5-2023

Understanding and Quantifying Phosphorus Transport from Septic Systems to Lake Auburn

Evan Ma
ema@bates.edu

Understanding and Quantifying Phosphorus Transport from Septic Systems to Lake Auburn

A Thesis Presented to

The Faculty of the Environmental Studies Department

Bates College

In partial fulfillment of the requirements for the

Degree of Bachelor of Arts

By

Evan Ma

Lewiston, ME

April 21st, 2023

Acknowledgements

The support of my advisor, Holly Ewing, made this thesis possible. When I first worked as a summer researcher for Holly, I hadn't yet taken any natural science classes at Bates and found myself enamored with everything I was learning. Seeing and hearing how Holly thinks about scientific ideas inspires me to be a better thinker and gives me a true excitement to learn, and the classes I've taken with her have been the most transformative to me during my time at Bates. Throughout the process of researching and writing, Holly has not only encouraged me to try new things, but also encouraged me to be more confident in my abilities. I am forever grateful for her mentorship, generosity with her time (sorry I make you late to things so often), and commitment to helping me and my peers develop as learners; thank you, Holly! I am also incredibly thankful for the support of Emily Arsenault and Camille Parrish throughout my time at Bates and in Holly's lab. Being able to go out and sample with everyone was such a privilege! My peers Eli Boesch Dining, Dylan Larose, and Abby Quinn were also incredibly helpful through this entire process. Also, thank you to Lars Gundersen who so kindly provided data layers, and whose thesis helped me conceptualize mine.

I would also like to thank the staff of the City of Auburn's Planning & Permitting Department for welcoming me so kindly as their intern last summer, for giving me so many opportunities to learn while in the office, and for continuing to support my academic and professional endeavors. I'd also like to thank them for providing data for this thesis, which I helped compile as part of my internship. The Auburn Water District also kindly provided data during my time as an intern. Thank you to Eric Cousens, John Blais, and Katherine Cook for your mentorship. In particular, I'd like to especially thank Rosemary Mosher for her continued support. It has been a real privilege to get to know her and stay in touch. Although her guidance with GIS was instrumental to the creation of this thesis, the personal support she has offered me has had an even greater impact.

To my parents, Carla Mastromatteo-Ma and Chi Ma, I'd like to thank you for always supporting my endeavors in life, and always wanting to see me be happy. This thesis is possible because of your encouragement and your interest in listening to me talk endlessly about septic systems. I am so thankful to my wonderful friends, who also listen to me talk endlessly about this thesis. I can't wait to celebrate all that you have accomplished and all that you will accomplish! Thank you to everyone mentioned above for believing in me, it truly means the whole world.

Glossary

Adsorption: The adhesion of phosphate to mineral surfaces through ion exchange, or the binding of phosphate to iron and aluminum oxides/hydroxides through ligand exchange
Al: Aluminum, a metal commonly involved in phosphorus retention processes
Attenuation: Removal of a contaminant from solution by processes that retain it in the soil (in the case of nitrate, may be the atmosphere). Also referred to as removal, retention, or immobilization in the context of septic systems
Biomat: An area of high biological activity near where the drain field pipes meet the soil. The biomat slows the downward percolation of water in the drain field.
Ca: Calcium, an element commonly involved in phosphorus retention processes
Calcareous: Refers to soils that are rich in calcium, and are often basic or neutral in pH
Drain Field: An area of soil that is used to treat wastewater from a septic system. Wastewater flows from the septic tank to be dispersed across the drainfield
Effective Porosity: The fraction of pore space in a soil that is active in transmitting water– some pores are dead ends or inaccessible to water and therefore not included in the effective porosity
Export Coefficient: A quantification method that assumes that each person generates a certain amount of a contaminant per unit of time. For example, a phosphorus export coefficient may be 0.7 kg of phosphorus per person per year.
Fe: Iron, a metal commonly involved in phosphorus retention processes
Hydraulic Conductivity: The transmissivity rate of soil (how much water can move through in a given time). High hydraulic conductivity means that water can move quickly through the soil
Impervious Surface: A type of land cover that is impenetrable, such as driveways, sidewalks, or roads, where water is unable to percolate downward
N: Nitrogen, a common nutrient in human wastewater that is linked to lake eutrophication
NO_3^-: Nitrate, the most common and mobile form of nitrogen in wastewater plumes
Non-calcareous: Refers to soils that are not rich in calcium that often have high levels of Fe and Al. These soils are often acidic
P: Phosphorus, a common nutrient in human wastewater that is linked to lake eutrophication
PO_4^{3-}: Phosphate, the most common form of phosphorus in wastewater, which is biologically available and mobile in groundwater
Retardation Factor: The degree by which the movement of a contaminant is slower than the movement of groundwater. For example, if phosphorus moves 20 times slower than groundwater, then the retardation factor is 20. Abbreviated as R_f
Saturated Zone: The zone in the soil that exists below the water table, where pore spaces are completely filled with water.
Seasonal High Water Table: The highest point the saturated zone reaches in a given year. This is often located by the presence of "mottling," which is evidence of certain soil processes
Vadose Zone: The zone in the soil that exists above the water table. Here, the pore spaces between soil minerals are filled with both air and water, rather than being completely filled with water. Also referred to as the "unsaturated zone"

Table of Contents

List of Figures, Tables, and Maps

Abstract

Rural areas often use septic systems to treat household wastewater, which may pose a phosphorus (P) loading risk to nearby water bodies if systems fail or if the soil types are unsuitable for P retention. In the Lake Auburn watershed, septic systems may be a source of phosphorus loading to Lake Auburn, an unfiltered drinking water supply. Site evaluations from municipal permits reveal patterns of septic system locations and soil types in septic drain fields. Many septic drain fields have shallow depths to groundwater or a restrictive layer, which may lead to inadequate P retention in the soil. Areas with a high density of septic systems near the two largest inlets to the lake may be P loading hotspots. Near the Basin inlet, shallow soil depths and proximity to the lake suggest that failing systems may be a source of P loading, and that there may not be robust P removal in these drain fields. A high density cluster of systems near Townsend Brook is located on a sand and gravel aquifer, with coarse sands that have less capacity to retain P than finer-textured soils. The creation of a model that simulates 200 years (1900 to 2100) of septic system operation demonstrates that septic systems may create a legacy P issue, because P loading was estimated to increase even after new development ceased. The model shows that policy changes may be able to decrease the septic system P load, but that the impact of such changes on P loading estimates may not be substantial for decades. In Auburn, such policy changes also have land use implications that would increase watershed sources of P loading through additional deforestation, impervious surface, and septic systems from new development. Other studies on Lake Auburn demonstrate that land-based P loading from new development may be higher than the P load reductions from improved wastewater treatment estimated in this study, suggesting that any change to septic system policy that does not also restrict development where it has previously been restricted is likely to lead to a net increase in the cumulative P load.

Executive Summary

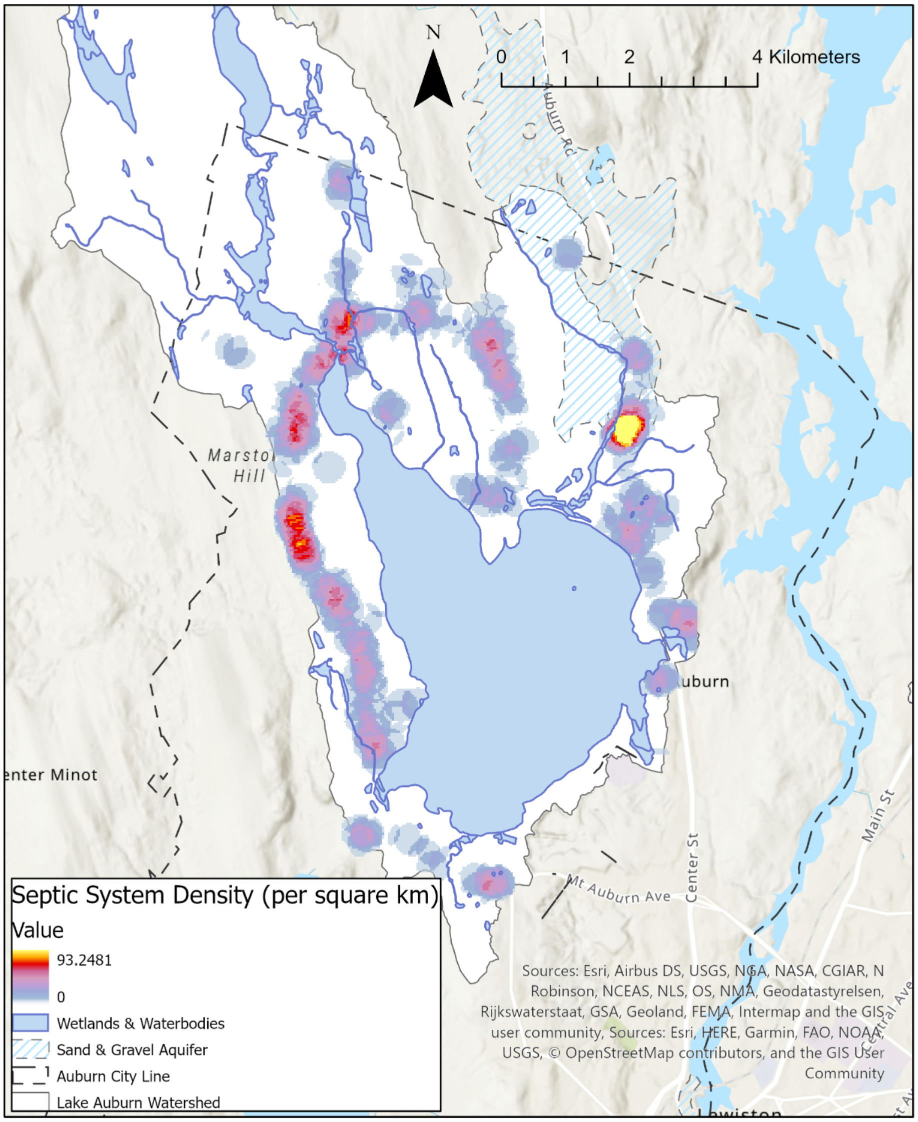
Septic systems are used to treat household wastewater in areas without sewer service and use native soil to remove pathogens and other contaminants from wastewater. Because phosphorus is present in wastewater, which can stimulate algal blooms (Schindler, 2006), improperly functioning septic systems are a potential threat to water quality that have recently become of interest for many rural watersheds (Withers et al., 2014). The transport of P from septic systems to lakes can either occur through the subsurface if the system is functioning properly, or through a more direct pathway when the system fails, which causes water to pond at the surface (Siegrist, 1987). However, systems that are "properly functioning" can contribute large amounts of P to groundwater if the soil types used in the septic drain field lead to poor P retention in sediments (Reide Corbett et al., 2002). Generally, septic systems are considered to contribute between 4 and 25% of the total phosphorus load in lake watersheds (Lusk et al., 2017).

In the Lake Auburn watershed, management of the lake is particularly crucial because it is an unfiltered drinking water supply that provides water to two Maine cities. One management action being considered by the City of Auburn is changing the septic system eligibility standards within the watershed to allow for the installation of mounded septic systems at the time of replacement (City of Auburn, 2022a). Changing the ordinance is intended to improve phosphorus retention in a subset of systems, but has the consequence of making new areas eligible for a septic system, which would cause land-use changes associated with development that load more phosphorus to the lake. Therefore, the objective of this study is to examine how the distribution of septic systems and soil characteristics across the Auburn portion of the Lake Auburn watershed may affect phosphorus loading from septic systems to the lake, and to use quantitative methods to understand the potential long-term impacts of septic systems. In order to do this, I collated municipal permits from the City of Auburn, which contain soil and design information about each septic system.

Areas of high septic system density near the Basin and Townsend Brook suggest that these areas may be loading hotspots (Map 0.1). The Basin has many systems located in close proximity to the lake shoreline and on soils with shallow depths to a restrictive layer, which may suggest that P retention is inadequate in these septic drain fields (Reide Corbett et al., 2002; Mechtensimer & Toor, 2017), and that both failing and well-functioning systems may contribute P to Lake Auburn in this area (Efroymson et al., 2007; Rakhimbekova et al., 2021). Septic systems in the Townsend Brook watershed are mostly located on sand and gravel deposits, where the coarse-textured soils in the septic drain fields are worse at retaining phosphorus than finely-textured soils (Carroll et al., 2005). This area has the highest density of septic systems in the Auburn portion of the watershed.

Failing systems may be a large potential source of P loading from Auburn's septic systems (Withers et al., 2011). Shallow depths to groundwater or a restrictive layer and coarse-textured soils are common soil characteristics throughout the watershed, and likely to lead to inadequate phosphorus retention in drain fields (Karathanasis et al., 2006).

Density of Septic Systems in the Auburn Portion of the Lake Auburn Watershed



Map 0.1: Density of septic systems in the Auburn portion of the Lake Auburn watershed. The highest density occurs near Townsend Brook, on a sand and gravel aquifer. Other high-density areas are near the Basin and North Auburn Road. Systems outside of the watershed within Auburn or within the watershed in other towns are not shown.

Because phosphorus moves slower than other wastewater constituents such as nitrate, chloride, and artificial sweeteners (Wilhelm et al., 1994; Robertson, 2021), and groundwater may move slowly (Todd, 1980), septic systems may create a legacy phosphorus issue where phosphorus from properly functioning systems may not reach the lake for decades (Roy et al., 2017). In order to investigate this potential effect, a model was created that spans 200 years (1900 to 2100) of septic system use in the Auburn portion of the Lake Auburn watershed. The model estimates that phosphorus loading may increase even in the absence of new development between 2023 and 2100 (Figure 0.1). Incorporating the proposed septic ordinance change, the model estimates little immediate effect of the change even in the absence of continued development. For example, there is estimated to be less than a 10% phosphorus loading reduction two decades after the proposed change is simulated to occur. Decreases in phosphorus loading occur in the model as older systems on shallow soils are replaced with mounded systems with greater phosphorus retention.

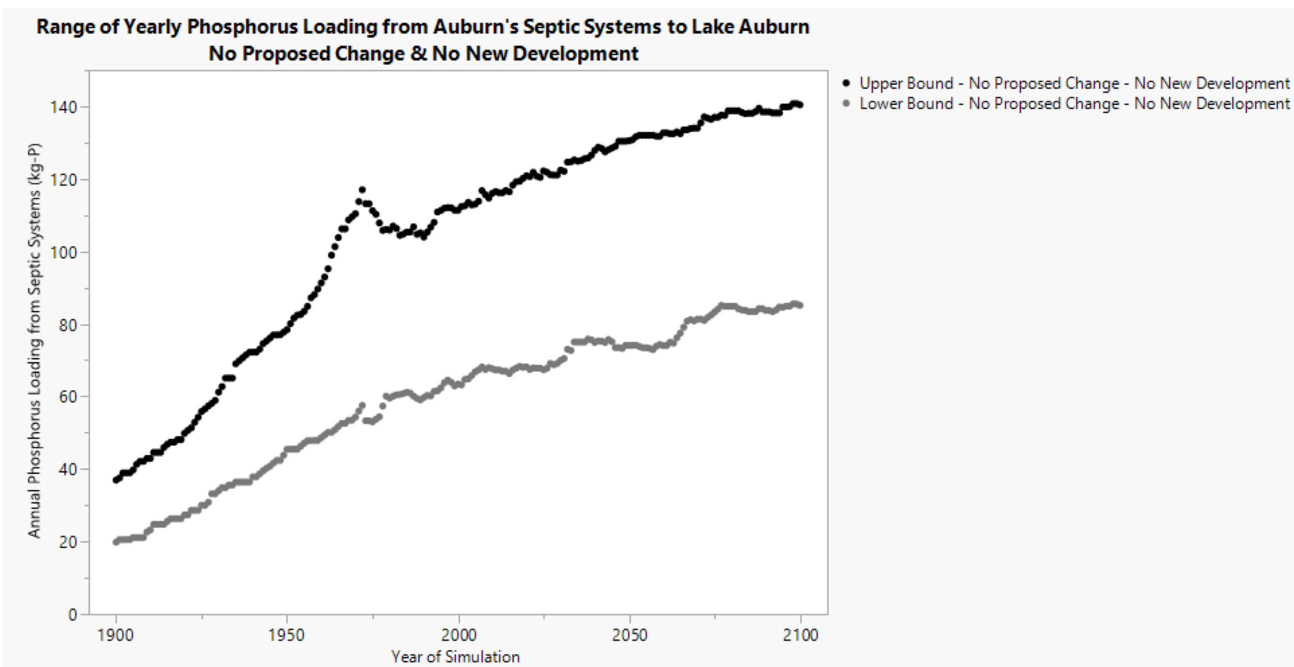


Figure 0.1: Upper and lower bounds of temporal model results, assuming no new development and no proposed ordinance change. The lower curve reflects transport to the lake assuming slow transport of phosphorus in groundwater, while higher phosphorus loading occurs if the soils under the septic systems transport phosphorus more quickly. Phosphorus loading is estimated to increase from 2024-2100 even though there is no new development, demonstrating that septic systems may pose a legacy issue, where phosphorus may take a long time to reach the lake.

Depending on how quickly phosphorus transport is assumed to occur, the model yields different results, with quick phosphorus transport leading to larger phosphorus load reductions (28.4 kg-P/year) and slow phosphorus transport leading to smaller reductions (5.8 kg-P/year) by the year 2100, assuming no new development occurs (Figure 0.2). However the proposed ordinance change would allow for more areas within the watershed to be developed by relaxing septic standards. As a result, phosphorus loading associated with continued development from loss of forested area, increased impervious surface, and more septic systems (Withers & Jarvie, 2008) may nullify any phosphorus loading reduction associated with improved wastewater treatment. For example, CEI (2010) considered a relaxation of the septic standards in one of their analyses and estimated that phosphorus loading would increase by 56.7 kg-P/year from land use changes alone as a result of changing the septic standard; when additional phosphorus loading from new septic systems was considered, this number increased to 158.3 kg-P/year (Figure 0.2).

This evidence from CEI (2010) demonstrates that the land use consequences of any proposed change in the septic ordinance that does not also continue to restrict development where it has been previously restricted would nullify the phosphorus loading reductions that were estimated in this study. Such land use changes would instead lead to a net increase in cumulative phosphorus loading in spite of improved wastewater treatment in certain septic systems (Figure 0.2).

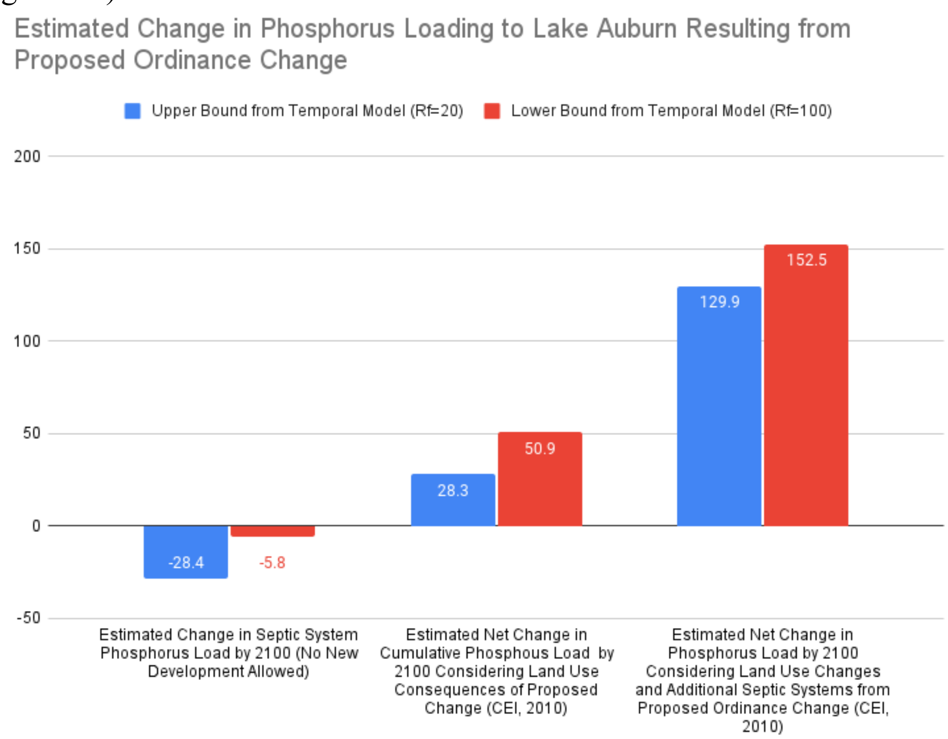


Figure 0.2: Estimated net change in cumulative P loading to Lake Auburn by 2100 considering the proposed ordinance change. Results from CEI (2010) are added to the lower and upper bounds from the temporal model, which demonstrate how land use changes allowed by proposed policies may cause a net P loading increase. The proposed change may reduce P loading only in the absence of new development.

Chapter I: Introduction

Septic systems utilize native soils to treat household wastewater in rural areas where municipal sewer lines do not exist. This form of on-site wastewater treatment is common in the United States, and is generally considered to be effective at removing pathogens from wastewater if the system is sited and installed properly (Mallin, 2013). However, septic systems can be a potential threat to water quality and have recently become the subject of attention for many rural watersheds because they may contribute nutrients such as nitrogen (N) and phosphorus (P) to lakes (Withers & Jarvie, 2008; Withers et al., 2014). This can be a major concern because increased nutrients often cause cyanobacterial blooms, which can lead to low dissolved oxygen, fish kills, and cyanotoxins (Pacheco et al., 2021; Carmichael & Boyer, 2016). In communities that have public uses for lakes, such as for drinking water or recreation, eutrophication from watershed sources such as septic systems, agriculture, or various land use changes is particularly alarming (Watson et al., 2016; Kong et al., 2022). Because phosphorus in particular is often considered to be the most limiting nutrient to lake ecosystems in northern latitudes (Kalff, 2002), preventing phosphorus loading from watershed sources is often regarded as necessary to prevent lake eutrophication (Schindler, 2006); septic systems may be responsible for 4 to 25% of the total phosphorus loading for surface water bodies (Lusk et al., 2017).

Various characteristics of a watershed dictate how septic systems may impact water quality. For example, soil characteristics of the drain field such as texture and depth to groundwater can impact how well the system retains nutrients (Karathanasis et al., 2006). Furthermore, their distribution across the landscape influences how much phosphorus may be loaded to nearby water bodies, with areas of high septic system density often contributing more P (Mallin, 2013; Iverson et al., 2018). Because phosphorus is involved in a variety of soil processes that delay its movement in groundwater (McBride 1984; Bedient et al., 1994), and groundwater may be slow to begin with (Todd, 1980), septic systems pose a legacy issue, where phosphorus does not reach a water body until years after the operation begins (Roy et al., 2017). However, with the oldest studied septic systems being less than fifty years old (Robertson et al., 2019), the long-term fate of phosphorus from septic systems is greatly unknown, though it is believed and assumed that P continues moving toward water bodies until reaching a shoreline (Robertson & Harman, 1999; Roy et al., 2017). Generally, larger setback distances are considered to allow more P to be retained in the soil (Lusk et al., 2017).

Considering the legacy impact of septic systems and the role of soils in nutrient attenuation, there are various ways to quantify nutrient loading from septic systems to lakes. These approaches include hydrologic models that also model nutrient loading from an entire watershed (Jeong et al., 2011; Geza et al., 2010; Appendix B), and export coefficient methods, which assume that each person exports a certain amount of N or P per year. Given the importance of time in assessing the impact of septic systems, studies that examine P loading from septic systems on long time scales are becoming increasingly common, and often demonstrate legacy phosphorus loading from septic systems (Oldfield et al., 2020a; Schellenger & Hellweger, 2019). Although these types of models have uncertainties like any other model, they can be an effective tool to quantitatively and visually demonstrate the geochemical and hydrologic ideas behind phosphorus transport from septic systems to water bodies.

Variation in soil types that lead to differences in wastewater treatment across a watershed may be especially critical to lake management in sensitive watersheds. For example, Lake Auburn, located in Auburn, Maine, is important to thousands of people because of its use as a public drinking water supply for the cities of Lewiston and Auburn, as well as certain areas of Poland, Maine. Beyond this, the Auburn Water District/Lewiston Water Division has an EPA filtration waiver due to the lake's great historic water quality, allowing them to provide unfiltered drinking water to over 59,000 people (CDM Smith, 2013). Management of the lake and its watershed have become topics of public debate because of water quality decline observed over the years. For example, cyanobacterial blooms followed by fish kills in 2012 and 2018 caused public concern over the health of the lake, leading to an alum treatment in 2019 that led to water quality improvement (CDM Smith, 2013). Beyond the importance of supplying clean water to the citizens of Lewiston, Auburn, and Poland, declining water quality is also a major concern because it places the Auburn Water District's EPA filtration waiver at risk, which would require the construction of a filtration plant that would cost 35-45 million dollars and additional yearly maintenance costs (FB Environmental, 2021). A 2021 report on the lake by FB Environmental Associates found that the construction of a filtration plant may be less economically feasible than management actions to maintain stellar water quality, and would have the consequence of increasing water bills for citizens of both cities, which would impose a social cost on the people of Lewiston and Auburn (FB Environmental, 2021).

A potential management action being discussed by the City of Auburn entails changing eligibility standards for a septic system within the watershed (City of Auburn, 2022b). The proposed change would allow for mounded septic systems, which may improve wastewater treatment in a subset of systems located on shallow soils (Bouma et al., 1975; City of Auburn, 2022c). However, it would also allow for more residential development in the watershed by relaxing certain requirements, which could lead to more deforestation, impervious surface, cleared land, and additional septic systems that would increase phosphorus loading to Lake Auburn (Easton et al., 2007; CEI, 2010). In the context of these discussions, the objective of this study is to examine the distribution of septic systems and soil characteristics throughout the Lake Auburn watershed, and use quantitative methods to understand how these characteristics may impact phosphorus loading from septic systems to Lake Auburn. It will also investigate and attempt to quantify the legacy effect of septic systems considering proposed management actions.

Chapter II: Background

2.1 Phosphorus and Septic Systems

2.1.1 Pathways of Nutrient Transport

Nutrients enter lakes through different routes of water flow which are often controlled by precipitation. For example, overland flow is direct surface runoff that enters tributaries of a lake or the lake itself. Though this water contains dissolved nutrients, it also transports nutrients bound in sediment through erosion, which makes it one of the most salient contributors to eutrophication (Lin et al., 2015). The amount of erosion that occurs depends on the soil type, land cover, and land uses (Sharma et al., 2011); agricultural and urban land uses load the largest amounts of phosphorus because of their high erosion potential and the limited ability for water to infiltrate through the soil (Daloğlu et al., 2012; Valtanen et al., 2014). Impervious surfaces, such as roads and driveways, load large amounts of phosphorus into lakes because no infiltration is able to occur (Withers & Jarvie, 2008). Conversely, forested areas load considerably less phosphorus than cleared areas (Dillon & Kirchner, 1975) because tree roots prevent soil erosion and vegetative cover limits the speed of water movement, thus improving soil infiltration (Kalff, 2002; Sun et al., 2018). Land use changes that deforest in favor of impervious or cleared areas are noted to lead to eutrophication due to the large amounts of phosphorus they contribute through overland flow (Sorrano et al., 1996).

Water also carries nutrients into lakes through subsurface flow, which consists of groundwater discharge into water bodies (Meinikmann et al., 2015) and near-surface runoff (Kalff, 2002). Precipitation or other sources of water may percolate through soil instead of traveling through surface or near-surface pathways, which can allow for significant treatment of the water through adsorption processes and plant uptake before it enters a water body. Both overland flow and subsurface flow contribute water and nutrients to stream flow. Within streams, there may be additional attenuation, or removal, of nutrients in sediments or by plants, which affects the mass of nutrients that reaches a lake (Darracq & Destouni, 2007). However, sources of pollution, such as septic systems that contribute wastewater directly into the soil, can cause groundwater pollution that can be a source of nutrient loading to water bodies through the subsurface (Withers et al., 2014; Fetter, 1993).

2.1.2 Septic System Design and Functioning

Septic systems function by utilizing soils to purify wastewater, which is also called effluent. While there are many different types of septic systems, those used in the City of Auburn are traditional septic systems which involve a septic tank and a drain field, as is allowed by city ordinance (City of Maine Ordinance Chapter 60, 2009). A reliance on the soil to treat septic effluent means that a location must have appropriate soils in order to have adequate treatment. In particular, in order to treat pathogens, the soil must not drain too fast, so as to allow water to percolate through without being treated (Brady & Weil, 2002); if it is too slow, water cannot percolate through the soil, leading to water pooling at the surface (Mallin, 2013). This is known as hydraulic failure of the system. When a septic system fails, the system must be replaced; septic systems typically have a lifespan around between 20-30 years before they fail due to buildups of organic matter at the drain field surface or rising water tables (Siegrist, 1987), though failure age varies depending on soil characteristics (Hill & Frink, 1980). Failing systems are often regarded as direct inputs to surface water bodies as ponding water may travel through a surface or near-surface pathway (Beal et al., 2005). This mechanism leads to the identification of the two major flow pathways for wastewater movement toward water bodies: a direct pathway from failing systems and a subsurface pathway.

Municipalities and states typically govern which types of soils are suitable for septic systems (State of Maine, 2014). For example, the City of Auburn requires 36 inches of vertical separation between the soil surface and the most “limiting factor,” which is where a site evaluator finds the seasonal high water table, bedrock, or other restrictive layer (City of Auburn Ordinance Chapter 60, 2009). This is to ensure that there is an adequate amount of soil to eliminate pathogens beneath the system. Many soils are unable to meet the requirements put in place by municipalities, which prevents the construction of new homes in certain areas (Brady & Weil, 2002). One way to make these areas “suitable” for a septic system includes the use of alternative septic systems that import additional soil in order to reach the depth to limiting factor requirement (Bouma et al., 1975). These are known as mounded septic systems, and consist of mounding imported soil on top of native soil in order to reach a certain cumulative soil depth (Mallin, 2013). The drain field exists within the mound, and the imported soil is often well suited for wastewater treatment (Bouma et al., 1975), which is intended to also improve nutrient attenuation. Using mounded systems to achieve greater soil depth may promote more complete P retention if the system is located on shallow soils (Reide Corbett et al., 2002).

2.1.3 The Biomat and Septic System Failure

One of the mechanisms that allows for proper septic system functioning involves the formation of a biomat near the infiltrative surface of the system that is rich in organic matter. The biomat tends to form and reach steady-state within a year of effluent application (Beal et al., 2005). Because septic effluent is nutrient rich, the microbial communities that form here differ from other soil microbial communities (Tomaras et al., 2009), and create a biomat with lower hydraulic conductivity than the surrounding soil, leading to lower infiltration rates (Siegrist, 1987). Because of this, the biomat allows for even spreading of the effluent across the drain field area. It also is responsible for maintaining unsaturated conditions beneath the drain field (Beach et al., 2005), which allows for better wastewater treatment. However, this same mechanism can be responsible for hydraulic failure of the system if the loading rate of effluent is greater than the infiltration rate of the biomat (Winstanley & Fowler, 2013), causing water to pond at the surface.

The robustness of the reduction of hydraulic conductivity in the biomat is primarily related to the loading rate of effluent, with higher loading leading to lower hydraulic conductivity (Siegrist, 1987; Beach et al., 2005; Bumgarner & McCray, 2007). Though it is intuitive that some nutrients are removed as the biomat forms, the role of the biomat in spreading effluent across the drain field and maintaining unsaturated conditions is considered its most important feature (Beal et al., 2005). In coarse-grained soils, formation of the biomat may be incomplete due to a lack of organic matter in coarse or sandy soils, as noted by field studies (Mechtensimer & Toor, 2016), which indicates that effluent may be less evenly distributed across drain fields with coarse-textured soils.

The buildup of bacterial plaque in the biomat may lead to hydraulic failure of the system when wastewater cannot percolate through the biomat as fast as it is applied (Siegrist, 1987). Failure may also occur due to low permeability soils, high water table, or lack of maintenance (Gunady et al., 2015). Therefore, finer-textured soils such as clay soils typically are more susceptible to failure, leading to surface ponding of effluent which is often contributed to water bodies through a surface or near-surface pathway (Beal et al., 2005; Withers et al., 2011). Septic systems are particularly susceptible to failure during large precipitation events, which raise the water table.

Failing septic systems may be a significant source of phosphorus loading to lakes because wastewater that is allowed to percolate through soils may take years to actually discharge into water bodies (Robertson & Harman, 1999), and P from a surface pathway does not receive treatment from the soil before entering water bodies. Studies using an artificial sweetener (acesulfame) as a tracer of wastewater in streams find that the percentage of wastewater from septic systems that enters a stream is low (Spoelstra et al., 2020) and varies greatly depending on weather and discharge conditions (Oldfield et al., 2020a). Therefore, failing septic systems may play an unexpected role in nutrient loading, as weather strongly influences how much effluent travels through this pathway. However, recent evidence finding that acesulfame may biodegrade during wastewater treatment (Castronovo et al., 2017; Van Stempvoort et al., 2020) indicates that more wastewater may reach water bodies through the subsurface than tracer studies suggest.

2.1.6 Phosphorus Attenuation in Drain Fields

In septic system drain fields, phosphorus (P) can be retained through various physical and chemical processes (Figure 2.1), and is most commonly present in the form of orthophosphates (Lusk et al., 2017), which are dissolved and biologically available. The adsorption of phosphorus refers to the adhesion of P to mineral surfaces through ion/ligand exchange processes that often involve clay particles and Fe/Al oxides/hydroxides (McBride, 1994; Singer & Munns, 2006; Sposito, 2008). Phosphorus can also be retained through mineral precipitation, where phosphorus binds with Fe, Al, or Ca to form a variety of solid secondary minerals such as variscite, strengite, or hydroxyapatite (Lusk et al., 2017; Robertson, 2021). While adsorption is reversible through desorption (Barrow, 1983), precipitation is considered to be a more permanent form of immobilization in the context of septic system drain fields (Robertson, 2008). Additional background on phosphorus immobilization and transport can be found in Appendix A.

A number of field studies aim to determine the ability of septic systems to attenuate P by using *in-situ* data to evaluate how soil type and soil depth affect adsorption and precipitation. Because the immobilization of P is directly related to contact between P in wastewater and soil mineral surfaces (Tan, 2011), it is intuitive that more soil correlates to more treatment. For example, Karathanasis et al. (2006) investigated the efficiency of septic systems with soil depths up to 60 cm, and found that P removal increased with increasing depth for all soil types. They also found that clay soils removed more P than sandy soils; this relationship was only significant when septic effluent was able to percolate through 60 cm of soil. Drain fields retain P especially well if effluent is able to percolate through 1 m of unsaturated soil (Reide Corbett et al., 2002;

Mechtensimer & Toor, 2017), with retention observed to lessen by a factor of ten below 1 m (Baer et al., 2019). Furthermore, there appears to be no correlation between water table depth and total P removal in drain fields with vadose zone depths between 1 m and 5 m (Robertson et al., 2019), which suggests that the first meter of unsaturated soil is particularly critical to wastewater treatment.

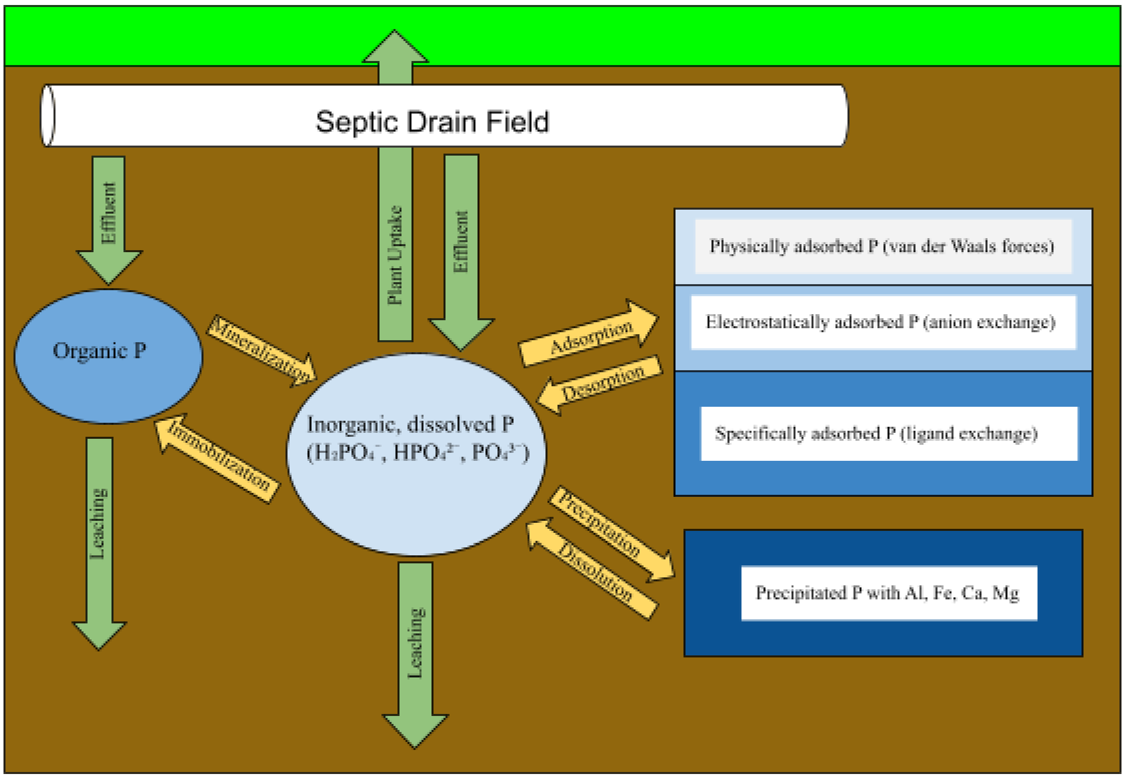


Figure 2.1: A conceptual diagram showing the most prominent movements and transformations of phosphorus in septic system drain fields. Green arrows represent the movement of phosphorus, while yellow arrows represent transformations of phosphorus. Deepening shades of blue signify more tightly immobilized P. The size of arrows or other shapes do not correlate to sizes of P stores.

