profile summarized by the USDA [35].

**Table 1.** Soil series data for each site compiled from the USDA [35] Web Soil Survey and hydrological features. Sites 100 and 200 were located within the same soil series. Typical depth to water is inferred and reported by the USDA based on the typical soil profile. Measured depth to water may vary. Ch= Chewacla; W= Wehadkee.

| Location | Soil Series | Description | Soil Texture | Typical Depth to Water (cm [in]) | Hydrologic Soil Group | Drainage Class |
|---|---|---|---|---|---|---|
| Site 100<br>Site 200 | Pacolet | Urban land complex; 10 - 15% slopes; saprolite derived from granite and gneiss and/or schist | Sandy loam to clay (Group II - IV) | > 203 (> 80) | B | Well drained |
| Site 300 | Chewacla and Wehadkee | 0 - 2% slopes, frequently flooded; loamy alluvium derived from igneous and metamorphic rock | Ch: Loam to clay loam (Group II - III)<br>W: Silt loam to clay loam (Group III) | Ch: 15 - 61 (6 - 24)<br>W: 0 - 30 (0 - 12) | B/D (Ch & W) | Ch: Somewhat poorly drained<br>W: Poorly drained |
| Site 400 | Altavista | 0 - 4% slopes; sandy loam; rarely flooded; derived from igneous and metamorphic rocks | Coarse sandy loam to clay loam (Group II - III) | 45 - 76 (18 - 30) | C | Moderately well drained |


**Figure 2.** A) Will Shingleton (BS Environmental Health Student) augering a borehole. B) Access risers to the septic tank shown in the foreground while a piezometer is installed in the background. C) Soil profile laid out for soil analysis. D) Grouting the piezometer within the borehole.

2.3 Sampling Protocol and Laboratory Analysis

During the 2021 – 2022 study, there were a total of 4 sampling events that occurred at Sites 100 and 300 in February, April, May, and June 2022. Site 200 was abandoned during the 2021 – 2022 NC Policy Collaboratory study due to drought conditions causing the piezometer and stream to be dry. The current project funded an additional 6 sampling events occurring at Sites 100, 300, and 400 in October, November, and December 2022 and January, February, and April 2023. Wastewater, groundwater, and surface water quality were analyzed at the other sites using in-field environmental measurements and grab sampling for nutrients and bacteria. Before sampling groundwater, a *Solinst* Temperature, Level, Conductivity meter was used to measure the depth to water at each piezometer. After recording the water depth, piezometers were purged using a new, disposable PVC bailer until approximately 2 bailer full volumes of water were removed, which allowed groundwater to recharge the piezometer. After bailing, groundwater was extracted via the bailer and poured into a clean calibration cup, which was then affixed to a HI-9829 (*Hanna Instruments*) multiprobe field meter to measure environmental parameters. These parameters included temperature, dissolved oxygen (DO), oxidation-reduction potential (ORP), pH, and specific conductance (SC). Additional groundwater was extracted and poured into a clean high-density polypropylene bottle and a sterile, 120-mL sampling vessel for nutrient and bacteria analysis, respectively. These steps were repeated for each piezometer using a new bailer, and the calibration cup and sensors were rinsed and primed with purged water before measuring environmental parameters. Wastewater samples were collected directly from septic tanks using a disposable, PVC bailer. Septic tanks were sampled last to prevent contamination of nutrients and bacteria.

Sites 100 and 300 are adjacent to 2 small streams that were also studied (Fig. 1). The HI-9829 calibration cup was rinsed, primed, and filled directly from the stream to analyze for the forementioned parameters in addition to turbidity. Stream discharge was also measured for surface waters by temporarily channelizing flow via natural or synthetic weir and recording the elapsed time to fill a volumetric bottle (measured in mL) and converted to L $sec^{-1}$. During some months, flow was too low to accommodate this method and discharge was calculated by multiplying the cross-sectional area of the active stream channel by the stream velocity. Stream velocity was estimated using the floating object method. Surface water samples were also collected by filling nutrient and bacteria bottles directly from the stream. All water samples were stored in an iced cooler and transported to research laboratories at East Carolina University for analysis.

Nutrient samples were analyzed at the Environmental Research Laboratory at East Carolina University. A vacuum filtration system was used to filter samples using 1.5- and 0.7-micron microfiber glass filters that were pretreated to remove organic material by combustion at 400°C in a muffle furnace. Stream samples were split into 2 equal sub-samples (approximately 125 – 300 mL per sub-sample) and filtered using 2 separate 1.5-micron filters. Sub-samples were required to analyze particulate nutrient concentrations and total suspended solids (TSS), both of which require a 1.5-micron filter for analysis. The first sub-sample was filtered, and the filter was stored in an aluminum foil packet preserved in the freezer until analysis occurred. The second sub-sample was used for TSS analysis, which involved filtering the sub-sample through a pre-weighed filter. The filter was then stored in an oven set to 104°C for at least 24 hours to allow water to evaporate. The filter was reweighed, and this process was repeated until the mass of the filter was equal across 3 separate measurements. The concentration was calculated by taking the difference in final and initial weight divided by the volume filtered and multiplied by 1,000,000 to convert from g $mL^{-1}$ to mg $L^{-1}$.

After filtering, the filtrate was recovered and analyzed for nutrients, chloride (Cl), and dissolved organic carbon (DOC). A SmartChem 170/200 (*KPM Analytics*) discrete analyzer was used to measure the concentrations of ammonium, nitrate, orthophosphate (henceforth referred to as phosphate), TDP (phosphate + dissolved organic phosphorus [DOP]), and Cl. The SmartChem was also used to measure particulate nitrogen (PN) and particulate phosphorus (PP) concentrations after filters were digested using the Kjeldahl method. A *Shimadzu* TOC/TN autoanalyzer using combustion catalytic oxidation was used to measure DOC and TDN. The total nitrogen (TN) and total phosphorus (TP) concentrations were estimated by summing the concentration of TDN and PN and TDP and PP, respectively.

*E. coli* concentrations were enumerated according to the *IDEXX* method and conducted at the Water Research Laboratory at East Carolina University. *E. coli* sampling began during the April 2022 sampling event (second event). A vial of nutrient broth (Colilert) was added to each sampling vessel containing 100 $mL^{-1}$ of sample, and vessels were vigorously agitated until the media dissolved. Samples were poured into a QuantiTray/2000, heat sealed, and labeled with sample identifier. Sample trays were then transferred into an incubator set to 35°C for 24 – 28 hours. After incubation, the trays were removed and exposed to a black light to identify the number of fluorescent wells, which indicate presence of *E. coli*. The number of large and small wells were summed, and the most probable number (MPN) table was used to match the well count with its associated concentration of *E. coli* (MPN 100 $mL^{-1}$). This process was repeated for each wastewater, groundwater, and surface water sample. The detection limit in undiluted samples ranged from 1 – 2,419.6 MPN 100 $mL^{-1}$, respectively. Wastewater samples were diluted using a factor between 2,000 – 10,000. Sites 100 and 300 routinely displayed characteristics of malfunction (e.g., ponded water, sulfur or "rotten egg" smell), thus groundwater and streams were also diluted during these events. Final concentrations were calculated by multiplying the dilution factor by the concentration acquired from the MPN table.

### 2.4 Data Analysis

Nutrient, bacteria, and environmental parameters were compiled into comparison groups, which included tank, drainfield, downgradient, and streams. During the 2021 – 2022 study, there was a total of 2 tanks (n= 8), 6 drainfield (n= 22) and 2 downgradient (n= 8) piezometers, and 2 streams (n= 8) that were sampled 4 times. One of the piezometers at Site 100 was dry during the May and June 2022 sampling events, thus the total number of drainfield samples was 22 instead of 24. During the 2022 – 2023 study, there was an addition of 1 tank, 5 drainfield, 1 downgradient, and 1 background piezometer. Sites 100, 300, and 400 were sampled 6 times during the current study equating to 18 tank, 65 drainfield, 13 downgradient, 12 stream, and 6 background samples. At Site 100, drainfield and downgradient piezometers were occasionally dry during sampling. One drainfield piezometer was dry in October 2022, the piezometer located 15 m downgradient of the system was dry on 4 sampling events, and one of the piezometers adjacent to the stream was vandalized between the February and April 2023 sampling events.

Pollutant data from each sampling location were pooled across sites to compare broad trends based on sampling location. Trends in pollutant data were also compared between sites to assess differences in system performance at the lot scale. The percent difference equation (Eq. 1) was used to calculate treatment efficiencies for pooled and individual sites between the tank and drainfield groundwater, the tank and downgradient groundwater, and the tank and the stream.

$$TE = \frac{Conc_T - Conc_X}{Conc_T} * 100 \qquad \text{Eq. 1}$$

where TE= treatment efficiency of pollutant; Conc= pollutant concentration; T= tank; and X= drainfield, downgradient, or stream sampling location.

A two-component mixing model was used to estimate pollutant removal percentages for nutrient and bacteria in groundwater piezometers between the tank and drainfield and tank and downgradient locations [26, 36]. The model was constructed based on the ratio of the difference in Cl concentrations between tank and groundwater and between tank and background (Eq. 2).

$$Fraction_{BG-GW} = \frac{Cl_T - Cl_X}{Cl_T - Cl_{BG}} \qquad \text{Eq. 2}$$

$$Fraction_{WW} = 1 - Fraction_{BG-GW}$$

where BG-GW= background groundwater; Cl= chloride concentration; T= tank; X= drainfield or downgradient piezometer; WW= wastewater

The fractional coefficients are used to predict the pollutant concentration in drainfield or downgradient groundwater by multiplying the pollutant's concentration within the tank by the fraction of wastewater percentage. These predicted pollutant concentrations assume that dilution was the only reduction mechanism. Thus, if observed concentrations are less than model predictions, it is assumed to represent a removal mechanism that removes pollutant mass or number of organisms. The percent difference equation (Eq. 1) was used to estimate mass reduction. In addition to groundwater mass reduction estimates, pollutant mass transport was assessed at Sites 100 and 300. Transport was estimated by multiplying stream discharge (L $sec^{-1}$) by pollutant concentration (mg $L^{-1}$) and units were converted to g $day^{-1}$ for nutrients and MPN $min^{-1}$ for *E. coli*.

Statistical analysis and figure development was conducted using the R Statistical Framework and R Studio [37, 38]. The 'readxl', 'ggplot2', and 'cowplot' packages were also used to develop figures [39-41]. The Kruskal-Wallis H test was used to determine if statistical differences existed in pollutant concentrations, masses, or loads for a categorical variable that contained > 2 groups. If results from this test yielded a significant difference ($p \leq 0.05$), then a pairwise Wilcoxon Rank Sum test was conducted to determine which groups differed. The Wilcoxon Rank Sum test was used to determine if pollutant concentration, masses, or loads differed significantly ($p \leq 0.05$) for a categorical variable with 2 groups.

## 2.5 Report Organization

Research findings are summarized in the next section of this report. The figures and tables included in the next section of the report include data collected from both 2022 – 2023 and 2021 – 2022 NC Policy Collaboratory studies. Data tables summarizing the 6 sampling events from only the 2022 – 2023 NC Policy Collaboratory study were included in Appendix C.

# 3 Research Findings

## 3.1 Nitrogen Concentrations

### *3.1.1 Pooled Sites*

Septic tanks contained the highest concentrations of TDN relative to other sampling locations (Fig. 3; Table 2). The median concentration of TDN in septic tanks pooled from all sites was 62.15 mg $L^{-1}$. This concentration was approximately 6 and 16 times greater than the median concentration of TDN within drainfield groundwater and adjacent streams, respectively. Furthermore, septic tanks contained a median TDN concentration that was approximately 65 times greater than background groundwater. These differences were statistically significant when comparing tanks to all other sampling locations ($p < 0.01$). Groundwater near drainfields contained a median concentration of TDN of 9.54 mg $L^{-1}$. Thus, the concentration reduction in median TDN between wastewater and drainfield groundwater was approximately 85% across all sites. The median concentration of TDN further reduced between groundwater near drainfields and adjacent streams. When pooling streams from Sites 100 and 300, the median TDN concentration was 3.81 mg $L^{-1}$, which represented a 60% reduction in median concentrations between drainfield groundwater and adjacent streams. This difference was significantly significant at $p \leq 0.02$. There was a 94% reduction in median concentrations of TDN between wastewater and nearby streams. Differences in concentration reduction likely occurred due to biogeochemical processes in soils beneath and adjacent to the drainfield. Similar processes and dilution may also occur in the surficial aquifer that contributed to differences in concentration reduction estimates. Despite a high treatment efficiency, groundwater near drainfields and nearby streams contained elevated TDN concentrations relative to background groundwater. Drainfield groundwater and streams contained median TDN concentrations that were approximately 10 and 3 times, respectively, greater than background concentrations (Table 2). These differences were statistically significant at $p < 0.01$.

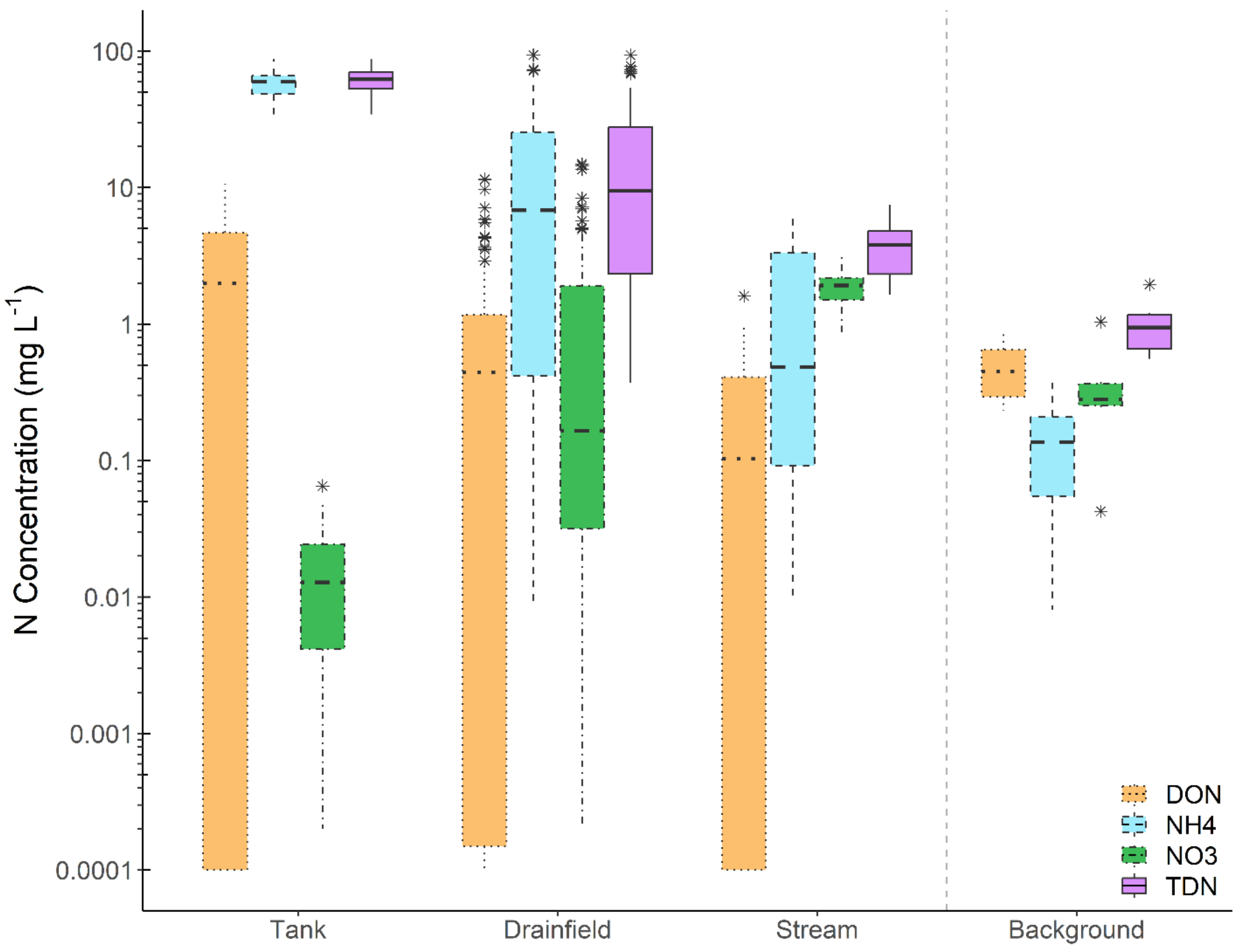


**Figure 3.** Nitrogen (N) concentrations of sampling locations pooled from Sites 100, 300, and 400. There is not a stream at Site 400, so data were only pooled from Sites 100 and 300. TDN= total dissolved nitrogen; NO3= nitrate-nitrogen; NH4= ammonium-nitrogen; DON= dissolved organic nitrogen.

