The Origins and Dynamics of Phosphorus in Maine's Lake Auburn Watershed

An Honors Thesis Presented to
The Faculty of the Environmental Studies Program

Bates College

In partial fulfillment of the requirements for the
Degree of Bachelor of Arts

By

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Lewiston, Maine
March 31st, 2020

ACKNOWLEDGEMENTS

When I came to Bates, I did not expect to write a scientific thesis. I had enjoyed natural and field science throughout high school, but found it intimidating. Then, as a second semester first-year, I took Scientific Approaches to Environmental Issues with my future advisor, Holly Ewing. This was the class that, far more so than any other class I took at Bates, changed the trajectory of my academic, intellectual, and career interests. Holly thinks and talks about the natural sciences in a way that makes sense to me and has consistently gone out of her way to help me and my learning. Throughout our three classes together, a summer research job (when the groundwork for this thesis was laid), and the process of researching, writing and revising this thesis, she has approached my position as a relative newcomer to science with humor, generosity, and mentorship and pushed me to the edge of my intellectual abilities. Thank you, Holly. Thanks also to Ellen Labbe, my high school biology teacher, who introduced me to the natural and field sciences and piqued my interest in learning more in college.

Thank you to Dan Fortin, Chris Curtis, and everyone at the AWD/LWD for providing me with data, teaching me the basics of watershed sampling (both as a summer research assistant with Holly and this year), and answering all my questions. Thanks also to Camille Parrish for GIS and moral support, and the entire Environmental Studies Committee for mid-semester feedback.

Thank you to my parents for enabling me to spend most of my childhood outside. During these early years, I learned to observe, to love being outside, and to care deeply about the natural world. Thanks for your love and support, and for your interest in my disparate academic and non-academic pursuits while at Bates. Thank you also for the financial sacrifices you made to send me to Bates.

Thanks finally to my partner for the past two years of Bates, Ariel. Thanks for your conversation, your love, and your patience with my occasionally obsessive academic and non-academic interests. Thanks for coming with me to the Carnegie basement, for helping me solve an array of thesis-related problems, and for being my study buddy. I'll miss our countless hours in Commons, in PGill atrium, and 126 Wood Street, but am looking forward to our next adventures.

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ABSTRACT

Land-use change and climate change-induced phosphorus (P) loading is a key driver of eutrophication in temperate lakes, including in Lake Auburn, the drinking water supply for more than 60,000 people in Lewiston and Auburn, Maine. Without decisive action to halt declining water quality, the cities could lose their EPA water filtration waiver and be required to build a $45+ million filtration plant within the next decade. Reversing the decline in water quality requires the identification of P loading hotspots and the development of targeted intervention strategies which maximize impact, minimize required staff time, and conserve scarce municipal resources. An examination of data collected since 2005 by the Auburn Water District/Lewiston Water Division (AWD/LWD) reveals that P concentrations in most Lake Auburn inlets have increased slightly in the past 15 years and are currently high enough to be of concern. Using the USDA's Soil and Water Assessment Tool (SWAT), the 2019 P load was estimated to be 1671 kg, an increase of 198% from the estimated 1980 load of 560 kg. This increase reflects dramatic changes in land use and climate in the watershed over the past four decades. The P load is predicted to increase by an additional 66% to 2768 kg under the most-likely projected climate change and watershed development scenario. Thus, a rapid and sustained response is needed to avert regular summertime hypoxia. These results allow for a more nuanced understanding of current sources of P, as well as a prediction of where additional future loading is likely to originate, which will allow lake managers to both identify the most urgent and cost-effective solutions to current loading, and proactively implement measures to mitigate future loading.

1: INTRODUCTION

Eutrophication is a large and growing problem in lakes, rivers, estuaries, and coastal oceans worldwide (Smith 1998; Carpenter 2003; Novotny & Olem 1994; Bartsch 1970), with negative impacts including increased phytoplankton biomass, shifts to toxic and bloom-forming species (Smith 1998), increased water murkiness, drinking water treatment issues, oxygen depletion (Hutchinson 1973), and fish kills (Lee 1972). Phosphorus (P) is a key driver of eutrophication because it is often a growth-limiting nutrient: the least abundant nutrient relative to the needs of primary producers (Kalff 2002; Novotny & Olem 1994). Thus, excess P loading can lead to excess plant growth and decay, and eutrophication (Kalff 2002). Human-induced land-use change is the first of two major drivers of P loading and eutrophication. Known as cultural eutrophication (Reckhow et al. 1980), it is caused by human activities like agriculture and forestry (Duda 1993), residential development (Reckhow & Simpson 1980), roads, and construction (Kitchell & Sanford 1992; Reckhow et al. 1980).

Long-term shifts in climate, including past, present, and projected anthropogenic climate change, are a second key driver of eutrophication and P loading. Warmer temperatures affect mixing regimes (North et al. 2014), cause more rain-on-snow precipitation events (Soranno et al. 1997), and worsen hypoxia (Rolighed et al. 2016; Tolle et al. 2015; Feuchtmayr et al. 2009; Sahoo & Schladow 2008). Mean temperatures in New England are expected to be at least 2ºC warmer in 2050 than they are today (USGCRP 2018). In addition to increasing temperatures, extreme precipitation events are predicted to cause short periods of intense loading (Lathrop et al. 1997). Between 1958 and 2016, the Northeastern United States saw a 55% increase in extreme precipitation events, the largest increase of any region in the United States, and extreme precipitation events are expected to increase by another 50% by mid-century (USGCRP 2018).

The combined effects of climate change and land-use change are likely to make it more difficult for water quality to improve in eutrophic and mesotrophic lakes (Rolighed et al. 2016) and easier for oligotrophic lakes to become eutrophic (Feuchtmayr et al. 2009), largely due to temperature-induced increases in productivity and internal P loading. Thus, nutrient loads for many lakes may need to be reduced below historic, preindustrial levels to counter the generally negative impacts of climate change on water quality (North et al. 2014; Rolighed et al. 2016).

Lake Auburn is a 914 ha mesotrophic lake in Maine's Androscoggin County which has been suffering from cultural eutrophication for several decades (Dudley 2004). This is of particular and urgent concern because the lake serves as the drinking water supply for close to 60,000 people in the twin cities of Lewiston and Auburn (Dudley 2004). Compounding matters, Lake Auburn's historically excellent water quality has earned the Auburn Water District and Lewiston Water Division (AWD/LWD) a filtration waiver, which allows them to distribute Clean Water Act-compliant water without filtration (US EPA 1991; CDM Smith 2013; CEI 2010; Dudley 2004). Should water quality continue to decline, the AWD/LWD is likely to lose the filtration waiver and be required to build a $45+ million filtration facility (CDM Smith 2013).

Diminishing the likelihood that a filtration plant will be needed requires decisive action to limit eutrophication and, especially, P loading. This necessitates the identification of loading hotspots and the development of targeted intervention strategies which maximize impact, minimize required staff time, and conserve scarce municipal resources. Accounting for watershed inputs using watershed data and models is one way to broadly understand the origins and dynamics of P within a watershed (Lang et al. 1988; Scavia et al. 2019; Ding et al. 2014; Dillon 1975). In watersheds with limited data, models are often used to predict P movements based on knowledge of other lakes (Reckhow & Chapra 1983). The Soil and Water Assessment

Tool (SWAT), developed by the United States Department of Agriculture (USDA), is a model which uses data about soil, slope, land use, and weather to predict how land management decisions could impact water, sediment, nutrient loading, and agricultural yields in complex watersheds. SWAT offers greater nutrient loading prediction efficiency and less uncertainty than other models, even in unmonitored watersheds (USDA n.d.; Shendge & Chockalingam 2018).

This study uses SWAT and, to a lesser extent, the AWD/LWD data to address the need for a more detailed understanding of the origins of P in the Lake Auburn watershed and the following questions: (1) What can existing AWD/LWD data about P and discharge reveal about recent spatial and temporal patterns in P loading? (2) How does P loading vary across the Lake Auburn watershed? Where are the loading hotspots? What locations should be prioritized for P loading mitigation measures? (3) How have changes in land use and climate since 1980 impacted watershed P loads? (4) What effect is projected mid-century development and climate change predicted to have on future P loading? Using AWD/LWD data to understand individual sub-watershed dynamics and as a comparison for the model results, SWAT was used to predict the effects of land-use change and climate change on past, present, and future P loading. Three key SWAT scenarios examined P loading as a function of 1980 land-use and climate data, 2019 land-use and climate data, and projected 2050 land-use and climate data. Secondary scenarios examined projected 2050 climate change alone, projected 2050 land-use change alone, and doubled projected 2050 land-use change alone.

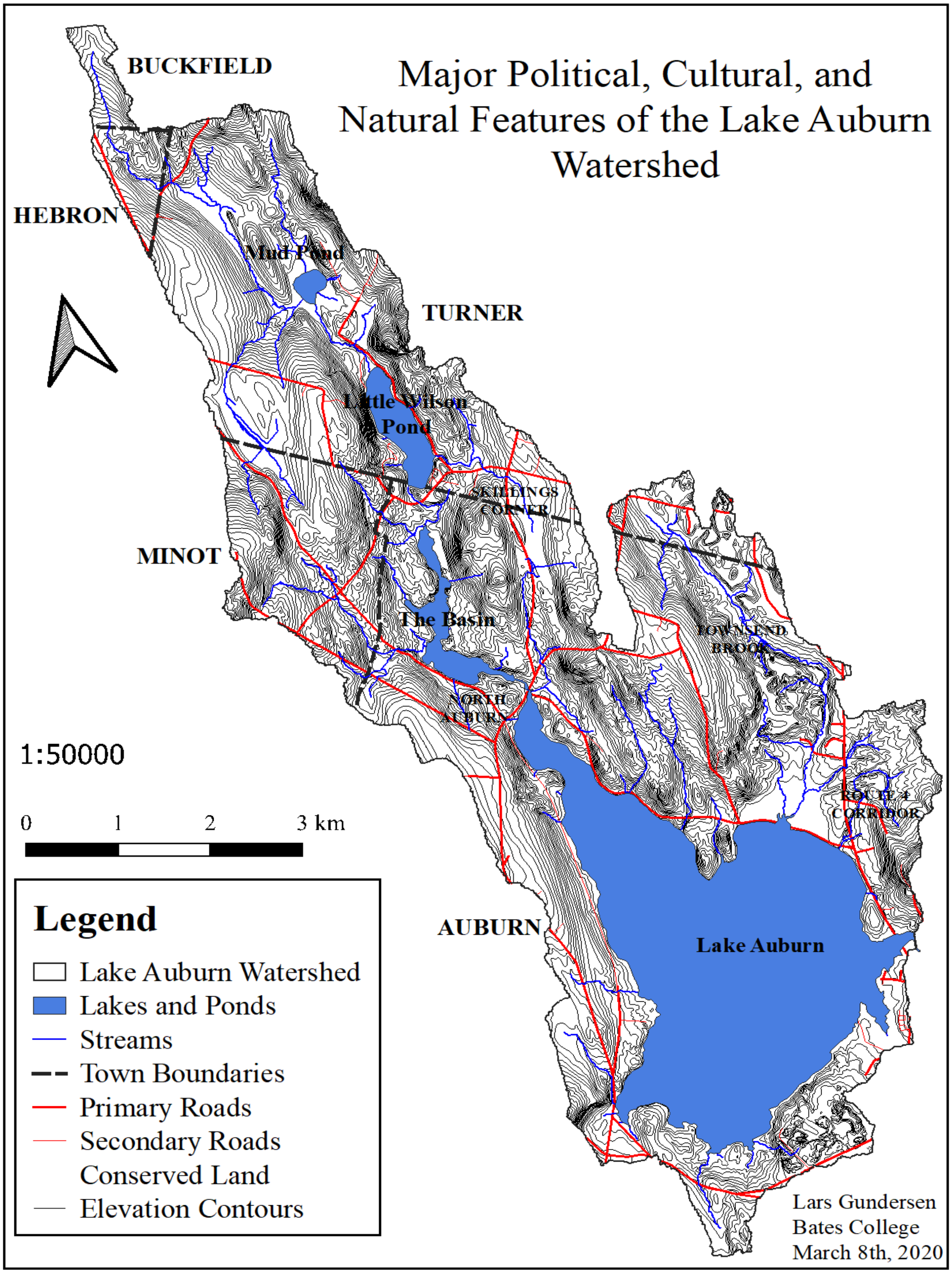
2: METHODS

2.1: ANALYSIS OF EXISTING DATA

2.1.1: Site Description and Contextualization

Lake Auburn is a historically oligotrophic, 914 ha mesotrophic lake in Maine's Androscoggin and Oxford counties (Dudley 2004). The lake has a mean elevation of 85 m, a total watershed area of 4740 ha, a maximum depth of 36 m, and a mean depth of 11 m. Lake Auburn is typically stratified from May to October (Dudley 2004). Since 1877, the lake has served as the drinking water supply for the twin cities Lewiston and Auburn, which have a combined population of about 60,000 (Dudley 2004). The cities have historically taken the maintenance of the lake's excellent water quality very seriously, implementing protection measures like a "No Bathing in the Lake" ordinance as early as the 1880s (CEI 2010). Since about 1900, the AWD/LWD have been responsible for watershed protection and water delivery, though many land conservation and educational outreach functions were shifted to the Lake Auburn Watershed Protection Commission (LAWPC) in the 1990s (CEI 2010).

The Lake Auburn watershed includes portions of the towns of Auburn, Turner, Minot, Hebron, and Buckfield (Map 1). Seventy-eight percent of the watershed is forested (mostly working forest), 10% is hay and pastureland, 4% is high-intensity development, 3% is wetland, 3% is low-intensity development, and 2% is cropland (CEI 2010). The LAWPC owns 81% of Lake Auburn's shoreline and 14% of the total watershed land, with an additional 7% of watershed land conserved through conservation easements and life estates (CEI 2010). In the late-1700s, virtually the entire watershed was cleared for farming. Agriculture continued for about 100 years before farmland started to be abandoned and forests started to regrow (Jones Associates, Incorporated (JAI) 2002). The AWD/LWD started buying land in the watershed for



Map 1: Major political, cultural, and natural features of the Lake Auburn Watershed.

conservation and forestry as farmland was abandoned and has managed more than 90% of its land for timber harvesting ever since (JAI 2002).

Lake Auburn has a mean water residence time of 4.1 years, and receives the majority of its water inputs (58-68%) from two streams: the Basin Stream, which drains three upstream ponds and provides the largest total contribution–albeit often inconsistently–and Townsend Brook, a cool, groundwater-fed, consistently-flowing stream (Hildreth 2008a; Hildreth 2008b; Dudley 2004). Direct precipitation totaling an average of 115 cm per year and other, intermittent streams provide the remaining 32-42% of total water inputs (Dudley 2004). Depending on annual precipitation patterns, the AWD/LWD withdraws 22-54% of the outflow. The non-evaporated remainder is released via a single, dammed outlet on the east side of the lake (Dudley 2004). Annual inflows and outflows each total about 27 million cubic meters (Dudley 2004).

Water quality in Lake Auburn has historically been excellent. Indeed, the AWD/LWD currently has an EPA filtration waiver, which allows it to distribute Clean Water Act-compliant water without filtration (US EPA 1991; CDM Smith 2013; CDM Smith 2014). To keep its filtration waiver, turbidity in Lake Auburn must not exceed 5 Nephelometric Turbidity Units (NTU) more than two times per year and more than five times in any ten-year period (US EPA 1991). These and other water quality stipulations have historically been easily attainable. Between 1977 and 2004, turbidity was consistently under 5 NTU, average Secchi readings were 7.4 m (range: 2-11 m), average total P readings were 8 µg/L (range: 7-14 µg/L), and average chlorophyll *a* concentrations were 2.8 µg/L (range: 1-17 µg/L) (Dudley 2004).

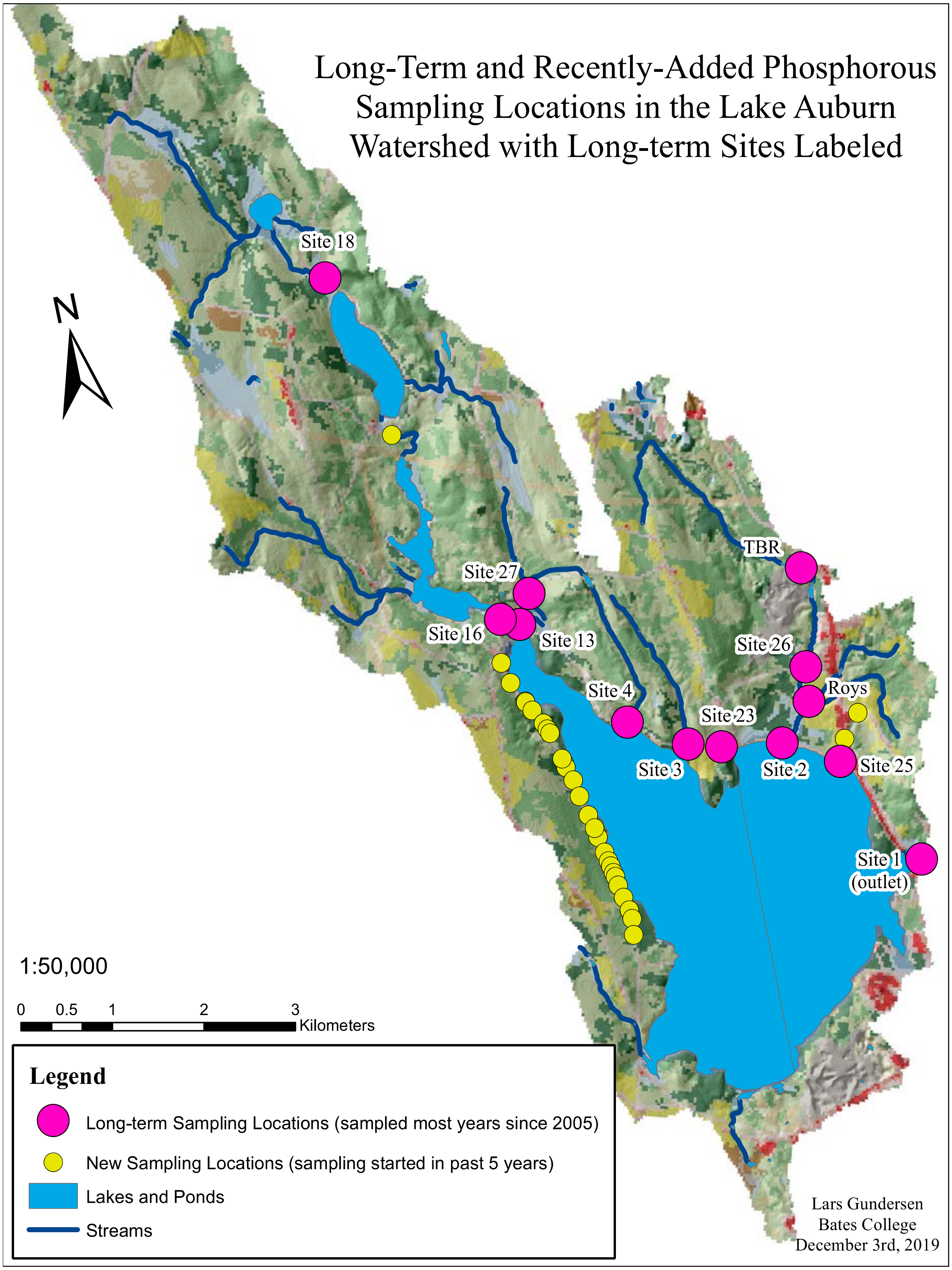
In 2011 and 2012, warmer temperatures, combined with storm events causing external nutrient loading, brought the long-term tenability of Lake Auburn's filtration waiver into question. Turbidity increased dramatically, despite remaining under 5 NTU, and there were cyanobacterial blooms and a trout kill (CDM Smith 2013; CDM Smith 2014). Total P concentrations in the epilimnion exceeded 14 µg/L, nearly double the long-term average, exacerbated perhaps by temporary mid-summer destratification which could have allowed P-rich hypolimnetic water to reach the surface and prompt an explosion in algal growth (CDM Smith 2013; CDM Smith 2014). This period of extreme loading in 2011 and 2012 solidified Lake Auburn's mesotrophic status (CEI 2010) and prompted an expansion in watershed sampling and protection measures. These events also raised the prospect that the AWD/LWD could lose its filtration waiver and be required to build a $45+ million filtration plant (CDM Smith 2013) (see Appendix A.1-A.3 for descriptions of eutrophication and nutrient loading and Appendix A.4-A.5 for descriptions of watershed protection measures and lake management options).

2.1.2: Existing Dataset Description

The AWD/LWD has 13 long-term P sampling sites (see Appendix B, ST 1 for full site descriptions). Most sites were sampled in 2005, and in most years from 2007 to the present. Any given year is missing data from between one and four sites. Additional sites have been added, mostly in the past two years, such that in 2019, there were 41 sites sampled (Map 2). However, the original 13 sites are sampled more frequently (8-10 times in 2019 versus 2-4 times for newer sites). After extensive compilation and reorganization (see Appendix C.1 for details), the dataset was analyzed, summarized, graphed, and mapped, with an emphasis on understanding how P concentrations and estimated loads vary within and across sites.

2.1.3: Phosphorus Concentration Analysis

Phosphorus concentrations were summarized and mapped to understand temporal variations across years and between months. Interannual variability was summarized as the range in P concentrations across all years from 2005 to 2019. Long-term means and minima were also



Map 2: Long-term and recently-added Water District Sampling locations, with long-term sites labeled. Long-term sites have more than 5 years of data. See Map 1 for broader watershed orientation.

calculated. Mean was calculated by taking the mean of each yearly mean in order to give data from each year equal weight regardless of how many times a site was sampled in a given year. Additionally, mean P concentration minima, means, maxima, and ranges were calculated for each month when sampling occurred (April to October) in order to understand seasonal trends across years. Again, a value for each month in each year was calculated before calculating an average across years, in order to give values from each year equal weight. These data were then mapped and graphed to understand how P concentrations vary across and between sites, both across and within years.

Spatial variability in P concentrations across sites was summarized and mapped as a yearly minimum, mean, and maximum for each site in each year. Additionally, average long-term minimum, mean, and maximum values (2005 and 2007-2018) were calculated for each year separately, and then as a single long-term average for each site with more than five years of data. Nine of the 13 long-term sites are missing data only from 2006, while the remaining four sites are missing data from 2006 as well as several other years (see Appendix B, ST 1 for and information on missing data). Sites with less than five years of data, which have mostly been added in the past few years, were excluded from the long-term analysis.

2.1.4: Phosphorus Load Calculations

Phosphorus load was estimated for streams where the AWD/LWD collects both discharge and P concentration data. Load was calculated for each site in which concurrent discharge and total P data have been collected at least once in 2019. Ten sites had these data for 2019, and they were all sampled between six and 10 times. Phosphorus load was calculated using two methods. The first method used a yearly mean concentration and discharge value for each site and was calculated using the formula:

$$\left(\left(avg\ annual\ discharge\left(\frac{m^3}{\text{sec}}\right)\right)x\ \left(avg\ annual\ [P]\ \left(\frac{kg}{m^3}\right)\right)\right)x\ \left(\left(\frac{60\ seconds}{\text{minute}}\right)\left(\frac{60\ minutes}{hour}\right)\left(\frac{24\ hours}{day}\right)\left(\frac{x\ days}{year}\right)\right)$$

The second method distributed concentration and discharge values to the midpoint day between adjacent sampling events and was calculated using the formula:

$$\left(\left(\left(discharge\ \left(\frac{m^3}{\text{sec}}\right)\right)x\ \left([P]\left(\frac{kg}{m^3}\right)\right)\right)x\ \left((days\ b/t\ midpts)\left(\frac{24\ hours}{day}\right)\left(\frac{60\ min}{hour}\right)\left(\frac{60\ sec}{\text{min}}\right)\right)\right) + (all\ other\ sectional\ loads)$$

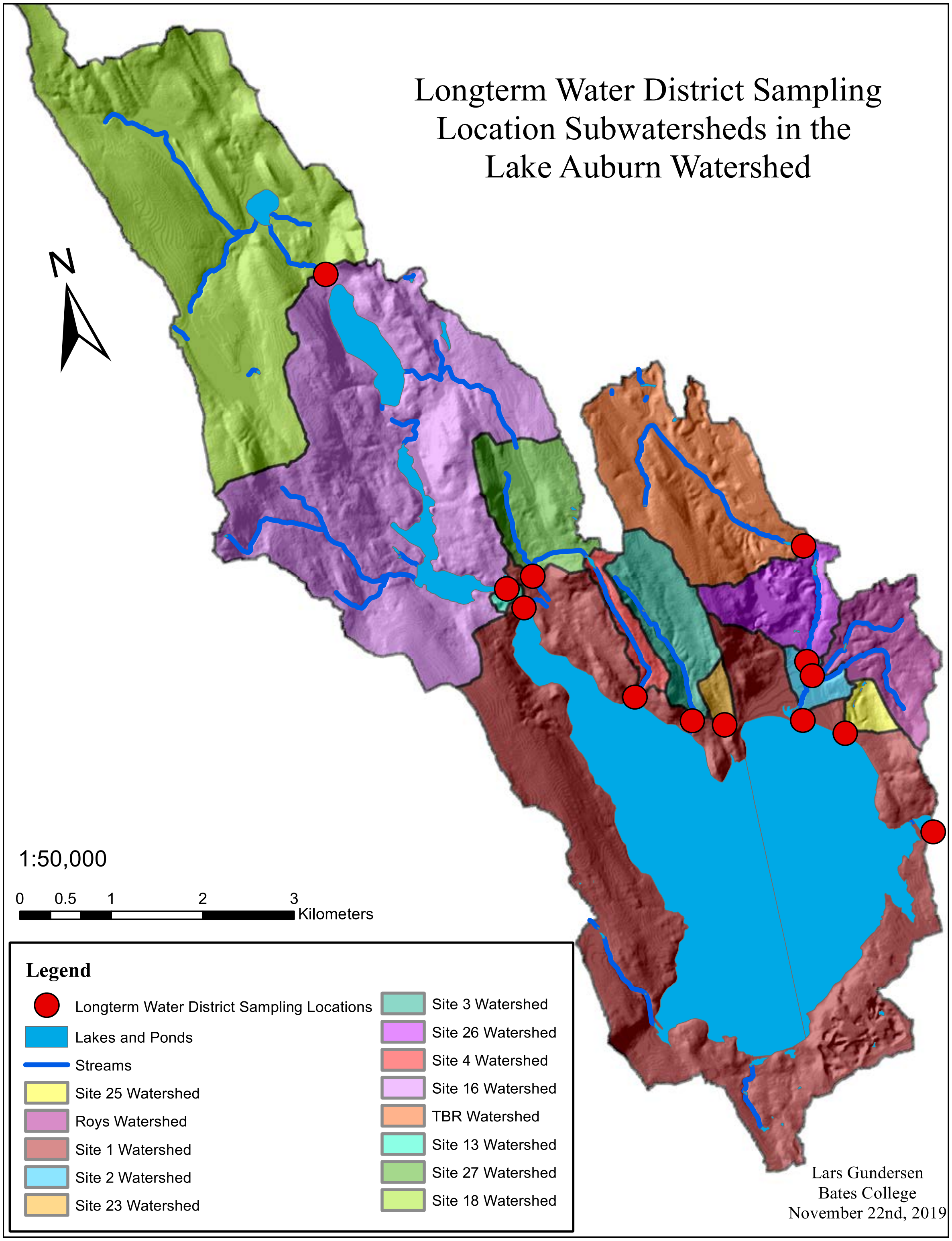
Both methods used two variations. One calculation assumed loading occurs year-round (x days = 365), and one assumed loading occurs during the ice-free season only (x days = 274) (see Appendix C.2, Figure 7a and 7b for more detailed calculation methods and Appendix B, ST 2 and 3 for full calculations). Load was calculated for sampling locations which are upstream of another sampling location, but these sites (Sites 18, 27, B-1, and R-2) were excluded from the final calculation to prevent double-counts.

As part of the calculation of total P load, the relationship between total P concentration and discharge was evaluated. For this analysis, only data from the two largest outlets (the Townsend Brook outlet (Site 2) and the Basin outlet (Site 13)) were included. These outlets have meters which collect 15-minute interval discharge data that can be merged with P concentration data. Thus, total P concentration values were merged with the nearest 15-minute interval discharge data by site, date, and time. For both sites, there were nearly as many matches as there were P concentration values (some discharge values were missing). There were at least three matches for every year since 2005 (excluding 2006), and often 10 or more matches in recent years. Non-match discharge data were excluded, as were several outliers which were recorded in error per AWD/LWD staff (see Appendix C.1).

2.1.5: The Relationship between Phosphorus Concentrations and Land Cover

The sub-watershed land draining through each long-term sampling location was identified in order to understand the effects of varying land covers across the watershed. A sub-watershed pour point was placed in the location of each sampling location with more than five years of data and the total area of each sub-watershed falling into each land cover category, in square meters, was calculated (Map 3; see Appendix C.1 for additional information on data compilation, analyses, and manipulations for maps and figures). Additionally, land covers were grouped into three broader "land cover classes" which broadly conform to the land cover classes used in the SWAT model introduced later: “developed” (a composite of “developed, open space;” “developed, low intensity;” “developed, medium intensity;” “developed, high intensity;” and “barren industrial land”); “agricultural” (“hay and pasture;” and “cultivated crops”); and “undeveloped” (“deciduous forest;” “evergreen forest;” “mixed forest;” “shrub/scrub;” “herbaceous;” “woody wetlands;” and “emergent herbaceous wetlands”). The “open water” land cover class was excluded from the analysis.

To compare P concentration and land cover across sub-watersheds, each broader land cover class was summed for each sub-watershed to understand sub- watershed land cover class distribution. The three broad land cover classes for each sub-watershed were then compared to AWD/LWD P concentration data. Relationships between land cover and P concentrations were compared to understand how the proportion of agricultural, developed, and undeveloped land in each sub-watershed affects P concentrations.



Map 3: Long-term sampling location sub-watersheds. A pour point was placed in the location of each long-term sampling location in order to delineate the sub-watershed area which drains through each sampling location.

2.2: Soil and Water Assessment Tool (SWAT) Watershed Analysis

2.2.1: Introduction to SWAT

The Soil and Water Assessment Tool (SWAT), developed by the US Department of Agriculture (USDA), is a model which uses inputs like soil, slope, land use, and weather to predict how land management decisions could impact water, sediment, nutrient loading, and agricultural yields in complex watersheds. It is popular due to its simplicity (ability to be used by non-experts), predictability (it has been validated in over 1000 temperate watersheds of all sizes), and stability (ease of running and lack of bugs) (USDA n.d.; Shendge & Chockalingam 2018; Zhang et al. 2019). SWAT offers greater nutrient loading prediction efficiency and less uncertainty than other models, even in unmonitored watersheds (USDA n.d.; Shendge & Chockalingam 2018) and was thus chosen for application to the Lake Auburn watershed (see Appendix A.6 for a fuller description of models and modelling and Appendix D for an explanation and depiction of why a model-based analysis was considered to be more appropriate than an existing data-based analysis).

2.2.2: Identification of Data Sources

SWAT relies on four key inputs: A digital elevation model (DEM) for topography, a land use raster layer, a soil raster layer, and hourly weather data for the time period of interest. While weather data can be downloaded from Texas A&M University as a properly-formatted spreadsheet, other inputs must come from external sources. The inputs used in this analysis are a 1/9 arc-second DEM, a 2016 land use raster, and a SSURGO soil raster (for all data source information, see Works Cited). Land use and soil data also require a properly-formatted lookup table with land use and soil type definitions and values. A default SSURGO table which automatically links to all SSURGO layers can be used as the soil lookup table, but land use data must be reclassified into SWAT land use classes (for reclassification choices for all scenarios, see Appendix B, ST 4). This table must be created separately, then joined with SWAT land use tables, each of which contain dozens of data points about land use class characteristics.

2.2.3: Default SWAT Scenario

The SWAT default scenario used the thresholds and inputs established by the SWAT demonstration scenario, which is automatically downloaded with the SWAT+ extension for QGIS, the GIS program most compatible with SWAT. Most of these default values have been tested and calibrated across a variety of watersheds, are recommended for use in all temperate watersheds, and are utilized by most SWAT users. Except for the land use, weather, and timeframe changes outlined in the next section, all analyses otherwise used default settings (Appendix B, ST 5). In addition to providing a prediction of current watershed-wide P concentrations and P loading, the outputs from the default analysis provide a baseline for comparison to scenarios with adjusted land use, weather, and/or timeframe data.

2.2.4: SWAT Scenarios

The three main ways in which scenarios can be created in SWAT is through land use, weather, and timeframe manipulations: Land uses can be split or excluded to simulate land-use change, weather data can be adjusted to reflect projected changes in climate, like rising temperatures and increased extreme precipitation, and past land use and weather data can be used to manipulate the timeframe and understand past changes in land use and climate. The six scenarios created in this analysis all rely on land use, weather and/or timeframe manipulations (Table 1).

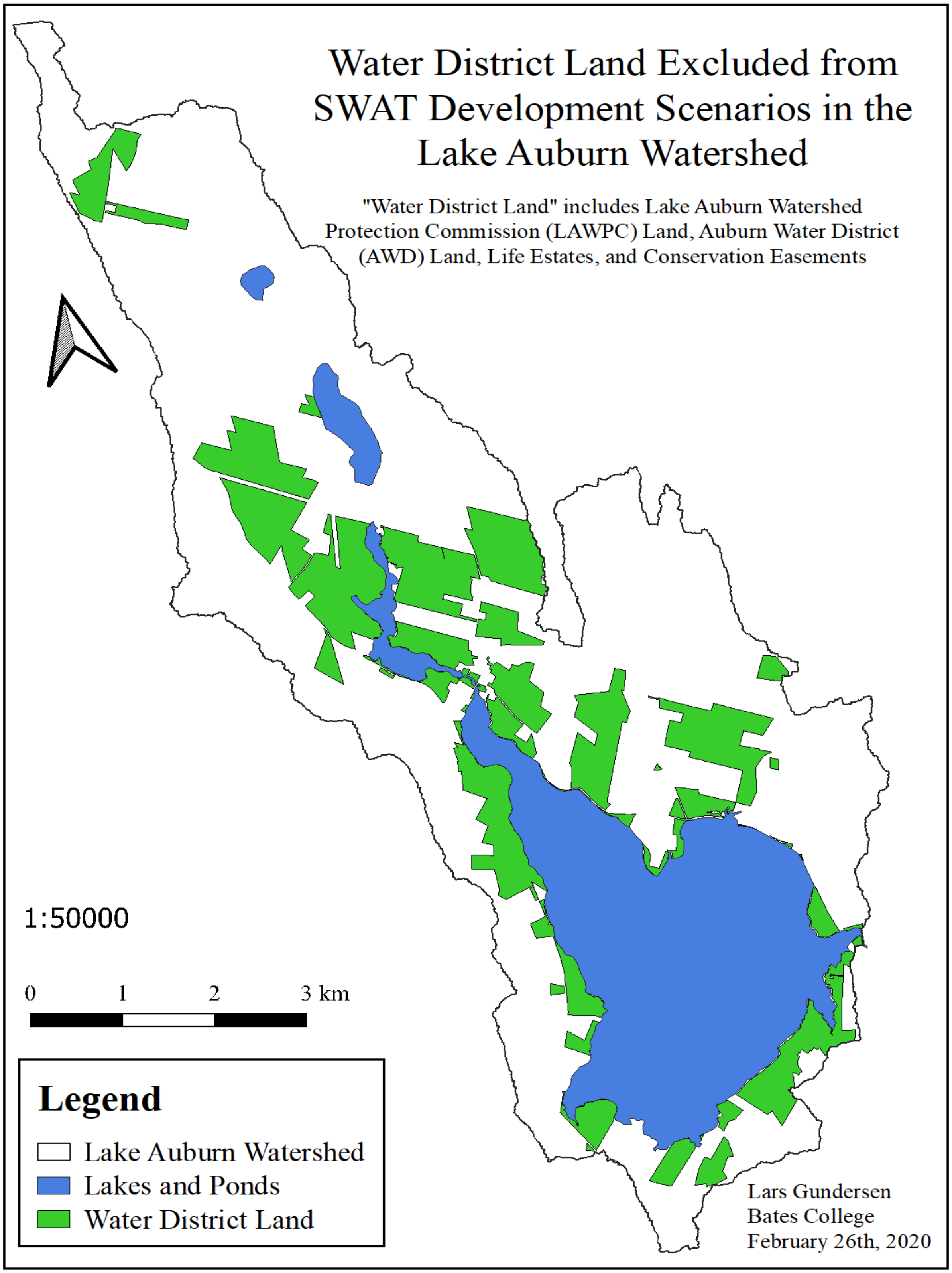
The City of Auburn projects that between 2010 and 2050, there will be 480 ha of development in the Lake Auburn watershed (City of Auburn Ordinance Chapter 60, 2010). To simulate this development, certain land uses were split to predict the impact of future development. There is a

Scenario	Scenario Description	Land Use Manipulations	Weather Manipulations	Timeframe Manipulations
Default Scenario with 2019 Data	Designed to predict current P loading using 2019 weather and land use data.	None (Water District land excluded).	None (2019 weather data used).	Prediction year: 2019
Default Scenario with 1980 Data	Designed to predict 1980 P loading using 1980 weather and land use data.	None (Water District land excluded).	None (1980 weather data used).	Prediction year: 1980
Projected Mid-Century Development Scenario	Designed to predict the impact of 480 ha of development on P loading using Auburn development estimates for 2050.	19% of land not developed or owned by the Water District designated as developed.	None (2019 weather data used).	Prediction year: 2050
Doubled Projected Development Scenario	Designed to predict the impact of 960 ha of development on P loading using doubled Auburn development estimates for 2050.	38% of land not developed or owned by the Water District designated as developed.	None (2019 weather data used).	Prediction year: 2050
Mid-Century Climate Change Scenario	Designed to predict the impact of climate change on P loading using mean climate change projections for 2050.	None (Water District land excluded).	Projected 2050 weather data used.	Prediction year: 2050
Development and Climate Change Scenario	Designed to predict the impact of climate change and projected development on P loading.	19% of land not developed or owned by the Water District designated as developed.	Projected 2050 weather data used.	Prediction year: 2050

Table 1: Scenario descriptions.

total of 3832 ha of land in the watershed (Appendix B, ST 6). Ten percent of it (343 ha) is already developed and is therefore not developable. Twenty-one percent (804ha) is owned by the LAWPC or is in conservation easements or life estates, meaning it will not be developed (this land is collectively referred to as "Water District land") (Map 4). Six percent of the land that remains (70 ha) is wetland that cannot be developed per city ordinance (City of Auburn Ordinance Chapter 60, 2010). Thus, there are 2513 ha of land in the watershed which could, theoretically, be developed.

After subtracting Water District land, developed land, and wetlands, there are 66 ha of barren land, 638 ha of deciduous forest, 286 ha of evergreen forest, 1081 ha of mixed forest, 96 ha of shrub/scrub, 24 ha of herbaceous grassland, 293 ha of hay/pasture, and 27 ha of cultivated crops



Map 4: Water district land excluded from SWAT development scenarios in the Lake Auburn watershed.

which could potentially be developed (Appendix B, ST 6). Thus, of the 2513 ha available for development, 3% is barren land, 25% is deciduous forest, 11% is evergreen forest, 43% is mixed forest, 4% is shrub/scrub, 1% is herbaceous, 12% is hay/pasture, and 1% is cultivated crops. Assuming that development will be divided across land uses based on their respective proportions of the developable land, 19% of each developable land cover class was reclassified as URBN (mixed residential). For the doubled development scenario, 38% of each developable land cover class was reclassified as URBN (mixed residential).

When land uses are split in SWAT, the model randomly assigns the land which it reclassifies based on the land which is "most likely" to be developed. Thus, land adjacent to existing development (such as land along roadways, and near existing residential areas) is developed first. As greater proportions of land are reclassified as developed, the model develops land further outward from existing development. It is important to note that despite these built-in steps to develop the "most likely" areas first, the land SWAT designates as developed is random. It is impossible to designate which areas are included in a land use split manually. Thus, though the development scenarios provide a well-calibrated overall estimate of load, some specific areas where the model predicts high loading may not reflect real-world conditions. In other words, in the development scenarios, some areas showing high loading may show high loading due to SWAT land use split assignments, rather than inherent conditions. Thus, watershed management decisions should not be based solely on development scenario outputs.

Projected climate change scenarios are simpler to simulate in SWAT. Under its mean projection, the National Oceanic and Atmospheric Administration (NOAA) predicts that extreme precipitation events will double, total rainfall will increase by about 25%, and temperatures will increase by 2ºC by 2050 in New England (USGCRP 2018; Michon 2019). To simulate these changes in climate, weather data were adjusted based on these assumptions. The mean monthly precipitation days and hours were cut in half to simulate a doubling of precipitation intensity, total mean monthly precipitation was increased by 25%, and mean monthly high and low temperatures were increased by 2ºC. Based on adjustments in mean monthly values, SWAT then extrapolates these changes to the tables with daily and hourly data, which come preinstalled in the program and do not require editing. The projected climate change scenario thus used these weather adjustments plus the original, unsplit land use data, while the development and climate change scenario used these weather adjustments plus the projected development land use splits (Table 1) (see Appendix A.7 for a description of projected climate change).

Past land use change is also simple to simulate in SWAT. A raster layer from the desired year may be added along with a lookup table. Thus, for the 1980s loading scenario, a land use raster from 1980 was added instead of the most recent land use raster, and a new lookup table matching the land use raster was added. The default scenario was then followed, except that the required weather data were 1977-1980, instead of 2016-2019 (Table 1).

2.3: Methods of SWAT Results, Existing Data, and Watershed Reports Comparison

Estimates of total annual P load provide the primary means of comparison both across SWAT scenarios and between this analysis and other Lake Auburn nutrient load estimates. The "Lake Auburn Watershed Management Plan" developed by Comprehensive Environmental Inc (CEI) in 2010 and the two-part "Diagnostic Study of Lake Auburn and its Watershed" follow-up report written by CDM Smith and CEI in 2013 are the main existing full-watershed reports. Thus, the total P load estimates created using SWAT will be compared to the total P load estimates developed by CEI and CDM. Both analyses use export coefficients (see Appendix A.6.3 and Appendix B, ST 7-9 for information on export coefficients), which account only for landcover, meaning the results are not based on the same assumptions as the SWAT results, which also account for soil, slope, and weather data. However, these results provide the only other estimates of P load for Lake Auburn and are therefore useful to help put the SWAT analysis results in context. For the analysis of existing AWD/LWD data, P load will be compared to CEI stream loading estimates only, as the CDM report did not break down P load by source.

The methods for estimating the total P load from streamflow were discussed above and were summed for all outlets to generate a total estimate (upstream sampling locations were excluded to prevent double counts). This estimate covers only the streams for which there are AWD/LWD data for 2019; thus, a few smaller streams were necessarily excluded from this analysis. The SWAT outputs are given in two formats: Hydrologic Response Units (HRUs) which unite adjacent cells with the same soil and land use, and Landscape Units (LSUs) which unite adjacent cells with the same landscape (floodplain or highland) and slope. Because land use varies more rapidly in most parts of the watershed than landscape and slope, most HRUs are far smaller than the LSUs (the Lake Auburn watershed contains about 300 LSUs and about 10,000 HRUs).

Though the total load estimates in this case varied by less than 1%, the SWAT developers recommend calculating load based on LSUs because LSUs neutralize many of the estimation extremes found in some HRUs, particularly in more variable watersheds. Thus, the five LSU output layers were loaded into GIS and joined into one sheet. After calculating geometry for each LSU (area in m2), load was calculated using the formula:

$$(Area\ (m^2)/10{,}000) \times\ (total\ phosphorus\ concentration\ \left(\frac{kg}{ha}\right))$$

After calculating the total load for each LSU, these values were summed by scenario to calculate the total P load for the watershed under each of the six scenarios detailed above (Table 1).

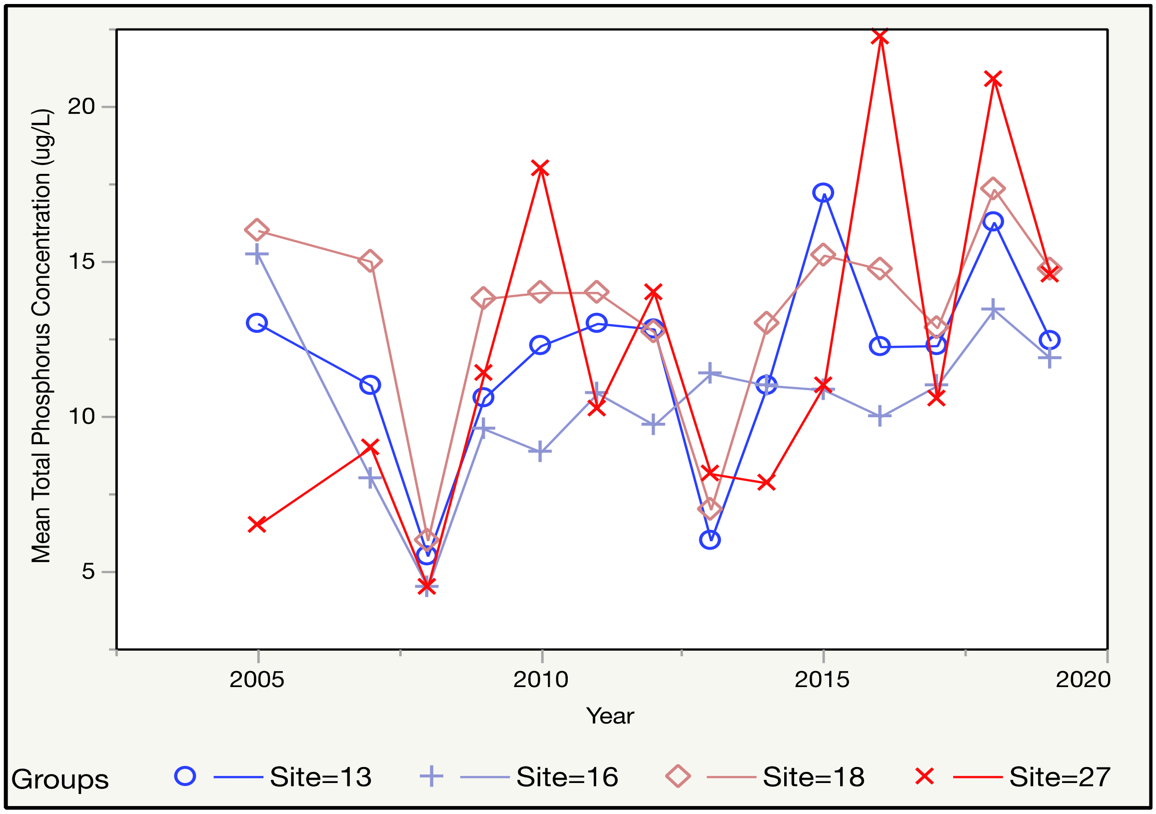
3: RESULTS

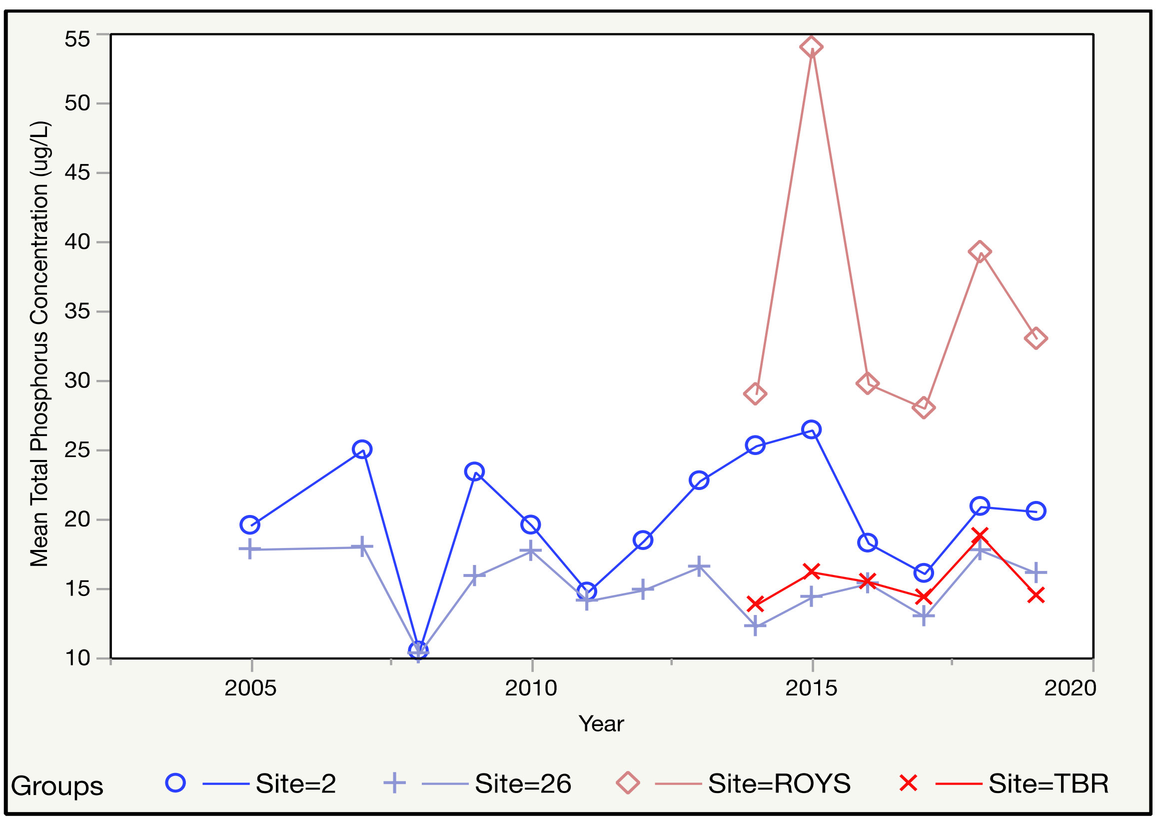
3.1: RESULTS OF THE ANALYSIS OF EXISTING DATA

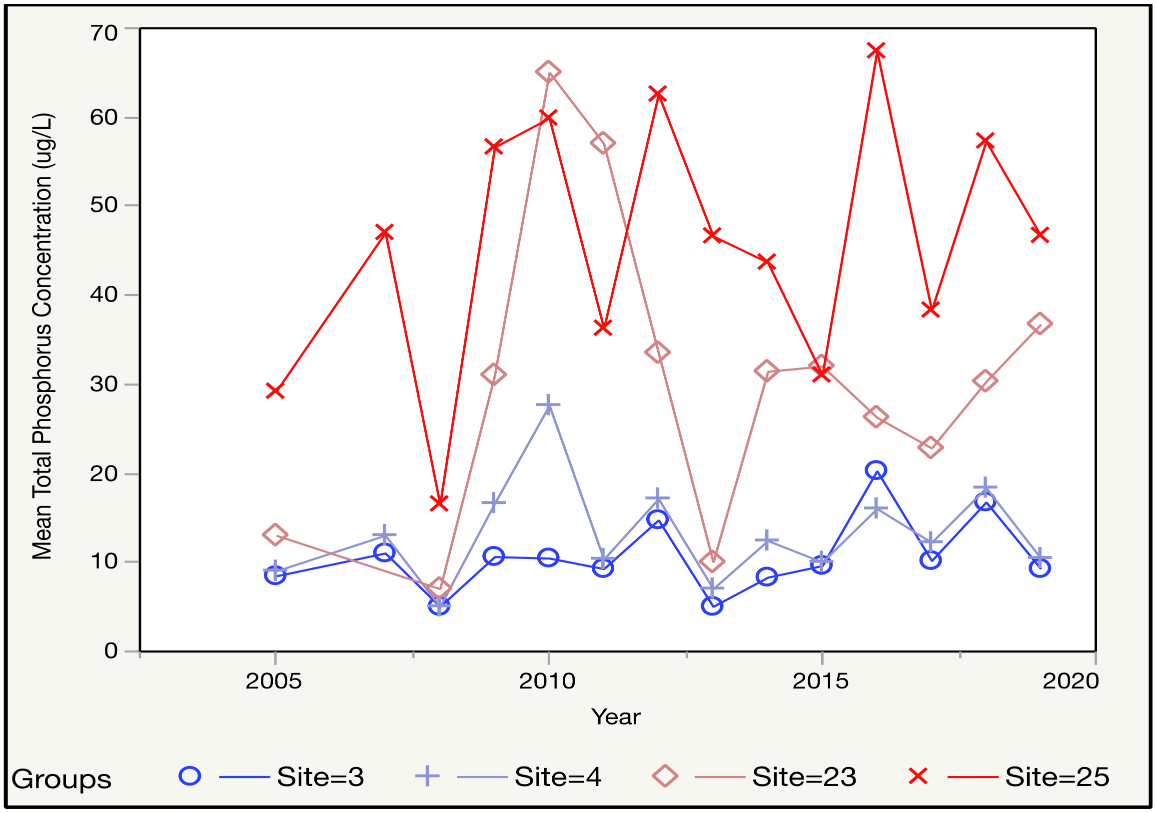
3.1.1: Spatial and Temporal Variation in Phosphorus Concentrations

Phosphorus concentrations vary substantially over time at all sites in the Lake Auburn watershed (see Maps 1 and 2 for site locations, Appendix E, SF 4 for all AWD/LWD data, and Appendix B, ST 1 for site descriptions and coordinates). Yearly means at long-term sites show high variability and few trends (Figure 1a-1c and Appendix E, SF 5a-5m). Yearly mean P concentrations within sub-watersheds generally show no increasing or decreasing trends in specific regions of the watershed. One exception may be the Johnson Road site (Site 27), which appears to have a trend of increasing P concentrations (Map 2 and Appendix E, SF 5k). All sites in the Basin sub-watershed have yearly means ranging from 8-20 ug/L (Figure 1a). With the exception of Roys, which is variable but always over 30 ug/L, all sites in the Townsend Brook sub-watershed have yearly means ranging from 10-25 ug/L (Figure 1b). Streams which are not part of the two major sub-watersheds varied substantially, from 5-70 ug/L, with Sites 23 and 25 having the high but variable concentrations, and Sites 3 and 4 having consistently low concentrations (Figure 1c). Across all years, sites with the smallest long-term concentration ranges, including the outlet (Site 1), and Mud Pond (Site 18), vary by about 25 ug/L, and sites with the largest long-term ranges, including the Townsend Brook outlet (Site 2), the First Brook outlet (Site 25), and the Basin outlet (Site 13), vary by as much as 400 ug/L (Map 5).