Gill et al. (2009) investigated the unsaturated zone of septic drain fields with a soil depth up to 1 m and concluded that, although a greater soil depth always contributes to greater P removal, the mineralogy of the soil tends to take on a larger importance. Subsoil layers containing high percentages of clay or the presence of calcium or Al/Fe oxides were noted to immobilize P especially well, regardless of depth. Their findings imply that coarse-grained soil types immobilize less P than clay soils (Gill et al., 2009). Other studies that assess the ability of soils to retain P after effluent application support the idea that sandy soil types adsorb less P than finer-textured soils due to their lesser surface area and limited ion exchange capacity (Carroll et al., 2005; Zanini et al., 1998; Mechtensimer & Toor, 2016). For example, Wilhelm et al. (1994) studied a septic drain field that was 95% sand and found that only 50% of P had been attenuated in the vadose zone before discharging into the groundwater.

Retention of P in septic system drain fields also differs depending on whether the soils are calcareous (rich in calcium) or non-calcareous. These types of soils differ greatly in pH and mineralogy, with calcareous soils generally being basic and having low amounts of Al and Fe-oxides, while non-calcareous soils are acidic and contain large amounts of Al and Fe. Because calcareous soils may lack metal cations and metal oxides that precipitate with P, they are more susceptible to P leaching in septic drain fields (Zanini et al., 1998; Robertson, 2003). A review of twenty-four septic systems by Robertson et al. (2019) included an investigation of P attenuation in the vadose zone and found that drain fields in non-calcareous soils retained 90% of P, while those in calcareous soils retained 66%. Although non-calcareous soils often outperform calcareous soils because of the availability of Fe and Al, the stability of Al-P mineral compounds, and low pH (Robertson, 2003; Robertson, 2012; Eveborn et al., 2012), calcareous soils still exhibit a large capability to retain P due to the formation of Ca-P precipitates (Robertson et al., 1998; Ige et al., 2005).

This is critical because studies on the vadose zone beneath septic drain fields find that mineral precipitation is the main mechanism responsible for P removal (Wilhelm et al., 1994; Robertson, 2008). Though adsorption is also a critical process, evidence suggests that increases in solid P are larger than increases in desorbable P after septic system operation (Robertson, 2012; Baer et al., 2019), demonstrating that more precipitation occurs than adsorption. This is confirmed by studies that found thick solid coatings of Fe-P, Al-P, and Ca-P compounds on individual grains of soil between 5 - 30 cm of the disposal pipes (Zanini et al., 1998; Robertson et al., 2002; Baer et al., 2019; Robertson et al., 2019). The largest amount of mineral precipitation occurs in the area within 50 cm of the pipes because the rapid oxidation of wastewater in this area facilitates precipitation (Robertson, 2012). Evidence that septic drain fields effectively remove phosphorus regardless of age and without significant decline also supports the idea that precipitation controls P retention in the vadose zone (Robertson et al., 1998). This is because, in drain fields used up to 44 years, no maximum P sorption capacity is reached (Robertson et al., 2019). It is important to note that many studies on the vadose zone beneath septic systems use systems that were confirmed to be installed properly (Karathanasis et al., 2006; Gill et al., 2009; Mechtensimer & Toor, 2016). Therefore, systems that were not properly installed may attenuate P less efficiently.

2.1.7 Phosphorus Transport to Water Bodies

Once wastewater leaves the vadose zone, it enters the saturated zone where it begins moving toward nearby water bodies. While there may be robust immobilization of P in the vadose zone (Mechtensimer & Toor, 2016), there is far less permanent P immobilization in the saturated zone because of a lack of mineral precipitation (Robertson, 1995; Robertson, 2008). This is because of the mobilization of iron-phosphorus compounds under anoxic or reducing conditions (Ptacek, 1998) such as those found in the saturated zone (Wilhelm, 1994). Therefore, reversible adsorption reactions mainly delay the transport of P through groundwater rather than immobilize it (Robertson & Harman, 1999).

Though wastewater constituents such as nitrate, sodium, or artificial sweeteners can move quickly in groundwater and discharge to water bodies (Spoelstra et al., 2017; Spoelstra et al., 2020), the movement of P in groundwater is delayed and it may take far longer for P from septic systems to discharge into lakes (Wilhelm et al., 1994). Beneath septic systems, the presence of P plumes traveling at rates slower than groundwater are well studied and documented (Harman et al., 1996; Robertson, 2008; Robertson et al., 2019; Rakhimbekova et al., 2021). The way P transport is delayed in the saturated zone is often quantified by comparing the groundwater velocity with the velocity of the plume. This is known as the retardation factor (R_f), which is the ratio of the groundwater velocity and the velocity of the contaminant in the subsurface (Bedient et al., 1994). Studies of septic system P plumes in Ontario, Canada, which are noted to have sandy and calcareous soil types, find values of R_f between 20-100 (Robertson et al., 1998), meaning phosphorus moves 20-100 times slower than groundwater in these soils.

R_f is specific to each individual P plume, and tends to be affected by the type of contaminant and soil qualities that influence adsorption (Ma et al., 2021). The retardation of phosphorus plumes occurs over long time periods such that phosphorus discharge into surface water bodies may not occur for many years after the septic system operation begins (Roy et al., 2017). Many studies on P plumes beneath septic drain fields use systems that were installed within fifty years of the research (Robertson et al., 2019; Rakhimbekova et al., 2021), meaning most plumes had not yet discharged into nearby lakes, though some studies project that P from septic systems over 100 meters away from the shoreline may begin discharging in the near future (Rakhimbekova et al., 2021). In calcareous, sandy soils, the velocity of P transport in groundwater is estimated between 1-2 meters per year with the largest plumes extending between 90 to 100 meters downgradient (Robertson et al., 2019). Non-calcareous soils are less susceptible to this degree of P leaching (Zanini et al., 1998; Robertson, 2003).

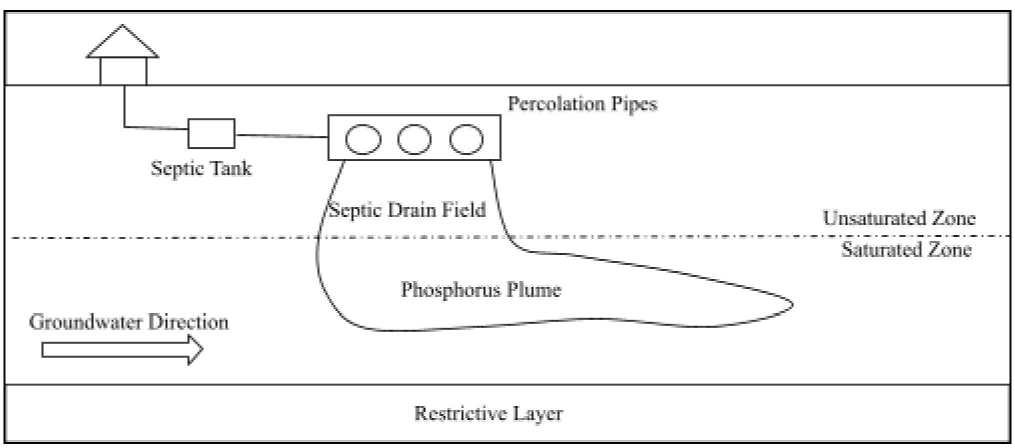


Figure 2.2: Simplified diagram of a phosphorus plume beneath a well-functioning septic system. P plumes form in groundwater after percolating through the unsaturated zone, which then move slowly toward water bodies. P is retained in the drain field, while less immobilization occurs in the plume. System failure is not portrayed. Adapted from Robertson et al. (2019).

The most well-studied P plume exists in Cape Cod, Massachusetts, where sewage disposal pits from the Massachusetts Military Reservation generated a massive P plume that traveled 520 m downgradient to discharge into Ashumet Pond (LeBlanc, 1984). However, because the pond only intercepted a portion of the plume, the furthest extent of the phosphorus plume was found to travel up to 760 m downgradient toward the Atlantic Ocean (McCobb et al., 2003). The sewage disposal site was in use from 1936 to 1995, with the plume discharging at concentrations of 3 mg/L at the lakeshore (McCobb et al., 2003); the plume was traced using electrical conductivity (EC) and boron (LeBlanc, 1984), which are common wastewater tracers (Robertson, 2021). A 3-dimensional reactive transport model was developed by Parkhurst et al. (2003) for this plume in order to investigate the extent of P loading to Ashumet Pond. Model results suggest that P did not begin discharging into the pond until 1965, about thirty years after the sewage disposal operation began, and P loading was simulated to persist for decades after decommissioning. At its peak, the simulation suggests that the maximum P load was 1,000 kg-P/yr in 1993, though the authors concede this may be an overestimate when compared to detailed *in-situ* groundwater data collected during this time (Parkhurst et al. 2003).

Though this example of a large P plume discharging to a local water body is far different from a typical septic system, because far more sewage was disposed than that of a single household, it provides evidence that distant sewage disposal systems can discharge P into lakes at high concentrations. Furthermore, it exemplifies the delay of P transport in the subsurface and demonstrates that there may be a legacy effect of sewage disposal systems, even years after they have been decommissioned. Few studies have investigated the legacy effect of septic systems after decommissioning, when septic systems stop loading wastewater to soils. Robertson & Harman (1999) studied two wastewater plumes after decommissioning and found that conservative constituents of wastewater, such as Na^+ and NO_3^-, continue migrating toward water bodies at the same velocity of groundwater. Concentrations of these ions returned to concentrations typically observed in groundwater. Phosphate, however, maintained high concentrations in the plume and continued moving downgradient toward surface water. Though the authors found that 85% of P from effluent had been retained in the vadose zone, the persistence of the plume demonstrates that the retardation of P in the saturated zone poses a legacy P issue (Robertson & Harman, 1999), which may affect water quality if these plumes reach the shoreline.

Another study on decommissioned septic systems illustrates that plumes from decommissioned systems consist of P that had previously reached the saturated zone while it was operational, rather than P retained in the vadose zone that became mobile again (Roy et al., 2017). This demonstrates the robust and near-permanent retention of P in the vadose zone and supports the idea that P in the saturated zone remains ultimately mobile. In Wasaga Beach, Canada, P plumes were approximated to travel 150 m for 115 years of operation (Roy et al., 2017). In the context of areas developed less than a century ago, this reveals that legacy P has likely not yet discharged into lakes, but may continue moving downgradient until reaching the shoreline. This is especially problematic considering that policy changes aiming to prevent P loading were implemented in the late 20th century. In Maine, for example, cesspools were often used instead of traditional septic systems, and wastewater often had elevated levels of P due to the use of phosphates in laundry detergents until they were banned in 1973 (Litke, 1999). It is possible for P loading from septic systems to increase even in the absence of continued development due to the delay of P (Schellenger & Hellweger, 2019), because new plumes continue to reach the shoreline. This implies that policy changes aimed to reduce P loading from septic systems may not actually decrease P loading until decades after they are implemented.

2.2 Relevant Modeling Approaches

2.2.1 Modeling approaches used for Lake Auburn

Previous studies on Lake Auburn have attempted to quantify phosphorus loading from septic systems. CEI (2010) uses an export coefficient approach that only considers septic systems within 300 feet (91.44 m) of the lake or tributary, with those within 50 feet (15.24 m) considered a direct input, receiving no P attenuation. An export coefficient assumes that each person produces a certain amount of phosphorus through a septic system per year. In this approach, CEI (2010) assumes that each individual exports 0.7 kg-P person^{-1}year^{-1}, that there are 2.6 individuals per household, and uses a soil retention factor of 0.8, which means that 80% of phosphorus is attenuated by the soil. After finding 287 systems within 300 ft of a shoreline and 41 systems within 50ft, they estimated P loading from septic systems to be 120.7 kg-P year^{-1} (133 kg-P year^{-1} after calibration by comparing results with in-lake phosphorus concentrations). However, FB Environmental (2021), a consultant group who performed a similar analysis, found that CEI accidentally used a soil retention factor of 0.2 instead of 0.8, which may have caused an overestimation in their analysis.

FB Environmental (2021) used a similar approach, using the same export coefficient and (corrected) attenuation factor, but instead using 2.325 people per household. This number was obtained through 2020 U.S. Census data. They considered systems within 300 ft (91.44 m) of the lake or a tributary, with those near tributaries undergoing additional P attenuation from the stream (FB Environmental, 2021). Only considering systems within a specified distance from the lake is a manner of accounting for the delay of P transport in groundwater. Since distant systems may have P plumes that do not reach a shoreline, they are not currently loading P to Lake Auburn even though they are operational. The use of a soil retention factor generalizes the ability of soil to retain P across the watershed. This is useful in the context of Lake Auburn because there are no existing field studies that evaluate septic system efficiency in this watershed, and there is limited accessible soil data aside from soil survey data. In the context of proposed changes to the City of Auburn's ordinance regarding septic systems within the Lake Auburn watershed (City of Auburn, 2022b) and a zoning change to reduce the allowed residential density (City of Auburn, 2022a), FB Environmental performed another analysis in 2022 that incorporates the proposed changes; they did this by increasing the soil retention factor from 0.8 to 0.9 to reflect better wastewater treatment (FB Environmental, 2022). This led them to predict that the proposed changes would marginally lower the total phosphorus load into Lake Auburn after considering the additional land use changes that may occur as a result of the ordinance change, though these results may be hard to contextualize due to changes in modeling assumptions between model runs (CEI, 2022).

2.2.2 Using Field Studies (Oldfield et al., 2020a)

Some studies use export coefficients to estimate P loading from septic systems in large catchments where fine-scale data may not be available that spans the entire region. For example, Oldfield et al. (2020a) attempts to quantify P loading from the Canadian portion of the Lake Erie watershed, choosing an export coefficient approach because hydrologically complex models require input data that is unavailable for such a large area. Therefore, using an export coefficient generalizes most of the P removal and transport processes that dictate P loading from septic systems to water bodies in order to make the study possible at this scale. Export coefficients also assume steady state– this means that each septic system contributes P to groundwater at a constant concentration regardless of age; septic systems are noted to reach steady state once sorption sites are taken up and the system has generally reached various equilibria (Robertson, 1995). The authors determined the location of each septic system through manual placement methods and through development of a geospatial model that places a septic system on each parcel of land, finding discrepancies between setback distances for each approach (Oldfield et al., 2020a).

Oldfield et al. (2020a) calculate their export coefficient using literature values of water use and wastewater composition, assuming that each person uses 221.8 L/day of water, and that there is a total P concentration of 10 mg/L in septic tank effluent. This gives an export coefficient of 0.81 kg-P $person^{-1}year^{-1}$. Because the soils in the Canadian Lake Erie Basin area are calcareous, the authors use a 66% attenuation factor based on evidence from Robertson et al. (2019), who find a mean value of 66% P attenuation by septic drain fields in calcareous soils in

Canada. The authors consider the delay of P transport by drawing on studies that used acesulfame as a tracer of wastewater, which suggest that only a portion of wastewater from septic systems is currently reaching tributaries (Spoelstra et al., 2020). Based on a study conducted in the same watershed (Oldfield et al., 2020b), they assume that only 24% of wastewater is currently reaching the streams (Oldfield et al., 2020a). They use these factors to determine that septic systems contribute between 1.7 - 5% of the total P load to Lake Erie if no systems are failing.

This export coefficient approach does not account for variability in soil characteristics across the landscape that are integral to P treatment. Instead, the authors use an attenuation factor determined from well-functioning septic systems on calcareous soils with over 1 m of unsaturated soil (Robertson et al., 2019), and use it as a lower bound of P loading. They determine the upper bound by removing the attenuation factor, assuming no P retention. Since soil characteristics are unknown, presenting an upper and lower bound demonstrates the extent of P loading in the case that each system removes P efficiently or not at all. The authors also consider the retardation of wastewater transport by assuming that 24% of effluent reaches tributaries. This accounts for the fact that many P plumes have not yet reached shorelines, and that some P is lost to a deeper aquifer system (Oldfield et al., 2020a). Applying this factor to the export coefficient method is useful because it assumes 24% of effluent from each system is discharging into a water body. Even though this does not reflect how P plumes discharge, since some plumes are discharging effluent while others are not, it works well to factor in each system under the assumption that it is unknown whether or not a specific septic system is discharging into a water body.

2.2.3 SANICOSE (Gill & Mockler, 2016)

A modified export coefficient model was developed by Gill & Mockler (2016) for application throughout Ireland, using 0.68 kg-P person^{-1}year^{-1} and 2.8 people per home. They consider three pathways by which P may enter a surface water body: a surface pathway from ponding of effluent, a near-surface pathway, and a groundwater pathway. The amount of effluent attributed to each pathway was determined by assigning each septic system a category based on a groundwater susceptibility map, soil permeability, depth to bedrock, and the distance to surface water. For example, low permeability soils were considered to load differing amounts of P through the surface pathway based on their distance to the water body. Similarly, septic systems with soils more susceptible to groundwater contamination had a higher amount of P contributed through groundwater. The attenuation of P was simulated by assigning a different attenuation factor for each type of soil (Table 2.1). Each factor was determined based on field studies, and specifically refers to the amount of P attenuated through 1 m of unsaturated soil (Gill & Mockler, 2016). Septic systems with shallow groundwater or bedrock had a separate attenuation factor (Table 2.1). The transport of P in the subsurface was simulated using rainfall data and groundwater recharge data from national databases as well as transport coefficients based on the soil permeability. These transport coefficients are similar to the approach by Oldfield et al. (2020b), who used evidence suggesting that only 24% of septic effluent reaches streams.

Application of the SANICOSE model in various watersheds in Ireland reveals that the relative contribution of P from septic systems is highest in small watersheds, and decreases as watershed size increases (Gill & Mockler, 2016). This model excels because it accounts for variations in soil type, groundwater depth, and distance from surface water bodies, as well as P loading from surface and subsurface pathways. While it uses coefficients based on field studies rather than equations that exactly resemble the hydrologic and geochemical phenomena that occur beneath septic systems, the authors posit that the semi-qualitative/quantitative approach is easy to understand and useful for predicting P loading without getting lost in the various parameters required for process-based hydrologic models that may be hard to define (Gill & Mockler, 2016).

SANICOSE Model Attenuation Factors	
Soil Characteristic	**Attenuation Factor**
Shallow Subsoil	**0.71**
High Permeability Soil	**0.90**
Medium Permeability Soil	**0.93**
Low Permeability Soil	**0.94**

Table 2.1: Attenuation factors used by Gill & Mockler (2016) in the SANICOSE model.

2.2.4 Transient Model (Oldfield et al., 2020a)

Beyond the export coefficient method used by Oldfield et al. (2020a), which generalizes the retardation of P in groundwater by assuming that 24% of effluent reaches tributaries, the same authors also develop a transient model which considers the setback distance of each septic system. They use the same export coefficient of 0.81 kg-P person^{-1}year^{-1}, and use evidence from

Robertson et al. (2019) to assume P plumes move at a rate between 1-2 m/yr. The transient model assumes that all septic system operations began in 1940, meaning it does not account for development patterns across the watershed. The model finds that P loading from septic systems to Lake Erie does not begin until 7 years after the simulation began, as the shortest setback distance was 15 m. As the simulation ended in 2020, only systems located within 80 m or 160 m (for 1 m/yr and 2 m/yr plume velocities, respectively) were actively discharging to water bodies. There were also large discrepancies in P loading estimates between two methods of placing septic system locations (Oldfield et al., 2020a), revealing that accurate placement of septic systems has a large effect on P loading in a temporal model due to varying setback distances. The transient model created by Oldfield et al. (2020a) demonstrates the legacy effect of septic systems and shows that the collective impact of septic systems from new development is not likely felt until decades after they are constructed.

2.2.5 Incorporating Policy Changes (Schellenger & Hellweger, 2019)

Temporal models can also use hydrologic equations and export coefficients to account for the retardation of P in groundwater. Schellenger & Hellweger (2019) evaluated P loading from septic systems to Oldham Pond in Massachusetts, United States, over a 300 year period, integrating policy changes that occurred during that time into the model. For example, septic systems at the beginning of the simulation are assumed to load far more P than contemporary systems because of the widespread use of cesspools and the use of phosphate (PO_4^{3-}) in laundry detergents. When conventional septic systems began to be used instead of cesspools and PO_4^{3-} was removed from detergents, septic systems in the model are assumed to load less P. The authors consider complex hydrology by using a 1-dimensional advection-dispersion-sorption equation to directly simulate groundwater flow. They also use a soil retention coefficient in order to consider vadose zone processes before P leaches to groundwater.

The model demonstrates that policy changes may decline the amount of P loaded to soils, and that P loading from septic systems may continue to increase in the absence of new development because new plumes continue to reach the pond. In the Oldham Pond Watershed, all septic system P plumes are projected to reach the pond after 2070. This type of model is applicable to areas where exact septic system locations are available as well as the year each location was first used for wastewater treatment. The use of the advection-dispersion-sorption equation signals that the model considers the transport of P in groundwater, though there may be a high degree of uncertainty in the model as a result of the various input parameters that may be difficult to define (Schellenger & Hellweger, 2019). Furthermore, the model does not consider differences in soil type that affect P retention in the vadose zone, such as shallow groundwater or texture. In spite of this, it is one of the only models that examines P transport from septic systems at such large time scales (300 years).

Chapter III: Methods

3.1.1 Data Collection

Data for septic systems within the Auburn portion of the Lake Auburn watershed were collected by employees of the City of Auburn and myself with data collection finishing in August 2022. Many septic systems were confirmed to exist either through municipal permits (HHE-200) or through the Auburn Water District (AWD). Each permit contains information about the design of the system, including the number of bedrooms in the home it serves and the year the previous system was installed, if available. It also includes soil information including the depth to limiting factor, which is the soil depth where the site evaluator located the seasonal high water table, hydrologically restrictive layer, or bedrock. Information from the AWD consists of the installation year, depth to limiting factor, and whether the limiting factor was groundwater or bedrock.

Furthermore, each permit features the exact location of the system either through GPS coordinates and/or a detailed drawing of the location of the system in relation to the property lines and building. These locations allowed each system to be mapped manually using ArcGIS Online. GPS coordinates were verified by comparing them to the detailed drawing. If the GPS location varied greatly from the drawing, the location from the drawing was used to map the system manually. If the drawing did not provide a clear location, the system was mapped within the property lines in a plausible location (void of trees, large vegetation, and close to building), by using aerial imagery. Systems from AWD datasets were mapped using this same method. Only systems within the watershed were mapped; therefore, if the parcel of land is partially within the watershed, but the system was installed outside of the watershed, the system was not included in the dataset. One septic system was permitted in 2019, though no structure had yet been built on the parcel. This system was included in analyses about soil characteristics under the assumption that it will be built in the future. However, it was included differently in analyses that quantify phosphorus loading.

Some buildings within the watershed do not have an associated permit even though their construction was allowed. Therefore, I used data layers from the City of Auburn including the city parcel lines, watershed boundary, and sewer service to obtain parcels within the watershed without a permit on-file for their septic system. Because some homeowners were granted easements to place their septic systems on a neighbor's land, it was necessary to manually account for these parcels and examine the data for other errors. The exact location of each septic system on these parcels was unknown; the location was determined by using the centroid of the parcel. Certain data such as the number of bedrooms for each home were imported from the City of Auburn's parcel map. The setback distance from each septic system to the nearest lake or major tributary was determined using ArcGIS Pro. Major tributaries were determined from the U.S Fish and Wildlife Service Wetland Inventory. All septic systems within the Auburn portion of the Lake Auburn watershed were merged into a single point layer. The density of septic systems for this area was calculated per square kilometer using ArcGIS Pro.

Most permits for septic systems contain visual descriptions of test soil pits that were recorded by site evaluators. Each test pit description indicates changes in soil texture, consistency, and color. They also mark where mottling (seasonal high water table), restrictive layer, or bedrock is observed, as well as the slope of the site. Each change in texture, consistency, color, or the presence of groundwater marked a new horizon for the purposes of data collection. The vadose zone of each test pit was characterized using the textures of the uppermost and lowest horizons, as well as the soil texture that makes up the largest percentage of the vadose zone by depth. This is henceforth referred to as the "dominant vadose zone texture." The soil pits themselves were described by 31 site evaluators, with the site evaluator's name being illegible for 11 test pits. These data were reconfigured using JMP Pro16 in order to join them with other data from septic system permits.

County-level soil survey data exists for the Lake Auburn watershed through the SSURGO database. Because soil survey data exists at a coarser scale than the test pits from site evaluation (exact location), these data may be less applicable at the site scale, but may reveal trends at the watershed level. Soil survey data from Androscoggin and Sagadahoc Counties, Maine were configured and overlaid with septic system permit data using ArcGIS Pro. Important fields include the hydrologic soil group and the depth to limiting factor, which was determined by the lowest value of the seasonal high water table, bedrock depth, or depth to restrictive layer for all soil types. Though SSURGO also divides the soil profile into horizons, a dataset of weighted means of select soil metrics is available through the USGS. These data were appended to the septic system permit data, and include saturated hydraulic conductivity and percent sand, silt, and clay. A 1/3 arc-second digital elevation model (DEM) from the USGS was used to obtain overland flow paths (streams) leading to large tributaries or Lake Auburn using ArcGIS

Pro. Other data include a map of significant sand and gravel aquifers from the Maine Department of Agriculture, Conservation & Forestry.

3.1.2 Soil Characteristics

Data for the depth to soil limiting factor from site evaluation was interpolated using Bayesian Empirical Kriging in ArcGIS Pro, and interpolated points for each test pit as well as points every 100 m around each waterbody or wetland, which were assigned a value of 0 (for surface water). Wetland data was obtained from the U.S. Fish and Wildlife Service Wetland Inventory. Because the lowest observed depth to limiting factor for all test pits was 4 inches (10.16 cm), all land areas assigned a value below 10 cm were assumed to have insufficient data, considering that the only data available for those areas were wetland data.

3.1.3 Export Coefficient Approaches

A variety of export coefficient methods were applied to these data in order to understand how model assumptions impact loading estimates. Methods similar to those previously used for Lake Auburn (CEI, 2010; FB Environmental, 2021) were replicated with few differences. A 300 ft (91.44 m) buffer was created around Lake Auburn and its major tributaries using ArcGIS Pro. All septic systems within the buffer were assumed to load P to water bodies, while septic systems outside the buffer were not considered. Phosphorus loading from each septic system to Lake Auburn is given by:

$$P = N \bullet \beta \bullet (1 - \alpha) \qquad (1),$$

where *P* is the mass (kg-P/year) of phosphorus loaded to Lake Auburn for a given year, *N* is the number of people her home, which was approximated by using the number of bedrooms each septic system serves (assuming one person per bedroom), β is the export coefficient (kg-P person^{-1}year^{-1}), and α is the attenuation factor (Table 3.1). Like other studies on Lake Auburn, a 0.7 kg-P person^{-1}year^{-1} export coefficient ($\beta = 0.7$) and a 0.8 attenuation factor ($\alpha = 0.8$) were applied to each system (CEI, 2010; FB Environmental, 2021). An attenuation factor of 0.8 assumes that 20% of phosphorus is not retained in soils. Additional attenuation in streams was not considered, which differs from (FB Environmental, 2021). The sum of loading from all systems was considered to be the 2023 P load; this is hereby referred to as the buffer method.

Because it is possible for septic systems beyond the buffer zone to contribute P to Lake Auburn (LeBlanc, 1984; Roy et al., 2017), another export coefficient method was developed based on Oldfield et al. (2020a), who used acesulfame as a tracer of wastewater and developed a model using a transport factor of 0.24 based on a tracer study conducted in the same watershed (Oldfield et al., 2020b). This will be referred to as the "whole watershed method." This approach allows distant systems to be included in the calculation by assuming that 24% of effluent from all systems are contributed to the lake and is given by the equation:

$$P = N \bullet \beta \bullet (1 - \alpha) \bullet \gamma \qquad (2),$$

where P is the mass (kg-P/year) of phosphorus loaded to Lake Auburn for a given year, N is the number of people her home, β is the export coefficient (kg-P person^{-1}year^{-1}), α is the attenuation factor, and γ is the transport coefficient (Table 3.1). Though some septic systems may be actively discharging P while others are not, the transport coefficient accounts for the fact that it is unknown which systems have begun discharging by assuming 24% of P from each system reaches Lake Auburn. The whole watershed method uses the same inputs as the buffer method, while considering all systems within the watershed and using $\gamma = 0.24$ (Oldfield et al., 2020a), within range of transport coefficients used by Gill & Mockler (2016).

A modified export coefficient method based on Gill & Mockler (2016) and Oldfield et al. (2020a) was developed to account for soil heterogeneity across the watershed by varying the attenuation factor (α) based on the soil type (Equation 3). The same export coefficient as CEI (2010) and FB Environmental (2021) was used in order to be consistent across all approaches; this value is within range of export coefficients used in other studies (Gill & Mockler, 2016; Schellenger & Hellweger, 2019; Oldfield et al., 2020a). This method assigns a permeability score to each septic system based on the vadose zone soil texture from site evaluation (Appendix C, Table C.1).

Buffer Method Inputs (Equation 1)				
Input Name	Symbol	Value	Unit	Source
Export Coefficient	β	0.7	kg-P $person^{-1}year^{-1}$	CEI (2010); FB Environmental (2021)
Attenuation Factor	α	0.8	fraction	CEI (2010); FB Environmental (2021)
Number of People per Home	N	Mean: 3.257	bedrooms	Municipal permits
Whole Watershed Approach with Soil Variability and System Failure Inputs (Equation 3; Figure 3.1)				
Input Name	Symbol	Value	Unit	Source
Equation 3				
Export Coefficient	β	0.7	kg-P $person^{-1}year^{-1}$	CEI (2010); FB Environmental (2021)
High Permeability Soil Attenuation Factor	α_h	0.75	fraction	Schellenger & Hellweger (2019)
Medium Permeability Soil	α_m	0.93	fraction	Gill & Mockler (2016)

Attenuation Factor				
Low Permeability Soil Attenuation Factor	α_L	0.94	fraction	Gill & Mockler (2016)
Shallow Soil Depth Attenuation Factor	α_s	0.71	fraction	Gill & Mockler (2016)
Transport Coefficient	γ	0.24	fraction	Oldfield et al. (2020a)
Fraction of P load attributed to failure	θ	0, 0.15, 0.3, 1	fraction	Gill & Mockler (2016); Figure 3.1
Number of People per Home	N	Mean: 3.257	bedrooms	Municipal permits
Figure 3.1				
Soil Permeability		low (3), medium (2), high (1)		Municipal permits, SSURGO database
Setback Distance	D		meters	Municipal permits, ArcGIS Pro
Age of System	A_c		years	Municipal permits

Table 3.1: Input parameters for all export coefficient methods. Literature sources for each value are included. The fraction of P load through a surface pathway depends on the setback distance, as used by Gill & Mockler (2016).

For systems without a site evaluation on-file, the hydrologic soil group was used, and assigned categories based on USDA definitions, as cited in Bean et al. (2021). Soil textures with ≥90% sand or hydrologic soil group A were assigned a permeability score of 1, soil textures described as silty clay or finer or hydrologic soil group D were assigned a permeability score of 3, and all other soil textures or hydrologic soil groups B and C were assigned a permeability score of 2 (Appendix C, Table C.1). These scores allow for each septic system to be assigned an attenuation factor. Any system with a depth to limiting factor less than 91.44 cm (36 in) received an attenuation factor of $\alpha_s = 0.71$, regardless of permeability score, based on Gill & Mockler

(2016). Using one attenuation factor for shallow subsoils regardless of soil texture is applicable considering evidence from Karathanasis et al. (2006), who find that differences in soil texture do not lead to statistically significant differences in P treatment until at least 60 cm of soil depth. For all septic systems with a depth to limiting factor greater than 91.44 cm, attenuation factors were assigned based on permeability score (Table 3.1). For a high permeability soil (permeability score of 1), an attenuation factor of $\alpha_h = 0.75$ was used, based on Schellenger & Hellweger (2019), who created a similar model in a watershed with similarly non-calcareous sands. Although Gill & Mockler (2016) use an attenuation factor of 0.9 for high permeability soil, many studies note variable P attenuation in sandy soils that is often not as robust as 90% (Robertson, 1995; Wilhelm et al., 1994; Roy et al., 2017). Therefore, a value of 0.75 was used. For a permeability score of 2, an attenuation factor of $\alpha_m = 0.93$ was used (Gill & Mockler, 2016). For a permeability score of 3, an attenuation factor of $\alpha_L = 0.94$ was used (Table 3.1).

Another version of the modified export coefficient method accounts for P loading through a surface pathway from failing systems. The amount contributed from failure differs based on soil permeability and setback distance as used by Gill & Mockler (2016). However, contrary to the approach by Gill & Mockler (2016), no attenuation of this P was considered, as many models assume that surfacing effluent from failing septic systems is a direct input to the lake or stream, receiving no additional treatment (Jeong et al., 2011; Sinclair et al., 2014). In addition to assuming that septic systems on low permeability soils (permeability score of 3) were contributing P through a surface pathway, I also assumed that systems installed over 29 years ago (median time until system replacement in Auburn) were failing and also contributed varying amounts of P through a surface pathway based on setback distance (Figure 3.1). This age is close to reported values of septic system lifespan in other New England states (Hill & Frink, 1980).

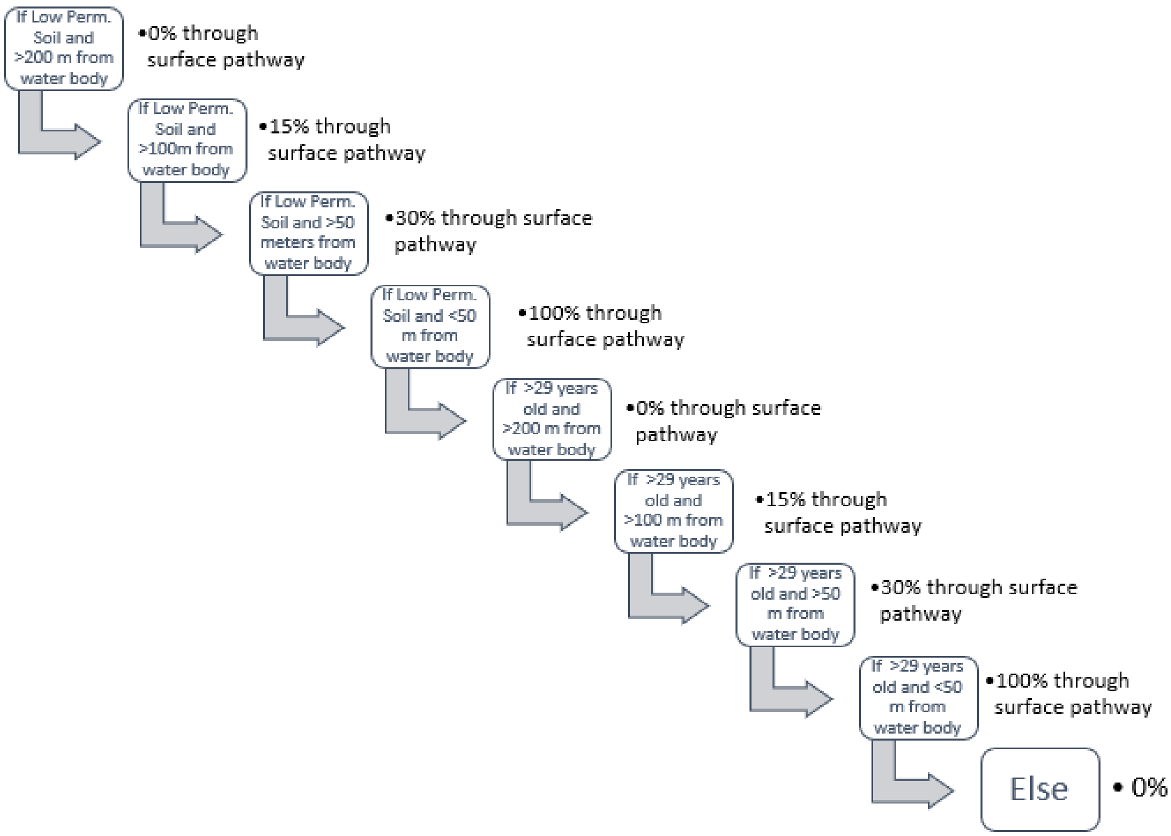


Figure 3.1: Flow diagram describing the percentage of septic effluent that is attributed through a surface pathway for each septic system on the basis of soil permeability, system age, and setback distances and percentage of load from Gill & Mockler (2016).