**Table 2.** Median (range) of concentrations of nitrogen (N) species summarized based on comparison group at each site and pooled.

| Site/Sampling Location | Concentration of N Species (mg $L^{-1}$) | | | |
|---|---|---|---|---|
| | TDN | $NO_3^-$-N | $NH_4^+$-N | DON |
| Site 100 | | | | |
| Tank | 67.08 (36.59 - 87.20) | 0.01 (<0.01 - 0.02) | 63.63 (36.57 - 87.20) | 1.92 (<0.01 - 6.10) |
| Drainfield | 11.02 (0.84 - 53.74) | 0.29 (<0.01 - 14.50) | 6.81 (0.10 - 53.69) | 0.44 (<0.01 - 4.28) |
| Downgradient | 1.91 (0.60 - 19.38) | 0.09 (<0.01 - 10.98) | 0.88 (0.05 - 8.41) | 0.72 (<0.01 - 2.16) |
| Stream | 2.29 (1.65 - 3.66) | 2.01 (0.80 - 2.67) | 0.09 (<0.01 - 0.50) | 0.19 (<0.01 - 0.98) |
| Site 300 | | | | |
| Tank | 53.90 (34.30 - 67.61) | 0.01 (<0.01 - 0.05) | 49.49 (34.26 - 66.22) | 2.92 (<0.01 - 10.61) |
| Drainfield | 20.04 (1.35 - 93.60) | 0.23 (<0.01 - 14.92) | 19.45 (0.19 - 93.58) | 0.91 (<0.01 - 11.48) |
| Stream | 4.86 (3.95 - 7.49) | 1.60 (1.19 - 3.15) | 3.37 (0.39 - 5.93) | <0.01 (<0.01 - 1.61) |
| Site 400 | | | | |
| Tank | 67.35 (55.83 - 79.06) | 0.03 (<0.01 - 0.07) | 64.03 (55.77 - 79.04) | 2.26 (<0.01 - 3.85) |
| Drainfield | 1.26 (0.37 - 3.13) | 0.06 (<0.01 - 2.24) | 0.16 (<0.01 - 2.37) | 0.42 (<0.01 - 1.02) |
| Pooled | | | | |
| Tank | 62.15 (34.30 - 87.20) | 0.01 (<0.01 - 0.07) | 59.67 (34.26 - 87.20) | 1.99 (<0.01 - 10.61) |
| Drainfield | 9.54 (0.37 - 93.60) | 0.17 (<0.01 - 14.92) | 6.81 (<0.01 - 93.58) | 0.44 (<0.01 - 11.48) |
| Stream | 3.81 (1.65 - 7.49) | 1.92 (0.80 - 3.15) | 0.49 (<0.01 - 5.93) | 0.10 (<0.01 - 1.61) |
| Background | 0.95 (0.56 - 1.95) | 0.28 (0.04 - 1.04) | 0.14 (<0.01 - 0.39) | 0.45 (0.23 - 0.91) |

Ammonium and nitrate were the dominant species of TDN when pooling sites (Table 2). Wastewater in septic tanks consisted predominantly of ammonium, followed by DON and nitrate. The mean percentage of ammonium, nitrate, and DON was 95.26%, 0.03%, and 4.71%, respectively. The median concentration of ammonium reduced by 89% between septic tanks and groundwater near drainfields (Table 2), presumably due to nitrification, adsorption, biological uptake, absorption, dilution, and/or dispersion. Nitrate concentrations support that nitrification occurred (Table 2). Groundwater near drainfields contained a greater median nitrate concentration (0.17 mg $L^{-1}$) relative to wastewater in septic tanks (0.01 mg $L^{-1}$). However, ammonium remained the dominant species of TDN in groundwater near drainfields when pooling sites (Table 2). The mean percentages were 59.36%, 21.91%, and 18.73% for ammonium, nitrate, and DON, respectively. The elevated concentration of ammonium in groundwater near drainfields suggests that septic systems may not be adequately treating wastewater. Since individual sites exhibited variability in system performance, this was discussed in greater detail in a later section. Nitrate was the dominant species of TDN when pooling streams from Sites 100 and 300 (Table 2). The mean percentage was 57.79%, 33.00%, and 9.21% for nitrate, ammonium, and DON, respectively. The shift in speciation suggests that the ammonium-rich groundwater may be nitrified after discharging to the stream where it becomes exposed to oxidizing conditions. There are also other septic systems in the area that drain to the streams. If these systems are adequately nitrifying septic system effluent, then they may be a source of nitrate to the streams.

Nitrogen speciation data also support that septic systems increase concentrations of labile nitrogen in groundwater and surface water. Groundwater and streams that received septic system effluent consisted mostly of either ammonium or nitrate. However, TDN concentrations in background groundwater consisted mostly of DON concentrations (Table 2). The mean percentage of DON, nitrate, and ammonium was 49.68%, 36.97%, and 13.34%, respectively. Thus, the increased TDN concentration and shift in dominant nitrogen speciation suggests that septic systems can alter the fate and transport of plant available forms of nitrogen. Thus, septic system performance is a key factor that can alter delivery of nitrogen to water resources.

*3.1.2 Lot-scale Trends in Nitrogen Concentrations*

Wastewater in septic tanks contained the highest concentrations of nitrogen at all sites (Fig. 4; Table 2). Median TDN concentrations in tanks ranged from approximately 54 – 67 mg $L^{-1}$ (Table 2). Overall, TDN concentrations in septic tanks ranged from 34.30 – 87.20 mg $L^{-1}$. Nitrogen treatment varied substantially at the lot scale. Groundwater near drainfields contained median concentrations of TDN that were approximately 84%, 63%, and 98% lower than wastewater at Sites 100, 300, and 400, respectively. TDN concentrations in drainfield groundwater significantly differed from wastewater at all sites ($p < 0.01$). Median TDN concentration in groundwater near drainfields remained elevated at Sites 100 and 300 (Fig. 4; Table 2). Sites 100 and 300 contained a median TDN concentration of 11.02 and 20.04 mg $L^{-1}$, respectively, in groundwater near drainfields. Groundwater near drainfields at Sites 100 and 300 contained median concentrations that were substantially greater than the median TDN in drainfield groundwater at Site 400 (1.26 mg $L^{-1}$) and background groundwater (0.95 mg $L^{-1}$). Furthermore, the range of TDN concentrations in groundwater near drainfields at Sites 100 and 300 were considerably greater than Site 400 and background groundwater. The maximum TDN concentrations observed at Sites 100 and 300 were similar to or exceeded TDN concentrations observed in wastewater in septic tanks (Table 2). Additionally, TDN concentrations in groundwater near drainfields at Sites 100 and 300 occasionally exceeded the minimum TDN concentration observed in wastewater from septic tanks. This occurred 25% (7 out of 28 samples) and 22.5% (9 out of 40 samples) of the time at Sites 100 and 300, respectively. During these occasions, the median (range) and mean (± standard deviation) of TDN was 43.58 mg $L^{-1}$ (38.58 – 53.74 mg $L^{-1}$) and 46.33 mg $L^{-1}$ (± 5.91 mg $L^{-1}$), respectively, in groundwater near the drainfield at Site 100. During these occasions at Site 300, TDN concentrations in drainfield groundwater were greater with a median (range) of 68.48 mg $L^{-1}$ (39.89 – 93.60 mg $L^{-1}$) and a mean (± standard deviation) of 61.57 mg $L^{-1}$ (± 19.38 mg $L^{-1}$). This phenomenon was not observed at Site 400 where the maximum TDN concentration was 3.13 mg $L^{-1}$, which was substantially lower than TDN concentrations in wastewater. Site 400 was also the only site where drainfield groundwater did not significantly differ from background groundwater ($p = 0.28$).

Concentrations of TDN in downgradient groundwater and streams were substantially lower than wastewater (Fig. 4; Table 2). At Site 100, median TDN concentrations in downgradient groundwater and the stream were both approximately 97% lower than wastewater concentrations. Both sampling locations were significantly different from wastewater and drainfield groundwater at $p \leq 0.01$. The median TDN concentration in downgradient groundwater was 1.91 mg $L^{-1}$, which was lower than the median TDN concentration in the stream (2.29 mg $L^{-1}$) and this difference was significant ($p = 0.02$). Thus, additional nitrogen inputs from other septic systems or other sources likely contributed to TDN concentration measured in the stream. At Site 300, the stream contained a median TDN concentration of 4.86 mg $L^{-1}$, which was about 91% lower than the median TDN in wastewater. Stream TDN concentrations were significantly different from TDN concentrations in septic tanks ($p < 0.01$) and groundwater near drainfields ($p < 0.01$) at Site 300. The median TDN concentration in the stream at Site 300 was more than double the median TDN concentration in the stream at Site 100 (Table 2). This difference was statistically significant ($p < 0.01$). The difference in nitrogen concentrations may have been due to differences in the distance from the system to the stream. The system at Site 100 is approximately 30 m away from the stream, whereas the system at Site 300 is about 15 m upgradient from the stream. Additionally, Site 100 has denser tree vegetation between the drainfield and the stream (Fig. 1). The increased distance and vegetation at Site 100 likely increased treatment potential. Both streams contained TDN concentrations that were approximately 2 – 5 times greater than background groundwater (Table 2). These data suggest that septic systems within this region can be potentially significant sources of nitrogen to water resources.

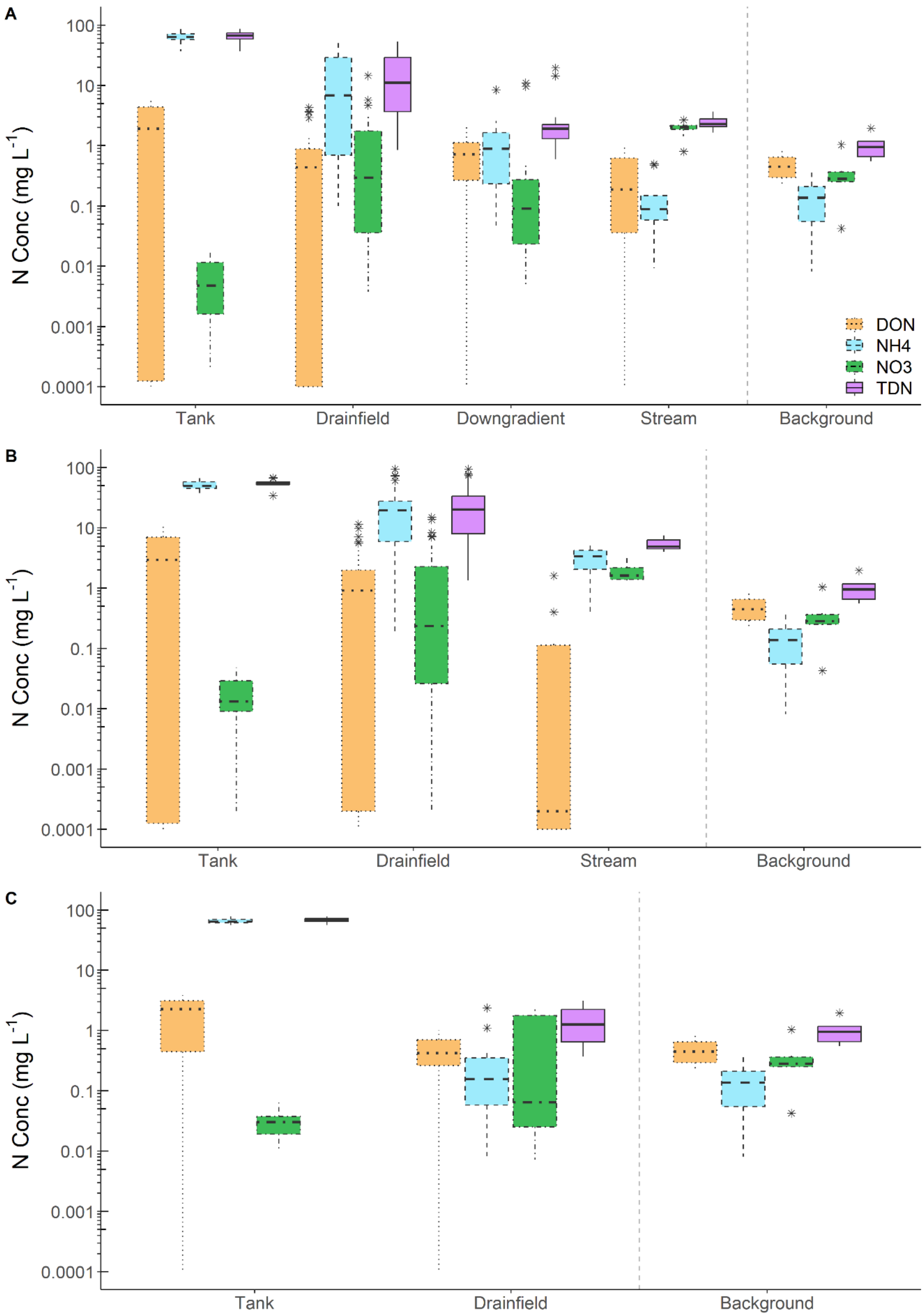


**Figure 4.** Nitrogen (N) concentrations for sampling locations at Sites 100 (A), 300 (B), and 400 (C) compared to background groundwater.

## 3.2 Phosphorus Concentrations

### *3.2.1 Pooled Sites*

Wastewater within pooled septic tanks consistently contained the highest concentration of TDP (Fig. 5). The median TDP concentration in wastewater was 7.23 mg $L^{-1}$. This median value was approximately 36 and 145 times greater than the median concentrations of TDP in groundwater near drainfields and in adjacent streams, respectively. Furthermore, the median concentration of TDP in septic tanks was more than 700 times greater than background groundwater. Wastewater TDP concentrations were significantly different from all other sampling locations at $p < 0.01$. Differences in median TDP concentrations between wastewater and water resources indicated that septic systems were effective at treating TDP. Groundwater near drainfields contained a median TDP concentration of 0.20 mg $L^{-1}$ (Table 3), which was 97% lower than median TDP concentration in wastewater. The median TDP concentration in the streams was 0.05 mg $L^{-1}$ (Table 3). This concentration was approximately 75% and 99% lower than median TDP concentrations in drainfield groundwater and wastewater, respectively. Both drainfield groundwater and streams still contained elevated concentrations of TDP compared to background groundwater (Table 3). The median TDP in background groundwater was 0.01 mg $L^{-1}$. Thus, median concentrations of TDP in groundwater near drainfields and streams were 20 and 5, respectively, times greater than background groundwater. Furthermore, TDP concentrations in background groundwater exhibited low variability ranging from < 0.01 to 0.03 mg $L^{-1}$, whereas groundwater and streams recharged by septic system effluent had larger variability in TDP concentrations (Fig. 5; Table 3). Drainfield groundwater and streams also contained several outliers, which was due to variability in lot-scale septic system performance and will be discussed in the next subsection.