Phosphorus concentrations also varied substantially within years. Mean monthly P concentrations at most sites were highest in June and July, and lowest in April, May, September, and October (Map 6). Increases in mean monthly concentrations across months were generally greatest between April and June (Map 5 & 6). Variability was greatest at many of the smallest


Mean Total Phosphorus Concentration (ug/L)
5
10
15
20
2005
2010
2015
2020
Year
Groups
Site=13
Site=16
Site=18
Site=27

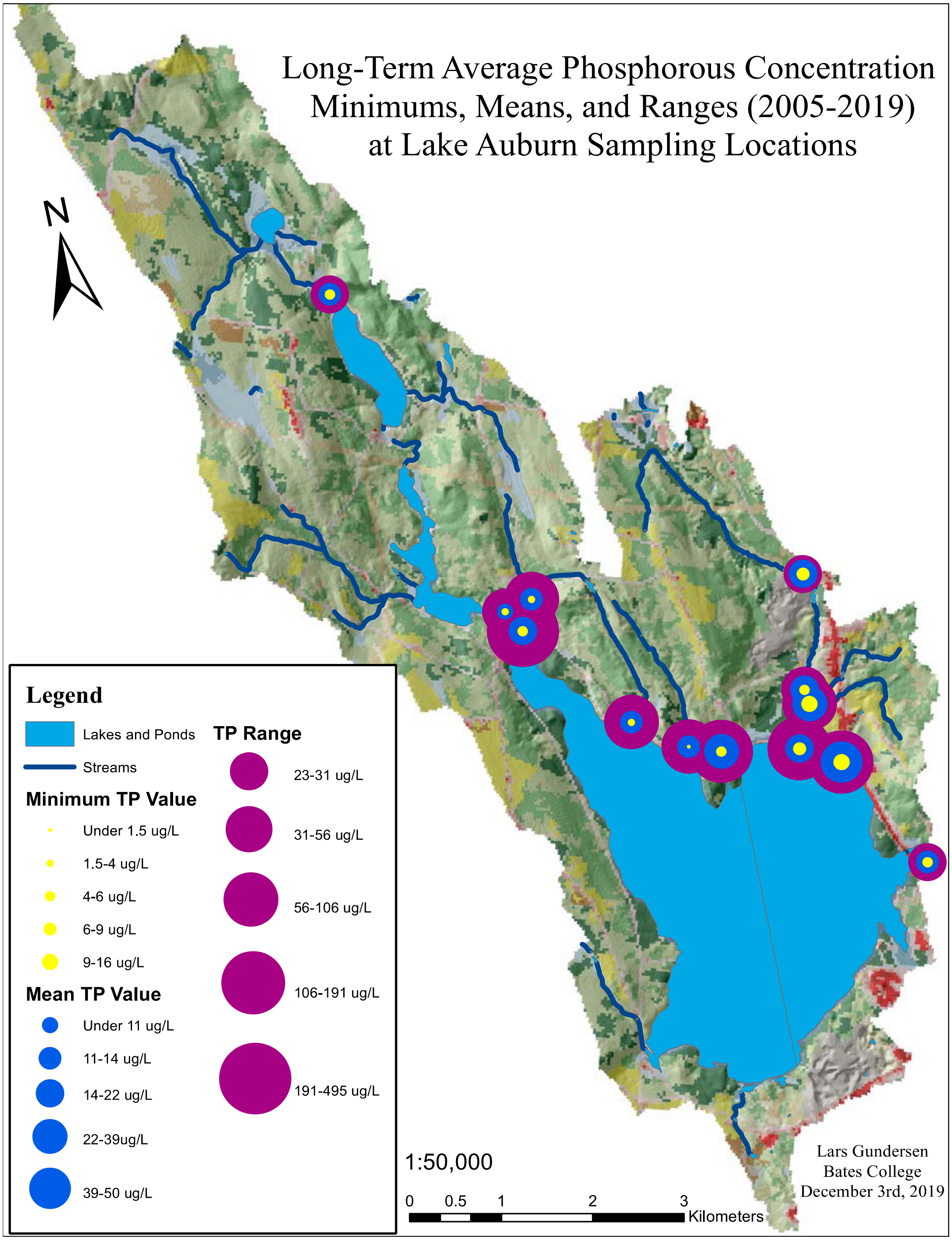

Mean Total Phosphorus Concentration (ug/L)
10
15
20
25
30
35
40
45
50
55
2005
2010
2015
2020
Year
Groups
Site=2
Site=26
Site=ROYS
Site=TBR



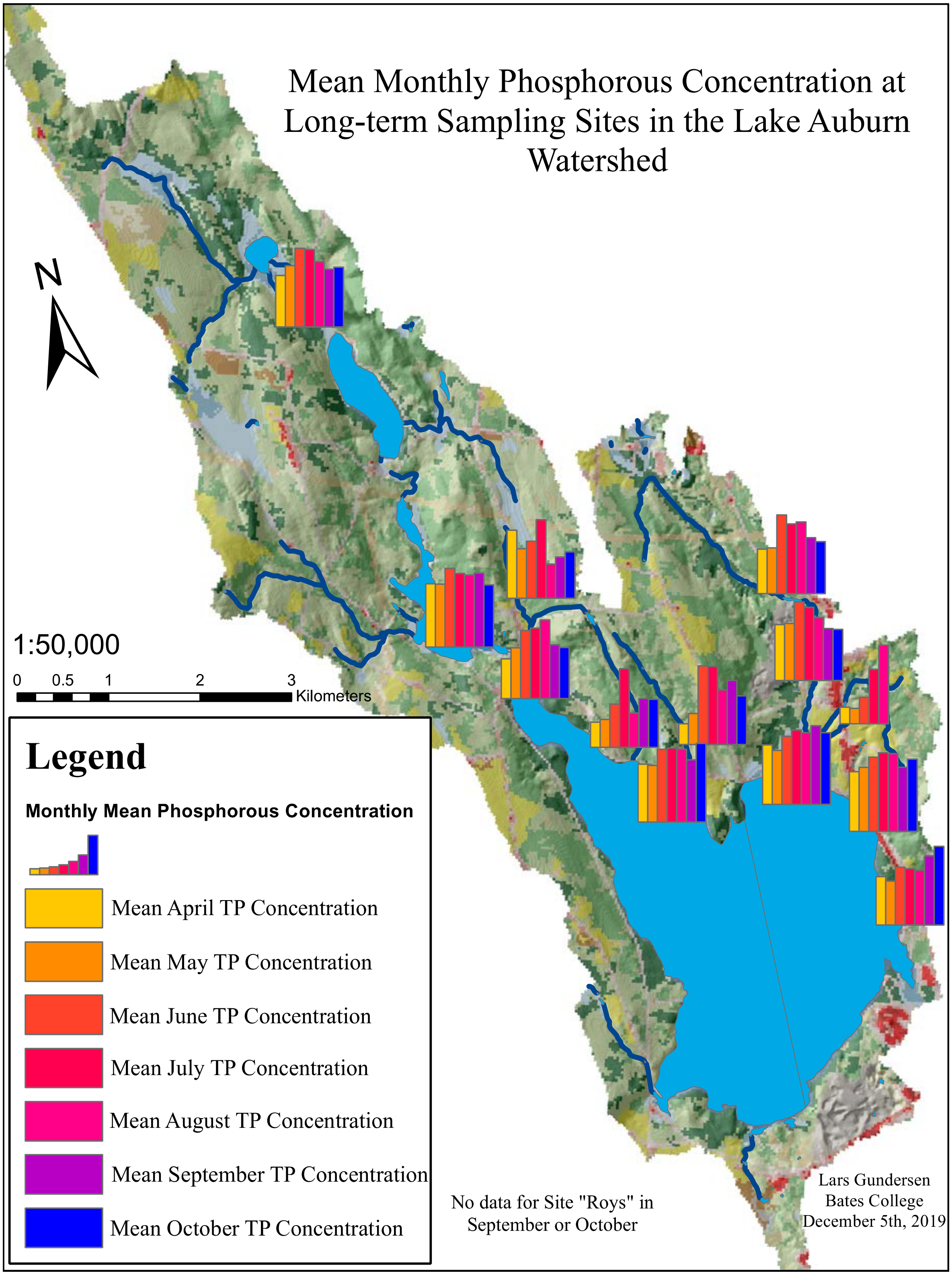
Figures 1a-1c: Mean total P concentration by site and year in the Basin drainage system (1a, top), the Townsend Brook drainage system (1b, middle), and non-Townsend or Basin drainage systems (1c, above). All sites are located in the watershed (not in the lake). See Maps 2 and 3, and Appendix B, ST 1 for site locations and descriptions.

inlets (Site 4, and Roys), and lowest at sites with consistently high P concentrations like First Brook (Site 25), and high-flow inlets like the Basin (Site 13) and Townsend Brook (Site 2) (Map 6). The outlet (Site 1) was an exception to the pattern of mid-summer peaks in P concentrations; here, P concentrations were lowest in April and May, then increased every month through October, the last month in which samples are taken (Map 6). The greatest springtime (April-June) increase was in the upper part of Townsend Brook (Site TBR), at the Horse Pond site on the north shore of the lake (Site 23), and at another north shore inlet (Site 4) (Appendix F, SM 1).

At all sites, variability was high, and trends were scarce (Figures 1a-1c). Indeed, variability within years was often nearly as substantial as variability across years. The most variable site



Map 5: Long-term average P concentration minimums, means, and ranges (2005-2019). Minimum is lowest value across all years; mean is mean of each yearly mean; range is range across all years.



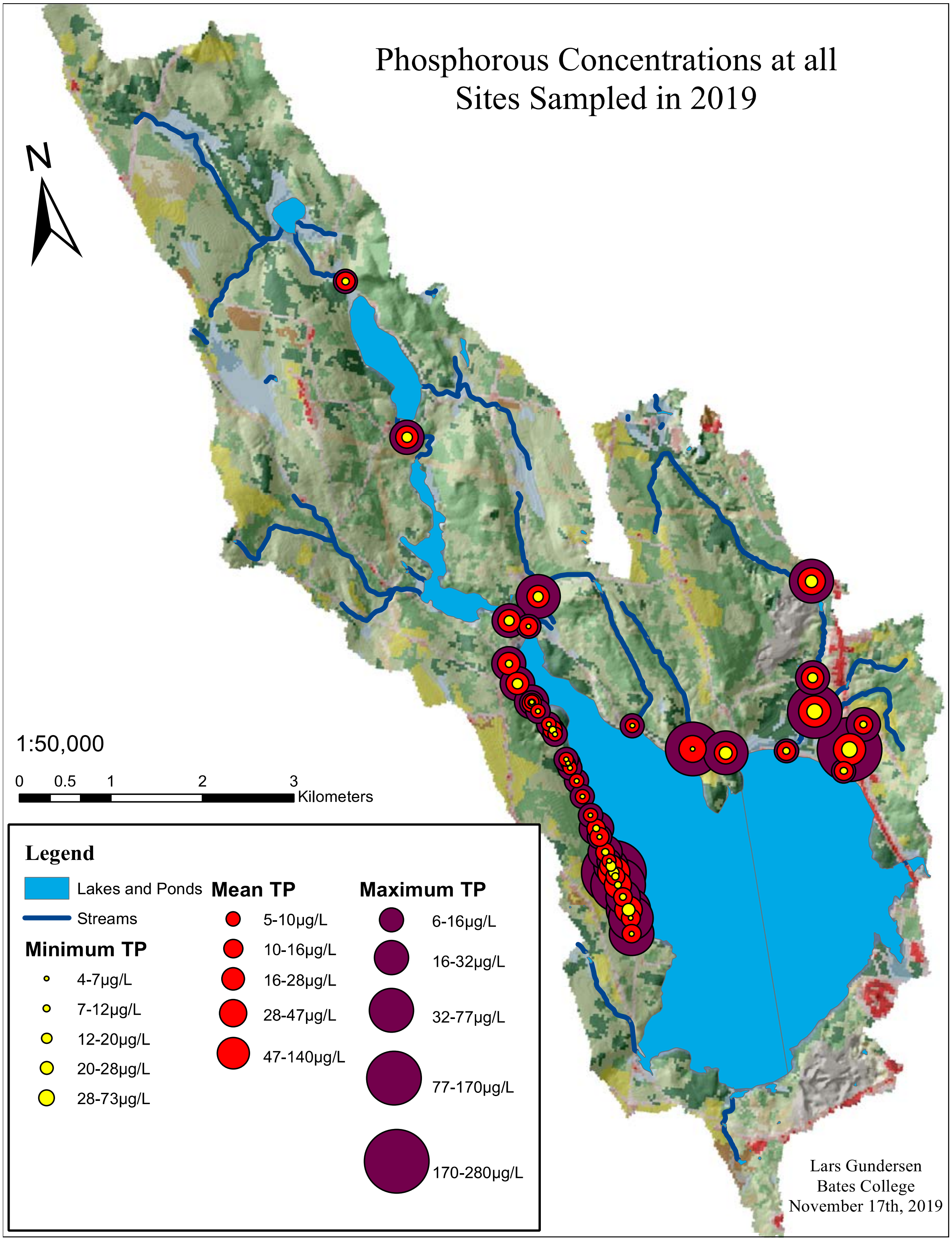
Map 6: Mean monthly P concentrations at long-term sites during the sampling season. Bars show relative comparisons within sites, meaning that each bar graph is on a different scale.

was the First Brook outlet site (Site 2). Here, the total range across all sampling years was 191 ug/L, while the range for individual years was often nearly 180 ug/L. The least variable site was in the upper part of Townsend Brook (Site TBR). Here, the total range across all sampling years was 23 ug/L, while the ranges for individual years ranged from 5-22 ug/L. Eight of the 13 long-term sampling locations varied by more than 50 ug/L across all years in which sampling occurred, with four of these sites varying by more than 100 ug/L. Just five sites varied by less than 50 ug/L, with only one varying by under 25 ug/L. Most of this variability is also reflected in yearly ranges.

Variation in P concentrations across sampling sites is substantial, and there are different dynamics in each inlet. Differences between the minimum, mean, and maximum value recorded at each site in 2019 (including sites where sampling began in 2018) demonstrate these variations in means and ranges across sites (Map 7). Some inlets, like Site 13 and Site 2, have relatively low concentrations. Some inlets, like Site 3, have very large variation between the minimum and maximum values. And other sites, like Site 25, have consistently high values (Map 7). Map 7 also provides perspective on the western shore of the lake, which was not sampled at all before 2018 and was thus excluded from the long-term data analysis. Phosphorus concentration values at several sites on the western shore are nearly as high as values at First Brook (Site 25), the long-term sampling location with the highest concentrations.

3.1.2: Major Stream Phosphorus Load Estimation

Phosphorus load for 2019 at sites with both concentration and discharge data demonstrates the impact of discharge on load. Inlets with relatively low concentrations but high flow, like the Basin inlet (Site 13) and Townsend Brook inlet (Site 2), contribute the largest loads, while several of the outlets with the highest concentrations but low flow, like the Taber's Driving



Map 7: Phosphorus concentrations at all sites sampled in 2019. Minimum is the minimum value recorded at each site; mean is the average of all P concentration data recorded at each site; and maximum is the highest value recorded at each site

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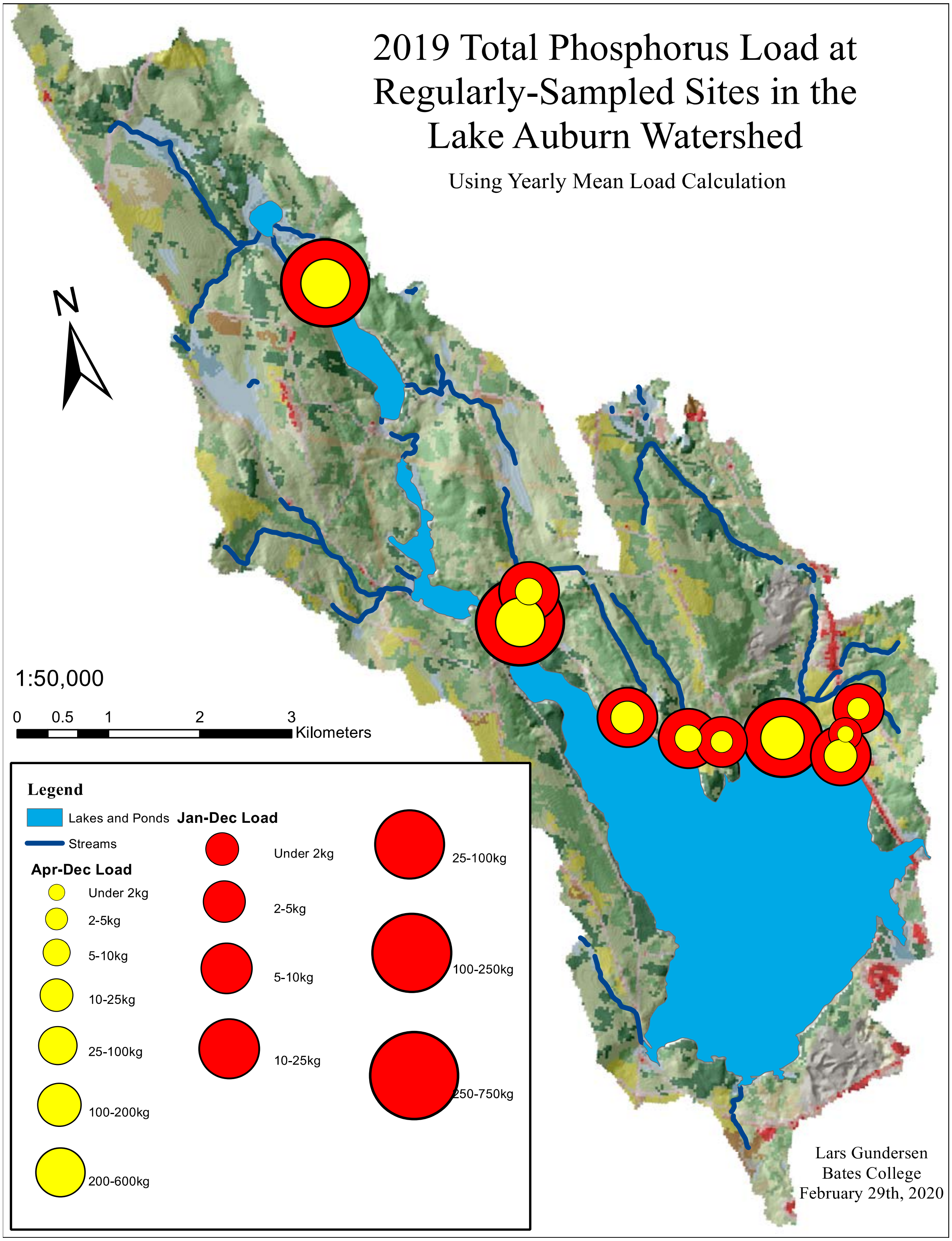
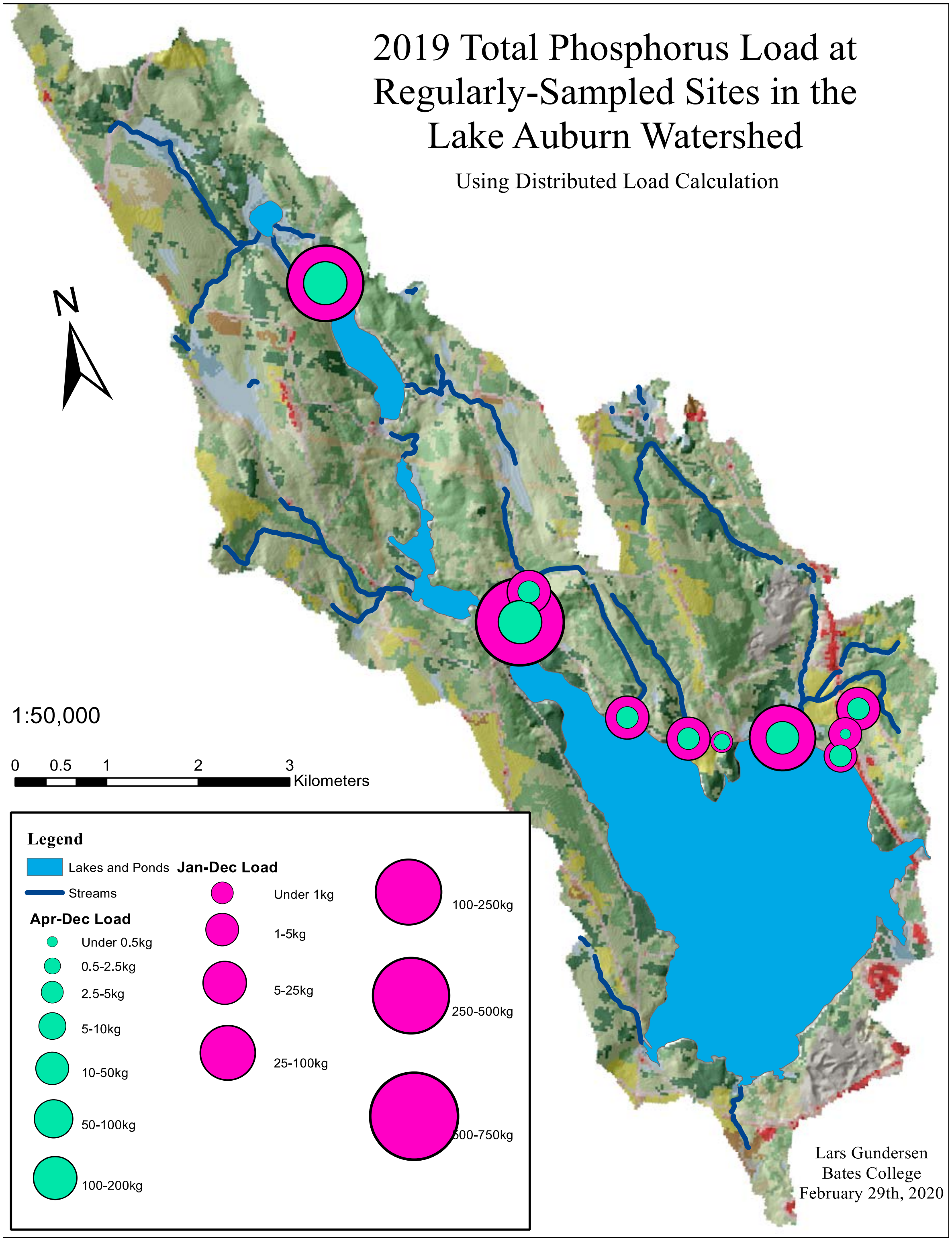


Map 8a: Estimated stream P load at long-term sampling locations, using yearly mean load estimate. See data manipulations for Map 8a in Appendix C for methods information.



Map 8b: Estimated stream P load at long-term sampling locations, using distributed load estimate. See data manipulations for Map 8b in Appendix C for methods information.

Range site (Site 3) and First Brook inlet (Site 25), contribute relatively smaller loads (Map 7 and 8a-8b). Across load calculation methods, the proportion of the total load originating from each stream remains fairly consistent. Based on AWD/LWD data, the Basin inlet is consistently responsible for over three-quarters (76-79%) of the measured stream load (some minor inlets are not measured), with Townsend Brook contributing about one-fifth (18-20%) of the measured load, and the remaining streams collectively contributing less than 5% of the measured load (Figure 2; Appendix B, ST 10). Sampling locations which are upstream of another sampling location were excluded from the final calculation of load to prevent double counts.

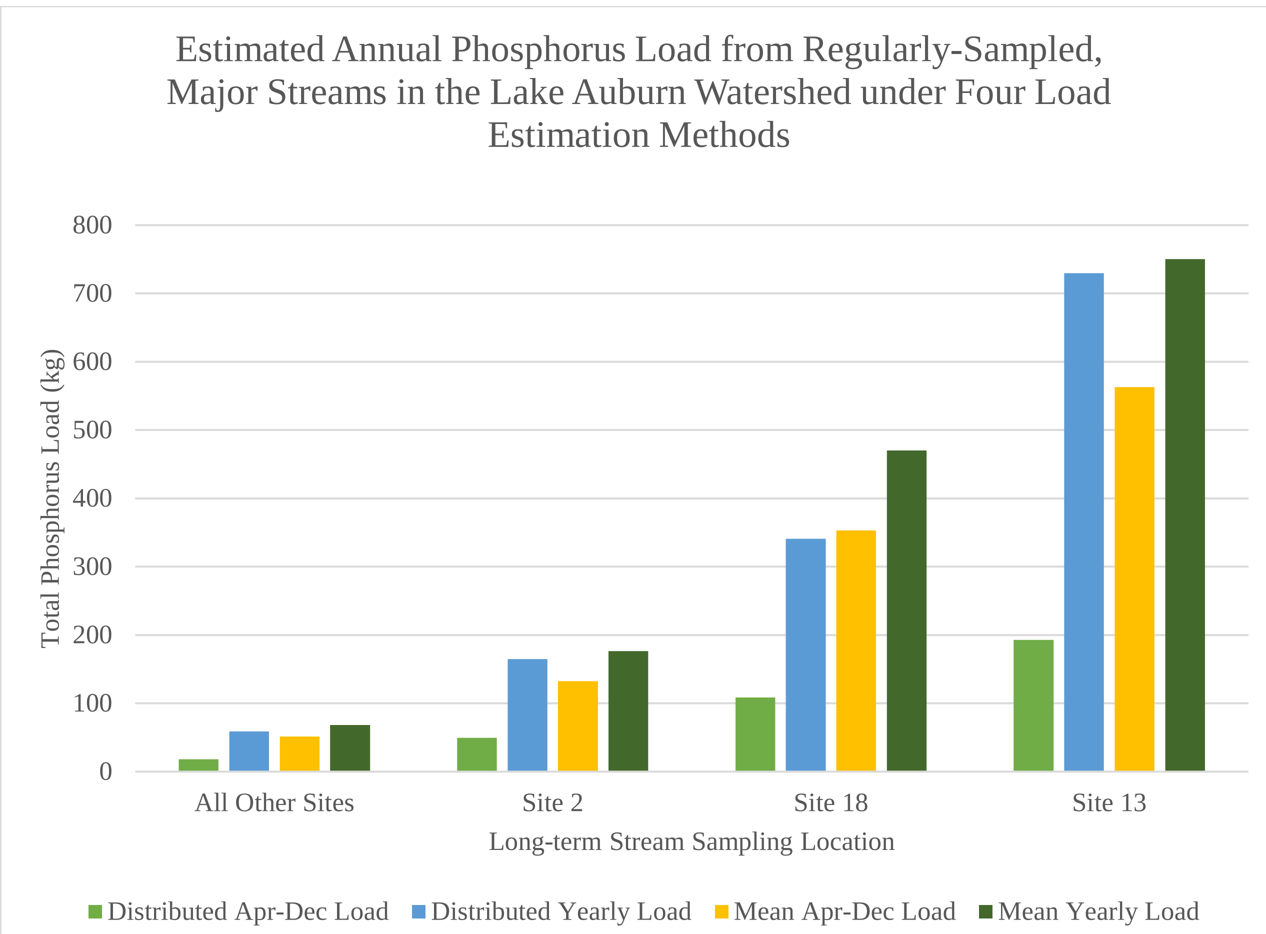


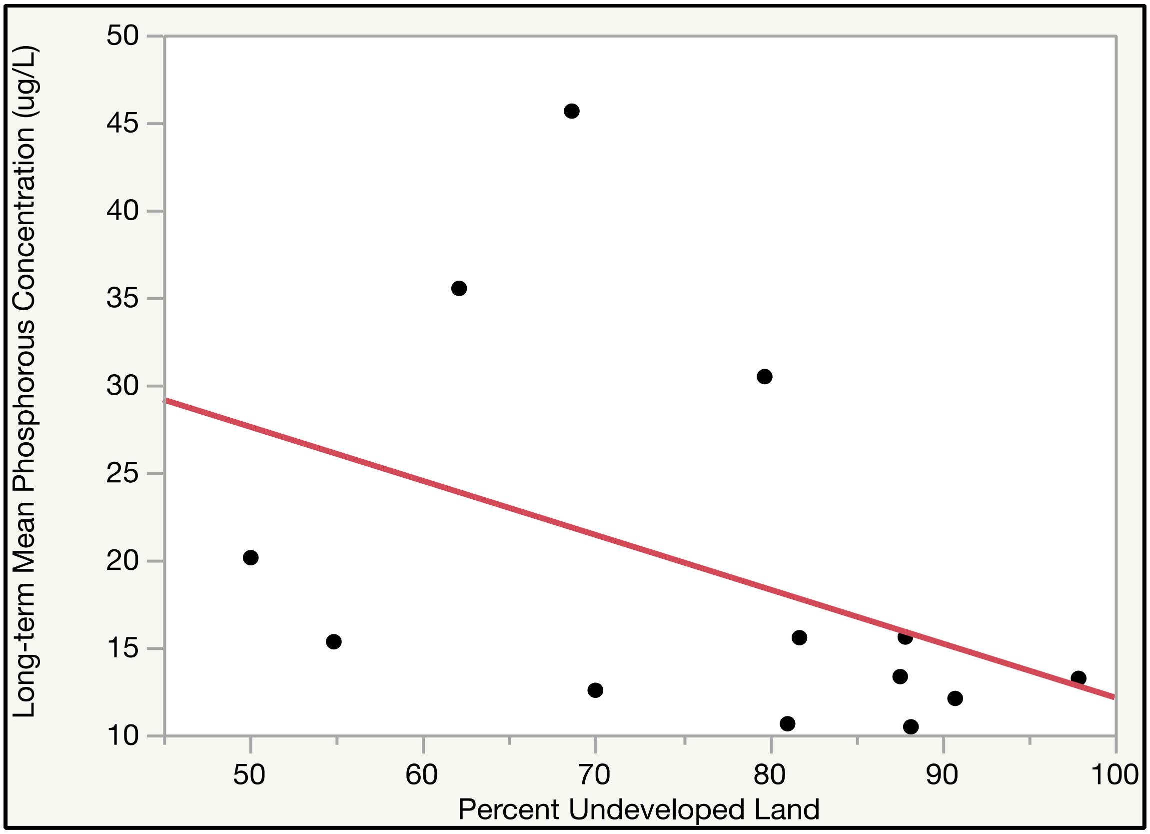
Figure 2: Estimated annual P load from regularly sampled major streams in the Lake Auburn watershed under four load estimation methods. See Maps 2 and 3, and Appendix B, ST 1 for site locations and descriptions.

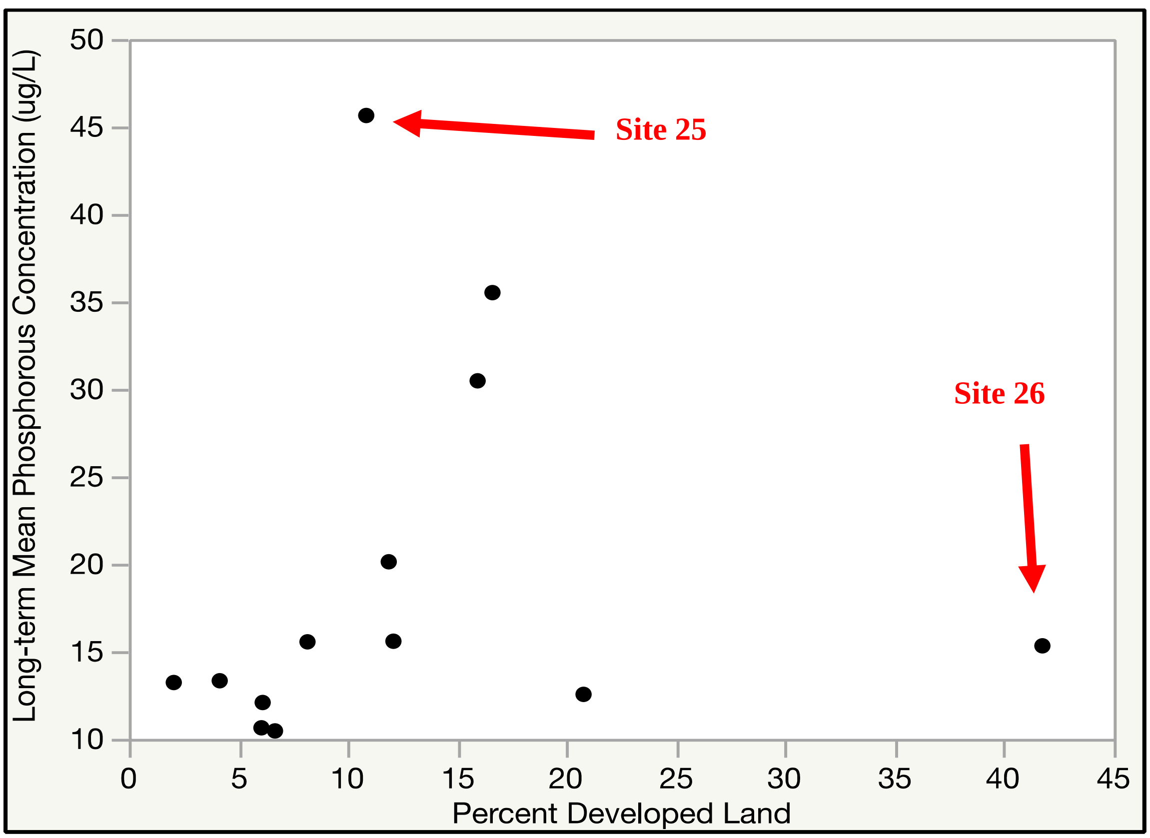
3.1.3: Concentration and Discharge Analysis

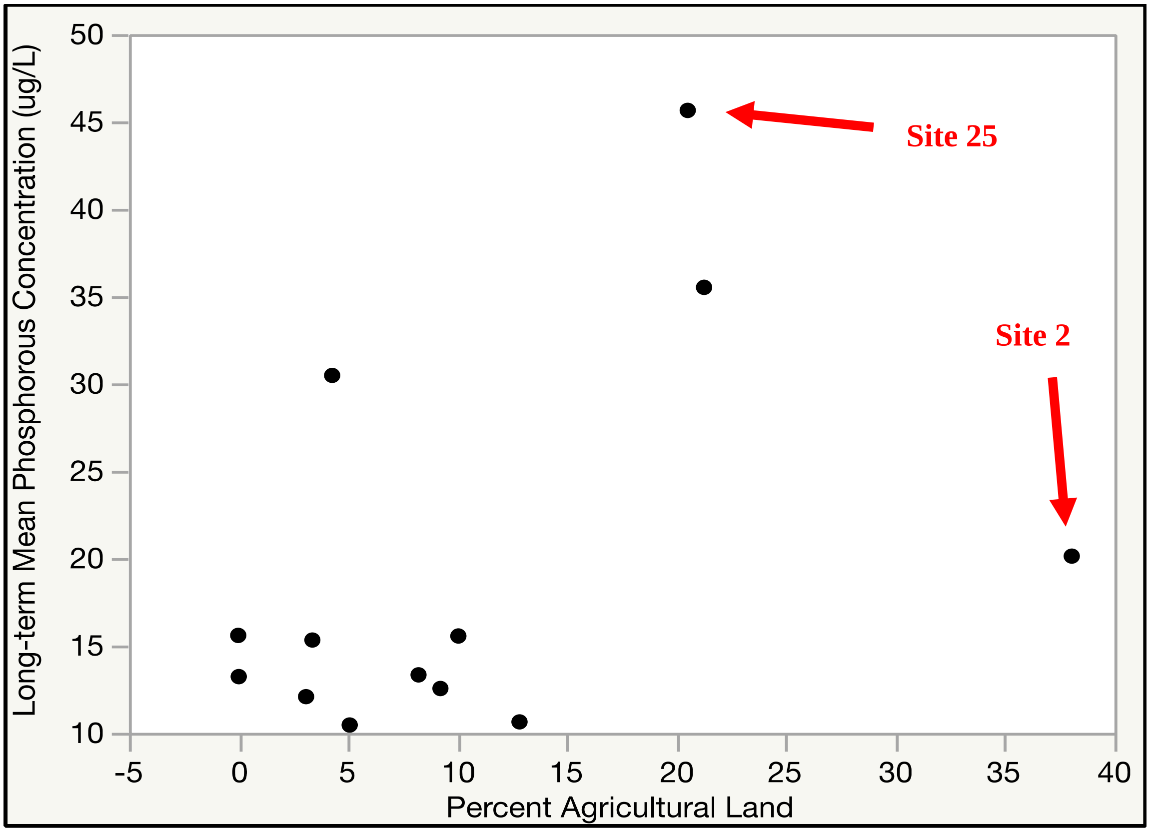
The concentration and discharge analysis gave differing results for the two sub-watersheds examined. At the Basin outlet (Site 13), neither low nor high discharge necessarily leads to high P concentrations, and there was no relationship between P concentrations and discharge (Appendix E, SF 6a and 6b). At the Townsend Brook outlet, the highest P concentrations consistently occurred when discharge was highest (Appendix E, SF 6c). Furthermore, when four P concentration outlier values, which AWD/LWD staff suspected were recorded in error, were excluded (see Appendix C.1), a statistically significant relationship emerged between P concentration and discharge (Appendix E, SF 6d). Nonetheless, it is important to note that even in this case, discharge only explains 37% of the variation, suggesting that one or several other factors besides discharge are more important determinants of P concentrations and that discharge is generally a poor predictor of P concentrations in the watershed.

3.1.4: Sub-Watershed Land Cover Analysis

In the sub-watersheds where data have been collected over the longest period, increasing percentages of developed and agricultural land are generally associated with higher P concentrations. There is a statistically significant negative relationship between undeveloped land and P concentrations in the watershed (Figure 3a). However, the R^2 value is 0.33, meaning that two-thirds of the variation in P concentrations is not explained by land cover. Similarly, developed and agricultural land is generally associated with higher P concentrations, though these relationships are not statistically significant (Figure 3b and 3c). The First Brook outlet (Site 25), which has the highest P concentrations, does not have the highest percentage of developed or agricultural land, while the mid-Townsend


Long-term Mean Phosphorous Concentration (ug/L)
50
45
40
35
30
25
20
15
10
50
60
70
80
90
100
Percent Undeveloped Land


Long-term Mean Phosphorous Concentration (ug/L)
50
45
40
35
30
25
20
15
10
0
5
10
15
20
25
30
35
40
45
Percent Developed Land
Site 25
Site 26



Figures 3a-3c: The relationship between P concentration and undeveloped land (3a, top), developed land (3b, middle), and agricultural land (3c, above) in the Lake Auburn watershed. R^2 values were 0.33, 0.14, and 0.23, respectively, and p-values were 0.032, 0.19. and 0.08, respectively. See Maps 2 and 3, and Appendix B, ST 1 for site locations and watershed orientation

Brook site (Site 26) has the highest percentage of developed land, but relatively low P concentrations. The Townsend Brook outlet (Site 2), meanwhile, has the highest percentage of agricultural land but relatively low P concentrations (Figure 3b and 3c). Thus, land cover alone is a poor predictor of P concentrations. Thus, in the absence of a plausible method of predicting watershed-scale P loading using AWD/LWD data, the SWAT model was used to predict P loading, with the goal of understanding watershed-wide dynamics and identifying P loading hotspots which may not be captured using AWD/LWD data.

3.2: Results of the SWAT Watershed Analysis

3.2.1: SWAT Phosphorus Loading Predictions under the Six Scenarios

Phosphorus load estimates from the six SWAT scenarios provide a macro-level means of comparison for past, present, and future P loading predictions in the watershed. The differences in loading estimates across these scenarios reveal how P loading has changed over the past 40 years, and how loading could increase over the next 40 years based on land management decisions and climate change. The total P load under the default scenario using 2019 data is 1671 kg per year (Figure 4). This represents a 198% increase from the 560 kg load estimated under the scenario using 1980 land use and weather data, which is used as the baseline for the purposes of the load analysis. Nineteen-eighty was chosen as the baseline because earlier land use data were not available. Indeed, the values from 1980 almost certainly represent an already-elevated estimate. However, 1980 data offers the earliest available point of

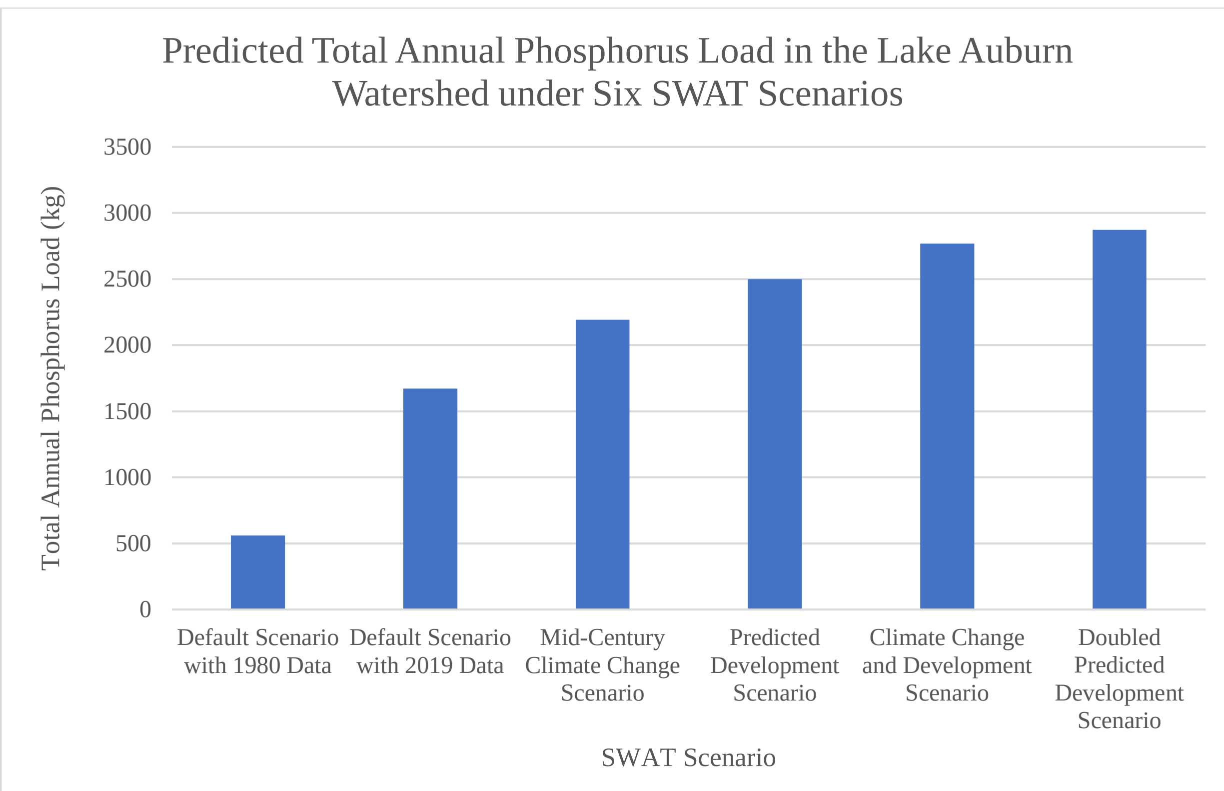


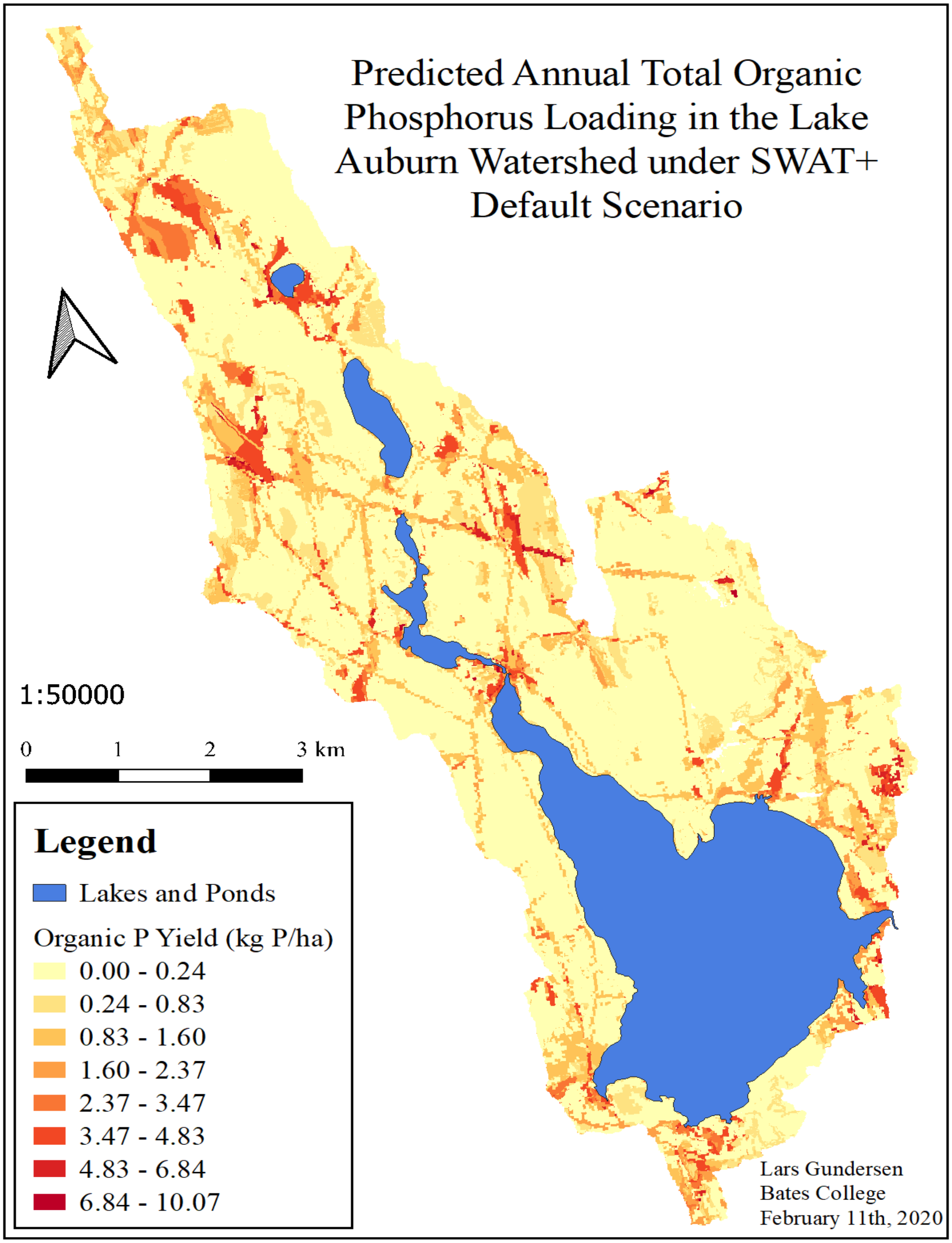
Figure 4: Predicted total P load in the watershed under the six SWAT scenarios.

reference, which makes it useful for comparison and for understanding the scope of the P loading increases which have taken place over the past 40 years.

All four future scenarios predict that this trend of increasing P loads will continue (Figure 4). Under the mid-century climate change scenario, the annual P load increases to 2198 kg by 2050, a 31% increase from current loading and a 291% increase from the 1980 baseline. If projected development in the watershed continues as planned, the annual P load is predicted to increase to 2500 kg by 2050, a 50% increase from current loading and a 346% increase from the 1980 baseline. If development occurs twice as rapidly as the 2010 Auburn development projections, annual P load is predicted to increase to 2872 kg by 2050, a 72% increase from current loading and a 413% increase from the 1980 baseline. Finally, under the most-likely scenario in which both climate change and projected development occur, the annual P load is predicted to increase to 2768 kg by 2050, a 66% increase from current loading and a 394% increase from the 1980 baseline (Figure 4).

3.2.2: Spatial Variability in SWAT Nutrient Loading Predictions

In addition to an overall loading estimate, SWAT predicts detailed, HRU-level loading estimates for the entire watershed. Under the default scenario, the model reveals substantial differences in loading across the watershed (Map 9a). Though all watershed land is contributing P to the total load, there appear to be loading hotspots near the North Auburn Dam, along the Route 4 corridor, in the southwest corner of the lake, around Skillings Corner, along Holbrook Road west of Little Wilson Pond, in East Hebron along the Turner town line, and along the lower portion of the Townsend Brook gulley (Map 9a; see Maps 1-3 for watershed orientation; see Appendix G, SM 2a-2d for higher-resolution maps of the areas around Mud Pond, Little Wilson Pond, the Basin, and Townsend Brook).

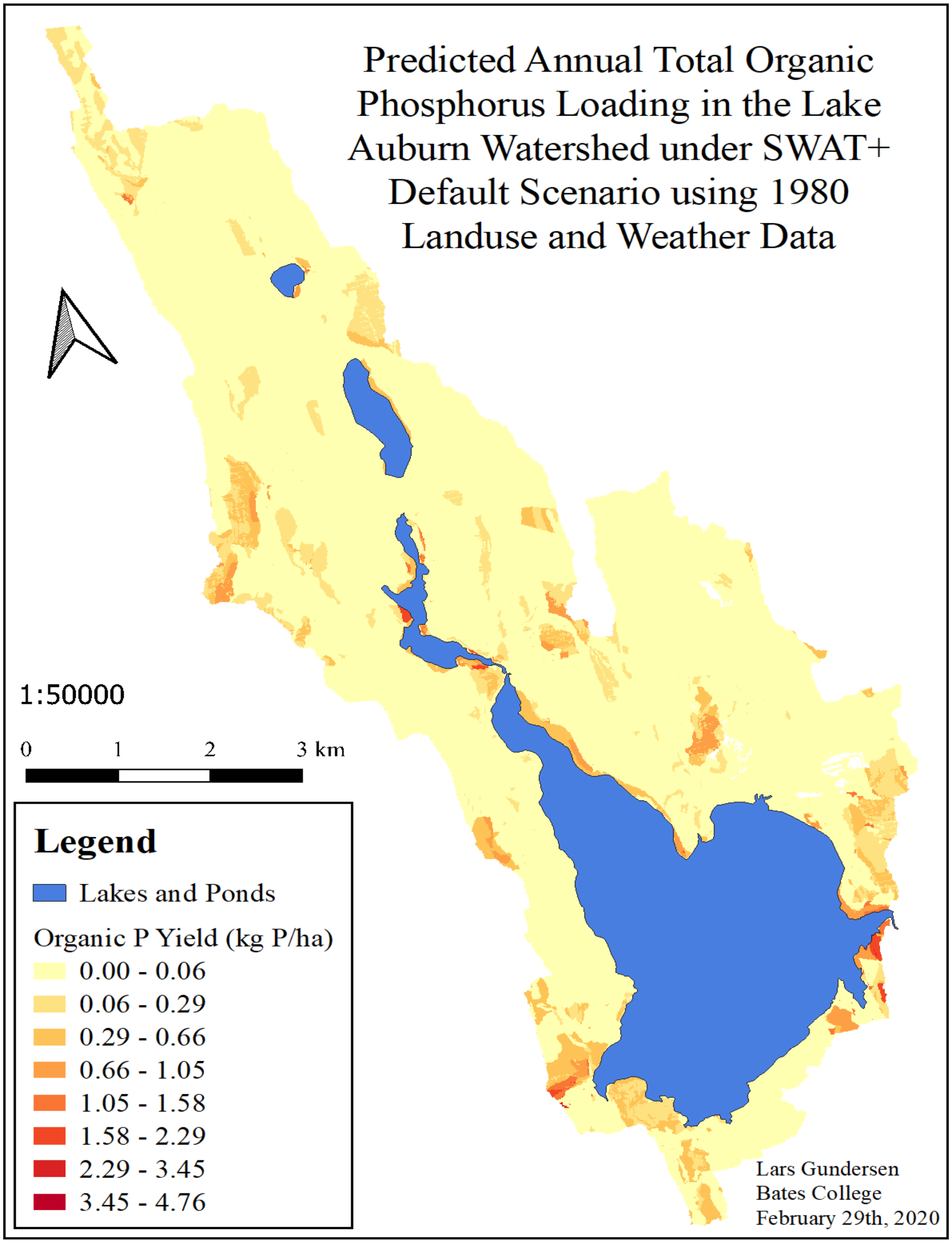


Map 9a: Predicted annual total organic P loading in the Lake Auburn watershed under SWAT+ default scenario.

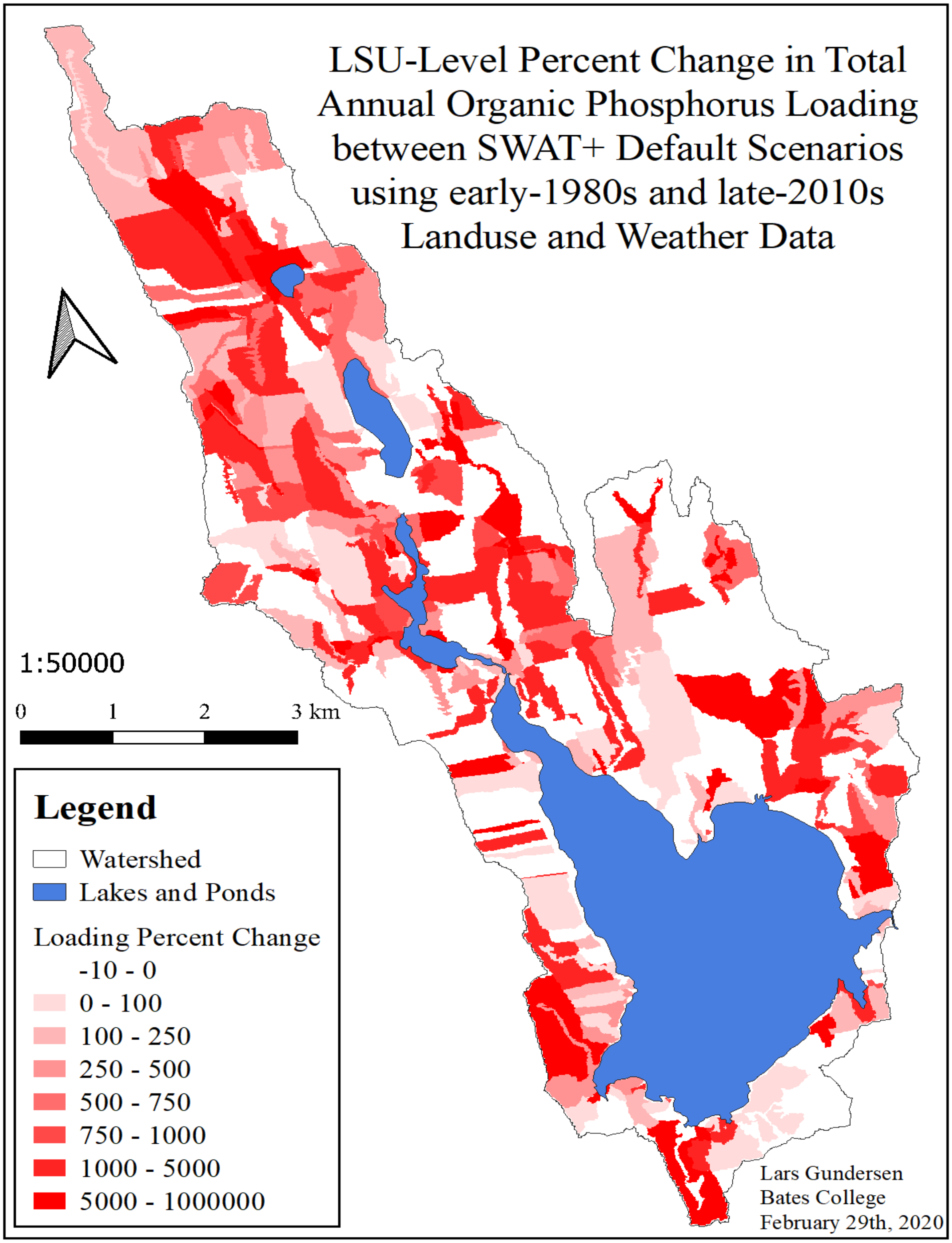
Loading has increased dramatically since 1980 according to the model (Map 9a, 9b, and 10a). Though hints of some current loading hotspots were visible in 1980, loading has increased by at least 100% in most of the watershed over the past four decades (Map 10a). HRU-level percent change in P loading could not be calculated for the 1980 scenario due to changes in land use (and thus, changes in HRU shape and location), but an LSU-level comparison reveals that loading has increased almost everywhere, but especially along the Route 4 corridor, in the southwest corner of the lake, around the Basin, Little Wilson Pond, and Mud Pond, and around Skillings Corner. Two exceptions to this increase are the areas along Lake Shore Drive on the northwest shore of the lake, and along the immediate shoreline of the Basin (Map 10a). In both of these places, loading appears to have decreased since 1980 (Map 9a and 9b). Loading has also remained stable in much of the upper Townsend Brook sub-watershed. (Map 10a).