Considering the watershed as a whole, soil permeability, system age, and setback distance, the total P load to Lake Auburn for each system was calculated using the equation:

$$P = (N \bullet \beta)(\theta + (1 - \theta))(1 - \alpha_i)(\gamma)) \qquad (3)$$

where P is the mass (kg-P/year) of phosphorus loaded to Lake Auburn for a given year, N is the number of people her home, β is the export coefficient (kg-P $person^{-1}year^{-1}$), θ is the percent of phosphorus attributed to the failing system determined from Figure 3.1, α_i is the attenuation factor for the given soil type, and γ is the transport coefficient. The sum of P loading from all systems was considered to be the 2023 P load. This method is referred to as the whole watershed with soil variation and system failure method.

3.1.4 Temporal Model

Using the modified export coefficient method from above, I created a temporal model based on Schellenger & Hellweger (2019) using JMP Pro16 that spans 200 years of wastewater disposal in the Auburn portion of the Lake Auburn watershed. The temporal model assumes that no new development occurs in the watershed, meaning no new septic systems are added beyond what already exists. Each septic system operation was simulated to begin based on the year the dwelling was built, based on the City of Auburn's parcel map. If no data existed, the median year of the above data set, 1949, was used. Only four data points used this median. Similar inputs as the whole watershed with soil variation and system failure method were used (Table 3.2). Notable differences are that the 0.24 transport coefficient was removed because P transport was being simulated directly, and that an average of 2.325 people per home (U.S. Census data from FB Environmental, 2021) was used instead of the number of bedrooms. This was because the number of people or bedrooms in a home is not constant over time, meaning the number of bedrooms would inaccurately reflect past loading. This altered the equation to become:

$$P = (N \bullet \beta_i)(\theta + (1 - \theta))(1 - \alpha_i)) \qquad (4),$$

where P is the mass (kg-P/year) of phosphorus loaded to Lake Auburn for a given year, N is the number of people her home, β_i is the export coefficient for the respective time period (kg-P person^{-1}year^{-1}), θ is the percent of phosphorus attributed to the failing system determined from Figure 3.2, and α_i is the attenuation factor for the given soil type.

In addition, the amount of P attributed from failing systems was modified. Instead of determining the percentage of P from failure from systems installed over 29 years ago, I assumed that each system failed every 29 years (median replacement age in Auburn). Because of computational limitations, I distributed the P loading from failure across the entire 29 year period of septic system operation. This also allows me to account for the fact that it is impossible to know which systems are failing and which are not at a given time. Depending on the setback distance of the system, this approach led to varying percentages of P attributed to failure for each system (Figure 3.2). Low permeability soils were also assumed to contribute P through a surface pathway, as used in previous methods (Gill & Mockler, 2016).

The speed of the P plume beneath each septic system was estimated with the Darcy equation given by Hölting & Coldewey (2018) and the retardation factor:

$$V_P = \left(\frac{K_{sat}}{\phi} \bullet \frac{dh}{dl}\right) \div R_f \qquad (5)$$

where V_P is the plume velocity (meters/day), K_{sat} is the saturated hydraulic conductivity (meters/day), ϕ is the effective porosity (fraction), $\frac{dh}{dl}$ is the change in hydraulic head, and R_f is the retardation factor (fraction). For each septic system, K_{sat} values from the SSURGO database, an assumed $\frac{dh}{dl}$ of 0.03 (from a median surface slope of 3% at all site evaluations), and effective porosity values assigned from Hölting & Coldewey (2018) based on the bottom-most soil texture of each system (Appendix C, Table C.2) were used. For R_f, upper and lower bounds were developed using $R_f = 20$ and $R_f = 100$, the full range of values from Robertson et al. (1998). Using upper and lower estimates is useful here given the uncertainty of how phosphorus moves in the Lake Auburn watershed, because no field studies on phosphorus transport in groundwater exist for this watershed. The fastest potential movement is simulated by setting $R_f = 20$, the lower end of literature values, likely simulating quicker phosphorus movement than in reality (Roy et al., 2017). However, $R_f = 20$ is close to the value used by Schellenger & Hellweger (2019), who used $R_f = 21.56$ to model septic P loading in a similarly non-calcareous, sandy watershed.

The P plume velocity allows calculation of the time it takes for P to begin discharging into Lake Auburn or a tributary based on the equation:

$$Year\ of\ First\ Discharge = \frac{D}{V_P} + Y_i \qquad (6),$$

where D is the setback distance in meters of a given septic system to the lake or major tributary, V_P is the plume velocity (m/year), and Y_i is the installation year of the system.

3.1.5 Role of Ban of Phosphate in Detergent

An important change that has occurred since septic systems were first installed in the watershed was the ban of phosphate (PO_4^{3-}) from laundry detergents in 1973 (Litke, 1999). This was incorporated by increasing the export coefficient for years prior to 1973 to $\beta = 1.214$ (Table 3.2), which was determined from 221.8 L/day of water use per person from Oldfield et al. (2020a) and 15 mg-P/L in household wastewater before 1973, which Schellenger & Hellweger (2019) derived from McCray et al. (2005). Then, the year when the P that originated following the ban on phosphate in detergent began discharging into the nearest water body was determined using the equation:

$$Year\ of\ Discharge\ of\ P\ Originating\ After\ 1973 = \frac{D}{V_P} + 1973\ (if\ Y_i < 1973) \quad (7),$$

where D is the setback distance in meters of a given septic system to the lake or major tributary, V_P is the plume velocity (m/year), and Equation 6 is used when $Y_i \geq 1973$.

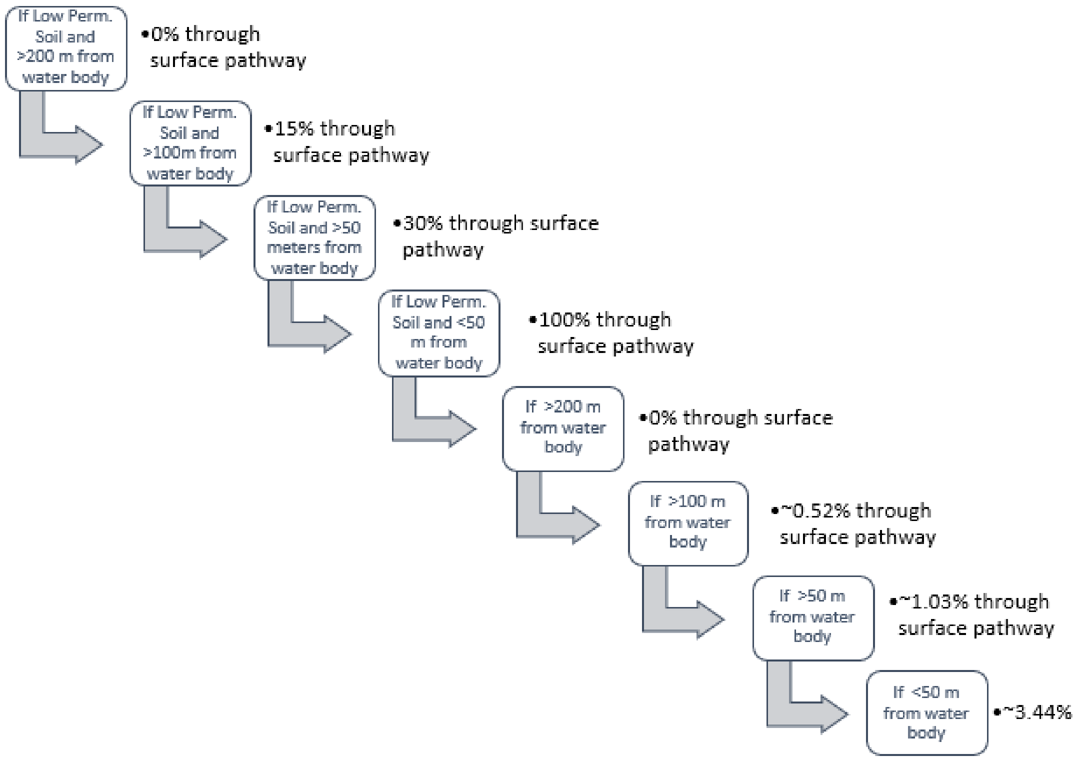


Figure 3.2: Flow diagram describing the percentage of septic effluent that is attributed through a surface pathway for each septic system, based on soil permeability, setback distance, and percentages of P load from different pathways as described by Gill & Mockler (2016). Failing systems are simulated by assuming a system fails once every 29 years (median age of replacement in Auburn), meaning the percentage is multiplied by a failure of probability of 1/29.

3.1.6 Investigation of Impact Septic Ordinance Change

To investigate the potential impact of Auburn's proposed septic ordinance change, an additional model was created which assumes the new ordinance is passed in 2023. This ordinance change may require that older systems that do not have the currently required 36 inches (91.44 cm) of drainage depth use a mounded septic system at the time of system replacement. Such systems would presumably have greater retention capacity because of the thicker vadose zone (Reide Corbett et al., 2002). If such systems were installed each time an older system on shallow soil failed, the year when P originating after the installation of such new systems would first discharge into a water body was calculated as:

$$Year\ of\ First\ Discharge\ After\ Mounded\ =\ \frac{D}{V_P} + Y_R \qquad (8),$$

where D is the setback distance in meters of a given septic system to the lake or major tributary, V_P is the plume velocity (m/year), and Y_R is the year that a given septic system is replaced after the proposed ordinance change, which was determined by:

$$If\ A_c\ is\ > 29,\ Y_R\ =\ 2023\ +\ 1 \qquad (9)$$

$$If\ A_c\ is\ < 29,\ Y_R = 2023\ +\ 1\ +\ (29\ -\ A_c) \qquad (10)$$

$$If\ A_c\ is\ missing,\ Y_R = 2023\ +\ \varepsilon \qquad (11),$$

where A_c is the age of the current system and ε is a random integer between 1 and 29. This set of equations assumes that systems would take at least a year to be replaced after any failure (if older than 29 years) (Equation 7), and that systems younger than 29 years will not be replaced until a year after they begin failing (Equation 8). Inherently, this assumes that the City of Auburn takes swift action to follow up with homeowners on septic system inspections and maintenance that would lead to replacements. If the age of the current system was unknown, the system was assumed to fail in a random year between 2024 and 2053 (a range of 1 to 29 years after the ordinance change). For this version of the model, different values of the attenuation factor (α_i) are applied based on how the proposed change would affect P retention beneath each septic system. Aspects of the proposed change that affect P retention are that 24 in (60.96 cm) of imported media may be added to increase soil depth, and that this imported media must be loamy sand in texture (City of Auburn, 2022c). I incorporated this into the modeling approach by assuming that all systems with a depth to limiting factor less than 12 in (30.48 cm) would still have shallow subsoils even with imported media, thereby using α_s. For systems with a depth to limiting factor greater than 12 in (30.48 cm) and less than 36 in (91.44 cm), α_m was applied and the permeability score was changed to 2, to simulate the installation of a mounded system (Equation 4; Table 3.2). For systems with a depth to limiting factor greater than 36 in (91.44 cm), the attenuation factor and permeability score were kept the same.

The temporal model uses each of the inputs described above on a time series to approximate P loading and transport to Lake Auburn. Combinations of P loading from failing systems and subsurface contributions of P are added to the lake based on the year in the simulation and the year in which the septic system begins operating (Figures 3.3, 3.4). For the temporal model, the one system that had been permitted but not yet constructed was simulated to begin operation in 2024.

Temporal Model Inputs (Equations 4 - 10; Figure 3.2)				
Input Name	Symbol	Value	Unit	Source
Equation 4				
Pre-1973 Export Coefficient	β_1	~1.214	kg-P person^{-1}year^{-1}	Schellenger & Hellweger (2019); Oldfield et al. (2020b)
Post-1973 Export Coefficient	β_2	0.7	kg-P person^{-1}year^{-1}	CEI (2010); FB Environmental (2021)

Number of People per Home	N	2.325	people	U.S. Census, as determined by FB Environmental (2021)
High Permeability Soil Attenuation Factor	α_h	0.75	fraction	Schellenger & Hellweger (2019)
Medium Permeability Soil Attenuation Factor	α_m	0.93	fraction	Gill & Mockler (2016)
Low Permeability Soil Attenuation Factor	α_L	0.94	fraction	Gill & Mockler (2016)
Shallow Soil Depth Attenuation Factor	α_s	0.71	fraction	Gill & Mockler (2016)
Fraction of P load attributed to failure	θ	0, 0.15, 0.3, 1	fraction	Gill & Mockler (2016); Figure 3.2
Figure 3.2				
Soil Permeability		low (3), medium (2), high (1)		Municipal permits, SSURGO database
Setback Distance	D		meters	Municipal permits, ArcGIS Pro
Probability of System Failure		1/29		Municipal permits
Equation 5				
Hydraulic Conductivity	K_{sat}		meters per day	SSURGO Database
Effective Porosity	ϕ	0.035 - 0.175		Hölting & Coldewey (2018)
Change in Hydraulic Head	$\frac{dh}{dl}$	0.03	fraction	Municipal permits
Retardation Factor	R_f	20,100	ratio	Robertson et al. (1998)
Equations 6 -11				
Setback Distance	D		meters	Municipal permits, ArcGIS Pro
Year Built	Y_i	Median: 1949	year	Municipal permits
Age of Current System	A_c		years	Municipal permits

Table 3.2: Inputs for the temporal model organized by which equation or figure of application. Inputs that influence more than one equation or figure are repeated.

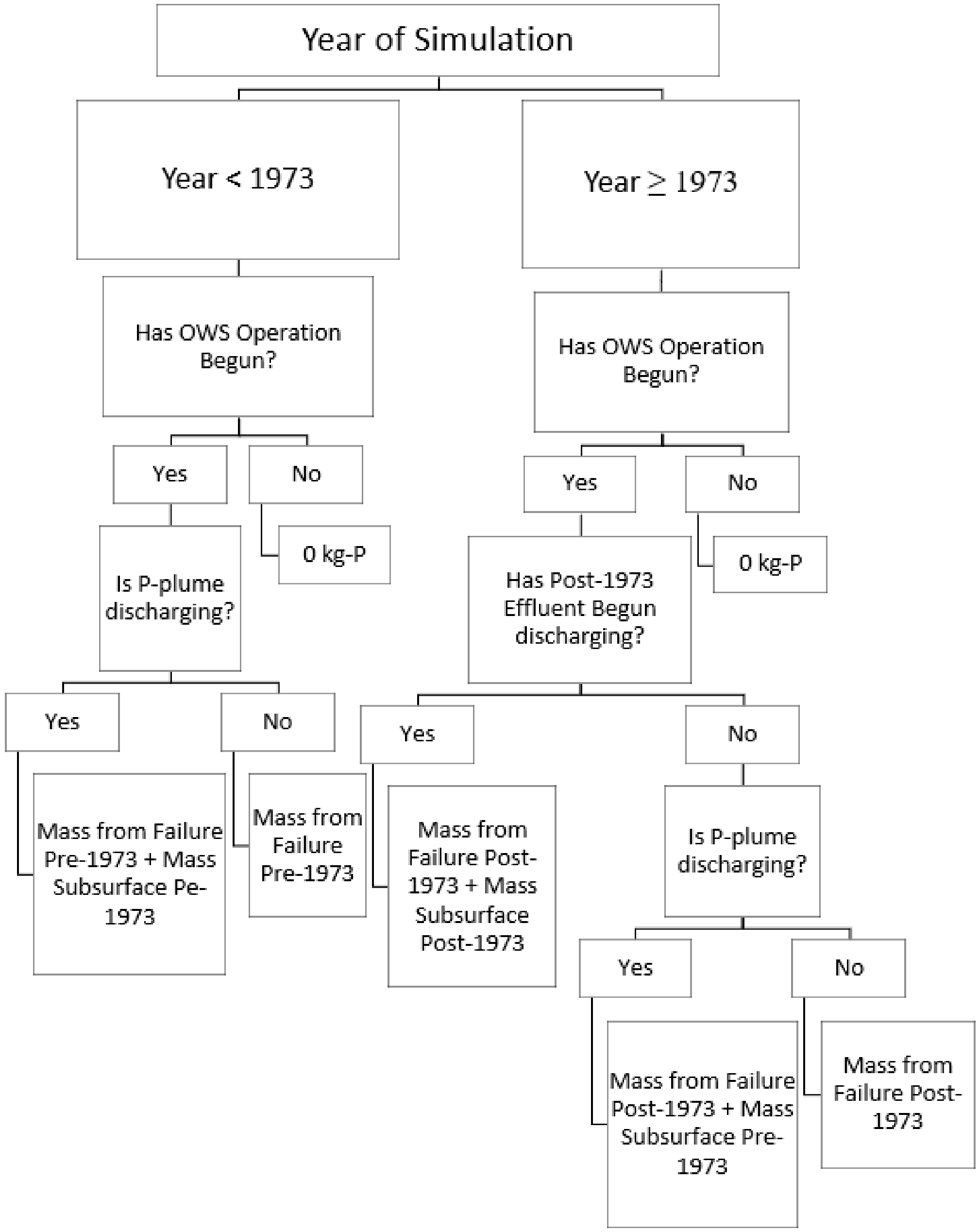


Figure 3.3: Flow diagram of the logic in the temporal model without implementation of the proposed ordinance change. Contributions from each septic system are identified through "if" statements that lead to some combination of P from failure and subsurface discharge, depending on the year of the simulation and the characteristics of the septic system. The mass of P determined at the end of the flow diagram is added to the cumulative load for the given year.

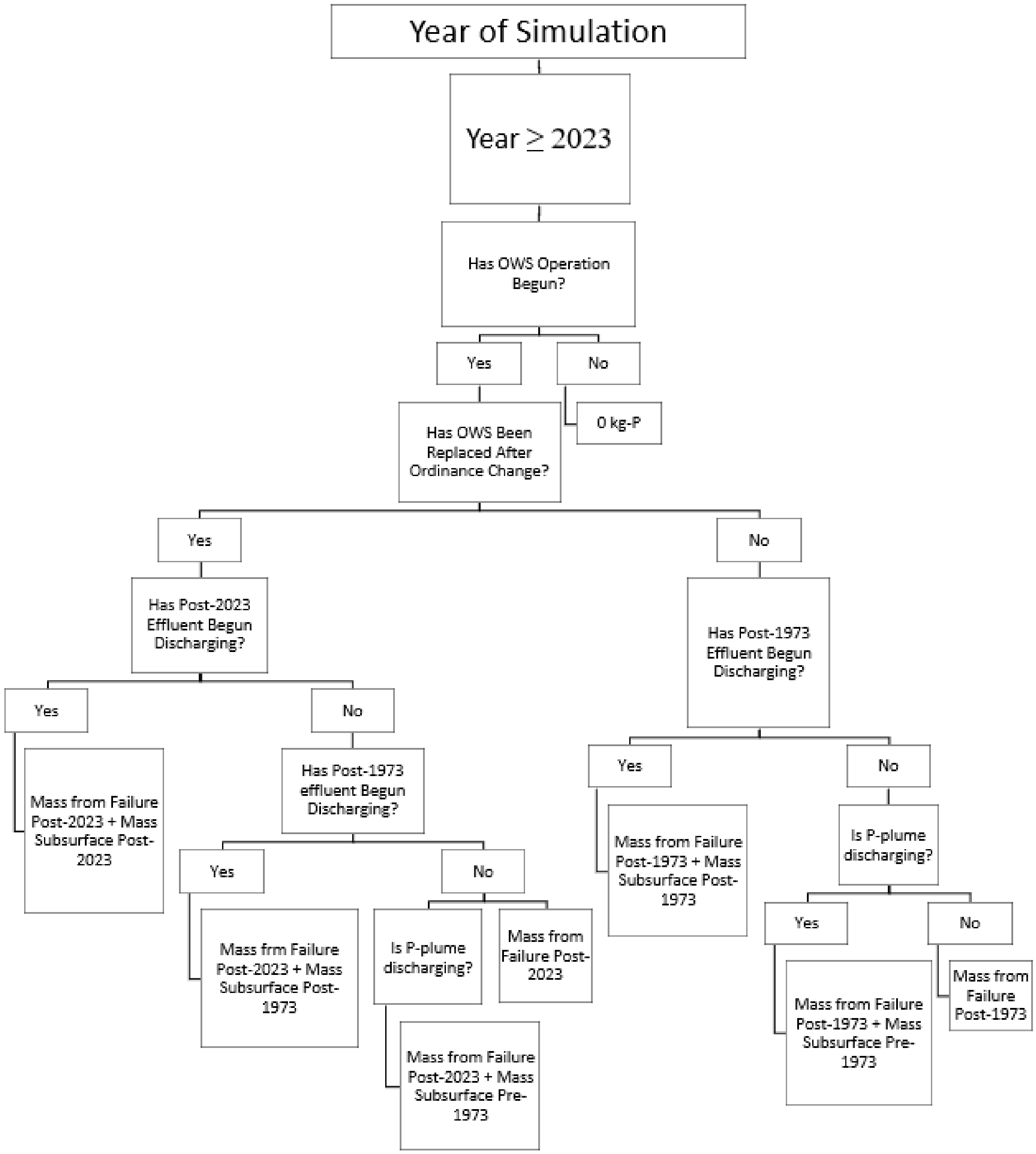


Figure 3.4: Flow diagram of the login in the temporal model, assuming the proposed ordinance change is passed in 2023. For years prior to 2023, Figure 3.3 is used. Contributions from each system are identified through "if" statements that lead to some combination of P from failure and subsurface discharge, depending on the year of the simulation and the characteristics of the septic system. The mass of P determined at the end of the flow diagram is added to the cumulative load for the given year.

Chapter IV: Results and Interpretation

4.1 Qualitative Analysis

4.1.1 Distribution of Septic Systems

Result: Based on municipal permits and the City of Auburn parcel map, there are 326 septic systems in the Auburn portion of the Lake Auburn watershed including one system that was permitted for 2019 but has not yet been built. Two areas of high septic system density exist in Auburn; one is at the northern inflow of the lake (the Basin) and the other is on the northeastern side of the lake near Townsend Brook (Map 4.3).

Most septic systems are in areas zoned rural residential and low density country residential, with few systems in the Agriculture and Resource Protection Zone (Map 4.1). On the west side of the lake, septic systems line both sides of North Auburn Rd, where an area of steep slope separates the road from Lake Auburn (Map 4.1, 4.2). Site evaluators noted seven septic systems on slopes of 15% or greater, all of which are located near overland flow paths (Map 4.2). A fifth of the systems (71 of 326, 21.8%) do not have at least a 300 ft (91.44 m) setback from a major tributary or the lake itself, while the remainder are not adjacent to the shoreline. Systems within close proximity to tributaries or the lake mostly cluster around the Basin inlet (Map 4.11).

Interpretation: Zoning regulations and other policies strongly impact the historical placement of septic systems. Systems placed on slopes of 15% or greater are more likely to contribute P through overland flow, as laterally moving effluent can emerge at the edge of the slope (Brady & Weil, 2002). All seven of the septic systems on steep slopes are located in close proximity to intermittent streams (Map 4.2), greatly increasing the likelihood of a surface connection between these systems and Lake Auburn when they fail.

Systems near the shoreline or tributaries begin discharging subsurface effluent to waterways more rapidly than systems farther from waterways, and they pose a much larger risk of contributing P into the lake when systems fail (Efroymson et al., 2007). Older systems are more likely to be failing due to biomat development preventing downward wastewater percolation (Siegrist, 1987). The Basin inlet is the area of greatest concern with regard to this considering the large cluster of systems within 300 ft (91.44 m) that also have shallow water table depths (Map 4.6). Other systems throughout the watershed on high runoff potential soils (hydrologic soil groups D, A/D, B/D, C/D) and near streams are also more likely to load P directly during failure (Maps 4.6, 4.9; Withers et al., 2014).

High density clusters of septic systems around the Basin inlet and Townsend Brook may be hotspots of P loading because high masses of P originate from these areas of high water flow. Both of these inlets are known to load high masses of P to Lake Auburn (Gundersen, 2020). The cluster of septic systems around the Basin are in an area that was developed early in the history of European colonization in the watershed (as discussed in Doolittle et al., 2018), and those near Townsend Brook result both from development along Route 4 and a high density residential area (Map 4.3).

4.1.2 Permits and Soils

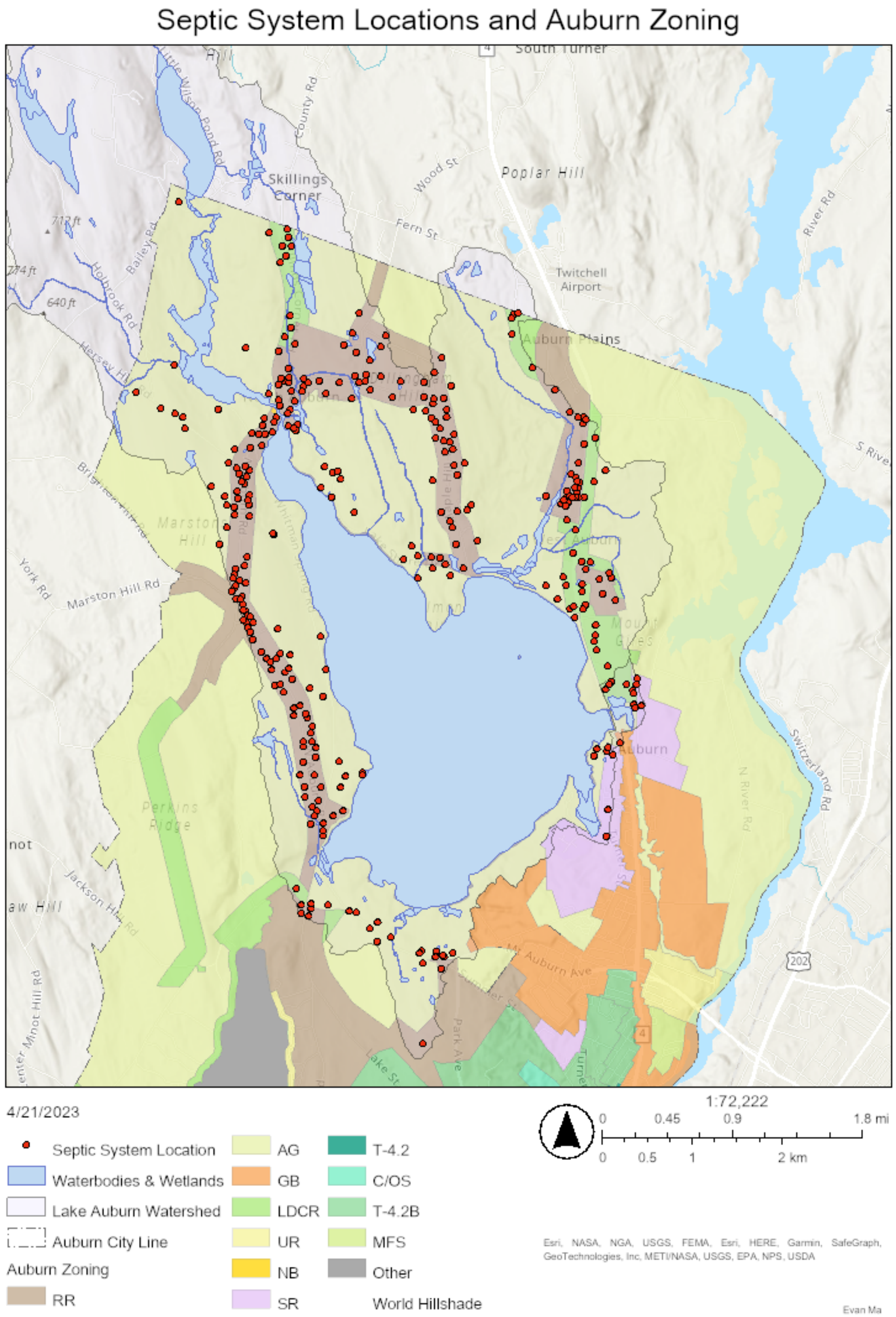
Result: The Auburn portion of the Lake Auburn watershed is characterized by shallow subsoils. While there are 326 septic systems in the Auburn portion of the watershed, only 234 of them have permits on file, and the majority of permitted systems (62.4%) have a depth to limiting factor less than the required 36 inches (Table 4.1). Of the systems with permits on file, 48 of the 88 septic systems that meet or exceed the 36 inch (91.44 cm) requirement are installed on coarse-textured soils such as sand and gravel. Many of these systems are part of the cluster near Townsend Brook, which overlie a sand and gravel aquifer (Map 4.3). Soil survey data suggest that the entire watershed is composed of non-calcareous, low pH soils that have a high percentage of sand and a low percentage of clay (Map 4.10).

Interpretation: The fact that 62.4% of permitted systems have a vadose zone depth of less than 36 inches (91.44 cm) (Table 4.1) indicates that a large percentage of septic systems in the Auburn portion of the Lake Auburn watershed may not appropriately attenuate P because they do not have sufficient soil depth (Mechtenismer & Toor, 2017), especially considering that drain fields must be installed 12 inches (30.5 cm) below the soil surface (City of Auburn Ordinance Chapter 60, 2009). If all systems with less than 36 inches (91.44 cm) of soil were required to be upgraded to a mounded system at the time of failure, this would decrease the percentage of systems with less than 36 inches (91.44 cm) of soil depth to 8.1% of permitted systems. Thus, more septic drain fields would have at least a meter of unsaturated soil, which is critical for proper P retention (Reide Corbett et al., 2002; Karathanasis et al., 2006; Baer et al., 2019). However, these systems would only be improved upon replacement, which may take decades until failure occurs and the system is replaced (Hill & Frink, 1980). Moreover, should the proposed change in the septic requirements allow for increased development in the watershed, such development would more than nullify the benefit of improving these systems (CEI, 2010; Section 4.4 below).

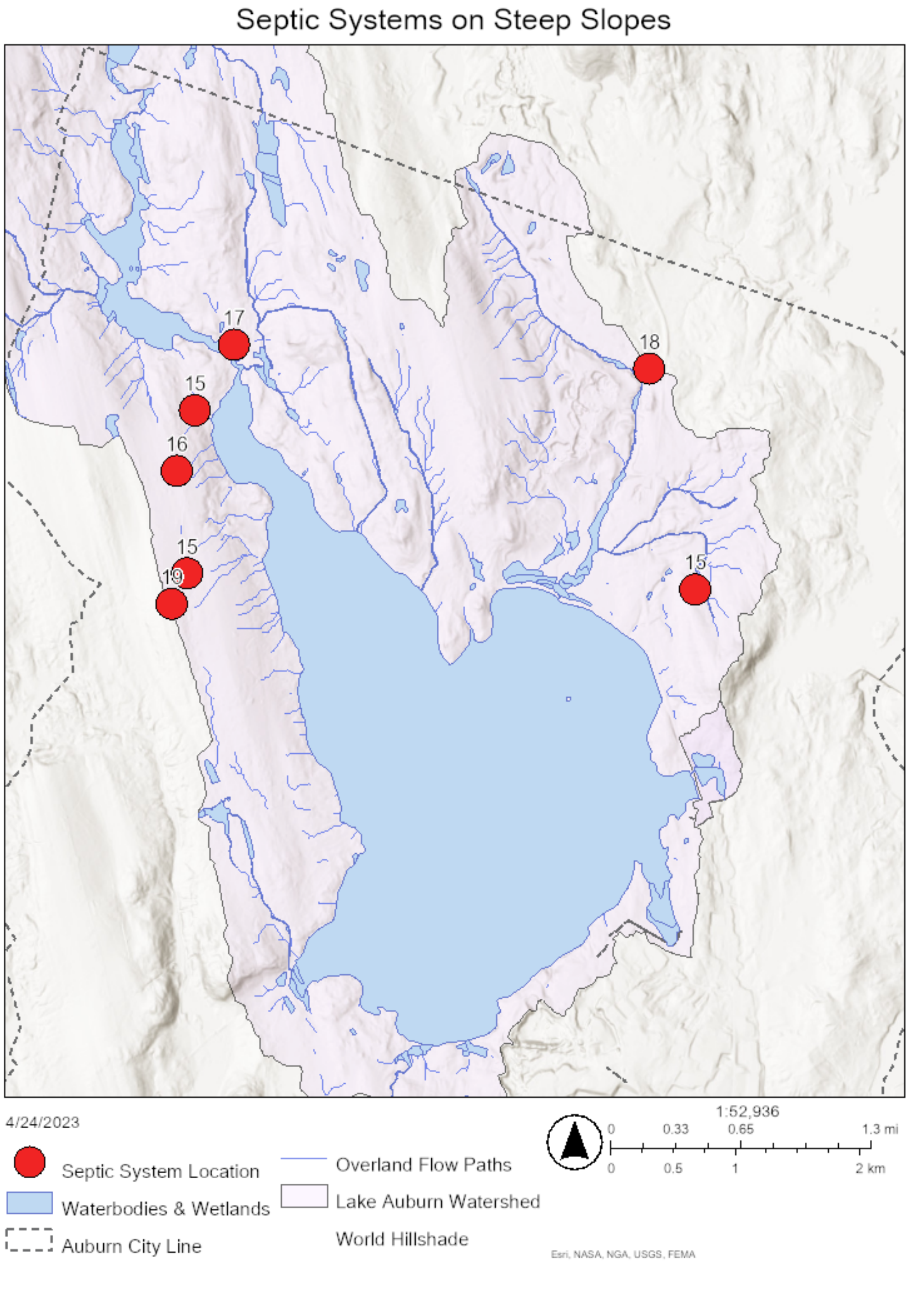
The location of the 88 (37.6%) that meet the 36 inch (91.44 cm) requirement on soils with >90% sand or gravel (Map 4.4), textures with low P retention capacity (McCray et al., 2005; Carroll et al., 2005) may be problematic for P retention even with an ordinance change. The proposed ordinance change would only improve P retention in these systems by removing the requirement that drain fields must be installed 12 inches (30.5 cm) below the soil surface (City of Auburn Ordinance Chapter 60, 2009). Therefore, these disposal fields may be placed on top of the soil surface at the time of replacement, thereby including finer-textured soils that overlay the sand and gravel. However, as most topsoil layers found in site evaluations were 6 inches (~15 cm) or less, any improvement at the time of replacement may be small.

The proposed ordinance also has language that prevents the construction of a septic system on sand and gravel within a 400 ft (121.9 m) buffer of a shoreline (City of Auburn, 2022b). Even with an increase of the buffer to 400ft (121.9 m) in which construction of septic systems is prohibited, any systems installed on the aquifer are likely to be a higher P loading risk in the long term due to the high hydraulic conductivity of underlying and adjacent soils (Map 4.8) (Harman et al., 1996). In sand aquifers, sewage disposal sites up to 520 m downgradient have been noted to contribute P in exceptional circumstances (LeBlanc, 1984), and P plumes from septic systems have been observed to move as far as 100 m in 44 years (Robertson et al., 2019). Considering that the area in northeastern Auburn (where the sand and gravel aquifer is) has had many residences for far longer than 44 years, it is intuitive that P from beyond the buffer area has likely reached the shoreline.

On the other hand, watershed soils are non-calcareous with low pH suggesting that some soils may have a high P retention capacity (Brady & Weil, 2002; Robertson, 2008), as long as Fe and Al are present (Borggaard et al., 1990; Freese et al., 1992); non-calcareous soils often have higher levels of iron (Fe) and aluminum (Al) oxides/hydroxides, and the low pH leads to higher availability of metal cations (Singer & Munns, 2006). Precipitation of P with Fe and Al and specific phosphate adsorption (Zanini et al., 1998), mechanisms that retain P more strongly than other types of adsorption (McBride, 1994; Filep, 1999; Sposito, 2008), are the most likely mechanisms of P retention. The lack of calcium in these soils indicates that precipitation with Ca is unlikely (Gill et al., 2009). The high percentage of sand throughout the watershed (Map 4.10) may lead to poor P retention because the coarse-grained minerals have low surface area, creating fewer sorption sites than on finer-grained soils (McGechan & Lewis, 2002; Carroll et al., 2005: Tan, 2011).