The data also exhibited a clear trend in phosphorus speciation when comparing water affected by wastewater to background groundwater (Fig. 5). Phosphate tended to be the dominant phosphorus species in wastewater, groundwater near drainfields, and in streams. More specifically, phosphate consisted of 89.71%, 73.90%, and 72.83% of TDP on average in wastewater, drainfield groundwater, and streams, respectively. However, DOP was the predominant phosphorus species in background groundwater contributing to 86.87% of TDP on average. The increased concentration of TDP and shift in dominant phosphorus speciation suggested that septic systems can be significant sources of phosphorus.

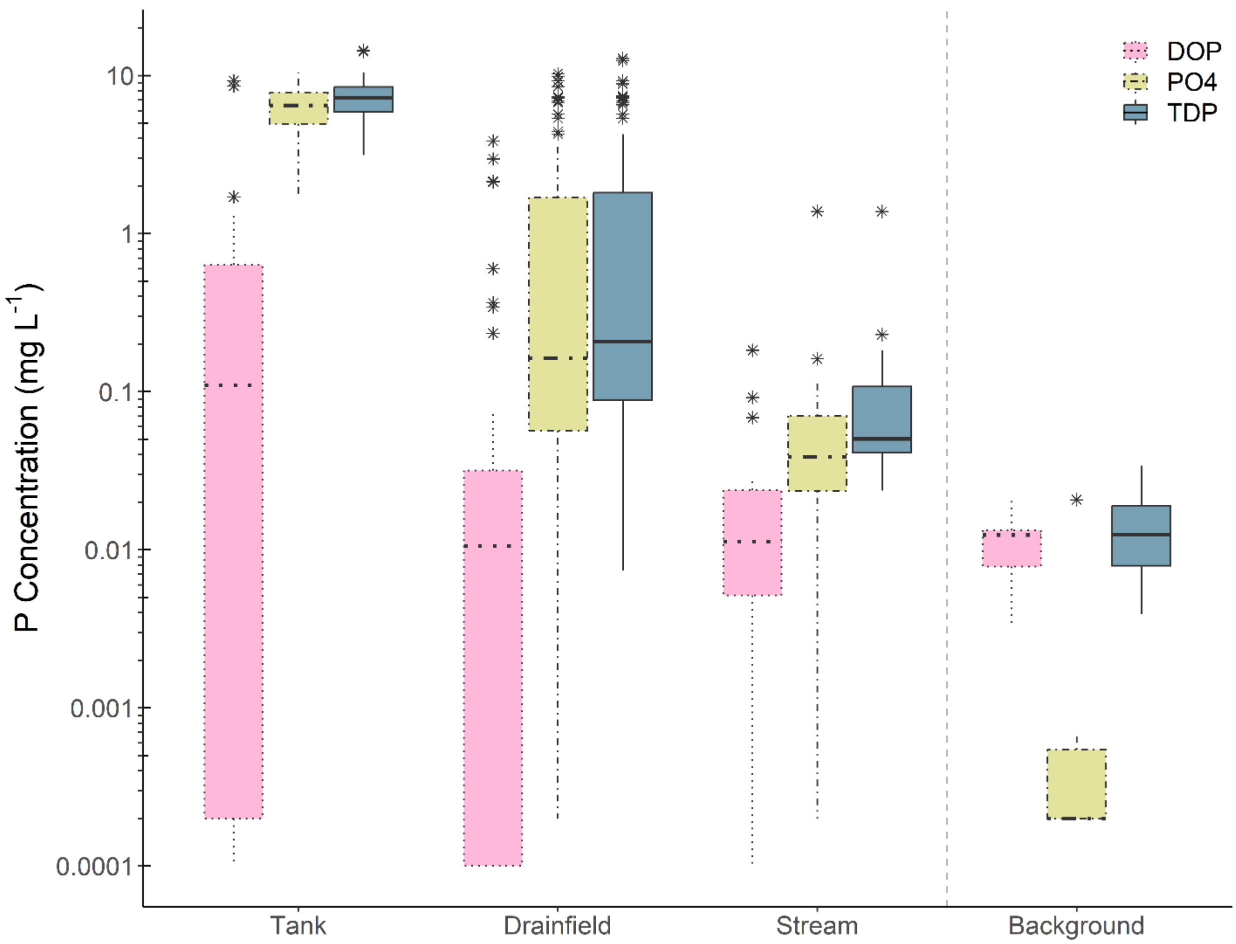


**Figure 5.** Phosphorus (P) concentrations of sampling locations pooled from Sites 100, 300, and 400. There is not a stream at Site 400, so data were only pooled from Sites 100 and 300. TDP= total dissolved phosphorus; PO4= phosphate-phosphorus; DOP= dissolved organic phosphorus.

**Table 3.** Median (range) of concentrations of phosphorus (P) species summarized based on comparison group at each site and pooled.

| Location | Concentration of P Species (mg $L^{-1}$) | | |
|---|---|---|---|
| | TDP | $PO_4^-$-P | DOP |
| Site 100 | | | |
| Tank | 7.91 (4.82 - 14.22) | 7.27 (4.82 - 10.42) | 0.04 (<0.01 - 9.30) |
| Drainfield | 0.39 (0.01 - 7.43) | 0.39 (<0.01 - 5.37) | <0.01 (<0.01 - 3.87) |
| Downgradient | 0.03 (<0.01 - 1.16) | 0.01 (<0.01 - 1.16) | 0.01 (<0.01 - 0.22) |
| Stream | 0.04 (0.02 - 1.38) | 0.04 (0.02 - 1.38) | <0.01 (<0.01 - 0.01) |
| Site 300 | | | |
| Tank | 5.65 (3.14 - 14.41) | 4.92 (1.75 - 5.94) | 0.32 (<0.01 - 8.61) |
| Drainfield | 0.29 (<0.01 - 12.82) | 0.22 (<0.01 - 10.27) | 0.02 (<0.01 - 2.97) |
| Stream | 0.10 (0.03 - 0.23) | 0.05 (<0.01 - 0.16) | 0.02 (<0.01 - 0.18) |
| Site 400 | | | |
| Tank | 8.26 (6.37 - 8.83) | 7.77 (6.30 - 8.50) | 0.21 (<0.01 - 0.71) |
| Drainfield | 0.10 (0.01 - 0.18) | 0.07 (<0.01 - 0.16) | <0.01 (<0.01 - 0.04) |
| Pooled | | | |
| Tank | 7.23 (3.14 - 14.41) | 6.47 (1.75 - 10.42) | 0.11 (<0.01 - 9.30) |
| Drainfield | 0.20 (<0.01 - 12.82) | 0.15 (<0.01 - 10.27) | 0.01 (<0.01 - 3.87) |
| Stream | 0.05 (0.02 - 1.38) | 0.04 (<0.01 - 1.38) | 0.01 (<0.01 - 0.18) |
| Background | 0.01 (<0.01 - 0.03) | 0.00 (<0.01 - 0.02) | 0.01 (<0.01 - 0.02) |

*3.2.2 Lot-scale Trends in Phosphorus Concentrations*

Septic system performance was highly efficient for all individual sites when comparing median TDP (Fig. 6; Table 3). The median TDP concentration of wastewater at Sites 100, 300, and 400 was 7.91 mg $L^{-1}$, 5.65 mg $L^{-1}$, and 8.26 mg $L^{-1}$, respectively. Concentrations of TDP were variable in all tanks ranging from approximately 3.14 – 14.41 mg $L^{-1}$. Groundwater near drainfields contained a median TDP concentration that was 95 – 99% lower than wastewater. Drainfield groundwater at Sites 100, 300, and 400 contained a median TDP concentration of 0.39 mg $L^{-1}$, 0.29 mg $L^{-1}$, and 0.10 mg $L^{-1}$, respectively (Table 3). This equated to a median concentration reduction of 95.09%, 95.33%, and 98.84% between wastewater and drainfield groundwater at Sites 100, 300, and 400, respectively. Differences between wastewater and drainfield groundwater were statistically significant at all sites ($p \leq 0.02$). While the treatment efficiencies are similar between sites, Site 400 contained TDP concentrations that were substantially lower than Sites 100 and 300. Drainfield groundwater at Site 400 was approximately 4 and 3 times lower than Sites 100 and 300, respectively. Similar to lot-scale trends in TDN, there were occasions when TDP concentrations at Sites 100 and 300 were consistent with wastewater strength (Fig. 6; Table 3). Concentrations of TDP in drainfield groundwater exceeded the minimum concentration in septic tanks 7% (2 out of 28) and 32.50% (13 out of 40) of the time at Sites 100 and 300, respectively. This phenomenon was never observed at Site 400. During these occasions at Site 100, the median (range) and mean (± standard deviation) concentration of TDP was 6.40 mg $L^{-1}$ (5.37 – 7.43 mg $L^{-1}$) and 6.40 mg $L^{-1}$ (± 1.46 mg $L^{-1}$), respectively, in groundwater near drainfields. During these occasions at Site 300, TDP concentrations in drainfield groundwater had a median (range) of 6.92 mg $L^{-1}$ (3.21 – 12.82 mg $L^{-1}$) and a mean (± standard deviation) of 7.55 mg $L^{-1}$ (± 2.77 mg $L^{-1}$).

Downgradient groundwater and streams typically contained the lowest TDP concentrations at Sites 100 and 300 (Table 3). At Site 100, the median TDP concentration in downgradient groundwater was 0.03 mg $L^{-1}$, which was 92% and >99% lower than drainfield groundwater and wastewater, respectively. These differences were statistically significant at $p < 0.01$. Both the median TDP concentration and TDP concentration range in downgradient groundwater was slightly lower than the stream (Fig. 6; Table 3). However, these differences were not statistically significant ($p = 0.12$). The stream at Site 100 contained a median TDP concentration of 0.04 mg $L^{-1}$, which was >99% lower than the median TDP concentration in wastewater. This difference was also statistically significant ($p < 0.01$). At Site 300, the stream contained a median TDP concentration of 0.10 mg $L^{-1}$ (Table 3). This was 98% lower than the median TDP concentration in wastewater and was a statistically significant difference ($p < 0.01$).

Despite the high treatment efficiency for TDP, groundwater and streams tended to contain elevated concentrations relative to background groundwater (Fig. 6; Table 3). The median TDP concentration in drainfield groundwater was between 10 and 39 times greater than background groundwater. Additionally, TDP concentrations were significantly different at $p < 0.01$ when comparing drainfield groundwater to background groundwater for each site. Streams at Sites 100 and 300 contained a median TDP concentration that was 4 and 10 times greater than background, respectively. This difference was also statistically significant for Site 100 ($p < 0.01$) and Site 300 ($p = 0.01$). These findings suggest that septic systems can be significant sources of phosphorus concentrations to groundwater and surface waters.

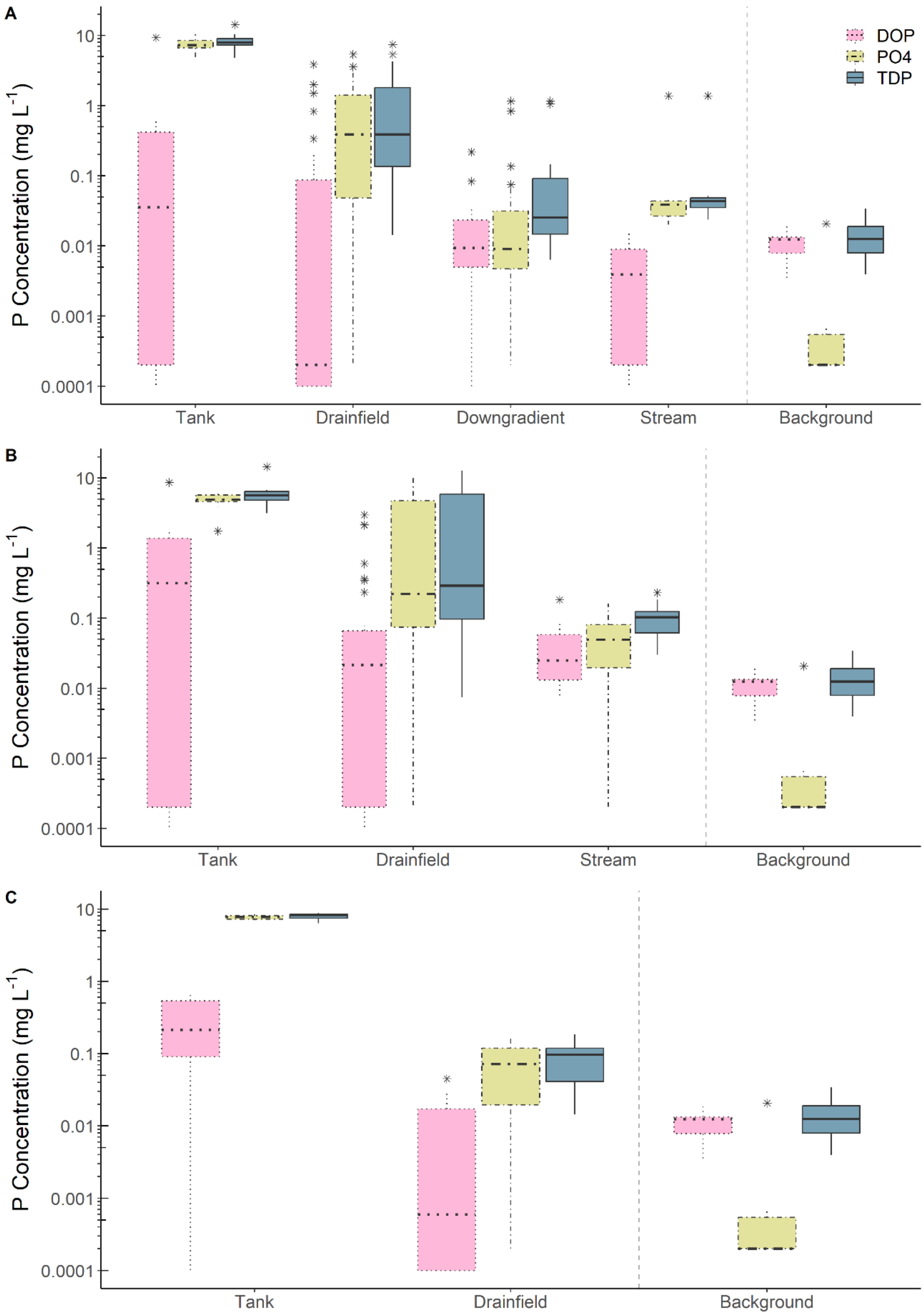


**Figure 6.** Phosphorus (P) concentrations for sampling locations at Sites 100 (A), 300 (B), and 400 (C) compared to background groundwater.