3.2.3: The Effects of Climate and Development on SWAT Predictions

If watershed development continues as planned for the next 40 years, the model predicts that P loading will also increase substantially (Map 9a, and Appendix G, SM 3a and SM 4a). Much of the increase in load appears to originate in and around locations which are already contributing an outsized load, such as around Mud Pond, around the North Auburn Dam, along the Route 4 corridor, around Skillings Corner, and near Taber's Driving Range on the north shore of the lake (Appendix G, SM 3a and SM 4a). There are also large swaths of land with little or no increase in loading, mostly in areas which are far away from roads and other development. It is also important to reiterate that in development scenarios, SWAT randomly assigns developed land based on where it considers development to be "most likely." Thus, though these maps present a well-calibrated estimate of overall load, future loading will not necessarily occur in the locations predicted by the model. In the scenario which accounts for



Map 9b: Predicted annual total organic P loading in the Lake Auburn watershed under SWAT+ default scenario using 1980 land use and weather data.



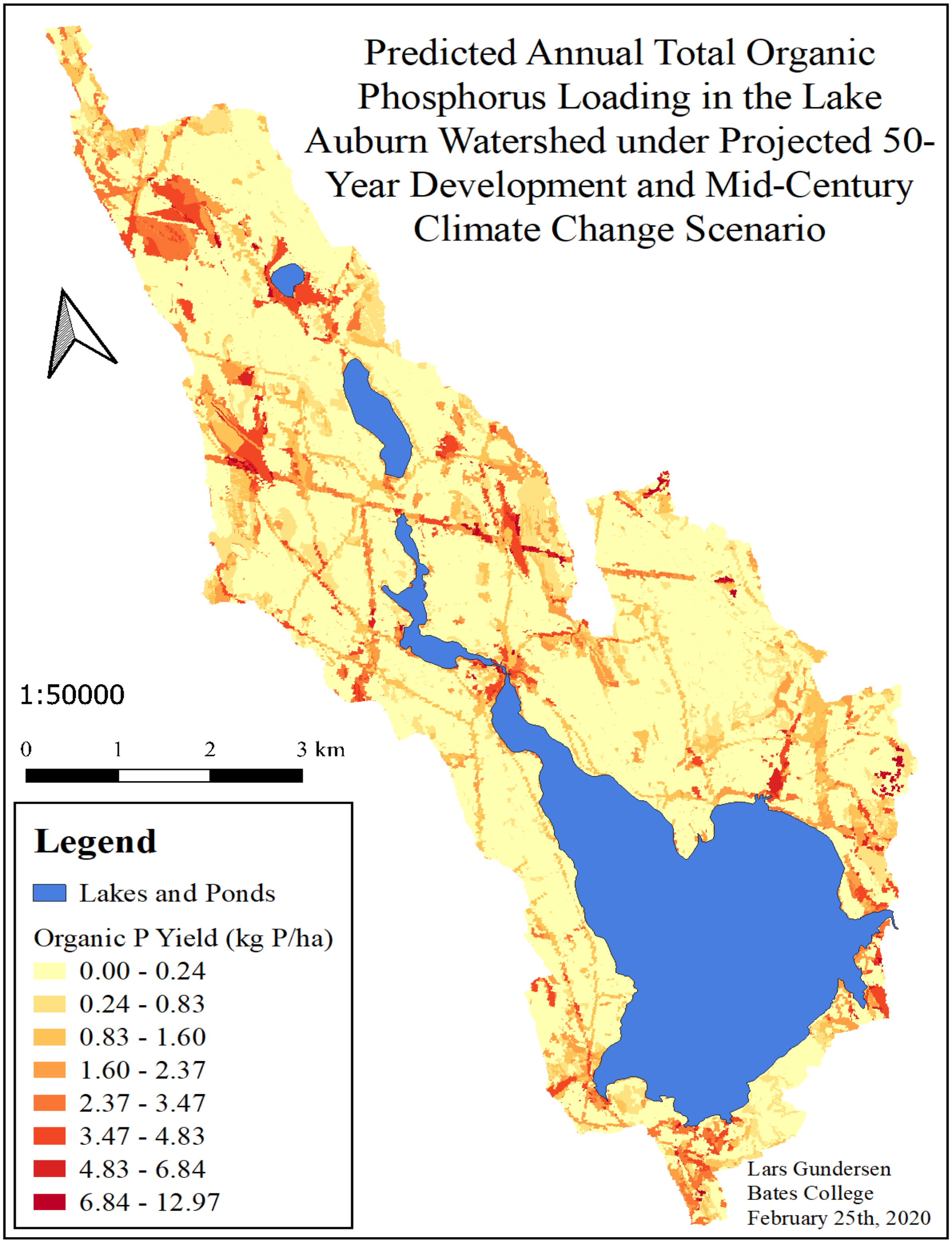
Map 10a: LSU-level percent change in total annual organic P loading between SWAT+ default scenarios using early-1980s and late-2010s land use and weather data.

more development than is currently projected (Appendix G, SM 3b and SM 4b), increases in P loading largely mirror the increases predicted in the projected development scenario (Appendix G, SM 3a). The main difference is that loading around existing roadways and development is even higher.

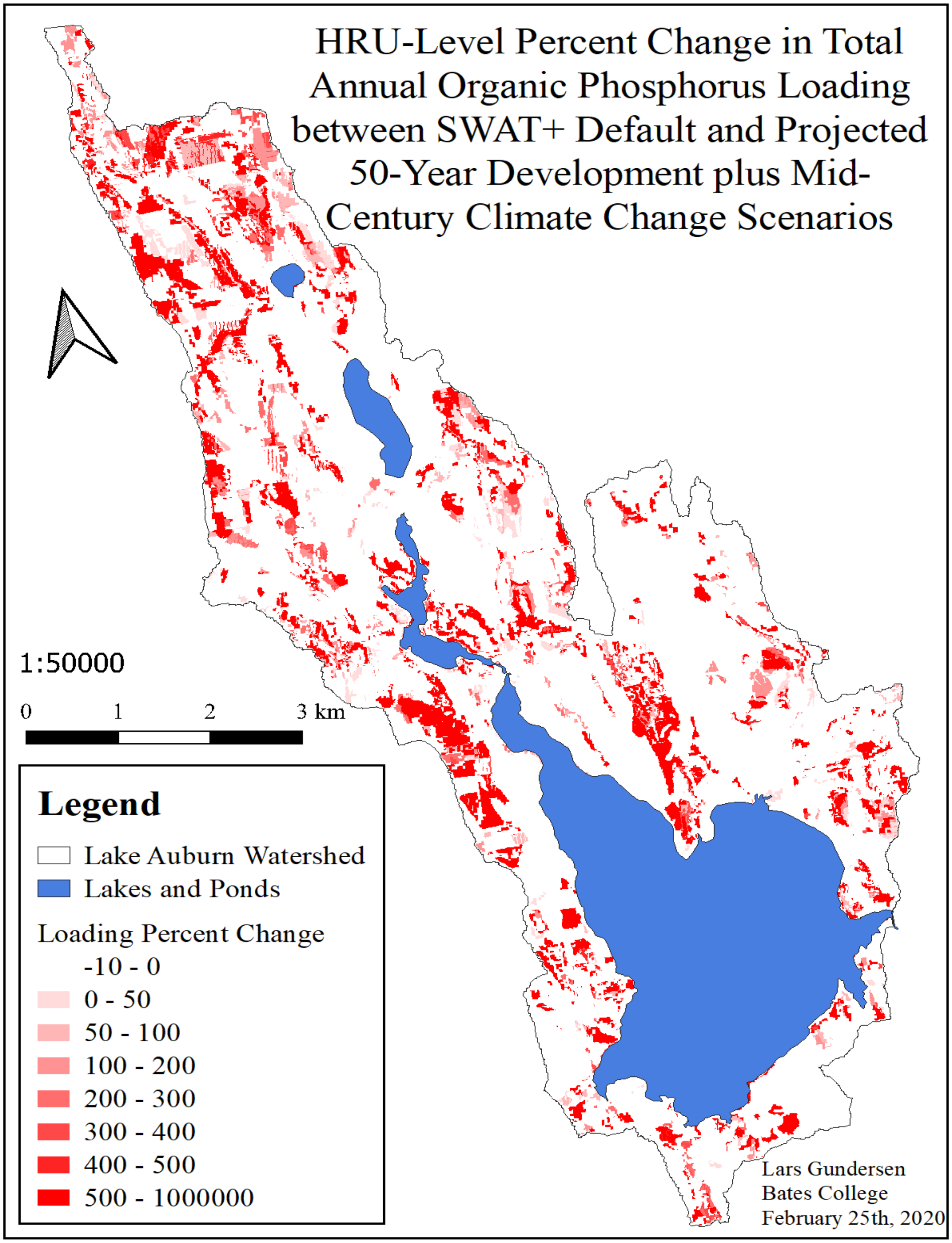
The effects of climate change on predicted mid-century P loading are similar to the development scenarios, but with a key distinction (Map 9a and Appendix G, SM 3c and SM 4c). In addition to evidence of increased loading along roadways and adjacent to existing development, the model predicts increased loading from areas with steep slopes (Appendix G, SM 4c). Thus, in addition to predicting higher loading in places like the Route 4 corridor, around Skillings Corner, near the North Auburn Dam, and along Holbrook Road, the model also predicts increased loading in places with steep slope like west of mid-Townsend Brook, north of Little Wilson Pond and east of Mud Pond, and east of Route 4 on the far eastern edge of the watershed (Appendix G, SM 4c; see Map 1 for slope map).

Under the most likely scenario in which both development and climate change take place as projected, SWAT again predicts that most additional loading will be clustered around roadways and existing development (Map 9c and 10b). Some loading was also predicted in areas with steep slopes, reflecting the projected effects of extreme precipitation events on future loading.

While most of the increase in P loading between 1980 and 2019 appears to have been driven by development, the impacts of climate change on loading increases are predicted to increase dramatically by 2050. Under the projected climate change scenario, climate change alone is predicted to increase annual loading by 527 kg, nearly equivalent to the entire 1980 load of 560 kg. Though additional scenarios would be needed to understand the relative importance of climate change and development on past and future P loading increases, a comparison of 1980



Map 9c: Predicted annual total organic P loading in the Lake Auburn watershed under the most likely projected 50- year development and mid-century climate change scenario.



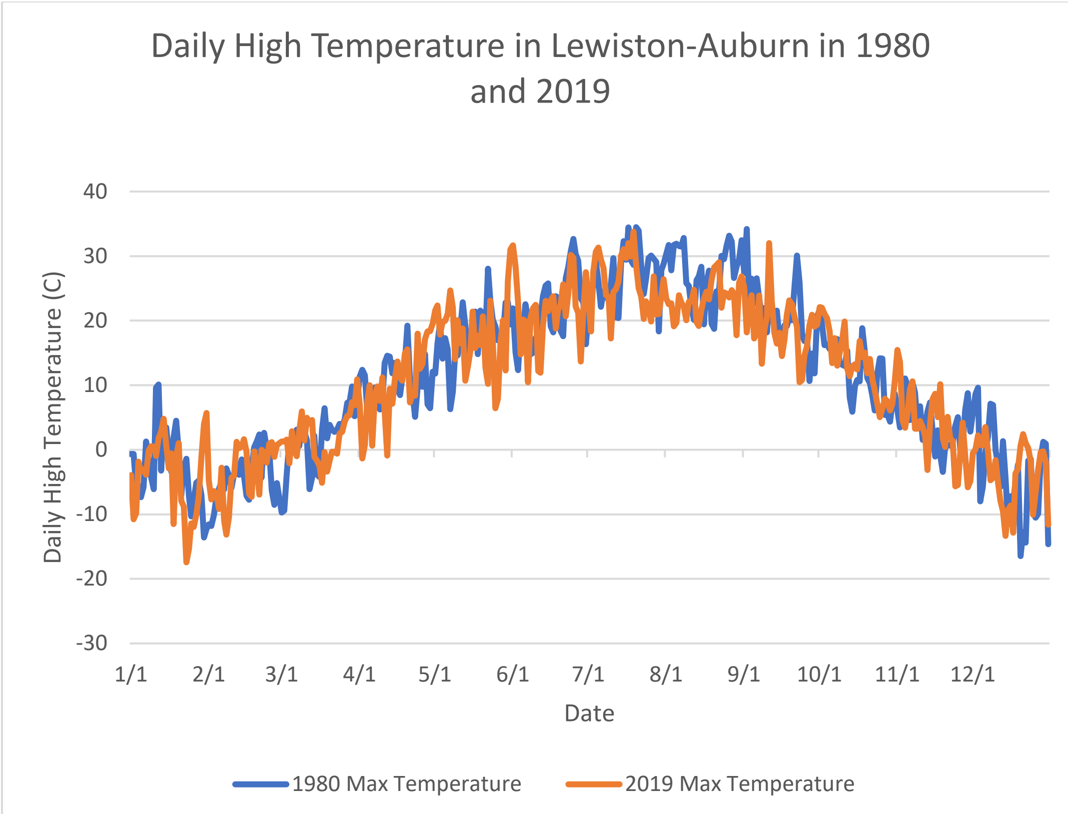
Map 10b: HRU-level percent change in total annual organic P loading between SWAT+ default and the most likely projected 50-year development plus mid-century climate change scenarios.

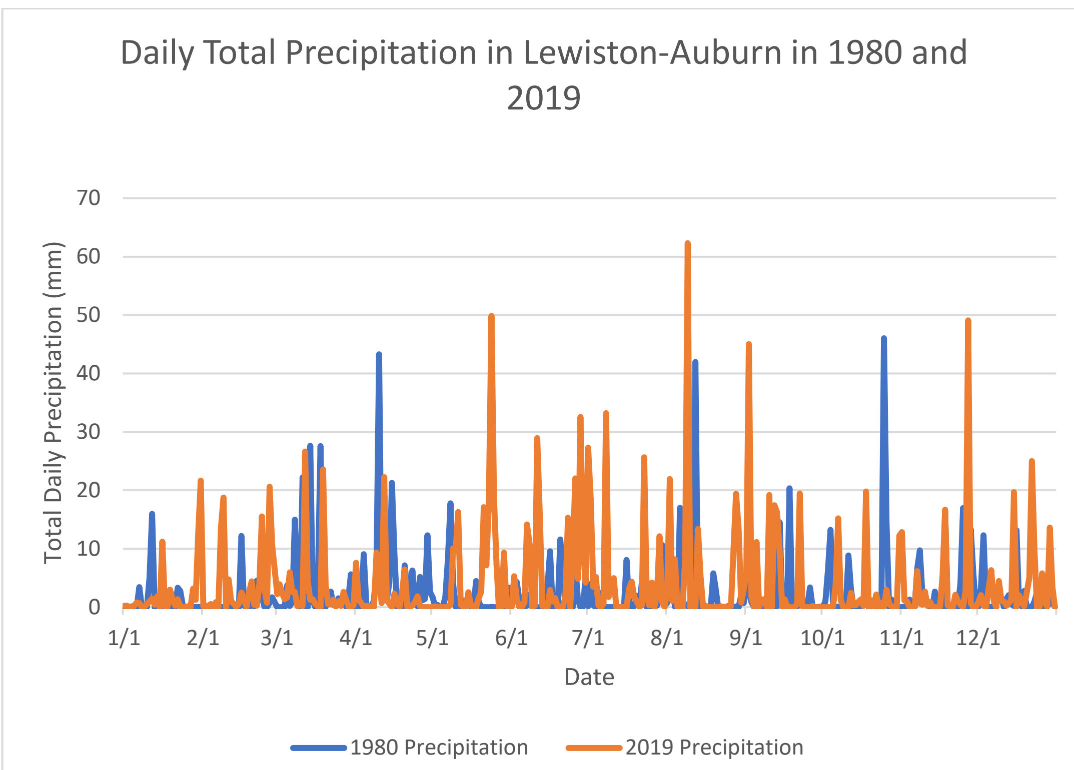
and 2019 weather data offers clues about the causes of P loading increases since 1980. Daily maximum temperature in Lewiston-Auburn has not changed dramatically between 1980 and 2019 (Figure 5a), but precipitation patterns have shifted (Figure 5b). There was more rainfall and, particularly, more heavy rainfall during the summer in 2019 compared to 1980. Of the 21 total days across both 2019 and 1980 in which daily precipitation totals were over 20 mm, 16 were in 2019 and five were in 1980 (Figure 5b). Though it is difficult to draw macro-level conclusions from a comparison of only two years, this comparison suggests that if the observed trends continue, the impact of climate change on P loading is likely to increase further.

3.2.4: Other SWAT Outputs and Considerations

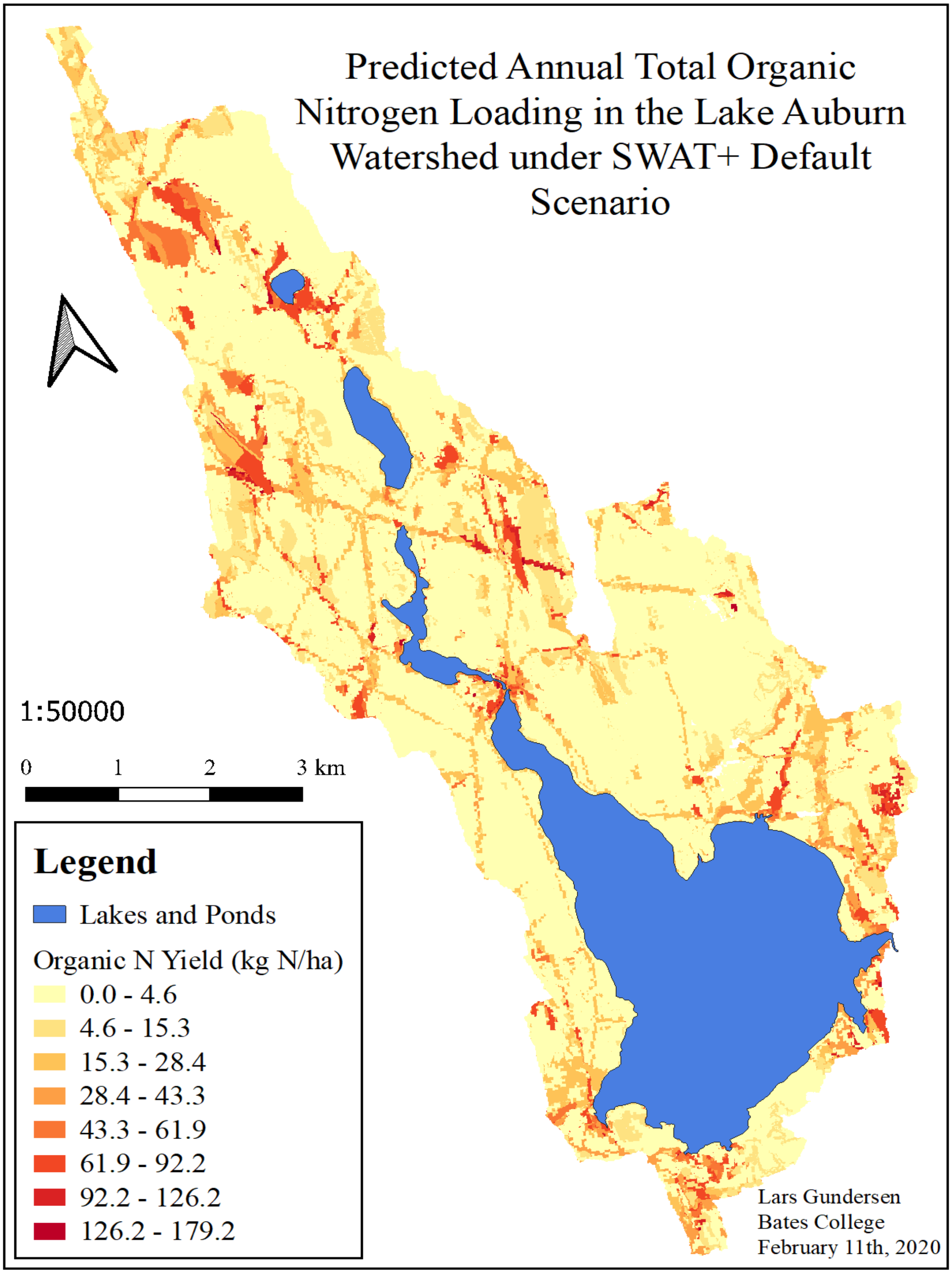
Where possible, this analysis uses HRUs instead of LSUs because they provide far greater resolution and allow for the identification of loading hotspots more accurately. However, because they divide watersheds into far fewer units, LSUs can be useful for understanding macro-level watershed dynamics. Compared to the HRU-level map for the default scenario, the LSU-level map shows similar general hotspot areas and allows for the visualization of broader hotspot regions (see Appendix G, SM 5a-5f for LSU level annual P yield maps). There is virtually no difference when calculating load using HRUs and LSUs. In this case, the values are within one percentage point of each other. SWAT also calculates lateral flow during precipitation events (see Appendix G, SM 6) and P transformations (see Appendix G, SM 7a-7d). Both are outside the scope of this project but are included for reference and out of an interest in providing the AWD/LWD with as much potentially useful information as possible.

Phosphorus loading is also not the only SWAT output. The model calculates dozens of other outputs on watershed management topics like nutrient loading, runoff, weather, and agriculture.





Figures 5a and 5b: SWAT daily high temperature (5a, top) and precipitation (5b, above) in Lewiston-Auburn in 1980 and 2019.



Map 11: Predicted annual total organic N loading in the Lake Auburn watershed under SWAT+ default scenario.

Most are beyond the scope of this project, but because it is likely that Lake Auburn is P and N (N) co-limited, N loading merits a brief exploration. Across all scenarios, SWAT predicts that locations of high N loading will very closely mirror locations of high P loading; the default P yield map (Map 9a), for example, is very similar to the default N yield map (Map 11). A similar phenomenon is visible across the five other scenarios (for all N yield maps, see Appendix G, SM 8a-SM 8e).

4: DISCUSSION

4.1: DISCUSSION OF EXISTING AWD/LWD DATA

4.1.1: Contextualizing the AWD/LWD Data

The absence of obvious, long-term trends and high intra- and interannual variability in the AWD/LWD data suggests that P concentrations have been consistently variable at most sites since 2005 (Figures 1a-1c). Indeed, the ability to see long-term trends in the data is obscured by intra-annual variability which is often nearly as substantial as interannual variability. The Johnson Road site (Site 27) is an exception because P concentrations are increasing (Appendix E, Figure 2k). While it is encouraging that P concentrations have not increased dramatically since sampling began in 2005, most sites regularly have P concentration values over 20 ug/L, which is concerning for long-term water quality. Phosphorus concentrations of 20 ug/L are generally considered to represent the upward bound of mesotrophic lakes (Dudley 2004; Carpenter 2003; Novotny & Olem 1994). Because lakes receive inputs from lower P concentration sources like groundwater and rainfall, stream concentrations of 20 ug/L will generally not result in lake concentrations of 20 ug/L. However, Lake Auburn gets about two-thirds of its water from streams (Hildreth 2008a; Hildreth 2008b; Dudley 2004), suggesting that continued increases in loading could cause Lake Auburn to become eutrophic (Carpenter 2003; Novotny & Olem 1994). Indeed, mean stream input concentrations must generally be under 10-15 ug/L for a lake to remain oligotrophic and under 20-25 ug/L for a lake to remain mesotrophic (Novotny & Olem 1994).

The substantial variation in P concentrations at most sites, both across and within years, is likely a reflection of both human and natural factors. In particular, because sampling plans and staff availability varied across years, there is variability in when sampling occurred, where and how the sample was taken, how much stream flow there was, how recently it had rained or snowed, how much it had rained or snowed, how rapidly the rain or snow had fallen, and the representativeness of the sample (e.g., whether it was taken in a rapidly-flowing area of a backwater, whether there was sediment in the water column or accidentally collected from the bottom of the stream). Showing the yearly maximum and minimum value displays this variability, and taking a yearly mean value offers a macro-level picture of stream conditions, but mean values are often skewed by particularly high or low values recorded as a result of variations in AWD/LWD sampling regimens (Map 7).[1]

4.1.2: The effects of climate and land use on Phosphorus loading

The variability of the AWD/LWD data likely also reflects the effects of weather and land-use change P loading. In addition to periodic, systematic sampling, the AWD/LWD also conducts sampling during storm events (usually times in which the forecast predicts more than one inch of rain or more than a half-inch of rain-on-snow) (personal communication, Dan Fortin (LWD), 9/17/2019). Phosphorus concentrations are nearly always higher during storm events, likely reflecting weather (heavy rain) and loading driven by land-use change (development, impervious surfaces/runoff, agriculture). At the Johnson Road site (Site 27), the only site with a notable trend of increasing concentrations, the increase could be driven by a combination of sub-watershed development and AWD/LWD logging operations (personal communication, Dan Fortin (LWD) 9/17/2019) (Figure 3a). A large swath of AWD/LWD land west of Skillings Corner Road and east of the Basin was clear-cut in the early-2010s, exposing a formerly forested area to rain-induced erosion that would result in P loading (personal communication, Dan Fortin

[1] At a site with only three or four annual data points, neither a mean nor a median is ideal, but if the data points for a site were, for example, 6 ug/L, 38 ug/L, and 40 ug/L, displaying a mean of 28 ug/L seems like a more reasonable depiction of stream conditions than displaying a median value of 38 ug/L.

(LWD) 9/17/19). It would be impossible to know the precise effect of this logging operation on P loading without additional historical data to document what happened, but Dan Fortin's explanation seems like a plausible one, as the effects of logging on P loading are often substantial and long-lasting (Foley et al. 2005; CDM Smith 2013). Other factors could also be at play, however, like leaky septic systems (CEI 2010), erosion along roads (Kratz et al. 1997; Huser et al. 2016a), and residential development (Ryding 1981; Carpenter & Cottingham 1997; Foley et al. 2005). These factors which affect load could have also played out decades ago, as in-lake conditions are often a reflection of changes in catchment conditions which happened 20-30 years ago (Ryding 1981).

The consistently high values at the First Brook outlet (Site 25) appear to be related to the substantial portion of the sub-watershed which is used as residential and agricultural land (Figure 3b). A large part of upper First Brook, including the small, plant-choked pond where the brook originates, is surrounded by agricultural and grazed land. Here, perhaps more than anywhere else in the watershed, are the negative effects of agriculture on water quality visible. Especially when it abuts streams and lakes, agriculture can drive rapid and extreme eutrophication (Carpenter & Cottingham 1997; Carpenter 2003; Duda 1993).

Particularly at the inlets with the lowest discharge, a common trend within years is that mid-summer P concentrations are higher than spring and fall concentrations (Map 6). This is not a trend that has been widely observed in the literature. A pair of large, multi-year Canadian studies, for example, found that nutrient concentrations are typically highest during snowmelt (Yates 2014; Rattan 2017). In the Lake Auburn watershed, this trend of high mid-summer concentrations is most visible in streams with the most variable discharge, suggesting that low mid-summer discharge could be concentrating P concentrations in low-flow streams.

This hypothesis is supported by the two largest and most consistently flowing inlets, the Basin (Site 16) and Townsend Brook (Site 2), not following this trend (Map 6). Indeed, P concentrations at the Basin and, especially, Townsend Brook are far more consistent than many of the smaller streams over the course of the ice-free season, perhaps reflecting continued flow through late-summer and less concentration of P load. Townsend Brook is the most consistently flowing inlet and had the least variation in P concentrations over the course of the summer. Discharge is more variable in the Basin than in Townsend Brook over the course of the summer, but the mid-summer decline in discharge is far less than at most of the smaller inlets. Here, concentrations increase slightly in mid-summer, but increase substantially less than concentrations at most of the smaller inlets (Map 6).

The trend at the outlet (Site 1) also differs from the trend seen in the smaller inlets. Here, P concentrations increase every month over the course of the summer, a trend which is common in temperate lakes (Søndergard et al. 2013; Kalff 2002; Welch & Cooke 2005; Jensen & Andersen 1992). It is unlikely that stream loading drives this increase in loading. The two largest inlets have fairly consistent flow, and discharge at the smaller outlets is extremely low in mid-summer. Thus, this increase is perhaps driven by a combination of two factors. First, some cyanobacteria may descend from the warm and well-lit epilimnion to the cooler, darker, and more nutrient-rich hypolimnion where they take up N and P, and return to the surface (Paerl 1988). These nutrients are then released through leakage and death (Cottingham et al. 2015), making the nutrients available to phytoplankton and increasing epilimnetic P concentrations (Elser et al. 2007; Schindler 1977). Second, declining lake volume, particularly in dry years (Dudley 2004), concentrates in-lake P concentrations.

4.1.3: Stream Phosphorus Load Estimates

Total P load offers a depiction of the relative contribution of various inlets which is, in many ways, a better indicator of loading hotspots. Indeed, though the First Brook inlet (Site 25) and Taber's Driving Range site (Site 3) have the highest P concentrations, each appears to contribute less than 1% of the total P load (Figure 2). Though 1% of the load represents, under the various stream load calculation methods, between about three and 10 kg of P per year, and high concentrations are extremely worrisome, P concentrations should only be one of the two metrics through which management priorities are evaluated. The second key metric is total P load. Both stream loading calculation methods showed that between 75 and 80% of the total stream P load originates in the Basin, and that 18 to 20% originates in Townsend Brook (Maps 8a and 8b; Figure 2). Thus, small variations in concentrations at these larger inlets will result in far more substantial variations in total load than small variations at the smaller sites. Based on the SWAT analyses (Maps 9a-9c), there is no reason to believe that the relative contributions of the various inlets have changed dramatically since 1980, though further analysis would be needed to determine precise estimates. It is also important to note that streams which are currently very small sources of water could contribute more flow in the future due to changing hydrologic conditions, and that the cumulative steam loading estimate using AWD/LWD is roughly half of the total load estimate predicted by SWAT. This means that there are many non-stream sources which are contributing a substantial load, that the stream load estimate using AWD/LWD data is a gross underestimate, or that the loading estimate predicted by SWAT is a gross overestimate.

Variation in estimates of stream P load are mostly a function of the methods used in the calculations. Indeed, stream P load estimates ranged from 253 to 928 kg per year across the distributed and yearly mean estimation methods and the ice-free season and year-round calculation methods (see methods and Appendix C, data manipulations for Maps 8a and 8b for load calculation methods). This variation demonstrates the impact of assumptions about loading on the loading estimate. Assuming that loading occurs year-round instead of during the ice-free season, for example, results in a more than three-fold difference in the estimate under the distributed load calculation method (253 kg versus 928 kg). This difference is driven mostly by the extremely high early spring values which, when distributed back to the midpoint of the latest fall value (usually in October), are applied to several winter months instead of a few spring weeks. Because each site had only six to 10 data points for the entire summer, the challenge when calculating load was deciding which values to use during the time between sampling events (which could be up to two months). Is it best to use one value until the next sample is taken? Should values be extended to the midpoint between values (as was done in the distributed method (Map 8b))? In cases where additional discharge data are available, should concentration values be applied to specific discharge ranges based on available data? If discharge is a poor predictor of concentration (as it was in this data set), is it better to average the concentration and discharge values across the entire year (as was done in the mean method (Map 8a))?

Ultimately, each method is rife with assumptions and pitfalls. The best practice for estimating load given concentration and discharge, known as the ratio estimator method (Quilbé et al. 2006) could not be utilized in the Lake Auburn watershed due to data limitations. In watersheds with at least weekly concentration and discharge data, a ratio can be established to predict concentration for a given amount of discharge (Quilbé et al. 2006). Six to 10 annual data points are insufficient to establish an adequately calibrated ratio. Many things can be learned from the AWD/LWD data, but a nuanced estimate of P load is not one of them. This is one reason why SWAT was adopted, despite the assumptions associated with models (see Appendix A.6).

4.1.4: Sub-Watershed Land Cover Analysis

Sub-watershed land cover is a poor predictor of P concentrations. Though there was a statistically significant negative relationship between the proportion of a sub-watershed that was undeveloped land and sub-watershed P concentrations (Figure 3a), the relationship between the proportion of a watershed with agricultural and developed land and P concentrations were less clear (Figures 3b and 3c). This makes sense because the "developed" land cover category lumps all forms of development. One sub-watershed could have lots of lightly developed land while another could have a smaller percentage of highly developed land which contributes a larger overall P load despite representing a smaller proportion of the sub-watershed (Bremigan et al. 2008; Carpenter & Cottingham 1997). Agricultural land, similarly, lumps hay/pasture and cultivated crops. A tiny percentage of crops could outweigh the P loading from a far larger proportion of hay and pastureland (Duda 1993).

The First Brook sub-watershed (Site 25) has the highest P concentrations but does not have the highest percentage of developed or agricultural land. However, of the developed and agricultural land in the sub-watershed, most development is high-intensity and a large portion of the agricultural land is cultivated crops, two factors which likely contribute to the high concentrations. Compounding matters, the pond where First Brook originates is surrounded by agricultural land, and large portions of the stream abut developed land; the spatial proximity of high intensity use to a stream often expands loading further (Carpenter & Cottingham 1997; Carpenter 2003; Duda 1993). The mid-Townsend Brook sub-watershed (Site 26), meanwhile, has the highest percentage of developed land, but most development is low-intensity and not adjacent to the stream. Similarly, the lower Townsend Brook sub-watershed (Site 2) has the highest percentage of agricultural land, but most of it is hay/pastureland in upper and distant parts of the watershed, perhaps contributing to the P concentrations here being lower than at Site 25 (Figures 3a-3c). An alternative explanation for why land cover is a poor predictor of P concentrations could be the decades-long lag which often exists between catchment nutrient inputs and higher nutrient concentrations being expressed in streams. In other words, current nutrient concentrations in sub-watershed streams could be reflecting the sub-watershed land cover of 20-40 years ago.

The greater spatial resolution in land cover, and the ability to see how land cover, soil, slope, and weather cumulatively affect flow, concentration, and load is the second major reason why SWAT was used. All SWAT results are predicated on the assumption that the Lake Auburn watershed behaves like the watersheds on which the model was calibrated. However, because land cover alone is a poor predictor of P concentrations and the AWD/LWD data allows only for a unnuanced estimate of load, the assumptions associated with SWAT seemed no greater than those associated with using the AWD/LWD data.

4.2: Discussion of SWAT Results

4.2.1: Contextualizing the SWAT Results

Based on the SWAT analysis, the annual Lake Auburn P load has nearly tripled since 1980, from 560 kg to 1671 kg (Figure 4). Based on the most-likely future SWAT scenario, this trend of increasing loads will continue through 2050, when the annual load is predicted to be 2768 kg, a nearly 400% increase from the 1980 baseline (Figure 4). Understanding the roles played by the two key drivers of eutrophication–one of which can be mostly addressed through local action (land-use change) and one of which is dependent on both local and global action (climate change)–will have major implications for the future of water quality in Lake Auburn.

Though the P load estimates from the existing stream data and SWAT analyses are not strictly comparable, it is worth assembling all known P load data for Lake Auburn in a single location. This also includes two watershed reports written by AWD/LWD consultants in the early-2010s (Table 2). The CEI (2010) report provides an extensive break-down of the different types of P loading, including an estimate of loading from streams. Its stream loading estimate of 685 kg per year broadly aligns with the stream loading estimates calculated in the existing data analysis. The total load estimate from the default SWAT scenario is also similar to the estimate developed by the consultants (Table 2). The SWAT default annual load estimate of 1671 kg is about 16% higher than the CEI (2010) estimate of 1432 kg and about 4% higher than the CDM Smith (2013) estimate of 1599 kg (Table 2). Here, it is important to note that just because loading estimates are similar does not mean they are accurate or "correct." All analyses could make the same incorrect assumptions or ignore or overemphasize certain watershed characteristics. It is also possible that the slight increase recorded in this analysis represents the real increase in loading since the early-2010s, with additional loading representing the increased load from

Source	Estimated Stream Loading (kg/year)	Estimated Total Loading (kg/year)
Analysis of AWD/LWD Data: Mean Apr-Dec	732	-
Analysis of AWD/LWD Data: Mean Jan-Dec	975	-
Analysis of AWD/LWD Data: Distributed Apr-Dec	253	-
Analysis of AWD/LWD Data: Distributed Jan-Dec	928	-
SWAT Analysis: Default Scenario	-	1671
SWAT Analysis: Predicted Development Scenario	-	2500
SWAT Analysis: Doubled Predicted Development Scenario	-	2872
SWAT Analysis: Mid-Century Climate Change Scenario	-	2192
SWAT Analysis: Climate Change and Development Scenario	-	2768
SWAT Analysis: Default Scenario using 1980s data	-	560
CEI (2010)	685	1432
CDM Smith (2013)	-	1599

Table 2: P load estimates from the two watershed reports, the analysis of existing AWD/LWD data, and the six SWAT scenarios. See Appendix C, data manipulations for Maps 8a and 8b for descriptions of mean and distributed load calculation methods.

recent development and climate change. There are no other known future scenario analyses of the watershed which could be used for comparison.

4.2.2: Modeled Changes in Phosphorus Loading since 1980

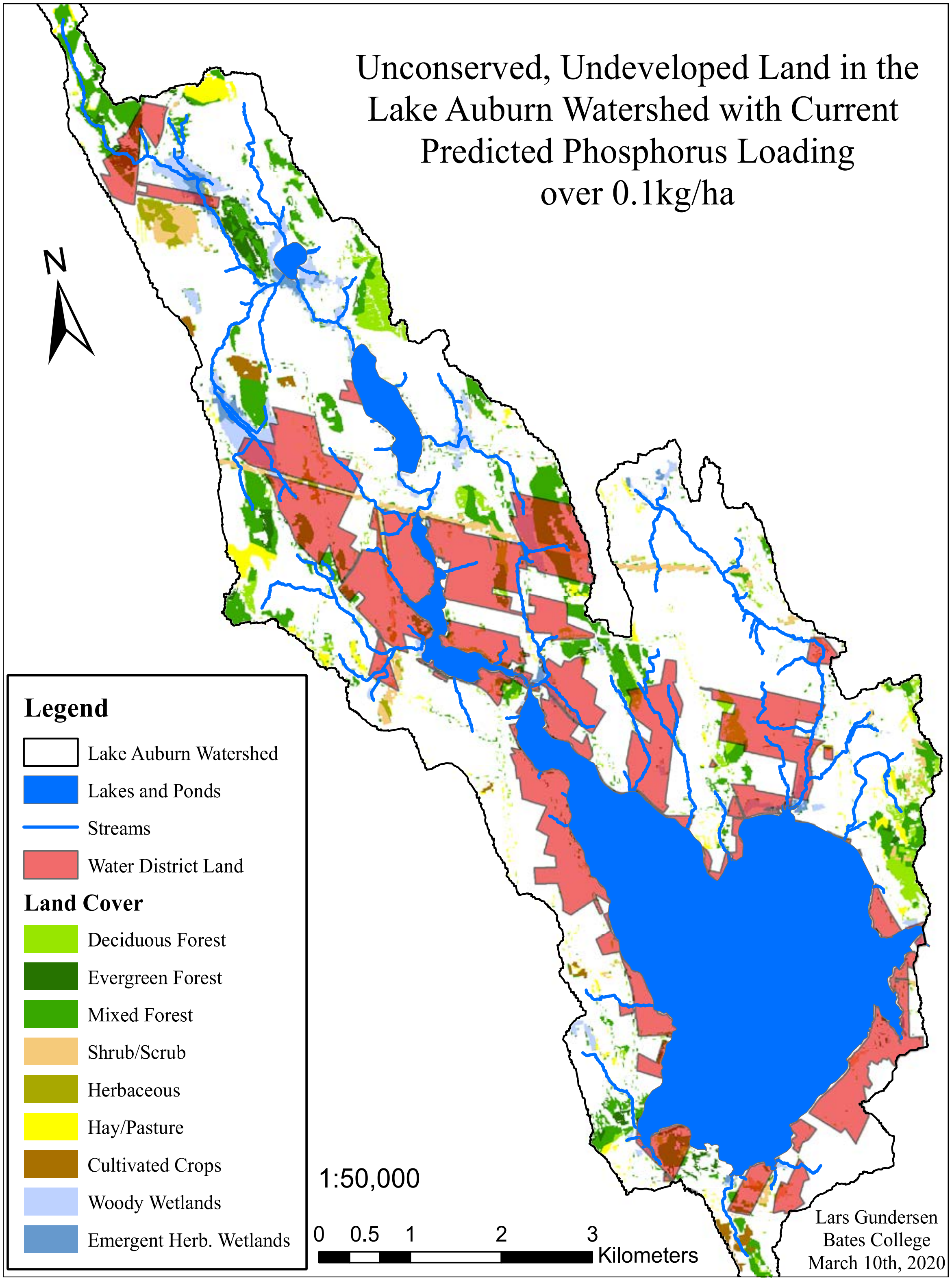
Based on the six SWAT scenarios, land-use change appears to be the primary driver of increased loading since 1980, with climate change playing a substantial secondary role which grows in the future scenarios. When comparing loading predictions under the 2019 and 1980 default scenarios (Figure 4 and Maps 9a, 9b, and 10a), as well as the land use map inputs, it is clear that there has been a lot of development in the watershed over the past 40 years. Most land-use change has consisted of commercial development around the Route 4 corridor, and residential development along Lake Shore Drive, the North Auburn Dam, Skillings Corner, and Holbrook Road (Maps 9a and 9b). Loading has also increased in places which were already developed in 1980, reflecting increasing development intensity in many parts of the watershed which were already developed, and the effects of heavy rainfall-driven loading. The main climate change effect to date appears to be increasing heavy rainfall (Figure 5a), which is also the impact of climate change most likely to cause an increase in P loading (Carpenter et al. 2018; Lathrop et al. 1997). Indeed, the CEI (2010) report on Lake Auburn estimated that extreme precipitation events alone contribute 44% of the lake's total P load. Here, it is also important to reiterate that 1980 is not a true baseline for either development or climate change; it was used because earlier land-use data are not available for the watershed.

Though further analyses would be needed to determine the precise characteristics of land-use change since 1980, it is visually apparent in the land use maps that residential development has been by far the largest driver of land-use change over the past 40 years. Extreme precipitation events have likely increased loading from crop and pastureland, but this effect has probably been less impactful than the effects of substantial residential development, mostly in formerly forested areas (USGCRP 2018). Further analyses would be needed to gain a more detailed understanding of the roles played by residential development versus agriculture. Similarly, to understand the exact extent to which land-use change and climate are responsible for increases in P load between 1980 and 2019, additional SWAT scenarios would have been needed. Running SWAT with 2019 land use data and 1980 weather data, and with 1980 land use data and 2019 weather data, and then comparing those results to the results of the scenarios using all 1980 and all 2019 data, would have allowed for SWAT estimates of climate and development-specific loads.

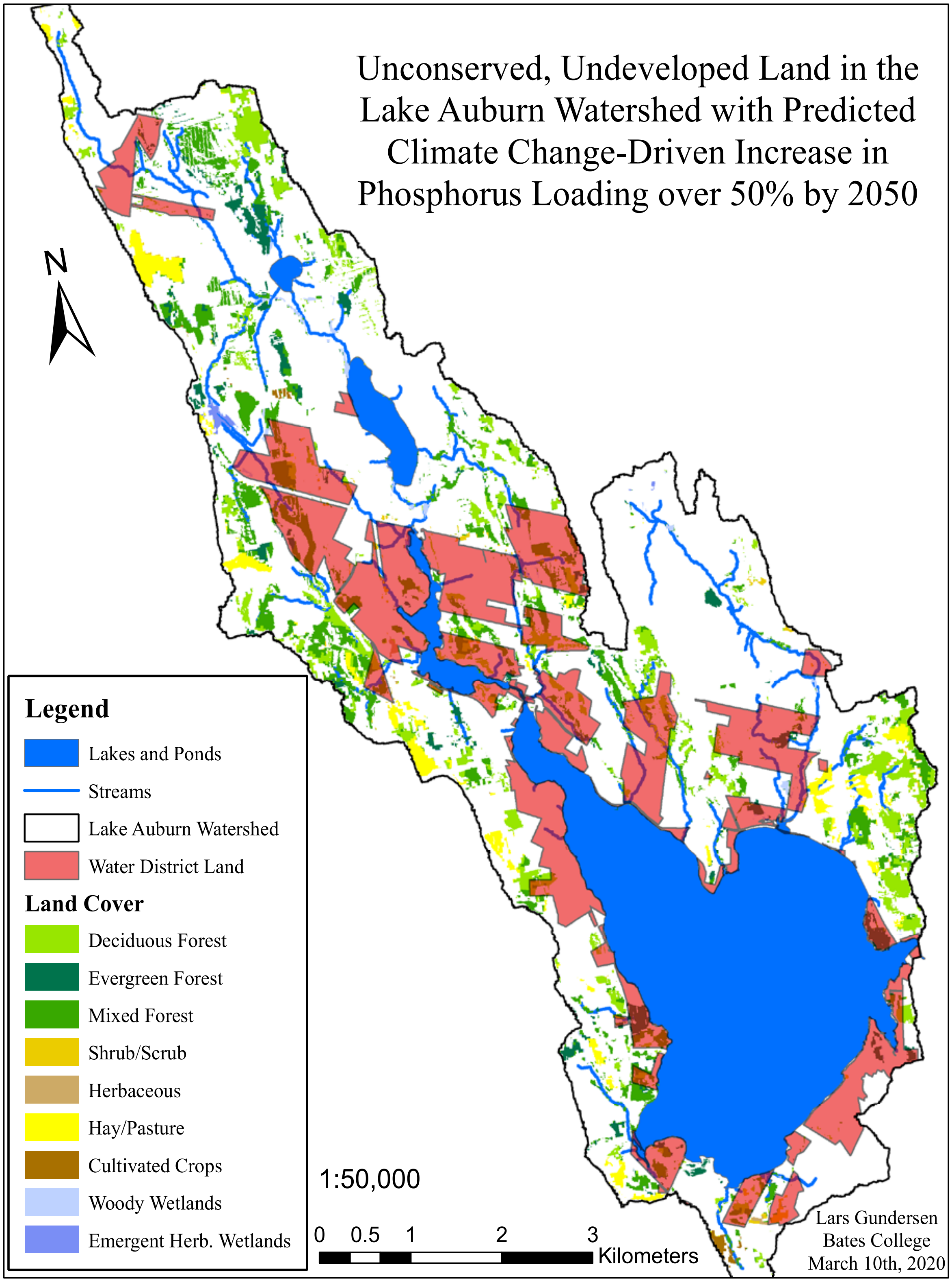
4.2.3: Predictions for Future Loading

Before considering the possible future effects of climate change, it is important to understand the dynamics of current P loading hotspots (Maps 9a and 12a). For the purposes of this analysis, loading hotspots were defined as having per-hectare P loading of over 0.1 kg. This value was chosen using the Jenks optimization method in GIS and splitting the load data into two categories. In addition to covering most developed land, loading hotspots cover substantial tracts of undeveloped land (Map 12a). All cropland, nearly all pasture most wetlands, as well as forests and shrubland in areas with steep slopes contribute at least 0.1 kg/ha of P per year (Map 12a). It is, however, important to note that large swaths of AWD/LWD land, mostly areas which are not forested or have steep slopes, also contribute annual P loads of more than 0.1 kg/ha (Map 12a). This suggests that buying land is not a standalone P load limiting strategy (Map 12a). Indeed, management actions which limit runoff (CEI 2010), halt logging in most places (Cooke et al. 1993), and replant forests are often needed once land is acquired (Novotny & Olem 1994; Huser et al. 2016a).

Phosphorus loading increases in large portions of the watershed under the projected climate change scenario (Figure 4, Map 12b and Appendix G, Map 11e). Again, using the Jenks optimization method in GIS, a P load increase of over 50% was identified as a particularly notable load expansion. Nearly all areas with current loading of 0.1 kg/ha were projected to have climate-change-induced increases in P loading of over 50%. These are locations that are developed or agricultural, have steep slopes, or are not forested, all conditions that are conducive to increased loading with extreme precipitation events (Carpenter et al. 2018; Lathrop et al. 1997). Substantially more forest with moderately steep slopes than was included in the original loading hotspot analysis has a projected climate change-driven increase in loading over 50%, perhaps reflecting the effects of increasing extreme precipitation (Map 12b). Identifying this land has the potential to allow for proactive P loading reductions before climate change increases the load from these areas.



Map 12a: Land in the Lake Auburn watershed with current predicted P loading over 0.1 kg /ha by land cover class. All land cover shown constitutes a loading hotspot, including some Water District land (which is shown for reference).



Map 12b: Land in the Lake Auburn watershed with predicted climate-change-driven increase in P loading over 50% by land cover class. All land cover shown is predicted to constitute a future loading hotspot, including some Water District land (which is shown for reference).

The maps showing P loading given different development scenarios are fraught when it comes to identifying loading hotspots. Assumptions which are built into SWAT land-use split decisions (the process through which SWAT models land-use change) rely on at least three key assumptions, which are worth briefly exploring. The first assumption is that development will be equally divided across land uses and will affect all land-cover types equally. Hay/pastureland and agricultural land may, for example, be more easily developed due to less required clearing of land, or less easily developed due to state and local farmland preservation incentives. Because there is no information from local municipalities on which types of land are most likely to be developed, it was assumed that an equal proportion of each land cover type would be developed.

The second main assumption is that SWAT makes good predictions about what land is “most likely” to be developed. When land uses are split in SWAT, the model assigns the percentage of reclassified land to the places where development is mostly likely: along existing roadways and around existing development. While this is generally a reasonable assumption, it does not allow for conservation land to be excluded from potential development. To resolve this issue, Water District land was recoded into separate land use categories, such that land-use splits would only affect non-Water District land. Thus, Water District land will not be reclassified, regardless of how close it is to existing development. The developers of SWAT claim that the overall effect of randomized, "most likely" land use splitting is negligible, but this component of the model is important to acknowledge. This is especially true when making management decisions. SWAT-designated high-load areas may reflect SWAT development assumptions rather than inherent qualities which cause an area to contribute a large load.

The final assumption is that all future development will be “mixed residential” (URBN). All split land was reclassified into this SWAT land-use category. This development category is the broadest in SWAT and was chosen because while most development in the watershed is expected to be residential (City of Auburn Ordinance Chapter 60, 2010), the density, distribution, and type of development is unknown. The mixed residential category accounts for variability in density, as well as some non-residential development, making it the natural choice for situations in which the exact type of development is unknown. Because SWAT reassigns land-use splits to the locations where the new land use is “most likely” to exist, the results of this analysis are not especially useful for predicting the precise locations of potential future development-driven loading. These scenarios are thus primarily useful for estimating how the watershed-wide P load will change if a certain amount of development occurs.

Despite these caveats, some broad conclusions about the impacts of land-use change on P loading can be drawn from the SWAT analyses. Based on both the projected development scenario and the doubled projected development scenario, the most problematic loading hotspots appear to be along the Route 4 corridor, in the lower part of Townsend Brook, around the North Auburn Dam, around Mud Pond, along Holbrook Road, around Skillings Corner, and northwest of Mud Pond (Appendix G, SM 3a). Most of the increases in loading are predicted around existing roads and development which, based on the model assumptions described above, is expected (Table 2; Figure 4; Map 9a; Appendix G, SM 3a and 3b, and SM 4a and 4b). The effects of development are predicted to result in particularly large P loads in areas with steep slopes, notably east of Route 4 on the far eastern edge of the watershed (Appendix G, SM 3a and 3b; see Map 1 for slope map). Thus, some of the most problematic contributors to P load are steep-sloped areas which have existing residential development and/or agricultural areas, including in the First Brook sub-watershed east of Route 4, the steep-sloped area northwest of Mud Pond, and along Holbrook Road west of the Basin (Map 1 and Appendix G, SM 3a and 3b).

Here, it is important to note that some very small loading hotspots, such as known road drainage hotspots along Spring Road on the western shore of the lake, are too small to be picked up by the land use maps used by SWAT. Other micro-hotspots may also not be captured by the model.

The cumulative impacts of land-use change and climate change, as simulated in the "most likely" future scenario, offer the most nuanced prediction of mid-century watershed P loading (Table 2, Figure 4, and Map 9c and 10b). Predictions from this scenario largely mirror predictions from the projected development scenario, but here, predicted loading is higher along roadways and in areas with steep slopes like east of Route 4, northwest and southeast of Mud Pond, and around Skillings Corner (Map 9c). Indeed, though land use explained 54-60% of nutrient concentration and water quality variance in a large survey of US lakes (Read et al. 2015), and increased forest cover is lower P concentrations (Nielson et al. 2012), weather also plays a crucial role. For example, North et al. (2014) studied an instructive example. In the late-1980s, the Lake of Zurich experienced increased hypoxia even as phosphorus inputs were reduced, large tracts of land were restored to their natural land cover, and increasingly oligotrophic conditions were expected (North et al. 2014). This change was attributed to a dramatic increase in Switzerland's mean air temperature during the 1980s, which increased water temperatures, causing changes in the lake's mixing regime, reduced dissolved oxygen concentrations in the hypolimnion, and increased hypoxia in the fall (North et al. 2014). Thus, it is impossible to separate land use or climate from P loading. However, because of the assumptions associated with SWAT land-use change predictions, it may still be most useful to use the climate change scenario to identify locations where increased climate-driven loading is predicted, then limit development in those areas (Map 12a and 12b, and Appendix G, SM 3c).

4.3: THE LAKE AUBURN PHOSPHORUS LOAD IN BROADER CONTEXT

Phosphorus concentrations in Lake Auburn (measured as kg of P per m^3 of lake volume) are already higher than in many Maine lakes. For example, P load ratios in Moosehead Lake (Maine's largest lake), Sebago Lake (Maine's deepest and second largest lake, and the drinking water supply for much of metropolitan Portland), and Rangeley Lake (one of several massive Androscoggin River headwater lakes) are 3%, 14%, and 33% of the Lake Auburn ratio, respectively (Table 3). Several mid-sized mesotrophic lakes have similar P load ratios to Lake Auburn. Load ratios in Damariscotta Lake, Long Pond, Cobbosseecontee Lake, and China Lake are 66%, 87%, 120%, and 133% of Lake Auburn concentrations, respectively (Table 3). Thus, water quality in Lake Auburn would be expected to be somewhat better than in Cobbosseecontee Lake and China Lake, and somewhat worse than in Damariscotta Lake and Long Pond.