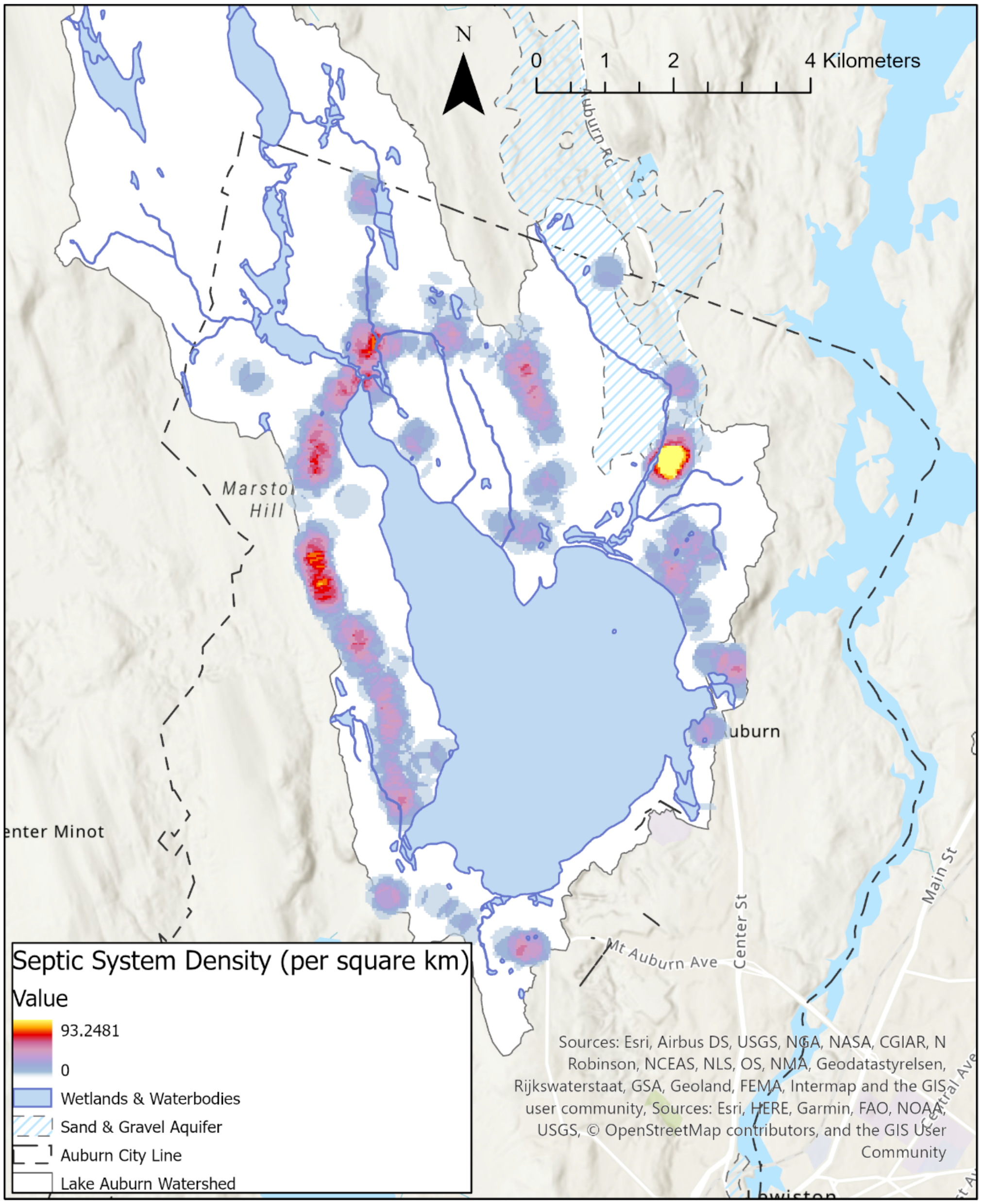


Map 4.1: The distribution of septic systems in Auburn is strongly related to zoning regulations. Zoning classifications of RR, LDCR, and SR are residential, while AG is the Agriculture and Resource Protection Zone.



Map 4.2: Seven septic systems are noted to be on slopes of 15% or greater. These systems exist close to overland flow paths (streams). Darker colors signify steeper slopes.

Density of Septic Systems in the Auburn Portion of the Lake Auburn Watershed



Map 4.3: Density of septic systems in the Auburn portion of the Lake Auburn watershed. The highest density (~90 septic systems/square km) occurs near Townsend Brook, on a sand and gravel aquifer. Other high-density areas are near the Basin and North Auburn Road. Systems outside of the watershed within Auburn or within the watershed in other towns are not shown.

4.1.3 The Basin Inlet Soil Characteristics

Result: A cluster of septic systems exists near the Basin inlet, the largest inflow to Lake Auburn that also contributes the largest amount of P to the lake (Gundersen, 2020). The soil textures in the vadose zone here are predominantly fine sandy loam or loam with shallow vadose zone depths (Map 4.5). Most systems are located on soils with less than 36 inches (91.44 cm) of unsaturated soil (Map 4.5). Of systems within this area that have at least a meter of unsaturated soil drain fields are medium sand, while there are only a few that have loamy soil textures (Map 4.5). Moreover, most of these systems discharge to the Lake Auburn shoreline rather than the actual inlet of the Basin, based on subwatersheds determined by Gundersen (2020).

Interpretation: The Basin not only has clusters of septic systems close to the shoreline, but also has soil depths far shallower than 36 inches (91.44 m), as found by site evaluators (Map 4.5) and as suggested by soil survey data (Map C.2). Though the fine-textured soil does not pose a concern with respect to P retention (Gill et al., 2009), the lack of unsaturated soil depth may lead to inadequate P attenuation due to shallow soils (Mechtensimer & Toor, 2017). The close proximity of these systems to the shoreline and the long time since they were installed likely means that P plumes from these systems actively discharge to Lake Auburn (Reide Corbett et al., 2002; Rakhimbekova et al., 2021).

Depth to Limiting Factor of Permitted Systems			
Depth (in)	**Depth (cm)**	**Number**	**%**
<12	**<30.48**	**19**	**8.1**
<36	**<91.44**	**127**	**54.3**
36+	**91.44+**	**88**	**37.6**
Total		**234**	**100**

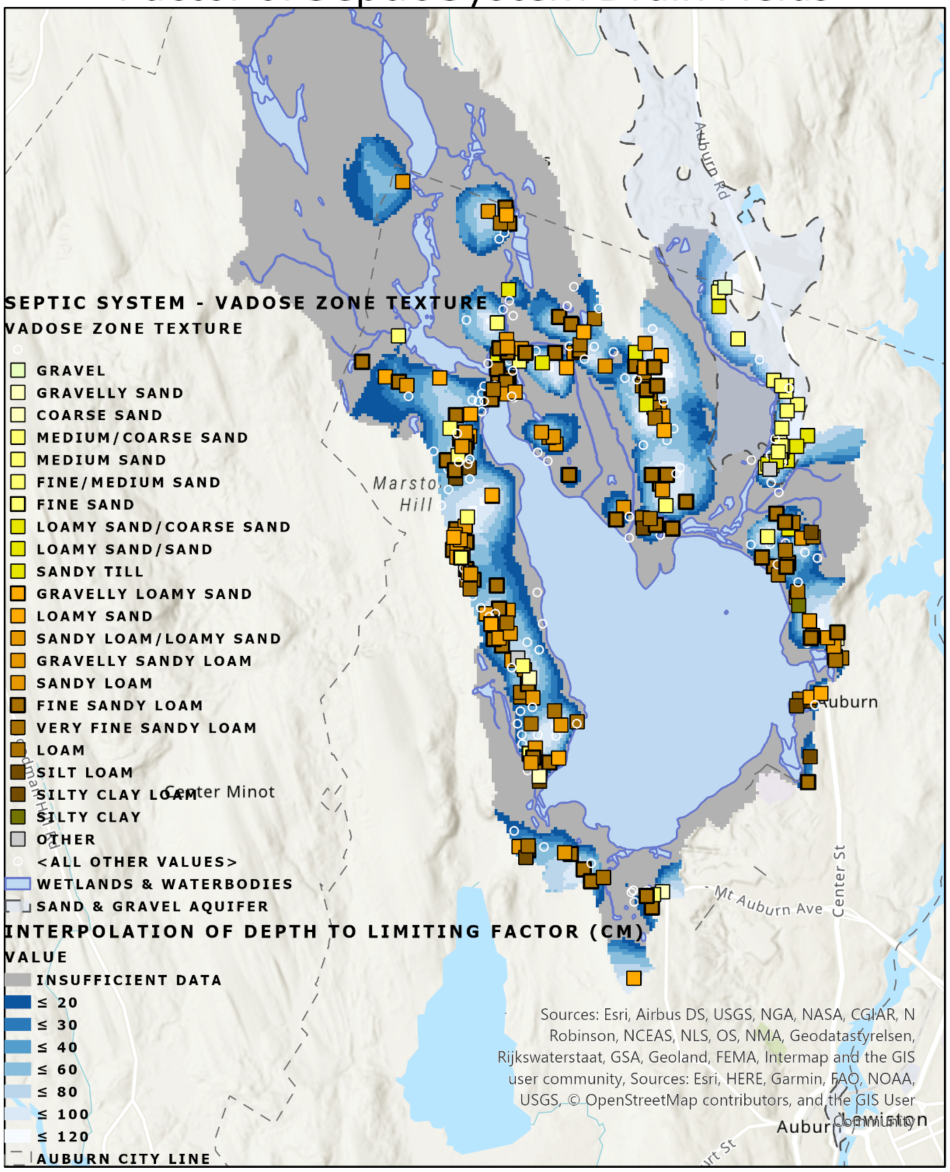
Table 4.1: Distribution of depth to limiting factor of all septic systems with site evaluations (234 of 326 systems). Systems with a depth to limiting factor less than 12 inches (30.5 cm) still would not have sufficient soil depth for P treatment if proposed ordinance change was passed, because a maximum of 24 inches of imported media can be used in mounds (City of Auburn, 2022a). Those over 12 inches (30.5 cm) but less than 36 inches (91.4 cm) would have improved P retention.

4.1.4 Townsend Brook Soil Characteristics

Result: Although the vast majority of septic systems within the Townsend Brook subwatershed are located on well-drained soils with well over a meter of unsaturated soil, site evaluations find that these soils are extremely sandy, with many on fine to coarse sands, and few on gravel (Map 4.7). Soil survey data also suggest that this entire area is over 90% sand, indicating that the soil texture is very coarse (Map 4.10); this is unsurprising considering that these systems overlie a sand and gravel aquifer which feeds Townsend Brook. This area contains both businesses and a housing subdivision with a high density of septic systems (Map 4.3). Hydraulic conductivity within the sand and gravel aquifer is on the order of 10 m/day according to the soil survey data (Map 4.8).

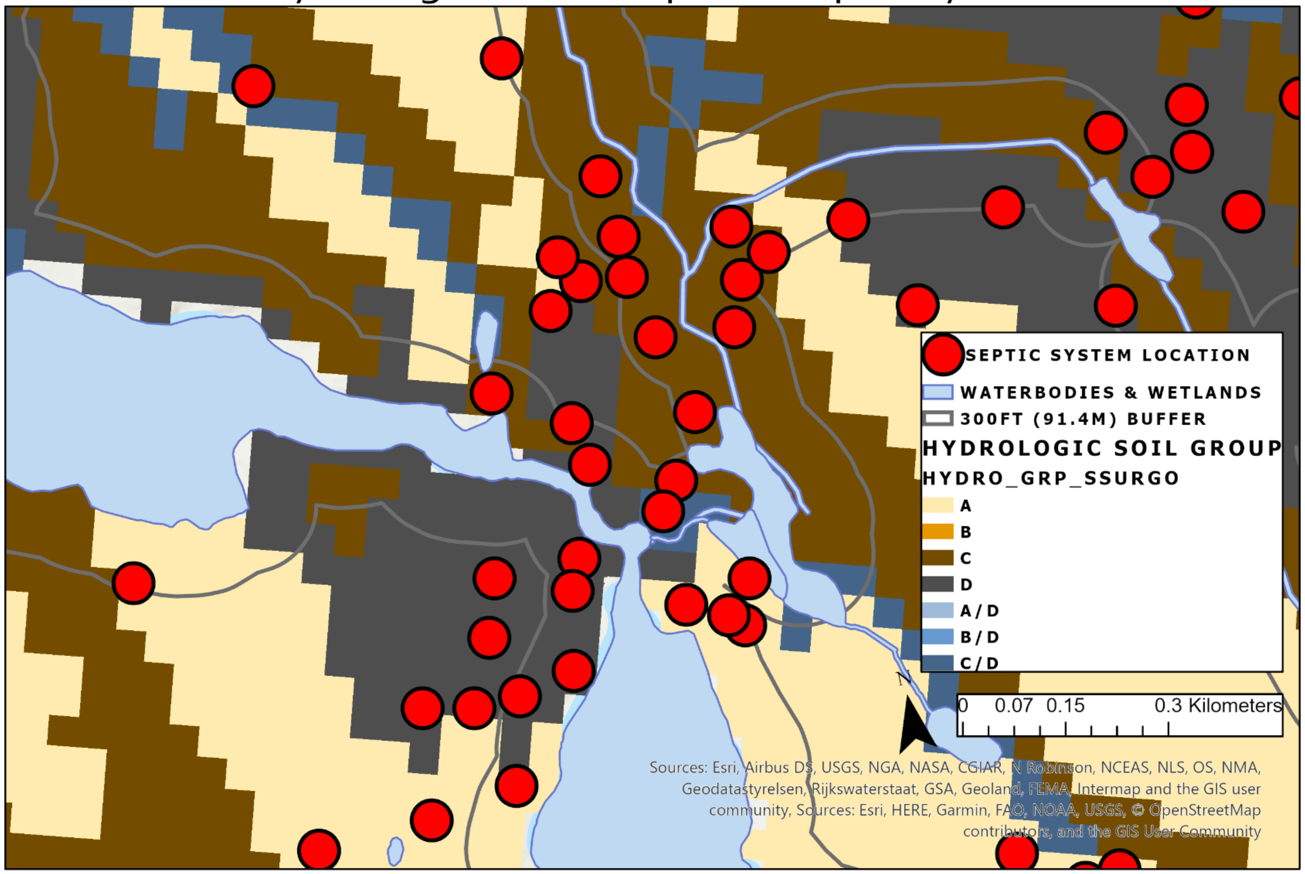
Interpretation: Townsend Brook, the second largest source of both water and P to Lake Auburn (Gundersen, 2020), is fed by a sand and gravel aquifer with a high density of septic systems (Map 4.3). The coarse-textured soils here are far worse at attenuating P than finer-textured soils due to having less surface area and ion exchange capacity (Carroll et al., 2005; Karathanasis et al., 2006; Gill et al., 2009), and therefore fewer sites for phosphate to adsorb with clay and Fe/Al oxides and hydroxides (Freese et al., 1992). Furthermore, some studies in sand aquifers have found minimal P retention in the unsaturated zone, even if it is well over a meter deep (Wilhelm et al, 1994; Robertson, 1995). The amount of Fe and Al in these soils is unknown, as there are no Fe/Al data in the soil survey dataset; this may be worth investigating because the absence of Fe/Al hydr(oxides) can result in minimal retention of P in the drain field (Borggaard et al., 1990; Gill et al., 2009) and retardation in groundwater (Ma et al., 2021). The density of septic systems in this area is the highest of anywhere in the watershed, so the Townsend Brook subwatershed may be a loading hotspot due to the high-density cluster of septic systems loading P to soils that have a lower P retention capacity due to their coarse-texture (McCray et al., 2005; Mechtensimer & Toor, 2016), making them more susceptible to P leaching (Nair et al., 2004) in spite of their well-drained nature. Furthermore, the fact that these systems discharge into a sand and gravel aquifer means that groundwater and therefore P moves quickly toward Lake Auburn. Even though P moves slower in the subsurface compared to other wastewater constituents (Robertson, 2021), the high speed of groundwater flow in this area means that septic systems located on the aquifer are more likely to be responsible for more rapid P transit to the lake. The hypothesis that these septic systems may load P to Townsend Brook and subsequently Lake Auburn would require field studies and ground-truthing.

Vadose Zone Texture and Depth to Limiting Factor of Septic System Drain Fields

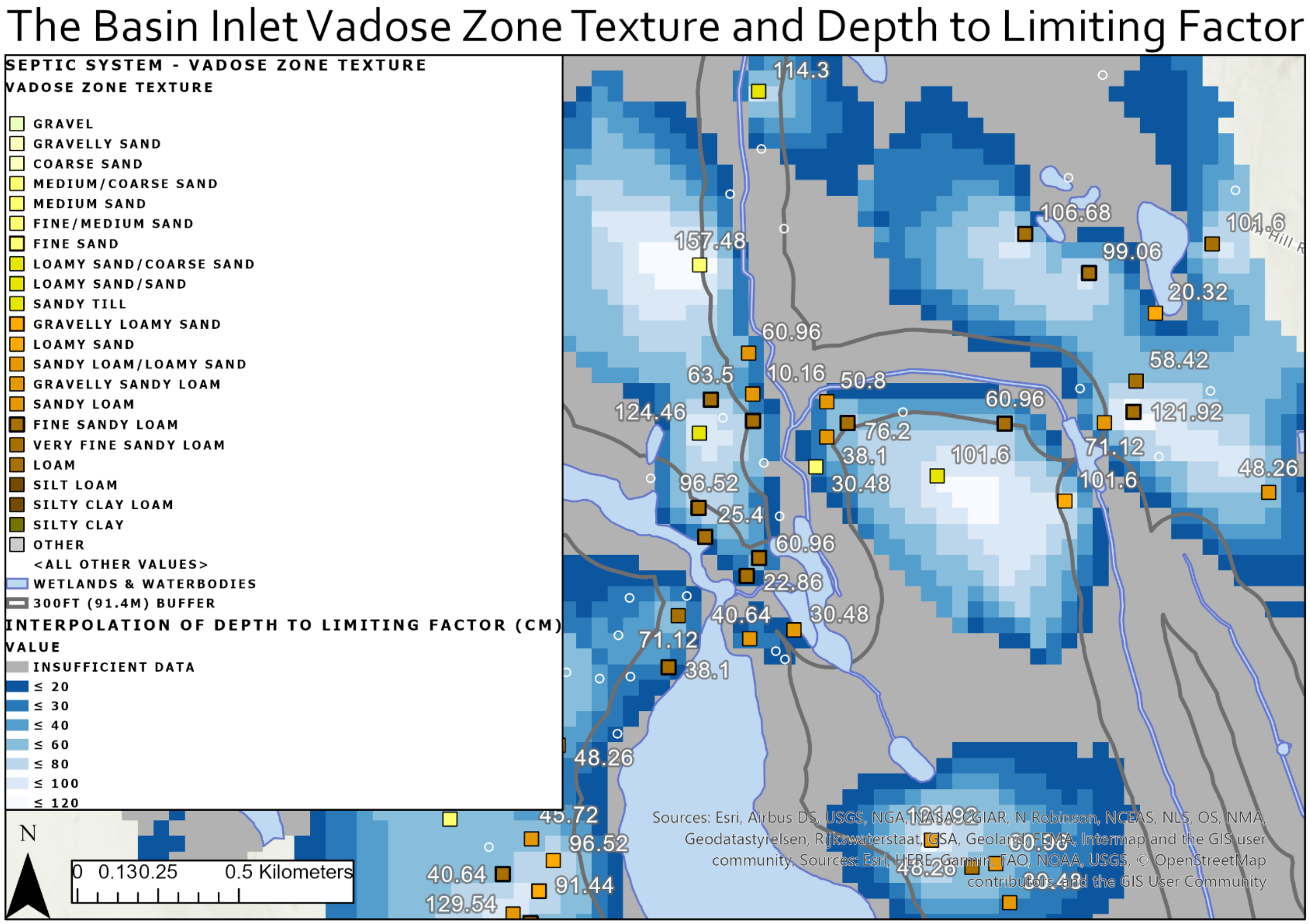


Map 4.4: Depth to limiting factor interpolated from site evaluations and dominant soil texture of the vadose zone. Darker colored squares represent finer-textured soils. Only septic systems with permits and site evaluations are shown. Systems without a site evaluation are marked with a white open circle.

The Basin Hydrologic Soil Group and Septic System Location

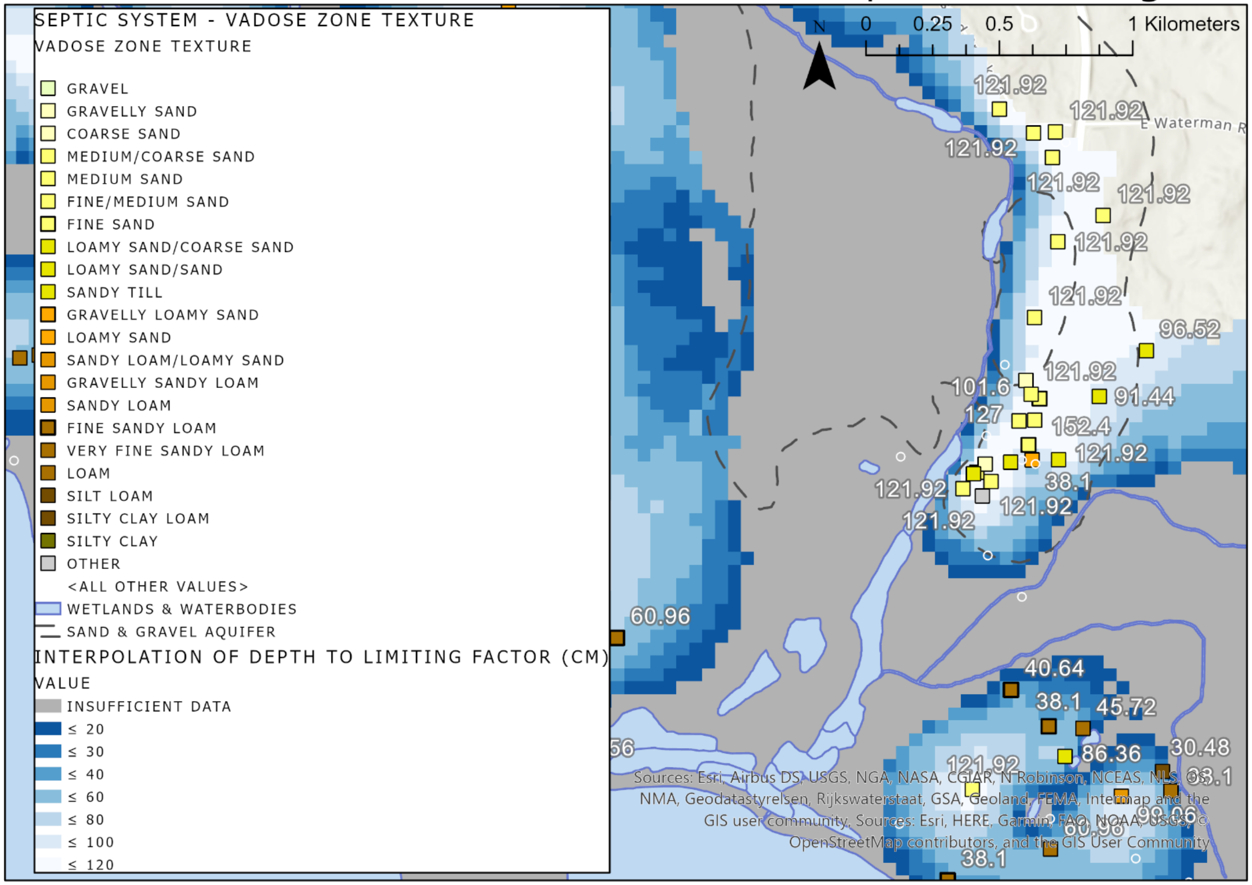


Map 4.6: Hydrologic soil group from SSURGO soil survey data around the Basin inlet. Hydrologic soil groups identify the runoff potential of soils based on soil texture, structure, and groundwater depth, with group A having the lowest runoff potential and group D or dual hydrologic soil groups having the highest runoff potential.



Map 4.5: Depth to limiting factor interpolated from site evaluations and vadose zone soil texture as identified by site evaluators around the Basin inlet. Only systems with site evaluations are shown, and interpolation only uses those data. Systems without site evaluations are shown as an open circle. Depth to limiting factor (cm) is displayed. Many systems are within a 300 ft (91.44 m) buffer.

Townsend Brook Vadose Zone Texture and Depth to Limiting Factor



Map 4.7: Depth to limiting factor interpolated from site evaluations and vadose zone soil texture. Value of soil depth (cm) is displayed. Many septic systems are located on the sand and gravel aquifer that feeds Townsend Brook, the second-largest contributor of waster and P to Lake Auburn. Only systems with site evaluations are displayed; systems without site evaluations are shown as an open circle.

4.1.5 Other Soil Characteristics

Result: Only 13 of the 230 systems with permits have bedrock as the limiting factor. Most systems have groundwater as the limiting factor, though site evaluators' examination of soil pits revealed that, below the seasonal high water table, a restrictive layer was usually also found. The most common soil texture was fine sandy loam, which occurred at 62 of the 224 assessed locations (27.6%). Site evaluators observed more variation in soil texture and depth to the limiting factor than was visible in soil survey data (Appendix E). No Munsell color was provided for pits at site evaluation, but few (27 of 224) had colors described to be reddish or orange. These were mostly described as "reddish brown," "orange brown," "red brown," or "reddish.".

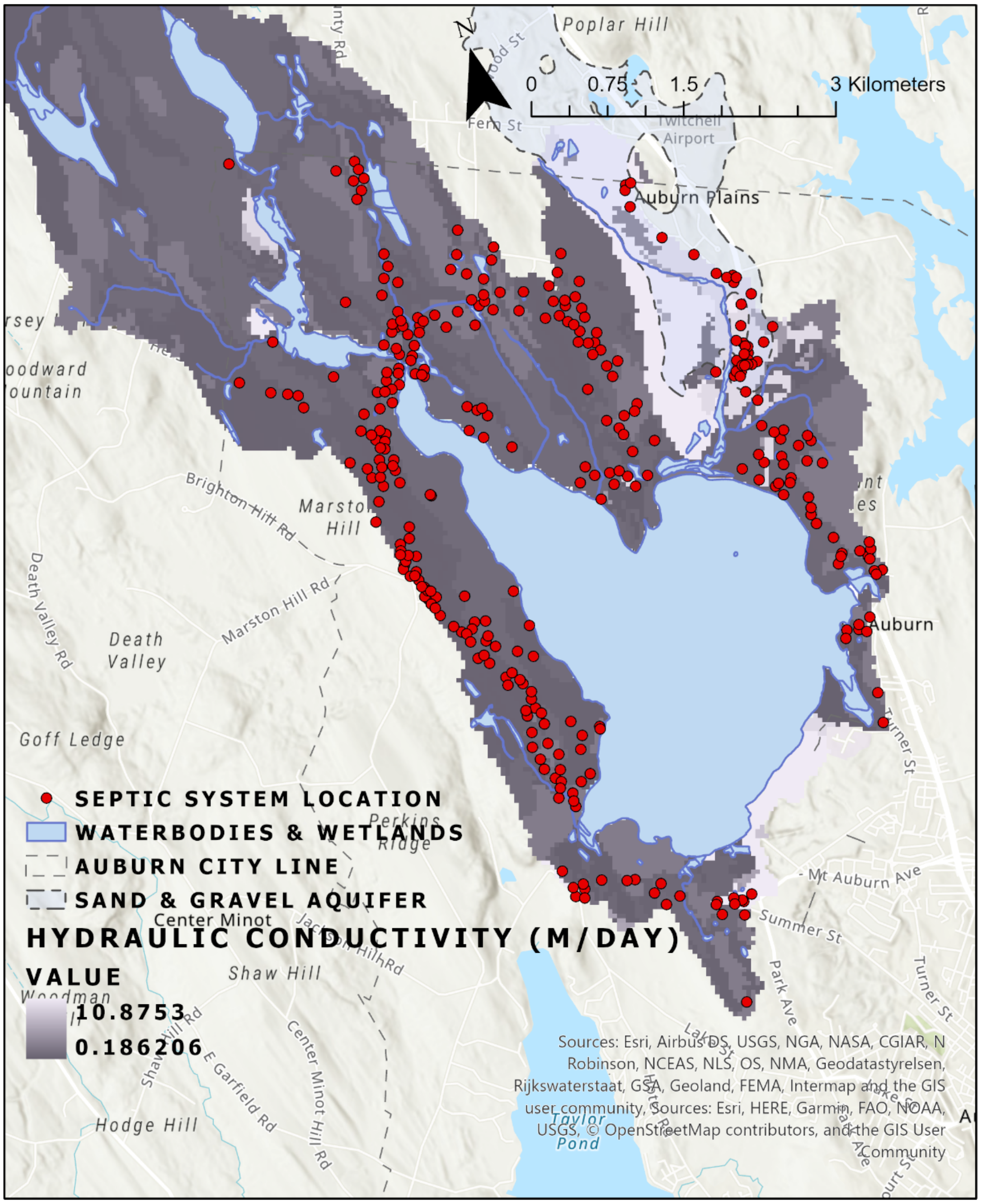
Interpretation: The installation of septic systems on fractured bedrock is a major concern because it provides a preferential flow pathway for effluent that transports wastewater, circumventing treatment (Fetter, 1993). However, this is not likely to be a widespread issue in Auburn since few systems within the Auburn portion of the watershed are located on bedrock. It is unknown if this bedrock is fractured. It may be worthwhile to investigate these few systems to understand whether they are a P loading risk, because wastewater from these systems may receive very little treatment.

The majority of systems appear to be on shallow subsoils with a restrictive layer and perched water table, which may pose a higher P loading risk to nearby water bodies by allowing for effluent to move laterally once reaching the restrictive layer (Gilliom & Patmont, 1983; Day, 2004). However, most of the research on septic systems and P plumes relates to sand aquifers

(Harman et al., 1996; Robertson, 2019), and the fate of P in systems on top of seasonally perched water tables is less widely understood (May, 2004). For soils in the watershed generally, most are of a texture with reasonable capacity to retain P (McCray et al., 2005), suggesting that shallow soil depths are likely more problematic. Site evaluations usually reveal a shallower depth to limiting factor than predicted by soil survey data, emphasizing the role of site evaluations for septic system eligibility.

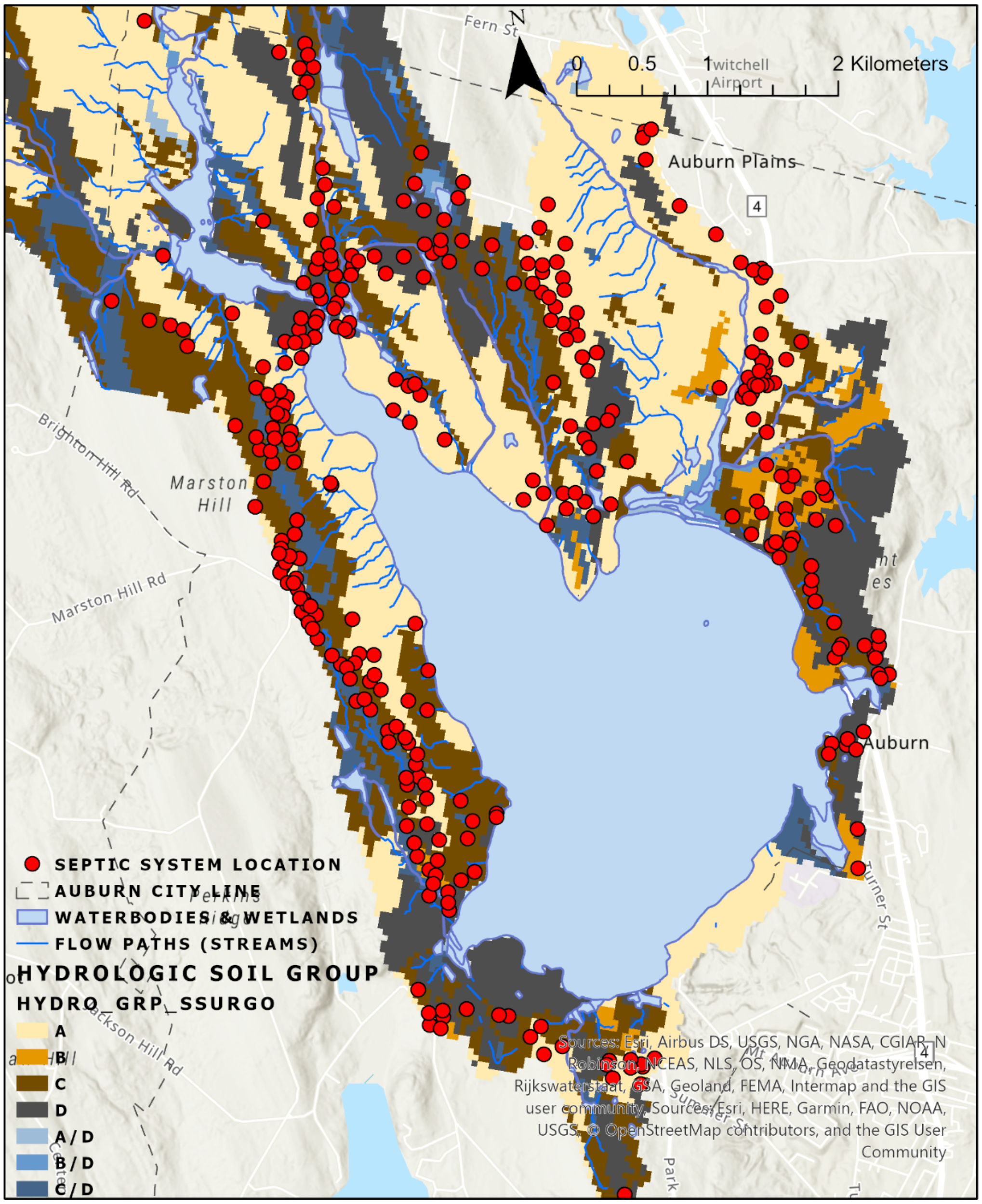
The 27 septic systems on soils that are red or orange in color likely have high P retention capacity because these soil colors typically signal the presence of sesquioxides, which are composed of Fe/Al oxides and hydroxides in short-range-order crystalline materials (Singer & Munns, 2006). Because these metals are so involved in P retention processes such as or phosphate adsorption to Fe/Al oxides/hydroxides or mineral precipitation to produce solid compounds such as variscite, strengite, or vivianite (Robertson, 2021), soils with high levels of Fe/Al often have high P retention capacity (Borggaard et al., 1990; Zhang et al., 2005). Therefore, drain fields with orangish or reddish soil colors are likely able to retain more P from wastewater than other septic drain fields in the watershed (Lusk et al., 2017). The limited observation of these soil colors may suggest that soils within the Auburn portion of the Lake Auburn watershed are low in Fe and Al, though field sampling and laboratory analyses are needed to confirm this.

Hydraulic Conductivity and Septic System Locations in Auburn



Map 4.8: Hydraulic conductivity (m/day) in the watershed from the SSURGO soil survey database. A high density cluster of septic systems exists on the sand and gravel aquifer which has high hydraulic conductivity.

Lake Auburn Watershed Hydrologic Soil Group and Septic System Location



Map 4.9: Hydrologic soil group from SSURGO database and overland flow paths. Systems of soils with high runoff potential (D, A/D, B/D, C/D) located near flow paths (streams) may contribute P from a surface pathway when they fail.