## 3.3 *Escherichia coli* Concentrations

### *3.3.1 Pooled Sites*

*E. coli* concentrations were elevated in septic tanks (Fig. 7; Table 4). Wastewater had a median and geometric mean value of 93,600 and 119,176 MPN 100 $mL^{-1}$, respectively. Both these values were considerably larger than median and geometric mean values in groundwater near drainfields, streams, and background groundwater (Fig. 7). Wastewater contained *E. coli* concentrations that were significantly different from all other sampling locations at $p < 0.01$. Drainfield groundwater and streams contained similar *E. coli* concentrations (Fig. 7). The median value in groundwater near drainfields was 1426.4 MPN 100 $mL^{-1}$, which was slightly greater than the median *E. coli* concentration in streams (1190 MPN 100 $mL^{-1}$). Additionally, the geometric mean values for drainfield groundwater and streams were similar (Table 4). *E. coli* concentrations were not significantly different between these two comparison groups ($p = 0.69$). Groundwater near drainfields exhibited the greatest variability in *E. coli* concentrations spanning 5 orders of magnitude (Fig. 7). This was most likely due to lot-scale trends in system performance. The lowest *E. coli* concentrations were observed in background groundwater (Fig. 7; Table 4). Background groundwater contained a median and geometric mean value of 4.2 and 4.6 MPN 100 $mL^{-1}$, respectively.

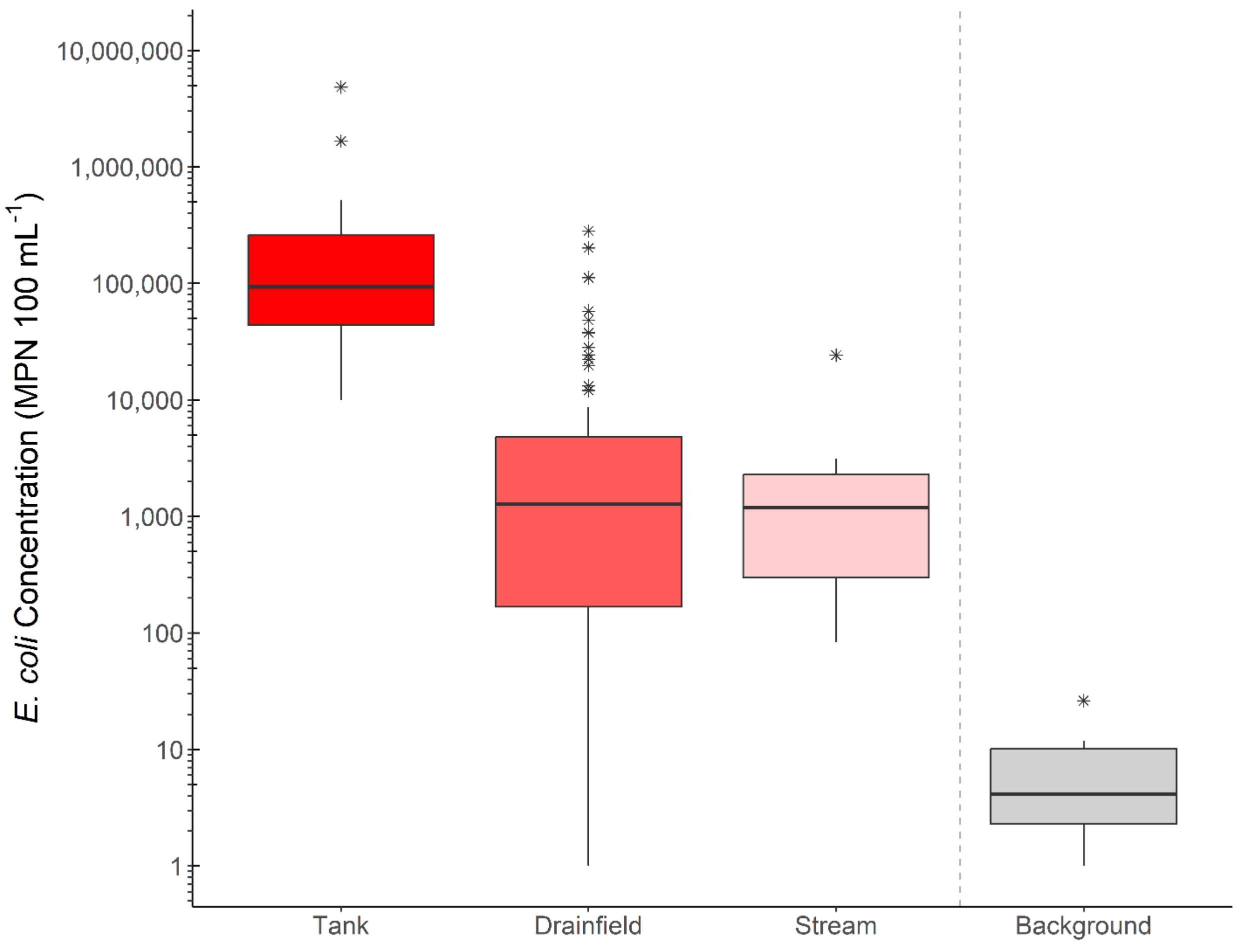


**Figure 7.** *Escherichia coli* (*E. coli*) concentrations of sampling locations pooled from Sites 100, 300, and 400. There is not a stream at Site 400, so data were only pooled from Sites 100 and 300.

**Table 4.** Median (range) and geometric mean values of *Escherichia coli* concentrations summarized based on comparison group at each site and pooled.

| Location | Median (Range) *(MPN 100mL$^{-1}$)* | Geometric Mean *(MPN 100mL$^{-1}$)* |
|---|---|---|
| Site 100 | | |
| Tank | 83400 (20500 - 1670000) | 119799 |
| Drainfield | 252.0 (1.0 - 60500) | 487.7 |
| Downgradient | 45.2 (1.0 - 1119.9) | 43.7 |
| Stream | 272.3 (83.3 - 2419.6) | 445.7 |
| Site 300 | | |
| Tank | 233200 (41200 - 4839200) | 210927 |
| Drainfield | 4419.6 (75.4 - 282720) | 4300 |
| Stream | 1986.3 (384.0 - 24196) | 1855.6 |
| Site 400 | | |
| Tank | 65200 (10000 - 181800) | 50220 |
| Drainfield | 248.1 (7.5 - 2419.6) | 202.7 |
| Pooled | | |
| Tank | 93600 (10000 - 4839200) | 119176 |
| Drainfield | 1426.4 (1.0 - 282720) | 1087.6 |
| Stream | 1190.0 (83.3 - 24196) | 909.4 |
| Background | 4.2 (1.0 - 26.2) | 4.6 |

*3.3.2 Lot-scale Trends in* E. coli *Concentrations*

*E. coli* concentrations were greatest at Site 300 relative to all other sampling locations (Fig. 8). The median concentration of *E. coli* in wastewater at Site 300 was 233,200 MPN 100 $mL^{-1}$, which was nearly 3 and 4 times greater than *E. coli* concentrations in septic tanks at Sites 100 and 400, respectively (Table 4). Similarly, the geometric mean value for wastewater at Site 300 was about 2 and 4 times greater than Sites 100 and 400, respectively. All sites had a median concentration reduction of *E. coli* that exceeded 98% between septic tanks and groundwater near drainfields. More specifically, median concentration of *E. coli* decreased by 98% at Site 300 and >99% at Sites 100 and 400. Each site contained *E. coli* concentrations in drainfield groundwater that were significantly different ($p < 0.01$) from wastewater. Treatment efficiency between Site 300 and Sites 100 and 400 only differed by 1.5%; however, both median and geometric mean values for *E. coli* in drainfield groundwater at Site 300 were considerably greater than those from Sites 100 and 400 (Table 4). Concentrations of *E. coli* in drainfield groundwater were occasionally similar to wastewater concentrations. This phenomenon occurred 14.29% (4 out of 28) and 12.50% (5 out of 40) at Sites 100 and 300, respectively. During these occasions at Site 100, groundwater near the drainfield had a median (range) and mean (± standard deviation) concentration of *E. coli* of 44,200 MPN 100 $mL^{-1}$ (24,196 – 60,500) and 43,274 MPN 100 $mL^{-1}$ (± 14,949), respectively. During these occasions at Site 300, *E. coli* concentrations had a median (range) and mean (± standard deviation) of 84,970 MPN 100 $mL^{-1}$ (24,400 – 282,270) and 121,242 MPN 100 $mL^{-1}$ (± 101,366), respectively, in drainfield groundwater. It is not clear if this phenomenon occurred at Site 400. There were 2 samples from drainfield groundwater when *E. coli* concentrations were beyond the maximum detection limit for an undiluted sample (2,419.6 MPN 100 $mL^{-1}$). The *E. coli* concentration could only be estimated at ">2,419.6 MPN 100 $mL^{-1}$". Thus, it may be possible that *E. coli* concentrations were similar to wastewater concentrations, but that could not be confirmed.

Concentrations of *E. coli* were several orders of magnitude lower in streams relative to wastewater concentrations (Table 4). The median concentration of *E. coli* in streams at Site 100 and 300 was 272.3 and 1986.3 MPN 100 $mL^{-1}$, respectively. This corresponded to a median concentration reduction of >99% for both streams. The stream at Site 300 had a median and geometric mean concentration of *E. coli* that was approximately 2 times lower than drainfield groundwater. At Site 100, the stream contained a median and geometric mean *E. coli* concentration that was similar to drainfield groundwater (Table 4). *E. coli* concentration in streams did not significantly differ from drainfield groundwater at Site 100 ($p = 1$) and Site 300 ($p = 0.16$). Downgradient groundwater at Site 100 contained the lowest *E. coli* concentrations at sites with septic systems (Table 4). The median and geometric mean concentration was 45.2 and 43.7 MPN 100 $mL^{-1}$, respectively. *E. coli* concentrations in downgradient groundwater were significantly different from all sampling locations at Site 100 and background groundwater ($p \leq 0.04$).

Groundwater and streams affected by wastewater discharges from septic systems contained elevated *E. coli* concentrations relative to background groundwater (Fig. 8; Table 4). Downgradient groundwater at Site 100 contained the lowest *E. coli* concentrations across all sites, but median *E. coli* concentration was still more than 10 times greater than background groundwater and this difference was significant ($p = 0.04$). At Site 300, drainfield groundwater and the stream contained median *E. coli* concentrations that were several orders of magnitude greater than background. Drainfield groundwater at Site 400 contained a median *E. coli* concentration that was nearly 60 times greater than background groundwater. These findings suggest that septic systems can be a significant source of *E. coli* in water resources.

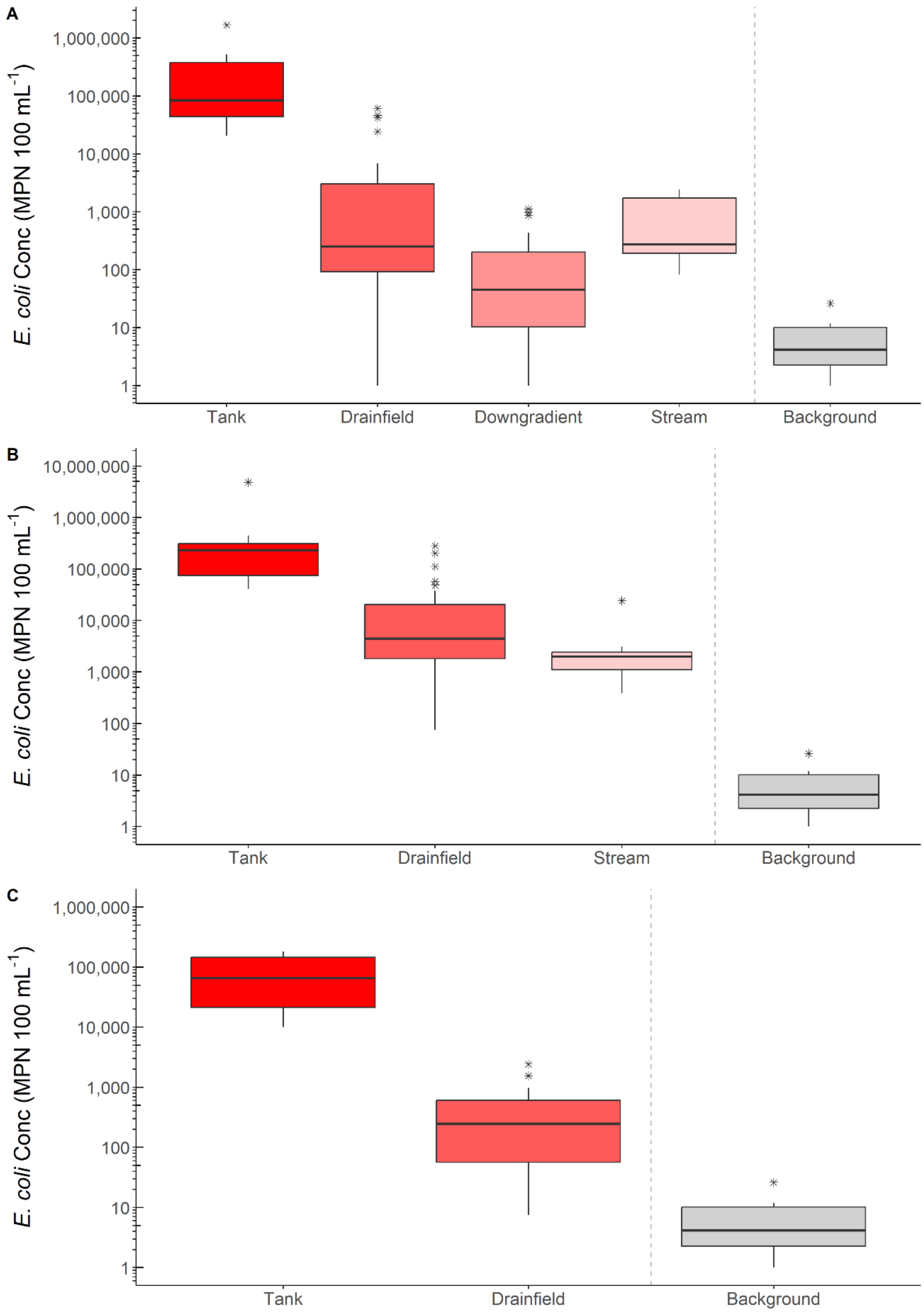


**Figure 8.** *E. coli* concentrations for sampling locations at Sites 100 (A), 300 (B), and 400 (C) compared to background groundwater. Note that the scale differs for plot B.

## 3.4 Pollutant Mass Reductions by Septic Systems

### *3.4.1 Nitrogen Mass Reductions*

Mass reduction of TDN ranged from approximately 25 – 97% between tank and groundwater (Table 5). The greatest mass reduction was estimated at Site 400, whereas Sites 100 and 300 reported much lower mass reduction. Groundwater near drainfields contained a median TDN mass reduction of 50%, 38%, and 97% at Sites 100, 300, and 400, respectively. The lowest estimates of TDN mass reduction were observed in a downgradient piezometer at Site 100 (~25%) and in drainfield groundwater at Site 300 (~38%). These low mass reduction estimates were likely driven by malfunctioning septic systems. Both sites occasionally exhibited indicators of a malfunctioning system (e.g., malodorous water ponded at surface or wastewater within tank freeboard). Despite these malfunctions, TDN mass reduction substantially increased at Site 100 between 15 m and 30 m downgradient of the system (Table 5). A vegetated buffer of trees and other emergent vegetation extended from the streambanks to approximately 5 – 10 m downgradient of the system. This vegetated zone was likely the main reason for the substantial increase in mass reduction between the downgradient piezometers at Site 100, especially when considering that labile forms of nitrogen were the dominant species (Table 2).