Phosphorus concentrations in Lake Auburn remain far lower than in Maine's most eutrophic lakes like Sabattus Pond and Annabessacook Lake (load ratios are 662% and 635% of Lake Auburn concentrations, respectively) (Table 3). Under the SWAT development and climate change scenario, Lake Auburn's load-to-volume ratio could surpass current load ratios in Cobbosseecontee and China Lakes, and nearly halve the gap in load ratios between Lake Auburn and Sabattus and Annabessacook Lakes. While these other lakes are likely to have similar issues with development and changes in climate, this comparison suggests that water quality in Lake Auburn could be considerably worse than Cobbosseecontee and China Lakes are now by mid-century (Table 3). Water quality is likely to remain substantially better than Sabattus and Annabessacook Lakes are now, at least through mid-century.

China Lake is a drinking water supply where a filtration facility was needed due to cultural eutrophication (Maine DEP 2001), and thus offers a particularly useful comparison for the future of Lake Auburn. This comparison suggests that if both land-use change and climate change proceed as projected through 2050, the P load ratio in Lake Auburn will be about 25% higher than the current ratio in China Lake (Table 3). Considering that water quality in China Lake is currently poor and that the lake required a filtration facility to meet Clean Water Act stipulations nearly 20 years ago (Maine DEP 2001), the “most likely” model prediction suggests that there is no doubt that a filtration plant will be needed by 2050 (and likely much sooner) in Lake Auburn. The “most likely” scenario is not, however, a scenario which is certain to happen. Because it is far easier to control land-use change at the local level than global climate change, local land management decisions will largely determine the future of Lake Auburn.

Without additional development in the watershed, the predicted P load in 2050 would be 2198 kg instead of 2768 kg (Figure 4; Table 3), a reduction of load of nearly one-quarter. This scenario also assumes that there will be no mitigation measures when there are an array of fairly simple and cost-effective interventions which could reduce the P load substantially. Indeed, interventions like better drainage along roads, larger buffer zones between agricultural and residential areas and water, and reforestation in areas with steep slopes have been shown to mitigate at least half of the effects of climate change-induced loading in some catchments (Jeppesen et al. 2009). If this estimate applies to Lake Auburn, these mitigation measures could further reduce load from 2198 kg to an estimated 1935 kg per year. Additional mitigation could also reduce the current load, perhaps by 25% (Jeppesen et al. 2009), which could allow the load in 2050 to be 1726 kg, an increase of only 55 kg from the current load (Table 3). An approach along these lines could allow Lake Auburn to remain mesotrophic and eliminate the need for a filtration facility. The fate of Lake Auburn is in Auburn’s hands. The decisions made over the coming years will determine whether the lake becomes eutrophic and potable only after extensive filtration, or a model for proactive management in the face of impending eutrophication via the dual forces of land-use change and climate change.

Lake	Source/Year	Location	Trophic Status	Estimated P Load (kg)	Load Pct of Lake Auburn	Lake Volume (m^3)	Volume Pct of Lake Auburn	Load/ Volume (kg/m^3)	Load/ Volume Percent of Default Lake Auburn
Moosehead Lake	Lakes of Maine/2004	Piscataquis/ Somerset County	Oligotrophic	2077	124	5.2×10^9	4,678	4.0×10^{-7}	3
Sebago Lake	Cumberland County Soil and Water Conservation District/2015	Cumberland County	Oligotrophic	8240	493	4.0×10^9	3,583	2.1×10^{-6}	14
Rangeley Lake	Maine DEP/2006	Franklin County	Oligotrophic	1795	107	3.6×10^8	323	5.0×10^{-6}	33
Damariscotta Lake	Damariscotta Lake Watershed Associaton/2015	Lincoln County	Oligotrophic/ Mesotrophic	1083	65	1.1×10^8	98	1.0×10^{-5}	66
Long Pond (Belgrade)	Maine DEP/2008	Kennebec County	Mesotrophic	1176	70	9.0×10^7	81	1.3×10^{-5}	87
Lake Auburn	**SWAT default analysis/2019**	**Androscoggin County**	**Mesotrophic**	**1671**	**100**	**1.1×10^8**	**100**	**1.5×10^{-5}**	100
Lake Auburn	**SWAT climate change scenario + interventions/2050**	**Androscoggin County**	**Mesotrophic**	**1726**	**103**	**1.1×10^8**	**100**	**1.6×10^{-5}**	103
Cobbossee-contee Lake	Maine DEP/2005	Kennebec County	Mesotrophic	2828	169	1.6×10^8	142	1.8×10^{-5}	120
Lake Auburn	**SWAT climate change scenario/2050**	**Androscoggin County**	**Mesotrophic**	**2198**	**132**	**1.1×10^8**	**100**	2.0×10^{-5}	132
China Lake	Maine DEP/2001	Kennebec County	Mesotrophic/ Eutrophic	2401	144	1.2×10^8	108	2.0×10^{-5}	133
Lake Auburn	**SWAT development and climate change scenario/2050**	**Androscoggin County**	**Mesotrophic/ Eutrophic**	**2768**	**166**	**1.1×10^8**	**100**	**2.5×10^{-5}**	166
Annabessa-cook Lake	Maine DEP/2004	Androscoggin County	Eutrophic	2817	169	3.0×10^7	27	9.6×10^{-5}	635
Sabattus Pond	Maine DEP/2004	Kennebec County	Eutrophic	2695	161	2.7×10^7	24	10×10^{-5}	662

Table 3: Comparison of selected Maine lakes to Lake Auburn based on P load, lake volume, and load ratio under current (SWAT default) and potential future scenarios.

5: SUMMARY AND RECOMMENDATIONS

5.1: SUMMARY OF MAJOR CONCLUSIONS

1. Though the existing AWD/LWD data offers many useful insights into macro-level watershed characteristics, there are not enough sampling locations or sampling events to make management decisions about potential problem areas *within* sub-watersheds based on these data. AWD/LWD data is also insufficient to calculate a nuanced estimate of stream load.

2. The SWAT model predicts that the 2019 P load for Lake Auburn was 1671 kg. This represents a 198% increase from the 1980 load of 560 kg. Under the most likely future scenario in which development and climate change proceed as projected through 2050, the model predicts load will increase to 2768 kg, a 66% increase from current loading and a 394% increase from 1980 (Figure 4). Thus, it is important to note that a successful intervention will not necessarily reduce load but could instead decrease the amount by which the load increases. This will have important implications for long-term water quality, whether and how soon a filtration facility is needed, and how expensive a potential filtration facility will be to operate (the cost will go up as P concentrations in the lake increase) (Warziniack et al. 2016).

3. All watershed land is contributing P to the total watershed load, but SWAT identified loading hotspots near the North Auburn Dam, along the Route 4 corridor, in the southwest corner of the lake, around Skillings Corner, along Holbrook Road west of Little Wilson Pond, in East Hebron along the Turner town line, and along the lower portion of the Townsend Brook gulley (Map 9a; see Maps 1-3 for watershed orientation and Appendix G, Maps 12a-12d for higher-resolution maps of the areas around Mud Pond, Little Wilson Pond, the Basin, and Townsend Brook).

5.2: RECOMMENDATIONS FOR POLICYMAKERS

1. *Ground-truth the loading hotspots identified by SWAT.* Visit areas in the watershed which the model has identified as high-load to look for signs of runoff and/or erosion-induced loading (Map 13). Draft and implement intervention strategies as needed.

2. *Do not ignore smaller inlets just because they contribute a small percentage of the total load.* Even the smallest inlets can yield kilograms of P per year. Small interventions in some of these streams could reduce a substantial portion of the stream's load, and help contribute to an overall reduction in watershed load.

3. *Continue to increase watershed sampling.* This could involve increasing the frequency of sampling, adding new sampling locations in areas upstream from outlets, and conducting in-lake sampling in the Basin, Little Wilson Pond, and Mud Pond. Focus particular attention on the Basin, Little Wilson Pond, and Mud Pond; this part of the watershed contributes about 80% of the total load according to the AWD/LWD data analysis.

4. *Conduct sampling in a way which improves stream load estimation capacity.* Always collecting both discharge and P concentration data, and conducting additional sampling, would dramatically improve future stream load estimates. Consider sampling both on a regular schedule (e.g., every two weeks for the entire ice-free season) as well as during extreme precipitation events.

5. *Considering limiting residential development and regulating agricultural land in the watershed.* Residential development is currently the main driver of land-use change in the watershed and cultivated crops contribute some of the highest per-hectare loads. Though it is unclear what proportion of land-use change-driven loading is caused by residential development and agriculture, these are two of the key causes of eutrophication in Lake Auburn. Consider monitoring residential developments and cultivated crop areas during heavy rainfall to look for signs of runoff and/or erosion-induced loading, strengthening buffer zone regulations, and limiting further residential development in the most sensitive areas.

6. *Prioritize buying and conserving undeveloped, high-load parcels of land.* Most of the undeveloped land which contributes a large P load is in places with steep slopes or near roads with poor drainage (see Map 12a for undeveloped, high-load land and Map 1 for slope map). Both the load and the amount of undeveloped, high-load land is predicted to increase with climate change (Map 12b). When the LAWPC considers purchasing land, undeveloped, high-load areas should be prioritized. However, buying land is not a standalone solution. Loading can still be very high without interventions to address steep slopes, halt logging operations, reforest disrupted land, and improve drainage along roadways. There are currently large swaths of Water District land which are predicted to contribute large P loads (Maps 12a-12b). Loading in these areas should be addressed.

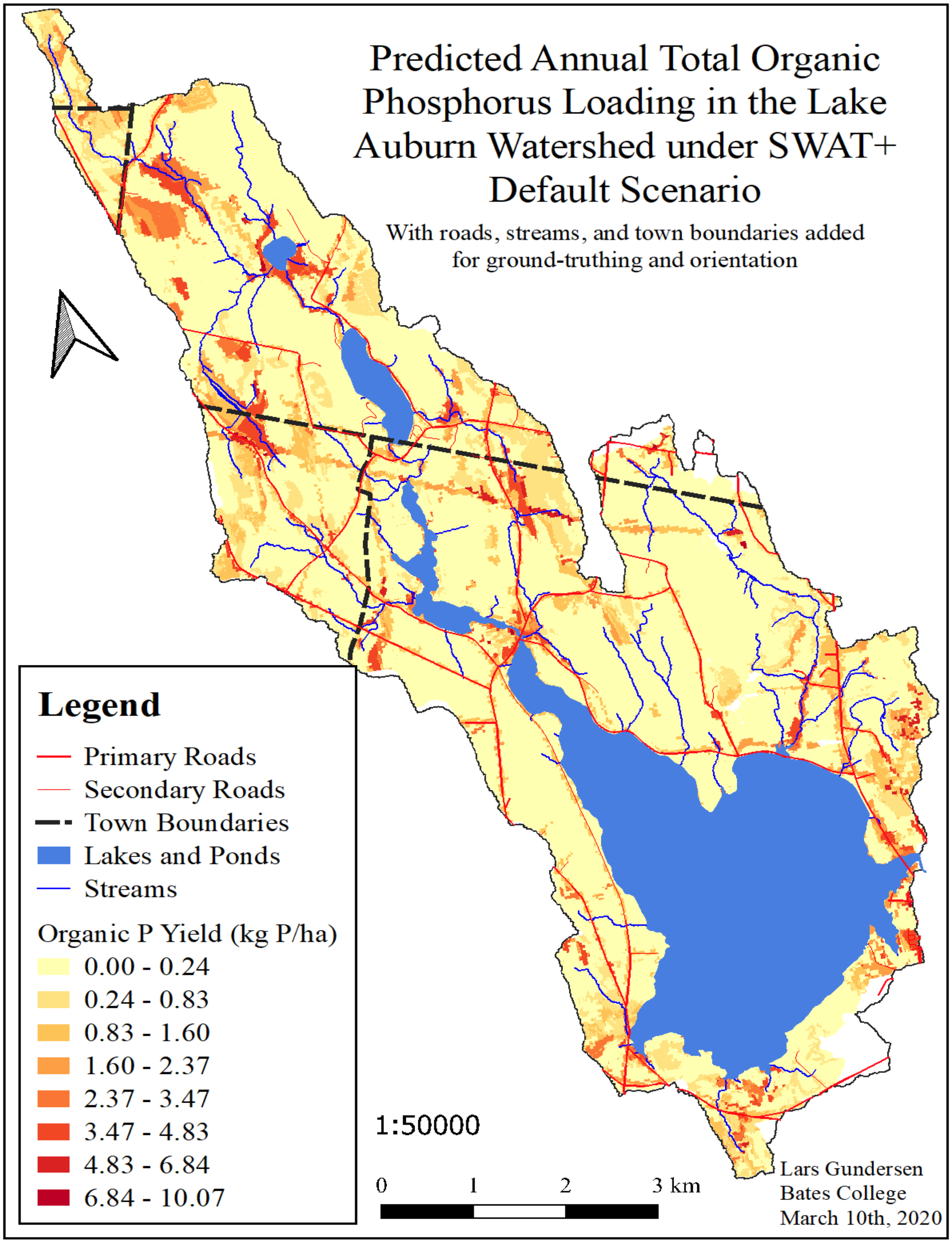
7. *Investigate potentially high loading around Mud Pond.* The AWD/LWD data analysis found that at least half of the Basin load originates in and around Mud Pond and the SWAT analysis also shows the area around Mud Pond as a loading hotspot. Consider conducting sampling in and around Mud Pond to confirm this prediction, then explore mitigation measures (see Appendix G, Map 12a for Mud Pond loading prediction map).

8. *Investigate P loading originating in the Townsend Brook sub-watershed.* Phosphorus concentrations in Townsend Brook are spatially variable. Concentrations at the outlet are low, but upstream concentrations are frequently high, particularly around Roys. Ground-truth potential loading hotspots based on both the AWD/LWD data and SWAT analyses, and draft mitigation measures (Maps 7, 9a, 9c, and appendix G, Map 12d for Townsend Brook loading prediction map).

9. *Consider P load mitigation strategies for First Brook.* Phosphorus concentrations in First Brook are the highest in the watershed, despite the overall load being small. Thankfully, discharge (and thus, load) at First Brook is low. Nonetheless, the brook contributes by far the largest load relative to discharge, mostly because the sub-watershed has so much developed and agricultural land.

10. *Pursue mitigation strategies for any AWD/LWD sampling location with P concentrations over 20 ug/L.* Four of the 13 long-term sampling locations have long-term mean P concentrations over 20 ug/L, and all 13 sites have at least one recorded value over 20 ug/L, with eight of the 13 sites having maximum concentration values over 100 ug/L. Sites with the highest mean P concentrations should be prioritized for interventions. Potential loading hotspots within sub-watersheds can be identified using the SWAT maps (Maps 9a and 13).

11. *Develop management options under the assumption that addressing P loading will also address N loading.* Based on the SWAT analysis, locations with high N loading generally mirror high P load areas. Thus, interventions to address P loading will also address N loading, should the lake be N and P co-limited.



Map 13: Predicted annual total organic P loading in the Lake Auburn watershed under SWAT+ default scenario, with roads, streams, and town boundaries added for ground-truthing and orientation. Roads symbolized in red because they all contribute at least 5 kg of P per hectare.

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DATA SOURCES:

All AWD/LWD data acquired via personal communication with Dan Fortin and/or Chris Curtis.

SWAT data was downloaded from the following sources:

DEM: https://viewer.nationalmap.gov/basic/
Land use (2016): https://www.mrlc.gov/national-land-cover-database-nlcd-2016
Land use (1980): https://water.usgs.gov/GIS/dsdl/ds240/index.html;
SSURGO Soil Data: https://websoilsurvey.sc.egov.usda.gov/App/WebSoilSurvey.aspx
Weather: https://globalweather.tamu.edu

APPENDICES

APPENDIX A: BACKGROUND ON LAKE AUBURN, NUTRIENT LOADING, AND EUTROPHICATION

A.1: Characterizing Nutrient Loading and Eutrophication

A.1.1: Characterizing Eutrophication

Most lakes have two major stable regimes, known as trophic states: Under the first regime, lakes are oligotrophic, meaning P inputs, phytoplankton biomass, and internal P loading is low, and the water is clear. Under the second regime, lakes are eutrophic, meaning P inputs, phytoplankton, and internal loading is high, and the water is turbid (Carpenter 2003; Novotny & Olem 1994). Eutrophication is the process of lakes transitioning from the oligotrophic to eutrophic (Carpenter 2003; Novotny & Olem 1994). Eutrophication is a large and growing problem in lakes, rivers, estuaries, and coastal oceans worldwide (Smith 1998). Eutrophication is driven largely by nutrient, mineral, and sediment over-enrichment (Bartsch 1970; Novotny & Olem 1994). Negative impacts include increased phytoplankton biomass, shifts to toxic and bloom-forming species (Smith 1998), increased water murkiness, drinking water treatment issues, oxygen depletion (Hutchinson 1973), and fish kills (Lee 1972).

In pristine water, aquatic plant and algal nutrients are not available at the ratios needed for maximum plant growth. The rarest nutrients in the water relative to the needs of plants and algae are known as limiting nutrients (Kalff 2002). The addition of a limiting nutrient to a body of water causes expanded primary production because all other necessary nutrients are already present in sufficient supply (Kalff 2002). Not all ecosystems are limited by the same nutrient (Novotny & Olem 1994), some ecosystems are limited by more than one nutrient (Carpenter 2003), and limiting nutrients can change over time as inputs and outputs shift, but in many aquatic ecosystems, plant and algal growth is limited by P (Kalff 2002). Phosphorus-limited aquatic ecosystems thus have sufficient quantities of all necessary nutrients besides P. When P is added, conditions become ideal for expanded primary production (Novotny & Olem 1994).

Eutrophication is both a natural and anthropogenic process (Reckhow et al. 1980). Under natural conditions, lakes "age" via a slow and mostly irreversible process of organic matter, nutrient, and sediment accumulation, which drives gradual eutrophication over millennia. Anthropogenic, "cultural eutrophication" is a far more rapid process in which human activities cause nutrient accumulation which drives eutrophication over years or decades (Reckhow et al. 1980). Cultural eutrophication, caused by human activities and development within watersheds (Ryding 1981), has become an increasingly pressing worldwide issue (Bartsch 1970; Lee 1972; Hutchinson 1973), including in Maine (CDM Smith 2013; CDM Smith 2014). Phosphorus enters lakes through erosion and runoff of sediments (Kitchell & Sanford 1992), agriculture and forestry (Duda 1993), and development (Reckhow & Simpson 1980), and because it is often a growth-limiting nutrient, these additions spur eutrophication (Carpenter 2003).

A.1.2: The Process of Eutrophication

Nutrient-poor lakes are clear because phytoplankton growth within them is strongly nutrient limited (Nurnberg 1984). This oligotrophic state is maintained by low nutrient inputs, which limit phytoplankton growth, which limits decomposition and maintains high enough dissolved oxygen levels in the hypolimnion for it to remain aerobic (Carpenter 2003). If nutrient levels, particularly in the epilimnion, increase, however, phytoplankton growth also increases. Dead plants sink to the bottom of the lake and decompose, an oxygen-intensive process which can use up hypolimnetic oxygen, causing anoxia (Nurnberg 1984). Hypolimnetic anoxia creates a feedback cycle in which increasing anoxia causes more P release from lake sediments (Section

2.4) which causes more phytoplankton growth, decomposition, anoxia, and ultimately, eutrophia (Nurnberg 1984).

Water quality and ecosystem services in lakes are maintained via feedbacks. Oligotrophia is sustained by nutrient retention in wetlands and forests, food web structures, and biogeochemical mechanisms which prevent excess nutrient recycling (Carpenter & Cottingham 1997). In eutrophic lakes, these processes are replaced by feedbacks linked to anthropogenic influences like agriculture, cities, development, and habitat destruction, which maintain the degraded state of the lake (Carpenter & Cottingham 1997). Agricultural and urban watersheds often face the greatest P loading challenges. In these as in all cases, nutrient loading reductions are crucial for eutrophication mitigation (Carpenter et al. 1999).

A.1.3: The Development of Nutrient Limitation (and Co-Limitation) Theory

Though the general relationship between springtime P and N inputs and summer algal blooms has been documented since at least the 1940s (Committee on Restoration of Aquatic Ecosystems 1992), there was debate among limnologists through the 1960s about whether carbon, N, or P was the main growth-limiting nutrient in freshwater ecosystems (Likens 1972). By the 1980s, a general consensus had emerged that P is the primary growth-limiting nutrient in most freshwater ecosystems (Welch & Cooke 1999; North et al. 2015; Ding et al. 2014; Lathrop et al. 1997; Likens 1972). In most lakes with high nutrient loading, P is not the only nutrient entering the water, however, and by the start of the 21st century, some limnologists were again questioning the primacy of P limitation in lakes, for in addition to being P limited, some freshwater aquatic ecosystems are N and P co-limited (Driscoll et al. 2003).

Anthropogenic N inputs have become an increasingly pressing issue in recent decades, with excess N in lakes and streams causing acidification, water quality degradation, and algal growth

(Driscoll et al. 2003). In the Northeastern US, atmospheric N deposition from fossil fuel combustion, largely in the Midwest, is the largest single source of N inputs to lakes (Kalff 2002). Though this source of N pollution is in long-term decline, it still has the potential to disrupt the balance between N and P limitation in lakes (Kalff 2002). In places with substantial atmospheric N deposition or abundant N-fixing life, N is often the second most limited nutrient in aquatic systems (Kalff 2002). Other lakes with lower deposition or fewer N-fixing species may be co-limited by P and N. Nonetheless, in most P-limited and P and N co-limited ecosystems, nutrient loading mitigation measures focus on P because it originates largely from external, anthropogenic sources. This makes P far easier to control than atmospheric deposition and/or fixation of N (Reckhow & Simpson 1980).

Many recent studies have proposed that N and P co-limitation is the most accurate characterization of nutrient limitation in most lakes (Elser et al. 1990; Rolighed et al. 2016; North et al. 2007; O'Donnell et al. 2017; Elser et al. 2009). In an early co-limitation study of Lake Tahoe, Elser et al. (1990) found that combined P and N enrichment led to far more algal growth than either P or N enrichment separately. This led them to conclude that the era of P-limitation primacy was over (Elser et al. 1990). A 2007 follow-up study, also led by Elser, came to similar conclusions: N and P limitation is equivalent in freshwater systems (Elser et al. 2007). They go on to say, however, that because global N and P availability has increased by 100% and 400% respectively due to anthropogenic activities, and because of widely disparate conditions in different lakes and streams, it is difficult to develop a consensus on whether P or N is the more important limiting nutrient (Elser et al. 2007). Thus, they propose joint nutrient controls which identify N and P as the primary limiting nutrients, and iron, silica, sulphur, and potassium as secondary limiting nutrients, depending on regional conditions (Elser et al. 2007).

Other recent studies have given further credence to the co-limitation hypothesis: O'Donnell et al. (2017) found that increasing temperatures are increasing phytoplankton growth in Lake Baikal, furthering both N and P limitation by increasing nutrient usage. In an enrichment experiment, they found that phytoplankton growth increased when both P and N were added together, but remained unchanged when either nutrient was added separately (O'Donnell et al. 2017). Another enrichment experiment in Lake Erie also found that phytoplankton growth is P and N co-limited, with biomass increasing only in scenarios when P and N were added together (North et al. 2007). Finally, the rate of atmospheric N deposition can impact the degree to which a lake is P and/or N-limited. Based on a study of 2053 lakes in Norway, Sweden, and Colorado, Elser et al. (2009) found that anthropogenic increases in atmospheric N impact variations in nutrient limitation across lakes and regions, with lakes in regions with substantial atmospheric N deposition experiencing P-limitation, and lakes in regions with low atmospheric N deposition experiencing N-limitation or P and N co-limitation (Elser et al. 2009). At the macro level, they also found that unforested, tundra catchments generally receive more atmospheric N deposition than forested catchments as a proportion of total loading (Elser et al. 2009).

Still, many studies continue to focus on P as the primary growth-limiting nutrient. In a 37-year study of Ontario's Experimental Lake 227, P was found to be the driver of eutrophication (Schindler et al. 2008; Schindler 1977). Lake 227 was fertilized with constant P inputs and decreasing N inputs, which declined to zero for the last 16 years of the study: after N fertilization ended, N fixation by cyanobacteria remained high enough for biomass growth to continue in proportion to the P inputs, leading to the conclusion that P input reductions should be the primary objective for eutrophication mitigation (Schindler et al. 2008; Schindler 1977).

A.1.4: Past Actions to Address Nutrient Loading and Eutrophication

Historically, one of the most effective ways to slow or reverse eutrophication was to improve sewage and wastewater treatment within sensitive watersheds, a solution which was widely implemented in developed countries by the 1980s (Bouveng 1978; Cullen & Forsberg 1988). Another major anthropogenic source of P has historically been detergents, which routinely contained 12% P through the 1970s (Forsberg 1981). Maine, Indiana, Illinois, Michigan, and Florida were the first states to ban P-based detergents in 1972, and these detergents were phased-out in most states by 1990, with notable exceptions being New Hampshire and California (Litke 1999). The reductions in P loading resulting from better sewage treatment and P detergent bans have shown that cultural eutrophication is reversible (Cullen & Forsberg 1988). Indiana's P detergent ban, for example, resulted in a reduction in per capita annual P pollution from 1.0 +/-0.04 kg/year to 0.5 +/-0.11 kg/year (Reckhow et al. 1980), and many Scandinavian lakes recovered from the beginnings of eutrophication following P detergent bans, advanced wastewater treatment system implementation, and agricultural modifications (Forsberg 1981).

Despite the fact that most incomplete lake recoveries are the result of insufficient P input reductions (Cullen & Forsberg 1988), nutrient loading reductions may not always have an immediately-apparent result: lakes often respond unpredictably to changes in inputs, with some responding better than others to reduced P inputs. Watersheds facing multiple anthropogenic pressures–like development, agriculture, and forestry–are often particularly vulnerable (Carpenter 2003); thus, the only guaranteed way to prevent eutrophication is to proactively limit P loading (Ryding 1981; Cullen & Forsberg 1988).

A.1.5: Current Responses to Nutrient Loading and Eutrophication

Eutrophic lakes are often categorized into three groups based on their resilience: reversible lakes can rapidly return to oligotrophy via P input reductions alone; Hysterectic lakes can slowly return to oligotrophy via extreme long-term P input reductions; And irreversible lakes cannot return to oligotrophy via P input reductions alone (Carpenter et al. 1999). These categories are useful for lake managers and governments seeking to focus recovery efforts on the most resilient lakes (Carpenter et al. 1999).

Once lake resiliency has been established, there are two broad approaches to lake recovery and eutrophication reduction: watershed management and in-lake treatment (Novotny & Olem 1994). All approaches aim to reduce the concentration of limiting nutrients in the water, namely P and sometimes N (Kalff 2002). Because N fixation and atmospheric deposition renders N input control difficult, most approaches focus on limiting P concentrations (Kalff 2002). Many lake managers seek to proactively reduce P before eutrophication becomes an issue, but a lack of understanding of lake regime shifts and constant, subtle changes in lake characteristics makes identifying baselines, determining plausible recovery objectives, and predicting rapid regime shifts very difficult (Carpenter 2003).

A.1.6: The Role of Cyanobacteria

Cyanobacteria can impact nutrient cycling and ecosystem resilience in lakes by accessing N and P not available to phytoplankton (Cottingham et al. 2015; Elser et al. 2007; Schindler 1977; Paerl 1988). Some cyanobacteria descend from warm and well-lit epilimnia to cooler, darker, and more nutrient-rich hypolimnia, gather N and P, then return to the surface (Paerl 1988). These nutrients are then released through leakage and death (Cottingham et al. 2015), making the nutrients available to phytoplankton and increasing epilimnetic nutrient concentrations (Elser et al. 2007; Schindler 1977). Cyanobacteria can occur in both oligotrophic and eutrophic lakes and can access hypolimnetic P during both anoxic and aerobic conditions, meaning this process has widespread implications, especially in vulnerable oligotrophic lakes, where slight nutrient increases can trigger eutrophia (Cottingham et al. 2015). Particularly concerning is the fact that the prevalence of cyanobacterial blooms can increase with warming temperatures (Paerl 1988), meaning that these threats can be expected to worsen with climate change (Cottingham et al. 2015; Jöhnk et al. 2008).

A.2: External Phosphorus Loading

A.2.1: Controls on External Phosphorus Loading

External P loading refers to the portion of a lake's P load which originates outside the lake. Whereas well-vegetated land in its natural state retains most of the nutrients deposited on it or contained within it, human actions often disrupt these natural buffers (Kalff 2002). Anthropogenic activities, like using P-based fertilizers, contribute directly to the amount of P in the watershed which is available to enter the lake, while other activities, like logging and development, cause disruption of soil and bedrock, destruction of buffer areas, erosion, and accelerated weathering, which allows the P within a landscape to become more mobile (Foley et al. 2005). Because P pollution sources are often diffuse, they can be extremely difficult to measure and regulate, compounding nutrient loading challenges (Novotny & Olem 1994).

Lake water quality is influenced by an array of local and regional factors, including physical features like length, fetch, and average and maximum depth, regional water characteristics, landscape position, geology, climate, and land use (Read et al. 2015). In many cases, land use is the largest single determinant of water quality, with variations in land use explaining 54-60% of nutrient concentration and water quality variance in a survey of US lakes (Read et al. 2015). Nutrient concentrations generally increase in watersheds with more development and agriculture (Bremigan et al. 2008). Indeed, variations in watershed agricultural extent alone explained 42% of N concentration variance and 39% of P concentration variance in a study of 210 Danish lakes

(Nielson et al. 2012). Increased forest cover, meanwhile, is associated with lower N and P concentrations (Nielson et al. 2012). However, nearly half of nutrient concentrations variation is not explained by land use (Bremigan et al. 2008). Bremigan et al. (2008) argue that this unexplained variance is explained by other local and regional factors, like basin and watershed morphometry, geology, and surface and groundwater dynamics.

Watershed characteristics, like the presence or absence of upstream lakes and wetlands, are another determinant of nutrient concentrations which are related to landscape position (Martin & Soranno 2006). Because lake P concentrations are influenced by nutrient and water inflows from the surrounding landscape (Zhang et al. 2012), seemingly similar lakes in differing landscape positions may have dramatically divergent chemical and biological properties (Kratz et al. 1997). Landscape position affects the relative importance of precipitation and groundwater inputs: upstream lakes receive more of their inputs from precipitation, and downstream lakes receive more of their inputs from groundwater (Kratz et al. 1997). A largely rain-fed lake will, for example, be more impacted by drought, which often increases mineral concentrations in the lake (Kratz et al. 1997). And by acting as a sink for inflowing nutrients, upstream lakes often cause lower P concentrations in downstream lakes (Zhang et al. 2012).

Wetlands also play a role in downstream P concentrations, though in a less definitive way: Zhang et al. (2012) found that wetlands can be a sink for nutrients via sediment retention or a source of nutrients via concentration through evaporation and macrophyte activity (Zhang et al. 2012). Fisher & Acreman (2004) found similarly that though wetlands frequently act as a sink for both P and N, some wetlands increase eutrophication in downstream lakes by increasing plant biomass, oxygen depletion, and nutrient concentrations (Fisher & Acreman 2004). Additionally, landscape setting can influence maximum primary production depth by controlling climate and light and can influence where human activities like shoreline development and road construction are possible (Kratz et al. 1997).

Precipitation patterns also play a role in determining external P loading variations: droughts result in low levels of runoff and erosion, while floods and heavy rains cause rapid inputs (Lathrop et al. 1997). In Lake Auburn, all of these factors are in play, especially logging, development (CDM Smith 2013; CDM Smith 2014), and the weathering of P-rich apatite, which is found in some parts of the watershed (Hildreth 2008a; Hildreth 2008b). Indeed, lake-specific characteristics account for 70% of the variation in water quality observed across US lakes (Read et al. 2015). The recognition that factors like geology, landscape characteristics, and human activities are the primary drivers of lake health has important implications for management: these characteristics make some lakes prone to having better water quality than others. Thus, scarce resources should be spent on the restoration and protection of lakes with characteristics likely to result in high water quality (Read et al. 2015).

A.2.2: External Phosphorus Loading Reductions and Water Quality

Eutrophication reversal has achieved mixed success. Sometimes, stopping or reversing P inputs is sufficient to reverse eutrophication, particularly in cold, oxygen-rich, deep lakes and lakes with rapid flushing rates. In many other cases, however, eutrophication is not reversible through P input reductions alone (Carpenter 2003). Nonetheless, P input reductions often precipitate long-term improvements in water quality. In a modeling study of Wisconsin's Lake Mendota, Lathrop et al. (1997) predicted that the lake would experience proportional improvements in water quality per unit reduction in P loading. Without P load reductions, they predicted that the probability of a cyanobacterial bloom in Lake Mendota on a given summer day was 60%, while with a 50% P load reduction, the probability of a cyanobacterial bloom on a given summer day drops to 20% (Lathrop et al. 1997).

Though the benefits of P load reductions are substantial, it can often take over a decade for reductions in P loading to yield reductions in hypoxia (Del Giudice et al. 2018; Lathrop et al. 1997). In Lake Erie, decadal cumulative external P loading, along with spring air temperatures, explain most of the interannual variability in summer hypoxia extent (Del Giudice et al. 2018), demonstrating that the lake responds to long-term P input variation as opposed to annual loading (Welch & Cooke 2005). The most effective eutrophication management steps thus involve precautionary P input reductions (Carpenter 2003). Long-term reductions in external P loading are less effective in reducing long-term hypoxic extent, however, when internal loading, from sediment and other sources, provides substantial portions of a lake's P load (Søndergaard et al. 2013).

A.3: Internal Phosphorus Loading

A.3.1: Characterizing Internal Phosphorus Loading

Internal P loading is one explanation for the slow declines in hypoxic extent observed in many lakes following external P load reductions. Internal loading refers to the large store of P and other nutrients which accumulates in eutrophied lakes and is repeatedly rereleased from sediment into the water column during hypoxia (Sas 1989; Welch & Cooke 2005). Phosphorus entering a lake has three potential fates: flowing out, remaining in the water, or being stored in the sediment (Kalff 2002; Carpenter 2003; Cottingham 2015; Nurnberg 1984). Phosphorus stored in sediment remains stored as long as the P is bound to oxygen, iron, calcium, aluminum, or another mineral (Kalff 2002). Under anoxic conditions, however, sediment-stored P is rereleased as organisms break oxygen and mineral bonds to use minerals for respiration in the absence of oxygen (Kalff

2002; Carpenter 2003; Cottingham 2015; Nurnberg 1984). The longer anoxia persists, the more P is released as increasingly large portions of the lake bottom is deprived of oxygen (Kalff 2002; Carpenter 2003; Cottingham 2015; Nurnberg 1984). Well-oxygenated, oligotrophic lakes thus store P for long period of time with little internal loading, whereas anoxic, eutrophic lakes constantly recycle sedimentary P, a process which maintains and expands eutrophia (Kalff 2002; Carpenter 2003; Cottingham 2015; Nurnberg 1984). Mesotrophic lakes like Lake Auburn are somewhere in the middle of these two extremes, with relatively short periods of anoxia causing relatively small releases of P, a process which is always on the verge of tipping into greater anoxia and eutrophia.

Internal P loading can also be driven by the resuspension of P-laden sediment particles, the rerelease of P particles through decomposition, low dissolved oxygen concentrations causing anoxia and expanded P release (Søndergaard et al. 2013), variations in wind affecting oxygenation (Sas 1989), variations in cloudiness affecting photosynthesis and variations in macrophyte activity (Kalff 2002), and variations in sediment type and water chemistry (Sas 1989; Jensen & Andersen 1992; Welch & Cooke 2005). Each of these factors can induce or prevent anoxia and, subsequently, internal loading. Internal P loading can represent a major proportion of a lake's P budget, though the exact proportion varies based on these watershed and in-lake characteristics (Nurnberg 1984). In Lake Mendota, Wisconsin, Soranno et al. (1997) found that the plurality of epilimnetic P is from hypo and metalimnetic entrainment. Though P entrainment varied spatially and temporally–with water column stability (mixing), weather (wind and storms), and P concentrations in the thermocline causing most of the variance–external P inputs only matched entrainment during extreme weather events like torrential rain (Soranno et al. 1997). In Saskatchewan's Lake Diefenbaker Reservoir, North et al. (2015) found that annual

P reloading from sediment represented about 25% of external loading. This sediment loading continued at higher rates through the winter (229 mg TP/m^2 in winter versus 169 mg TP/m^2 in summer), which caused increased phytoplankton growth and reduced dissolved oxygen concentrations for the spring, creating a cycle of increasing hypoxia and internal loading (North et al. 2015).

Several other lake-specific characteristics contribute to internal P loading. Average depth is important for determining algal biomass because P is more readily recycled from sediments within the photic zone, where wind resuspension and bioturbation occur (Welch & Cooke 2005). Internal loading can also happen in deeper parts of a lake, but P from the hypolimnion is unlikely to reach the epilimnion in stratified lakes without unusually substantial mixing and entrainment (Welch & Cooke 2005). Like depth, water temperature affects nutrient release from sediments (Liikanen et al. 2002). Phosphorus flux is correlated positively with temperature and carbon dioxide production and negatively with dissolved oxygen content: temperature and dissolved oxygen explained 47% of internal P flux variance in Finnish lakes, with the role of temperature increasing in anoxic conditions (Liikanen et al. 2002). Finally, water residence times can impact P concentrations, both in water and in sediments (Schindler et al. 1987). In Ontario's intentionally over-fertilized Experimental Lake 227, increased P concentrations were caused by declining water renewal rates following an end of experimental fertilization rather than by internal loading from sediments, because P return from sediments never exceeded 4% of the lake's total P budget (Schindler et al. 1987). Thus, lake-specific characteristics can influence the relative importance of different sources of nutrient loading in different lakes under different conditions.

A.3.2: Factors Driving Anoxia and its Symptoms

Phosphorus accumulated during aerobic conditions is released during anoxic conditions, a process propelled by an array of factors (Kalff 2002). Hypolimnetic dissolved oxygen depletion rates depend on in-lake P concentrations, lake depth, and mean depth and temperature of the hypolimnion, with anoxia being more common in shallower and warmer lakes (Carpenter et al. 1999). Sediment with molar aluminum-to-reducible-iron ratios under three and aluminum hydroxide-to-reducible iron-bound P ratios under 25 is particularly conducive to internal loading under anoxic conditions (Psenner et al. 1984). Applying these findings to Lake Auburn, Doolittle et al. (2018) found molar aluminum to reducible iron ratios between 0.2 and 1.7 and molar aluminum hydroxide to reducible iron-bound P ratios between 2 and 14.5, showing that high internal P loading is likely during periods of anoxia (Doolittle et al. 2018; Doolittle 2015). Sediment in anoxic lakes retains less P than oxic lakes, meaning that anoxic hypolimnia are likely more P-rich, and suggesting that models for oxic lakes overestimate P retention in anoxic lakes, where the lack of oxygen causes different sediment behavior (Nurnberg 1984).

A.4: Local and State Watershed Protection Measures, Laws, and Ordinances

For as long as Lake Auburn has been used as a public drinking water supply, the City of Auburn has undertaken an array of measures to protect the lake's water quality. In the 1880s, the City Council passed an ordinance, still on the books today, stating that "…No Person shall swim, bathe, or engage in other recreational activities on or near Lake Auburn which involve a reasonable likelihood that he will become totally or partially immersed in the Lake or in any stream emptying into the same, nor shall any person wash or clean any clothing or animal in the same, or deposit any filth, waste, or rubbish in or near the Lake" (City of Auburn Ordinance Chapter 22, 2010). More recent ordinances, like Auburn's Zoning Ordinance Section 5.3 aim to maintain water quality and ecosystem services by placing stringent groundwater contamination standards on agricultural areas, requiring a 50ft (15m) buffer strip between water and agricultural land, banning earth disturbances and vegetation removal within 50ft (15m) of water, and mandating that no more than 30ft (9m) of every 100ft (30m) of shoreline be cleared (City of Auburn Ordinance Chapter 60, 2010). Additional ordinances, like Auburn's Zoning Ordinance Section 5.4 require progressively larger buffer zones in areas with slopes steeper than 10%, stipulate that no more than 40% of trees in a given areas can be cleared during any 10 year period, and order that new buildings must be built at least 100ft (30m) from any body of water (City of Auburn Ordinance Chapter 60, 2010) (see Appendix B, ST 11 for buffer retention rates).

Auburn also has ordinances directly related to P control. Auburn Zoning Ordinance Section 5.7, which is intended to protect Lake Auburn and Taylor Pond from continued P loading, outlines building and erosion control standards which aim to reduce external loading (City of Auburn Ordinance Chapter 60, 2010). This section also estimates that an additional 1188 acres (480ha) in the Lake Auburn watershed will be developed over the next 50 years, but states that the P control mechanisms in the ordinance are sufficient to prevent excess loading (City of Auburn Ordinance Chapter 60, 2010).

Another major source of P in the watershed is septic systems, and Auburn's Zoning Ordinance Section 5.3 dictates that leach fields are only allowed in places where the water table is more than 3ft (1m) below the surface, that there must be at least 2ft (0.67m) between the bottom of leach fields and the groundwater, that leach fields cannot be within 300ft (91m) of lakes and streams when soil is more than 70% sand, and that if daily sewage flow is more than 2000 gallons (3785L), septic systems must be 1000ft (300m) from water (City of Auburn Ordinance Chapter 60, 2010). Even with this ordinance, which renders septic standards for homes located within the Lake Auburn watershed stricter than Maine state standards, the standards are more lenient than state standards in southern New England (CEI 2010). Indeed, whereas Maine law requires 1.5ft (0.5m) vertical separation between the bottom of leach fields and groundwater, laws in Vermont, Massachusetts, Connecticut, and Rhode Island require 3ft (1m) vertical separation (Vermont Agency of Natural Resources 2019; Connecticut Department of Public Health 2018; Rhode Island Department of Environmental Management 2010; Massachusetts Department of Environmental Protection 2016). Compounding matters, soils in the Lake Auburn watershed are coarser and more gravelly than in most other parts of Maine (Hildreth 2008a; Hildreth 2008b), resulting in less filtration before effluent reaches groundwater (CEI 2010). Septic systems can leach into lakes from places as far as 100m from the shoreline, and more lenient vertical separation rules cause more loading into lakes (CEI 2010).

A.5: Choices for Human Interventions

A.5.1: Characterizing Human Interventions

One of the issues with preventing and reducing nutrient loading is that the human activities which cause eutrophication also generally create profits (Carpenter et al. 1999). Meanwhile, the symptoms of eutrophication–like increased phytoplankton, growth of toxic species, decreases in water clarity, odor and taste issues, oxygen depletion, and fish kills–contribute to a decline in the ecosystem services provision (Carpenter et al. 1999). Carpenter et al. (1999) thus argue that to maximize the economic benefits of lakes, P inputs should be kept below the calculated levels required to maintain the current state of a lake, allowing for ecosystem services–which are far more economically valuable in the long-term–to continue (Carpenter et al. 1999). Carpenter & Cottingham (1997) conclude similarly that the underlying problem of lake restoration is that those who contribute to eutrophication do not benefit from remediation as much as those who did not contribute to eutrophication. They identify the need for conservation incentives and state that the United States must develop social and institutional mechanisms to achieve this, but do not make any proposals or suggestions (Carpenter & Cottingham 1997). Indeed, governments and lake managers must create incentives to maintain and improve lake water quality. This objective is complicated by the often-impossible challenge of the public wanting a quick solution leading to a rapid recovery at a low cost (Lathrop et al. 1997; Ryding 1981; Carpenter et al. 1999).

Because ecosystem-focused eutrophication management measures–namely reducing external nutrient loading–are frequently expensive and time-consuming to implement (Cooke et al. 1993; Huser et al. 2016a), lake managers and limnologists have developed several one-off (Cooke et al. 1993), relatively rapidly and easily-implemented interventions (Copetti et al. 2016) to address eutrophication, known collectively as geoengineering. Geoengineering strategies are human actions that intervene with biogeochemical processes to control eutrophication (Lürling et al. 2016). They are often employed when external nutrient loading reductions are insufficient to reverse eutrophication (Lürling et al. 2016; Carpenter et al. 1999). Since its first trials on Seattle's Lake Washington in the 1960s, geoengineering has held the promise of silver bullet lake management, a promise which has so far failed to materialize (Kalff 2002). Before engaging in any geoengineering strategy, it is important to conduct a watershed analysis and create a nutrient budget (Huser et al. 2016), in order to ensure proper dosage and maximize program effectiveness (Spears et al. 2016; Lürling et al. 2016). Even with significant planning, geoengineering often fails when external nutrient loading is not sufficiently reduced (Cooke et al. 1993), highlighting a general consensus that geoengineering is most effective when it complements strategies to dramatically reduce external nutrient loading (Welch & Cooke 1999; Welch & Cooke 2005; Spears et al. 2016; Lürling et al. 2016). Cost and application challenges also make many proposed strategies nonviable in practice; the vast majority of geoengineering projects have thus utilized just two methods: P inactivation with alum, and P inactivation with lanthanum-modified bentonite (Lürling et al. 2016).

A.5.2: Phosphorus Inactivation with Alum

One of the best-studied P management interventions is P inactivation with alum (Welch & Cooke 1999; Cooke et al. 1993; Welch & Cooke 2005). Phosphorus inactivation controls sediment P release by binding inorganic sediment P with aluminum salt (aluminum sulfate and/or sodium aluminate) (Welch & Cooke 1999; Huser et al. 2016a; Huser et al. 2016b). Though the costs of P inactivation have dropped since the method was first developed in the 1970s, this intervention is generally only used when all loading reduction options have been exhausted (Cooke et al. 1993; Welch & Cooke 1999). The first comprehensive evaluation of alum treatment effectiveness is Welch & Cooke's (1999) study of 21 lakes across the Northeast, Upper Midwest, and Pacific Northwest which measured alum treatment effectiveness via reductions in lake total P, internal loading rate, and chlorophyll levels both immediately after the treatment and for a 20 year post-application period. In polymictic lakes, internal loading was reduced in six of nine lakes by an average of two-thirds, and total P was reduced by an average of 50% in all lakes for between five and 11 years. In dimictic lakes, internal loading and total P concentrations were both reduced by an average of 80% for four to 21 years (Welch & Cooke 1999).

Two 2016 studies broadly confirmed these findings: in an analysis of 114 alum-treated lakes in the US, Canada, and Europe, Huser et al. (2016b) found that 90% of lakes saw water quality improvements in the first two years after application, that the average treatment lasted 21 years for stratified lakes and 5.7 years for shallow lakes, and that treatment effectiveness variance was explained largely by alum dose (47%) and watershed to lake area ratio (32%) (Huser et al. 2016b). Correct alum dosage is imperative for treatment success, and good estimates of mobile P are necessary for correct dosage calculations (Huser et al. 2016b). A second study by Huser et al. (2016a) in Minneapolis' Chain of Lakes affirmed the finding that alum treatment dose is generally the best predictor of P concentration reductions. Differences in alum application rates in four Minneapolis lakes led to alum treatment effectiveness times varying from four to 20 years (Huser et al. 2016a). In the three lakes where the treatment was effective for the fewest years, external loading was reduced by less than 1%, whereas in the lake where the treatment was effective for 20 years, external loading was reduced by 49% (Huser et al. 2016a). Twenty-five years after alum application, Huser et al. (2016a) note that the treatment did not solve eutrophication in any of the lakes but that when the alum treatment was complemented by substantial external loading reductions, one lake saw improved water quality for 20 years (Huser et al. 2016a). They also calculate that alum treatments are about 50 times less expensive than reducing external loading through land use changes, road reengineering, and improved stormwater management. Based on these findings, they suggest that alum treatments every 15-20 years are a sustainable and cost-effective management option (Huser et al. 2016a).

A.5.3: Phosphorus Inactivation with Lanthanum-Modified Bentonite

Phosphorus inactivation with lanthanum-modified bentonite (LMB), known commercially as Phoslock, is a less-utilized but similarly-functioning alternative to P inactivation with alum (Spears et al. 2016; Copetti et al. 2016). It has been used in about 200 lakes worldwide, 50% of which are in Europe, 30% of which are in Australia and New Zealand, and 13% of which are in North America (Copetti et al. 2016). Lab, mesocosm, and lake studies have found it to be highly effective in doses far below human toxicological levels, making it potentially useful in reservoirs, and potentially in ecologically-sensitive areas (Copetti et al. 2016). However, there has been little research on the potentially negative effects of LMB on benthic invertebrates and primary producers, and its cost is about ten times greater than alum (Copetti et al. 2016).

Though additional research is needed to understand the full ecological implications of LMB use, an analysis of 18 European lakes treated with LMB found that total P values decreased from an average of 0.08mg/L to an average of 0.03mg/L, chlorophyll *a* concentrations decreased from an average of 119μg/L to an average of 74μg/L, and Secchi depths increased from an average of 398cm to an average of 506cm during the first two years (Spears et al. 2016). Yasseri & Epe (2016) meanwhile, found that lanthanum to P ratios in sediment cores must be greater than one for the dose to be most effective, based on a study of Germany's Eichbaumsee (Yasseri & Epe 2016).

A.5.4: Other Geoengineering Approaches

An array of other interventions–many poorly-studied, already widely-implemented, unfeasible, requiring significant political will, or with large ecological consequences–have been proposed or attempted. Nutrient diversion (better waste treatment) has been widely implemented in developed countries (Cooke et al. 1993). Dilution and flushing (adding low-nutrient water to a system) is poorly-studied and unfeasible on large scales (Welch & Cooke 1999). Urban runoff protection (treatment and management of storm water) has been widely implemented and would require political will (funding) to expand (Novotny & Olem 1994). Hypolimnetic withdrawal (siphoning, pumping, or selectively discharging nutrient-rich hypolimnetic water) is poorly-studied and unfeasible on large scales (Cooke et al. 1993). Artificial circulation (preventing or limiting stratification via mixing) is poorly-studied and unfeasible on large scales (Kalff 2002). Food-web manipulations (eliminating or introducing fish species) is poorly studied, has potentially negative ecological consequences, and would thus require political will (Cooke et al. 1993). And aeration and sediment oxygenation is poorly-studied (Prepas & Burke 1997).

A few of these interventions merit further exploration. Dredging is often proposed as a more permanent solution to internal P loading than inactivation with alum, because dredging, in theory, removes sediment-stored P from a lake completely (Welch & Cooke 2005; Novotny & Olem 1994). Though it cost about 30 times more than an alum treatment (about $18,000 per hectare versus about $600 per hectare in 2005) and has profound ecological consequences, dredging often lasts for over 50 years, while alum treatments are rarely more than a short term fix (Welch & Cooke 2005). Aeration, either forced destratification via circulation pumps or hypolimnetic oxygenation, has also been shown to reduce internal loading, particularly when accompanied by external loading reductions (Kalff 2002). Hypolimnetic oxygenation is potentially a simpler, cheaper, more easily-implemented alternative to forced destratification, though more research is required: Prepas & Burke (1997) injected oxygen into the hypolimnion of Amisk Lake, Alberta for two years, increasing mean summer dissolved oxygen concentrations from 1.0 to 4.6 mg/L and decreasing mean P concentrations from 123 to 56 μg/L. Because hypolimnetic hypoxia causes increased P release from sediments, oxygenation is a potentially cost-effective ($30,000 in capital costs and $50,000 per year to inject one ton of oxygen per day in 1997) management option for lakes with low dissolved oxygen concentrations and large proportions for total nutrient loading coming from internal loading (Prepas & Burke 1997). And though P concentrations were reduced less in the epilimnion than in the hypolimnion, reductions in algal growth were substantial because of internal loading reductions (Prepas & Burke 1997).

A.5.5: The Viability of Geoengineering as an Effective Stand-Alone Response

There is some disagreement about whether interventions beyond geoengineering are necessary to maintain and improve lake water quality. Steinman (2019) argues that geoengineering is a short-term solution which should complement reductions in external nutrient loading based on his long-term observations of Spring Lake, Michigan. In the ten years following an alum treatment, P levels increased between 12% and 110% in different parts of the lake, which Steinman (2019) understands to mean that an alum treatment alone is insufficient to address nutrient loading and eutrophication. Osgood (2019) meanwhile, argues that it is more cost-effective to simply apply an alum treatment to a lake whenever P concentrations increase above a determined threshold rather than address external loading issues. Under the scenario proposed by Osgood (2019), lakes with nutrient loading issues and lakes at risk of eutrophication would be treated with alum every five to 20 years indefinitely, with treatment frequency reflecting lake-specific characteristics described earlier. Steinman (2019), however, maintains that this approach amounts to treating the symptom rather than the underlying problem of external nutrient loading. Furthermore, using alum whenever a lake faces nutrient loading or eutrophication issues reduces the urgency of maintaining environmental best practices for local stakeholders, which has the potential to further degrade water quality (Steinman 2019).

One problem with measuring geoengineering effectiveness is that treatment is almost always merged with concerted attempts to dramatically reduce external loading (Welch & Cooke 1999). In the rare case of Wisconsin's Eau Galle Reservoir, which was treated with alum without any attempt to reduce P loading, sediment P was briefly reduced, but algal blooms continued because external loading continued unabated (Welch & Cooke 1999). Thus, alum treatments are often effective in conjunction with dramatic external P loading reductions (Welch & Cooke 1990), but there is no evidence to suggest that alum treatment is effective as either a short or long-term stand-alone solution (Huser et al. 2016b).

Though a complete reliance on geoengineering is controversial and rarely successful, P inactivation, especially when paired with external nutrient loading reduction measures, can dramatically reduce lake recovery times (Hupfer et al. 2016). In Germany's Arendsee, an alum treatment precipitated an immediate improvement in water quality, then gradual, concurrent reductions in nutrient loading ensured its long-term effectiveness (Hupfer et al. 2016). This type of approach, namely merging multiple strategies to address nutrient loading and eutrophication, is gaining increasing credence in limnological circles (Hupfer et al. 2016; Steinman 2019). Waajen et al. (2016) tested the effectiveness of various two-pronged approaches to lake management by dividing two Dutch ponds into six sections, each of which tested a different approach: dredging with biomanipulation (fish control and macrophyte introduction) with and without additions of polyaluminium chloride (a flocculant); Phoslock (LMB) application with biomanipulation (fish control and macrophyte introduction) with and without polyaluminium chloride; Biomanipulation alone; And control (Waajen et al. 2016). Biomanipulation plus other measures which reduced sediment P release were the most effective: dredging and biomanipulation reduced chlorophyll *a* concentrations from a range of 52.5-168.9µg/L to a range of 5.3-6.2µg/L while biomanipulation plus Phoslock reduced chlorophyll *a* concentrations to a range of 5.9-7.6µg/L (Waajen et al. 2016). Combining methods and approaches can reverse eutrophication and increase lake resiliency without relying on a single approach or expecting one-off human interventions to act as silver bullets.