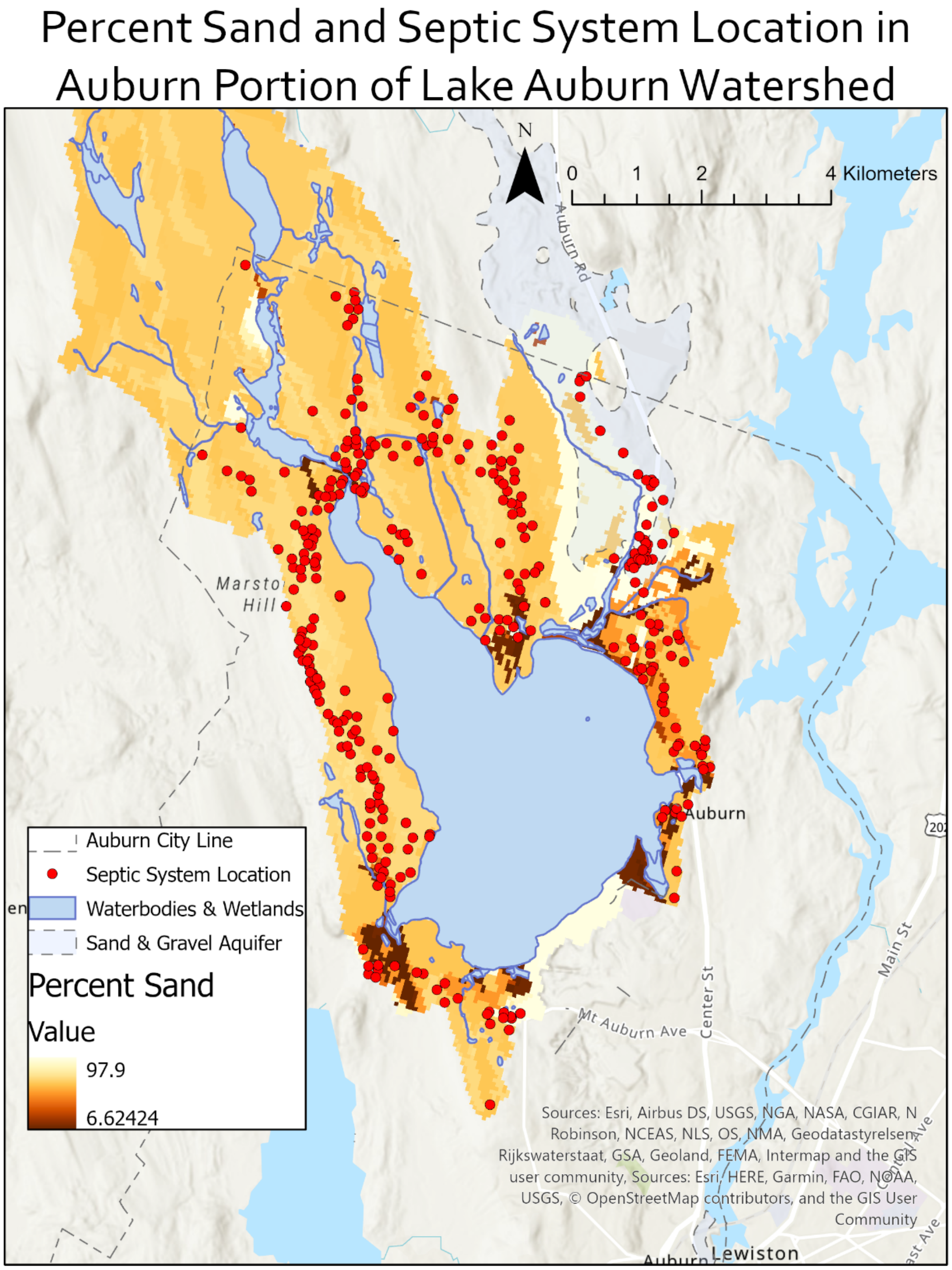


Figure 4.10: Percent sand from the SSURGO weighted means dataset and septic system location in the Auburn portion of the Lake Auburn watershed. The soils in the watershed have a high percentage of sand with the exception of a few areas. High percent sand may impact P sorption capacity.

4.2 Quantitative Analysis

4.2.1 Export Coefficient Approaches

Result: All export coefficient methods quantify P loading from septic systems in the Auburn portion of the Lake Auburn watershed only. Within this area, there are 71 septic systems within 300 ft (91.44 m) of Lake Auburn and its tributaries (Map 4.11). The buffer method, quantifying P loading to water bodies using just these systems, suggests that septic systems contribute 30.9 kg-P/year (Equation 1; Table 3.1). The buffer method was also modified to account for differing soil characteristics (Table 3.1), which increased the projected P load to 41.5 kg-P/year. These systems were mostly in the area surrounding the Basin inlet.

Methods quantifying P loading from the whole watershed rather than just the 300 ft (91.44 m) buffer include a transport coefficient (Equation 2; Table 3.1), and suggest that 35.4 kg-P/year are contributed to Lake Auburn from septic systems within the city lines. When this approach is modified to account for soil types and groundwater depth, this number increases to 44.4 kg-P/year. Furthermore, additional loading from system failure is accounted for, the whole-watershed calculation that considers soil variability suggests that P loading from septic systems may be 63.9 kg-P/year (Figure 4.1). Of this total, 20.7 kg is attributed to failing systems, while the remaining 43.2 kg would come from properly working systems discharging through the subsurface and groundwater pathways (Figure 4.1).

Interpretation: Findings from the buffer method can be compared with the findings of CEI (2010) who used similar methods and yielded a calculation of 120.7 kg-P/year. Because they calculated the P load from septic systems for the entire watershed, while this study only focuses on the Auburn portion of the watershed, it is impossible to compare them directly. However, the fact that the CEI (2010) estimate is so much higher than the buffer method estimate of 30.7 kg-P/year suggests either that different streams were used to create the buffer, or more P loading from shoreline septic systems occurs in the upper-watershed towns than in Auburn. Both possibilities are bolstered by the fact that CEI (2010) identified 287 residences within 300 ft of the lake or a tributary, whereas only 71 were located in this study. If methodological approaches were similar, this suggests that shoreline septic systems are likely a lesser issue in the Auburn portion of the watershed than they are in the upper-watershed. In the Auburn portion of the Lake Auburn watershed, most of the systems within 300 ft (91.44 m) of a water body surround the Basin inlet, many of which discharge into the Lake Auburn shoreline rather than the Basin, and the proximity of these systems to the lake may pose a P loading issue in this area (CEI, 2010; FB Environmental, 2021).

Expanding the scope of the estimation by using the "whole-watershed method" demonstrates that the loading estimate increases if calculations include contributions from systems outside of the 300 ft (91.44 m) buffer. This is a reasonable assumption as plumes have been noted to migrate over 90 m in other studies (LeBlanc 1984; Robertson et al., 2019; Rakhimbekova et al., 2021).

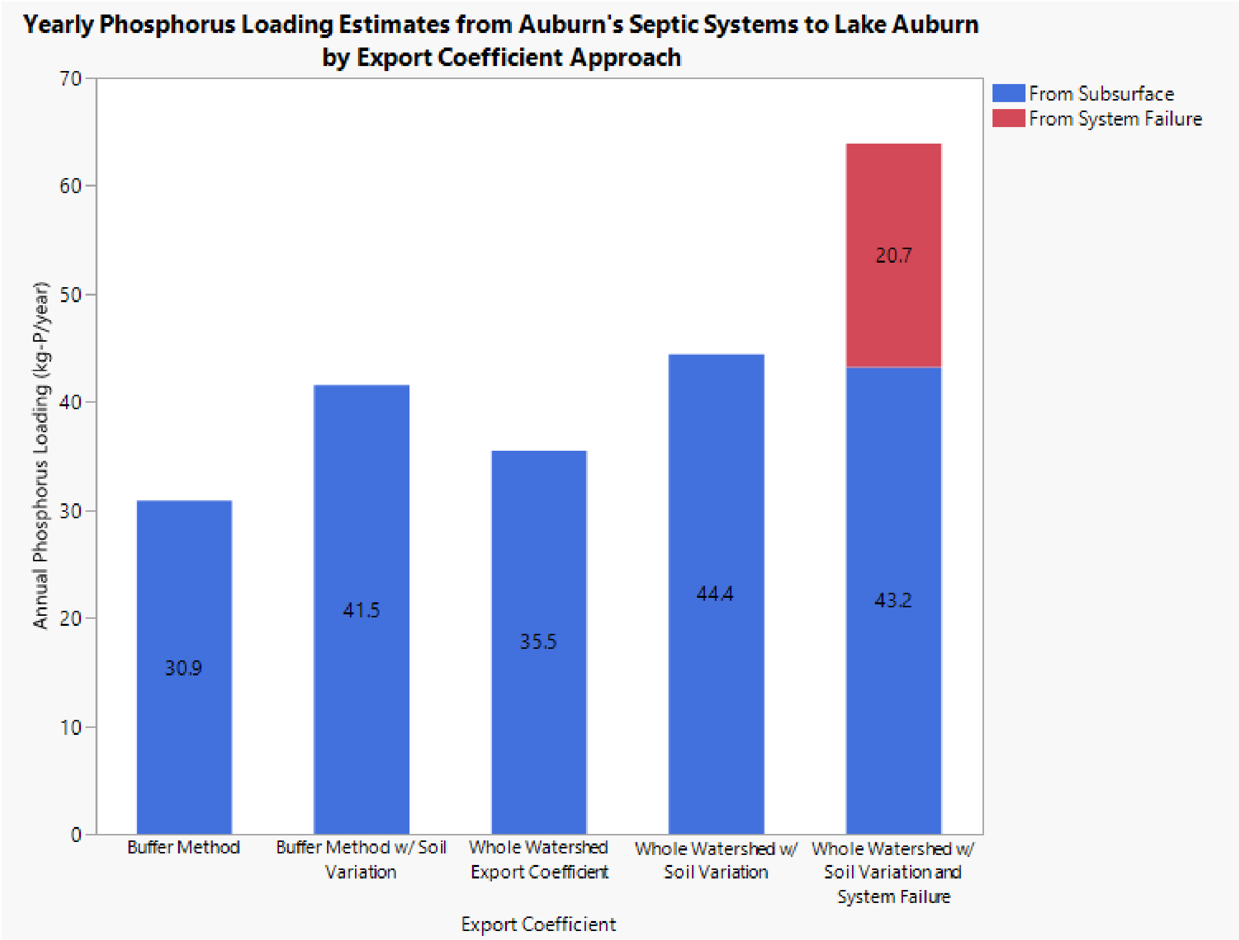
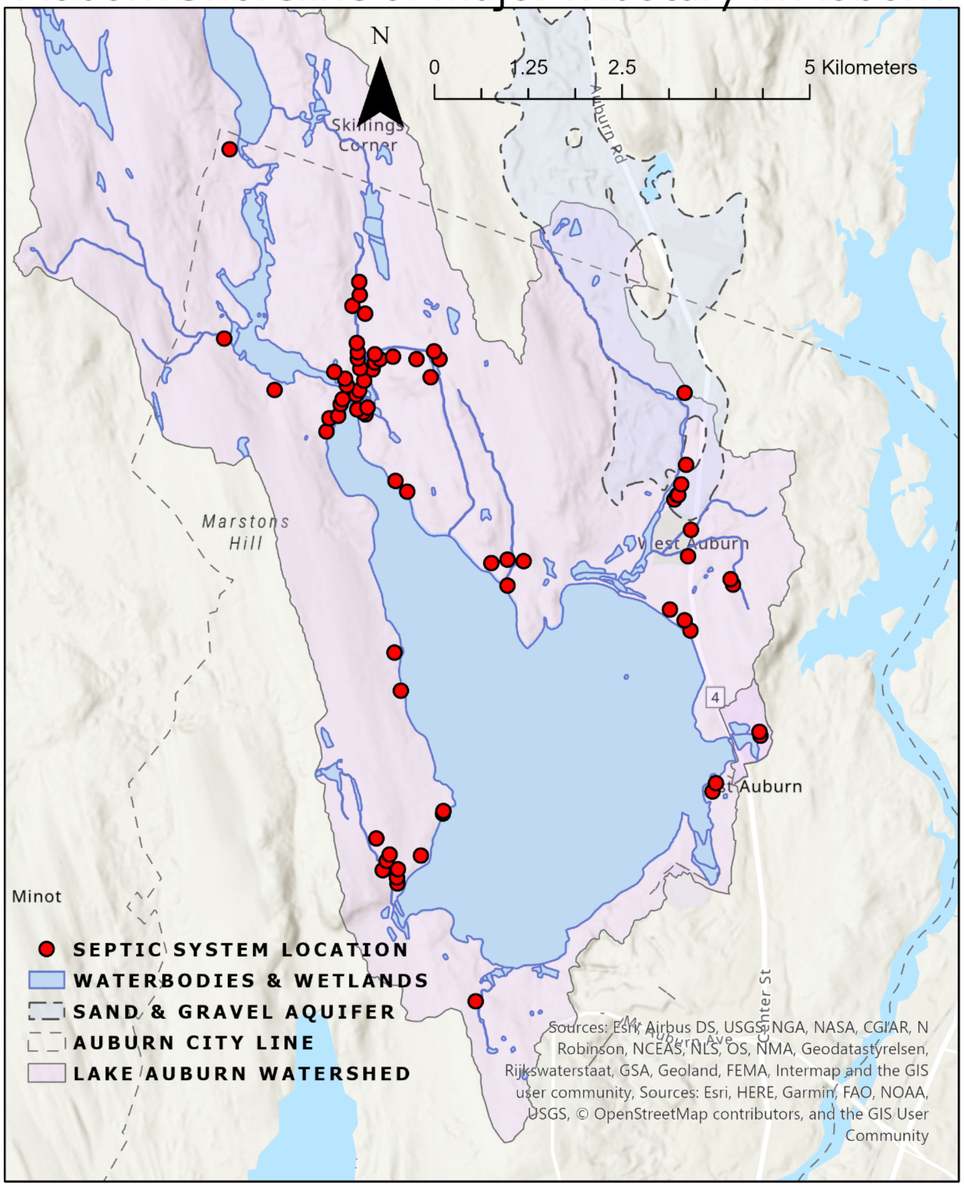


Figure 4.1: Phosphorus loading (kg-P/year) for 2023 as estimated using a variety of methods.

Modifying both the buffer method and the whole-watershed method to account for soil variability within the Lake Auburn watershed demonstrates that shallow vadose zone depths and coarse-textured soils may lead to higher P loading estimates since these soils are less effective at P treatment (Reide Corbett et al., 2002). Shallow vadose zone depths generally pose a larger issue than coarse-textured soils across the watershed; in the loading estimate, 202 of 325 systems use the attenuation factor for shallow subsoils, while 83 systems attenuate less P due to coarse-textured soil. These two soil characteristics that lead to poor P retention are widespread throughout the watershed, and this is reflected in higher loading estimates when soil characteristics are considered..

Accounting for P loading from failing systems demonstrates that hydraulic failure may be a large contributor to the yearly P load from septic systems (Figure 4.1). Unlike subsurface flow, contributions from failing systems can be considered to be more “instant,” likely reaching Lake Auburn within the year, while subsurface flow from properly functioning systems may take years to reach the shoreline (Roy et al., 2017). This model scenario suggests that 30% of the total P load from septic systems to Lake Auburn may come from failing systems. Though none of these methods account for the proposed ordinance change, they do highlight shallow groundwater and failing systems as potentially large sources of P loading to Lake Auburn. The proposed ordinance change would improve systems placed on sites with shallow depths to groundwater by introducing imported media (Bouma et al., 1975). However, this would only occur as the systems are replaced, which may take decades.

Septic Systems within 300ft (91.4m) of Lake Auburn Shoreline or Major Tributary in Auburn



Map 4.11: Septic systems within 300 ft (91.44 m) of Lake Auburn or major tributaries. Most cluster around the Basin inlet. Major tributaries were determined from stream layers used by Gundersen (2020) and from the USFWS Wetland Inventory.

4.3 Temporal Model

4.3.1 Model Results

Result: The temporal model (Equations 4-10, Table 3.2) was able to approximate P loading from septic systems at long time scales and returned different results based on the retardation factor (R_f) used. Using $R_f = 20$ leads to higher loading estimates for all years, while $R_f = 100$ estimates P loading to be lower (Figure 4.2). Regardless of retardation factor, the model suggests that P loading to waterways increased steadily after 1900 until the phosphate detergent ban in 1973. After 1973, P loading temporarily decreased, reaching relative minima 3-17 years after the ban before continuing to rise again (Figure 4.2). Looking at the highest estimate of loading using $R_f = 20$, it is clear that P loading is simulated to fluctuate within the overall rising trend depending on when effluent from various systems reaches the lake or a tributary (Figure 4.3).

After the last septic system begins operating in 2024, no new development is simulated to occur in the model, meaning no new septic systems begin operating past this year. In spite of this, for both values of R_f, the model simulates increasing P loading for the remaining 76 years of the time series, reaching a maximum one year before the simulation ends (Figure 4.2). At the end of the simulation, the estimated P load for 2100 is 85.1 to 140.3 kg-P/year, bounded by the $R_f = 100$ and $R_f = 20$ scenarios, respectively, an increase of 17 to 20 kg-P relative to the 2023 estimate for this model.

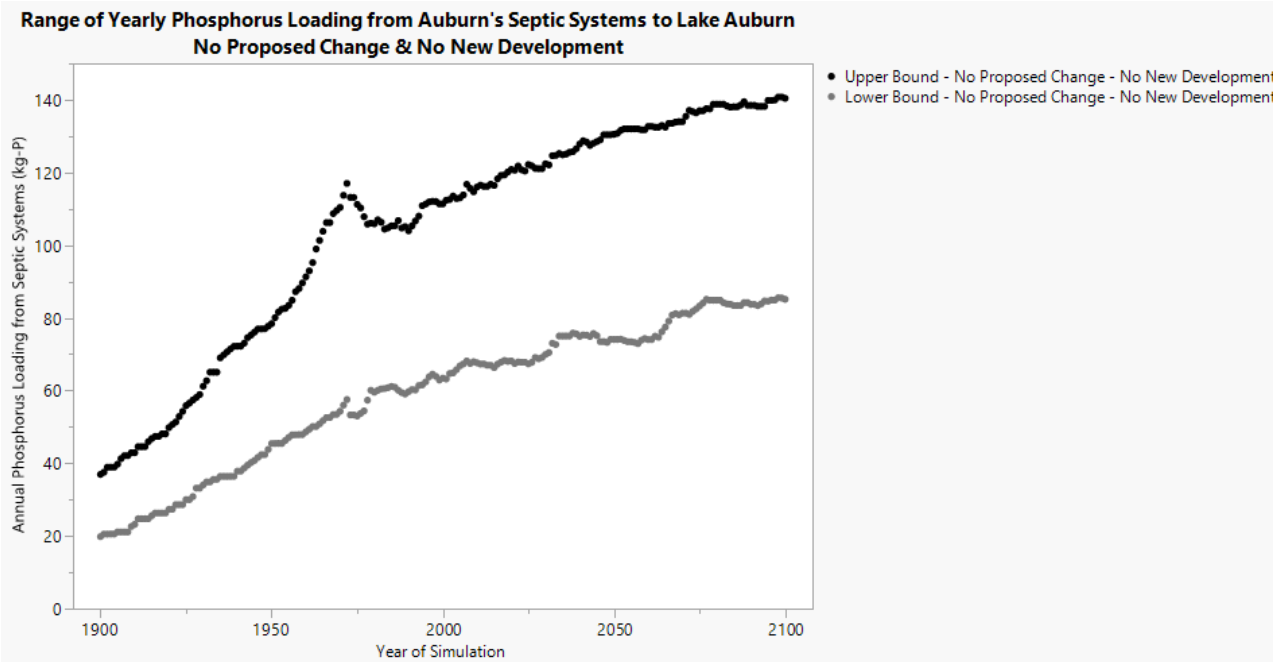


Figure 4.2: Temporal model (Equations 4-10; Table 3.2) results for end-member retardation factors (Rf), assuming no new development and no proposed ordinance change. The lower curve reflects transport to the lake assuming slow transport of P in groundwater (Rf=100), while higher P loading occurs if the soils under the septic systems transport P more quickly P (Rf=20) P loading is estimated to increase from 2024-2100 in the absence of new development, demonstrating that septic systems may pose a legacy P issue.

Interpretation: Septic systems within the Lake Auburn watershed create a legacy P issue. The simulation demonstrates that, even in the absence of new development, P loading may still increase by 17 to 20 kg-P/year from 2023 to 2100 because plumes from distant systems continue to reach the shoreline with each year (Figure 4.2). Septic systems in the Lake Auburn watershed may not begin contributing P to Lake Auburn for many years after they are installed, which may cause P loading from septic systems to rise in the future even if development halts, consistent with other studies (Sinclair et al., 2014; Roy et al., 2017; Oldfield et al., 2020a). The legacy effect is evident between 2000-2020 in the simulation, as there were more new residences in the 2000s (26 new homes from 2000-2009) than in the 2010s (4 new residences from 2010-2019), but loading rates do not immediately reflect these changes (Figure 4.3, 4.4). The mismatches between P loading increase over time and development patterns is explained by the proper retention of P within a septic drain field and the retardation of phosphorus in groundwater (Robertson et al., 1998; Oldfield et al., 2020a), delaying the arrival of P plumes from older systems at the shoreline. These phenomena are also responsible for the continued increase in P loading through the end of the simulation even in the absence of new development (Figure 4.2).

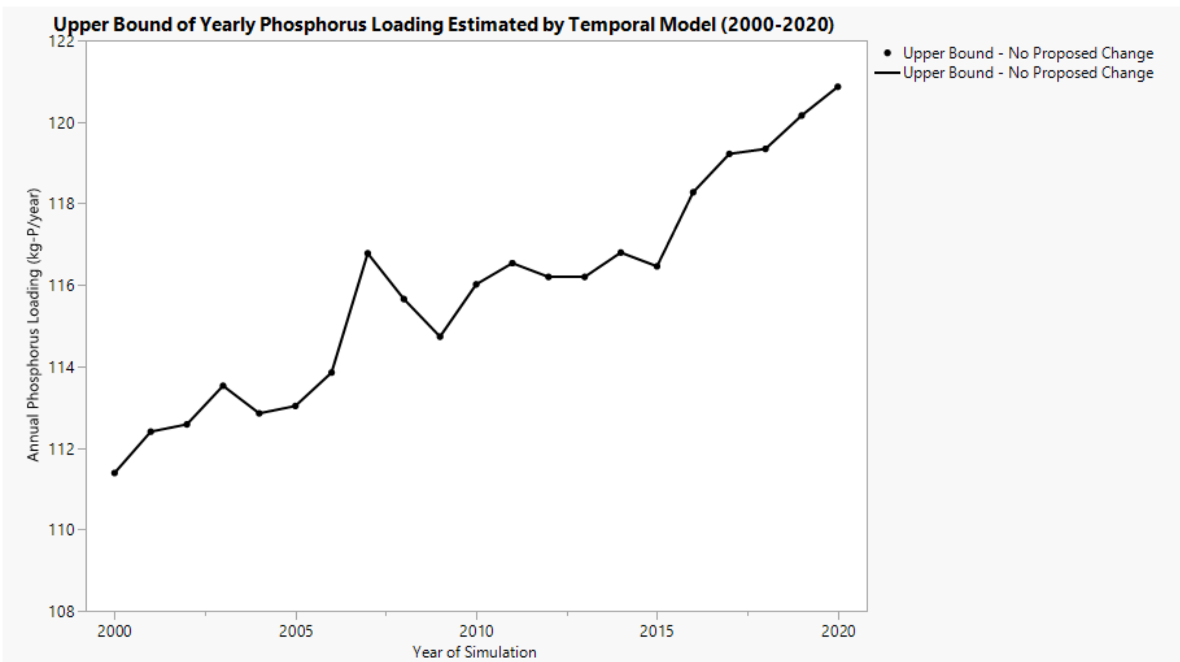


Figure 4.3: Upper bound (retardation factor, Rf=20) of annual P loading between 2000-2020 as projected by the temporal model. The temporal model estimates that annual loading fluctuates due to the long-term effect of the 1973 phosphate detergent ban, which has its effects delayed due to the retardation of phosphorus in groundwater. Despite the fact that there was minimal development between 2010 and 2020, a P loading spike is simulated at the end of the decade due to the legacy effect.

The 1973 phosphate detergent ban causes a noticeable decline shortly after the policy change is implemented (3-17 years) (Figure 4.2), similar to model simulations by Schellenger & Hellweger (2019). The immediacy of the decline is mostly due to a reduction of P from failing systems and systems closest to water bodies have begun discharging lower P concentration groundwater. Many more years are required for the effect of the 1973 phosphate ban to substantially decrease inputs from systems farther from the lake, as is visible in the slower growth of the yearly P load from 1973 to 2100 in spite of the increased development in the latter half of the 1900s (Figure 4.2). Effects of the change are most delayed in model simulations with a high retardation factor (Figure 4.2). This demonstrates the long time scales it may take for policy changes to affect septic system P loading.

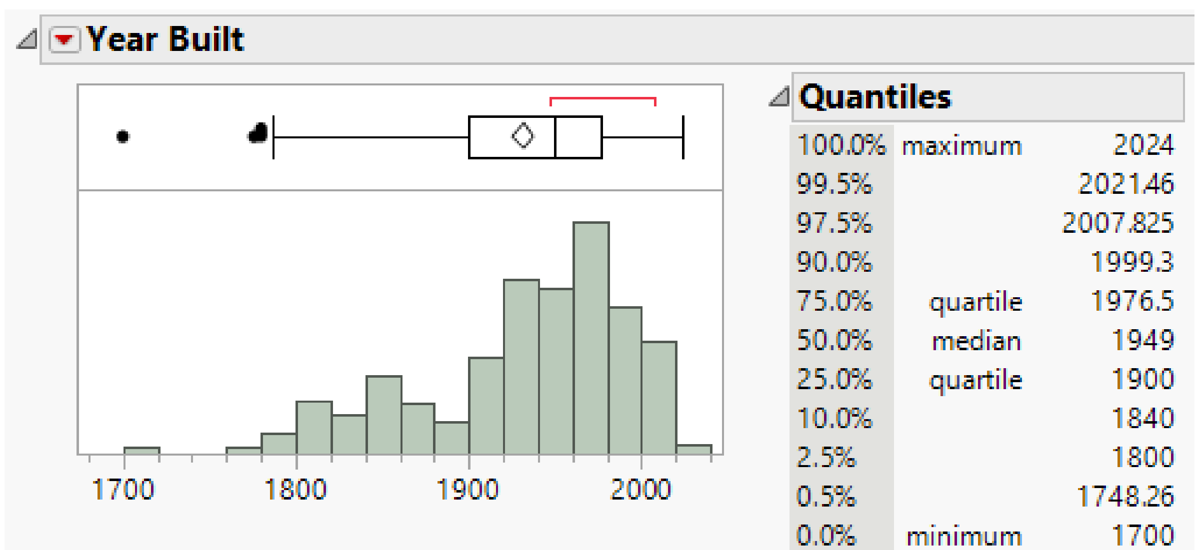


Figure 4.4: Distribution of the year each residence was built within the Auburn portion of the Lake Auburn watershed. Development spiked between 1920-1980 and slowed to a rate of 4 new residences per decade between 2010-2019.

4.3.2 Estimation for 2023

Result: The temporal model returns values of 67.8 and 120.8 kg-P/year for 2023 for $R_f = 100$ and $R_f = 20$, respectively. These estimates are between 106 and 189% of the whole watershed with soil variation and system failure method, on which the temporal model was based. The temporal model also provides estimates far higher than other export coefficient approaches (Figure 4.5).

Interpretation: Removing the transport coefficient from the export coefficient methods and simulating P transport directly through the temporal model results in markedly higher P loading estimates in 2023. One of the factors most responsible for this discrepancy is the delay of P transport in groundwater (Bedient et al., 1994; Robertson et al., 1998), which means that high P groundwater from before 1973 is simulated to reach the Auburn shoreline in modern times. Another factor is that the transport coefficient used in the "whole-watershed method" was developed for a watershed in Canada (Oldfield et al., 2020b), which is unlikely to reflect field conditions in Auburn.

When the retardation factor is lower, more P plumes from systems with large setbacks are predicted to reach the shoreline. For example, in 2023, there are 219 (67%) P plumes contributing phosphorus to Lake Auburn through the subsurface in 2023 for $R_f = 20$. For $R_f = 100$, there are 99 P plumes that have reached the shore, or 30% of all operating systems.

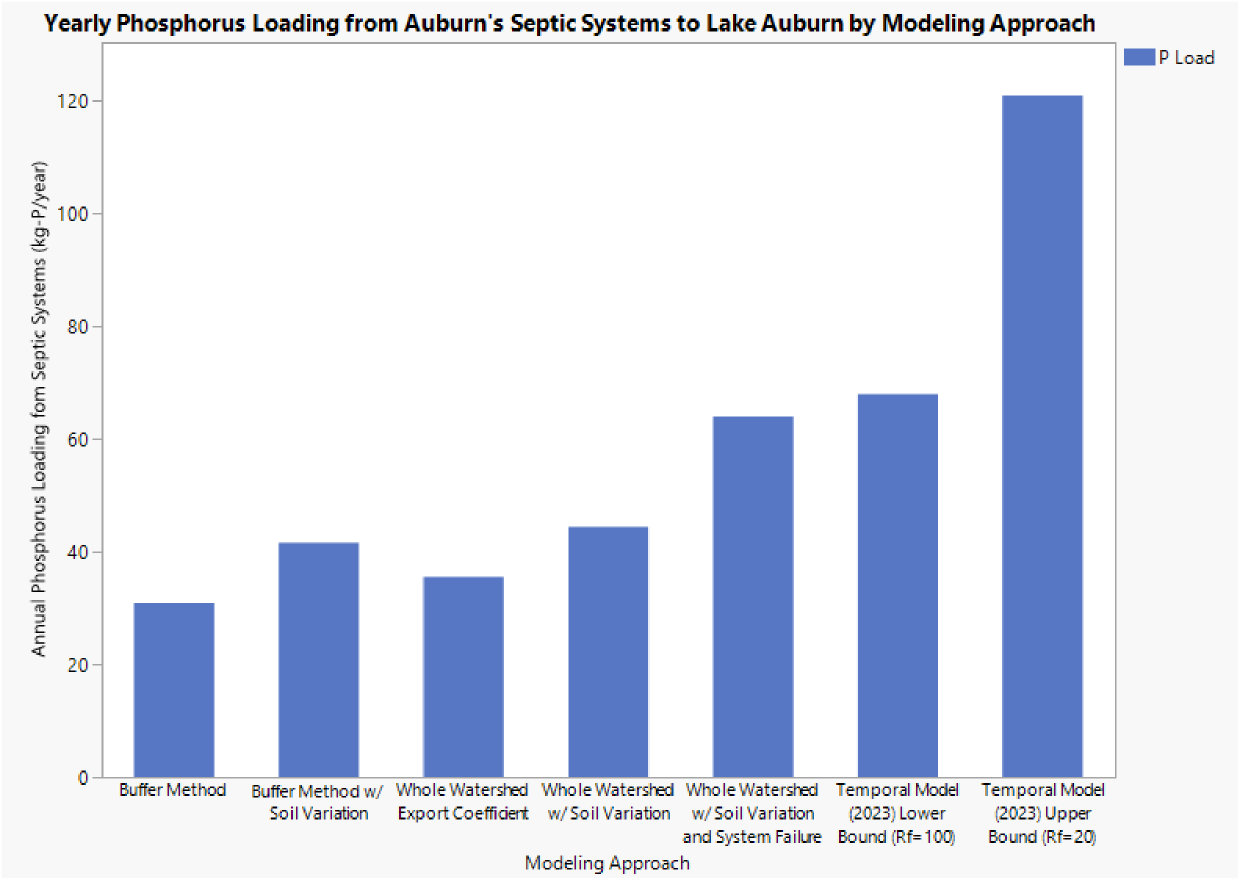


Figure 4.5: Loading estimates for 2023 using export coefficient methods and the 2023 values from the temporal model. Differing values of Rf produce results that are higher than other estimates for all values of Rf.

Comparing these numbers to the buffer method, which assumes that only the 71 systems within the 300 ft (91.44 m) buffer contribute P, or other methods using a transport coefficient, which assumes that only 24% of effluent reaches the shoreline, it is clear that the number of P plumes from septic systems that are predicted to discharge to a water body has a big influence on the estimated P load. The temporal model with the lower retardation factor predicts that a large number of residences built pre-1900 (Figure 4.4) outside of the 91.44 m buffer are likely contributing P to the lake in spite of large setback distances, assuming plumes move between 1-2 m/year (Robertson et al., 2019). Additionally, because the model includes the hydraulic conductivity of the surficial aquifer, the high hydraulic conductivity of the sand and gravel aquifer located near Townsend Brook allows for P transport from septic systems outside of the 300 ft (91.44 m) buffer zone.

4.3.3 Integration of Proposed Ordinance Change

Result: Integration of the proposed ordinance, assumed to take effect in 2023, causes changes to the simulation beginning in the year 2024. In general, this leads to a decrease in P loading to the lake as older systems on shallow soils are replaced with newer systems with greater vadose-zone retention (Figure 4.6). However, large reductions of P loading do not occur instantaneously. For example, if $R_f = 20$, which assumes that phosphorus transport from septic systems is quick, loading from septic systems does not decline by 10 kg-P/year (less than 10% of the 2023 load for this scenario) until 2043, 20 years after the ordinance change is assumed to happen. By the end of the simulation in 2100, the annual P load is estimated to decrease by 28.4 kg-P/year. If higher retardation of phosphorus is assumed ($R_f = 100$), there is only an estimated 5.8 kg-P/year reduction from the 2023 value by 2100, and the overall rate of decline is small (Figure 4.6).

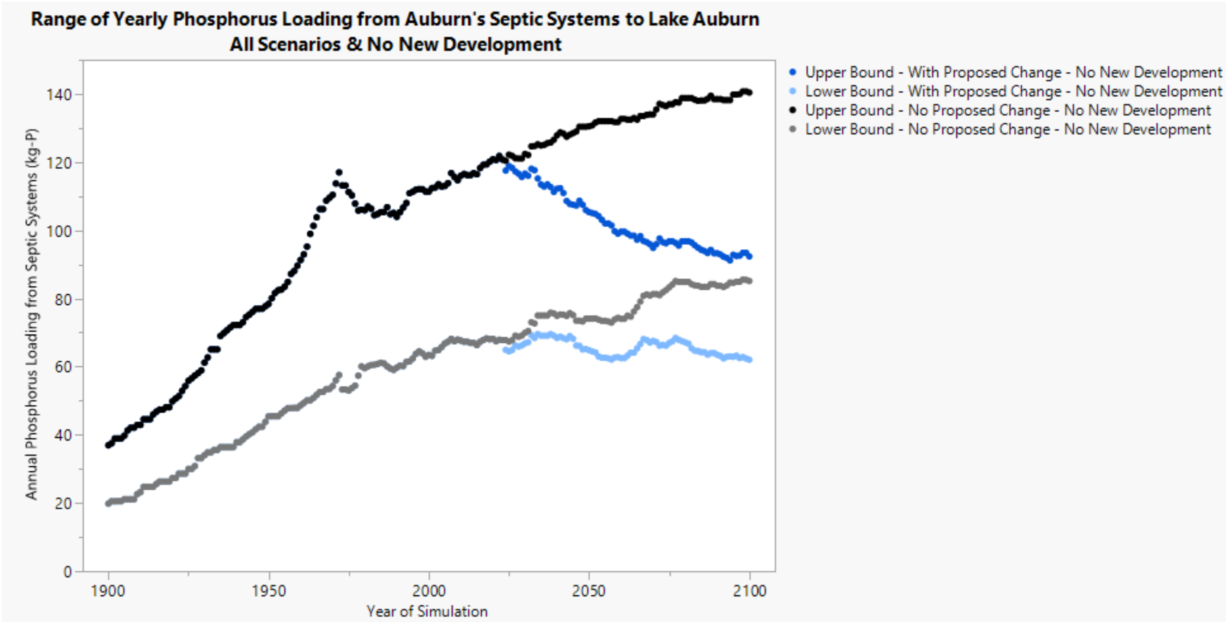


Figure 4.6: Temporal predictions of annual P loading from septic systems to Lake Auburn, incorporating the proposed septic ordinance change beginning in 2023. For a low retardation factor (Rf=20), P load reductions are larger and occur quicker than if a higher retardation factor (Rf=100) is used. Reductions are not immediate and do not exceed 10 kg-P/year until two decades after the ordinance change occurs, demonstrating the legacy effect of septic systems. The model assumes each system fails and is replaced after it operates for 29 years, with P loading from failure distributed across the entire 29 year period (Section 3.1.4).

Interpretation: Though the proposed ordinance change eventually leads to reduced annual P loading declines, there is little immediate effect of the proposed policy change on the P load. The retardation of phosphorus in groundwater delays the effect of the proposed change; short-term changes result from the replacement of septic systems on low permeability soils that are more susceptible to failure. In the long term, wastewater from after the change takes many years to reach the shoreline, leading to estimated decreases by 5.8 to 28.4 kg-P in the year 2100. This change in septic system loading is smaller than the recommended P loading reduction for the whole watershed from FB Environmental (2021), who presented various loading estimates for the entire watershed and suggested that the annual P load be reduced to 900 kg-P/year in order to maintain water quality. Because it takes years for systems to be replaced, and then additional time for the effect of improved wastewater treatment to be observed, the potential effect of the proposed ordinance change on P loading may not be substantial until many decades have passed. Therefore, the proposed change to the septic policy is unlikely to lead to immediate and substantial P loading reductions.

Because P transport in groundwater is delayed, it may be likely that the implementation of any septic ordinance change could have the quickest effect if immediate efforts focused on septic systems that are failing (old or on low permeability soil), on steep slopes, located close to the shoreline, or on the aquifer. These septic systems are most likely to have a surface connection to Lake Auburn or contribute P to the lake through the subsurface the quickest when properly functioning. The identification and replacement of failing systems may also reduce P loading.

The temporal model's reliance on literature values, soil survey data, and many modeling assumptions that simplify hydrologic processes (explained in Chapter III and explored further in Appendix D) may limit the precision of these numerical P loading estimates. However, the trends observed over the course of the simulation reflect the scientific ideas discussed in Chapter II, such as the retardation of phosphorus in groundwater and that septic systems over 300 ft (91.44 m) from the shoreline may contribute P to Lake Auburn, especially if they are located in areas with high hydraulic conductivity (Harman et al., 1996; Rakhimbekova et al., 2021).