Mass reduction estimates suggested that mass removal processes accounted for most of the reduction in TDN. The median concentration reduction for drainfield groundwater at Sites 100, 300, and 400 were 84%, 63%, and 98%, respectively. Comparing these reductions to the mass reduction estimates (Table 5) suggests that dilution was not the primary reduction mechanism for most sampling locations. A similar phenomenon was observed for the piezometers located 30 m downgradient of the system at Site 100. There was a 98% concentration reduction and 88% mass reduction between median TDN in the tank and groundwater located 30 m downgradient. However, this trend was not observed in groundwater collected from the piezometer 15 m downgradient of the system. The median TDN concentration of groundwater at this location was about 75% lower than wastewater, but mass reductions were estimated to be approximately 25%. Thus, dilution accounted for most of the treatment at this piezometer. These results suggest that mass removal processes (e.g., denitrification, adsorption, plant and microbe uptake, anaerobic ammonium oxidation, volatilization) contributed to most of the treatment observed, except for sampling locations heavily influenced by malfunctions that may circumvent removal processes.

**Table 5.** Two-component mixing model using median chloride (Cl) and total dissolved nitrogen (TDN) concentrations to estimate mass reduction between tank and specified groundwater location.

| Location | Cl ($mg\ L^{-1}$) | Fraction WW | Fraction GW | Predicted TDN ($mg\ L^{-1}$) | Observed TDN ($mg\ L^{-1}$) | Cl/TDN Ratio | TDN Mass Reduction (%) |
|---|---|---|---|---|---|---|---|
| Site 100 | | | | | | | |
| Tank | 69.76 | 1.00 | 0.00 | | 67.08 | 1.04 | |
| DF | 26.54 | 0.33 | 0.67 | 22.28 | 11.02 | 2.41 | 50.53% |
| 15 m | 26.65 | 0.33 | 0.67 | 22.39 | 16.88 | 1.58 | 24.60% |
| 30 m | 18.04 | 0.20 | 0.80 | 13.47 | 1.63 | 11.04 | 87.87% |
| BG | 5.04 | 0.00 | 1.00 | | 0.95 | 5.33 | |
| Site 300 | | | | | | | |
| Tank | 62.07 | 1.00 | 0.00 | | 53.90 | 1.15 | |
| DF | 39.39 | 0.60 | 0.40 | 32.46 | 20.04 | 1.97 | 38.27% |
| BG | 5.04 | 0.00 | 1.00 | | 0.95 | 5.33 | |
| Site 400 | | | | | | | |
| Tank | 52.47 | 1.00 | 0.00 | | 67.35 | 0.78 | |
| DF | 31.37 | 0.56 | 0.44 | 37.40 | 1.26 | 24.95 | 96.64% |
| BG | 5.04 | 0.00 | 1.00 | | 0.95 | 5.33 | |

*3.4.2 Phosphorus Mass Reductions*

Mass reduction of TDP ranged from 58 – 99% between tank and groundwater (Table 6). The highest and lowest estimates of TDP mass reduction occurred at Site 100. When comparing drainfield groundwater, Site 400 had a median mass reduction of 98%, followed by Site 300 at 91% and Site 100 at 85%. The lowest mass reduction of TDP occurred in the piezometer 15 m downgradient of the septic system at Site 100. The spatial trends in mass reductions of TDP were identical to TDN, suggesting that the malfunctioning systems had a similar effect on TDP treatment. A substantial increase in TDP mass reduction was also observed between downgradient piezometers at Site 100, which was likely due to plant activity in the vegetated buffer. Phosphate is a plant available nutrient and was the dominant TDP species at all sampling locations at Site 100 except for downgradient piezometers (Table 3).

Estimates of TDP mass reductions also suggest that removal processes likely accounted for most of the treatment. The median concentration reduction of TDP in groundwater near drainfields was approximately 95%, 95%, and 99% for Sites 100, 300, and 400, respectively. Piezometers located 15 and 30 m downgradient of the system at Site 100 had a median concentration reduction of TDP of 86% and 99%, respectively. Comparing these reductions to median mass reductions (Table 6) suggests that mass removal processes accounted for most of the TDP treatment at all sites. Thus, processes that remove phosphorus (e.g., adsorption, mineralization, and biological uptake) accounted for most of the treatment observed at these sites.

**Table 6.** Two-component mixing model using median chloride (Cl) and total dissolved phosphorus (TDP) concentrations to estimate mass reduction between tank and specified groundwater location.

| Location | Cl ($mg\ L^{-1}$) | Fraction WW | Fraction GW | Predicted TDP ($mg\ L^{-1}$) | Observed TDP ($mg\ L^{-1}$) | Cl/TDP Ratio | TDP Mass Reduction (%) |
|---|---|---|---|---|---|---|---|
| Site 100 | | | | | | | |
| Tank | 69.76 | 1.00 | 0.00 | | 7.91 | 8.81 | |
| DF | 26.54 | 0.33 | 0.67 | 2.63 | 0.39 | 68.50 | 85.26% |
| 15 m | 26.65 | 0.33 | 0.67 | 2.64 | 1.10 | 24.13 | 58.20% |
| 30 m | 18.04 | 0.20 | 0.80 | 1.59 | 0.02 | 774.22 | 98.53% |
| BG | 5.04 | 0.00 | 1.00 | | 0.01 | 406.70 | |
| | | | | | | | |
| Site 300 | | | | | | | |
| Tank | 62.07 | 1.00 | 0.00 | | 5.65 | 11.00 | |
| DF | 39.39 | 0.60 | 0.40 | 3.40 | 0.29 | 135.89 | 91.47% |
| BG | 5.04 | 0.00 | 1.00 | | 0.01 | 406.70 | |
| | | | | | | | |
| Site 400 | | | | | | | |
| Tank | 52.47 | 1.00 | 0.00 | | 8.26 | 6.35 | |
| DF | 31.37 | 0.56 | 0.44 | 4.59 | 0.10 | 326.49 | 97.90% |
| BG | 5.04 | 0.00 | 1.00 | | 0.01 | 406.70 | |

## 3.5 Stream Nutrient Concentrations and Mass Exports

### *3.5.1 Concentrations of Total Nitrogen and Total Phosphorus*

Nutrient concentrations in streams differed considerably between Sites 100 and 300 (Table 7). The median concentration of TN at Site 300 was 5.29 mg $L^{-1}$, which was more than double the median concentration at Site 100. This difference was statistically significant at $p < 0.01$. Concentrations of TN consisted primarily of TDN species (Table 7), which mostly consisted of nitrate at Site 100 and ammonium at Site 300 (Table 2). Site 300 had a median TDN concentration of 4.86 mg $L^{-1}$, which was approximately double the median at Site 100 and this difference was also statistically significant ($p < 0.01$). The stream at Site 300 also contained elevated PN concentrations relative to Site 100, but this difference was not significant ($p = 0.08$). Phosphorus concentrations exhibited similar trends when comparing the streams at these 2 sites (Table 7). Median concentrations of TP at Sites 100 and 300 were 0.11 and 0.59 mg $L^{-1}$, respectively. Thus, the median concentration of TP at Site 300 was more than 5 times greater, which was a significant difference ($p < 0.01$). Most of the TP concentration consisted of PP at both streams (Table 7). The stream at Site 300 also contained PP concentrations that were substantially greater than Site 100 and this difference was significant ($p < 0.01$).

Median concentrations of TP and PP were similar at Site 300, suggesting that this stream tended to contain elevated PP relative to TDP. Furthermore, PP concentrations were highly variable, ranging from 0.13 – 5.12 mg $L^{-1}$, whereas TDP concentrations tended to be less variable (Table 7). At Site 100, median concentrations of PP and TDP were more similar, and the range of TDP concentrations were more variable than PP. The differences in PP concentrations between streams may be related to TSS. Phosphorus may become adsorbed to sediment particles and transported to surface waters during erosion events. The stream at Site 300 contained considerably greater TSS concentrations relative to Site 100 and this difference was statistically significant ($p = 0.04$).

**Table 7.** Median (range) of nitrogen, phosphorus, sediment, carbon, and chloride (Cl) concentrations in the streams at Site 100 and 300. TN= total nitrogen; PN= particulate nitrogen; TDN= total dissolved nitrogen; TP= total phosphorus; PP= particulate phosphorus; TDP= total dissolved phosphorus; TSS= total suspended solids; DOC= dissolved organic carbon.

| Parameter | Concentration (mg $L^{-1}$) | |
|---|---|---|
| | **100 Stream** | **300 Stream** |
| TN | 2.45 (2.16 - 3.99) | 5.29 (4.28 - 9.88) |
| PN | 0.19 (0.09 - 0.61) | 0.33 (0.14 - 2.41) |
| TDN | 2.29 (1.65 - 3.66) | 4.86 (3.95 - 7.49) |
| TP | 0.11 (0.07 - 1.48) | 0.59 (0.22 - 5.18) |
| PP | 0.06 (0.03 - 0.13) | 0.54 (0.13 - 5.12) |
| TDP | 0.04 (0.02 - 1.38) | 0.10 (0.03 - 0.23) |
| TSS | 39.62 (26.84 - 82.80) | 241.11 (16.00 - 1384.44) |
| DOC | 8.24 (7.07 - 14.10) | 14.79 (10.35 - 19.43) |
| Cl | 11.30 (6.35 - 15.61) | 18.19 (12.57 - 21.97) |

*3.5.2 Nutrient Mass Export and* E. coli *Loads from Streams*

Nutrient mass export by streams tended to be elevated in Site 300 compared to Site 100. Median mass export of TDN and TN at Site 300 was approximately 56 and 67 g $day^{-1}$, respectively. These median values were nearly twice that of TDN (30 g $day^{-1}$) and TN (39 g $day^{-1}$) exports from the stream at Site 100. However, differences in nitrogen exports were not statistically significant for TDN ($p = 0.19$) nor TN ($p = 0.19$). Differences in mass exports of phosphorus between streams tended to be more distinct (Fig. 10). Median export of TDP and TP at Site 300 was approximately 1.5 and 14 g $day^{-1}$, respectively. These exports were approximately 2 and 7 times greater than median exports of TDP (0.6 g $day^{-1}$) and TP (2 g $day^{-1}$), respectively, at Site 100 (Fig. 10). Differences in TP masses were statistically significant at $p = 0.05$, but TDP masses between streams were not statistically significant ($p = 0.19$).

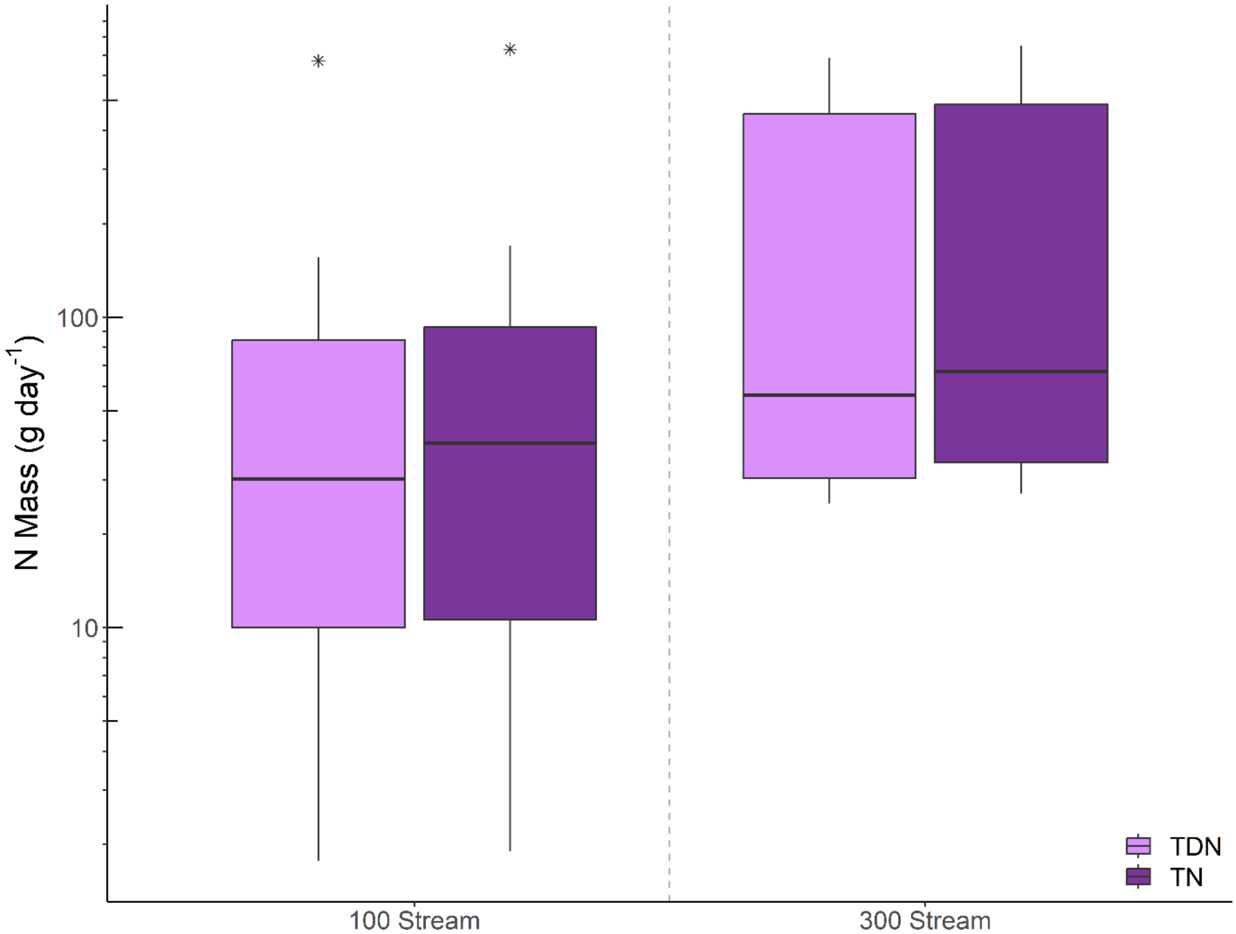


**Figure 9.** Mass exports of total dissolved nitrogen (TDN) and total nitrogen (TN) from streams at Sites 100 and 300.

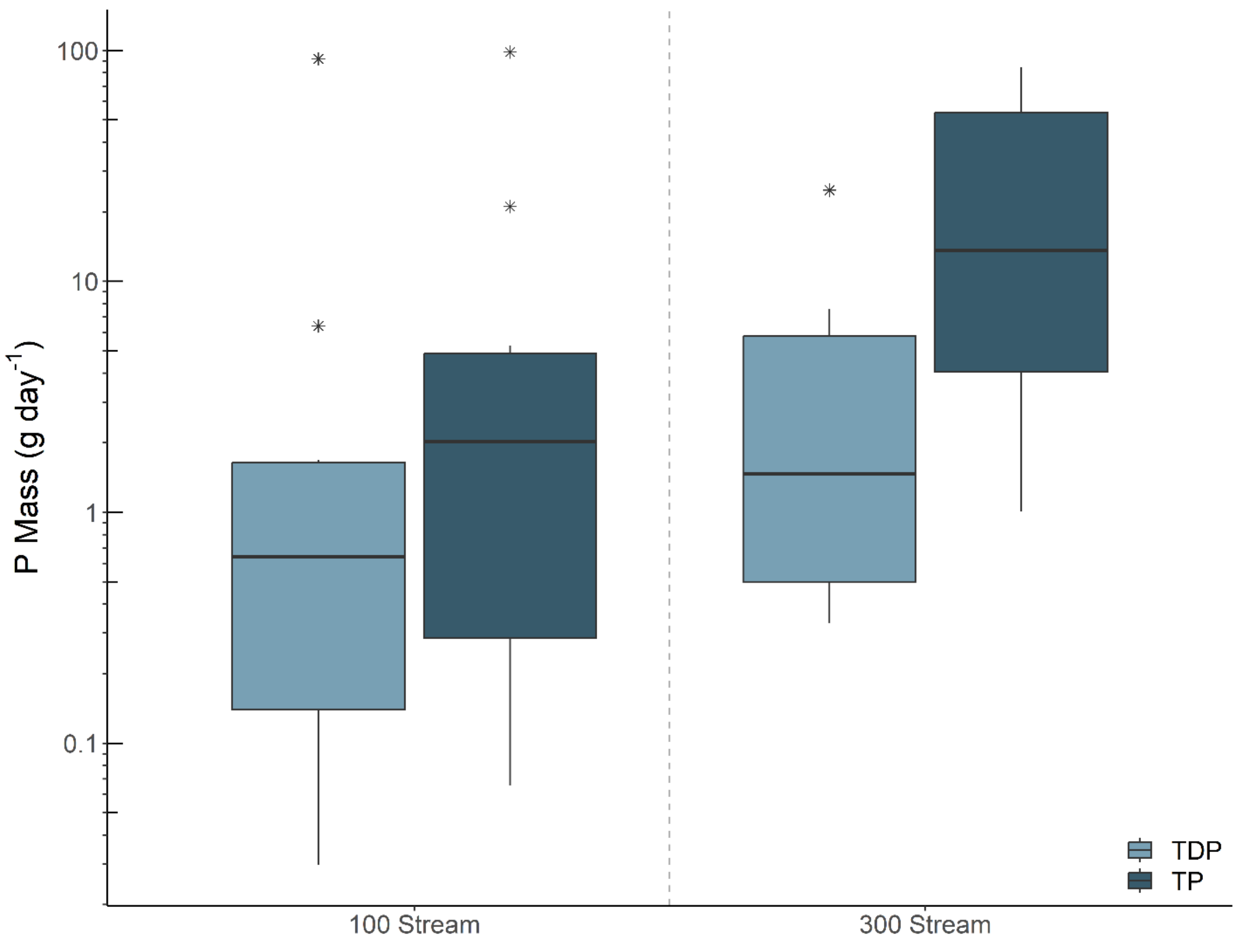


**Figure 10.** Mass exports of total dissolved phosphorus (TDP) and total phosphorus (TP) from streams at Sites 100 and 300.