Despite the promise of some geoengineering techniques, it is unlikely that an ecologically-benign, completely effective approach will emerge in the foreseeable future (Cooke et al. 1993;

Welch & Cooke 1999; Kalff 2002; Novotny & Olem 1994; Steinman 2019). Thus, creating a more resilient system which can maintain its trophic state when subject to external or internal disturbances, (Ludwig et al. 1997) ought to be the most important lake management objective. Increasing resilience will require proactive and precautionary action to limit external P loading, difficult political decisions which sometimes prioritize ecology and long-term ecosystem services over short-term economics, and community engagement to ensure local stakeholders understands what is at stake. This approach will reduce long-term costs (Steinman 2019), improve ecosystem service delivery (Novotny & Olem 1994; Welch & Cooke 1999), and preserve recreational opportunities and quality of life (Cooke et al. 1993; Kalff 2002).

A.6: Phosphorus Budgets and Modeling

A.6.1: Phosphorus Budgets

Creating a P budget is one way to broadly understand the origins and dynamics of P within a watershed (Lang et al. 1988; Scavia et al. 2019; Ding et al. 2014; Dillon 1975). Phosphorus budgets quantify P movements within a lake and its watershed with the goal of determining nutrient source and sink values (Reckhow & Chapra 1983). There are multiple methods for calculating P budgets, but it is important to note that the results of all P budgets are estimates (Reckhow & Chapra 1983). It is impossible to measure all P flows within a watershed, so P budgets rely on models, which predict loading based on knowledge of other lakes and any physical measurements which have been made (Reckhow & Chapra 1983). Advances in modeling have led to increasing predictive power and spatial nuance, but results remain estimates (USDA n.d.; Shendge & Chockalingam 2018). A few examples of P budgets illustrate the choices modelers make, how differences in lake characteristics play out, and how model and landscape variability lead to different results.

In a 1988 P budget for Lake St. Clair, Lang et al. (1988) found that Lake Huron, the lake's largest input, contributed 52% of P loading, other non-point sources from within the watershed contributed 43%, and 5% came from the atmosphere, shoreline erosion, and point sources. They also measured outflow and determined internal loading to be the difference between the external load and outflow concentrations (Lang et al. 1988). Whereas this study found that P inputs and outputs were nearly equal, and that internal loading was low to nonexistent (Lang et al. 1988), another study of the Lake St. Clair-Detroit River system conducted by Scavia et al. (2018) found that the lake retains 20% of P inputs (Scavia et al. 2019). Better methods are likely largely responsible for these different results, but increasing P inputs from nearby urban and suburban areas are likely also to blame (Scavia et al. 2019). Because different methods were used in each model, it is impossible to determine to what extent differences in methods and differences in land use explain temporal variation (USDA n.d.; Shendge & Chockalingam 2018; Zhang et al. 2019).

Lakes in different environments often have divergent P budgets (Ding et al. 2014; Dillon 1975). Lake Mead, a dammed reservoir in a drought-prone region, is a major P sink for the Colorado River, retaining about three-quarters of the P that enters it (Ding et al. 2014). Concentrations of P also increase substantially during periods of drought and subsequently intensified P concentration, according to long-term government data and 2000 grab samples collected throughout the reservoir (Ding et al. 2014). Varying flushing rates can also affect outcomes: based on a P budget for Cameron Lake, Ontario, Dillon (1975) proposed the hypothesis that doubled P loading from doubled flow is different from doubled loading from doubled inflow concentrations, and found that despite very high P loading, Cameron Lake remains oligotrophic due to its high flushing rate. A similarly-situated lake adjacent to Cameron

Lake, Four Mile Lake, has similar P concentrations despite experiencing about one-twentieth of the P loading, due to its much slower flushing rate (Dillon 1975).

A.6.2: Characterizing Modeling

Models are simplified representations of real systems, concepts, or processes (Reckhow & Chapra 1983; US EPA n.d.). They are commonly used to predict future conditions based on current trends, or to understand unstudied or poorly-studied systems based on information from similar, better-studied systems (Reckhow & Chapra 1983; US EPA n.d.). Models are useful for organizing, summarizing, and presenting information, and for helping policymakers understand the system(s) a policy could impact (Reckhow & Chapra 1983). The most effective models are designed with a clear project objective, have carefully-defined parameters of interest, utilize methods consistent with available data, and use multiple methods or models to understand uncertainty and improve predictions (US EPA n.d.). Other important considerations include spatial and temporal variability, assumptions, causality (or lack thereof), cost, and sampling design and/or data acquisition (Reckhow & Chapra 1983; US EPA n.d.). Because P is often the growth-limiting nutrient in aquatic systems and is more easily controlled than N in co-limited lakes, most lake models emphasize P loading (Reckhow et al. 1980).

Models can greatly improve understandings of systems where it is not possible to conduct in-depth, lake-specific sampling. If it includes enough inputs for lake-specific factors identified earlier, a model for northern temperate lakes, for example, is transferrable to most temperate lakes, meaning many lakes can be cost-effectively assessed and easily compared (Reckhow et al. 1980). General models ignore inherent differences across lakes, however (Beaulac & Reckhow 1982), meaning they are usually less accurate and reliable than in-lake measurement (Reckhow et al. 1980). Though the complexity and uniqueness of lakes and watersheds makes modeling difficult, there are also shared, nearly-universal characteristics of aquatic systems which models can leverage (Reckhow & Chapra 1983). Models reflect current understandings of systems, and as with any tool, better inputs, applications, and system understandings will make model predictions better (US EPA n.d.). It is essential to note that the importance of model accuracy increases the closer a model comes to being used to make lake management decisions with ecological and economic impacts. In these cases, uncertainty must be minimized and quantified, and the process must be publicized (US EPA n.d.).

A.6.3: Export Coefficient Modeling

In the 1970s and 80s, export coefficients were developed as one of the first general watershed modeling approaches. Export coefficients offer a general nutrient loading estimate per unit area of a given land cover, without regard to soil, slope, climate, or other watershed characteristics (Haith et al. 1992; Reckhow 1980; Reckhow & Simpson 1980; Dillon & Rigler 1975; Beaulac & Reckhow 1982). Export coefficient modeling remains popular because there are minimal data requirements: it is possible to develop a reasonable estimate of P loading with watershed land cover data alone (Khadam & Kaluarachchi 2006). However, export coefficients are usually estimated through field studies of small, single-land-use watersheds, leading to wide ranges in values resulting from differences in measurement and estimation, and natural, spatial, and temporal variability across sites (Strickling & Obenour 2018; Beaulac & Reckhow 1982). Indeed, Khadam & Kaluarachchi (2006) determined that export coefficients for agriculture varied by 72% across temperate climate studies, while export coefficients for urban areas and forest varied by 59% and 58% respectively. Thus, decisions about which export coefficient to use can dramatically change P loading estimates (Khadam & Kaluarachchi 2006).

Despite the potential for substantial inaccuracy, the use of export coefficients for P loading estimation is widespread, and some results are comparable to observed loading. In the United Kingdom's Frome Catchment, P loading was estimated using both export coefficients and sampling: export coefficient modeling found the annual total P load to be 25,605kg, while the measured load was 23,400kg (Hanrahan et al. 2001). This analysis included factors like precipitation and slope, which made it much more accurate than original, export coefficient models (Hanrahan et al. 2001). Many current modeling projects, including the CEI (2010) Lake Auburn P budget, use the traditional, 1980s approach which only considers land cover. Given advances in modeling since this approach was developed, these results are often less nuanced than results from newer models, which often involve many inputs. Multiple-input models typically do a better job accounting for watershed-specific characteristics, and also provide more points of reference for model calibration and validation, which further reduces uncertainty (Shendge & Chockalingam 2018; Zhang et al. 2019).

A.6.4: Geographic Information System (GIS) Modeling

Most models developed after the creation of Geographic Information Systems (GIS) rely on it for greatly expanded spatial and temporal nuance. Soranno et al. (1996) used GIS to model non-point source P loading from land and surface waters in the Lake Mendota, Wisconsin watershed. Using topography, land cover, and pixel-level water flow to map how water travels to the lake, they showed that over half of the watershed contributes little to P loading (Soranno et al. 1996). They also found that urban areas contribute large amounts of P even in the driest years, whereas agricultural and vegetated areas contribute large amounts of P only in wet years and during heavy rain (Soranno et al. 1996). In a similar study of the West Branch of the Delaware River, Endreny & Wood (2003) concluded similarly that while export coefficient modeling offers a reasonably accurate picture of watershed-scale P loading, GIS modeling, which incorporates buffers, slope, and soil, offers a more accurate result which also allows for the identification of disproportionate load areas (Endreny & Wood 2003).

A.6.5: Soil and Water Assessment Tool (SWAT) Modeling

Seeking an improved method for estimating nutrient loading and the impact of various land use decisions, the US Department of Agriculture (USDA) developed the Soil and Water Assessment Tool (SWAT). SWAT is a multi-process model incorporating hydrology, ecology, agriculture, and water quality designed to predict how land management decisions could impact water, sediment, and agricultural yields in complex watersheds. It is popular due to its simplicity (ability to be used by non-experts), predictability (consistently accurate predictions), and stability (ease of running and lack of bugs) (USDA n.d.; Shendge & Chockalingam 2018; Zhang et al. 2019). Using sub-daily precipitation data and many other inputs like slope and soil, SWAT offers greater nutrient loading prediction efficiency and less uncertainty than other models, even in watersheds without monitoring data (USDA n.d.; Shendge & Chockalingam 2018). Underlining its effectiveness in complex, data scarce catchments, Ndomba et al. (2008) found in a study of Tanzania's mostly-unmonitored Pangani River Basin that the model did a good job predicting loading. Relying on elevation, soil type, land cover, and daily rainfall, temperature, humidity, solar radiation, and windspeed data, they found that loading estimates in the watershed had a Nash-Sutcliffe coefficient of 0.65 for monthly flows (a Nash-Sutcliffe coefficient of 1 signifies perfect correlation between model predictions and observed conditions and a coefficient of 0 signifying no relationship) (Ndomba et al. 2008). Given the lack of data in this watershed, SWAT performed well, and far better than any other model tested (Ndomba et al. 2008). In watersheds with more data available, SWAT predictions are even closer to conditions measured in the field and/or validations comparing past SWAT predictions and actual outcomes (USDA n.d.; Zhang et al. 2019).

A.6.6: Other Considerations: Spatial and Temporal Variability, Buffers, and Rainfall Intensity

Recognizing that export coefficient models are unable to quantify uncertainties in model structure, parameters, or predictions (Strickling & Obenour 2018), and that they do not include spatial or temporal variability in weather and management (Xia et al. 2016; Cooper et al. 2014), several other approaches have been proposed for estimating nutrient loading. Xia et al. (2016) proposed a Dynamic Parameter Model (DPM) which emphasizes stream flow variability as the key metric of temporal variability and slope as the key metric of spatial variability. Cooper et al. (2014) developed a "spatio-temporal model" based on Wales' Conwy Catchment which identified soil and landcover as the best spatial predictors of water quality. And Strickling & Obenour (2018) created a Bayesian hierarchical model incorporating precipitation, land use, point source discharges, and livestock operations. All of these provide useful insights and innovations about how different landscape factors influence P loading.

Buffer strips can act to stop pollutants that might otherwise reach water (Weller et al. 2011) and can also be used to determine what proportion of sediment runoff that originates in a given location will enter the water (Novotny & Olem 1994). In a study of 321 sub-watersheds in the Chesapeake Bay watershed, buffer strips blocked 95% of N runoff in the coastal plains, 35% in the Piedmont, and 39% in the mountains (Weller et al. 2011). In addition to being a management consideration, buffers are useful in GIS mapping and P budget modeling: Novotny & Olem (1994) created a nutrient loading scale for locations at various distances from water. They found that locations 22.5m or more from water generally contribute no nutrients, with caveats for places with particularly steep slopes, loose soil, or sparse vegetation (Novotny & Olem 1994).

Loading potential from various distances can be used in GIS to understand areas of disproportionate loading (Novotny & Olem 1994).

Rainfall intensity is another important determinant of nutrient loading. In simulated experiments with rainfall intensities of 30, 50, 65, and 100mm/hour and slopes of 0, 5, and 10º, Zhang et al. (2018) found that compared to bare soil, tall grass could reduce suspended solids runoff by 86-99.5%, total P by 44-89.9%, and particulate P by 92-98.5%. Suspended solids and total P losses were much greater with increasing slope and, especially, rainfall intensity (Zhang et al. 2018). Similarly, Serrano-Muela et al. (2015) found that rainfall intensity is the most important contributor to overland flow and nutrient loss, based on a study of a 2012 flood in the Spanish Pyrenees. Consideration of slope and rainfall intensity is thus crucial for accurate P budget modeling (see Appendix B, ST 12-14 for information on runoff, infiltration and slope).

A.7: Long-term Shifts in Climate and Anthropogenic Climate Change

A.7.1: Increases in Temperature

Long-term shifts in climate, including past, present, and projected anthropogenic climate change, have implications for eutrophication mitigation and recovery. North et al. (2014) studied an instructive example: in the late-1980s, the Lake of Zurich experienced increased hypoxia even as P inputs were reduced and increasingly oligotrophic conditions were expected (North et al. 2014). This paradox was attributed to a dramatic increase in Switzerland's mean air temperature during the 1980s, a change which increased water temperatures and caused changes in the lake's mixing regime, reducing dissolved oxygen concentrations in the hypolimnion, and increasing hypoxia in the fall (North et al. 2014). In light of climate change, it is important to assess factors beyond the scope of nutrient loading when addressing eutrophic conditions, and to consider that nutrient loads reductions may need to be below historic, preindustrial rates to counter the generally-negative impacts of climate change on water quality (North et al. 2014).

In a study of Danish lakes, Rolighed et al. (2016) found that P loading would need to be reduced by 60% to maintain current summer chlorophyll *a* levels under a 6ºC temperature increase scenario (Rolighed et al. 2016). Through simulations of climate warming scenarios ranging from +1º to +6ºC, they found that phytoplankton–and especially cyanobacteria–will benefit from warmer temperatures, effectively making it more difficult for eutrophic lakes to recover and easier for oligotrophic lakes to become eutrophic (Rolighed et al. 2016). Another Danish study had similar findings: under a 6ºC climate warming scenario, phytoplankton, and especially cyanobacteria, will increase dramatically, with increased biomass expected to reduce Secchi reading depths by 83-89% (Trolle et al. 2015). Similarly, Feuchtmayr et al. (2009) found that warmer water resulting from warmer ambient air temperatures will exacerbate most of the problems of eutrophic lakes. Aquatic plant biomass is expected to increase dramatically but ecological diversity will decline. Floating macrophytes which benefit from both light and nutrient-rich epilimnic waters are expected to shade out phytoplankton living deeper in the water column (Feuchtmayr et al. 2009). Warmer water temperatures will have implications extending beyond phytoplankton biomass: Lake Tahoe, for example, is expected to see an annual mean temperature increase of 0.015ºC for at least the first half of the 21st century, and a model predicts it will experience reduced mixing, increased dissolved oxygen limitation in the hypolimnion, and declining Secchi readings (Sahoo & Schladow 2008).

A.7.2: Extreme Precipitation Events

In addition to temperature, climate-change-induced increases in precipitation extremes could play a major role in nutrient loading variations across months and years. Drought is predicted to cause decreased external loading but increased nutrient concentrations in lakes with low water levels; extreme precipitation events and flooding, meanwhile, are predicted to cause short periods of intense loading (Lathrop et al. 1997). In Italy's Lake Maggiore, which is historically oligotrophic and has been recovering from 20th century eutrophication since the 1980s, there is a strong relationship between extreme precipitation events and phytoplankton growth (Morabito et al. 2018). The pulse of nutrients rapidly entering the lake during and after extreme rainfall causes a spike in epilimnetic nutrient concentrations, causing subsequent spikes in phytoplankton growth, especially cyanobacteria (Morabito et al. 2018). Extreme precipitation events and extreme levels of P discharge are also linked in Wisconsin's Lake Mendota, where 11 of the 12 largest 24 hour precipitation events (events with more than 51mm of rainfall) since 1901 happened between 1994 and 2015 (Carpenter et al. 2018). Carpenter et al. (2018) found that external P loading increased linearly with increasing rainfall intensity, and that more precipitation falling as torrential rain and rain-on-snow will increase P loading and make recovery from eutrophication more difficult (Carpenter et al. 2018). Between 1958 and 2016, the Northeastern United States saw a 55% increase in extreme precipitation events, the largest increase of any region in the country (USGCRP 2018; Michon 2019). With further increases expected under even the most modest climate change scenarios, extreme-precipitation-induced nutrient is expected to be a growing issue, particularly in the Northeastern US (USGCRP 2018; Michon 2019).

Appendix B: Tables with Background Information, Preliminary Research, Detailed Methods Information, and Calculations

ST 1: AWD/LWD sampling sites descriptions (listed from southeast to northwest (see Maps 1-3 for watershed orientation and site locations). Location descriptions from firsthand observation; general P assessment and seasonal flow variation information from Dan Fortin (LWD))

Site Name	Site Location	Location Description	Relative Phosphorus Level Assessment	Seasonal Flow Variations	Years Without Data (2005-2019)	Site Coordinates
Site 1	Lake Auburn outlet	The outlet is located on the eastern shore where ME-4 crosses a small arm of the lake, separating the outlet from the rest of the lake.	Usually representative of overall in-lake conditions.	A dam regulates flow. There is always some flow, but it is much reduced in late summer to maintain lake water levels.	2006, 2007, and 2019	44.14655 -70.22900
Site 25	First Brook outlet	Site 25 is located at the Lake Shore Drive crossing of First Brook, 10-15m from the where the brook flows into the lake.	Usually the highest P concentrations in the watershed.	Often dry in July and August except when it rains.	2006	44.15652 -70.23525
Roys	Roys streams and Townsend Brook convergence	The Roys site is at the downhill edge of Pine Acres Golf Course, near where two streams draining the golf course merge with Townsend Brook.	Phosphorus concentrations are often high.	The streams usually go dry in late summer, except during rain events.	2005-2013	44.16282 -70.23790
Site 2	Townsend Brook outlet	The Townsend Brook outlet is at the Lake Shore Drive road crossing. There is a large swampy area on the opposite side of the lake.	Phosphorus concentrations are usually moderate.	Townsend Brook is largely spring fed, and thus flows throughout the ice-free season.	2006	44.15937 -70.24261
Site 26	Townsend Brook and Tot Lot Stream convergence	Site 26 is located just below the convergence of Townsend Brook and the spring-fed Tot Lot stream, which emerges from a small pond just above the Tot Lot and flows through the playground.	Phosphorus concentrations are typically low. Tot Lot stream concentrations are extremely low.	Flow is fairly consistent throughout the ice-free season.	2006	44.16615 -70.23758

Site Name	Site Location	Location Description	Relative Phosphorus Level Assessment	Seasonal Flow Variations	Years Without Data (2005-2019)	Site Coordinates
Site 23	Horse Pond	Site 23 is located at the road crossing of Lake Shore Drive and a small stream which flows near a horse paddock.	Phosphorus concentrations are variable, but often high.	The stream flows very intermittently, and rarely makes it to the lake.	2006 and 2007	44.16045 -70.25080
TBR	Townsend Brook Road (upper Townsend Brook)	The Townsend Brook Road site is located down a small hill from the road where Townsend Brook flows out of a bog and becomes a more rapidly-moving stream.	Phosphorus concentrations are generally low.	The stream flows year-round, but has more flow in the spring.	2005-2013	44.17575 -70.23547
Site 3	Taber's Driving Range	Site 3 is located at this stream's Lake Shore Drive road crossing adjacent to Taber's Driving Range.	Phosphorus concentrations are low.	This stream is spring-fed, and thus flows consistently during the ice-free season.	2006	44.16105 -70.25531
Site 4	Northwest shore intermittent stream	This site is located at the location where an intermittent stream crosses Lake Shore Drive.	Phosphorus concentrations are low.	Flow is inconsistent, and the stream is usually dry by mid-July.	2006	44.16428 -70.26263
Site 13	Basin Bridge	Site 13 is located where the Lake Shore Drive ends at North Auburn Road. Just before the end, Lake Shore Drive crosses the Basin stream, and the sampling location is there.	Phosphorus concentrations are typically fairly low, but load is high due to high flow.	This is the largest source of water to the lake, though inputs are variable and the stream sometimes goes mostly dry in August.	2006	44.17565 -70.27467
Site 16	Basin Dam	The Basin Dam site is located just upstream of Site 13, at the Basin Dam. This site is before the Johnson Road Stream enters the Basin Stream.	Phosphorus concentrations are usually low.	Flow is regulated by a dam. There is usually some flow, except in particularly dry Augusts.	2006	44.17692 -70.27634

Site Name	Site Location	Location Description	Relative Phosphorus Level Assessment	Seasonal Flow Variations	Years Without Data (2005-2019)	Site Coordinates
Site 27	Johnson Road	Site 27 is located at the crossing of Johnson Road and the intermittent Johnson Road Stream.	Phosphorus concentrations are often moderate.	This stream often goes dry by July.	2006	44.17842 -70.27267
Site 18	Wilson Pond inlet	Site 18 is located at the inlet to Wilson Pond, the middle pond in the 3-pond Basin chain.	Phosphorus concentrations are variable, but never more than moderate.	There is flow throughout the ice-free season, though flow declines somewhat in August.	2006	44.21237 -70.29198

ST 2: Calculation of mean P load for AWD/LWD sampling sites with concentration and discharge data

Site	Mean Discharge (m^3/sec)	Mean Concentration (ug/L)	Mean Concentration (µg/L)	Mean Concentration (µg/m^3)	Daliy TP load (kg)	Jdays Apr-Dec	Jdays Full Year
2	0.273	20.5	20.5	20,500	0.483	274	365
3	0.039	9.4	9.4	9429	0.032	274	365
4	0.044	10.2	10.2	10,167	0.038	274	365
13	1.94	12.3	12.3	12,250	2.06	274	365
18	1.01	14.9	14.9	14,857	1.29	274	365
23	0.004	40.3	40.3	40,333	0.015	274	365
25	0.011	49.0	49.0	49,000	0.046	274	365
27	0.025	15.0	15.0	15,000	0.032	274	365
B-1	0.001	162.8	162.8	162,800	0.017	274	365
R-2	0.001	88.3	88.3	88,250	0.006	274	365

ST 3: Calculation of distributed P load for AWD/LWD sampling sites with concentration and discharge data

Date	Jday	Site	Apr-Dec Start JDay	Year-Round Start Jday	End JDay	Apr-Dec Days Using Value	Year-Round Start Jday	Discharge (m^3/sec)	TP (ug/L)	TP (µg/L)	TP (µg/L)	Daily TP Load (kg)	Apr-Dec Daily TP*days(kg)	Year-Round Daily TP*days (kg)
4/1/19	91	2	91	0	98.5	7.5	98.5	0.544	27	27	30000	1.26875	9.52	124.97
4/16/19	106	2	98.5	98.5	112.5	14	14	0.343	19	19	20000	0.5627	7.88	7.88
4/29/19	119	2	112.5	112.5	127	14.5	14.5	0.213	15	15	20000	0.27583	4	4
5/15/19	135	2	127	127	149.5	22.5	22.5	0.152	15	15	20000	0.19708	4.43	4.43
6/13/19	164	2	149.5	149.5	168	18.5	18.5	0.200	22	22	20000	0.38087	7.05	7.05
6/21/19	172	2	168	168	174	6	6	0.644	29	29	30000	1.61399	9.68	9.68
6/25/19	176	2	174	174	198	24	24	0.071	22	22	20000	0.13466	3.23	3.23
8/8/19	220	2	198	198	365	136	167	0.017	15	15	20000	0.02188	2.98	3.65
4/1/19	91	3	91	0	98.5	7.5	98.5	0.132	11	11	10000	0.12558	0.94	12.37
4/16/19	106	3	98.5	98.5	112.5	14	14	0.041	7	7	10000	0.02481	0.35	0.35
4/29/19	119	3	112.5	112.5	127	14.5	14.5	0.028	5	5	10000	0.01201	0.17	0.17
5/15/19	135	3	127	127	149.5	22.5	22.5	0.037	5	5	10000	0.01615	0.36	0.36
6/13/19	164	3	149.5	149.5	170	20.5	20.5	0.020	9	9	10000	0.01549	0.32	0.32

6/25/19	176	3	170	170	198	28	28	0.017	9	9	10000	0.01333	0.37	0.37
8/8/19	220	3	198	198	365	136	167	0.000	20	20	20000	0.00025	0.03	0.04
4/1/19	91	4	91	0	98.5	7.5	98.5	0.114	11	11	10000	0.10823	0.81	10.66
4/16/19	106	4	98.5	98.5	112.5	14	14	0.065	8	8	10000	0.04498	0.63	0.63
4/29/19	119	4	112.5	112.5	127	14.5	14.5	0.021	8	8	10000	0.01424	0.21	0.21
5/15/19	135	4	127	127	149.5	22.5	22.5	0.011	9	9	10000	0.00843	0.19	0.19
6/13/19	164	4	149.5	149.5	170	20.5	20.5	0.049	13	13	10000	0.05482	1.12	1.12
6/25/19	176	4	170	170	365	164	195	0.003	12	12	10000	0.00281	0.46	0.55
4/1/19	91	13	91	0	98.5	7.5	98.5	5.256	13	13	10000	5.9031	44.27	581.46
4/16/19	106	13	98.5	98.5	112.5	14	14	3.874	7	7	10000	2.34284	32.8	32.8
4/29/19	119	13	112.5	112.5	127	14.5	14.5	3.126	10	10	10000	2.70102	39.16	39.16
5/15/19	135	13	127	127	149.5	22.5	22.5	0.974	10	10	10000	0.84162	18.94	18.94
6/13/19	164	13	149.5	149.5	168	18.5	18.5	0.929	15	15	20000	1.20372	22.27	22.27
6/21/19	172	13	168	168	174	6	6	0.861	16	16	20000	1.19001	7.14	7.14
6/25/19	176	13	174	174	198	24	24	0.430	15	15	20000	0.55782	13.39	13.39
8/8/19	220	13	198	198	365	136	167	0.085	12	12	10000	0.08808	11.98	14.71

4/1/19	91	18	91	0	98.5	7.5	98.5	1.739	17	17	20000	2.55374	19.15	251.54
4/16/19	106	18	98.5	98.5	112.5	14	14	2.907	10	10	10000	2.51165	35.16	35.16
4/29/19	119	18	112.5	112.5	127	14.5	14.5	0.981	13	13	10000	1.10231	15.98	15.98
5/15/19	135	18	127	127	149.5	22.5	22.5	0.656	9	9	10000	0.51049	11.49	11.49
6/13/19	164	18	149.5	149.5	170	20.5	20.5	0.602	17	17	20000	0.88349	18.11	18.11
6/25/19	176	18	170	170	198	28	28	0.133	19	19	20000	0.21848	6.12	6.12
8/8/19	220	18	198	198	365	136	167	0.008	19	19	20000	0.01395	1.9	2.33
4/1/19	91	23	91	0	98.5	7.5	98.5	0.005	16	16	20000	0.00694	0.05	0.68
4/16/19	106	23	98.5	98.5	112.5	14	14	0.005	14	14	10000	0.00631	0.09	0.09
4/29/19	119	23	112.5	112.5	127	14.5	14.5	0.008	9	9	10000	0.00591	0.09	0.09
5/15/19	135	23	127	127	149.5	22.5	22.5	0.003	10	10	10000	0.00282	0.06	0.06
6/13/19	164	23	149.5	149.5	170	20.5	20.5	0.004	23	23	20000	0.00832	0.17	0.17
6/25/19	176	23	170	170	365	164	164	0.000	170	170	170000	0.00115	0.19	0.19
4/1/19	91	25	91	91	98.5	7.5	7.5	0.017	73	73	70000	0.10966	0.82	0.82
4/16/19	106	25	98.5	98.5	112.5	14	14	0.020	51	51	50000	0.08984	1.26	1.26
4/29/19	119	25	112.5	112.5	127	14.5	14.5	0.013	29	29	30000	0.03375	0.49	0.49

5/15/19	135	25	127	127	149.5	22.5	22.5	0.009	40	40	40000	0.03004	0.68	0.68
6/13/19	164	25	149.5	149.5	168	18.5	18.5	0.001	43	43	40000	0.00417	0.08	0.08
6/21/19	172	25	168	168	174	6	6	0.022	77	77	80000	0.14935	0.9	0.9
6/25/19	176	25	174	174	198	24	24	0.003	42	42	40000	0.01262	0.3	0.3
8/8/19	220	25	198	198	365	136	167	0.000	37	37	40000	0.00027	0.04	0.04
4/1/19	91	27	91	0	98.5	7.5	98.5	0.038	39	39	40000	0.12885	0.97	12.69
4/16/19	106	27	98.5	98.5	112.5	14	14	0.062	12	12	10000	0.06458	0.9	0.9
4/29/19	119	27	112.5	112.5	127	14.5	14.5	0.007	10	10	10000	0.00608	0.09	0.09
5/15/19	135	27	127	127	149.5	22.5	22.5	0.036	7	7	10000	0.02198	0.49	0.49
6/13/19	164	27	149.5	149.5	170	20.5	20.5	0.003	12	12	10000	0.00333	0.07	0.07
6/25/19	176	27	170	170	365	164	164	0.001	10	10	10000	0.00081	0.13	0.13
4/1/19	91	B-1	91	0	98.5	7.5	98.5	0.002	270	270	270000	0.05256	0.39	5.18
4/16/19	106	B-1	98.5	98.5	112.5	14	14	0.001	200	200	200000	0.0171	0.24	0.24
4/29/19	119	B-1	112.5	112.5	127	14.5	14.5	0.001	110	110	110000	0.0094	0.14	0.14
5/15/19	135	B-1	127	127	148.5	21.5	21.5	0.001	94	94	90000	0.00803	0.17	0.17
6/21/19	172	B-1	148.5	148.5	334	185.5	216.5	0.001	140	140	140000	0.01143	2.12	2.47

4/1/19	91	R-2	91	0	98.5	7.5	98.5	0.003	130	130	130000	0.02845	0.21	2.8
4/16/19	106	R-2	98.5	98.5	112.5	14	14	0.000	81	81	80000	0.0035	0.05	0.05
4/29/19	119	R-2	112.5	112.5	127	14.5	14.5	0.001	50	50	50000	0.00362	0.05	0.05
5/15/19	135	R-2	127	127	149.5	22.5	22.5	0.001	53	53	50000	0.0058	0.13	0.13
6/13/19	164	R-2	149.5	149.5	168	18.5	18.5	0.000	74	74	70000	0.00268	0.05	0.05
6/21/19	172	R-2	168	168	174	6	6	0.001	110	110	110000	0.00646	0.04	0.04
6/25/19	176	R-2	174	174	198	24	24	0.000	78	78	80000	0.00224	0.05	0.05
8/8/19	220	R-2	198	198	365	136	167	0.000	130	130	130000	0.00009	0.01	0.01

ST 4: SWAT land use classification choices

NLCD Land Cover Class	Non-Development Scenarios SWAT Land Use Code	Non-Water District Land SWAT Land Use Code for Development Scenarios	Water District Land SWAT Land Use Code for Development Scenarios
Developed, Open Space	URLD (developed, low density)	URLD (developed, low density)	None
Developed, Low Intensity	URLD (developed, low density)	URLD (developed, low density)	None
Developed, Medium Intensity	URML (developed, medium density)	URML (developed, medium density)	None
Developed, High Intensity	UTRN (developed, high intensity/transportation)	UTRN (developed, high intensity/transportation)	None
Barren Land	BARR (barren)	BARR (barren)	BARR_WD (barren)
Deciduous Forest	FRSD (deciduous forest)	FRSD_TEMS (deciduous forest)	FRSD_SUMS (deciduous forest)
Evergreen Forest	FRSE (evergreen forest)	FRSE_TEMS (evergreen forest)	FRSE_SUMS (evergreen forest)
Mixed Forest	FRST (mixed forest)	FRST_TEMS (mixed forest)	FRST_SUMS (mixed forest)
Shrub/Scrub	SHRB (shrubland)	SHRB (shrubland)	SHRB_WD (shrubland)
Herbaceous	GRAS (grassland)	GRAS (grassland)	GRAS_WD) (grassland)
Hay/Pasture	PAST (pasture)	RNGE_SUST (hay/rangeland)	RNGE_TEST (hay/rangeland)
Cultivated Crops	AGRL (generic agricultural land)	AGRL (generic agricultural land)	None
Woody Wetlands	WEWO (wooded wetland)	WEWO (wooded wetland)	WEWO (wooded wetland)
Emergent Herbaceous Wetland	WEHB (herbaceous wetland)	WEHB (herbaceous wetland)	WEHB (herbaceous wetland)

ST 5: Decisions on SWAT actions requiring a choice for the six scenarios

Step	SWAT Action Requiring a Choice	SWAT Default Scenario	Projected Mid-Century Development Scenario	Doubled Projected Development Scenario	Mid-Century Climate Change Scenario	Development and Climate Change Scenario	SWAT Default Scenario with 1980s Land Use and Weather Data
Step 1	DEM	1/9 arc-second	1/9 arc-second	1/9 arc-second	1/9 arc-second	1/9 arc-second	1/9 arc-second
	Create Streams	Streams from DEM: 9/90ha thresholds for channels/streams	Streams from DEM: 9/90ha thresholds for channels/streams	Streams from DEM: 9/90ha thresholds for channels/streams	Streams from DEM: 9/90ha thresholds for channels/streams	Streams from DEM: 9/90ha thresholds for channels/streams	Streams from DEM: 9/90ha thresholds for channels/streams
	Create Outlet	Outlet at Lake Auburn outlet	Outlet at Lake Auburn outlet	Outlet at Lake Auburn outlet	Outlet at Lake Auburn outlet	Outlet at Lake Auburn outlet	Outlet at Lake Auburn outlet
	Grid	No	No	No	No	No	No
	Create Floodplains	10m Buffer and DEM inversion	10m Buffer and DEM inversion	10m Buffer and DEM inversion	10m Buffer and DEM inversion	10m Buffer and DEM inversion	10m Buffer and DEM inversion
Step 2	Land Use	NCLD 2016 land use	NCLD 2016 land use	NCLD 2016 land use	NCLD 2016 land use	NCLD 2016 land use	NCLD 1980 land use
	SSURGO	USDA ssurgo	USDA ssurgo	USDA ssurgo	USDA ssurgo	USDA ssurgo	USDA ssurgo
	Slope	10%	10%	10%	10%	10%	10%
	Floodplain	DEM inversion	DEM inversion	DEM inversion	DEM inversion	DEM inversion	DEM inversion
	Channel	2% merge	2% merge	2% merge	2% merge	2% merge	2% merge
	Land Use Split	None	19% of BARR, FRSD, FRSE, & FRST_TEMS, SHRB, GRAS, RNGE_SUST, AGRL	38% of BARR, FRSD_TEMS, FRSE_TEMS, & FRST_TEMS, SHRB_WD, GRAS_WD,	None	19% of BARR, FRSD_TEMS, FRSE_TEMS, & FRST_TEMS, SHRB, GRAS, RNGE_SUST, AGRL	None

				RNGE_TEST, AGRL			
	Filters	By land use, soil, and slope (5%)	By land use, soil, and slope (5%)	By land use, soil, and slope (5%)	By land use, soil, and slope (5%)	By land use, soil, and slope (5%)	By land use, soil, and slope (5%)
Step 3	**Weather Data**	US 2016-2019 weather data	US 2016-2019 weather data	US 2016-2019 weather data	US 2016-2019 weather data	US 2016-2019 weather data	US 1977-1980 weather data
	Weather Data Changes	None	None	None	Precipitation time halved; precipitation increased by 25%; monthly mean temperatures increased by 2ºC	Precipitation time halved; precipitation increased by 25%; monthly mean temperatures increased by 2ºC	None
	Weather Generator	Augusta Airport	Augusta Airport	Augusta Airport	Augusta Airport	Augusta Airport	Augusta Airport
	Run Years	2016-2019	2016-2019	2016-2019	2016-2019	2016-2019	1977-1980
	Calibration Years	2016-2018	2016-2018	2016-2018	2016-2018	2016-2018	1977-1979
	Output Years	2019	2019	2019	2019	2019	1980

ST 6: Land use split calculations for development scenarios

Land Cover Class	Total Land (ha)	Land Cover Class % of Total Land	Water District Land (ha)	Water District % of Total	Developable?	Developable (non-Water District) (ha)	% Developable Land in LC	Amount of Land to Develop for ProDev (ha)	% Land to Develop for ProDev	% Land to Develop for 2x ProDev
Developed, Open Space	215	6%	0	0%	No	0	0	0	0	0
Developed, Low Intensity	84	2%	0	0%	No	0	0	0	0	0
Developed, Medium Intensity	37	1%	0	0%	No	0	0	0	0	0
Developed, High Intensity	7	1%	0	0%	No	0	0	0	0	0
Barren Land	92	2%	26	28%	Yes	66	3%	13	19%	38%
Deciduous Forest	857	22%	218	25%	Yes	638	25%	122	19%	38%
Evergreen Forest	414	11%	127	31%	Yes	286	11%	55	19%	38%
Mixed Forest	1397	36%	316	23%	Yes	1081	43%	207	19%	38%
Shrub/Scrub	126	3%	29	23%	Yes	96	4%	18	19%	38%
Herbaceous	36	1%	12	34%	Yes	24	1%	5	19%	38%
Hay/Pasture	299	8%	5	2%	Yes	293	12%	56	19%	38%
Cultivated Crops	27	1%	0	0%	Yes	27	1%	5	19%	38%
Woody Wetlands	203	5%	60	30%	No	0	0	0	0	0
Emergent Herbaceous Wetlands	39	1%	10	26%	No	0	0	0	0	0
Totals (where applicable)	3832	n/a	804	21%	n/a	2513	n/a	480	n/a	n/a

ST 7: Export Coefficients from the Literature

Parameter	Beaulac & Reckhow (1982) Kg/Ha/Yr	CEI 2010 Kg/Ac/Yr	CEI (2010) Kg/Ac/Day	CEI (2010) Kg/Ha/Yr	Dillon & Kirchner (1974) Mg/M2/Yr-Low	Dillon & Kirchner (1974) Mg/M2/Yr-Avg	Dillon & Kirchner (1974) Mg/M2/Yr-High	Dillon & Rigler (1975) Mg/M2/Yr
Deciduous woodland				0.00600				
Coniferous woodland				0.00600				
Forest with igneous rock					2.50000	4.80000	7.70000	4.70000
Mixed forest with igneous rock					8.10000	11.70000	16.00000	10.20000
Forest with sedimentary rock					6.70000	10.70000	14.50000	11.70000
Forest/pasture with sedimentary rock					20.50000	28.80000	37.00000	23.30000
Climax hardwood forest								
Mixed pine and hardwood								
Wetland				0.00600				
Scrub/grass/orchard								
Hay/grass	0.25000			0.20600				
Grazed turf/pasture	0.85000			0.20600				
Tilled land				0.20600		46.00000		
Suburban/rural development			0.00081					
Urban			0.00445		110.00000		1660.00000	
Septic tanks								
Solid manure storage area								
Precipitation		0.08100						

Endreny & Wood (2003) Kg/Cap/Yr-Min	Endreny & Wood (2003) Kg/Cap/Yr-25 Pct	Endreny & Wood (2003) Kg/Cap/Yr-50 Pct	Endreny & Wood (2003) Kg/Cap/Yr-75 Pct	Endreny & Wood (2003) Kg/Cap/Yr-Max	Endreny & Wood (2003) Kg/Ha/Yr-Min	Endreny & Wood (2003) Kg/Ha/Yr-25 Pct	Endreny & Wood (2003) Kg/Ha/Yr-50 Pct	Endreny & Wood (2003) Kg/Ha/Yr-75 Pct	Endreny & Wood (2003) Kg/Ha/Yr-Max
					0.03000	0.04000	0.07000	0.12000	0.19000
					0.04000	0.06000	0.20000	0.28000	0.31000
					0.13000	0.21000	0.28000	0.69000	0.97000
					0.10000	0.10000	0.10000	0.10000	0.10000
					0.13000	0.21000	0.28000	0.69000	0.97000
					0.13000	0.21000	0.28000	0.69000	0.97000
					0.19000	0.49000	0.93000	2.45000	4.85000
0.74000	1.16000	1.46000	1.52000	3.00000					
					0.18000	0.19000	0.24000	0.35000	0.54000

Frink (1991) Kg/Ha/Yr	Haith et al.(1992) Kg/Ha/Day	Haith et al.(1992) Kg/Ha/Yr	Haith et al. (1992) Kg/Ha/Yr	Hanrahan et al. (2001) Kg/Ha/Yr	Reckhow et al.(1980)a Kg/Ha/Yr	Reckhow et al. (1980)a Kg/Cap/Yr	Reckhow et al. (1980)b Kg/Ha/Yr-Low	Reckhow et al. (1980)b Kg/Ha/Yr-Most Likely	Reckhow et al. (1980)b Kg/Ha/Yr-High
			0.00600	0.02000	0.47000		0.10000	0.20000	0.30000
			0.00600	0.02000			0.10000	0.20000	0.30000
					0.90000				
0.12000					0.27000				
			0.00600						
				0.02000					
0.55000			0.20600	0.20000	0.64000		0.20000	0.40000	1.30000
0.53000			0.20600	0.20000	0.85000		0.20000	0.40000	1.30000
2.30000			0.20600	0.66000					
1.18000	0.00200			0.83000	1.08000		0.35000	0.90000	2.70000
1.49000	0.01100			0.83000			0.35000	0.90000	2.70000
						1.47700			
					356.00000				
		0.20000					0.15000	0.30000	0.50000

Reckhow et al. (1980)b Kg/Cap/Yr-Low	Reckhow et al. (1980)b Kg/Cap/Yr-Most Likely	Reckhow et al. (1980)b Kg/Cap/Yr-High	Reckhow & Simpson (1980) Kg/Cap/Yr-Low	Reckhow & Simpson (1980) Kg/Cap/Yr-Most Likely	Reckhow & Simpson (1980) Kg/Cap/Yr-High	Reckhow & Simpson (1980) Kg/Km2/Yr-Low	Reckhow & Simpson (1980) Kg/Km2/Yr-Most Likely	Reckhow & Simpson (1980) Kg/Km2/Yr-High
						2.00000	22.50000	45.00000
						2.00000	22.50000	45.00000
						10.00000	105.00000	300.00000
						50.00000	190.00000	500.00000
0.30000	0.60000	1.00000	0.30000	0.65000	1.80000			
						15.00000	35.00000	50.00000

ST 8: Consensus Export Coefficients (from Haith et al. (1992); Hanrahan et al. (2001); Reckhow (1980); Reckhow & Simpson (1980); Endreny & Wood (2003); Dillon & Rigler (1975); CEI (2010); Beaulac & Reckhow (1982); Frink (1991); Dillon & Kirchner (1974); Reckhow et al. (1980))

Parameter	Literature Estimates for Parameter	Mean Export Coefficient Value (Kg/M2/Yr)	Minimum Export Coefficient Value (Kg/M2/Yr)	Maximum Export Coefficient Value (Kg/M2/Yr)	Export Coefficient Range (Kg/M2/Yr)	Median Export Coefficient Value (Kg/M2/Yr)
Climax hardwood forest	1	0.00009000	0.00009000	0.00009000	0.00000000	0.00009000
Coniferous woodland	14	0.00001584	0.00000060	0.00004500	0.00004440	0.00001500
Deciduous woodland	15	0.00001498	0.00000060	0.00004700	0.00004640	0.00001000
Forest with igneous rock	4	0.00000493	0.00000250	0.00000770	0.00000520	0.00000475
Forest with sedimentary rock	4	0.00001090	0.00000670	0.00001450	0.00000780	0.00001120
Grazed turf/pasture	14	0.00005016	0.00001300	0.00013000	0.00011700	0.00003400
Hay/grass	14	0.00003180	0.00001000	0.00013000	0.00012000	0.00002030
Mixed forest and pasture with igneous rock	4	0.00001150	0.00000810	0.00001600	0.00000790	0.00001095
Mixed forest and pasture with sedimentary rock	4	0.00002740	0.00002050	0.00003700	0.00001650	0.00002605
Mixed pine and hardwood	2	0.00001950	0.00001200	0.00002700	0.00001500	0.00001950
Precipitation	13	0.00254963	0.00001500	0.03278025	0.03276525	0.00003000
Scrub/grass/orchard	6	0.00003833	0.00000200	0.00009700	0.00009500	0.00002450
Solid manure storage area	1	0.03560000	0.03560000	0.03560000	0.00000000	0.03560000
Suburban/rural development	8	0.00009724	0.00000090	0.00027000	0.00026910	0.00008650
Tilled land	13	0.00007894	0.00001000	0.00030000	0.00029000	0.00004600
Urban	17	0.00026085	0.00000493	0.00166000	0.00165507	0.00011000
Wetland	2	0.00000060	0.00000060	0.00000060	0.00000000	0.00000060

ST 9: Consensus Septic Export Coefficients (from Haith et al. (1992); Hanrahan et al. (2001); Reckhow (1980); Reckhow & Simpson (1980); Endreny & Wood (2003); Dillon & Rigler (1975); CEI (2010); Beaulac & Reckhow (1982); Frink (1991); Dillon & Kirchner (1974); Reckhow et al. (1980))

Parameter	Literature Estimates for Parameter	Mean Export Coefficient Value (Kg/Cap/Yr)	Minimum Export Coefficient Value (Kg/Cap/Yr)	Maximum Export Coefficient Value (Kg/Cap/Yr)	Export Coefficient Range (Kg/Cap/Yr)	Median Export Coefficient Value (Kg/Cap/Yr)
Septic tanks	12	1.16725	0.30000	3.00000	2.70000	1.08000

ST 10: Phosphorus load estimates for streams with long-term AWD/LWD data under two load estimate assumptions (distributed and mean), each with two timeframes.

Site	Incl. in Total Load	Dist. Apr-Dec 2019 Load (kg)	% of Total	Dist. Jan-Dec 2019 Load (kg)	% of Total	Mean Apr-Dec 2019 Load (kg)	% of Total	Mean Jan-Dec 2019 Load (kg)	% of Total
2	Yes	49	20%	165	18%	132	18%	176	18%
3	Yes	2.6	1.0%	14	1.5%	8.8	1.2%	12	1.2%
4	Yes	3.5	1.4%	13	1.4%	11	1.4%	14	1.4%
13	Yes	193	76%	730	79%	563	77%	750	77%
18	No	108	-	341	-	353	-	470	-
23	Yes	0.65	0.26%	1.3	0.14%	4.0	0.55%	5.4	0.55%
25	Yes	4.6	1.8%	4.6	0.50%	13	1.73%	17	1.7%
27	No	2.7	-	14	-	8.8	-	12	-
B-1	No	3.42	-	8.2	-	4.8	-	6.3	-
R-2	No	0.61	-	3.2	-	1.7	-	2.3	-
Sum		**253**	**100%**	**928**	**100%**	**732**	**100%**	**975**	**100%**

ST 11: Buffer Retention (Endreny & Wood 2003)

Buffer Strip Width (m)	Percent Runoff Retention	Percent Runoff Reaching Water
1	32	68
5	58	42
10	66	34
15	71	29
20	75	25
25	78	22
30	80	20

ST 12: Runoff and Infiltration (Tarboton 2003 (Green-Ampt, Horton, Philip); USDA 2004)

Method	Rainfall Amount (in)	Rainfall Amount (cm)	Rainfall Intensity (cm/hr)	Infiltration (cm)	Runoff (in)	Runoff (cm)	Percent Infiltration	Percent Runoff
Green-Ampt		0.3	1.2	0.3		0	100	0
Green-Ampt		0.4	1.6	0.4		0	100	0
Green-Ampt		0.5	2	0.5		0	100	0
Green-Ampt		0.6	2.4	0.599995		0.00005	99.999167	0.0083333
Green-Ampt		0.7	2.8	0.554		0.146	79.142857	20.857143
Green-Ampt		0.8	3.2	0.497		0.303	62.125	37.875
Green-Ampt		0.4	1.6	0.4		0	100	0
Green-Ampt		0.6	2.4	0.441		0.159	73.5	26.5
Green-Ampt		0.6	2.4	0.422		0.178	70.333333	29.666667
Horton		0.3	1.2	0.3		0	100	0
Horton		0.4	1.6	0.4		0	100	0
Horton		0.5	2	0.5		0	100	0
Horton		0.6	2.4	0.6		0	100	0
Horton		0.7	2.8	0.668		0.032	95.428571	4.5714286
Horton		0.8	3.2	0.518		0.282	64.75	35.25
Horton		0.4	1.6	0.396		0.004	99	1
Horton		0.6	2.4	0.351		0.249	58.5	41.5
Horton		0.6	2.4	0.311		0.289	51.833333	48.166667
Philip		0.3	1.2	0.3		0	100	0
Philip		0.4	1.6	0.4		0	100	0
Philip		0.5	2	0.5		0	100	0

Philip		0.6	2.4	0.6		0	100	0
Philip		0.7	2.8	0.6997		0.0003	99.957143	0.0428571
Philip		0.8	3.2	0.635		0.165	79.375	20.625
Philip		0.4	1.6	0.4		0	100	0
Philip		0.6	2.4	0.52		0.08	86.666667	13.333333
Philip		0.6	2.4	0.481		0.119	80.166667	19.833333
USDA	1	2.54		2.54	0	0	100	0
USDA	2	5.08		5.0292	0.02	0.0508	99	1
USDA	3	7.62		7.112	0.2	0.508	93.333333	6.6666667
USDA	4	10.16		8.89	0.5	1.27	87.5	12.5
USDA	5	12.7		10.3886	0.91	2.3114	81.8	18.2

ST 13: Runoff and Slope (Zhang et al. 2018)

Parameter	Slope	Rainfall Intensity (mm/hr)	Land Use	Runoff Rate (g/m2/hr)	Runoff Rate (mg/m2/hr)	Percent Runoff	Runoff Rate (kg/m2/hr)
Suspended solids	0	30	Bare Land		0.03		0.000000003
Total P	0	30	Bare Land	0.1			0.0001
Dissolved P	0	30	Bare Land	0.06			0.00006
PP/TP	0	30	Bare Land			60	
Suspended solids	0	50	Bare Land		1		0.0000001
Total P	0	50	Bare Land	0.66			0.00066
Dissolved P	0	50	Bare Land	0.1			0.0001
PP/TP	0	50	Bare Land			15.15	
Suspended solids	0	65	Bare Land		1.19		0.000000119
Total P	0	65	Bare Land	0.87			0.00087
Dissolved P	0	65	Bare Land	0.09			0.00009
PP/TP	0	65	Bare Land			10.34	
Suspended solids	0	100	Bare Land		5.37		0.000000537
Total P	0	100	Bare Land	2.3			0.0023
Dissolved P	0	100	Bare Land	0.22			0.00022
PP/TP	0	100	Bare Land			9.57	
Suspended solids	5	30	Bare Land		10.75		0.000001075
Total P	5	30	Bare Land	6.05			0.00605

Dissolved P	5	30	Bare Land	5.39			0.00539
PP/TP	5	30	Bare Land			89.09	
Suspended solids	5	50	Bare Land		24.41		0.000002441
Total P	5	50	Bare Land	11.91			0.01191
Dissolved P	5	50	Bare Land	9.97			0.00997
PP/TP	5	50	Bare Land			83.71	
Suspended solids	5	65	Bare Land		96.99		0.000009699
Total P	5	65	Bare Land	33.08			0.03308
Dissolved P	5	65	Bare Land	32.71			0.03271
PP/TP	5	65	Bare Land			98.88	
Suspended solids	5	100	Bare Land		166.68		0.000016668
Total P	5	100	Bare Land	64.59			0.06459
Dissolved P	5	100	Bare Land	60.82			0.06082
PP/TP	5	100	Bare Land			94.16	
Suspended solids	10	30	Bare Land		20.65		0.000002065
Total P	10	30	Bare Land	8.94			0.00894
Dissolved P	10	30	Bare Land	8.03			0.00803
PP/TP	10	30	Bare Land			89.82	
Suspended solids	10	50	Bare Land		81.84		0.000008184
Total P	10	50	Bare Land	26.79			0.02679
Dissolved P	10	50	Bare Land	24.72			0.02472
PP/TP	10	50	Bare Land			92.27	

Suspended solids	10	65	Bare Land		108.59		0.000010859
Total P	10	65	Bare Land	25.37			0.02537
Dissolved P	10	65	Bare Land	24.13			0.02413
PP/TP	10	65	Bare Land			95.11	
Suspended solids	10	100	Bare Land		159.32		0.000015932
Total P	10	100	Bare Land	68.1			0.0681
Dissolved P	10	100	Bare Land	64.69			0.06469
PP/TP	10	100	Bare Land			94.99	
Suspended solids	0	30	Tall Grass		0		0
Total P	0	30	Tall Grass	0			0
Dissolved P	0	30	Tall Grass	0			0
PP/TP	0	30	Tall Grass			0.11	
Suspended solids	0	50	Tall Grass		0		0
Total P	0	50	Tall Grass	0			0
Dissolved P	0	50	Tall Grass	0			0
PP/TP	0	50	Tall Grass			0.6	
Suspended solids	0	65	Tall Grass		0		0
Total P	0	65	Tall Grass	0			0
Dissolved P	0	65	Tall Grass	0			0
PP/TP	0	65	Tall Grass			0.01	
Suspended solids	0	100	Tall Grass		0		0

Total P	0	100	Tall Grass	0			0
Dissolved P	0	100	Tall Grass	0			0
PP/TP	0	100	Tall Grass			2.1	
Suspended solids	5	30	Tall Grass		0.04		0.000000004
Total P	5	30	Tall Grass	0.54			0.00054
Dissolved P	5	30	Tall Grass	0.07			0.00007
PP/TP	5	30	Tall Grass			12.44	
Suspended solids	5	50	Tall Grass		0.66		0.000000066
Total P	5	50	Tall Grass	3.34			0.00334
Dissolved P	5	50	Tall Grass	0.15			0.00015
PP/TP	5	50	Tall Grass			4.62	
Suspended solids	5	65	Tall Grass		5.64		0.000000564
Total P	5	65	Tall Grass	9.47			0.00947
Dissolved P	5	65	Tall Grass	1.05			0.00105
PP/TP	5	65	Tall Grass			10.95	
Suspended solids	5	100	Tall Grass		15.44		0.000001544
Total P	5	100	Tall Grass	19.87			0.01987
Dissolved P	5	100	Tall Grass	4.58			0.00458
PP/TP	5	100	Tall Grass			23.06	
Suspended solids	10	30	Tall Grass		0.72		0.000000072
Total P	10	30	Tall Grass	1.05			0.00105
Dissolved P	10	30	Tall Grass	0.16			0.00016

PP/TP	10	30	Tall Grass			15.63	
Suspended solids	10	50	Tall Grass		0.54		0.000000054
Total P	10	50	Tall Grass	5.23			0.00523
Dissolved P	10	50	Tall Grass	0.86			0.00086
PP/TP	10	50	Tall Grass			16.36	
Suspended solids	10	65	Tall Grass		10		0.000001
Total P	10	65	Tall Grass	10.34			0.01034
Dissolved P	10	65	Tall Grass	1.37			0.00137
PP/TP	10	65	Tall Grass			13.26	
Suspended solids	10	100	Tall Grass		22.25		0.000002225
Total P	10	100	Tall Grass	30.69			0.03069
Dissolved P	10	100	Tall Grass	5.2			0.0052
PP/TP	10	100	Tall Grass			16.93	

ST 14: Soil Maximum Saturation Capacity (Tarboton 2003)

Soil Texture	Saturation (cm/hr)
Sand	63.36
Loamy sand	56.16
Sandy loam	12.49
Silt loam	2.59
Loam	2.5
Sandy clay loam	2.27
Silty clay loam	0.612
Clay loam	0.882
Sandy clay	0.781
Silty clay	0.371
Clay	0.461

APPENDIX C: DATA COMPILATION AND ANALYSIS

C.1: Compilation of the AWD/LWD Dataset

The data received from the Auburn Water District/Lewiston Water Division (AWD/LWD) required extensive reformatting prior to analysis in JMP. The same column often included multiple types of data, rows were inconsistently filled, and large amounts of data were missing (Supplementary Figure 1). All Water District data was thus reformatted in Excel to have consistent columns (each column holds only one type of information/data). This process also allowed for the identification of missing data, and several follow-up conversations with Water District officials. All discharge information was converted into metric units.