4.4 Discussion of Land Use Implications

The proposed ordinance change may have consequences on land use that could lead to higher P loading and have the potential to completely nullify the P load decreases associated with improved wastewater treatment. As the ordinance would allow for mounded systems in the Auburn portion of the watershed, it has the potential to reduce the requirement of 36 inches (91.44 cm) to the limiting factor to a 12 inch (30.5 cm) requirement. Therefore, areas that currently are ineligible for a septic system would become "suitable," and residences could be placed on those lots. City records include 14 septic system applications that were rejected within the past 20 years because they did not have 36 inches to the limiting factor (located during data collection). There are likely more that are not on file, considering that municipal records are incomplete and that over 65% of permitted systems do not reach the 36 inch standard. All of the soils on lots with rejected applications had at least 12 inches to the limiting factor, indicating that they would be eligible for a septic system if the proposed ordinance was instated. Furthermore, it can be assumed that there is intent to build on these parcels considering that applications exist for these lots. These parcels of land are all along existing roads, meaning they would be the easiest to develop because their development would not require the extension of existing roadways.

Considering that the estimated decrease in yearly P loading from septic systems between 2023 to 2100 is, at a maximum, 28.4 kg-P/year, the effect of land based P loading from increased development may nullify these P load reductions. Previous studies and reports on Lake Auburn that quantify future phosphorus loading predict P loading increases well over 28.4 kg-P/year due to land use changes and climate change (CEI, 2010; Gundersen, 2020; FB Environmental, 2021).

The most compelling evidence for this comes from the CEI (2010) Lake Auburn Watershed Management Plan, which contains cumulative P loading estimates under different buildout scenarios. In particular, they compared P loading from a standard buildout analysis with another buildout analysis that considered a relaxation of the current septic standards, which would allow for development on land areas with less than 36 inches to the limiting factor (this is the same effect as the proposed ordinance and would only affect Auburn). They found that the yearly phosphorus load from the Auburn portion of the watershed increases by 56.7 kg-P/year from land use changes alone at full buildout; when additional P loading from new septic systems in Auburn was considered, this number increased to 158.3 kg-P/year. This evidence from CEI (2010) demonstrates that the land use consequences of any proposed change in the septic ordinance that does not also continue to restrict building where it has been previously restricted would nullify the estimated reduction of 28.4 kg-P/year from improving existing septic systems, and would instead lead to an increase in cumulative P loading (Figure 4.7). In short, relaxing the 36 inch (91.44 cm) requirement while allowing for the use of mounded systems would lead to better wastewater treatment in cases where existing systems have shallow restrictive layers, but land use changes that become possible with the relaxation of septic system standards would increase P loading to such a degree that there may be a net increase upwards of ~130 kg-P/year by 2100 (Figure 4.7).

In an effort to counteract the potential development made possible by the proposed septic ordinance, the City of Auburn proposed a zoning change for many residential areas within the watershed to require larger lot sizes and road frontage for a new residence (City of Auburn, 2022a). Even though most of the parcels with rejected septic applications are small and would not comply with more stringent zoning requirements, they would be considered "legally nonconforming" because the parcel was formed before the zoning change (City of Auburn Ordinance Chapter 60, 2009). Therefore, they would all remain buildable. The changes modeled by CEI (2010) are also likely minimally affected by proposed zoning changes to reduce the allowed density of new development within the watershed because many areas that are ineligible for a septic system in the zone are on legally nonconforming lots, which are buildable

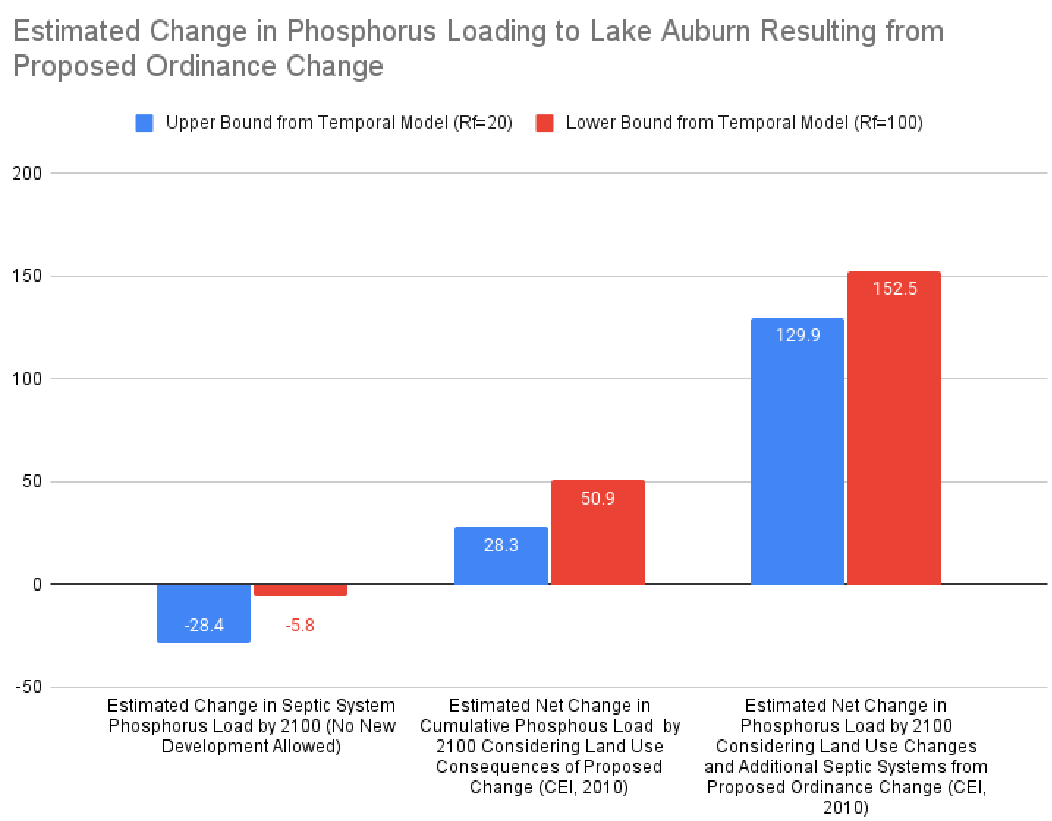


Figure 4.7: Estimated net change in cumulative P loading to Lake Auburn by 2100 considering the proposed ordinance change. Results from CEI (2010) are added to the lower and upper bounds from the temporal model, which demonstrate how land use changes allowed by proposed policies may cause a net P loading increase. The proposed ordinance change may reduce P loading only in the absence of new development within the Lake Auburn watershed.

Other studies on Lake Auburn support this finding. Using the Soil & Water Assessment Tool, Gundersen (2020) estimated an increase in P loading by 829 kg-P/year if the amount of watershed development anticipated by the City of Auburn (City of Auburn Ordinance Chapter 60, 2009) were to occur. This is well above the 28.4 kg-P/year maximum decrease in P loading estimated in this study with a change in the septic ordinance. Even studies that considered a proposed zoning change that would reduce the allowed density of residential development in the Lake Auburn watershed (City of Auburn, 2022a), found only a marginal (6 kg-P/year) decrease in the cumulative P load as a result of the combination of both septic and zoning policy changes (FB Environmental, 2022). However, other changes to the modeling approach were made between model runs, including removing additional development in the Agriculture and Resource Protection Zone (FB Environmental, 2022), which suggests that the P loading decrease predicted in the study may be too high (CEI, 2022). Therefore,without efforts to continue restricting development on lots that are currently ineligible for a septic system, the P contributions from land-use changes would exceed any reductions in P loading from improved retention in septic drain fields.

Chapter V: Conclusion

5.1 Summary of Major Findings

5.1.1 Watershed Characteristics and Potential Sources of Loading

- The area around the Basin inlet may be a phosphorus (P) loading hotspot because of the large number of systems within 300 ft (91.44 m) of the lake shoreline that also do not have at least 36 inches (91.44 cm) to the limiting factor. As a result, there may not be as much P retention in these soils (Mechtensimer & Toor, 2017).
- The Townsend Brook subwatershed has a high-density cluster of systems on sand and gravel deposits, which are the media of an aquifer, which feeds the second-largest inlet to Lake Auburn. The coarse-textured soils here are far worse at attenuating P than finer-textured soils (Carroll et al., 2005).
- 62.4% of septic systems in the Auburn portion of the Lake Auburn watershed with permits on file are installed on shallow soils. Of these, 19 of 146 systems on shallow soils have less than 12 inches (30.5 cm)of soil and would not have 36 inches (91.44 cm) to the limiting factor even if mounded systems were permitted. The remaining systems (215 of 234) could achieve 36 inches (91.44 cm) of vertical separation if mounded systems were permitted.
- Phosphorus loading estimates demonstrate that shallow soil depths, coarse-textured soils, and failing systems may be large contributors of P loading from septic systems in the Auburn portion of the Lake Auburn watershed. The use of different methods emphasizes that field studies are necessary to inform modeling efforts.

5.1.3 Legacy Phosphorus and Potential Effect of Policy Changes

- The full impact of septic system P contributions in the Lake Auburn watershed likely have not yet been observed due to the retardation of P in groundwater (Roy et al., 2017). In the absence of new development, P loading from Auburn's septic systems is projected to increase for the entire 76 year period of the model simulation after development ceases.
- Only in the absence of new development may policy changes decrease the P load from septic systems. Moreover, the impact of any change in septic policy may not be observed for decades due to the retardation of P in soils. For example, the proposed septic ordinance change was predicted to reduce P loading by less than 10% two decades after a policy change was simulated in the model.
- The proposed ordinance change may reduce P loading from septic systems by between 5.8 and 28.4 kg-P/year by 2100, but only in the absence of new development. However, should new development be permitted to occur, land use changes such as the loss of forested area, increased impervious surface, and the addition of new septic systems would

increase P loading (Withers & Jarvie, 2008). Incorporating analyses done by CEI (2010) suggests that the proposed change in septic policy could cause a net increase of 129.9 to 152.5 kg-P/year by 2100 when land use changes and additional septic systems are considered.

- Allowing the use of mounded systems would improve wastewater treatment in the 146 septic systems known to be on shallow soils. However, the cumulative effect is not substantial until years after each system is replaced, which may take decades. Model simulations show that the decrease in P loading may not be felt in full by 2100 because of the retardation of P in soils. Therefore, this measure alone cannot guarantee immediate and substantial reductions in P loading.

5.2 Recommendations for Future Research

- Organizing efforts to identify old septic systems that are failing and systems that do not have a site evaluation on file within Auburn may lead to replacements of systems that pose a P loading risk. It may also allow for a greater understanding of the septic systems within the watershed.
- Expanding the inventory of septic systems into the headwater towns of Turner, Hebron, Minot, and Buckfield would create a more complete database and would allow for analyses that encompass the entire watershed.
- A variety of information about soil characteristics and the transport of phosphorus from septic systems toward Lake Auburn in the Lake Auburn watershed would dramatically improve modeling efforts that quantify P loading. Investigating septic system plumes for various wastewater contaminants such as NO_3^-, PO_4^{3-}, and other constituents such as Na^+, Cl^-, boron, or electrical conductivity (specific conductance). Methods for these types of studies are laid out in various publications, and additional information is included in Appendix F.

Works Cited

Baer, S., Robertson, W., Spoelstra, J., & Schiff, S. (2019). Phosphorus and nitrogen loading to Lake Huron from septic systems at Grand Bend, ON. *Journal of Great Lakes Research*, *45*(3), 642–650. https://doi.org/10.1016/j.jglr.2019.03.003

Barrow, N. J. (1983). A mechanistic model for describing the sorption and desorption of phosphate by soil. *Journal of Soil Science*, *34*(4), 733–750. https://doi.org/10.1111/j.1365-2389.1983.tb01068.x

Beach, D. N. H., McCray, J. E., Lowe, K. S., & Siegrist, R. L. (2005). Temporal changes in hydraulic conductivity of sand porous media biofilters during wastewater infiltration due to biomat formation. *Journal of Hydrology*, *311*(1–4), 230–243. https://doi.org/10.1016/j.jhydrol.2005.01.024

Beal, C. D., Gardner, E. A., & Menzies, N. W. (2005). Process, performance, and pollution potential: A review of septic tank - soil absorption systems. *Soil Research*, *43*(7), 781. https://doi.org/10.1071/SR05018

Bean, G. M., Ransom, C. J., Kitchen, N. R., Scharf, P. C., Veum, K. S., Camberato, J. J., Ferguson, R. B., Fernández, F. G., Franzen, D. W., Laboski, C. A. M., Nafziger, E. D., Sawyer, J. E., & Nielsen, R. L. (2021). Soil hydrologic grouping guide which soil and weather properties best estimate corn nitrogen need. *Agronomy Journal*, *113*(6), 5541–5555. https://doi.org/10.1002/agj2.20888

Bedient, P. B., Rifai, H. S., & Newell, C. J. (1994). *Ground Water Contamination: Transport and Remediation*. PTR Prentice Hall.

Borggaard, O. K., Jdrgensen, S. S., Moberg, J. P., & Raben-Lange, B. (1990). Influence of organic matter on phosphate adsorption by aluminium and iron oxides in sandy soils. *Journal of Soil Science*, *41*(3), 443–449. https://doi.org/10.1111/j.1365-2389.1990.tb00078.x

Bouma, J., Converse, J. C., Otis, R. J., Walker, W. G., & Ziebell, W. A. (1975). A Mound System for Onsite Disposal of Septic Tank Effluent in Slowly Permeable Soils with Seasonally Perched Water Tables. *Journal of Environmental Quality*, *4*(3), 382–388. https://doi.org/10.2134/jeq1975.00472425000400030022x

Brady, N. C., & Weil, R. R. (2002). *The nature and properties of soils* (13th ed). Prentice Hall.

Bumgarner, J., & McCray, J. (2007). Estimating biozone hydraulic conductivity in wastewater soil-infiltration systems using inverse numerical modeling. *Water Research*, *41*(11), 2349–2360. https://doi.org/10.1016/j.watres.2007.02.040

Carmichael, W. W., & Boyer, G. L. (2016). Health impacts from cyanobacteria harmful algae blooms: Implications for the North American Great Lakes. *Harmful Algae*, *54*, 194–212. https://doi.org/10.1016/j.hal.2016.02.002

Carroll, S., Goonetilleke, A., Khalil, W. A.-S., & Frost, R. (2005). Assessment via discriminant analysis of soil suitability for effluent renovation using undisturbed soil columns. *Geoderma*, *131*(1–2), 201–217. https://doi.org/10.1016/j.geoderma.2005.03.022

Castronovo, S., Wick, A., Scheurer, M., Nödler, K., Schulz, M., & Ternes, T. A. (2017). Biodegradation of the artificial sweetener acesulfame in biological wastewater treatment and sandfilters. *Water Research*, *110*, 342–353. https://doi.org/10.1016/j.watres.2016.11.041

CDM Smith. (2013). *Diagnostic Study of Lake Auburn and its Watershed: Phase I*. Accessed 18 April, 2023. https://lakeauburnwater.org/?page_id=882

City of Auburn, Maine Office of Planning & Permitting (City of Auburn). (2022a). *Lake Auburn Staff Report Watershed Protection Updates RR to LDCR.* Accessed 15 April, 2023. https://www.auburnmaine.gov/CMSContent/Planning/071222/Lake%20Auburn%20Staff%20Report_%20Watershed%20Protection%20Updates%20RR%20to%20LDCR.pdf

City of Auburn, Maine Office of Planning & Permitting (City of Auburn). (2022b). *Staff Report Lake Auburn Watershed Protection Updates*. Accessed 15 April, 2023. https://www.auburnmaine.gov/CMSContent/Planning/04122022/Staff%20Report_Lake%20Auburn%20Watershed%20Protection%20Updates%20(1).pdf

City of Auburn, Maine Office of Planning & Permitting (City of Auburn). (2022c). *Workshop Septic Systems 4.12.22*. Accessed 17 April, 2023. https://www.auburnmaine.gov/CMSContent/Planning/04122022/Workshop%20Septic%20Systems%204.12.22.pdf

City of Auburn, Maine Ordinance Chapter 60. (2009). *Zoning*. Accessed 14 April, 2023. https://library.municode.com/me/auburn/codes/code_of_ordinances?nodeId=PTIICOOR_CH60ZO

Comprehensive Environmental, Incorporated (CEI). (2010). *Lake Auburn Watershed Management Plan*. Accessed 20 January, 2023. https://lakeauburnwater.org/?page_id=882

Comprehensive Environmental, Incorporated (CEI). (2022). *Peer Review of FB Environmental Associates A Regulatory, Environmental, and Economic Analysis of Water Supply Protection in Auburn, Maine dated October 2021 and the supplemental Lake Auburn Model Technical Memorandum dated August 1, 2022*. Accessed 11 March, 2023. https://lakeauburnwater.org/?page_id=882

Daloğlu, I., Cho, K. H., & Scavia, D. (2012). Evaluating Causes of Trends in Long-Term Dissolved Reactive Phosphorus Loads to Lake Erie. *Environmental Science & Technology*, *46*(19), 10660–10666. https://doi.org/10.1021/es302315d

Darracq, A., & Destouni, G. (2007). Physical versus biogeochemical interpretations of nitrogen and phosphorus attenuation in streams and its dependence on stream characteristics. *Global Biogeochemical Cycles*, *21*(3), n/a-n/a. https://doi.org/10.1029/2006GB002901

Day, L. (2004). Septic Systems as Potential Pollution Sources in the Cannonsville Reservoir Watershed, New York. *Journal of Environmental Quality*, *33*(6), 1989–1996. https://doi.org/10.2134/jeq2004.1989

Dillon, P. J., & Kirchner, W. B. (1975). The effects of geology and land use on the export of phosphorus from watersheds. *Water Research*, *9*(2), 135–148. https://doi.org/10.1016/0043-1354(75)90002-0

Doolittle, H. A., Norton, S. A., Bacon, L. C., Ewing, H. A., & Amirbahman, A. (2018). The internal and watershed controls on hypolimnetic sediment phosphorus release in Lake Auburn, Maine, USA. *Lake and Reservoir Management*, *34*(3), 258–269. https://doi.org/10.1080/10402381.2018.1423588

Easton, Z. M., Gérard-Marchant, P., Walter, M. T., Petrovic, A. M., & Steenhuis, T. S. (2007). Identifying dissolved phosphorus source areas and predicting transport from an urban watershed using distributed hydrologic modeling. *Water Resources Research*, *43*(11). https://doi.org/10.1029/2006WR005697

Efroymson, R. A., Jones, D. S., & Gold, A. J. (2007). An Ecological Risk Assessment Framework for Effects of Onsite Wastewater Treatment Systems and Other Localized Sources of Nutrients on Aquatic Ecosystems. *Human and Ecological Risk Assessment: An International Journal*, *13*(3), 574–614. https://doi.org/10.1080/10807030701341142

Eveborn, D., Kong, D., & Gustafsson, J. P. (2012). Wastewater treatment by soil infiltration: Long-term phosphorus removal. *Journal of Contaminant Hydrology*, *140–141*, 24–33. https://doi.org/10.1016/j.jconhyd.2012.08.003

FB Environmental Associates (FB Environmental). (2021). *A Regulatory, Environmental, and Economic Analysis of Water Supply Protection in Auburn, Maine*. Accessed 20 January, 2023. https://lakeauburnwater.org/?page_id=882

FB Environmental Associates (FB Environmental). (2022). *Memorandum on Modeling 2022 Proposed Ordinance Changes*. Accessed 20 January, 2023. https://www.auburnmaine.gov/pages/government/planning-board-agendas

Fetter, C. W. (1993). *Contaminant Hydrogeology*. Macmillan Pub. Co.; Maxwell Macmillan Canada; Maxwell Macmillan International.

Filep, G. (1999). *Soil chemistry: Processes and constituents*. Akadémiai Kiadó.

Freese, D., Zee, S. E. A. T. M., & Riemsdijk, W. H. (1992). Comparison of different models for phosphate sorption as a function of the iron and aluminium oxides of soils. *Journal of Soil Science*, *43*(4), 729–738. https://doi.org/10.1111/j.1365-2389.1992.tb00172.x

Geza, M., McCray, J. E., & Murray, K. E. (2010). Model Evaluation of Potential Impacts of On-Site Wastewater Systems on Phosphorus in Turkey Creek Watershed. *Journal of Environmental Quality*, *39*(5), 1636–1646. https://doi.org/10.2134/jeq2009.0354

Gill, L. W., & Mockler, E. M. (2016). Modeling the pathways and attenuation of nutrients from domestic wastewater treatment systems at a catchment scale. *Environmental Modelling & Software*, *84*, 363–377. https://doi.org/10.1016/j.envsoft.2016.07.006

Gill, L. W., O'Luanaigh, N., Johnston, P. M., Misstear, B. D. R., & O'Suilleabhain, C. (2009). Nutrient loading on subsoils from on-site wastewater effluent, comparing septic tank and secondary treatment systems. *Water Research*, *43*(10), 2739–2749. https://doi.org/10.1016/j.watres.2009.03.024

Gilliom, R. J., & Patmont, C. R. (1983). Lake phosphorus loading from septic systems by seasonally perched groundwater. *Journal (Water Pollution Control Federation)*, 1297-1305.

Gunady, M., Shishkina, N., Tan, H., & Rodriguez, C. (2015). A Review of On-Site Wastewater Treatment Systems in Western Australia from 1997 to 2011. *Journal of Environmental and Public Health*, *2015*, 1–12. https://doi.org/10.1155/2015/716957

Gundersen, L. (2020). The Origins and Dynamics of Phosphorus in Maine's Lake Auburn Watershed. *Honors Theses*. 333. https://scarab.bates.edu/honorstheses/333

Harman, J., Robertson, W. D., Cherry, J. A., & Zanini, L. (1996). Impacts on a Sand Aquifer from an Old Septic System: Nitrate and Phosphate. *Ground Water*, *34*(6), 1105–1114. https://doi.org/10.1111/j.1745-6584.1996.tb02177.x

Hill, D. E., & Frink, C. R. (1980). Septic System Longevity Increased by Improved Design. *Journal (Water Pollution Control Federation)*, *52*(8), 2199–2203. http://www.jstor.org/stable/25040860

Hölting, B., & Coldewey, W. G. (2018). *Hydrogeology* (8th ed.). Springer Berlin Heidelberg.

Ige, D. V., Akinremi, O. O., Flaten, D. N., Ajiboye, B., & Kashem, M. A. (2005). Phosphorus sorption capacity of alkaline Manitoba soils and its relationship to soil properties. *Canadian Journal of Soil Science*, *85*(3), 417–426. https://doi.org/10.4141/S04-064

Iverson, G., Humphrey, C. P., O'Driscoll, M. A., Sanderford, C., Jernigan, J., & Serozi, B. (2018). Nutrient exports from watersheds with varying septic system densities in the North Carolina Piedmont. *Journal of Environmental Management*, *211*, 206–217. https://doi.org/10.1016/j.jenvman.2018.01.063

Jeong, J., Santhi, C., Arnold, J. G., Srinivasan, R., Pradhan, S., & Flynn, K. (2011). Development of Algorithms for Modeling Onsite Wastewater Systems within SWAT. *Transactions of the ASABE*, *54*(5), 1693–1704. https://doi.org/10.13031/2013.39849

Kalff, J. (2002). *Limnology*. Prentice Hall, Upper Saddle River.

Karathanasis, A. D., Mueller, T. G., Boone, B., & Thompson, Y. L. (2006). Nutrient removal from septic effluents as affected by soil thickness and texture. *Journal of Water and Health*, *4*(2), 177–195. https://doi.org/10.2166/wh.2006.0015

Kong, X., Ghaffar, S., Determann, M., Friese, K., Jomaa, S., Mi, C., Shatwell, T., Rinke, K., & Rode, M. (2022). Reservoir water quality deterioration due to deforestation emphasizes the indirect effects of global change. *Water Research*, *221*, 118721. https://doi.org/10.1016/j.watres.2022.118721

LeBlanc, D. R. (1984). *Sewage plume in a sand and gravel aquifer, Cape Cod, Massachusetts* (Report No. 2218). U.S. Geological Survey water-supply paper.

Lin, C., Ma, R., & He, B. (2015). Identifying Watershed Regions Sensitive to Soil Erosion and Contributing to Lake Eutrophication—A Case Study in the Taihu Lake Basin (China). *International Journal of Environmental Research and Public Health*, *13*(1), 77. https://doi.org/10.3390/ijerph13010077

Litke, D. W. (1999). *Review of Phosphorus Control Measures in the United States and Their Effects on Water Quality* (Report No. 99-4007). U.S. Geological Survey Water-Resources Investigations Report. Accessed 11 March, 2023. https://pubs.usgs.gov/wri/wri994007/pdf/wri99-4007.pdf

Lusk, M. G., Toor, G. S., Yang, Y., Mechtensimer, S., De, M., & Obreza, T. A. (2017). A review of the fate and transport of nitrogen, phosphorus, pathogens, and trace organic chemicals in septic systems. *Critical Reviews in Environmental Science and Technology*, *47*(7), 455–541. https://doi.org/10.1080/10643389.2017.1327787

Ma, J., Ma, Y., Wei, R., Chen, Y., Weng, L., Ouyang, X., & Li, Y. (2021). Phosphorus transport in different soil types and the contribution of control factors to phosphorus retardation. *Chemosphere*, *276*, 130012. https://doi.org/10.1016/j.chemosphere.2021.130012

Mallin, M. A. (2013). Septic Systems in the Coastal Environment. In *Monitoring Water Quality* (pp. 81–102). Elsevier. https://doi.org/10.1016/B978-0-444-59395-5.00004-2

McBride, M. B. (1994). *Environmental chemistry of soils*. Oxford University Press.

McCobb, T. D., LeBlanc, D. R., Walter, D. A., Hess, K. M., Kent, D. B, & Smith, R. L. (2003). *Phosphorus in a ground-water contaminant plume discharging to Ashumet Pond, Cape*

Cod, Massachusetts, 1999 (Report No. 02-4306). U.S. Geological Survey Water-Resources Investigations Report.

McCray, J. E., Kirkland, S. L., Siegrist, R. L., & Thyne, G. D. (2005). Model Parameters for Simulating Fate and Transport of On-Site Wastewater Nutrients. *Ground Water*, *43*(4), 628–639. https://doi.org/10.1111/j.1745-6584.2005.0077.x

McGechan, M. B., & Lewis, D. R. (2002). Sorption of Phosphorus by Soil, Part 1: Principles, Equations and Models. *Biosystems Engineering*, *82*(1), 1–24. https://doi.org/10.1006/bioe.2002.0054

Mechtensimer, S., & Toor, G. S. (2016). Fate, mass balance, and transport of phosphorus in the septic system drainfields. *Chemosphere*, *159*, 153–158. https://doi.org/10.1016/j.chemosphere.2016.05.084

Mechtensimer, S., & Toor, G. S. (2017). Septic Systems Contribution to Phosphorus in Shallow Groundwater: Field-Scale Studies Using Conventional Drainfield Designs. *PLOS ONE*, *12*(1), e0170304. https://doi.org/10.1371/journal.pone.0170304

Meinikmann, K., Hupfer, M., & Lewandowski, J. (2015). Phosphorus in groundwater discharge – A potential source for lake eutrophication. *Journal of Hydrology*, *524*, 214–226. https://doi.org/10.1016/j.jhydrol.2015.02.031

Nair, V. D., Portier, K. M., Graetz, D. A., & Walker, M. L. (2004). An Environmental Threshold for Degree of Phosphorus Saturation in Sandy Soils. *Journal of Environmental Quality*, *33*(1), 107–113. https://doi.org/10.2134/jeq2004.1070

Oldfield, L. E.., Rakhimbekova, S., Roy, J. W., & Robinson, C. E. (2020a). Estimation of phosphorus loads from septic systems to tributaries in the Canadian Lake Erie Basin. *Journal of Great Lakes Research*, *46*(6), 1559–1569. https://doi.org/10.1016/j.jglr.2020.08.021

Oldfield, L. E., Roy, J. W., & Robinson, C. E. (2020b). Investigating the use of the artificial sweetener acesulfame to evaluate septic system inputs and their nutrient loads to streams at the watershed scale. *Journal of Hydrology*, *587*, 124918. https://doi.org/10.1016/j.jhydrol.2020.124918

Oliver, C. W., Radcliffe, D. E., Risse, L. M., Habteselassie, M., Mukundan, R., Jeong, J., & Hoghooghi, N. (2014). Quantifying the Contribution of On-Site Wastewater Treatment Systems to Stream Discharge Using the SWAT Model. *Journal of Environmental Quality*, *43*(2), 539–548. https://doi.org/10.2134/jeq2013.05.0195

Pacheco, J. P., Iglesias Frizzera, C., Goyenola, G., Teixeira de-Mello, F., Fosalba, C., Baattrup-Pedersen, A., Meerhoff, M., & Jeppesen, E. (2021). Invasion of Ceratium furcoides in subtropical lakes in Uruguay: Environmental drivers and fish kill record during its bloom. *Biological Invasions*, *23*(11), 3597–3612. https://doi.org/10.1007/s10530-021-02600-w

Parkhurst, D. L., Stollenwerk, K. G., & Colman., J. A. (2003) *Reactive-transport simulation of phosphorus in the sewage plume at the Massachusetts Military Reservation, Cape Cod, Massachusetts* (Report No. 03-4017). U.S. Geological Survey Water-Resources Investigations Report.

Ptacek, C. J. (1998). Geochemistry of a septic-system plume in a coastal barrier bar, Point Pelee, Ontario, Canada. *Journal of Contaminant Hydrology*, *33*(3–4), 293–312. https://doi.org/10.1016/S0169-7722(98)00076-X

Rakhimbekova, S., O'Carroll, D. M., Oldfield, L. E., Ptacek, C. J., & Robinson, C. E. (2021). Spatiotemporal controls on septic system derived nutrients in a nearshore aquifer and their discharge to a large lake. *Science of The Total Environment*, *752*, 141262. https://doi.org/10.1016/j.scitotenv.2020.141262

Reide Corbett, D., Dillon, K., Burnett, W., & Schaefer, G. (2002). The spatial variability of nitrogen and phosphorus concentration in a sand aquifer influenced by onsite sewage treatment and disposal systems: A case study on St. George Island, Florida. *Environmental Pollution*, *117*(2), 337–345. https://doi.org/10.1016/S0269-7491(01)00168-3

Robertson, W. D. (1995). Development of steady-state phosphate concentrations in septic system plumes. *Journal of Contaminant Hydrology*, *19*(4), 289–305. https://doi.org/10.1016/0169-7722(95)00022-N

Robertson, W. D. (2003). Enhanced Attenuation of Septic System Phosphate in Noncalcareous Sediments. *Ground Water*, *41*(1), 48–56. https://doi.org/10.1111/j.1745-6584.2003.tb02567.x

Robertson, W. D. (2008). Irreversible Phosphorus Sorption in Septic System Plumes? *Groundwater*, *46*(1), 51–60. https://doi.org/10.1111/j.1745-6584.2007.00389.x

Robertson, W. D. (2012). Phosphorus Retention in a 20-Year-Old Septic System Filter Bed. *Journal of Environmental Quality*, *41*(5), 1437–1444. https://doi.org/10.2134/jeq2011.0427

Robertson, W. D. (2021). *Septic System Impacts on Groundwater Quality*. The Groundwater Project, Guelph, Ontario, Canada.

Robertson, W. D., & Harman, J. (1999). Phosphate Plume Persistence at Two Decommissioned Septic System Sites. *Ground Water*, *37*(2), 228–236. https://doi.org/10.1111/j.1745-6584.1999.tb00978.x

Robertson, W. D., Schiff, S. L., & Ptacek, C. J. (1998). Review of Phosphate Mobility and Persistence in 10 Septic System Plumes. *Ground Water*, *36*(6), 1000–1010. https://doi.org/10.1111/j.1745-6584.1998.tb02107.x

Robertson, W. D., Van Stempvoort, D. R., & Schiff, S. L. (2019). Review of phosphorus attenuation in groundwater plumes from 24 septic systems. *Science of The Total Environment*, *692*, 640–652. https://doi.org/10.1016/j.scitotenv.2019.07.198

Roy, J. W., Spoelstra, J., Robertson, W. D., Klemt, W., & Schiff, S. L. (2017). Contribution of phosphorus to Georgian Bay from groundwater of a coastal beach town with decommissioned septic systems. *Journal of Great Lakes Research*, *43*(6), 1016–1029. https://doi.org/10.1016/j.jglr.2017.09.001

Schellenger, F. L., & Hellweger, F. L. (2019). Phosphorus loading from onsite wastewater systems to a lake (at long time scales). *Lake and Reservoir Management*, *35*(1), 90–101. https://doi.org/10.1080/10402381.2018.1541031

Schindler, D. W. (2006). Recent advances in the understanding and management of eutrophication. *Limnology and Oceanography*, *51*(1part2), 356–363. https://doi.org/10.4319/lo.2006.51.1_part_2.0356
Siegrist, R. L. (1987). Soil Clogging During Subsurface Wastewater Infiltration as Affected by Effluent Composition and Loading Rate. *Journal of Environmental Quality*, *16*(2), 181–187. https://doi.org/10.2134/jeq1987.00472425001600020016x
Sinclair, A., Jamieson, R., Madani, A., Gordon, R. J., Hart, W., & Hebb, D. (2014). A Watershed Modeling Framework for Phosphorus Loading from Residential and Agricultural Sources. *Journal of Environmental Quality*, *43*(4), 1356–1369. https://doi.org/10.2134/jeq2013.09.0368
Singer, M. J., & Munns, D. N. (2006). *Soils: An Introduction* (6th ed). Pearson Prentice Hall.
Sharma, A., Tiwari, K. N., & Bhadoria, P. B. S. (2011). Effect of land use land cover change on soil erosion potential in an agricultural watershed. *Environmental Monitoring and Assessment*, *173*(1–4), 789–801. https://doi.org/10.1007/s10661-010-1423-6
Soliman, M. M. (Ed.). (1998). *Environmental Hydrogeology*. Lewis Publishers.
Soranno, P. A., Hubler, S. L., Carpenter, S. R., & Lathrop, R. C. (1996). Phosphorus Loads to Surface Waters: A Simple Model to Account for Spatial Pattern of Land Use. *Ecological Applications*, *6*(3), 865–878. https://doi.org/10.2307/2269490
Spoelstra, J., Senger, N. D., & Schiff, S. L. (2017). Artificial Sweeteners Reveal Septic System Effluent in Rural Groundwater. *Journal of Environmental Quality*, *46*(6), 1434–1443. https://doi.org/10.2134/jeq2017.06.0233
Spoelstra, J., Schiff, S. L., & Brown, S. J. (2020). Septic systems contribute artificial sweeteners to streams through groundwater. *Journal of Hydrology X*, *7*, 100050. https://doi.org/10.1016/j.hydroa.2020.100050
Sposito, G. (2008). *The chemistry of soils* (2nd ed). Oxford University Press.
State of Maine. (2014). *Subsurface Wastewater Disposal Rules*. Accessed 14 April, 2023. https://www.maine.gov/dhhs/mecdc/environmental-health/dwp/documents/SubsurfaceWastewaterDisposalRulesProposal.pdf
Sun, D., Yang, H., Guan, D., Yang, M., Wu, J., Yuan, F., Jin, C., Wang, A., & Zhang, Y. (2018). The effects of land use change on soil infiltration capacity in China: A meta-analysis. *Science of The Total Environment*, *626*, 1394–1401. https://doi.org/10.1016/j.scitotenv.2018.01.104
Tan, K. H. (2011). *Principles of Soil Chemistry* (4th ed). CRC Press.
Todd, D. K. (1980). *Groundwater Hydrology* (2d ed). Wiley.
Tomaras, J., Sahl, J. W., Siegrist, R. L., & Spear, J. R. (2009). Microbial Diversity of Septic Tank Effluent and a Soil Biomat. *Applied and Environmental Microbiology*, *75*(10), 3348–3351. https://doi.org/10.1128/AEM.00560-08
Valtanen, M., Sillanpää, N., & Setälä, H. (2014). The Effects of Urbanization on Runoff Pollutant Concentrations, Loadings and Their Seasonal Patterns Under Cold Climate. *Water, Air, & Soil Pollution*, *225*(6), 1977. https://doi.org/10.1007/s11270-014-1977-y

Van Stempvoort, D. R., Brown, S. J., Spoelstra, J., Garda, D., Robertson, W. D., & Smyth, S. A. (2020). Variable persistence of artificial sweeteners during wastewater treatment: Implications for future use as tracers. *Water Research*, *184*, 116124. https://doi.org/10.1016/j.watres.2020.116124

Watson, S. B., Miller, C., Arhonditsis, G., Boyer, G. L., Carmichael, W., Charlton, M. N., Confesor, R., Depew, D. C., Höök, T. O., Ludsin, S. A., Matisoff, G., McElmurry, S. P., Murray, M. W., Peter Richards, R., Rao, Y. R., Steffen, M. M., & Wilhelm, S. W. (2016). The re-eutrophication of Lake Erie: Harmful algal blooms and hypoxia. *Harmful Algae*, *56*, 44–66. https://doi.org/10.1016/j.hal.2016.04.010

Wilhelm, S. R., Schiff, S. L., & Robertson, W. D. (1994). Chemical fate and transport in a domestic septic system: Unsaturated and saturated zone geochemistry. *Environmental Toxicology and Chemistry*, *13*(2), 193–203. https://doi.org/10.1002/etc.5620130203

Winstanley, H. F., & Fowler, A. C. (2013). Biomat Development in Soil Treatment Units for On-site Wastewater Treatment. *Bulletin of Mathematical Biology*, *75*(10), 1985–2001. https://doi.org/10.1007/s11538-013-9881-y

Withers, P. J. A., & Jarvie, H. P. (2008). Delivery and cycling of phosphorus in rivers: A review. *Science of The Total Environment*, *400*(1–3), 379–395. https://doi.org/10.1016/j.scitotenv.2008.08.002

Withers, P. J., Jarvie, H. P., & Stoate, C. (2011). Quantifying the impact of septic tank systems on eutrophication risk in rural headwaters. *Environment International*, *37*(3), 644–653. https://doi.org/10.1016/j.envint.2011.01.002

Withers, P. J., Jordan, P., May, L., Jarvie, H. P., & Deal, N. E. (2014). Do septic tank systems pose a hidden threat to water quality? *Frontiers in Ecology and the Environment*, *12*(2), 123–130. https://doi.org/10.1890/130131

Zanini, L., Robertson, W. D., Ptacek, C. J., Schiff, S. L., & Mayer, T. (1998). Phosphorus characterization in sediments impacted by septic effluent at four sites in central Canada. *Journal of Contaminant Hydrology*, *33*(3–4), 405–429. https://doi.org/10.1016/S0169-7722(98)00082-5

Data Sources:

SSURGO-NRCS Soil Survey Data: https://websoilsurvey.nrcs.usda.gov/app/
USGS Weighted Means of SSURGO Data:
https://water.usgs.gov/GIS/metadata/usgswrd/XML/ds866_ssurgo_variables.xml#stdorder
USFWS National Wetland Inventory:
https://www.fws.gov/program/national-wetlands-inventory/wetlands-data
USGS 1/3 arc-second Digital Elevation Model:
https://www.usgs.gov/the-national-map-data-delivery/gis-data-download
Significant Sand and Gravel Aquifer Map from the Maine Department of Agriculture, Conservation & Forestry: https://www.maine.gov/dacf/mgs/pubs/digital/aquifers.htm

A special thank you to the City of Auburn Planning & Permitting Department for providing me with access to municipal data for this project and for their cooperation. I would also like to thank the Auburn Water District for providing me with information on septic systems within Auburn during Summer 2022.