*E. coli* loadings from streams yielded similar trends as nutrient masses (Fig. 11). The stream at Site 300 typically contained greater *E. coli* loadings compared to Site 100. The median *E. coli* loading from Site 300 was 116,141 MPN $min^{-1}$, which was nearly twice as much as Site 100 (60,095 MPN $min^{-1}$). Despite these differences, the *E. coli* loadings tended to overlap between streams, thus loads did not significantly differ between streams ($p = 0.19$).

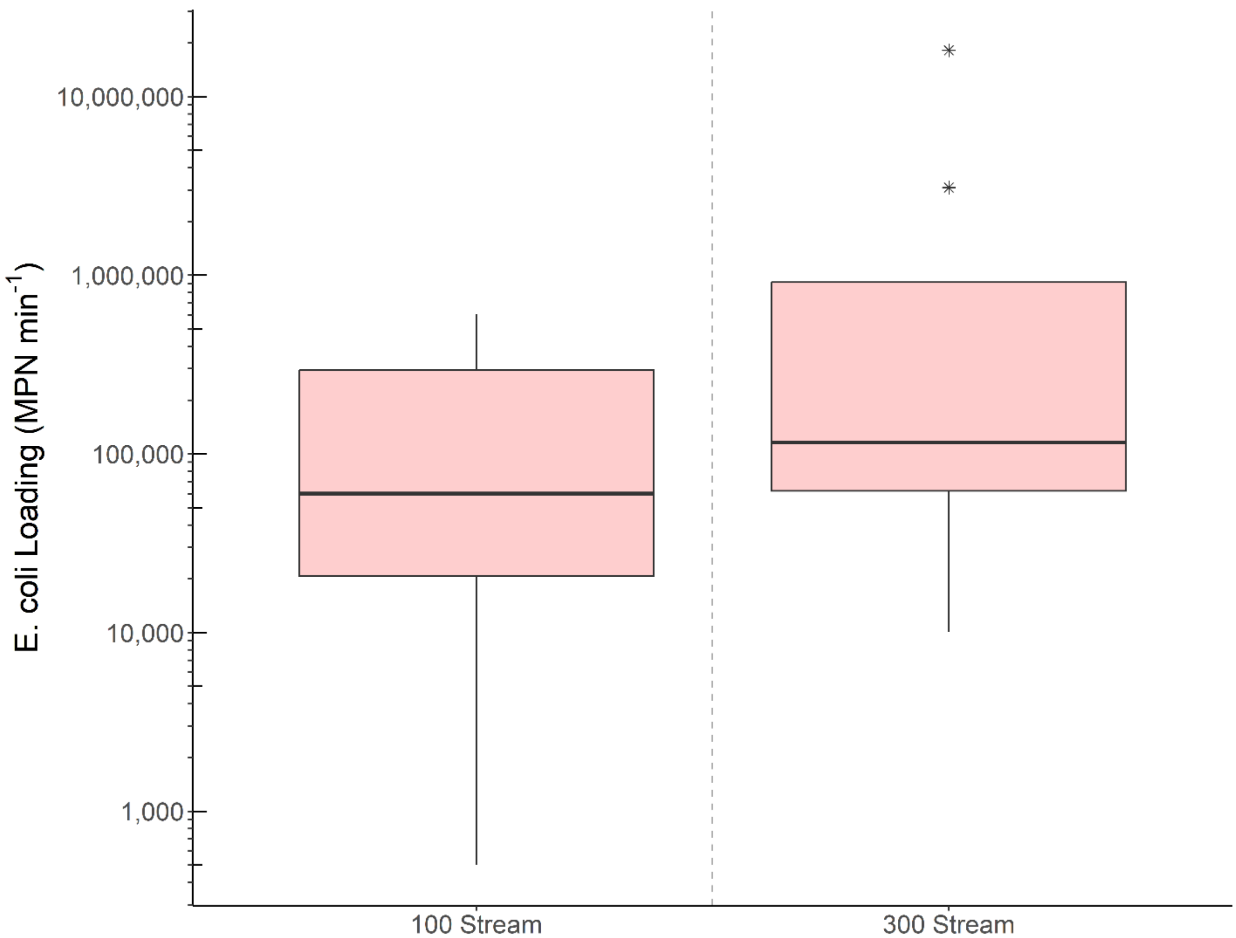


**Figure 11.** *Escherichia coli* loadings from streams at Sites 100 and 300.

The lack of statistical significance between streams for most of the nutrient and bacteria pollutants was likely due to the similarity in stream discharge. Site 300 tended to contain elevated concentrations relative to Site 100, whereas Site 100 tended to contain greater stream discharge. Median stream discharges were 11.9 L $min^{-1}$ and 8.1 L $min^{-1}$ for Sites 100 and 300, respectively (Table 8). Although discharge was highly variable at both sites and differences in discharge also were not statistically significant ($p = 0.38$).

3.6 Physicochemical Parameters

In addition to nutrient and *E. coli* data, physicochemical parameters indicated that septic systems influenced water quality (Table 8). SC, pH, ORP, and DO data suggest that groundwater and surface water was affected by septic system effluent. Wastewater can be a source of dissolved ions that increase SC values and septic tanks contained the greatest values of SC. Groundwater near drainfields contained the next highest values of median SC, followed by downgradient groundwater, and streams. Groundwater and surface water recharged by septic system effluent also contained elevated median SC values relative to background groundwater (Table 8). Another factor that can be indicative of wastewater influence is pH. Nitrification is a process by which ammonium transforms into nitrate and relies on aerobic soils. During the oxidation of ammonium ($NH_4^+$), nitrogen releases hydrogen ions and subsequently binds with oxygen to form nitrate ($NO_3^-$), water ($H_2O$), and some hydrogen ions are released. The released hydrogen ions cause a reduction of pH. Groundwater near drainfields consistently contained a lower pH relative to wastewater, which may be an indicator that nitrification occurred. However, nitrification may have been limited at Sites 100 and 300 due to insufficient separation distance, which is a zone of aerated soil measured from the trench bottom to water table (Fig. 12). North Carolina requires a minimum separation distance of at least 30 cm (12 in) for Group II – IV soils or 45 cm (18 in) for Group I soils. Insufficient separation distance could inhibit nitrification, which would explain the tendency for ammonium to be the predominant species of TDN in drainfield groundwater. Thus, reduced pH was not likely due to nitrification, but may instead be due to mixing of septic system effluent and background groundwater. Background groundwater had a median pH of 5.84, which was about 1 pH unit lower than wastewater in tanks (Table 8). Values of pH also typically decreased as distance away from the septic system increased. At Site 300, the stream contained lower pH values relative to wastewater and drainfield groundwater collected from this site. This result, along with the increased nitrate concentrations (Table 2), suggested that nitrification occurred after ammonium-rich groundwater discharged to the stream.

Both ORP and DO suggest that wastewater may have also affected water resources. The lowest values of ORP and DO were typically observed in the septic tanks, which was expected given the anaerobic environment. ORP and DO values increased in groundwater and surface water relative to wastewater (Table 8). Site 300 typically contained the lowest ORP and DO values in both drainfield groundwater and streams relative to the other sites, which likely occurred due to inadequate treatment of wastewater. Both ORP and DO in the stream at Site 300 tended to be much lower than the stream at Site 100.

**Table 8.** Median (range) of physicochemical parameters summarized by site and sampling location. T= tank; DF= drainfield; DG= downgradient (Site 100 only); S= stream; DTW= depth to water; Temp= temperature; SC= specific conductance; ORP= oxidation-reduction potential; DO= dissolved oxygen; Turb= turbidity; Q= discharge.

| Location | DTW (cm) | Temp (°C) | SC (µS $cm^{-1}$) | pH | ORP (mV) | DO (mg $L^{-1}$) | Turb (FNU) | Q (L $min^{-1}$) |
|---|---|---|---|---|---|---|---|---|
| Site 100 | | | | | | | | |
| T | | 17.2 (11.4 - 23.7) | 1033 (676 - 1145) | 7.04 (6.75 - 7.99) | -263.5 (-470.0 - -224.0) | 1.0 (<0.1 - 2.6) | | |
| DF | 70.10 (32.92 - 204.22) | 13.8 (8.0 - 18.6) | 394 (69 - 967) | 6.56 (5.37 - 7.88) | -31.0 (-218.0 - 240.0) | 2.9 (<0.1 - 6.4) | | |
| DG | 88.70 (61.57 - 115.82) | 14.3 (10.3 - 19.1) | 241 (150 - 382) | 5.73 (5.33 - 7.29) | -5.2 (-219.0 - 114.0) | 2.0 (<0.1 - 5.2) | | |
| S | | 12.6 (6.5 - 18.7) | 134 (112 - 170) | 6.14 (5.90 - 7.60) | 29.8 (-167.0 - 157.3) | 6.4 (2.6 - 9.1) | 5.3 (0.0 - 169.0) | 11.9 (0.6 - 127.3) |
| Site 300 | | | | | | | | |
| T | | 18.6 (11.8 - 25.5) | 870 (692 - 1148) | 7.06 (6.72 - 7.93) | -256.0 (-326.0 - -215.1) | 1.0 (<0.1 - 2.6) | | |
| DF | 57.91 (0.00 - 130.45) | 14.9 (6.0 - 21.7) | 604 (99 - 1734) | 6.51 (5.72 - 8.36) | -122.0 (-276.0 - 17.5) | 1.2 (<0.1 - 3.9) | | |
| S | | 14.7 (10.0 - 19.3) | 211 (186 - 264) | 5.87 (5.58 - 7.12) | -66.9 (-212.0 - -21.0) | 2.4 (0.6 - 3.8) | 8.0 (0.0 - 509.0) | 8.1 (2.6 - 99.6) |
| Site 400 | | | | | | | | |
| T | | 17.3 (13.8 - 19.6) | 937 (839 - 1124) | 6.95 (6.80 - 7.23) | -260.5 (-273.0 - -241.0) | 1.6 (<0.1 - 2.5) | | |
| DF | 307.70 (256.03 - 355.70) | 16.9 (16.0 - 18.0) | 342 (137 - 484) | 5.78 (5.19 - 6.72) | 31.9 (-157.0 - 75.0) | 2.6 (0.2 - 3.5) | | |
| Pooled | | | | | | | | |
| T | | 18.1 (11.4 - 25.5) | 930 (676 - 1148) | 7.00 (6.72 - 7.99) | -256.5 (-470.0 - -215.1) | 1.4 (<0.1 - 2.6) | | |
| DF | 68.58 (0.00 - 355.70) | 16.0 (6.0 - 21.7) | 431 (69 - 1734) | 6.44 (5.19 - 8.36) | -90.1 (-276.0 - 240.0) | 2.5 (<0.1 - 6.4) | | |
| S | | 14.4 (6.5 - 19.3) | 178 (112 - 264) | 6.07 (5.58 - 7.60) | -39.3 (-212.0 - 157.3) | 3.7 (0.6 - 9.1) | 6.7 (0.0 - 509.0) | 8.1 (0.6 - 127.3) |
| BG | 124.97 (108.51 - 132.89) | 15.4 (12.3 - 19.3) | 90 (52 - 151) | 5.84 (5.41 - 8.20) | -49.0 (-131.2 - 27.2) | 5.3 (1.9 - 6.1) | | |

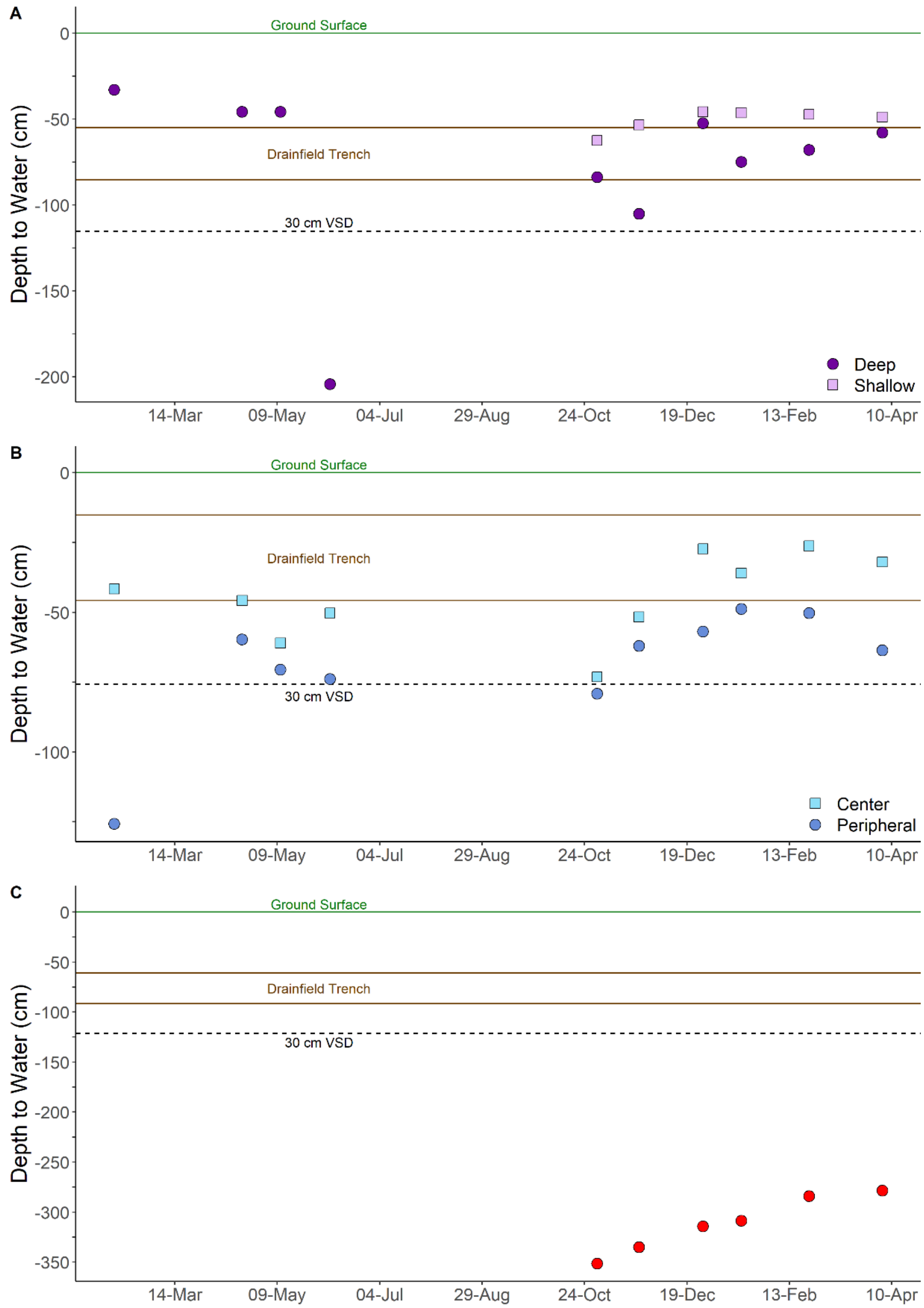


**Figure 12.** Depth to water in drainfield piezometers to estimated trench bottom and the 30 cm vertical separation distance (VSD) requirement for Sites 100 (A), 300 (B), and 400 (C). Site 100 only used data from the shallow and deep piezometer from the nest (Fig. 1). At Site 300, drainfield piezometers were grouped based on location relative to surfaced wastewater. At Site 400, the median of the 3 drainfield piezometers was used.