Bethel Steele compiled the "parameter" data for all years: This included all of the parameters (i.e.: total P, conductivity, pH, etc.) collected at AWD/LWD sampling sites since 2005. This compilation was done in R.

Following compilation, the data were analyzed in JMP. Below is a description of the data manipulation done to create each map and figure (where applicable). Descriptions for the SWAT maps are in the methods section, as the creation of these maps did not require data compilation for individual map creation; general SWAT data compilation information follows the descriptions of individual maps.

	A	B	C	D	E	F	G	H	I	J	K	L	M
1													
2	LAKE AUBURN WATERSHED GRAB SAMPLES STREAM FLOWS												
3	SAMPLED BY:	DAF,MEV	Model: Fp211 Flow Probe; Global Water SN 1304000539								actual	actual	thoeretical
4	ROUND PIPES								A(SQFT)		CFS	GPM	GPD
5	DATE	TIME	GPS #	SITE NAME	H(FT)	D(FT)	H/D	constant	C*D2	AVE VEL	G*H	I*448.83	K*1440
6	5/7/14	720	N44 08.527 W070 13.537	OUTLET	0.8	10	0.08	0.0294	2.94	7.7	22.638	10,148.62	14,614,006.18
7		755	N44 09.969 W070 14.244	*TOT LOT LEFT	0.6	2	0.3	0.1982	0.7928	4	3.1712	1,421.65	2,047,174.50
8				*TOT LOT CENTER	0.6	2	0.3	0.1982	0.7928	3.3	2.61624	1,172.86	1,688,918.96
9				*TOT LOT RIGHT	0.7	4	0.175	0.0923	1.4768	0.4	0.59072	264.82	381,340.48
10		810	N44 09.391 W070 14.115	SITE 25	0.4	2	0.2	0.1118	0.4472	0.4	0.17888	80.19	115,476.34
11		825	N44 09.562 W070 14.554	SITE 2	2.6	4	0.65	0.5404	8.6464	0.5	4.3232	1,938.09	2,790,850.41
12		830	N44 09.663 W070 15.311	SITE 3	0.3	6	0.05	0.0147	0.5292	4.6	2.43432	1,091.31	1,571,480.14
13		855	N44 10.705 W070 16.360	*SITE 27 LEFT	0.1	3	0.033	0.0069	0.0621	6.3	0.39123	175.39	252,559.31
14				*SITE 27 RIGHT	0.1	3	0.033	0.0069	0.0621	6.3	0.39123	175.39	252,559.31
15		900	N44 12.742 W070 17.509	SITE 18	2.2	4	0.55	0.4426	7.0816	1.2	8.49792	3,809.62	5,485,849.25
16		910	N44 11.766 W070 17.215	*SITE 17 LEFT	2.1	6	0.35	0.245	8.82	1.1	9.702	4,349.41	6,263,145.50
17				*SITE 17 RIGHT	1.1	6	0.183	0.0961	3.4596	1.3	4.49748	2,016.22	2,903,357.21
18		930	N44 10.151 W070 16.613	SITE 5A	0.2	2	0.1	0.0409	0.1636	0.05	0.00818	3.67	5,280.62
19	*STANDING ON CULVERT TAKING READINGS												
20	SQUARE CULVERT												
21													
22				SITE NAME	H(FT)	W(FT)	AREA			AVG VEL			
23		835	N44 09.857 W070 15.769	SITE 4	0.2	2.83	0.566			0.2	0.1132	50.74756	73,076.49
24		840	N44 10.539 W070 16.480	SITE 13	4.75	20	95			0.2	19	8517.7	12,265,488.00
25													
26													
27	STREAMS DIRECTLY INTO LAKE												

SF 1: Example of water district file format. Most files continued for hundreds of cells both down and to the right in this format.

C.2: Data Manipulations for In-Text Maps and Figures

C.2.1: Data Manipulations for In-Text Maps

Map 3 (Long-term sampling location sub-watersheds. A pour point was put in the location of each long-term sampling location in order to delineate the sub-watershed area which drains through each sampling location): A sub-watershed pour point was placed in the location of each long-term sampling location in order to delineate the sub-watershed area which drains through each sampling location. Sampling locations were moved up to 15m (perpendicular to the stream to preserve stream position and not move the sampling point up or downstream) to be placed on the nearest possible location with high flow accumulation. This is within the margin of error of the GPS units used. A land cover layer was extracted by sub-watershed, and the area of each land cover class, in meters, was calculated. Once delineated, a land cover Excel file was created with the area of each sub-watershed in each land cover class (in meters). The percentage of each land cover class in each sub-watershed was also calculated. Consolidated land cover classes were also calculated: Developed (a composite of Developed, open space; Developed, low intensity; Developed, medium intensity; Developed, high intensity; and Barren industrial land); Agricultural (Hay and pasture; and Cultivated crops); and Undeveloped (Deciduous forest; Evergreen forest; Mixed forest; Shrub/scrub; Herbaceous; Woody wetlands; and Emergent herbaceous wetlands). Open water was excluded from the land cover analysis. For the two larger sub-watersheds, namely Townsend Brook and the Basin, a sub-watershed was created at each long-term sampling location, thus delineating those sub-watersheds into multiple smaller sub-watersheds. This was done to better understand nuance within these sub-watersheds.

Map 5 (Long-term average P concentration minimums, means, and ranges (2005-2019). Minimum is lowest value across all years; mean is mean of each yearly mean; range is range across all years): The data for the sites with a long-term record were summarized in JMP twice: First, a yearly total P concentration minimum, mean, and range was taken by site, followed by a summary of yearly values across all years. The result was the minimum value recorded across all years, the mean across all years, and the range across all years with each year weighted equally regardless of the number of sampling events in each year. This file was then joined to long-term site layer by site in GIS. Symbols are exaggerated by 25% with each layer (i.e.: a TP range of 20 is symbolized as 25% larger than a TP mean of 20, which is 25% larger than a minimum TP of 20). This was done to make the map more readable. When data for a site was missing for a given year, that year was not included in the calculation of mean.

Map 6 (Mean monthly P concentrations at long-term sites during the sampling season (April-October)): The data for the sites with a long-term record were summarized in JMP twice: First, a mean total P concentration value for each month in each year was calculated by site, followed by a summary of mean monthly values across years. The result was such that the mean value for each year was weighted equally regardless of the number of sampling events in each month and year. This file was then joined as a new field to a "sites with long-term data" layer in GIS and symbolized.

Map 7 (Phosphorus concentrations at all sites sampled in 2019. Minimum is the minimum value recorded at each site; mean is the average of all P concentration data recorded at each site; and maximum is the highest value recorded at each site): The parameters file was summarized in JMP to determine minimum, mean, and maximum total P concentration value by site for 2019. This file was then joined to a combined long-term and new site layer in GIS and symbolized.

Map 8a (Estimated stream P load at long-term sampling locations, using yearly mean load estimate): Phosphorus load was estimated for sites where the Auburn Water District and Lewiston Water Division (AWD/LWD) collected both discharge data and P concentration data. The ice-free period was defined as April 1st to December 31, and load was calculated for each site in which concurrent discharge and total P data had been collected at least once in 2019. Ten sites had this data for 2019, and they were all sampled between six and 10 times. Using the ice-out (91) and ice-in (365) Julian days, the total number of ice-free, loading days was estimated as 274. All calculations were done in Excel, then joined to the long-term site layer by site in GIS and symbolized. For the year-round calculation, the earliest sampling event of the year was extended to January 1, 2019, and the latest sampling event of the year was extended to December 31, 2019, bringing the total number of days to 365 (see Appendix B, ST 2 for full load calculation information and ST 10 for load estimates).

Map 8b (Estimated stream P load at long-term sampling locations, using distributed load estimate): Phosphorus load was estimated for sites where the Auburn Water District and Lewiston Water Division (AWD/LWD) collected both discharge data and P concentration data. The ice-free period was defined as April 1st to December 31, and load was calculated for each site in which concurrent discharge and total P data had been collected at least once in 2019. Ten sites had this data for 2019, and they were all sampled between six and 10 times. Using the ice-out (91) and ice-in (365) Julian days, the total number of ice-free, loading days was estimated as 274. Then, using the Julian days of each sampling event, the midpoint Julian day between sampling events was identified. The values for each sampling event were then applied to the days between the midpoint of the previous sampling event and the midpoint of the next sampling event. Site 3,for example, the total P value of 9ug/L recorded on June 25th (Julian day 176) and applied to Julian days 170 through 198. This is because the previous and next sampling events were on June 13th (Julian day 164) and August 8th (Julian day 220), respectively. Thus, 170 is the midpoint between 164 and 176, and 198 is the midpoint between 176 and 220 (see Appendix B, ST 3 for full calculations and Appendix B, ST 10 for total load estimates). Once the midpoints were identified, the number of days for which the total P value applied was calculated. Next, the total P concentration value (in ug/L) was multiplied by 10^9 to get total P concentration in kilograms per liter (kg/L). Then, this value was multiplied by 1000 to get total P concentration in kilograms per cubic meter (kg/m^3). The discharge value (in m^3/sec) was then multiplied by the total P concentration value (in kg/m^3) to get the amount of P discharged per second. The per-second load was then multiplied by 60 to get load per minute, 60 again to get load per hour, and 24 to get daily load. The daily load value was then multiplied by the number of days for which that value was to be used (based on the nearest midpoints), giving the sectional total P value for the days surrounding each sampling event. These sectional values were then summed by site to calculate to total P load for the 274 ice-free Julian days in 2019 (Appendix B, ST 10). All calculations were done in Excel, then joined to the long-term site layer by site in GIS and symbolized. For the year-round calculation, the earliest sampling event of the year was extended to January 1, 2019, and the latest sampling event of the year was extended to December 31, 2019, bringing the total number of days to 365.

The creation of all SWAT maps is contained within the SWAT+ extension for QGIS. Thus, no data manipulations were needed and all procedures are contained in the methods and/or appendices. Initial orientation maps (Maps 1, 2, and 4) also required no data manipulations.

C.2.2: Data Manipulations for In-Text Figures

Figure 1a (Mean total P concentration by site and year at sites in the Basin drainage system): The data for the sites with a long-term record were summarized by yearly mean and site and coded by major drainage system (Townsend Brook, the Basin, and direct drainage to the lake (not part of the Basin or Townsend Brook sub-watershed)). Each was made into a separate subset. The data for the sites in the Basin sub-watershed were graphed by site. X input: Year; Y input: Total P concentration. One value of 430ug/L for Site 13 on March 19th, 2014 was removed because it is implausibly high. Dan Fortin at the AWD/LWD who likely collected the data believes this value was recorded in error.

Figure 1b (Mean total P concentration by site and year at sites in the Townsend Brook drainage system): The data for the sites in the Townsend Brook sub-watershed (see description of Figure 1a) were graphed by site. X input: Year; Y input: Total P concentration.

Figure 1c (Mean total P concentration by site and year at non-drainage system sites): The data for the sites not belonging in the Townsend Brook and Basin sub-watersheds (see description of Figure 1a) were graphed by site. X input: Year; Y input: Total P concentration.

Figure 2 (Estimated annual P load from regularly-sampled major streams in the Lake Auburn watershed under four load estimation methods): This figure is a different symbolization of the data analyses done for Maps 8a and 8b).

Figure 3a (Undeveloped land and P concentrations in long-term sampling location sub-watersheds): The data for the sites with a long-term record were merged with the land cover file by site (see description of Map 2 for sub-watershed delineation protocol). In JMP, a fit y by x scatterplot was created. X input: Percent undeveloped land; Y input: Long-term mean P concentration (mean of yearly means for 2005-2019). A line was fitted due to the graph's statistical significance. R^2: 0.33; p-value: 0.032.

Figure 3b (Developed land and P concentrations in long-term sampling location sub-watersheds): The same protocol as for Figure 3a was followed except with percent developed land as the X input. No line was added because the relationships is not statistically significant. R^2: 0.14; p-value: 0.19.

Figure 3c (Agricultural land and P concentrations in long-term sampling location sub-watersheds): The same protocol as for Figure 3a was followed except with percent agricultural land as the X input. No line was added because the relationship is not statistically significant. R^2: 0.23; p-value: 0.08. The same graphs were also made for each individual land cover class in each sub-watersheds. Because relationships between individual classes were weaker, the summary land cover classes are used for the purposes of this analysis.

Figure 4 (Predicted total P load in the watershed under the six SWAT scenarios): SWAT load estimates were graphed in Excel.

Figure 5a (Daily high temperature in Lewiston-Auburn in 1980 and 2019): SWAT temperature data was graphed in Excel.

Figure 5b (Daily precipitation in Lewiston-Auburn in 1980 and 2019): SWAT precipitation data was graphed in Excel.

C.3: Data Manipulations for Appendix Maps and Figures

C.3.1: Data Manipulations for Appendix Maps

Map 7 (Spring P concentrations (April, May, and June) at long-term sampling locations): The data for the sites with a long-term record were summarized in JMP twice: First, a mean total P concentration value for each month by site was taken, then a mean of all years by site. This was done to give equal weight to each year of data regardless of the number of sampling events in that year. Symbol sizes were exaggerated by 25% per parameter for enhanced readability: for example, if the minimum total P concentration at a given site was 10µg/L, it would be symbolized using a 10 point circle; a mean total P concentration value of 10µg/L would therefore be symbolized using a 12.5 point circle, and the maximum would use a 15 point circle. This file was then joined as a new field to the "sites with long-term data" layer in GIS and symbolized.

C.3.2: Data Manipulations for Appendix Figures

SF 2a-SF 2d: The Townsend Brook discharge file was graphed in JMP using fit y by x by year (data for 2013-2016). X input: Water depth (m); Y input: Discharge (m^3/sec).

SF 3: All Townsend Brook depth and discharge data was graphed in an overlay plot. X input: Date; Y inputs: Water depth (m) and Discharge (m^3/sec).

Figure 1 (Overlay plot of P concentration by date for all water district data collected since 2005): The parameters file was graphed as an overlay plot in JMP. X input: Date; Y input: Total P concentration. These are the raw values collected by AWD/LWD across all sites and all sampling events with nothing excluded.

Figures 2a-2m (mean total P concentration by year at all sites with long-term data, symbolized as needle plots): The data for the sites with a long-term record were summarized as a yearly mean by site. Needle plots were then created by site. X input: Year; Y input: Mean total P concentration.

Figure 6a: (Discharge and total P concentration in the Basin (all times in which both discharge and concentration data were collected since 2005)): All discharge and total P concentration data from the Basin outlet (Site 13) was merged. In JMP, a fit y by x scatterplot was created. X input: Discharge; Y input: Total P concentration. R^2: 0.0000783.

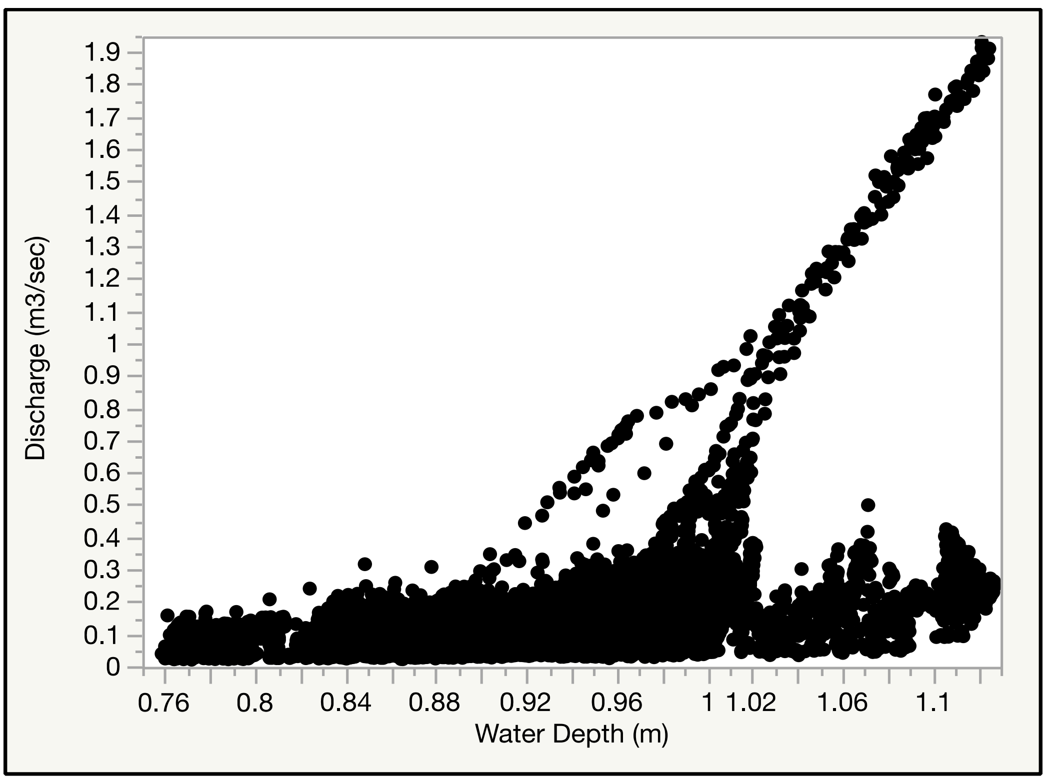
Figure 6b: Discharge and total P concentration in the Basin (all times in which both discharge and concentration data were collected since 2005 with two outliers excluded: 4/1/2014 and 4/29/2019). The two outliers were excluded from the plot created under 6a. R^2: 0.0000783.

Figure 6c: (Discharge and total P concentration in Townsend Brook (all times in which both discharge and concentration data were collected since 2005)): All discharge and total P concentration from the Townsend Brook (Site 2) was merged. In JMP, a fit y by x scatterplot was created. X input: Discharge; Y input: Total P concentration. R^2: 0.0841.

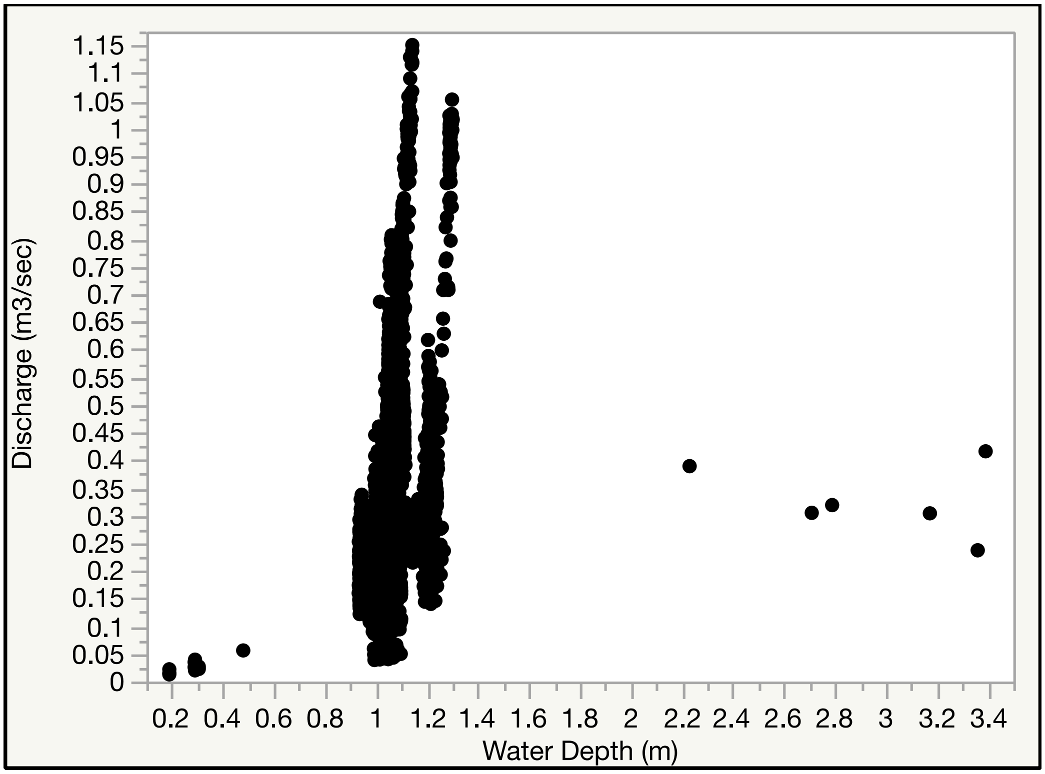
Figure 6d: Discharge and total P concentration in Townsend Brook (all times in which both discharge and concentration data were collected since 2005 with four outliers excluded: 3/19/2014, 8/14/2014 11:15am, 8/14/2014 5:30pm, and 9/30/2015). The four outliers were excluded from the plot created under 6c. Dan Fortin at the AWD/LWD who likely collected the data believes that the 200ug/L concentration value was recorded in error. A line was fitted due to the graph's statistical significance. R^2: 0.3705.

APPENDIX D: INITIAL PROJECT PLANS AND CHOICES OF METHODS

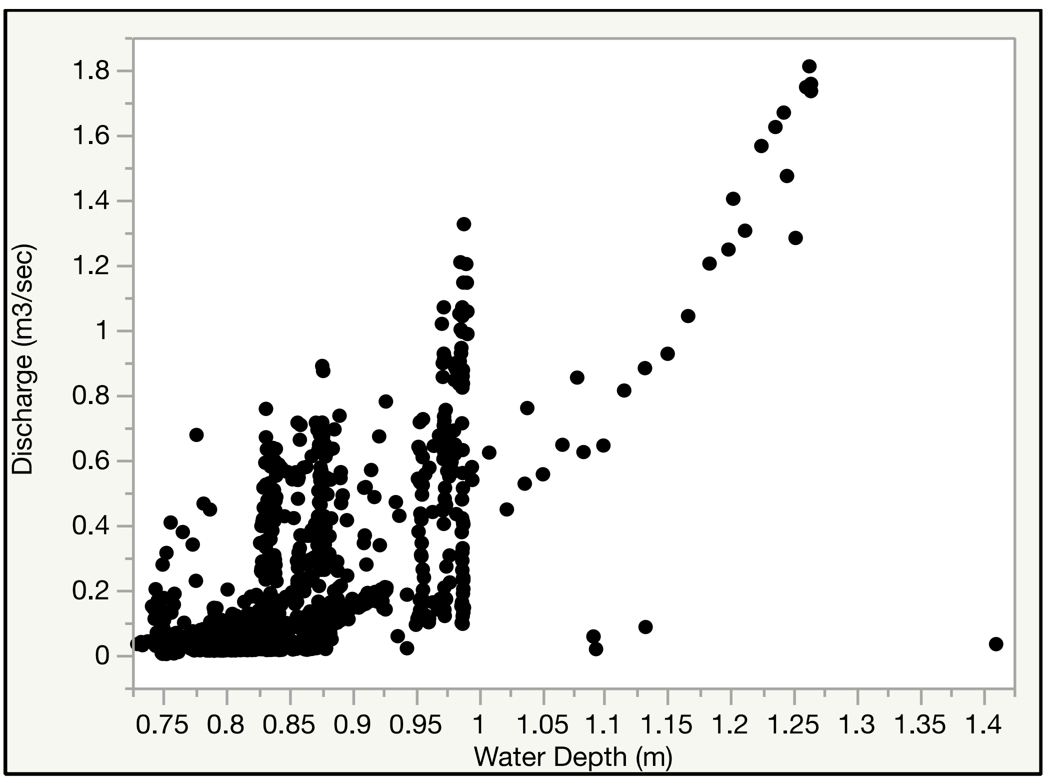
The initial plan for this project relied far more heavily on AWD/LWD data than the final iteration. The idea was to use AWD/LWD on discharge and P concentration, particularly at the Townsend Brook and Basin outlets (where most streamflow enters) to estimate P loading. There are flow meters at the Townsend Brook and Basin outlets which measure water depth in the outlet culverts; we initially thought that there would be a relationship between the culvert depth data, which is collected at 15-minute intervals, and discharge measured periodically (usually 5-15 times per year) by AWD/LWD staff. Had this relationship existed, we could have used the existing discharge data to estimate continuous discharge. There is no strong relationship between culvert depth and discharge (SF 2a-SF 2d). Furthermore, there are large gaps in the data (SF 3), which would have necessitated excluding entire years and parts of most years. Thus, this approach was rapidly abandoned. These example figures help justify why this approach was not pursued further.



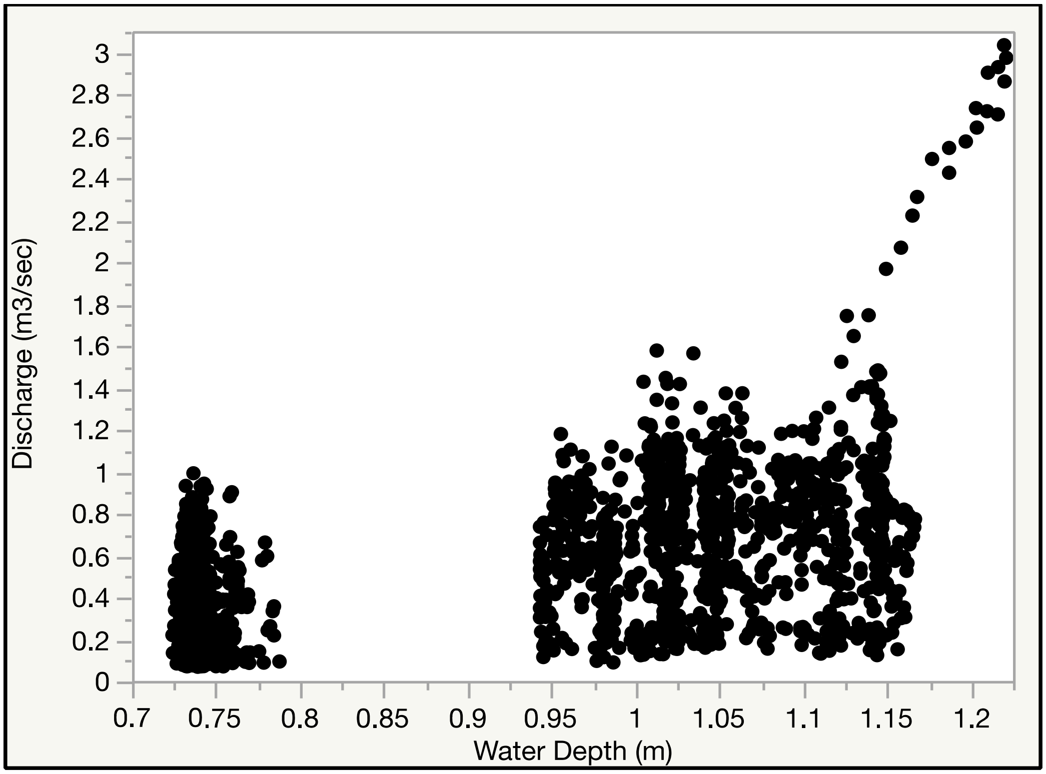
SF 2a: Relationships between outlet culvert water depth and discharge at Site 2 (Townsend Brook outlet) in 2013.



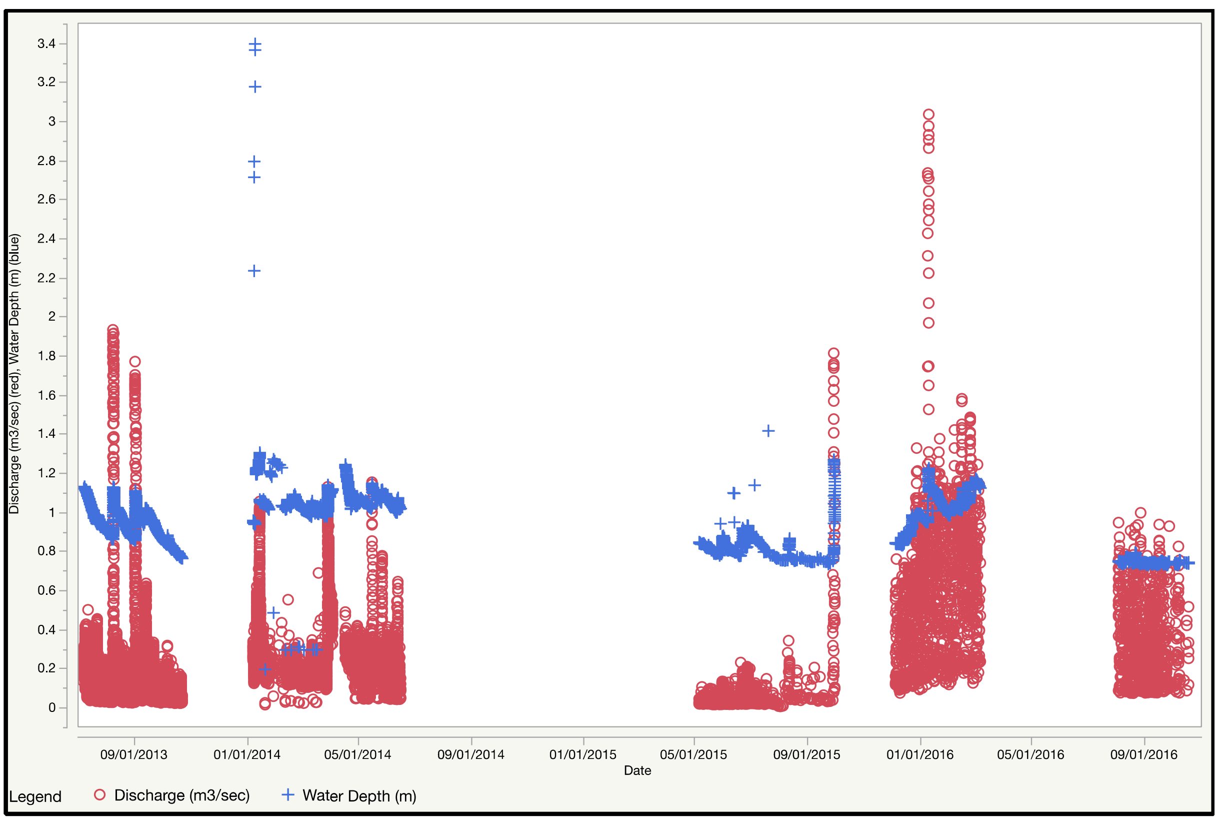
SF 2b: Relationships between outlet culvert water depth and discharge at Site 2 (Townsend Brook outlet) in 2014.



SF 2c: Relationships between outlet culvert water depth and discharge at Site 2 (Townsend Brook outlet) in 2015.

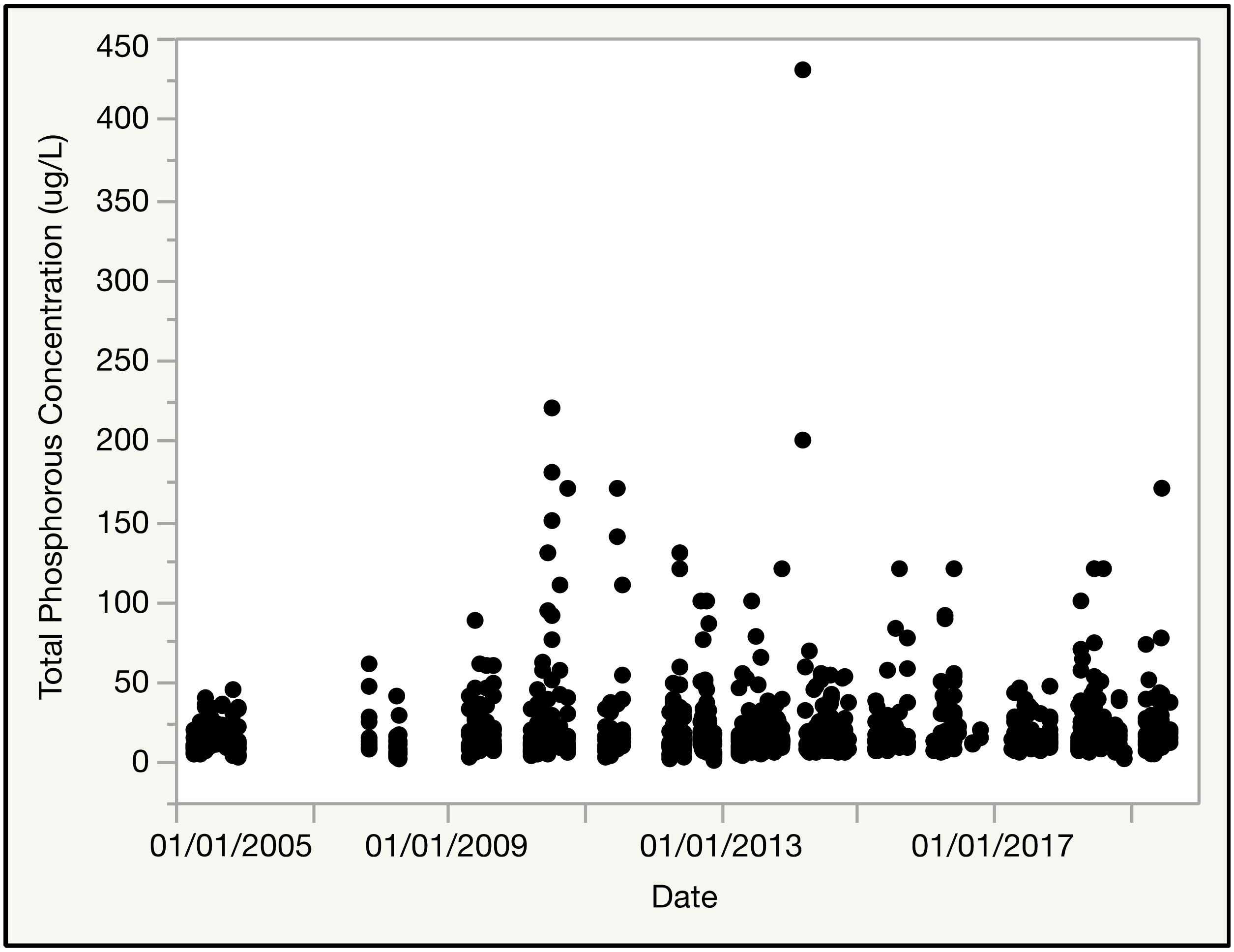


SF 2d: Relationships between outlet culvert water depth and discharge at Site 2 (Townsend Brook outlet) in 2016.

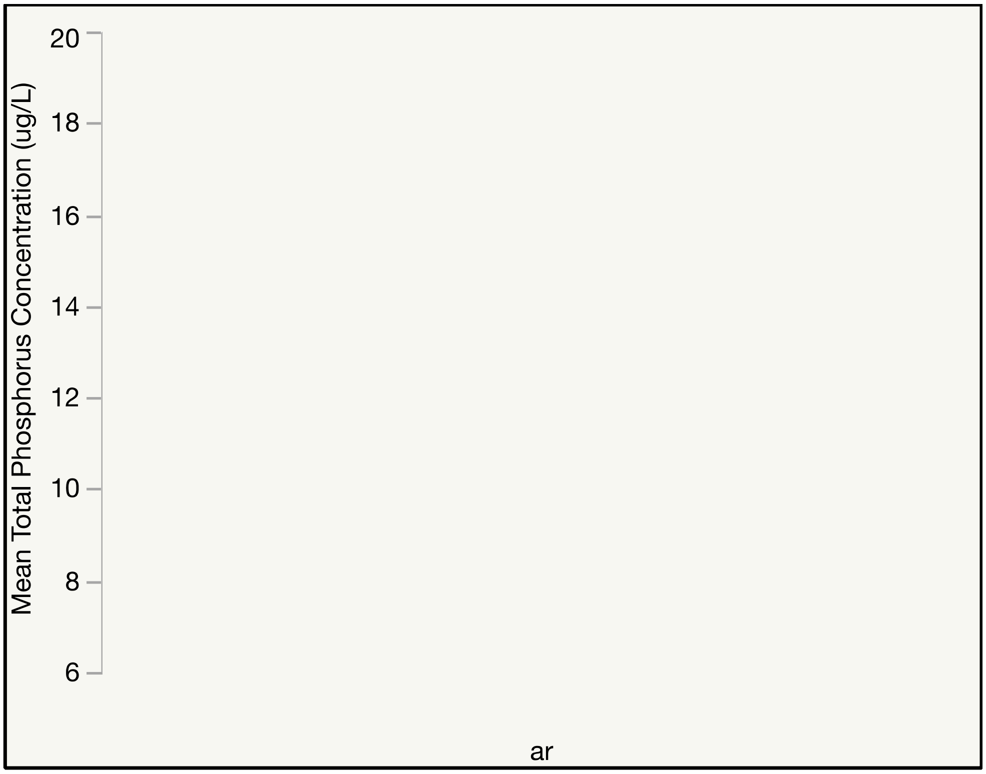


SF 3: All recorded discharge and water depth data at Site 2 (Townsend Brook outlet).

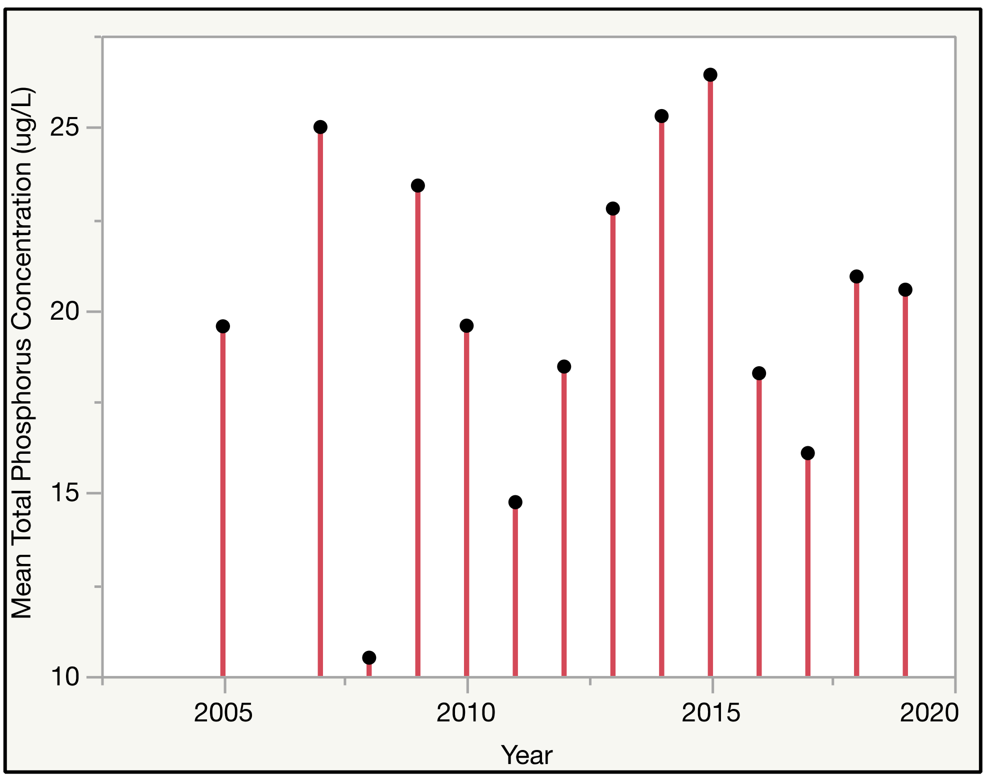
APPENDIX E: SUPPLEMENTARY FIGURES



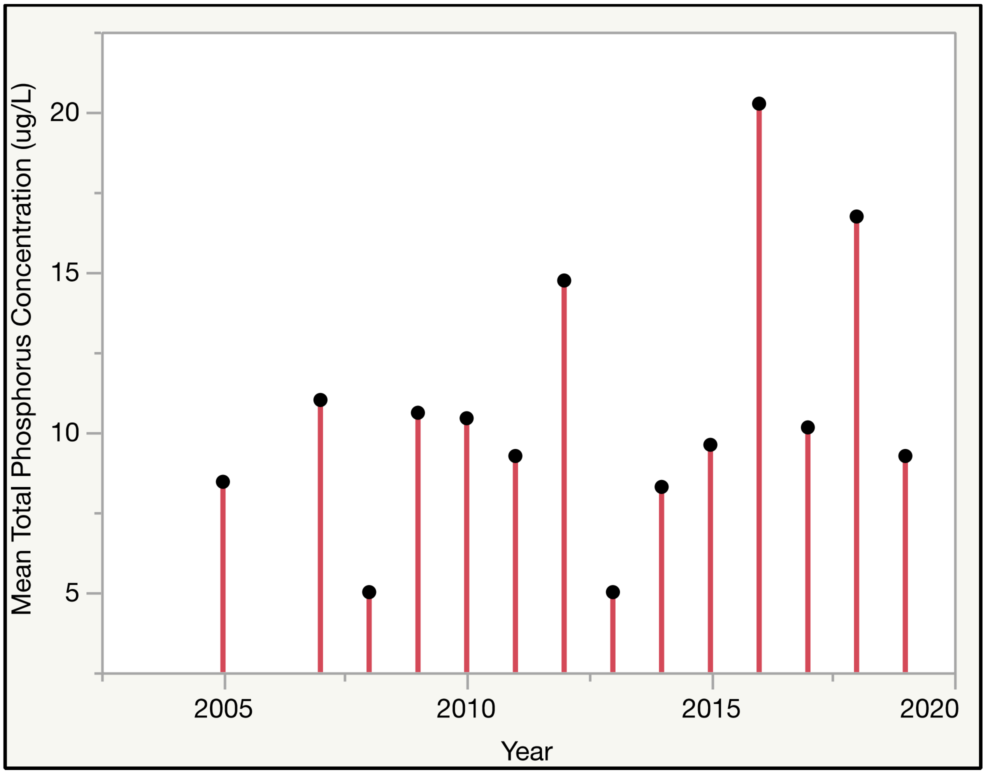
SF 4: Overlay plot of P concentration by date for all water district data collected between 2005 and 2019.



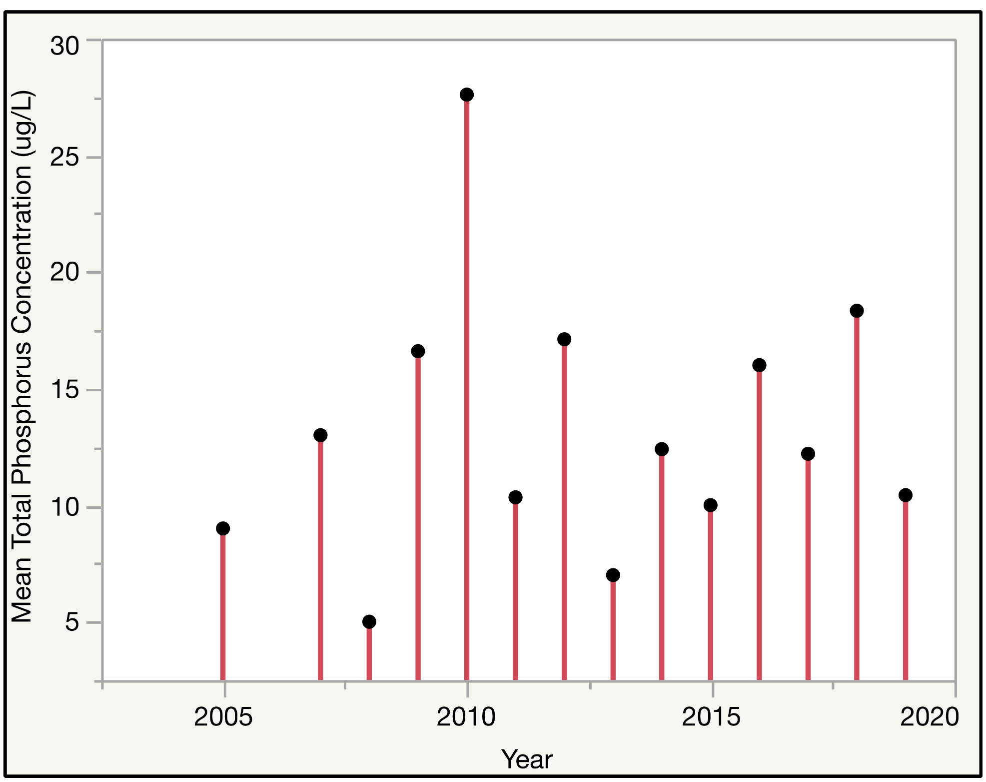
SF 5a: Mean total P concentration by year at Site 1 (Lake Auburn outlet).



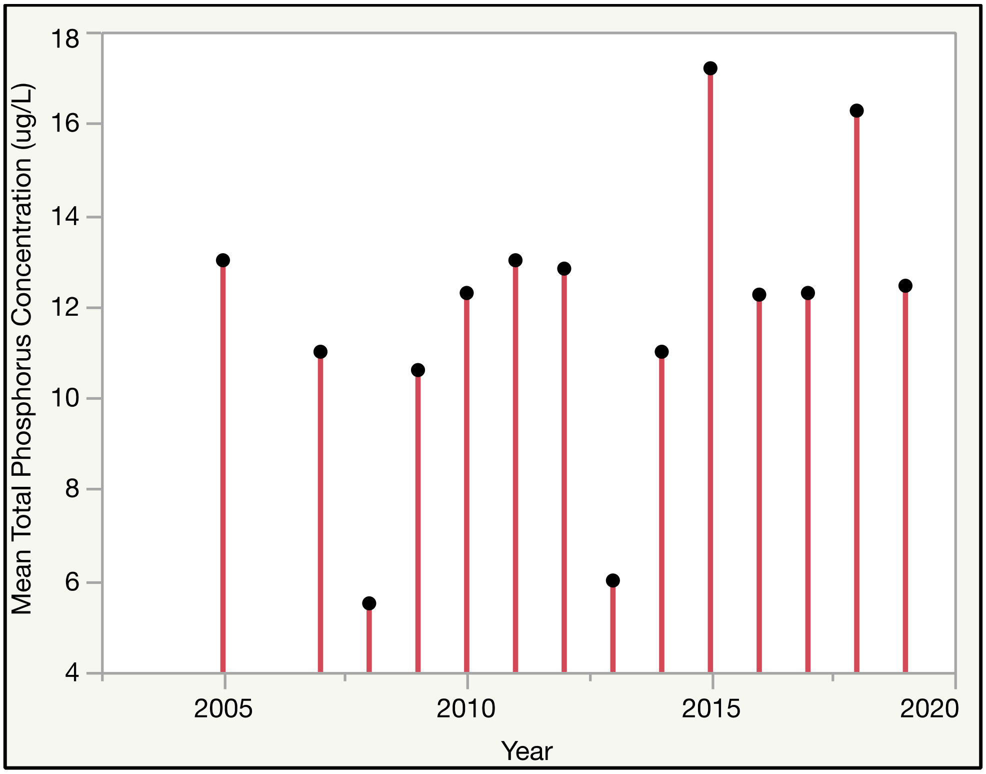
SF 5b: Mean total P concentration by year at Site 2 (Townsend Brook outlet).



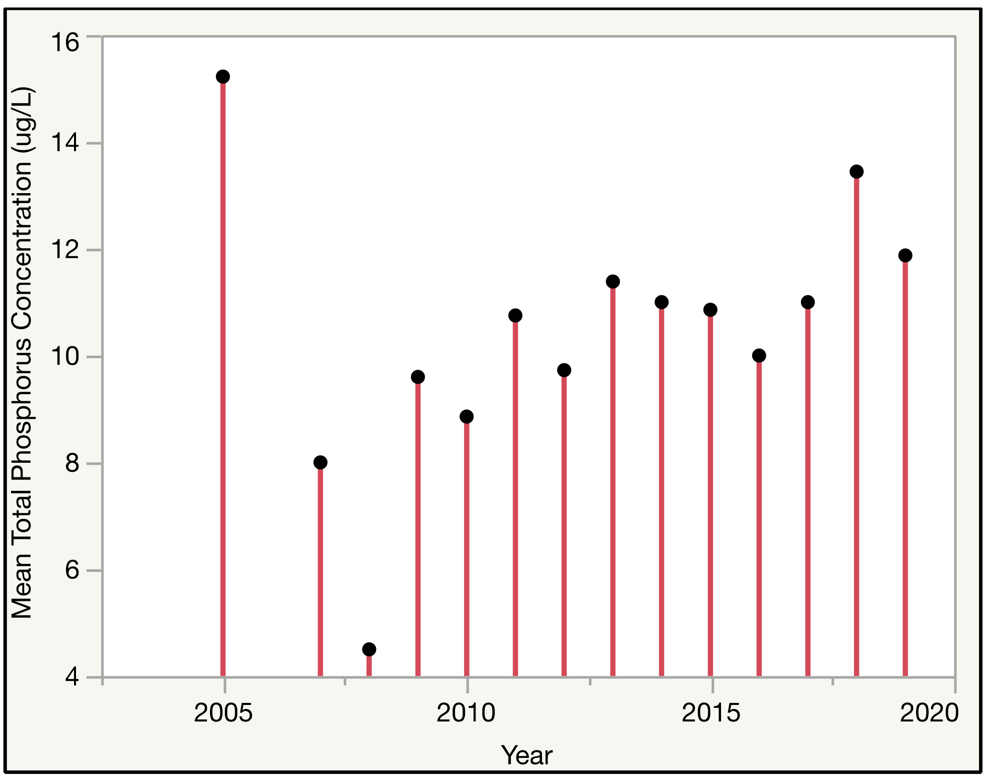
SF 5c: Mean total P concentration by year at Site 3 (Taber's Driving Range).



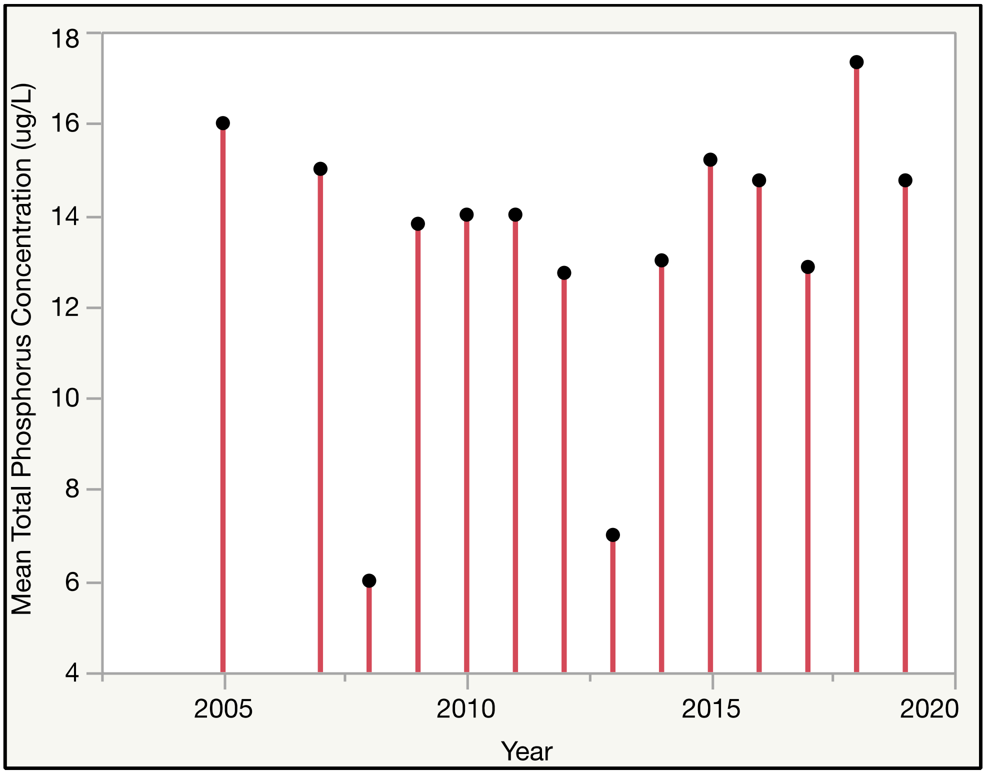
SF 5d: Mean total P concentration by year at Site 4 (northwest shore).



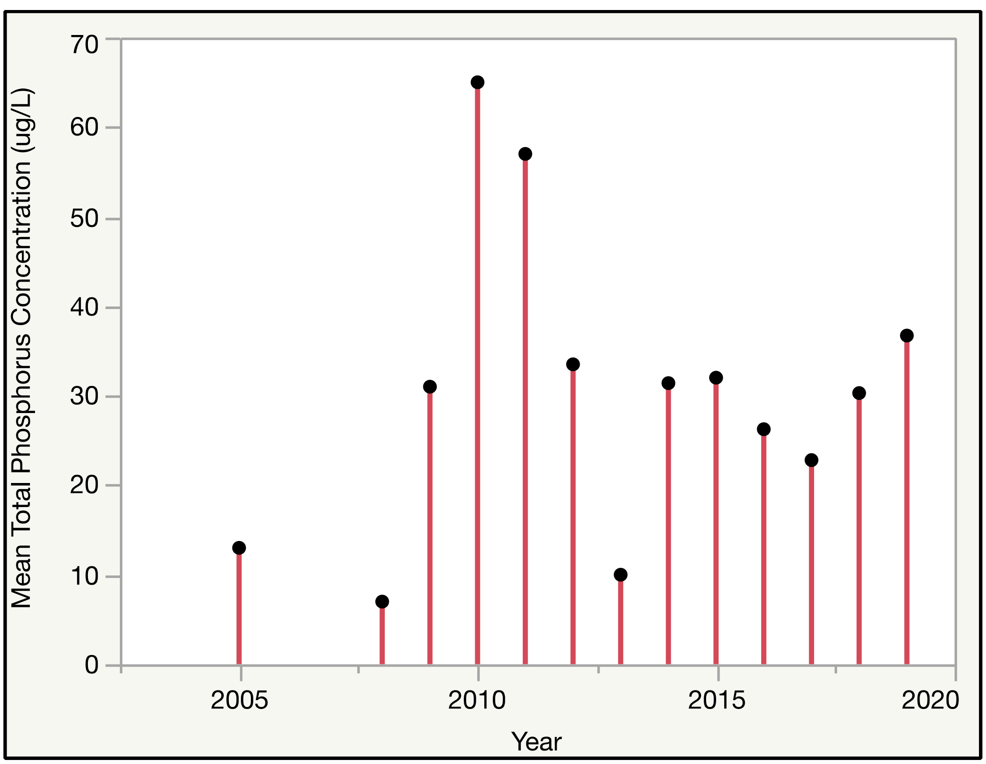
SF 5e: Mean total P concentration by year at Site 13 (Basin outlet).