Appendix A: Additional Background

A.1.1 Phosphorus Geochemistry

Physical and chemical processes govern the mobility of phosphorus in soils. A number of processes immobilize P, which are often referred to in aggregate as phosphorus attenuation, retention, or removal in the context of soil contamination. About 85% of the phosphorus in wastewater exists in the form of inorganic phosphorus, also called orthophosphates ($H_2PO_4^-$, HPO_4^{2-}, PO_4^{3-}) (Lusk et al., 2017), which is dissolved and biologically available. The remaining P is in organic form, which can be transformed into inorganic P through biological processes; similarly, inorganic P can be immobilized into organic P as soil microorganisms utilize it in cellular processes and biomass (Weihrauch & Opp, 2018).

While orthophosphates are mobile in soils and can leach into groundwater, they can be retained weakly or strongly through adsorption and precipitation reactions. Physical adsorption refers to the adhesion of phosphate anions to mineral surfaces through van der Waals forces, which are weak intermolecular forces that require close proximity of a phosphate ion to the mineral surface (Tan, 2011). Because these forces are weak, physical adsorption is easily reversible. Adsorption also refers to ion exchange processes where electrostatic forces bind an ion to an oppositely charged site on the mineral surface. This type of ion exchange is easily reversible; ions such as nitrate (NO_3^-) tend to participate in this type of sorption (Filep, 1999). While phosphate participates in electrostatic ion exchange, it also is involved in a process called specific phosphate adsorption, which binds phosphate more strongly than other ions (Figure A.1). Specific phosphate sorption occurs on specific sites on the exchange complex that consist of Al or Fe oxides and hydroxides, or on clay particles (Sposito, 2008). Instead of typical anion exchange, a process called ligand exchange occurs where phosphate is exchanged for $-OH^-$ (Filep, 1999; Singer & Munns 2006), thus releasing it dissolved form. McBride (1994) describes that ligand exchange is different from anion exchange because it releases $-OH^-$ into solution, is specific to a subset of ions like phosphate, is less reversible than electrostatic adsorption, and leads to a more negative charge at the mineral surface. Specific sorption is a process so dominant that high quantities of Fe-oxides and Al-oxides in a particular soil can be an indicator of high capacity to retain P (Borggaard et al., 1990; Zhang et al., 2005).

While soils have an anion exchange capacity (AEC) similar to the widely known cation exchange capacity (CEC), AEC is not a perfect reflection of the amount of phosphorus a soil may retain due to the fact that specific phosphate adsorption may occur on neutral or negatively charged sites, instead of just positive sites (Filep, 1999; Singer & Munns, 2006). However, soils do have a maximum threshold for phosphorus retention (Zhang et al., 2005), often referred to as the maximum sorption capacity. This capacity increases with decreasing pH, as a result of more positive sites for adsorption (Brady & Weil, 2002) and because low pH increases in the availability of Fe and Al (Singer & Munns, 2006). In alkaline environments, where there is less Fe/Al, high amounts of exchangeable Ca can indicate high P retention (Ige et al., 2005).

Because minerals with a higher surface area have more sorption sites, the amount of phosphate a soil may adsorb is strongly related to the surface area and thus size of soil particles (Tan, 2011). Therefore, soil qualities such as texture and the presence of organic matter strongly influence adsorption capacity (McGechan & Lewis 2002), with sands having a lower sorption capacity than clay soils. A review of reported values of maximum sorption capacity by McCray et al. (2005) finds that median maximum P sorption capacity of sand is 40 mg-P/kg-soil while the capacity of sandy loam is 405 mg-P/kg-soil, which is evidence of the far worse ability of sands to immobilize P. This is because coarse-textured soils such as sands often have less ion exchange capacities due to having less surface area (Singer & Munns, 2006), leading to a lesser ability to adsorb phosphorus (Carroll et al., 2005).The potential for P sorption to organic matter indicates that moderate amounts of organic matter improve the retention of P (Atalay, 2001), while too much organic matter is detrimental to P sorption because it competes with P for sorption sites (Daly et al., 2001). The mineralogy of the soil has a large effect on sorption capacity alongside texture and clay percentage (Fink et al., 2016; Ige et al., 2005) because different mineralogies dictate the amounts of Fe and Al that may be present (Borggaard et al., 1990). Though the different types of adsorption bind phosphates at different strengths, all adsorption processes are reversible through desorption (Barrow, 1983). Similar to how P is tightly held at low pH, desorption is low at low pH, and increases as pH increases (Filep, 1999).

Phosphates may also be immobilized more permanently through precipitation reactions involving base and metal cations (Figure A.1). Unlike adsorption, precipitation involves chemical reactions that transform phosphates into solid mineral compounds. Therefore, precipitation is not limited by the number of possible sorption sites and instead by the presence of its constituents (Filep, 1999). Since these reactions involve base and metal cations, they are pH dependent. Non-calcareous soils are acidic and have larger available stores of Al and Fe, which precipitate with phosphate to produce minerals such as variscite [$AlPO_4 \cdot 2H_2O$]

(Robsertson, 2021). In calcareous soils, which are basic, phosphate tends to precipitate with Ca or Mg to form compounds such as hydroxyapatite [$Ca_5(PO_4)_3(OH)$] (Lusk et al., 2017). Due to a lack of availability of Al and Fe in non-acid conditions, calcium precipitation with phosphate controls the solubility of P in calcareous soils (McBride, 1994).

Precipitation is a pH dependent process because it relates to the availability of certain elements and the solubility of P minerals. Generally, P is more soluble near-neutral pH (Eveborn et al., 2012). This results partially from the fact that metal cations are more available at low pH, while base cations like calcium are most available in basic conditions. Therefore, studies have noted that an increase in pH only increases the immobilization of phosphorus if the soil pH is already high (Barrow, 1984); in essence, this suggests that changes in pH only increase P retention if the pH becomes less neutral. Because Al and Fe tend to be the most common precipitates, the immobilization of phosphorus is typically regarded to be highest at low pH (Robertson, 2012).

Organic phosphorus may also leach down the soil profile. It can be considered to leach more readily than inorganic P because it adsorbs to mineral surfaces less strongly than phosphate and is not involved in reactions involving Al or Fe (Frossard et al., 1989). Authors who collected leachate from beneath septic drain fields found that the percentage of organic phosphorus in solution increased from 16% to 53% after being treated by the soil (Mechtensimer & Toor, 2016). This indicates that the soil does not immobilize organic phosphorus as efficiently as it does inorganic phosphorus; both forms of P may pose issues if leached into groundwater due to their effect on surface water bodies (Kalff, 2002). Microbial P is only able to leach if it is held in deceased microbes (Weihrauch & Opp, 2018), indicating that microbial organic P is immobile if the microbes are alive.

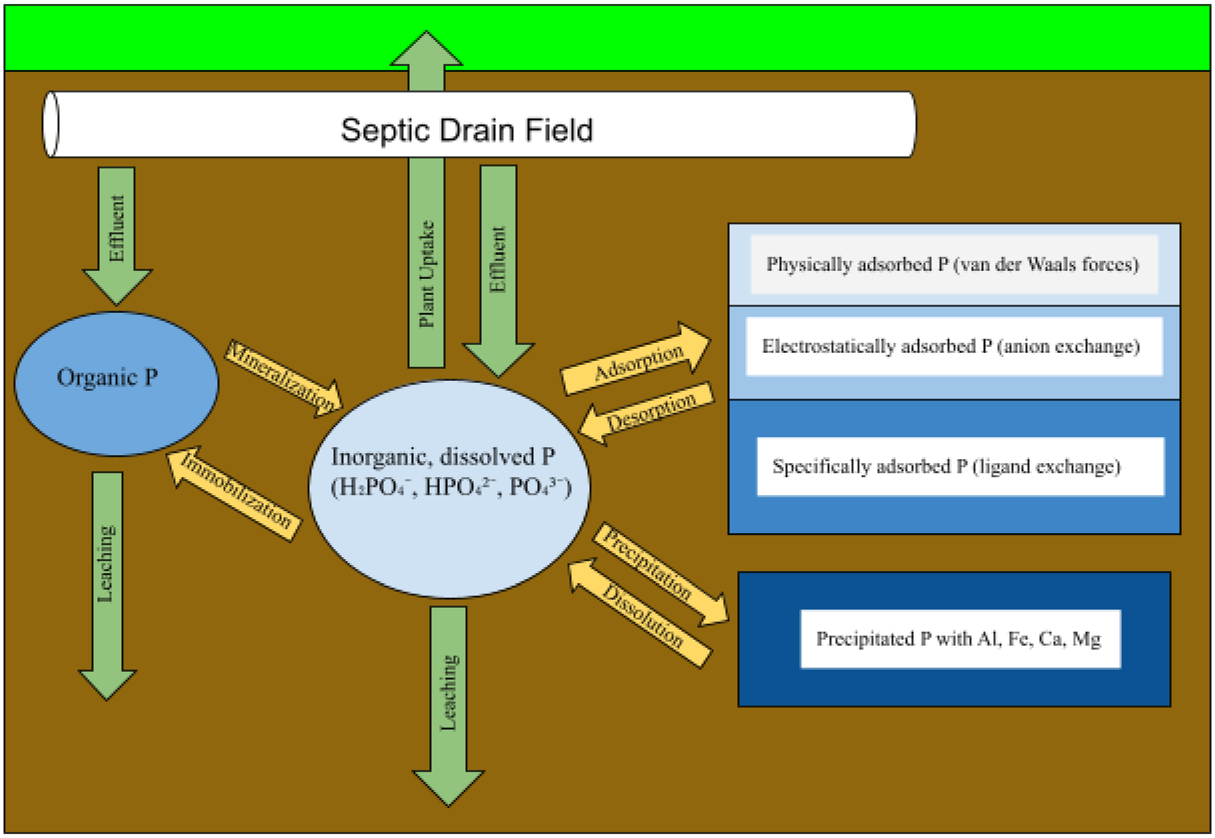


Figure A.1: A conceptual diagram showing the most prominent movements and transformations of phosphorus in septic drain fields. Green arrows represent the movement of phosphorus, while yellow arrows represent transformations of phosphorus. Deepening shades of blue signify more tightly immobilized P. The size of arrows or other shapes do not correlate to sizes of P stores.

Scholars often quantify phosphorus immobilization from adsorption and precipitation processes in soils by creating isotherms that reflect how P retention operates in different soils. A phosphorus sorption isotherm seeks to show a relationship between the equilibrium concentration of P in solution and the amount of P sorbed per unit mass of soil. This relationship demonstrates how the amount of P sorbed is dependent on the concentration of the solute.

Because the physical and chemical processes that govern phosphorus mobility are complex and hard to distinguish (Boulding, 1995), sorption isotherms generalize all phosphorus immobilization under the term “sorption,” which includes physical adsorption, specific phosphate adsorption, and precipitation (Evangelou, 1998). In this way, sorption isotherms may generalize the immobilization of P. Some studies point out that isotherms may be limited in their effectiveness because of this generalization and because of other assumptions that cause mismatches between the isotherm and empirical data (Cucarella & Renman, 2009).

There are three main types of isotherms used in P loading models. The linear isotherm is the most basic of the three. Due to the linear nature of this isotherm, it reaches no maximum value, meaning it cannot accurately simulate the maximum sorption capacity of a soil. The slope of a linear isotherm depends upon the texture of the soil among other variables, such as the bulk density of the soil (McCray et al., 2005). Freundlich and Langmuir isotherms, which are two types of non-linear isotherms, are more complex than the linear isotherm. The sorption of phosphate most closely fits the Langmuir isotherm (McBride, 1994; Evangelou, 1998), and has the ability to account for a maximum sorption capacity. However, because the concentration of P in household wastewater typically falls within the linear portion of non-linear isotherms (McCray et al., 2005), many modelers opt to use the linear isotherm; the simplicity of the isotherm also makes it mathematically easier to use (Filipović et al., 2016).

A.1.2 Hydrogeology

The area below the septic drain field is composed of two hydrologically distinct areas through which water moves toward water bodies. The vadose zone is located directly below the infiltration pipes and consists of unsaturated soil, meaning the pore spaces between mineral particles are filled with both air and water. Below the vadose zone, there is a saturated zone where pore spaces are completely filled with groundwater. While the transition between these zones tends to be relatively discrete, there often exists a large capillary zone in finer textured soils where groundwater is drawn upward through a process similar to capillary action (Bedient et al., 1994), causing redoximorphic features such as mottling to be observed (Brady & Weil, 2002). Although the capillary zone is technically above the true water table, evidence of the capillary zone can mark the seasonal high water table (Brady & Weil, 2002; State of Maine, 2014).

In the vadose zone beneath a septic drain field, percolation of wastewater tends to be vertical in nature, though the way wastewater is dispersed is heterogeneous and percolation is not perfectly one-dimensional due to dispersion (Filipović et al., 2016). Hydrogeologists who have developed mathematical models of vadose zone hydrology posit that accurate simulation of flow in the unsaturated zone is imperative for the assessment of contaminant transport through groundwater (Schnoor, 1996; Szymkiewicz, 2013).

The rate at which water moves through the vadose zone is influenced predominantly by climatic factors and soil characteristics such as the unsaturated hydraulic conductivity (Stephens, 1996), which is often represented as a function of moisture. Movement of water through the unsaturated zone is often considered more complex and slower than movement through the saturated zone due to the presence of air in pore space which can inhibit the infiltration of water (Bedient et al., 1994; Hölting & Coldewey, 2018). In spite of this, the presence of macropores, fractured bedrock, or soil lenses can create preferential flow paths for water that make downward percolation faster (Fetter, 1993; Toor et al., 2005).

Because water in the saturated zone can be vertically contained by bedrock or hydrologically restrictive layers of finely-textured soils (Soliman et al., 1998), water flow is mostly lateral and downgradient (Stephens 1996). Groundwater flow tends to follow topography toward surface water bodies. Although the saturated hydraulic conductivity tends to be higher than the unsaturated hydraulic conductivity (Fetter, 1993), groundwater still moves slowly at rates that may span from 2 meters per year to 2 meters per day (Todd, 1980). In both the saturated and unsaturated zone, water flow may trap contaminants in less connected or smaller pores (Singer & Munns, 2006), as water tends to flow through the largest and most connected pores instead.

Groundwater movement is described by Darcy's Law, which states that the rate of lateral flow through saturated porous soil is proportional to the elevation decrease over a distance (Schnoor, 1996), also known as the slope or relief. Darcy velocity can be calculated by the product of the saturated hydraulic conductivity (K_{sat}) and the change in groundwater head ($\frac{dh}{dl}$). Though it does not represent the actual velocity of water moving through the soil, it can be used to find the pore-water velocity by dividing by the effective porosity (ϕ) of the soil (Hölting & Coldewey, 2018). In septic systems, wastewater that enters the saturated zone after infiltration through the vadose zone moves laterally and downgradient until entering a surface water body, while some may be lost to a deeper aquifer system (Boulding, 1995; Todd, 1980).

Since groundwater moves slowly, an influx of pollutants from septic systems can create identifiable wastewater plumes within the saturated zone, with contaminant plumes differing in length and concentration based on the mobility of the contaminant. For example, a study by Robertson (2008) compares nitrate and phosphate plumes over a sixteen year period, finding that nitrate plumes are longer and deeper than phosphate plumes, and that the richest zone of nitrogen extends beyond the entire phosphorus plume, demonstrating that nitrogen is far more mobile than phosphorus. While inorganic nitrogen (primarily in the form of NO_3^-) is highly mobile and readily moves along the speed of groundwater (Wilhelm et al., 1994), phosphate (PO_4^{3-}) is far less mobile as it is readily adsorbed. Because of this, plumes of phosphorus from septic systems move at rates far slower than the velocity of the groundwater (Ptacek, 1998; Rakhimbekova et al., 2021), as reversible adsorption reactions delay the transport of P in the saturated zone.

Back to Section 2.1.6

A.2 Works Cited for Appendix A

Atalay, A. (2001). Variation in Phosphorus Sorption with Soil Particle Size. *Soil and Sediment Contamination: An International Journal*, *10*(3), 317–335. https://doi.org/10.1080/20015891109284

Barrow, N. J. (1983). A mechanistic model for describing the sorption and desorption of phosphate by soil. *Journal of Soil Science*, *34*(4), 733–750. https://doi.org/10.1111/j.1365-2389.1983.tb01068.x

Barrow, N. J. (1984). Modelling the effects of pH on phosphate sorption by soils. *Journal of Soil Science*, *35*(2), 283–297. https://doi.org/10.1111/j.1365-2389.1984.tb00283.x

Bedient, P. B., Rifai, H. S., & Newell, C. J. (1994). *Ground Water Contamination: Transport and Remediation*. PTR Prentice Hall.

Borggaard, O. K., Jdrgensen, S. S., Moberg, J. P., & Raben-Lange, B. (1990). Influence of organic matter on phosphate adsorption by aluminium and iron oxides in sandy soils. *Journal of Soil Science*, *41*(3), 443–449. https://doi.org/10.1111/j.1365-2389.1990.tb00078.x

Boulding, R. (1995). *Practical handbook of soil, vadose zone, and ground-water contamination: Assessment, prevention, and remediation*. Lewis Publishers.

Brady, N. C., & Weil, R. R. (2002). *The nature and properties of soils* (13th ed). Prentice Hall.
Carroll, S., Goonetilleke, A., Khalil, W. A.-S., & Frost, R. (2005). Assessment via discriminant analysis of soil suitability for effluent renovation using undisturbed soil columns. *Geoderma*, *131*(1–2), 201–217. https://doi.org/10.1016/j.geoderma.2005.03.022
Cucarella, V., & Renman, G. (2009). Phosphorus Sorption Capacity of Filter Materials Used for On-site Wastewater Treatment Determined in Batch Experiments-A Comparative Study. *Journal of Environmental Quality*, *38*(2), 381–392. https://doi.org/10.2134/jeq2008.0192
Daly, K., Jeffrey, D., & Tunney, H. (2006). The effect of soil type on phosphorus sorption capacity and desorption dynamics in Irish grassland soils. *Soil Use and Management*, *17*(1), 12–20. https://doi.org/10.1111/j.1475-2743.2001.tb00003.
Evangelou, V. P. (1998). *Environmental soil and water chemistry: Principles and applications*. John Wiley.
Eveborn, D., Kong, D., & Gustafsson, J. P. (2012). Wastewater treatment by soil infiltration: Long-term phosphorus removal. *Journal of Contaminant Hydrology*, *140–141*, 24–33. https://doi.org/10.1016/j.jconhyd.2012.08.003
Fetter, C. W. (1993). *Contaminant Hydrogeology*. Macmillan Pub. Co.; Maxwell Macmillan Canada; Maxwell Macmillan International.
Filep, G. (1999). *Soil chemistry: Processes and constituents*. Akadémiai Kiadó.
Filipović, V., Ondrašek, G., & Filipović, L. (2016). Modelling Water Dynamics, Transport Processes and Biogeochemical Reactions in Soil Vadose Zone. In M. S. Javaid (Ed.), *Groundwater—Contaminant and Resource Management*. InTech. https://doi.org/10.5772/63496
Fink, J. R., Inda, A. V., Bavaresco, J., Barrón, V., Torrent, J., & Bayer, C. (2016). Adsorption and desorption of phosphorus in subtropical soils as affected by management system and mineralogy. *Soil and Tillage Research*, *155*, 62–68. https://doi.org/10.1016/j.still.2015.07.017
Frossard, E., Stewart, J. W. B., & St. Arnaud, R. J. (1989). Distribution and mobility of phosphorus is grassland and forest soils of Saskatchewan. *Canadian Journal of Soil Science*, *69*(2), 401–416. https://doi.org/10.4141/cjss89-040
Hölting, B., & Coldewey, W. G. (2018). *Hydrogeology* (8th ed.). Springer Berlin Heidelberg.
Ige, D. V., Akinremi, O. O., Flaten, D. N., Ajiboye, B., & Kashem, M. A. (2005). Phosphorus sorption capacity of alkaline Manitoba soils and its relationship to soil properties. *Canadian Journal of Soil Science*, *85*(3), 417–426. https://doi.org/10.4141/S04-064
Kalff, J. (2002). *Limnology*. Prentice Hall, Upper Saddle River.
Lusk, M. G., Toor, G. S., Yang, Y., Mechtensimer, S., De, M., & Obreza, T. A. (2017). A review of the fate and transport of nitrogen, phosphorus, pathogens, and trace organic chemicals in septic systems. *Critical Reviews in Environmental Science and Technology*, *47*(7), 455–541. https://doi.org/10.1080/10643389.2017.1327787
McBride, M. B. (1994). *Environmental chemistry of soils*. Oxford University Press.

McCray, J. E., Kirkland, S. L., Siegrist, R. L., & Thyne, G. D. (2005). Model Parameters for Simulating Fate and Transport of On-Site Wastewater Nutrients. *Ground Water*, *43*(4), 628–639. https://doi.org/10.1111/j.1745-6584.2005.0077.x
McGechan, M. B., & Lewis, D. R. (2002). Sorption of Phosphorus by Soil, Part 1: Principles, Equations and Models. *Biosystems Engineering*, *82*(1), 1–24. https://doi.org/10.1006/bioe.2002.0054
Mechtensimer, S., & Toor, G. S. (2016). Fate, mass balance, and transport of phosphorus in the septic system drainfields. *Chemosphere*, *159*, 153–158. https://doi.org/10.1016/j.chemosphere.2016.05.084
Ptacek, C. J. (1998). Geochemistry of a septic-system plume in a coastal barrier bar, Point Pelee, Ontario, Canada. *Journal of Contaminant Hydrology*, *33*(3–4), 293–312. https://doi.org/10.1016/S0169-7722(98)00076-X
Rakhimbekova, S., O'Carroll, D. M., Oldfield, L. E., Ptacek, C. J., & Robinson, C. E. (2021). Spatiotemporal controls on septic system derived nutrients in a nearshore aquifer and their discharge to a large lake. *Science of The Total Environment*, *752*, 141262. https://doi.org/10.1016/j.scitotenv.2020.141262
Robertson, W. D. (2008). Irreversible Phosphorus Sorption in Septic System Plumes? *Groundwater*, *46*(1), 51–60. https://doi.org/10.1111/j.1745-6584.2007.00389.x
Robertson, W. D. (2012). Phosphorus Retention in a 20-Year-Old Septic System Filter Bed. *Journal of Environmental Quality*, *41*(5), 1437–1444. https://doi.org/10.2134/jeq2011.0427
Robertson, W. D. (2021). *Septic System Impacts on Groundwater Quality*. The Groundwater Project, Guelph, Ontario, Canada.
Schnoor, J. L. (1996). *Environmental modeling: Fate and transport of pollutants in water, air, and soil*. Wiley.
Singer, M. J., & Munns, D. N. (2006). *Soils: An Introduction* (6th ed). Pearson Prentice Hall.
Sposito, G. (2008). *The chemistry of soils* (2nd ed). Oxford University Press.
State of Maine. (2014). *Subsurface Wastewater Disposal Rules*. Accessed 14 April, 2023. https://www.maine.gov/dhhs/mecdc/environmental-health/dwp/documents/SubsurfaceWastewaterDisposalRulesProposal.pdf
Stephens, D. B. (1996). *Vadose Zone Hydrology*. CRC, Lewis Publishers.
Szymkiewicz, A. (2013). *Modelling Water Flow in Unsaturated Porous Media: Accounting for Nonlinear Permeability and Material Heterogeneity*. Springer.
Tan, K. H. (2011). *Principles of Soil Chemistry* (4th ed). CRC Press.
Todd, D. K. (1980). *Groundwater Hydrology* (2d ed). Wiley.
Toor, G. S., Condron, L. M., Cade-Menun, B. J., Di, H. J., & Cameron, K. C. (2005). Preferential phosphorus leaching from an irrigated grassland soil. *European Journal of Soil Science*, *56*(2), 155–168. https://doi.org/10.1111/j.1365-2389.2004.00656.x
Wilhelm, S. R., Schiff, S. L., & Robertson, W. D. (1994). Chemical fate and transport in a domestic septic system: Unsaturated and saturated zone geochemistry. *Environmental Toxicology and Chemistry*, *13*(2), 193–203. https://doi.org/10.1002/etc.5620130203

Weihrauch, C., & Opp, C. (2018). Ecologically relevant phosphorus pools in soils and their dynamics: The story so far. *Geoderma*, *325*, 183–194. https://doi.org/10.1016/j.geoderma.2018.02.047

Zhang, H., Schroder, J. L., Fuhrman, J. K., Basta, N. T., Storm, D. E., & Payton, M. E. (2005). Path and Multiple Regression Analyses of Phosphorus Sorption Capacity. *Soil Science Society of America Journal*, *69*(1), 96. https://doi.org/10.2136/sssaj2005.0096dup

Appendix B: Review of Other Modeling Approaches

B.1.1 WARMF (Geza et al., 2010)

The Watershed Analysis Risk Management Framework (WARMF) is a model that estimates nutrient loading for an entire watershed. Geza et al. (2010) used WARMF to estimate P loading from septic systems in a large (126 km^2) watershed in Colorado. As a hydrologic model, WARMF uses algorithms to describe water movement, calculating stream flow, groundwater flow, soil moisture, and a variety of other metrics relating to water flow (Herr et al., 2001). For septic systems, the model uses isotherms and flow paths to simulate P sorption and transport, and requires inputs such as P concentration in effluent, and wastewater generation per person. It also requires the use of easily accessible datasets such as a digital elevation model (DEM), soil data from STASTGO, land cover from the National Land Cover Dataset, and meteorological data (Geza et al., 2010), which allows for parameters such as the maximum sorption capacity, adsorption isotherms, porosity, initial soil P concentration, and hydraulic conductivity to be determined. Up to five soil layers may be defined in WARMF, which may have different properties that affect the movement and attenuation of P. WARMF uses these data to simulate the movement of effluent toward streams and accounts for additional P attenuation within streams, both through adsorption and particulate settling.

One of the benefits of WARMF is that it uses easily accessible data and simulates the exact processes involved in P retention and transport. However, there are a large number of parameters that must be determined either through literature values or model calibration that may be difficult to define. Moreover, the use of national datasets may not capture the spatial heterogeneity that exists across a watershed. For example, Geza et al. (2010) found that the STASTGO soil survey data only revealed three soil types across the 126 km^2 watershed, and only one sorption isotherm was used for native soil. Sensitivity analysis identified P-sorption capacity as a sensitive parameter, meaning changes in P-sorption capacity caused large changes in the results relative to other parameters. Since P-sorption capacity varies greatly with soil type (McCray et al., 2005), a lack of specific soil data may make models less accurate.

B.1.2 SWAT (Jeong et al., 2011)

In order to simulate P loading from septic systems using the Soil & Water Assessment Tool (SWAT), Jeong et al. (2011) adapted a biomat algorithm from Siegrist et al. (2005), which is now integrated into SWAT (Texas A&M University, 2012). SWAT, similar to WARMF, requires many inputs from national datasets, and calculates P loading for the entire watershed rather than exclusively from septic systems. Previous thesis work by Gundersen (2020) uses the SWAT model to estimate P loading into Lake Auburn; descriptions of the model and its functions can be found there. In simulating septic system processes and biomat formation, SWAT considers factors such as air temperature, soil moisture, and ground cover to simulate the growth of bacteria in the biomat, as well as bacterial mortality rates and the buildup of bacterial plaque. Through this, it estimates the P removed in the drain field using a linear sorption isotherm and simulates the lifespan of the septic system. When the septic system fails, the model simulates a certain number of days until the system is fixed where the system loads P through overland flow.

The model has been used to estimate the typical lifespan of septic systems based on input parameters determined by Jeong et al. (2011), accounting for variability in septic lifespan by using Monte Carlo simulations. They found that the model estimated a median lifespan of 20 years and a range of 5 to >35 years, which was in range of reported values (Jeong et al., 2011). When estimating P loading, the authors found that using SWAT in the reference watershed underestimated the amount of P loading to water bodies because the soil types in the watershed were sandier than could be captured by SWAT. Because SWAT only used one sorption isotherm, the model was unable to consider variations in soil type that are present in the watershed (Jeong et al., 2011). This reveals that the use of hydrologic models that only utilize one isotherm may be unable to accurately reflect soil types throughout a watershed, which is critical to simulating P removal in septic systems. This could be exacerbated by coarse-scale soil survey data that does not reflect the variations present in smaller watersheds.

SWAT may also require various input parameters that may be hard to define, though default parameter values are provided by Texas A&M University (2012). While the default parameters provide a value when no data specific to the study area are available, it is possible that results may not reflect the true dynamics of P movement from septic systems because the default parameters are not appropriate for the area. For example, an input parameter is the time until a failing system is fixed, which has a default value of 70 days (Texas A&M University, 2012). If the reference watershed is in an area with less education about septic system maintenance, this value may be larger than 70 days, and the amount of P transported to the lake may not be accurately simulated.

B.1.3 SWAT with POWSIM (Sinclair et al., 2014)

Some studies seek to alter SWAT to better reflect watershed characteristics or the transport of P in the subsurface. Sinclair et al. (2014b) modifies SWAT by creating a Phosphorus Onsite Wastewater Simulator (POWSIM) to reflect a Nova Scotia watershed, where lateral flow sand filters (LFSF) are used in septic drain fields, which consist of imported sand. The authors account for the mass of sand involved in P treatment, as well as use an equation to find the mass of soil in the wastewater plume that may contribute to P treatment and retardation. Furthermore, the authors simulate P treatment by using piecewise linear sorption isotherms and maximum sorption capacity in order to consider both adsorption and precipitation processes. The isotherms function by using a large slope (indicating high P retention) to consider all P removal processes before the maximum sorption capacity is reached, and using a smaller slope (indicating less retention) when the cumulative amount of P removal exceeds the maximum sorption capacity, reflecting how mineral precipitation still occurs while adsorption ceases. The isotherms were developed by fitting empirical P-removal data collected in the same watershed to equations (Sinclair et al. 2014a). This allowed them to account for differences between the three different types of imported sand filter media and native soil. Only one isotherm was used for native soil, likely because they characterize the area as having shallow groundwater, which indicates that the amount of native soil is likely small.

When applied to a Nova Scotia watershed, Sinclair et al. (2014b) lumped septic systems into four clusters and used median values for each cluster to determine input values. They approximated septic systems failure time based on a local survey on septic system maintenance.

They found that SWAT with POWSIM more accurately predicted P loading into water bodies than SWAT alone for this watershed, based on field data (Sinclair et al., 2014b). Advantages to POWSIM are that it uses isotherms developed from field studies in the same watershed, meaning these isotherms more accurately reflect the systems in Nova Scotia than a default isotherm may. It also accounts for hydraulic failure of septic systems and accurately considers the retardation of P in the groundwater, finding that peak P loading from septic systems did not occur until over 30 years after the simulation began (Sinclair et al., 2014b).

Even though POWSIM is only applicable to the reference watershed in Nova Scotia, it excels in considering soil factors such as vadose zone depth, hydraulic conductivity, and site slope, while also considering processes of P removal and transport. Its other strengths lie in the ways it captures septic system design in Nova Scotia and utilizes data from the same watershed in its calculations. However, it does not consider complex hydrology as is used by process-based models like WARMF or SWAT (POWSIM is used in conjunction with SWAT, but does not use any SWAT algorithms). Though the model was designed to be used for lateral flow sand filter septic systems that are common in Nova Scotia, it is clear that this type of model could be modified or reproduced for areas serviced by different types of septic systems. The types of input parameters needed for POWSIM are obtainable only through field studies, as the development of isotherms requires empirical data. (Back to Chapter I)

B.2 Works Cited for Appendix B

Geza, M., McCray, J. E., & Murray, K. E. (2010). Model Evaluation of Potential Impacts of On-Site Wastewater Systems on Phosphorus in Turkey Creek Watershed. *Journal of Environmental Quality*, *39*(5), 1636–1646. https://doi.org/10.2134/jeq2009.0354

Gundersen, L. (2020). The Origins and Dynamics of Phosphorus in Maine's Lake Auburn Watershed. *Honors Theses*. 333. https://scarab.bates.edu/honorstheses/333

Herr, J., Weintraub, L., & Chen, C. W. (2001). *User's Guide to WARMF: Documentation of Graphical User Interface*. Electric Power Research Institute. Accessed 22 March, 2023. https://scdhec.gov/sites/default/files/docs/HomeAndEnvironment/Docs/WARMF_UsersGuide_2001.pdf

Jeong, J., Santhi, C., Arnold, J. G., Srinivasan, R., Pradhan, S., & Flynn, K. (2011). Development of Algorithms for Modeling Onsite Wastewater Systems within SWAT. *Transactions of the ASABE*, *54*(5), 1693–1704. https://doi.org/10.13031/2013.39849

McCray, J. E., Kirkland, S. L., Siegrist, R. L., & Thyne, G. D. (2005). Model Parameters for Simulating Fate and Transport of On-Site Wastewater Nutrients. *Ground Water*, *43*(4), 628–639. https://doi.org/10.1111/j.1745-6584.2005.0077.x

Siegrist, R. L., J. McCray, L. Weintraub, C. Chen, J. Bagdol, P. Lemonds, S. Van Cuyk, K. Lowe, R. Goldstein, and J. Rada. (2005). *Quantifying site‐scale processes and watershed‐scale cumulative effects of decentralized wastewater systems*. Project No. WU‐HT‐00‐27. Prepared by the Colorado School of Mines for the National Decentralized Water Resources Capacity Development Project, Washington University, St. Louis, Mo.

Sinclair, A., Jamieson, R., Gordon, R. J., Madani, A., & Hart, W. (2014a). Modeling Phosphorus Treatment Capacities of On-Site Wastewater Lateral Flow Sand Filters. *Journal of Environmental Engineering*, *140*(2), 04013002. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000796

Sinclair, A., Jamieson, R., Madani, A., Gordon, R. J., Hart, W., & Hebb, D. (2014b). A Watershed Modeling Framework for Phosphorus Loading from Residential and Agricultural Sources. *Journal of Environmental Quality*, *43*(4), 1356–1369. https://doi.org/10.2134/jeq2013.09.0368

Texas A&M University. (2012). *Chapter 35 SWAT Input Data: .SEP.* Accessed 19 April, 2023. https://swat.tamu.edu/media/69404/ch35_input_sep.pdf

Appendix C: Supplemental Methodology

Soil Permeability Score Assigned by Vadose Zone Texture			
Soil Permeability	Permeability Score	Attenuation Factor (α_i)	Vadose Zone Soil Texture from Site Evaluation
High Permeability	1	0.75	gravel, gravelly sand, gravelly loamy sand, medium sand, coarse sand, fine sand, loamy sand/sand, fine/medium sand, medium/coarse sand, sandy till, loamy sand/coarse sand, Hydrologic Soil Groups A, A/D
Medium Permeability	2	0.93	loamy sand, sandy loam/loamy sand, gravelly sandy loam, sandy loam, fine sandy loam, very fine sandy loam, silt loam, silty clay loam Hydrologic Soil Groups B, B/D, C, C/D, other
Low Permeability	3	0.94	silty clay, Hydrologic Soil Group D

Table C.1: Each septic drain field was assigned a soil permeability score based on its texture for use in quantification methods that account for soil variation or system failure. Permeability score was determined either from site evaluation or using hydrologic soil group from the SSURGO database.