## 4. Summary of Research Findings

The goal of this study was to assess lot-scale septic system performance by 3 septic systems located in Raleigh Belt geology. Wastewater contained the highest concentrations of pollutants, with a median TDN, TDP, and *E. coli* concentration of 62.15 mg $L^{-1}$, 7.23 mg $L^{-1}$, and 93,600 MPN 100 $mL^{-1}$ when pooling the data. Septic systems were most effective at reducing *E. coli* concentrations with groundwater near drainfields containing median concentrations that were 98% to >99% lower than wastewater concentrations. Groundwater downgradient of the septic system (Site 100 only) and adjacent streams (Sites 100 and 300) contained median *E. coli* concentrations that were >99% lower than wastewater. Septic systems also contained high reduction efficiencies for median TDP concentrations. Drainfield groundwater contained a median TDP concentration that was 95% – 99% lower than wastewater concentrations. This median reduction increased to >99% in downgradient groundwater and 98% – >99% in streams. Concentration reductions of TDN exhibited the most variability between sites. Drainfield groundwater contained median TDN concentrations that were 63% – 98% lower than wastewater. The median TDN concentration in downgradient groundwater was 97% lower than wastewater, and in the streams, it ranged from 91% – 97%. Nutrient mass reductions also varied between sites. Mass reduction of TDP ranged from 85% – 98% between tanks and drainfield groundwater. Downgradient groundwater was subdivided by distance downslope from the system at Site 100. The lowest TDP mass reduction between tank and groundwater occurred at the piezometer located 15 m with a reduction of 58%. However, TDP mass reduction estimates increased to 99% in piezometers located 30 m downgradient. Estimates of TDN mass reduction were lower than TDP but yielded similar trends. Mass reduction of TDN ranged from 38% – 97% between tanks and drainfield groundwater. The lowest mass reduction of TDN was 25% when comparing wastewater to a piezometer 15 m downgradient. However, estimates of TDN mass reduction increased to 88% in downgradient piezometers located 30 m downslope. Concentration and mass reductions suggest that some systems were highly effective at reducing pollutant concentrations, whereas others were not as effective.

Differences in septic system performance were most likely related to differences in site characteristics that affected treatment. The highest concentration and mass reductions were observed at Site 400. Despite high treatment efficiencies for nutrients and *E. coli* at this site, drainfield groundwater contained contaminant concentrations that were elevated relative to background. This site borders a forested area, and the septic system is approximately 45 m (~150 ft) away from the nearest stream, thus additional treatment likely occurred in the subsurface before reaching surface waters. Site 300 typically had the lowest treatment efficiency, while treatment at Site 100 was typically intermediary. However, there were 2 occasions when performance declined substantially at Site 100. During all but 2 sampling events, the piezometer located 15 m downgradient of the system was dry. However, on 28 December 2022 and 24 February 2023, this piezometer yielded groundwater that contained a median concentration of TDN and TDP that was greater than all other sampling locations (Tables 5 and 6). During these sampling events, surfaced effluent was noted near the forementioned piezometer. Thus, the most plausible reason for the variability in performance between sites was likely related to septic system malfunction. Sites 100 and 300 routinely contained DTW that violated the 30 cm vertical setback distance. Malfunctions likely inhibited biogeochemical processes that reduce or remove nitrogen, phosphorus, and *E. coli* at these sites thereby reducing treatment potential. Thus, the elevated nutrient and *E. coli* concentration and SC coupled with the reduced ORP and DO values in groundwater at Sites 100 and 300 compared to groundwater at Site 400 were likely driven by malfunctions. Furthermore, water quality data from the stream at Site 300 suggests that malfunctioning septic systems can degrade surface water quality if the systems are in close proximity to surface water features.

Results from the current study suggest that inadequately performing septic systems exhibit high potential to negatively affect water quality in downstream surface waters. Efforts to identify and repair malfunctioning septic systems could substantially improve septic system performance thereby reducing delivery of nutrients and bacteria, such as *E. coli* and others, to water resources. A program to identify malfunctioning systems can take shape in a variety of ways via field inspections coupled with geospatial analysis. Geospatial analysis can be an effective starting point in identifying underperforming and/or malfunctioning septic systems. The USDA Web Soil Survey data can be integrated into a GIS database along with other hydrologic and municipal layers to evaluate potential hotspots within a sub-watershed based on soil series information and location of septic systems. Malfunctioning septic systems can be identified at relatively low-cost via site inspections to identify and document visual and olfactory indicators of septic system failure (e.g., upwelling of malodorous water located near septic system components). Furthermore, water levels in septic tanks can be monitored to ensure that sufficient freeboard exists in the tank. Monitoring water table depth to identify violations of vertical separation distance offers another solution to identify malfunctioning septic systems. In the current study, there were some sampling events that occurred where no obvious visual or olfactory indicators of malfunction were noticed; however, the depth to water data indicated there was insufficient vertical separation distance. Locating malfunctional and/or underperforming septic systems can help guide retrofit activities for individual septic systems and/or sub-watersheds with elevated system densities. For malfunctional systems, these activities could include repairing and/or upgrading septic systems to improve system performance. For underperforming systems, activities could include engineering nature-based solutions to restore or enhance biogeochemical processes to improve treatment efficiency. Functional septic systems can treat wastewater to similar standards as municipal wastewater systems [42, 43]. Thus, identifying and remediating inadequately performing septic systems is a key component of reaching water quality goals, especially in nutrient-sensitive watersheds that heavily rely on septic systems for wastewater management.

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## APPENDIX A: CITIZEN PHOTOS

Visitors to Lake Benson Park have publicly shared photographs of scenery at the park over the past several years. In reviewing these photos, one can identify several algal bloom events from 2019 – 2022, which occurred in late Summer or early Fall. A limitation of these photos is that it is difficult to spatially pinpoint the exact location where the image was shot. Photos were taken by park guests from 2019 to 2022 and were available at the Google page for Lake Benson Park as of 28 Jul 2022 (see link below).
https://www.google.com/maps/uv?pb=!1s0x89ac61b15209a937%3A0xa00af406a9ab5c39!3m1!7e115!4shttps%3A%2F%2Flh5.googleusercontent.com%2Fp%2FAF1QipNfRbW6vuvVoeU4ezSbRmlHqY7Sx5ToCcFf0dMV%3Dw213-h160-k-no!5slake%20benson%20park%20-%20Google%20Search!15sCgIgAQ&imagekey=!1e10!2sAF1QipNfRbW6vuvVoeU4ezSbRmlHqY7Sx5ToCcFf0dMV&hl=en&sa=X&ved=2ahUKEwiXsZb-5pv5AhXBElkFHQdwDNMQoip6BAh3EAM

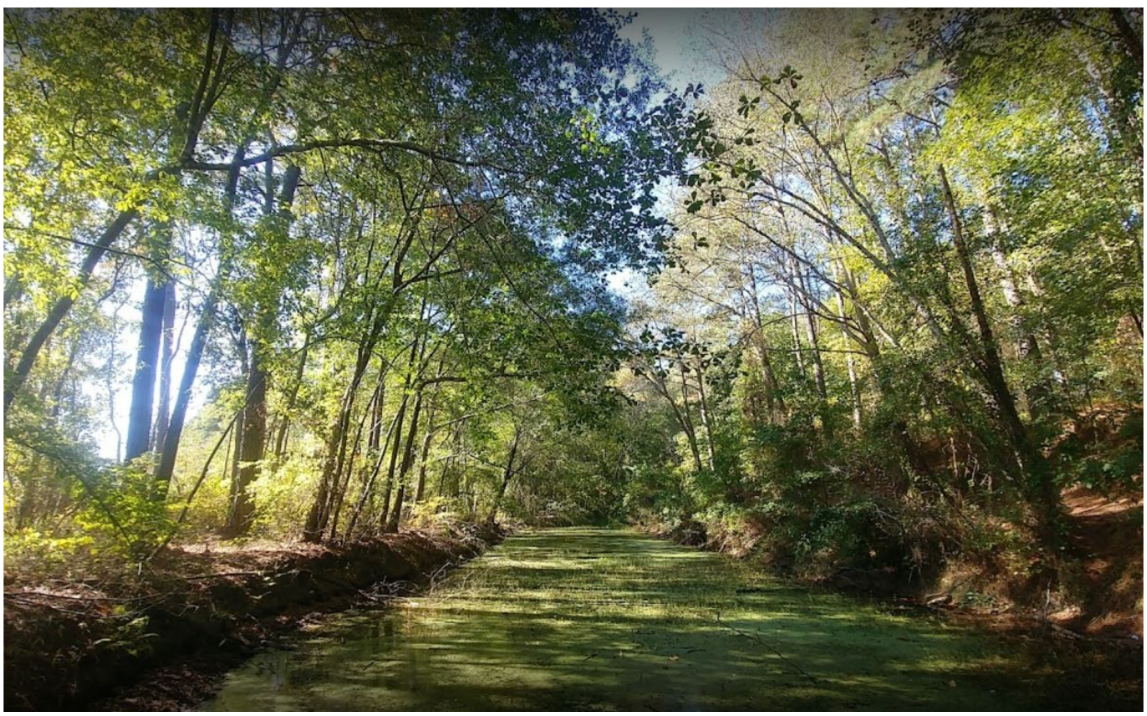

**Figure A1.** Algal bloom in October 2019 at Lake Benson Park.

**Figure A2.** Algal bloom event at Lake Benson Park in September 2020.

**Figure A3.** An algal bloom event at Lake Benson Park on 6 July 2022.

**Figure A4.** Another algal bloom at Lake Benson Park on 6 Jul 2022.

## APPENDIX B: BACKGROUND MAP

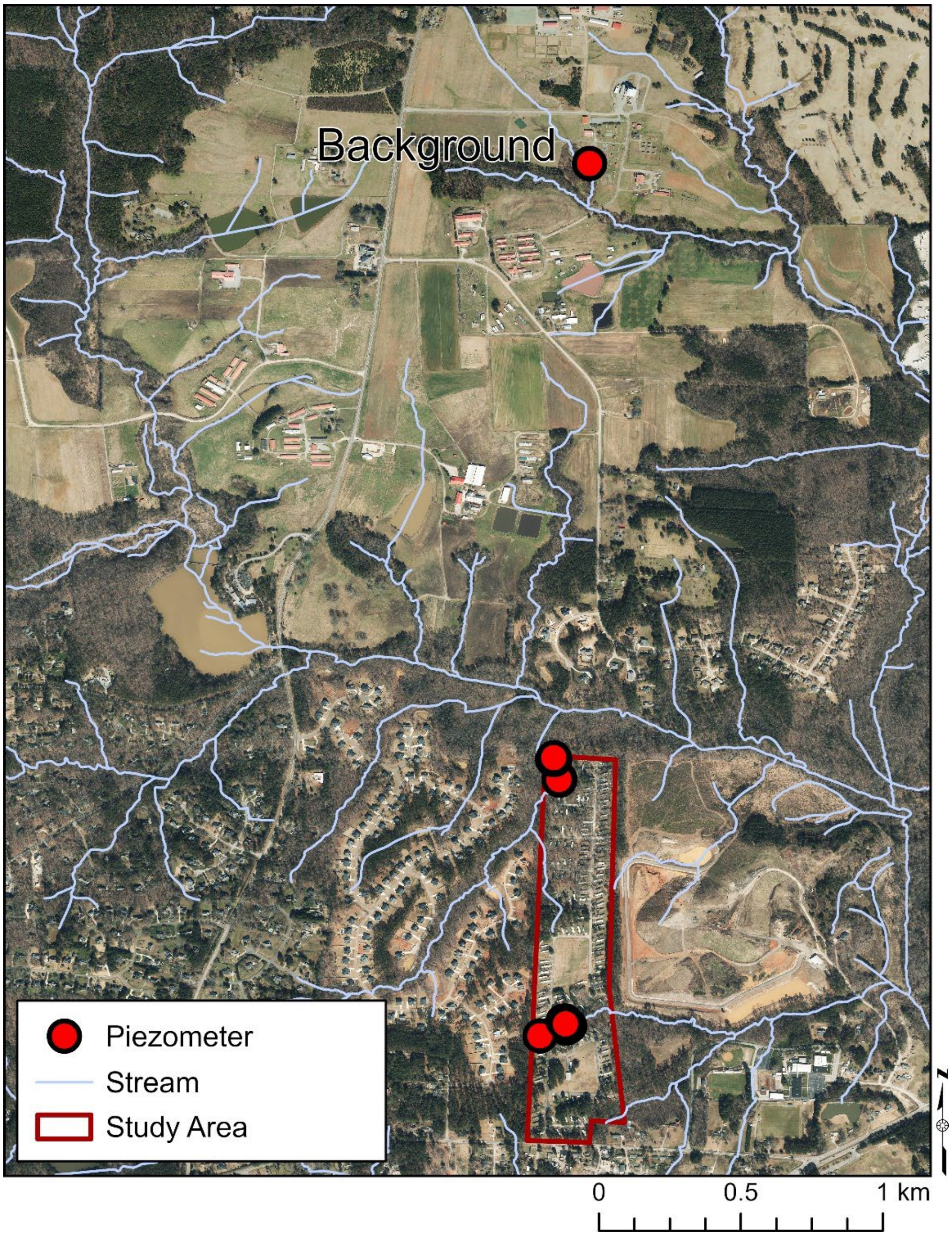


**Figure B1.** Map showing location of background piezometer relative to study area, which is within 3.2 km (2 miles) of the study area.