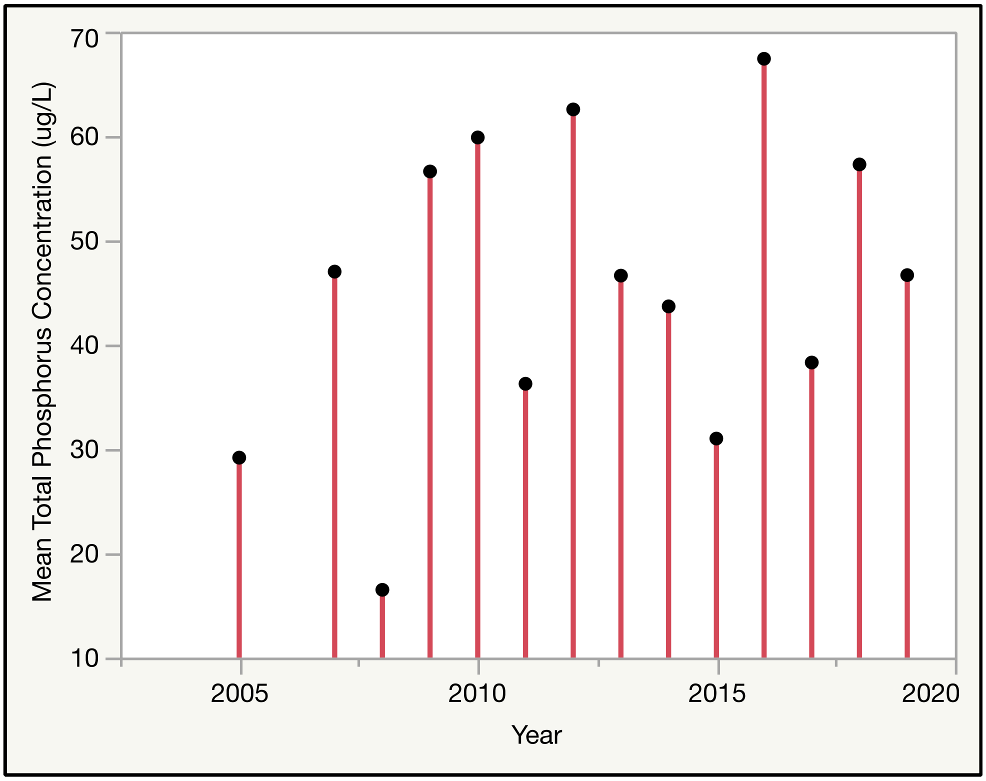
SF 5f: Mean total P concentration by year at Site 16 (Basin Dam).



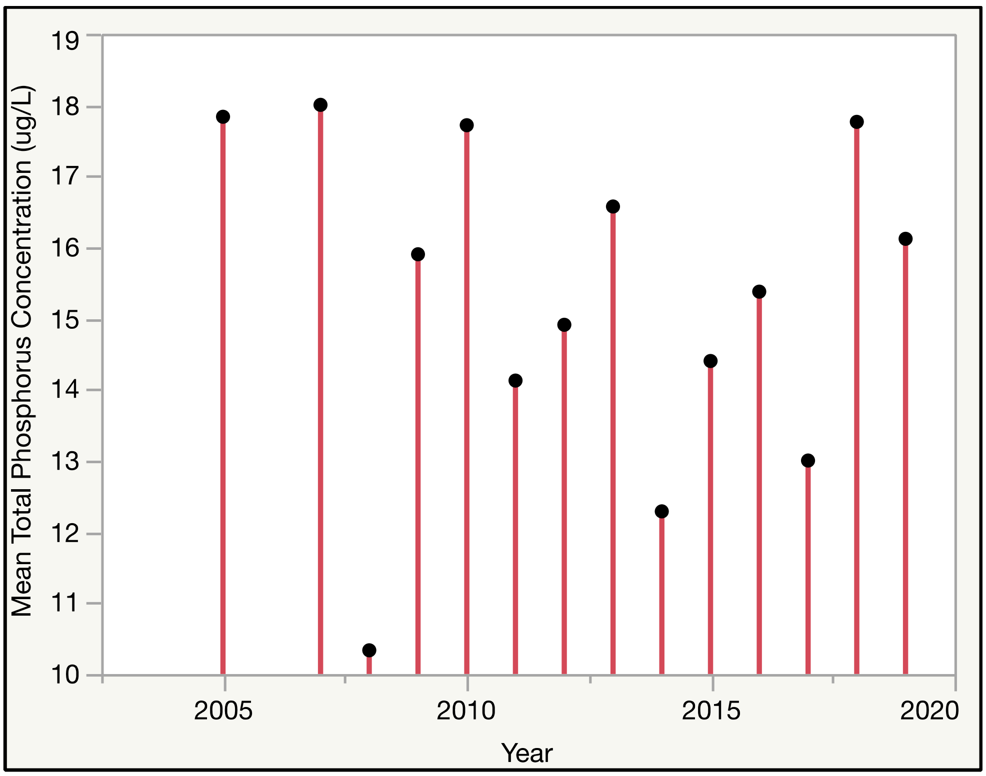
SF 5g: Mean total P concentration by year at Site 18 (Mud Pond outlet).



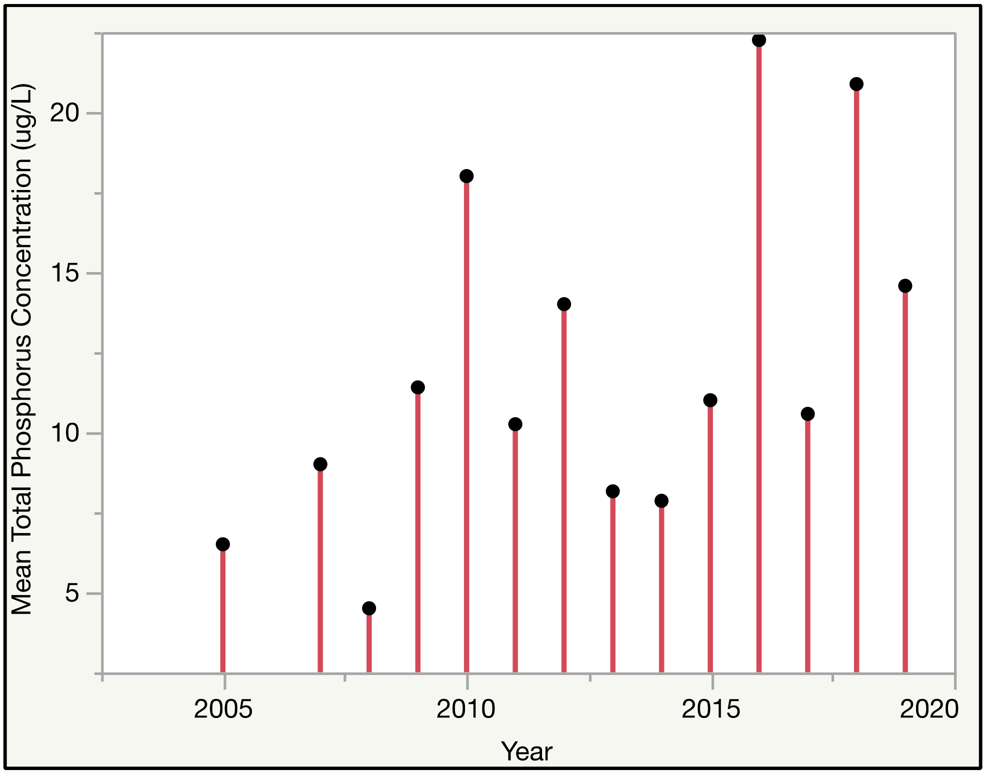
SF 5h: Mean total P concentration by year at Site 23 (Horse Pond/north shore).



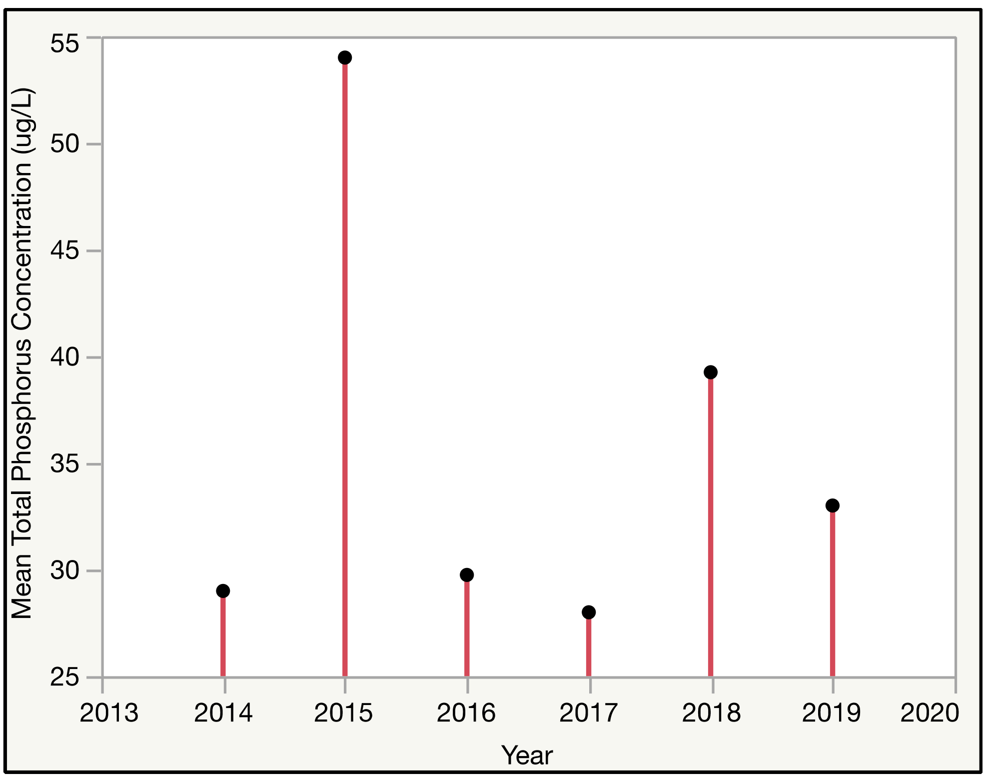
SF 5i: Mean total P concentration by year at Site 25 (First Brook).



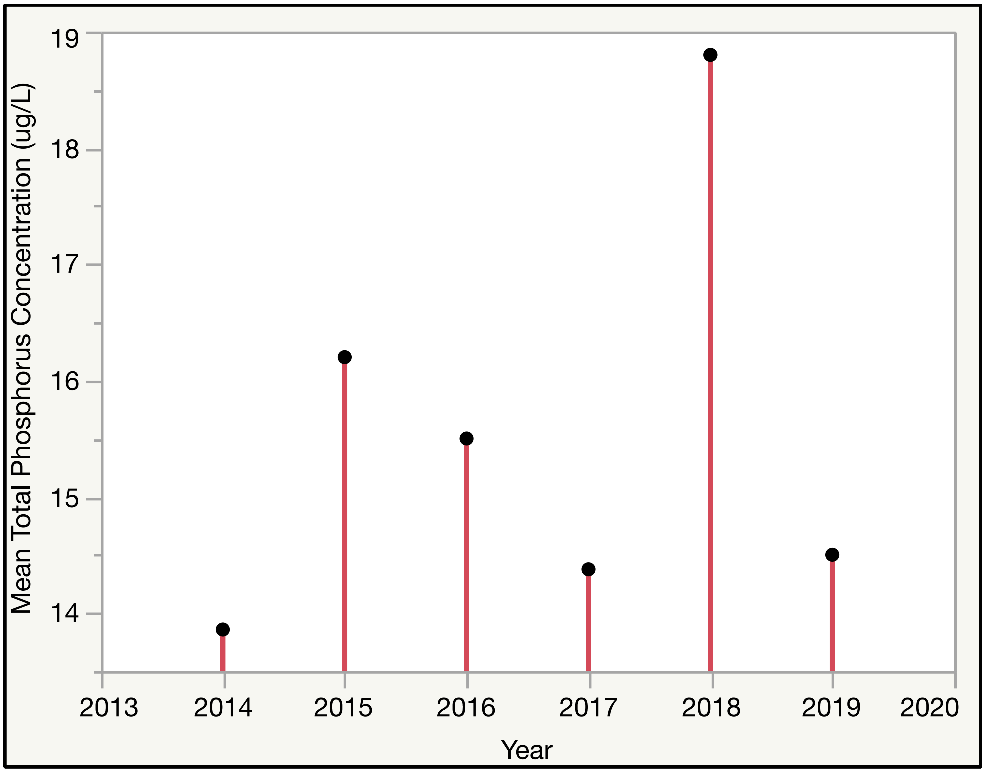
SF 5j: Mean total P concentration by year at Site 26 (Tot Lot/Townsend Brook).



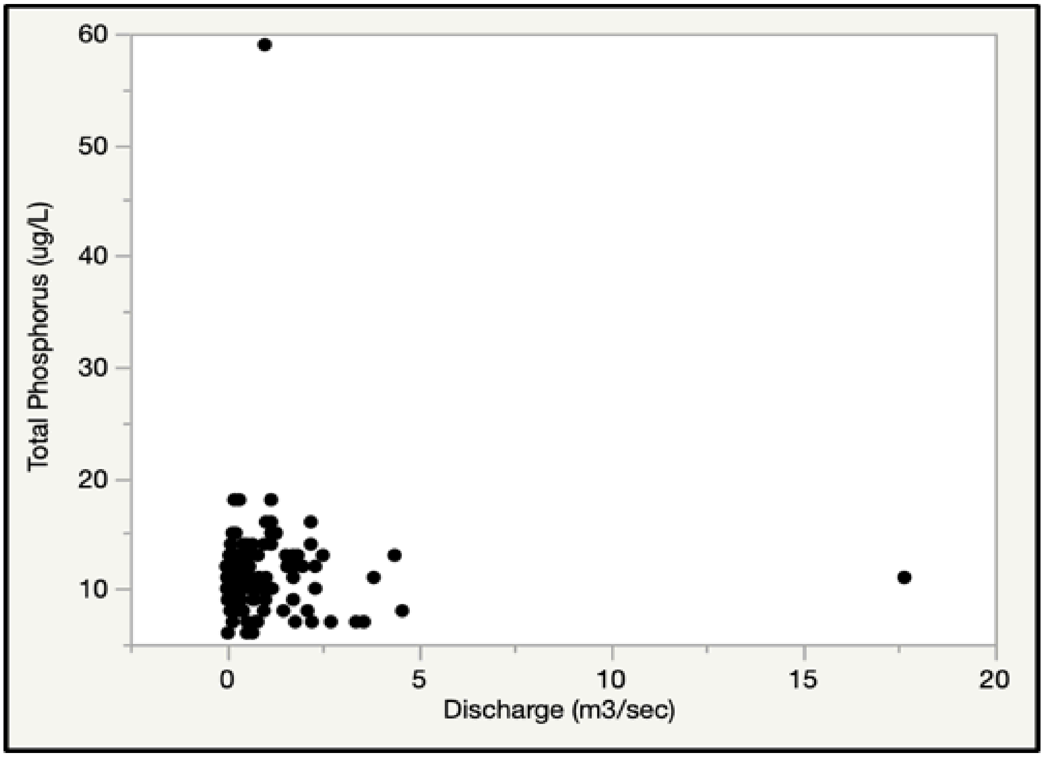
SF 5k: Mean total P concentration by year at Site 27 (Johnson Road).



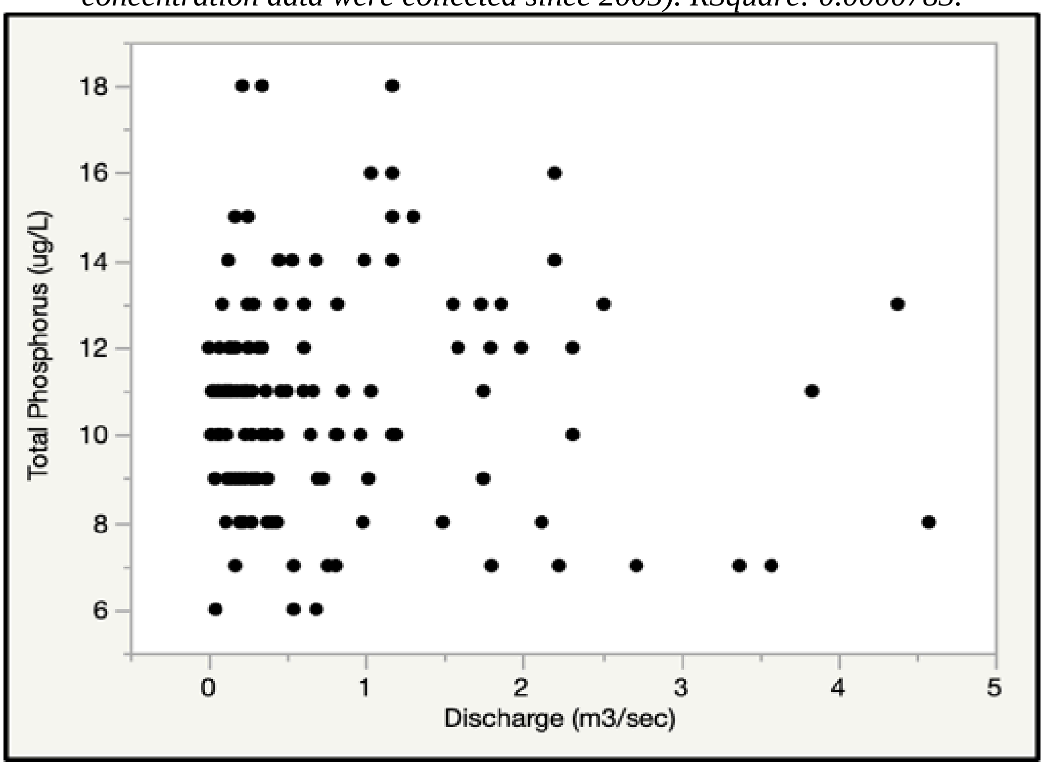
SF 5l: Mean total P concentration by year at Site Roys (Roy's Golf Course/Townsend Brook).



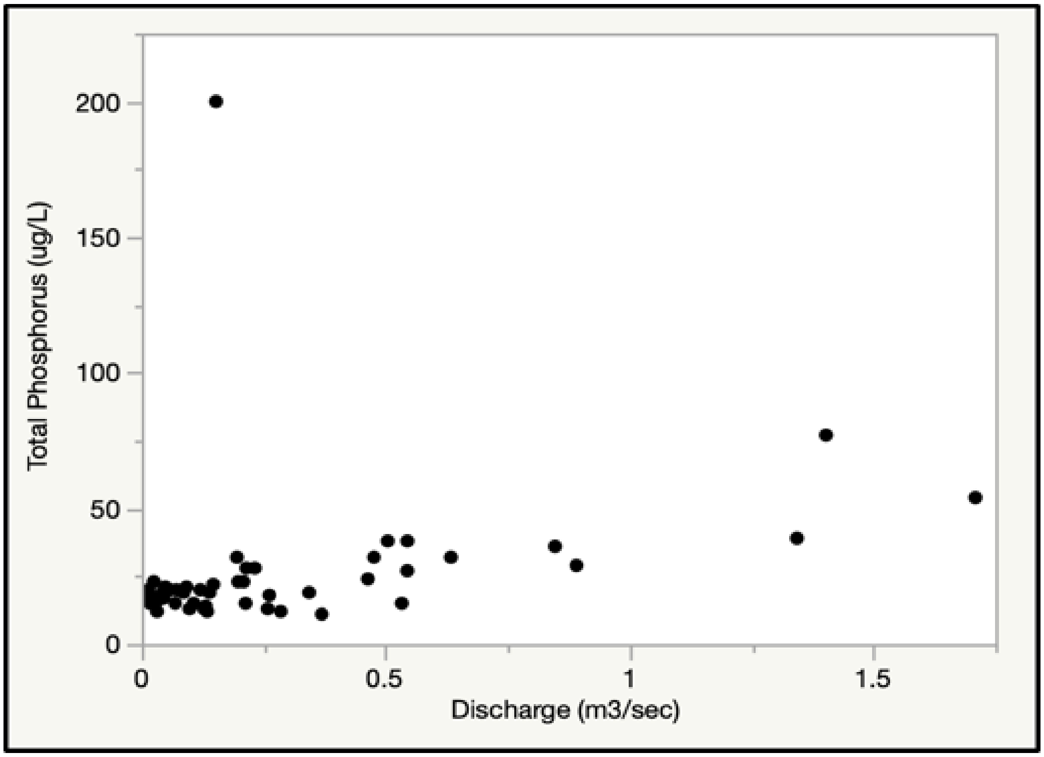
SF 5m: Mean total P concentration by year at Site TBR (Townsend Brook Road/Townsend Brook).



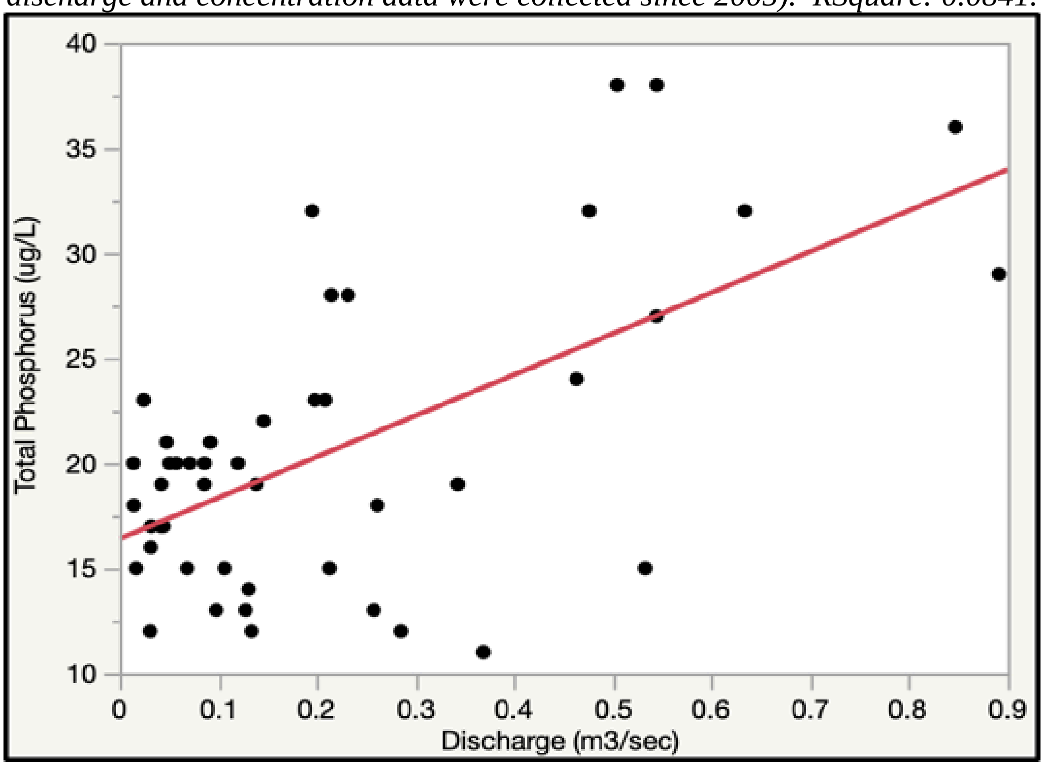
SF 6a: Discharge and total P concentration in the Basin (all times in which both discharge and concentration data were collected since 2005). RSquare: 0.0000783.



SF 6b: Discharge and total P concentration in the Basin (all times in which both discharge and concentration data were collected since 2005 with two outliers excluded: 4/1/2014 and 4/29/2019). RSquare: 0.0000783.

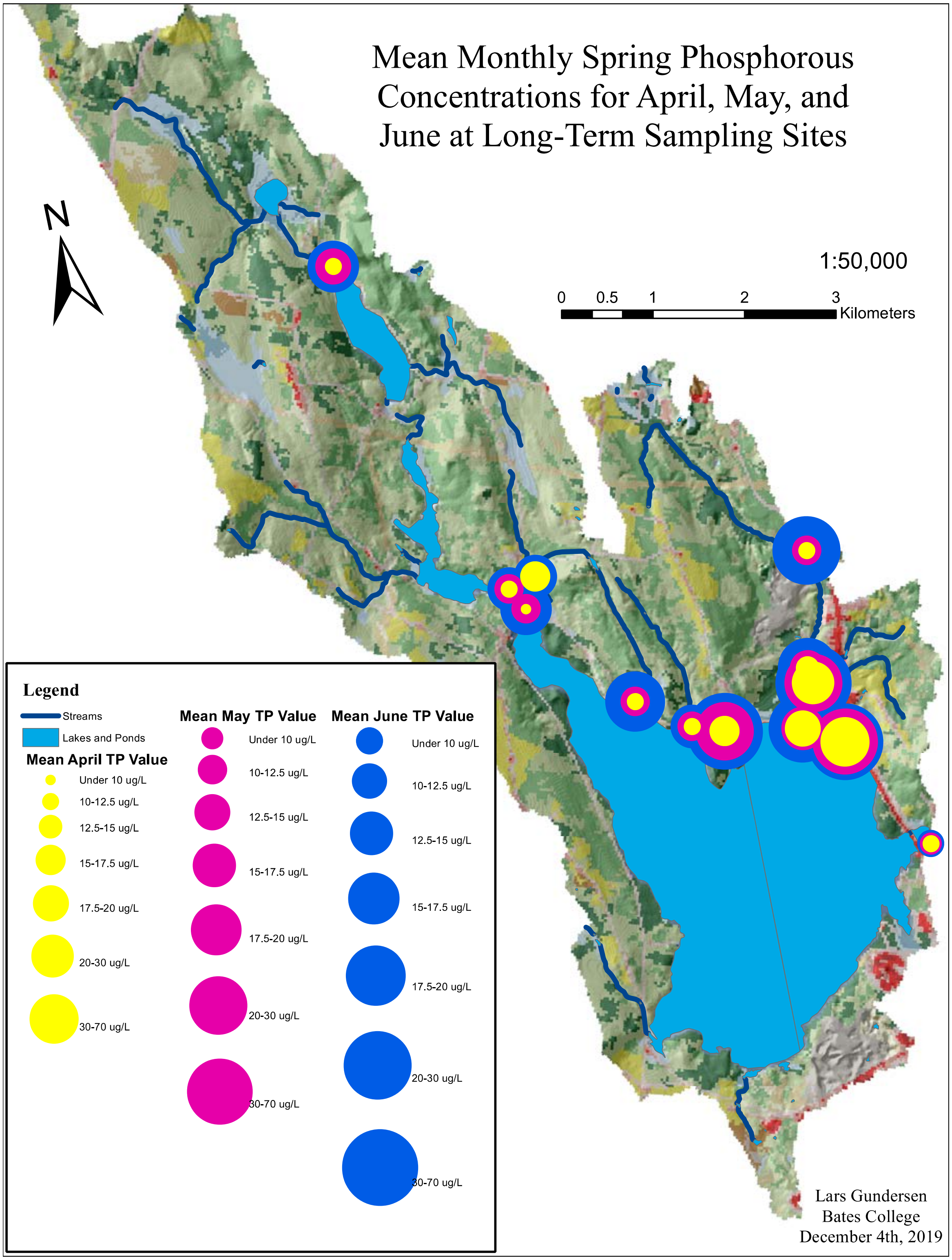


SF 6c: Discharge and total P concentration in Townsend Brook (all times in which both discharge and concentration data were collected since 2005). RSquare: 0.0841.



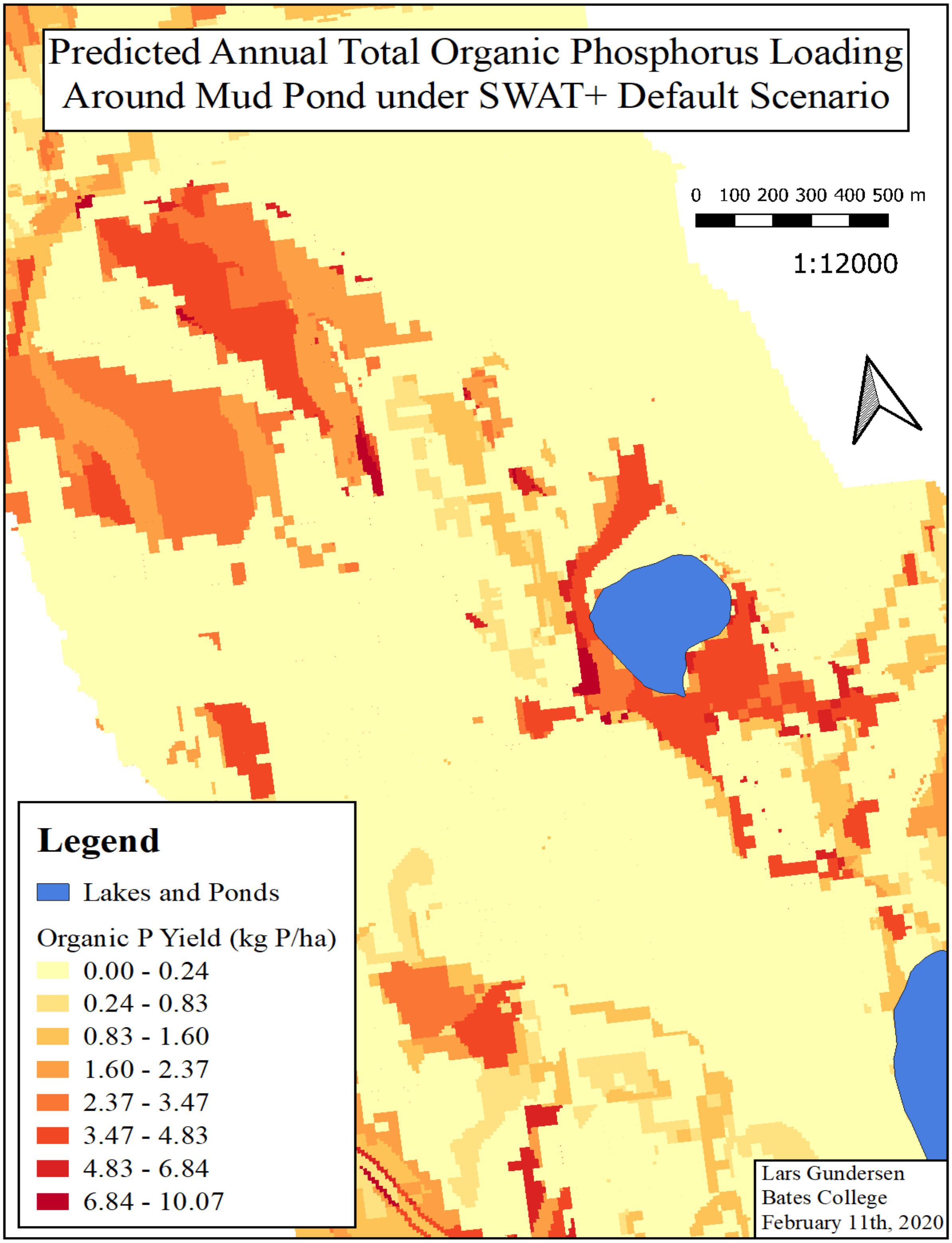
SF 6d: Discharge and total P concentration in Townsend Brook (all times in which both discharge and concentration data were collected since 2005 with four outliers excluded: 3/19/2014, 8/14/2014 11:15am, 8/14/2014 5:30pm, and 9/30/2015). RSquare: 0.3705.

APPENDIX F: SUPPLEMENTARY EXISTING DATA ANALYSIS MAPS

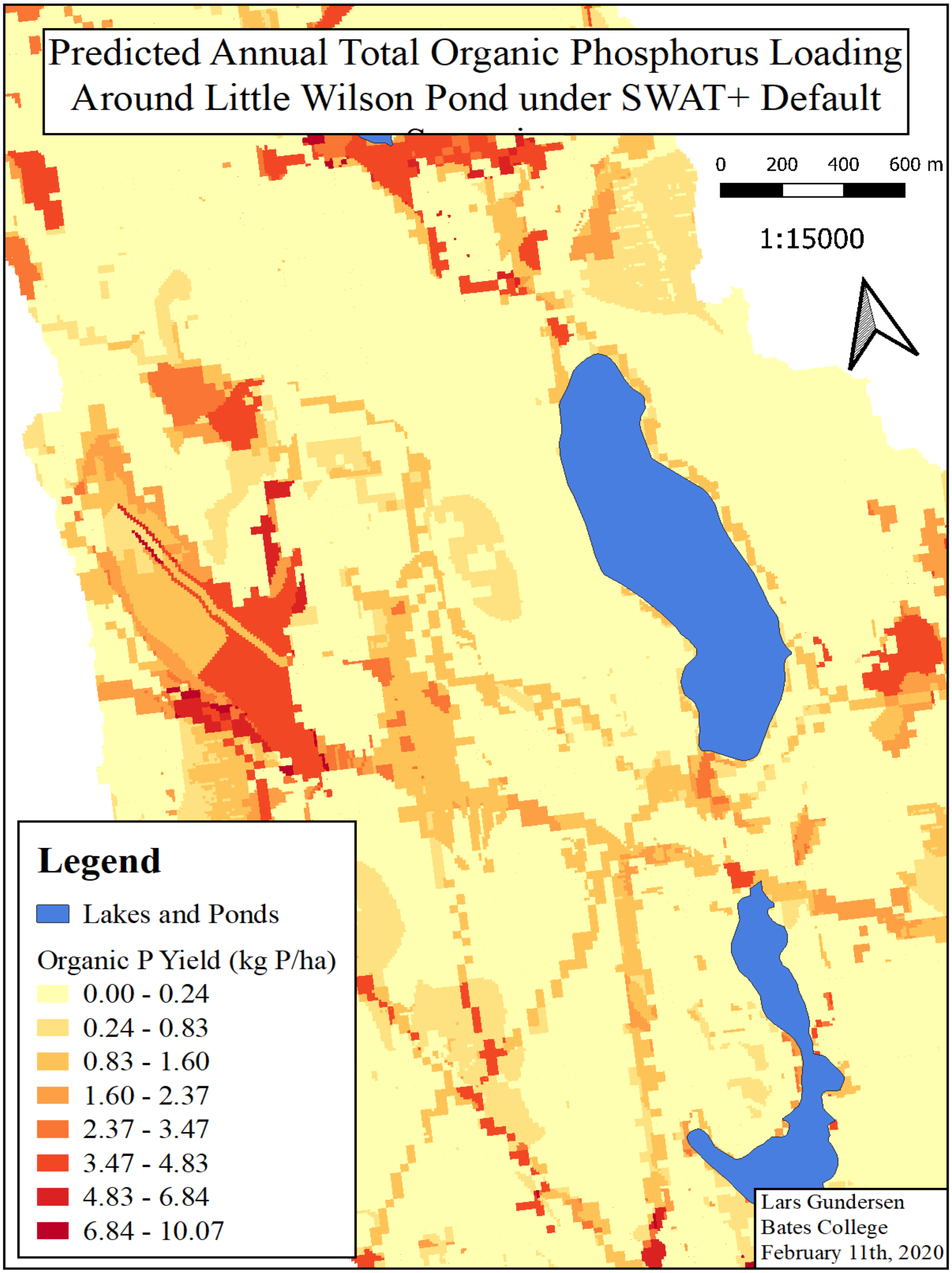


Map 7: Spring P concentrations (April, May, and June) at long-term sampling locations.

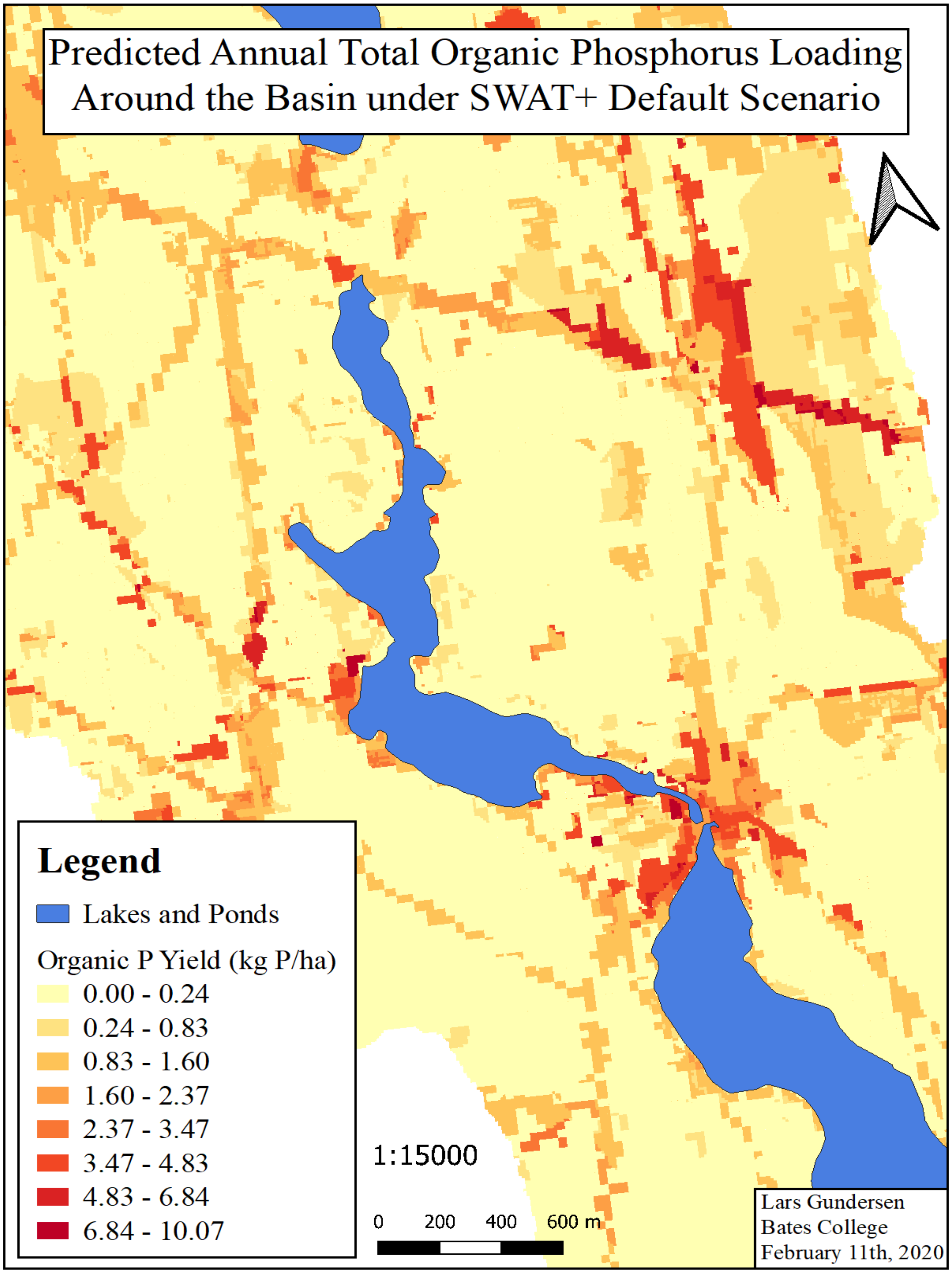
APPENDIX G: SUPPLEMENTARY SWAT MAPS



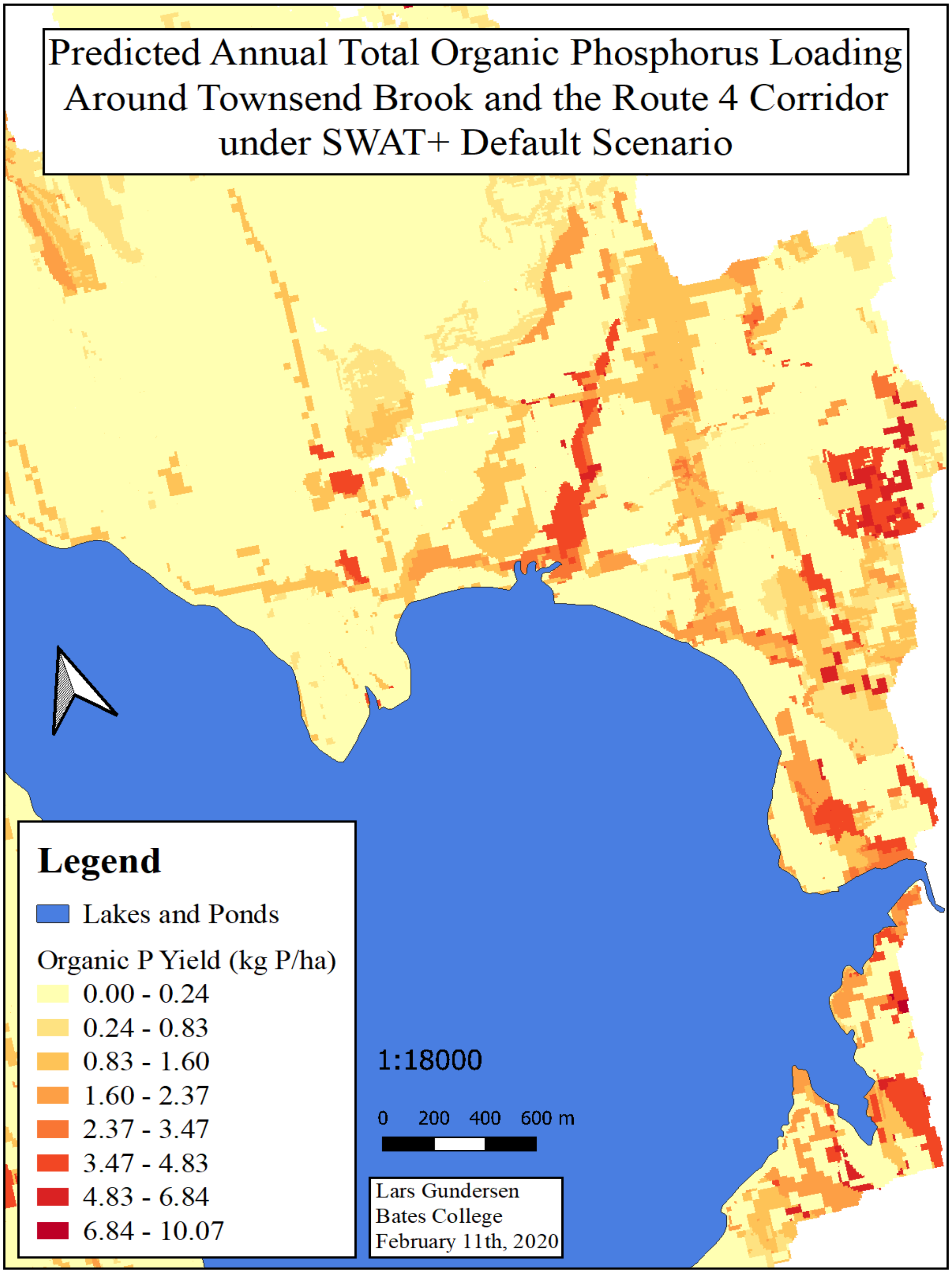
SM 2a: Predicted annual total organic P loading around Mud Pond under SWAT+ default scenario.



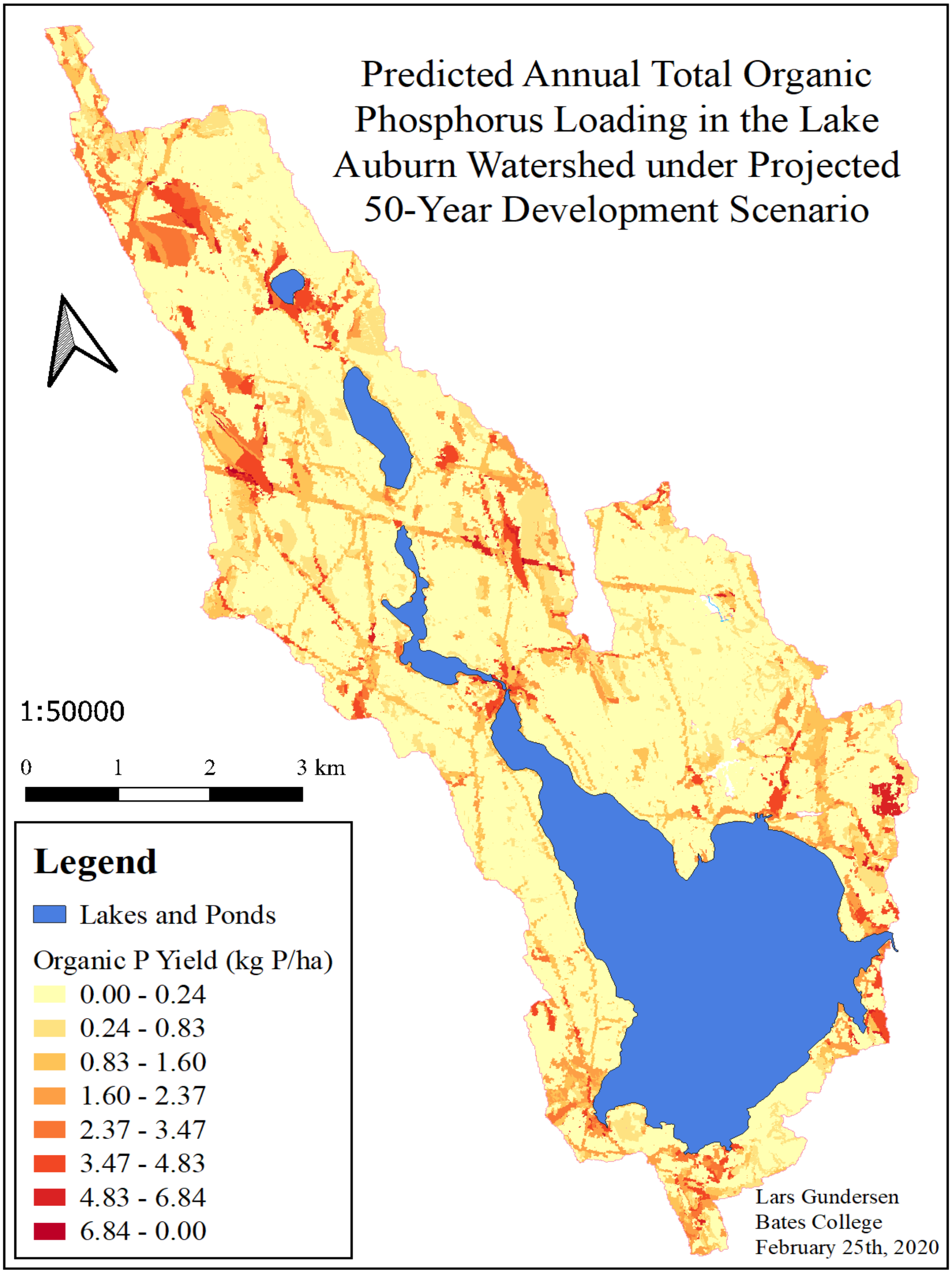
SM 2b: Predicted annual total organic P loading around Little Wilson Pond under SWAT+ default scenario.



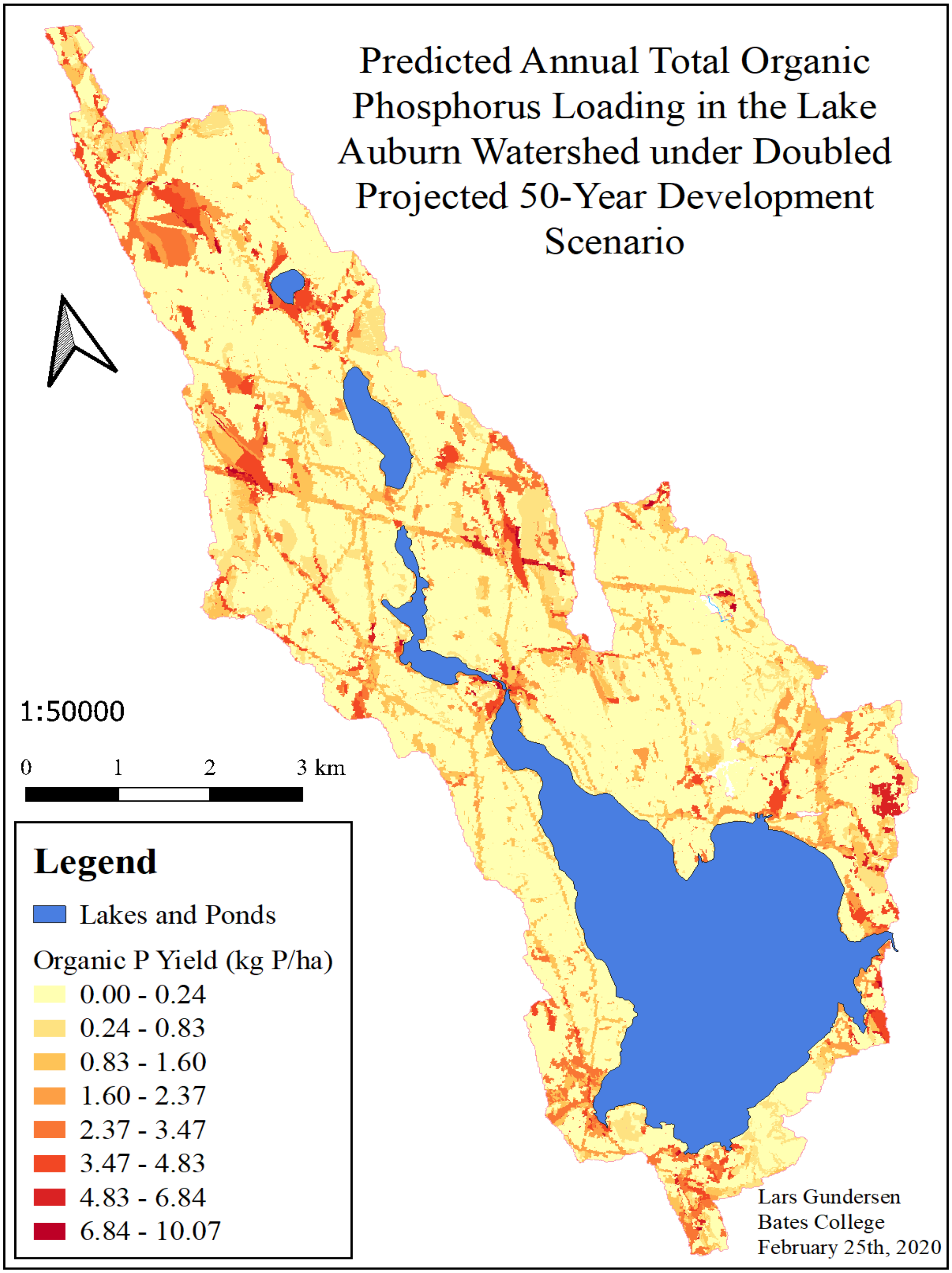
SM 2c: Predicted annual total organic P loading around the Basin under SWAT+ default scenario.



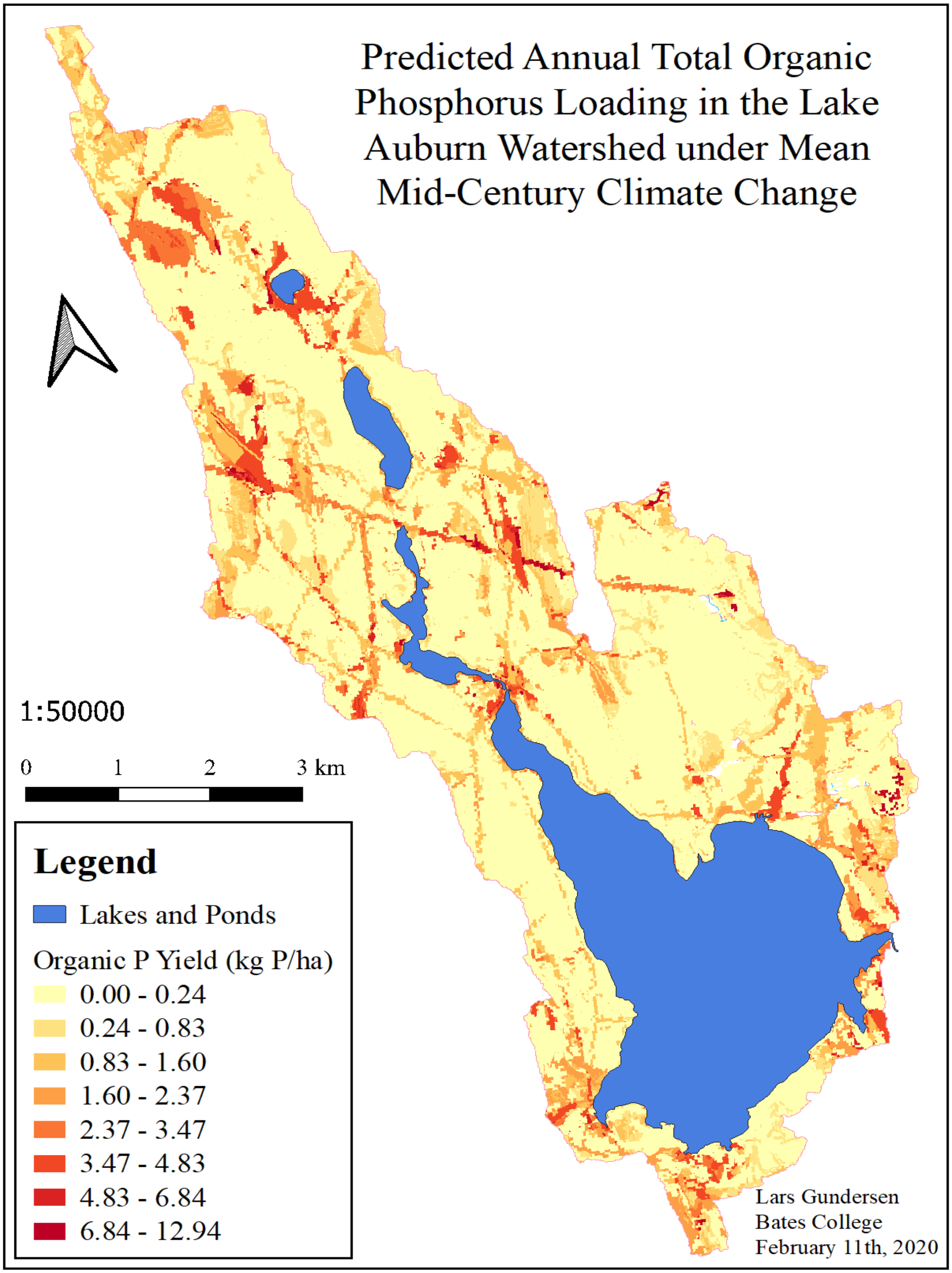
SM 2d: Predicted annual total organic P loading around Townsend Brook and the Route 4 corridor under SWAT+ default scenario.