Effective Porosity Values Assigned by Vadose Zone Texture		
Soil Texture (Hölting & Coldewey (2018)	Effective Porosity (ϕ)	Bottom-most Horizon of Vadose Zone from Site Evaluation
sandy gravel	0.225	gravel
gravelly sand	0.175	gravelly sand, gravelly loamy sand
medium sand	0.125	medium sand, coarse sand, fine sand, loamy sand/sand, fine/medium sand, medium/coarse sand, sandy till, loamy sand/coarse sand, Hydrologic Soil Groups A, A/D
silty sand	0.10	loamy sand, sandy loam/loamy sand, gravelly sandy loam, Hydrologic Soil Groups B, B/D
sandy silt	0.75	sandy loam, fine sandy loam, very fine sandy loam, silt loam, other
clayey silt	0.055	silty clay loam, Hydrologic Soil Groups C, C/D
silty clay	0.035	silty clay, Hydrologic Soil Group D

Table C.2: Effective porosity values were assigned to soil types based on Hölting & Coldewey (2018). The bottom-most horizon of the vadose zone was assigned an effective porosity value based on the soil texture it most closely resembles. Effective porosity was used in Equation 5, in which the Darcy equation was applied to calculate the P plume velocity.

Back to Section 3.1.3
Back to Section 3.1.4

Appendix D: Limitations of Temporal Model

D.1.1 Introduction

Although the temporal model provides estimates of P loading into the future, the numbers presented in this study are not likely to reflect actual P loading from septic systems to Lake Auburn due to the number of assumptions made, the reliance on literature values, and the inability to calibrate the model against *in-situ* data. Other models that quantify P loading from septic systems are calibrated as part of a hydrologic model that estimates P loading for the entire watershed (Jeong et al., 2011; Geza et al., 2010) or can be compared to *in-situ* data (Sinclair et al., 2014; Oliver et al., 2014). In addition, sensitivity analyses are integral to understanding how model results may change if its inputs were varied. Neither calibration nor a sensitivity analysis were conducted in this study because the model was not run alongside other hydrology-based models. A full range of values for R_f was presented in lieu of a sensitivity analysis, though other inputs, such as effective porosity and hydraulic conductivity may be similarly sensitive. As the model inputs may be incorrect due to the use of literature values, the model may not reflect field conditions. The trends observed in the results take on a larger importance.

The model created to simulate 200 years of P loading from Auburn's septic systems to Lake Auburn relies on many assumptions described in Chapter III. Other limitations and potential improvements to the model are outlined in order to understand the ways in which the model may not reflect field conditions and inform future modeling approaches if this study were to be replicated with a larger focus on modeling.

D.1.2 Modeling Assumptions

Buffer Method Assumptions			
Modeling Approach	**Assumption**	**Implication**	**Source**
Only include systems within 300 ft (91.4 m) of the lake or major tributary.	Only systems within 300 ft (91.4 m) of the lake or major tributary contribute phosphorus due to the delay of P transport and various field studies that have found phosphorus plumes a maximum of 100m from the drain field.	Systems beyond the buffer are not included in the loading estimate though they may contribute P from hydraulic failure or from the subsurface, especially if a residence has existed there for over a century or the system is located on the aquifer.	**Literature Sources:** Harman et al. (1996); Robertson et al. (2019); Rakhimbekova et al. (2021) **Modeling:** CEI (2010); FB Environmental (2021)
Constant attenuation factor of 0.8.	All systems remove the same amount of phosphorus regardless of their placement. Use of an attenuation factor implies that systems have reached steady state.	Systems attenuate P variably based on soil depth and texture. Using a constant attenuation factor assumes that differences in soil types equal out to 80% attenuation on average. However, this approach obscures how soils across the landscape attenuate P differently, especially if certain soil characteristics are more common throughout the watershed than others, which leads to attenuation that is either more or less than 80% on average. Steady state implies that there is no variation in P retention over time– this only occurs once the septic system has been in operation long enough to reach various equilibria.	**Literature Sources:** Robertson (1995); Harman et al. (1996) **Modeling:** CEI (2010); FB Environmental (2021); Schellenger & Hellweger (2019); Oldfield et al. (2020b);
Use number of bedrooms instead of average number	Assumes that there is one person per bedroom, and that the number of bedrooms is an	There are varying amounts of people per bedroom in homes. Some homes have empty bedrooms while other homes have 2+ people per bedroom. However, using the number of bedrooms considers how	

Buffer Method Assumptions			
Modeling Approach	**Assumption**	**Implication**	**Source**
Only include systems within 300 ft (91.4 m) of the lake or major tributary.	Only systems within 300 ft (91.4 m) of the lake or major tributary contribute phosphorus due to the delay of P transport and various field studies that have found phosphorus plumes a maximum of 100m from the drain field.	Systems beyond the buffer are not included in the loading estimate though they may contribute P from hydraulic failure or from the subsurface, especially if a residence has existed there for over a century or the system is located on the aquifer.	**Literature Sources:** Harman et al. (1996); Robertson et al. (2019); Rakhimbekova et al. (2021) **Modeling:** CEI (2010); FB Environmental (2021)
Constant attenuation factor of 0.8.	All systems remove the same amount of phosphorus regardless of their placement. Use of an attenuation factor implies that systems have reached steady state.	Systems attenuate P variably based on soil depth and texture. Using a constant attenuation factor assumes that differences in soil types equal out to 80% attenuation on average. However, this approach obscures how soils across the landscape attenuate P differently, especially if certain soil characteristics are more common throughout the watershed than others, which leads to attenuation that is either more or less than 80% on average. Steady state implies that there is no variation in P retention over time– this only occurs once the septic system has been in operation long enough to reach various equilibria.	**Literature Sources:** Robertson (1995); Harman et al. (1996) **Modeling:** CEI (2010); FB Environmental (2021); Schellenger & Hellweger (2019); Oldfield et al. (2020b);
of people per household from U.S. Census Data	effective proxy for the number of people per home.	different masses of P may originate from different areas of the watershed based on household size. Moreover, septic system design standards are developed based on the number of bedrooms.	

Whole Watershed Method with Soil Variation and System Failure Assumptions			
Modeling Approach	**Assumption**	**Implication**	**Source**
Consider all systems within the watershed while using a transport coefficient of 0.24	All systems have the capability to contribute P to the lake. Due to setback distances and the retardation of phosphorus in groundwater compared to other wastewater constituents, not all systems are discharging to the lake. Therefore, it is assumed that 24% of P that reaches the groundwater from each system is being transported to the lake.	Some P plumes have reached the shoreline while others have not. Assuming that 24% of P that has reached groundwater is being transported to the lake does not reflect how P transport from septic systems to lakes actually occurs. However, it accounts for the fact that it is impossible to know which P plumes are discharging and which are not by assuming that 24% from each system reaches the lake. This inherently assumes that the soils beneath each system transport P the same way, which may not reflect field conditions. Furthermore, this value is based on field studies carried out in Canada, where watershed characteristics such as soils and development history differ from Auburn. Therefore, this transport coefficient is a great reflection of the watershed used in the field study, but may not accurately represent P transport in the Lake Auburn watershed. However, no field studies on septic systems exist for Auburn. This is the only method that indirectly considers P loss to a deeper aquifer system.	**Literature Sources:** Oldfield et al. (2020b); Spoelstra et al. (2020) **Modeling:** Gill & Mockler, 2016; Oldfield et al. (2020a)
Multiple attenuation factors based on soil permeability and vadose zone depth	OWS drain fields attenuate P differently based on soil texture and if they have at least a meter of unsaturated soil before the soil limiting factor	Different soil types attenuate phosphorus differently due to textural differences that dictate the ability of the soil to retain P. Having a shallow vadose zone depth also leads to worse P retention, with depths of over a meter being known to attenuate P more robustly. However, other variations in soils, such as the mineralogy, which dictates how much Fe, Al, and Ca are in the soil, also strongly influence P retention. Soil pH also has an effect on P retention. These factors are not accounted for.	**Literature Sources:** Zanini et al. (1998); Karathanasis et al. (2006); Robertson (2003); Gill et al. (2009); Mechtensimer & Toor (2016, 2017);; Baer et al. (2019); Robertson (2019) **Modeling:** Gill & Mockler (2016)

Varying percentages of P load attributed to failing systems based on age, setback distance and soil permeability	Failing systems contribute P through a surface pathway, where no attenuation is considered. The further a system is from the shoreline, less effluent travels through this pathway because some level of soil infiltration occurs over that distance.	Failing systems over 200m away from the shoreline are not modeled to contribute P through a surface pathway, even though they are failing and may contribute P in this way. Failing systems within 50m of the lake are assumed to contribute all of their effluent directly to the lake even though this may not be the case in reality. Low-permeability soils are considered to always be failing even though they may function properly and may fail more quickly than systems on other types of soil. The median age of replacement in auburn (29 years) is within range of median failure ages for Connecticut and Massachusetts.	**Literature Sources:** Hill & Frink (1980)**;** Withers et al. (2012); Withers et al. (2014) **Modeling:** Withers et al. (2012); Gill & Mockler (2016)

Temporal Model Assumptions			
Modeling Approach	**Assumption**	**Implication**	**Source**
Attenuation factors are used to simulate vadose zone removal processes. However, no removal is simulated to occur in the saturated zone.	There is robust P removal in the vadose zone as a result of precipitation reactions that occur less in the saturated zone. Adsorption processes are simulated to only delay the transport of P to the lake in groundwater. P that reaches the groundwater table is ultimately mobile until it discharges into the lake or tributary.	There is a reduced amount of permanent P retention in the saturated zone, though there inevitably is some. The modeling approach does not account for this small amount of attenuation. The model also does not account for P loss if water moves through an aquitard to a deeper aquifer system. Dispersion processes are also not accounted for. It is also assumed that all P discharged to groundwater reaches the lake, meaning that the interception of P plumes by wells is not considered.	**Literature Sources:** Todd (1980); Wilhelm et al. (1994); Harman et al. (1996); Robertson (2008); Robertson et al. (2019) **Modeling:** Roy et al. (2017); Oldfield et al. (2020b)
Applied retardation factors of 20 and 100 to pore-water velocity	Phosphorus is delayed in groundwater compared to more conservative constituents such	The phosphorus plume for each system moves slower than groundwater by a factor of 20 or 100 in the simulation. Using a value of 20 is likely to simulate phosphorus moving quicker than	**Literature Sources:** Bedient (1994); Robertson et al. (1998);

calculations using the Darcy equation to calculate the P-plume velocity beneath each septic system, which is used to determine when the plume discharges.	as nitrate, sodium, chloride, or artificial sweeteners. Therefore, it moves toward water bodies slower than the speed of groundwater. Values of the retardation factor vary between septic system operations, and no field studies exist in Auburn to inform what a value may be.	reality, because it is at the lower end of literature values. Similarly, using a value of 100 is likely to simulate phosphorus moving more slowly than in reality. Therefore, simulations using these values serve as a range of possibilities. It is impossible to know what a representative value for the retardation factor may be without field studies in this watershed.	Roy et al. (2017); Robertson et al. (2019); Rakhimbekova et al. (2021) **Modeling**: Roy et al. (2017); Schellenger & Hellweger (2019)
Adjusted the percentage of P load attributed to a failing system by multiplying the percentage by 1/29, the median age of system replacement in the Auburn portion of the Lake Auburn Watershed. Failing systems on low-permeability soils were kept the same.	Each septic system is assumed to fail for one year before it is replaced after its 29th year. Because of computation constraints, the percent attributed to failure is distributed equally across the entire 29 year lifespan of the system. (This is likely computationally possible with more advanced modeling). Failing systems are also assumed to be replaced within a year after they fail.	Instead of a system failing during a failure year and contributing more P through a surface pathway during that year, the mass of P from failure is distributed across the whole lifespan. This leads to overestimates if a system is not failing and underestimates on failure years. However, this approach accounts for the fact that it is impossible to know when a system is failing, and generalizes the effect of system failure as a result. This type of generalization is similar to modeling approaches that spatially lump septic systems into a single unit, which then assumes that they all fail on the same failure year (Sinclair et al., 2014). The benefit of the modeling approach used here is that, by evenly distributing the P load from system failure, variability over time from this factor is eliminated, allowing other trends to be revealed through the simulation. Moreover, the model assumes that each system is replaced almost as soon as it fails, which is a misrepresentation of reality, since systems may fail for many years before they are noticed or replaced. However, other models simulate the replacement of septic systems this way (Jeong et al., 2011). This would lead to an underestimate of the results. Assuming a failure age of 29 years may also inadequately capture how some systems fail very quickly after installation, while others operate for far longer	**Modeling:** Sinclair et al. (2014); Jeong et al. (2011)

		without failing.	
After the proposed ordinance change is simulated in 2023, systems are replaced in 2024 if they are older than 29 years old, or are replaced once they reach this age. Systems that have their age unknown are replaced on a random year between 2024 and 2053 (range of 29 years)..	The quick replacement of these systems assumes that the City of Auburn takes quick action to implement the proposed change through community outreach, education on septic system maintenance, and financial support. It also assumes that all systems are replaced within 29 years of the proposed change.	It is unlikely that the City of Auburn is able to conduct community outreach and identify failing systems such that all systems older than 29 years old are replaced within the year following the implementation of the ordinance. This would lead to the impact of the proposed change being observed sooner in the model than in actuality. Many systems will not be replaced until they are far older than 29 years old and after they have been failing for many years (Withers et al., 2014). Therefore, the impact of the proposed change will be observed quicker in the model than it would in actuality, and the nutrient load from failing systems may be underestimated. Use of median age of replacement in Auburn (29 years) is close to reported median lifespan of systems in Connecticut and Massachusetts.	**Literature Sources:** Hill & Frink (1980)
No adaptation of methods for cesspool use or other forms of wastewater disposal (outhouses)	Assumes that waste disposal through cesspools and outhouses contributed the same amount of P per person as is contributed through a traditional septic system, and that attenuation is the same.	P attenuation and loading to soils is not the same in cesspools and outhouses as it is in traditional septic systems. However, by the year in which the model has the most modern implications, traditional septic systems were widely used, and results from previous years are less applicable to modern questions, though wastewater treatment from these periods may impact model results far into the simulation. Historical data on wastewater quantities and P attenuation using cesspools/outhouses is not widely available. Therefore, integration of these forms of wastewater treatment would require more assumptions that may not be accurate.	**Modeling:** Schellenger & Hellweger (2019)

Use of the Darcy equation to estimate plume velocity with a constant dh/dl of 0.03	Change in hydraulic head was assumed to be 0.03 because no data on this is available. This was based on the median slope of all system locations based on site evaluations.	The change in hydraulic head influences the pore-water velocity. Assuming one value across the watershed assumes that areas of generally steeper slope and areas of gentler slopes behave the same; this is not true. However, other modeling approaches use slope as a proxy for change in hydraulic head, even though groundwater elevation does not necessarily reflect topography.	**Modeling:** Sinclair et al. (2014)
Use of the average number of people per home from the U.S. Census data, as determined by FB Environmental (2021)	2.325 people per home used in export coefficient calculation	No variability in household size is considered. This was chosen here because it is impossible to approximate household size over a 200 year period. Household population and the size of a home may change over time (additions), which means that using the number of bedrooms would likely lead to overestimations.	**Modeling:** FB Environmental (2021)

D.1.3 Model Limitations

Limitation: By only accounting for P attenuation in the unsaturated zone, similar to other studies (Oldifled et al., 2020a), the model presents a worst-case-scenario by assuming that all P that reaches the groundwater table moves ultimately toward Lake Auburn. Though there has been observed to be smaller amounts of attenuation occurring in the groundwater zone immediately below the disposal field (Robertson et al., 2019), the main shortcoming of this modeling approach is that it cannot account for loss of P to a deeper aquifer system or the removal of P if the plume is intercepted by a neighboring well. These factors are pertinent to this watershed because of the use of wells and the abundance of septic systems located on perched water tables with an restrictive layer, which may lose small amounts of water to a deeper aquifer.

Potential Solution: Use of the advection-dispersion-sorption (ADS) equation as used by Schellenger & Hellweger (2019) would account for the dispersion and sorption processes that are responsible for additional P retention in the saturated zone. Schellenger & Hellweger (2019) use the 1-dimensional ADS, though higher-dimension equations exist. If only one septic plume is being modeling, complex geochemical models exist that consider additional immobilization of P in the saturated zone (Parkhurst et al., 2003).

Sinclair et al. (2014) considers the loss of P to a deeper aquifer system by assuming that a percentage of P is lost. However, after sensitivity analysis with a range of 0-10 percent, they assume that 0% of P is lost to a deeper aquifer. This type of approach would be most applicable for improving the temporal model created in this study. To my knowledge, there are no studies quantifying P loss from the interception of a wastewater plume to neighboring wells.

Limitation: The use of soil survey data or literature values for hydraulic conductivity and effective porosity may not reflect field conditions, which would influence the results of the Darcy equation and the plume velocity calculations.

Potential Solution: Field sampling and laboratory analysis of soil samples collected throughout the watershed may help better assign values for these metrics, which would make application of the Darcy equation more precise. Furthermore, use of the "Darcy flow" tool in ArcGIS Pro would be helpful to creating a similar model.

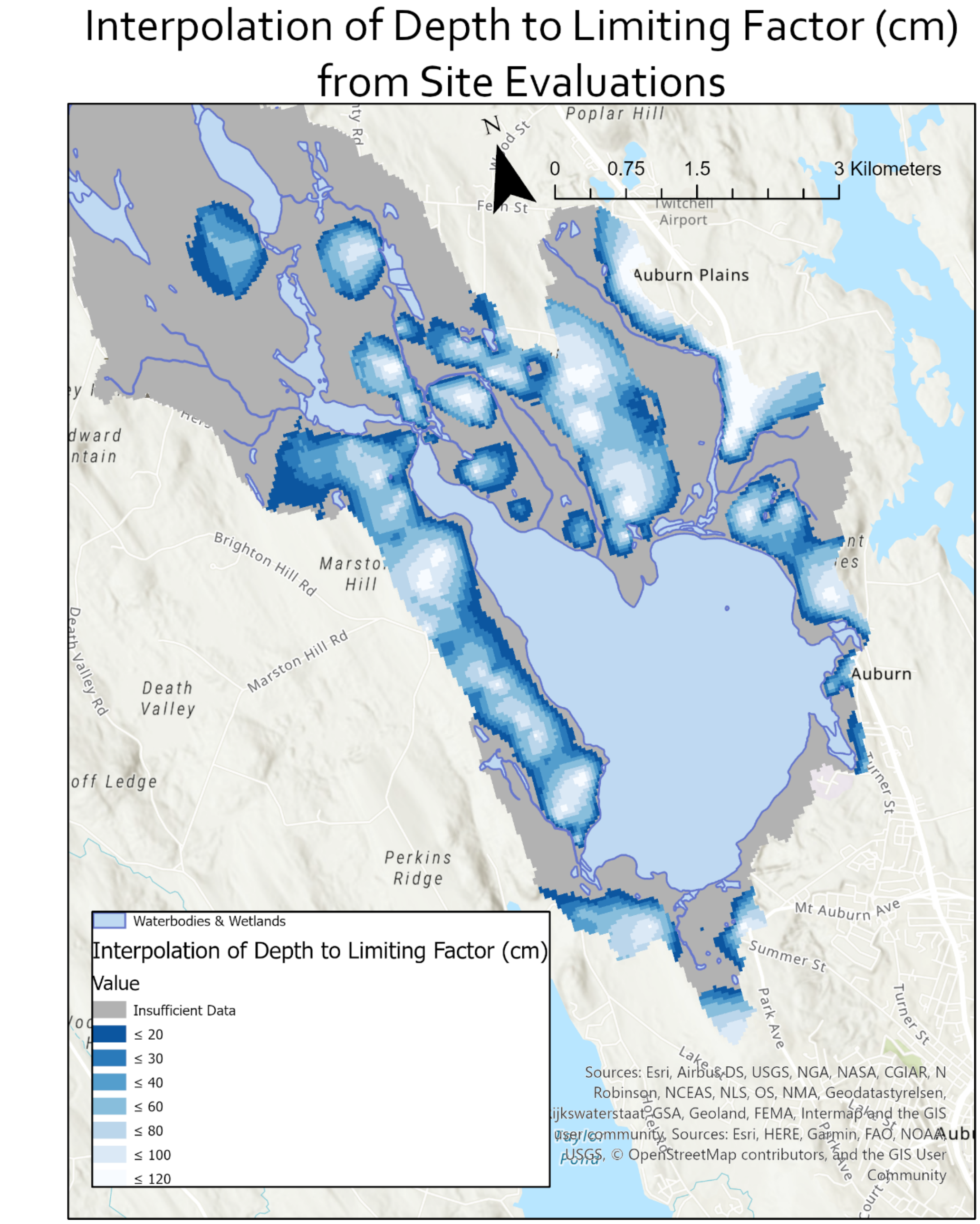
Limitation: The temporal model uses the straight-line setback distance from each septic system to the nearest lake or shoreline. However, groundwater and surface water do not usually flow in a straight line, and typically follow topography or a hydraulic gradient. The use of straight-line distance may underestimate the travel distance between each septic system and the nearest shoreline, which affects loading estimates used in this study.

Potential Solution: Accurate and correct use of the "Darcy flow" and "particle track" tools in ArcGIS Pro would allow for groundwater flow paths to be calculated, which would inform the groundwater path each P plume takes for well-functioning systems. For failing systems, the pathway could be determined using distance tools in ArcGIS Pro alongside a digital elevation model to obtain the flow path from the septic system and the nearest shoreline.

Limitation: None of the approaches in this study account for additional P attenuation in streams if the septic system is located near a stream instead of the Lake Auburn shoreline.

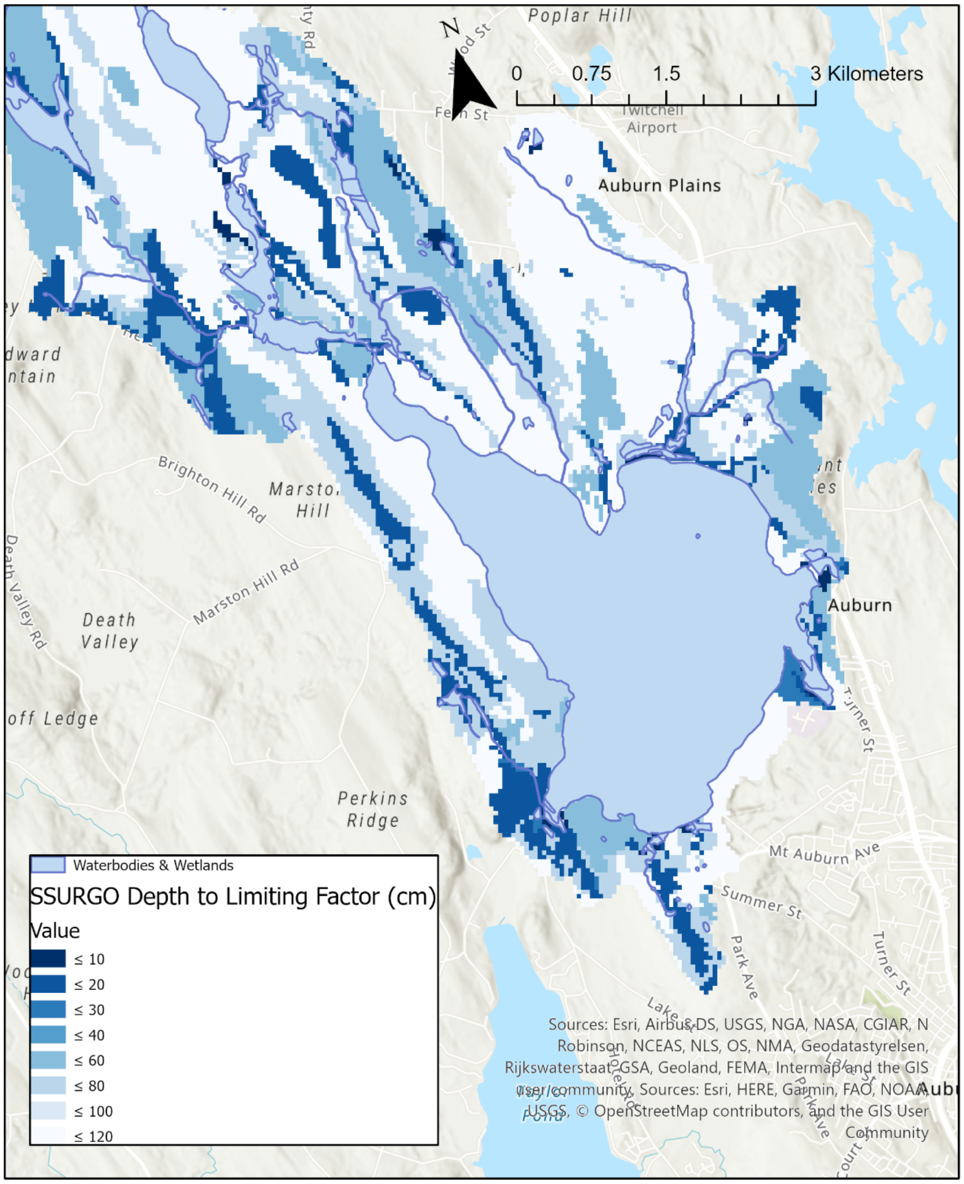
Potential Solution: Incorporating additional P attenuation in streams through an approach similar to FB Environmental (2021) is possible for this type of study. (Back to Section 4.3.3)

Appendix E: Maps Comparing Soil Survey Data and Site Evaluations

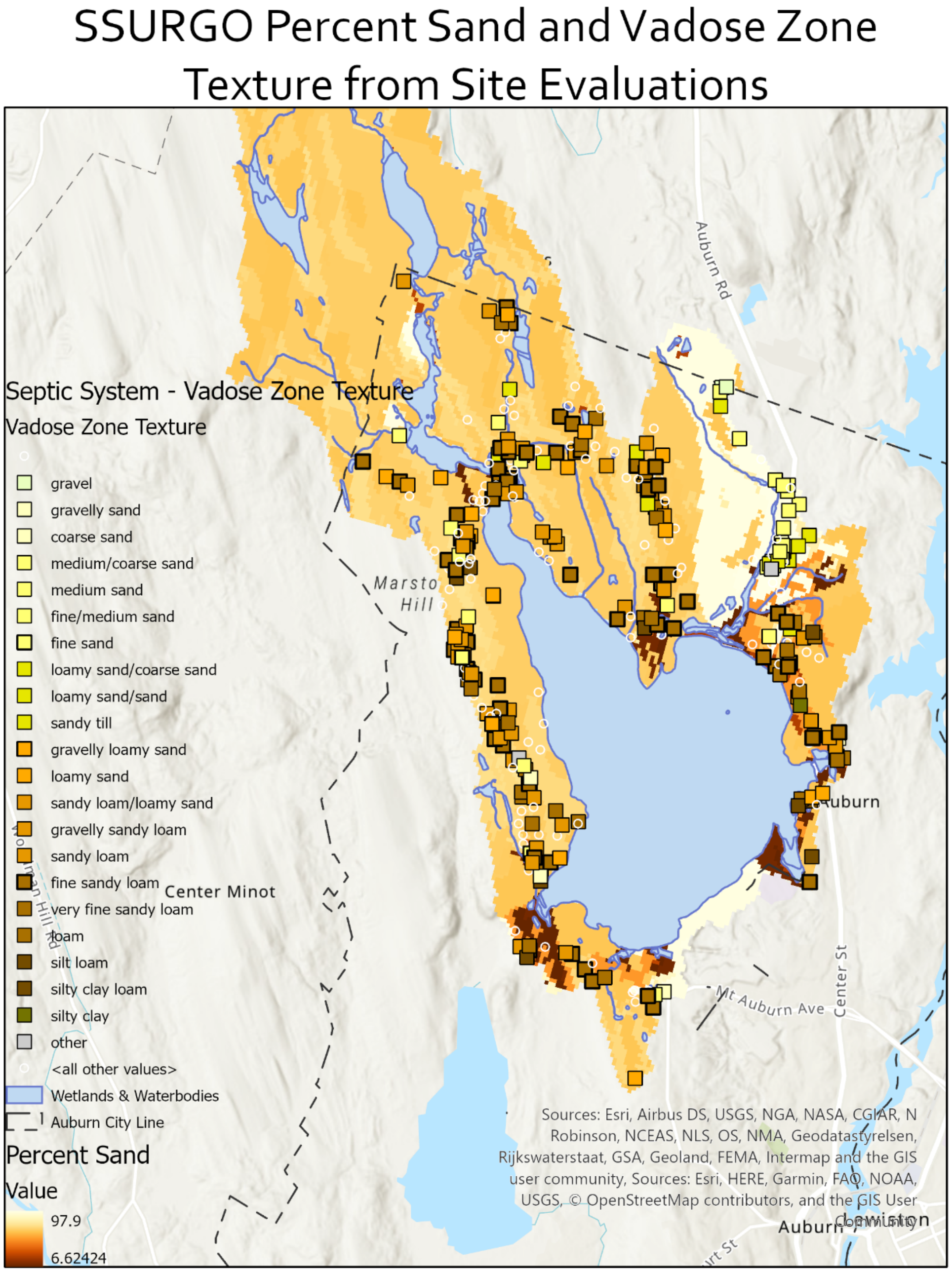


Map E.1: Depth to limiting factor from site evaluations interpolated across the Auburn portion of the watershed. Site evaluators found more variability than can be seen in soil survey data.

SSURGO Depth to Limiting Factor (cm)



Map E.2: Depth to limiting factor (cm) determined from SSURGO soil survey data throughout the watershed.



Map E.3: SSURGO weighted mean of percent sand for the entire soil profile and soil texture of the vadose zone as described during site evaluation. More variability exists in site evaluation than can be observed in soil survey data.

(Back to Section 4.1.5)

Appendix F: Suggestions for Future Field Studies

F.1 Possible Data to Collect

Field studies are integral to informing models, and carrying out a field study on septic systems in the Lake Auburn watershed could influence how P loading is quantified in this watershed, as well as identify areas of high P loading. Studies on P plumes provide the most information about soils and P transport that is useful for thinking about the cumulative effects of P loading on Lake Auburn. Many P plume studies use extensive networks of piezometers or monitoring wells to collect groundwater samples, as described by Cherry et al. (1983), Roy et al. (2017), Baer et al. (2019), and in Supporting Materials from Robertson et al. (2021). Data collection either takes place over a few days, capturing a singular moment, or periodically over the course of many years, to track the progression of a plume. Many studies also take advantage of previous studies that characterize groundwater movement within the area of interest. The following is a recounting of various data collected by researchers in the literature, and it is likely not feasible to collect all or even most of these data in a single study.

Potential Types of Sampling and Data Collection for Phosphorus Plume Assessment			
Data Type	Type of Sample	Reasoning	Source with Methods
Soil Characteristics			
Water Extractable (desorbable) P, major cations	Soil sample	Understanding the ion exchange capacity of the soil is important to understanding P retention capacity.	Mechtensimer & Toor (2016)

Acid-extractable / oxalate extractable Al, Fe, Ca, P	Soil sample	The amount of Al, Fe, and Ca are critical to understanding the mechanisms of P retention. Background P is necessary to understand changes in P masses after septic system operation.	Zanini et al. (1998); Eveborn et al. (2012)
Bulk density, texture	Soil sample	Texture is important to qualitatively understanding P retention capacity, while bulk-density is used in sorption isotherms and other equations.	
pH	Soil sample/pore water sample	pH strongly influences P retention.	
Vadose Zone Retention			
TP, SRP, TN, nitrate, ammonium concentration in effluent	Wastewater sample (with replicates)	Understanding the P concentration in effluent is essential to knowing what percentage of P is retained in the vadose zone.	Mechtensimer & Toor (2016)
TP, SRP, TN, nitrate, ammonium concentration at water table	Pore-water sample	Allows for quantification of vadose zone retention. Ammonium and nitrate allow for an understanding of the oxidation of wastewater.	Mechtensimer & Toor (2016)
Plume Characteristics			
Characterization of groundwater movement	Monitoring network	Allows for an understanding of the groundwater movement beneath the septic system, which influences plume velocity	Roy et al. (2017); Reide Corbett et al. (2002)

		and the movement of various constituents.	
pH, Eh, electrical conductivity, temperature	*In-situ* measurement (groundwater)	Septic effluent can have an acidifying effect on the surrounding soils, which can influence P retention and retardation depending upon the buffering capacity of the soil and the redox conditions. Electrical conductivity may be an effective tracer if other sources of ions are minimal.	Rakhimbekova et al. (2021); Robertson (2008)
Boron, artificial sweeteners, chloride, sodium, other cations	Groundwater sample	Tracers of wastewater. Artificial sweeteners can signal that wastewater is present. Tracers tend to move conservatively in groundwater, which allows the extent of the wastewater plume to be determined since the P plume is likely smaller.	Spoelstra et al. (2017, 2020); Oldfield et al. (2020); Roy et al. (2017); Wilhelm et al. (1994)
TP, SRP, TN, nitrate, ammonium	Groundwater sample	Nutrients limiting to lake ecosystems that may be of interest. Nitrate and ammonium can demonstrate how complete the oxidation of wastewater was in the vadose zone as well as reflect redox conditions. P concentrations signal how fast phosphorus may move in the system, while denitrification may occur to permanently remove N.	Robertson (2008);

(Back to Section 5.2)

F.2 Works Cited for Appendix F

Baer, S., Robertson, W., Spoelstra, J., & Schiff, S. (2019). Phosphorus and nitrogen loading to Lake Huron from septic systems at Grand Bend, ON. *Journal of Great Lakes Research*, *45*(3), 642–650. https://doi.org/10.1016/j.jglr.2019.03.003

Cherry, J. A., Gillham, R. W., Anderson, E. G., & Johnson, P. E. (1983). Migration of contaminants in groundwater at a landfill: A case study. *Journal of Hydrology*, *63*(1–2), 31–49. https://doi.org/10.1016/0022-1694(83)90222-6

Oldfield, L. E., Roy, J. W., & Robinson, C. E. (2020). Investigating the use of the artificial sweetener acesulfame to evaluate septic system inputs and their nutrient loads to streams at the watershed scale. *Journal of Hydrology*, *587*, 124918.

Rakhimbekova, S., O'Carroll, D. M., Oldfield, L. E., Ptacek, C. J., & Robinson, C. E. (2021). Spatiotemporal controls on septic system derived nutrients in a nearshore aquifer and their discharge to a large lake. *Science of The Total Environment*, *752*, 141262.

Robertson, W. D. (2008). Irreversible Phosphorus Sorption in Septic System Plumes? *Groundwater*, *46*(1), 51–60. https://doi.org/10.1111/j.1745-6584.2007.00389.x

Robertson, W. D., Van Stempvoort, D. R., & Schiff, S. L. (2019). Review of phosphorus attenuation in groundwater plumes from 24 septic systems. *Science of The Total Environment*, *692*, 640–652. https://doi.org/10.1016/j.scitotenv.2019.07.198

Roy, J. W., Spoelstra, J., Robertson, W. D., Klemt, W., & Schiff, S. L. (2017). Contribution of phosphorus to Georgian Bay from groundwater of a coastal beach town with decommissioned septic systems. *Journal of Great Lakes Research*, *43*(6), 1016–1029. https://doi.org/10.1016/j.jglr.2017.09.001

Spoelstra, J., Senger, N. D., & Schiff, S. L. (2017). Artificial Sweeteners Reveal Septic System Effluent in Rural Groundwater. *Journal of Environmental Quality*, *46*(6), 1434–1443. https://doi.org/10.2134/jeq2017.06.0233

Spoelstra, J., Schiff, S. L., & Brown, S. J. (2020). Septic systems contribute artificial sweeteners to streams through groundwater. *Journal of Hydrology X*, *7*, 100050. https://doi.org/10.1016/j.hydroa.2020.100050

Wilhelm, S. R., Schiff, S. L., & Robertson, W. D. (1994). Chemical fate and transport in a domestic septic system: Unsaturated and saturated zone geochemistry. *Environmental Toxicology and Chemistry*, *13*(2), 193–203. https://doi.org/10.1002/etc.56201302