## APPENDIX C: SUPPLEMENTARY DATA TABLES

The following data tables replicate the major analyses in the body of the report. The data summarized here only includes sampling events from the 2022 – 2023 NC Policy Collaboratory. For all tables, the following abbreviations were commonly used:

- TN= total nitrogen
- PN= particulate nitrogen
- TDN= total dissolved nitrogen
- $NO_3^-$-N= nitrate-nitrogen
- $NH_4^+$-N= ammonium-nitrogen
- DON= dissolved organic nitrogen
- TP= total phosphorus
- PP= particulate phosphorus
- TDP= total dissolved phosphorus
- $PO_4^-$-P= orthophosphate
- DOP= dissolved organic phosphorus
- Cl= chloride
- *E. coli*= *Escherichia coli*
- DTW= depth to water
- Temp= temperature
- SC= specific conductance
- ORP= oxidation-reduction potential
- DO= dissolved oxygen
- Turb= turbidity
- Q= discharge

**Table C1.** Median (range) of nitrogen (N) species summarized by location at the lot scale and pooled across sites.

| Location | Concentration of N Species (mg $L^{-1}$) TDN | $NO_3^-$-N | $NH_4^+$-N | DON |
|---|---|---|---|---|
| Site 100 | | | | |
| Tank | 60.66 (36.59 - 75.44) | 0.01 (<0.01 - 0.02) | 59.55 (36.57 - 73.18) | 1.10 (<0.01 - 6.10) |
| Drainfield | 17.84 (0.84 - 53.74) | 0.17 (<0.01 - 14.50) | 14.14 (0.10 - 53.69) | 0.07 (<0.01 - 4.28) |
| Downgradient | 1.48 (0.60 - 19.38) | 0.08 (0.02 - 10.98) | 0.94 (0.16 - 8.41) | 0.61 (<0.01 - 2.16) |
| Stream | 2.09 (1.65 - 3.66) | 1.88 (0.80 - 2.53) | 0.15 (0.04 - 0.50) | 0.07 (<0.01 - 0.98) |
| Site 300 | | | | |
| Tank | 56.41 (34.30 - 67.61) | 0.01 (<0.01 - 0.05) | 51.86 (34.26 - 66.22) | 3.31 (<0.01 - 10.61) |
| Drainfield | 29.90 (1.35 - 93.60) | 0.57 (<0.01 - 14.92) | 25.33 (0.23 - 93.58) | 0.28 (<0.01 - 9.70) |
| Stream | 4.64 (3.95 - 6.43) | 2.11 (1.19 - 3.15) | 2.55 (0.39 - 4.09) | 0.00 (<0.01 - 1.61) |
| Site 400 | | | | |
| Tank | 67.35 (55.83 - 79.06) | 0.03 (<0.01 - 0.07) | 64.03 (55.77 - 79.04) | 2.26 (<0.01 - 3.85) |
| Drainfield | 1.26 (0.37 - 3.13) | 0.06 (<0.01 - 2.24) | 0.16 (<0.01 - 2.37) | 0.42 (<0.01 - 1.02) |
| Pooled | | | | |
| Tank | 64.13 (34.30 - 79.06) | 0.01 (<0.01 - 0.07) | 61.11 (34.26 - 79.04) | 1.99 (<0.01 - 10.61) |
| Drainfield | 8.71 (0.37 - 93.60) | 0.17 (<0.01 - 14.92) | 5.75 (<0.01 - 93.58) | 0.34 (<0.01 - 9.70) |
| Stream | 3.81 (1.65 - 6.43) | 1.93 (0.80 - 3.15) | 0.49 (0.04 - 4.09) | <0.01 (<0.01 - 1.61) |
| Background | 0.95 (0.56 - 1.95) | 0.28 (0.04 - 1.04) | 0.14 (<0.01 - 0.39) | 0.45 (0.23 - 0.91) |

**Table C2.** Median (range) of phosphorus (P) species summarized by location at the lot scale and pooled across sites.

| Location | Concentration of P Species (mg $L^{-1}$) | | |
|---|---|---|---|
| | TDP | $PO_4^-$-P | DOP |
| Site 100 | | | |
| Tank | 7.89 (4.82 - 14.22) | 6.69 (4.82 - 9.01) | 0.13 (<0.01 - 9.30) |
| Drainfield | 0.64 (0.01 - 7.43) | 0.64 (<0.01 - 5.37) | 0.00 (<0.01 - 3.87) |
| Downgradient | 0.03 (<0.01 - 1.16) | 0.01 (<0.01 - 1.16) | 0.01 (<0.01 - 0.22) |
| Stream | 0.04 (0.02 - 1.38) | 0.03 (0.02 - 1.38) | 0.01 (<0.01 - 0.01) |
| Site 300 | | | |
| Tank | 6.21 (3.14 - 14.41) | 5.24 (1.75 - 5.94) | 1.37 (<0.01 - 8.61) |
| Drainfield | 3.21 (0.08 - 12.82) | 3.05 (0.02 - 10.27) | 0.03 (<0.01 - 2.97) |
| Stream | 0.11 (0.03 - 0.23) | 0.02 (<0.01 - 0.16) | 0.05 (0.01 - 0.18) |
| Site 400 | | | |
| Tank | 8.26 (6.37 - 8.83) | 7.77 (6.30 - 8.50) | 0.21 (<0.01 - 0.71) |
| Drainfield | 0.10 (0.01 - 0.18) | 0.07 (<0.01 - 0.16) | 0.00 (<0.01 - 0.04) |
| Pooled | | | |
| Tank | 7.23 (3.14 - 14.41) | 6.47 (1.75 - 9.01) | 0.39 (<0.01 - 9.30) |
| Drainfield | 0.27 (0.01 - 12.82) | 0.16 (<0.01 - 10.27) | 0.01 (<0.01 - 3.87) |
| Stream | 0.05 (0.02 - 1.38) | 0.03 (<0.01 - 1.38) | 0.01 (<0.01 - 0.18) |
| Background | 0.01 (<0.01 - 0.03) | 0.00 (<0.01 - 0.02) | 0.01 (<0.01 - 0.02) |

**Table C3.** Median (range) and geometric mean values of *E. coli* concentrations summarized by location at the lot scale and pooled across sites.

| Location | Median (Range) *(MPN 100mL$^{-1}$)* | Geometric Mean *(MPN 100mL$^{-1}$)* |
|---|---|---|
| Site 100 | | |
| Tank | 49500 (20500 - 86400) | 49902 |
| Drainfield | 610.2 (7.5 - 60500) | 682.4 |
| Downgradient | 161.6 (1.0 - 967.2) | 60.9 |
| Stream | 251.7 (83.3 - 2419.6) | 402.0 |
| Site 300 | | |
| Tank | 97200 (41200 - 4839200) | 170554 |
| Drainfield | 2419.6 (75.4 - 282720) | 3049 |
| Stream | 1839.8 (550.0 - 24196) | 2282.2 |
| Site 400 | | |
| Tank | 65200 (10000 - 181800) | 50220 |
| Drainfield | 248.1 (7.5 - 2419.6) | 202.7 |
| Pooled | | |
| Tank | 67200 (10000 - 4839200) | 75327 |
| Drainfield | 814.6 (7.5 - 282720) | 856.4 |
| Stream | 1190.0 (83.3 - 24196) | 957.8 |
| Background | 4.2 (1.0 - 26.2) | 4.6 |

**Table C4.** Two-component mixing model using median chloride (Cl) and total dissolved nitrogen (TDN) concentrations for each sampling location.

| Location | Cl (mg $L^{-1}$) | Fraction WW | Fraction GW | Predicted TDN (mg $L^{-1}$) | Observed TDN (mg $L^{-1}$) | Cl/TDN Ratio | TDN Mass Reduction (%) |
|---|---|---|---|---|---|---|---|
| Site 100 | | | | | | | |
| Tank | 58.69 | 1.00 | 0.00 | | 60.66 | 0.97 | |
| DF | 31.21 | 0.49 | 0.51 | 29.58 | 17.84 | 1.75 | 39.69% |
| 15 m | 26.65 | 0.40 | 0.60 | 24.43 | 16.88 | 1.58 | 30.88% |
| 30 m | 22.55 | 0.33 | 0.67 | 19.80 | 1.43 | 15.82 | 92.80% |
| BG | 5.04 | 0.00 | 1.00 | | 0.95 | 5.33 | |
| Site 300 | | | | | | | |
| Tank | 58.82 | 1.00 | 0.00 | | 56.41 | 1.04 | |
| DF | 42.04 | 0.69 | 0.31 | 38.82 | 29.90 | 1.41 | 22.98% |
| BG | 5.04 | 0.00 | 1.00 | | 0.95 | 5.33 | |
| Site 400 | | | | | | | |
| Tank | 52.47 | 1.00 | 0.00 | | 67.35 | 0.78 | |
| DF | 31.37 | 0.56 | 0.44 | 37.40 | 1.26 | 24.95 | 96.64% |
| BG | 5.04 | 0.00 | 1.00 | | 0.95 | 5.33 | |

**Table C5.** Two-component mixing model using median chloride (Cl) and total dissolved phosphorus (TDP) concentrations for each sampling location to estimate mass reduction.

| Location | Cl (mg $L^{-1}$) | Fraction WW | Fraction GW | Predicted TDP (mg $L^{-1}$) | Observed TDP (mg $L^{-1}$) | Cl/TDP Ratio | TDP Mass Reduction (%) |
|---|---|---|---|---|---|---|---|
| Site 100 | | | | | | | |
| Tank | 58.69 | 1.00 | 0.00 | | 7.89 | 7.44 | |
| DF | 31.21 | 0.49 | 0.51 | 3.85 | 0.64 | 48.73 | 83.36% |
| 15 m | 26.65 | 0.40 | 0.60 | 3.18 | 1.10 | 24.13 | 65.24% |
| 30 m | 22.55 | 0.33 | 0.67 | 2.58 | 0.64 | 35.22 | 75.13% |
| BG | 5.04 | 0.00 | 1.00 | | 0.01 | 406.70 | |
| | | | | | | | |
| Site 300 | | | | | | | |
| Tank | 58.82 | 1.00 | 0.00 | | 6.21 | 9.47 | |
| DF | 42.04 | 0.69 | 0.31 | 4.27 | 3.21 | 13.08 | 24.75% |
| BG | 5.04 | 0.00 | 1.00 | | 0.01 | 406.70 | |
| | | | | | | | |
| Site 400 | | | | | | | |
| Tank | 52.47 | 1.00 | 0.00 | | 8.26 | 6.35 | |
| DF | 31.37 | 0.56 | 0.44 | 4.59 | 0.10 | 326.49 | 97.90% |
| BG | 5.04 | 0.00 | 1.00 | | 0.01 | 406.70 | |

**Table C6.** Median (range) of nitrogen species, phosphorus species, total suspended solids (TSS), dissolved organic carbon (DOC), and chloride (Cl) for the stream at Sites 100 and 300.

| Parameter | Concentration (mg $L^{-1}$) | |
|---|---|---|
| | **100 Stream** | **300 Stream** |
| TN | 2.35 (2.16 - 3.99) | 5.01 (4.28 - 6.72) |
| PN | 0.26 (0.16 - 0.61) | 0.33 (0.28 - 0.46) |
| TDN | 2.09 (1.65 - 3.66) | 4.64 (3.95 - 6.43) |
| | | |
| TP | 0.11 (0.08 - 1.48) | 0.58 (0.24 - 0.72) |
| PP | 0.07 (0.04 - 0.13) | 0.45 (0.13 - 0.61) |
| TDP | 0.04 (0.02 - 1.38) | 0.11 (0.03 - 0.23) |
| | | |
| TSS | 42.71 (35.60 - 82.80) | 500.52 (34.67 - 1384.44) |
| DOC | 7.50 (7.07 - 8.77) | 12.56 (10.35 - 15.96) |
| Cl | 13.47 (10.99 - 15.61) | 18.34 (17.65 - 21.97) |

**Table C7.** Median (range) of physicochemical parameters summarized by site and sampling location. T= tank; DF= drainfield; DG= downgradient (Site 100 only); S= stream; DTW= depth to water; Temp= temperature; SC= specific conductance; ORP= oxidation-reduction potential; DO= dissolved oxygen; Turb= turbidity; Q= discharge.

| Location | DTW (cm) | Temp (°C) | SC ($\mu S\ cm^{-1}$) | pH | ORP (mV) | DO ($mg\ L^{-1}$) | Turb (FNU) | Q ($L\ min^{-1}$) |
|---|---|---|---|---|---|---|---|---|
| Site 100 | | | | | | | | |
| T | | 16.6 (11.4 - 20.1) | 938 (676 - 1054) | 7.12 (6.75 - 7.99) | -243.9 (-310.0 - -224.0) | 1.6 (<0.1 - 2.6) | | |
| DF | 70.10 (45.72 - 176.17) | 14.2 (8.0 - 18.0) | 554 (69 - 967) | 6.66 (5.37 - 7.88) | -46.9 (-218.0 - 184.0) | 3.1 (<0.1 - 4.7) | | |
| DG | 88.70 (64.62 - 115.82) | 14.0 (10.3 - 17.4) | 241 (150 - 382) | 5.70 (5.33 - 7.29) | 5.0 (-57.9 - 91.5) | 3.1 (0.1 - 5.2) | | |
| S | | 13.2 (6.5 - 16.3) | 142 (129 - 170) | 6.08 (5.90 - 7.60) | 29.8 (-42.0 - 72.1) | 6.7 (5.2 - 9.1) | 5.3 (2.7 - 39.3) | 27.7 (<0.1 - 127.3) |
| Site 300 | | | | | | | | |
| T | | 18.0 (11.8 - 21.0) | 895 (692 - 1148) | 7.31 (6.72 - 7.93) | -256.0 (-305.0 - -242.9) | 1.6 (<0.1 - 2.6) | | |
| DF | 56.69 (0.00 - 91.44) | 13.2 (6.0 - 18.1) | 789 (99 - 1734) | 6.76 (5.72 - 8.36) | -121.1 (-276.0 - 17.5) | 2.5 (<0.1 - 3.9) | | |
| S | | 14.2 (11.1 - 17.3) | 201 (186 - 224) | 5.72 (5.58 - 7.12) | -70.4 (-193.0 - -46.2) | 2.9 (2.2 - 3.8) | 26.1 (3.1 - 509.0) | 60.7 (<0.1 - 99.6) |
| Site 400 | | | | | | | | |
| T | | 17.3 (13.8 - 19.6) | 937 (839 - 1124) | 6.95 (6.80 - 7.23) | -260.5 (-273.0 - -241.0) | 1.6 (<0.1 - 2.5) | | |
| DF | 307.70 (256.03 - 355.70) | 16.9 (16.0 - 18.0) | 342 (137 - 484) | 5.78 (5.19 - 6.72) | 31.9 (-157.0 - 75.0) | 2.6 (0.2 - 3.5) | | |
| Pooled | | | | | | | | |
| T | | 16.9 (11.4 - 21.0) | 917 (676 - 1148) | 7.07 (6.72 - 7.99) | -256.0 (-310.0 - -224.0) | 1.6 (<0.1 - 2.6) | | |
| DF | 70.10 (0.00 - 355.70) | 16.1 (6.0 - 18.1) | 482 (69 - 1734) | 6.51 (5.19 - 8.36) | -57.6 (-276.0 - 184.0) | 2.6 (<0.1 - 4.7) | | |
| S | | 14.2 (6.5 - 17.3) | 178 (129 - 224) | 5.94 (5.58 - 7.60) | -44.1 (-193.0 - 72.1) | 4.5 (2.2 - 9.1) | 10.9 (2.7 - 509.0) | 38.3 (<0.1 - 127.3) |
| BG | 124.97 (108.51 - 132.89) | 15.4 (12.3 - 19.3) | 90 (52 - 151) | 5.84 (5.41 - 8.20) | -49.0 (-131.2 - 27.2) | 5.3 (1.9 - 6.1) | | |