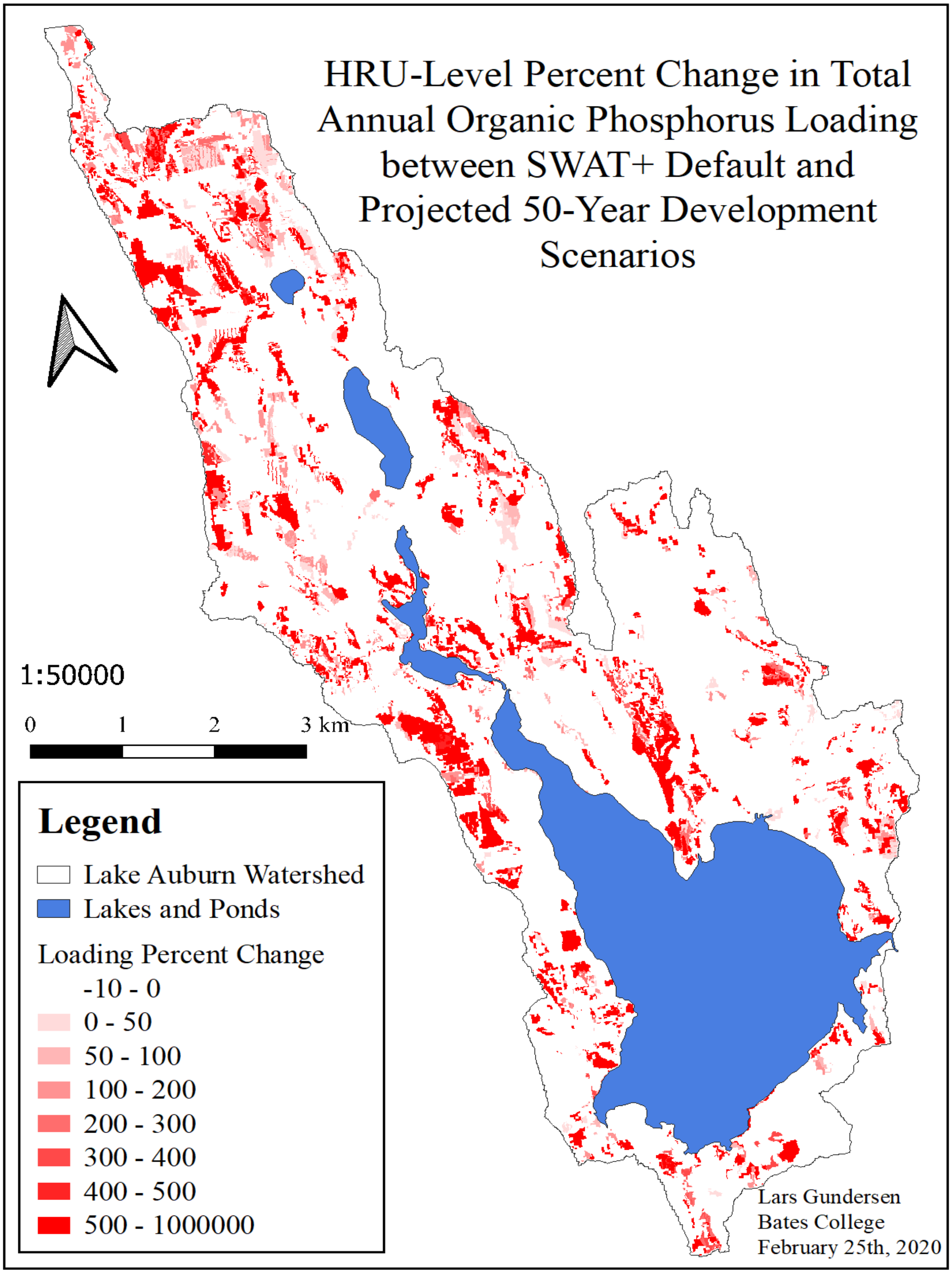
SM 3a: Predicted annual total organic P loading in the Lake Auburn watershed under projected 50-year development scenario.



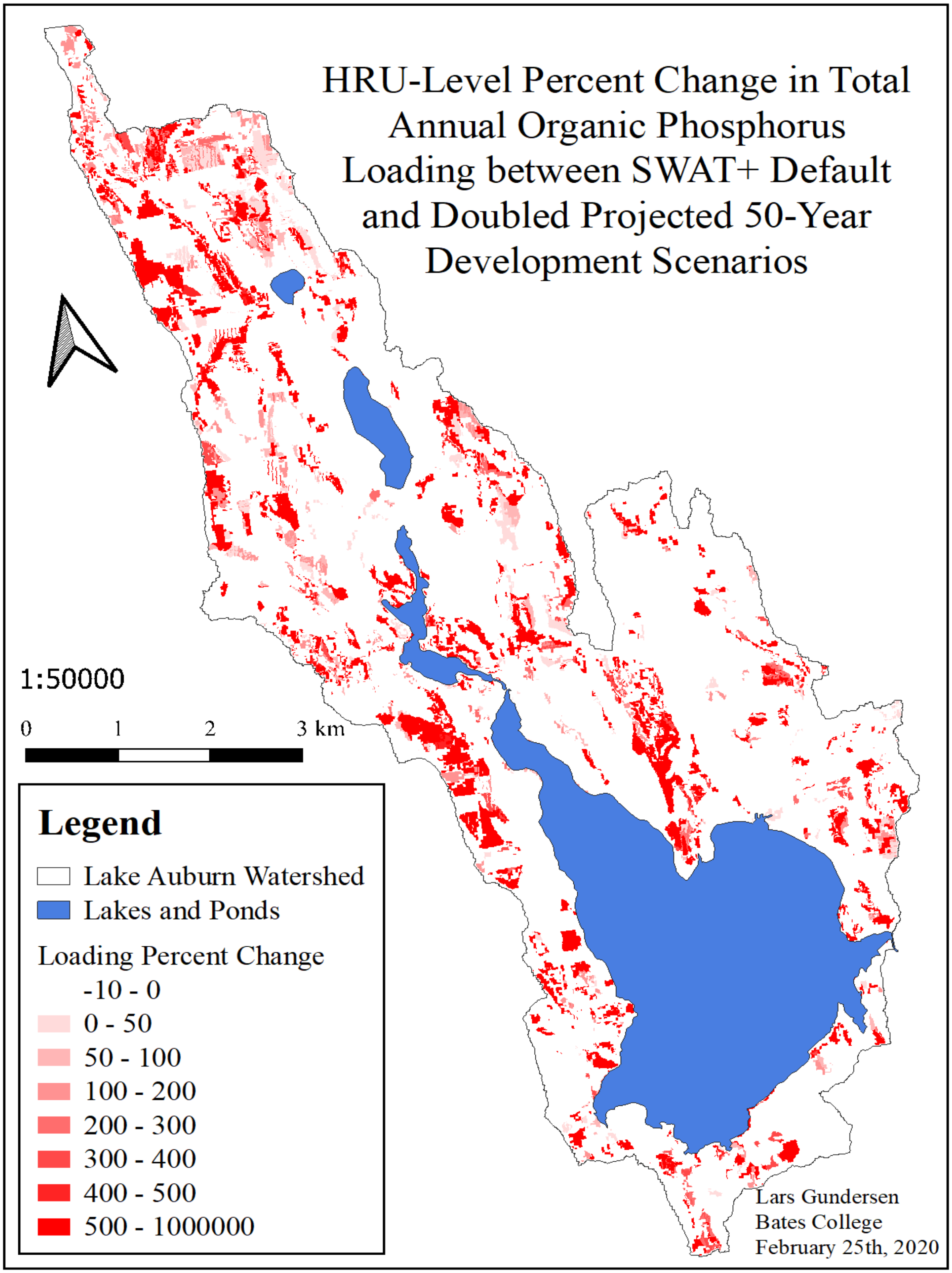
SM 3b: Predicted annual total organic P loading in the Lake Auburn watershed under doubled projected 50-year development scenario.



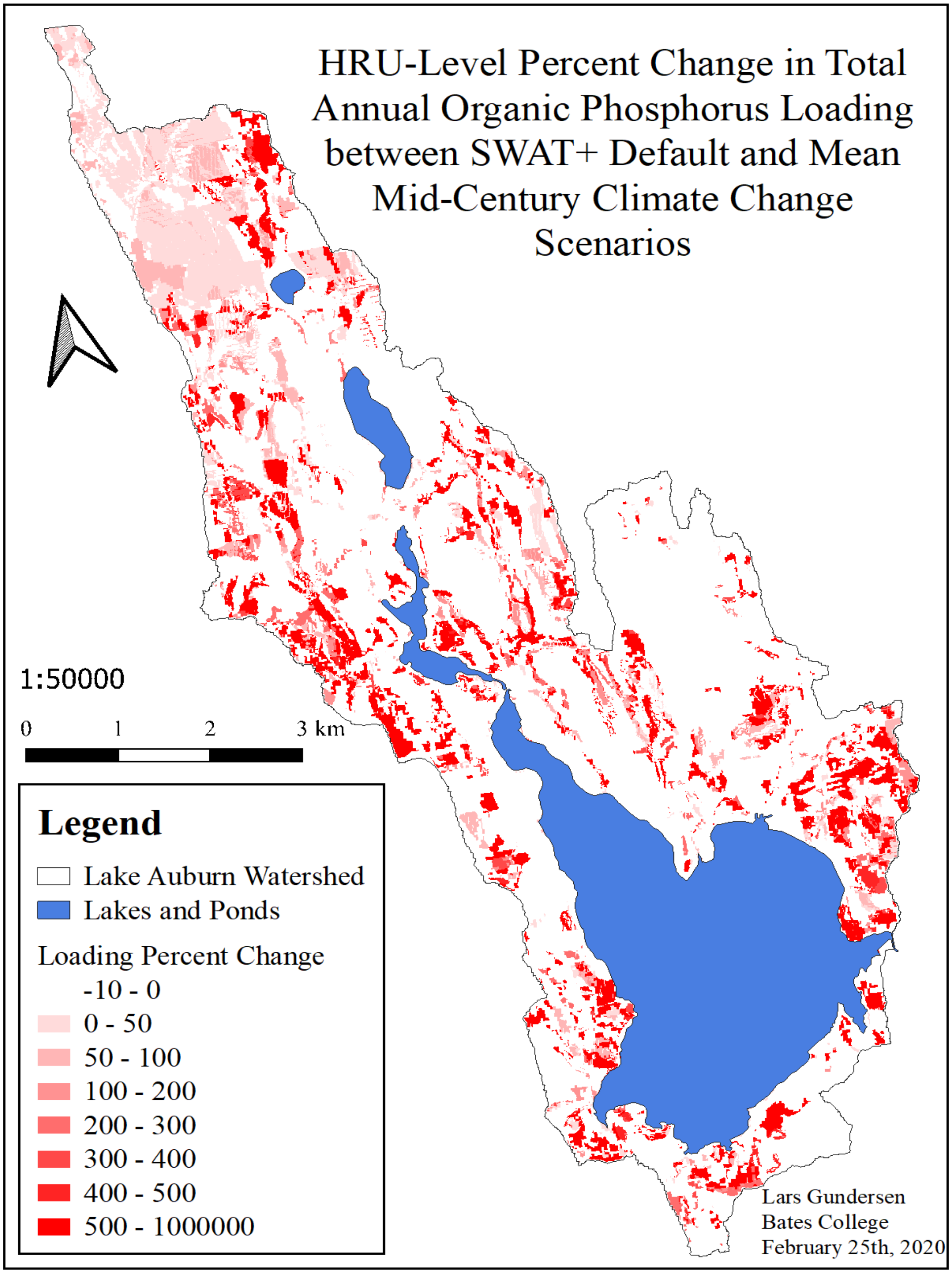
SM 3c: Predicted annual total organic P loading in the Lake Auburn watershed under mean mid-century climate change scenario.



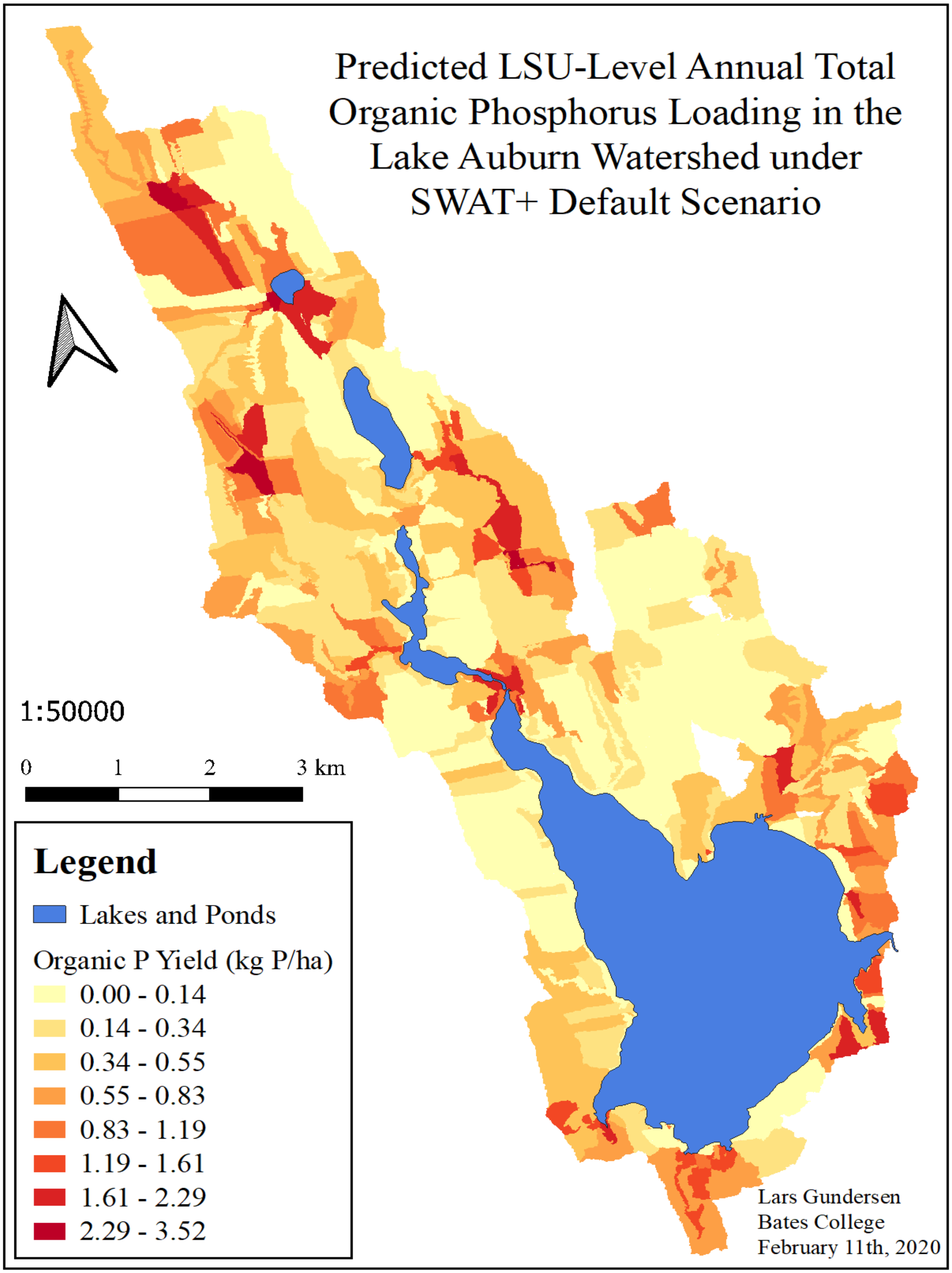
SM 4a: HRU-level percent change in total annual organic P loading between SWAT+ default and projected 50-year development scenarios.



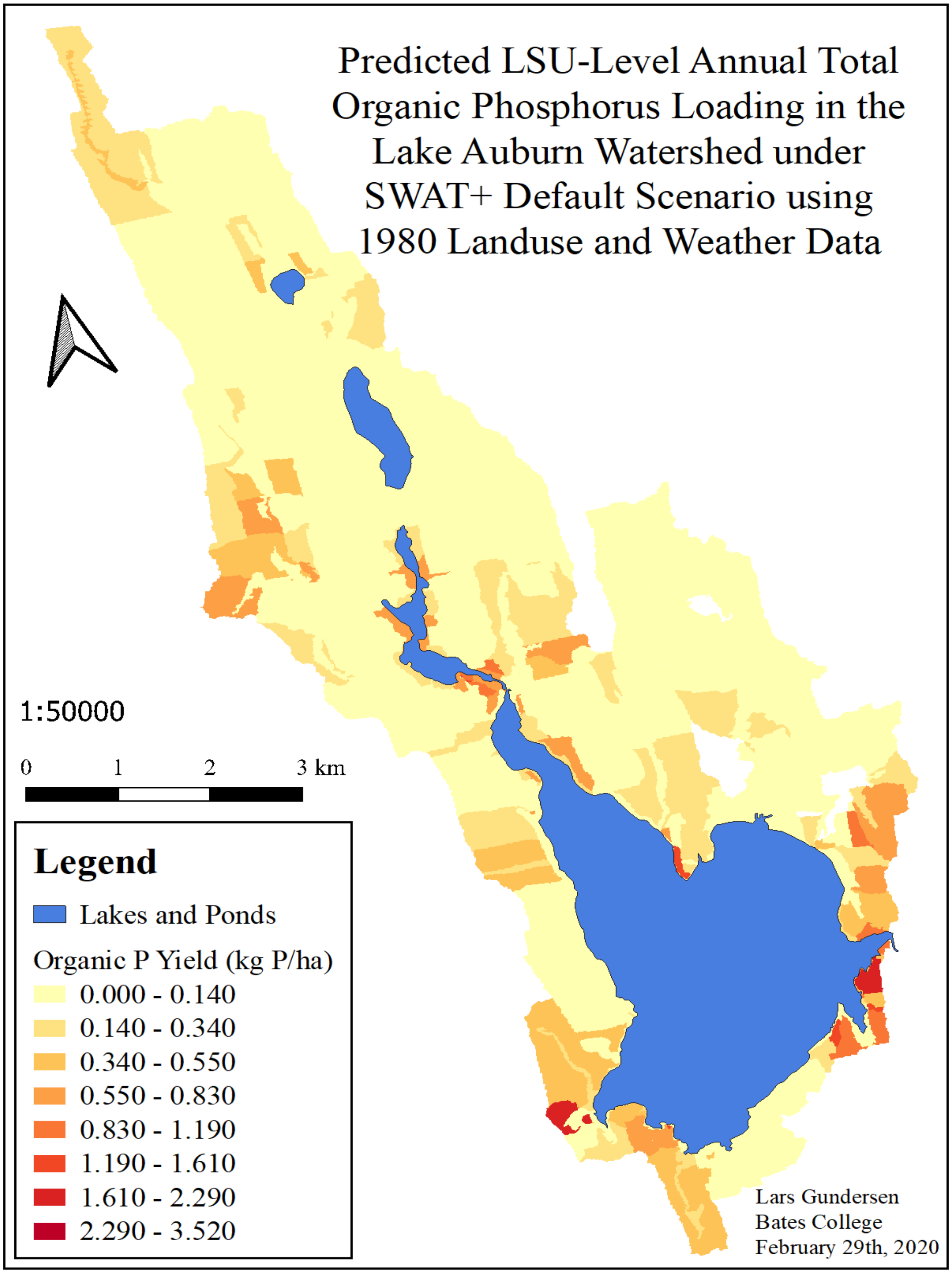
SM 4b: HRU-level percent change in total annual organic P loading between SWAT+ default and doubled projected 50-year development scenarios.



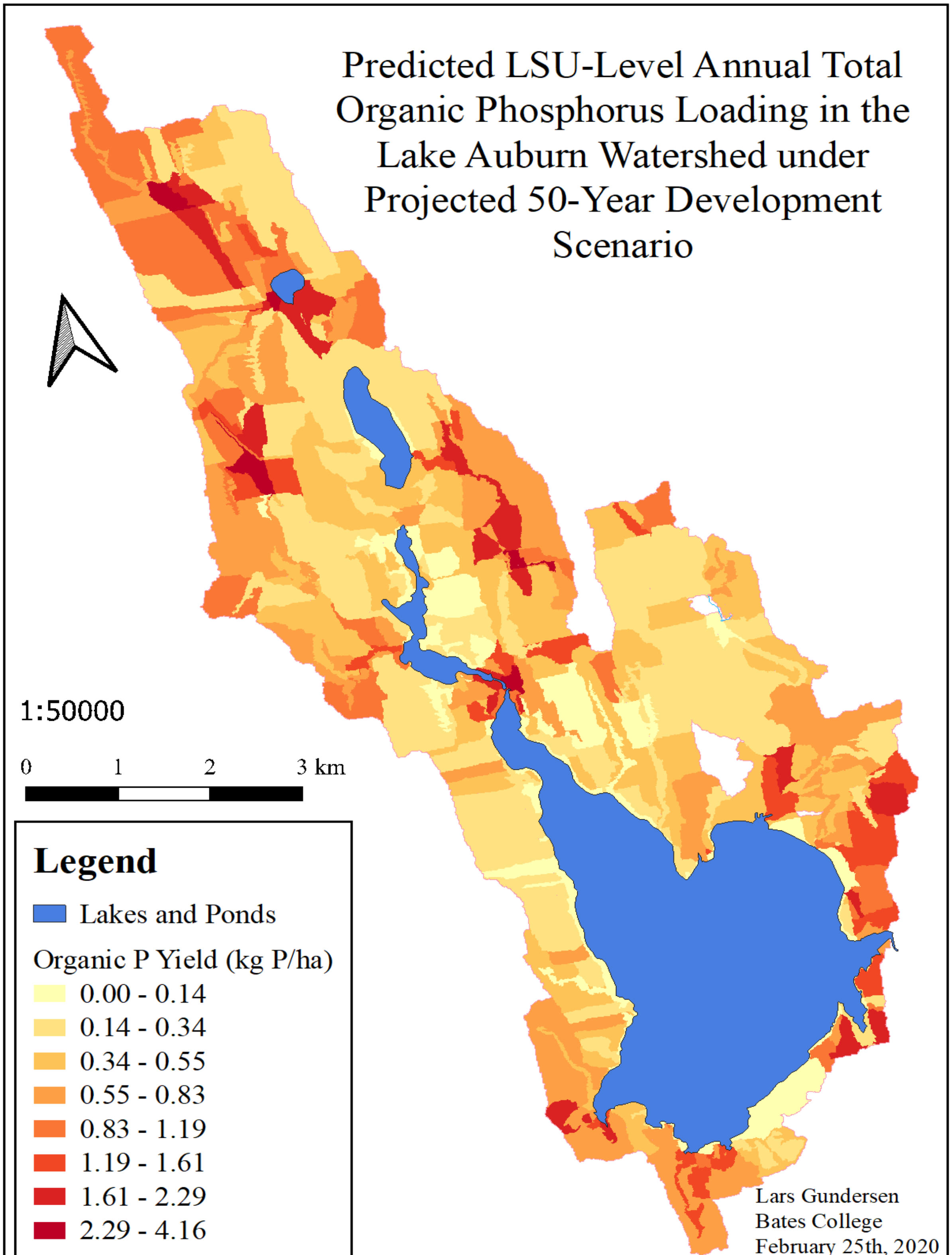
SM 4c: HRU-level percent change in total annual organic P loading between SWAT+ default and mean mid-century climate change scenarios.



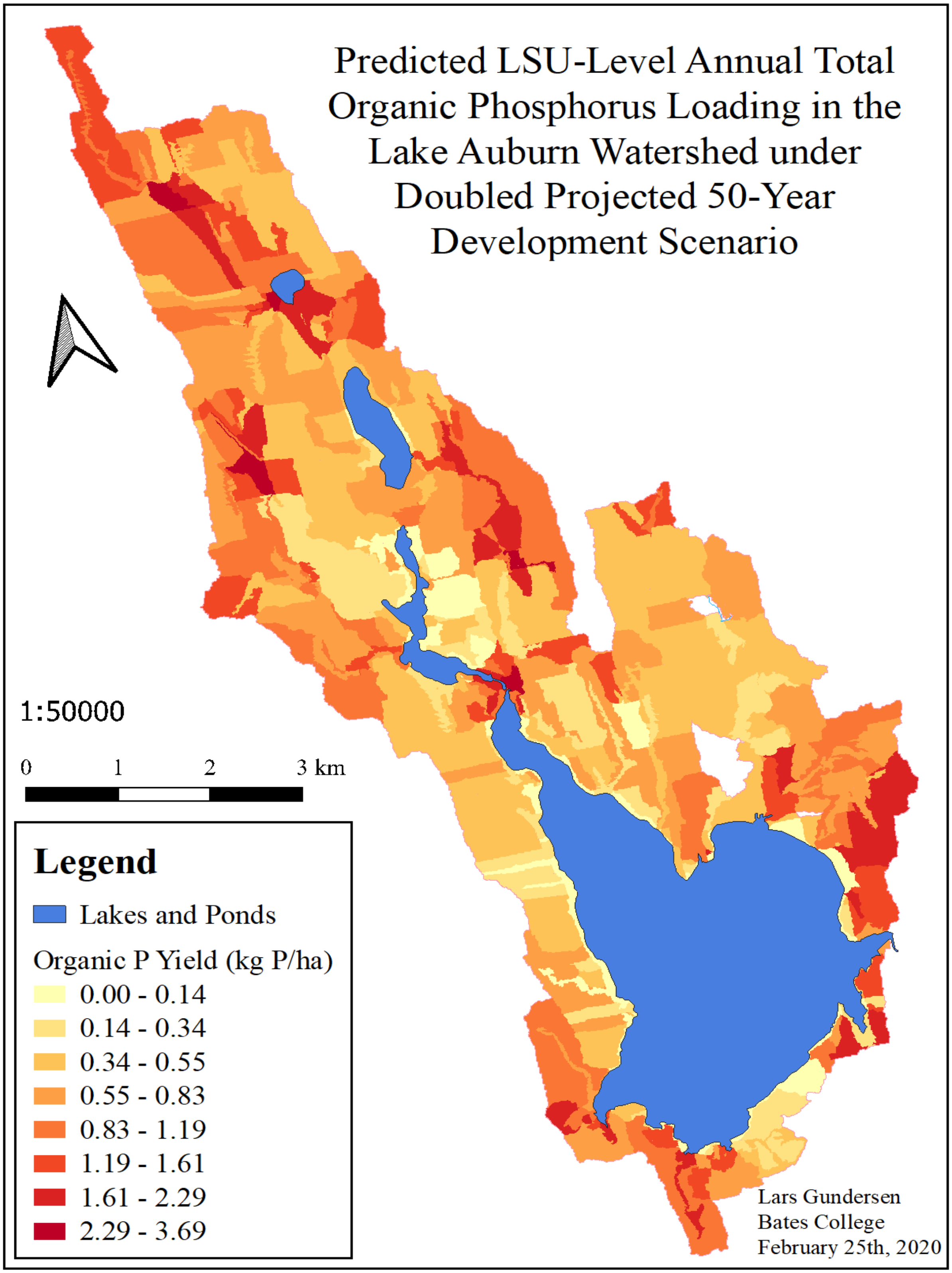
SM 5a: Predicted LSU-level annual total organic P loading in the Lake Auburn watershed under SWAT+ default scenario.



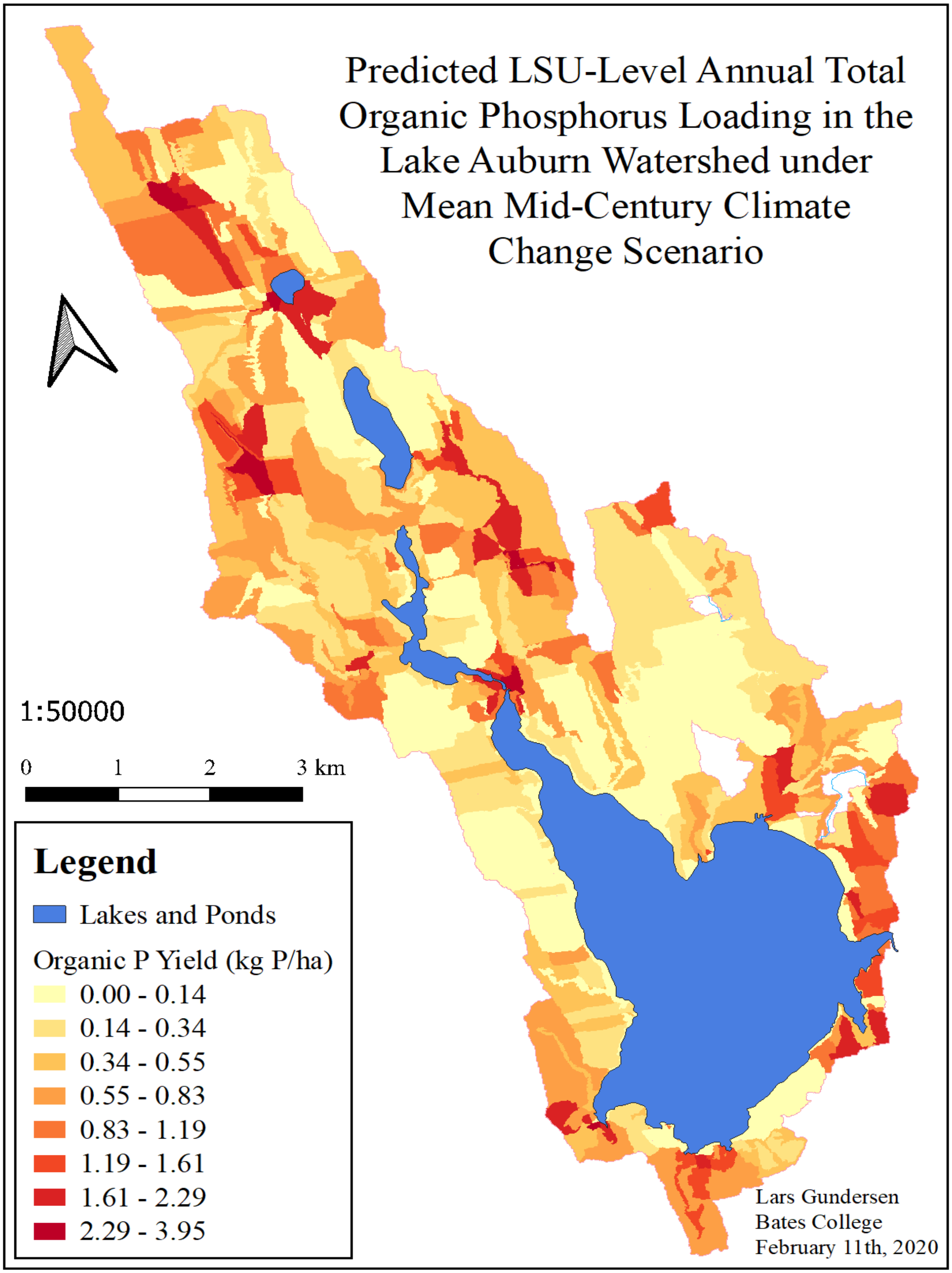
SM 5b: Predicted annual total organic P loading in the Lake Auburn watershed under SWAT+ default scenario using 1980 land use and weather data



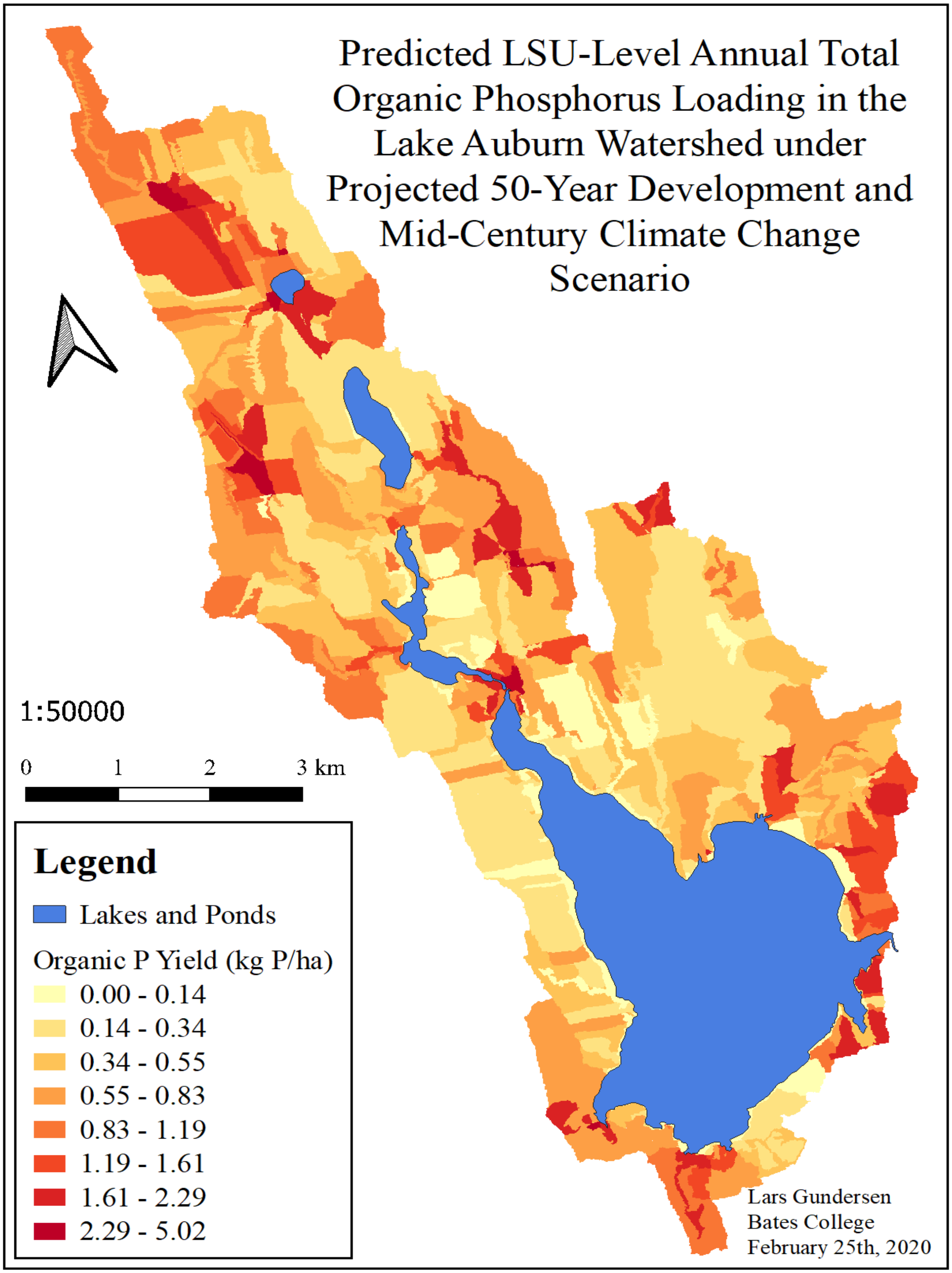
SM 5c: Predicted annual total organic P loading in the Lake Auburn watershed under projected 50-year development scenario



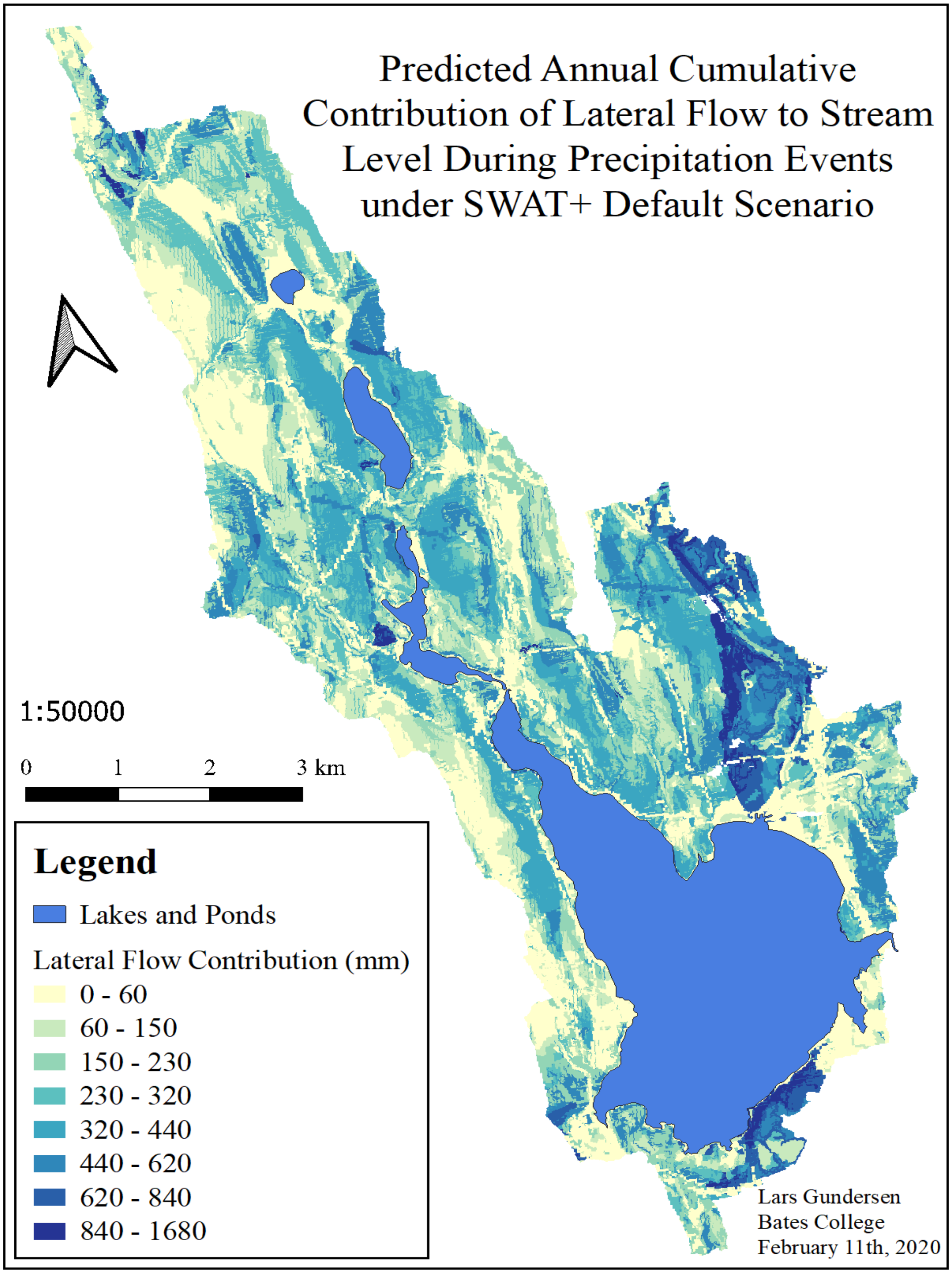
SM 5d: Predicted annual total organic P loading in the Lake Auburn watershed under doubled projected 50-year development scenario



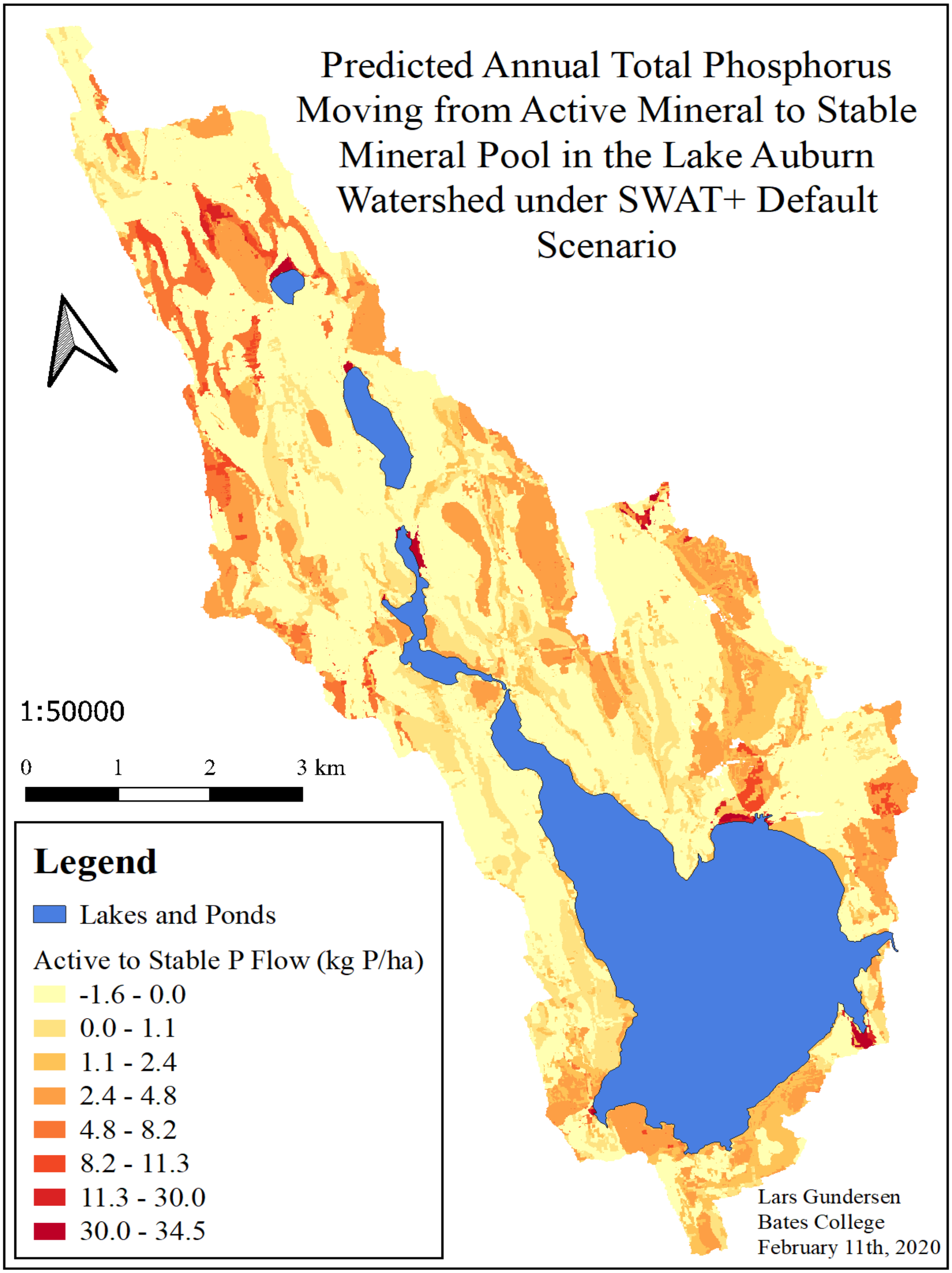
SM 5e: Predicted annual total organic P loading in the Lake Auburn watershed under mean mid-century climate change scenario



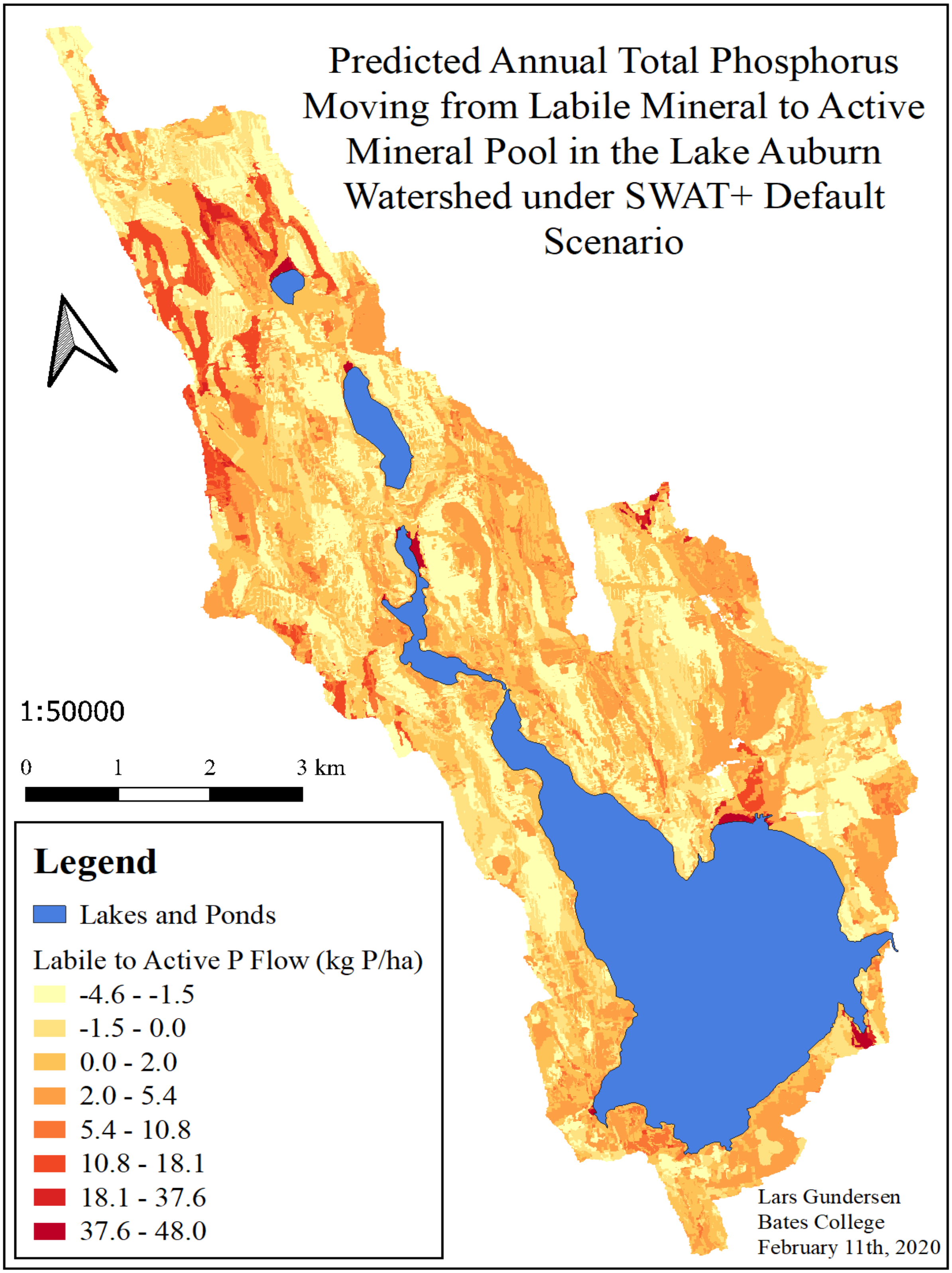
SM 5f: Predicted annual total organic P loading in the Lake Auburn watershed under projected 50-year development and mid-century climate change scenario



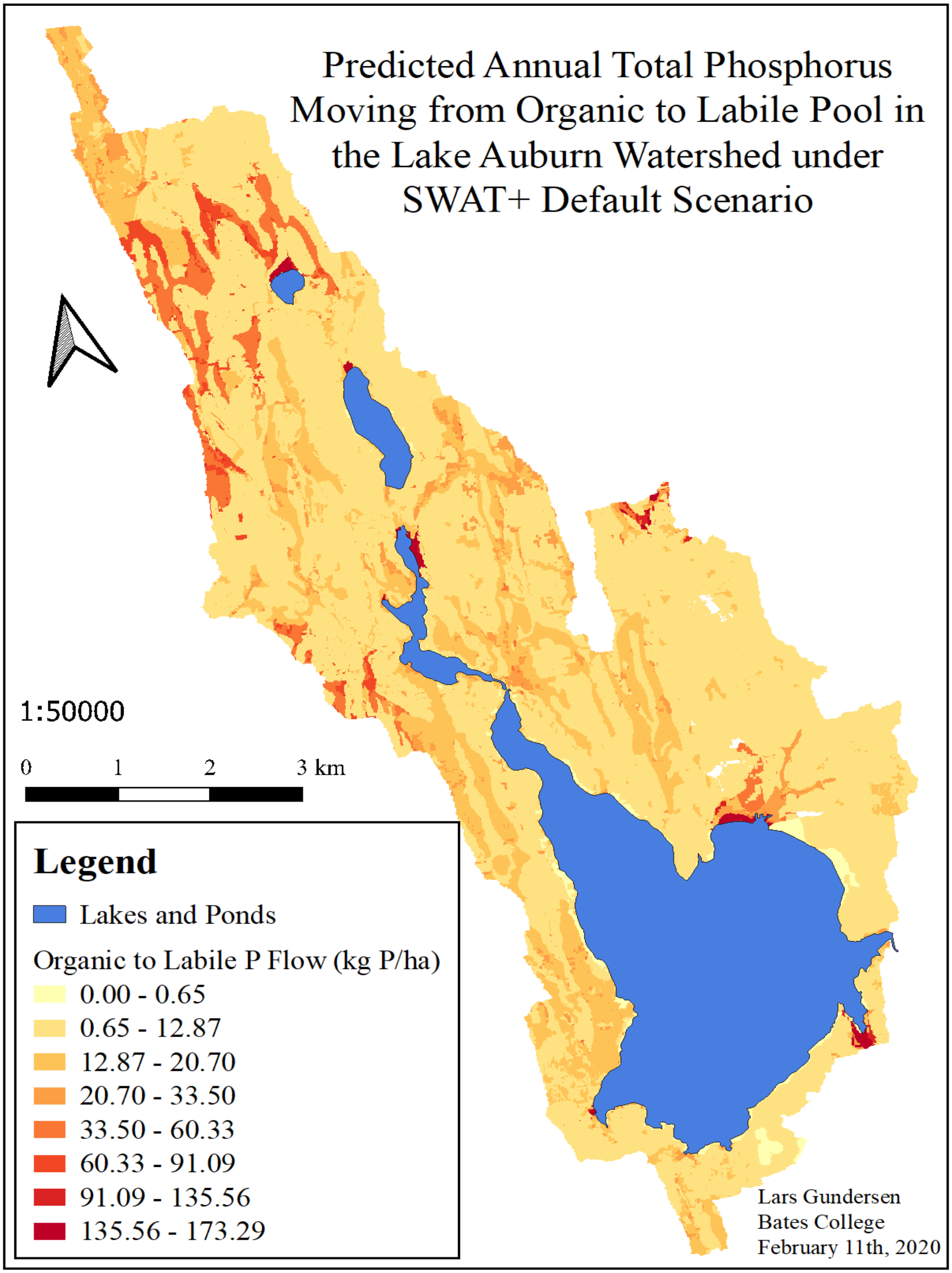
SM 6: Predicted HRU-level annual cumulative contribution of lateral flow to stream level during precipitation events under SWAT+ default scenario



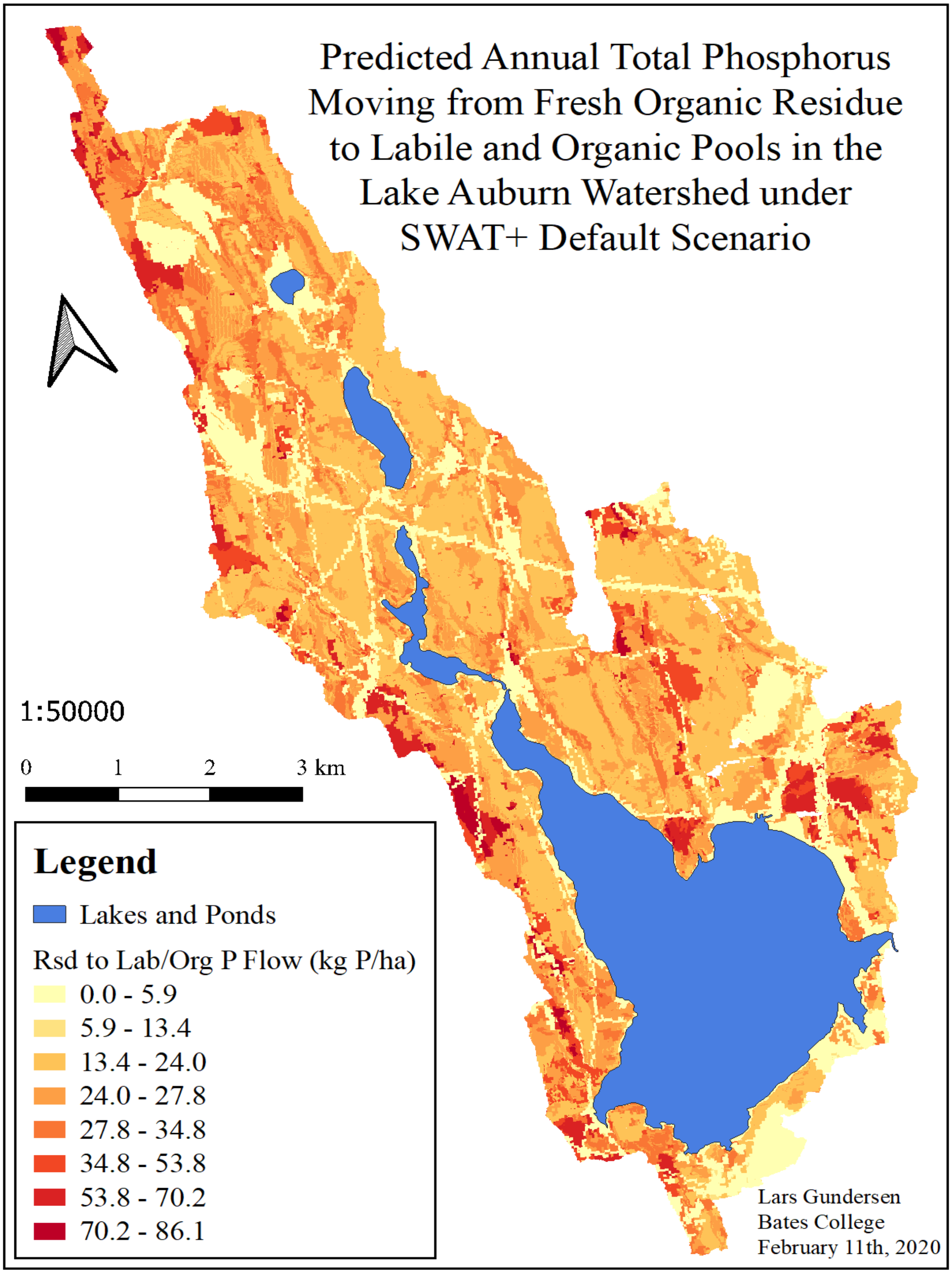
SM 7a: Predicted HRU-level annual total P moving from active mineral to stable mineral pool in the Lake Auburn watershed under SWAT+ default scenario



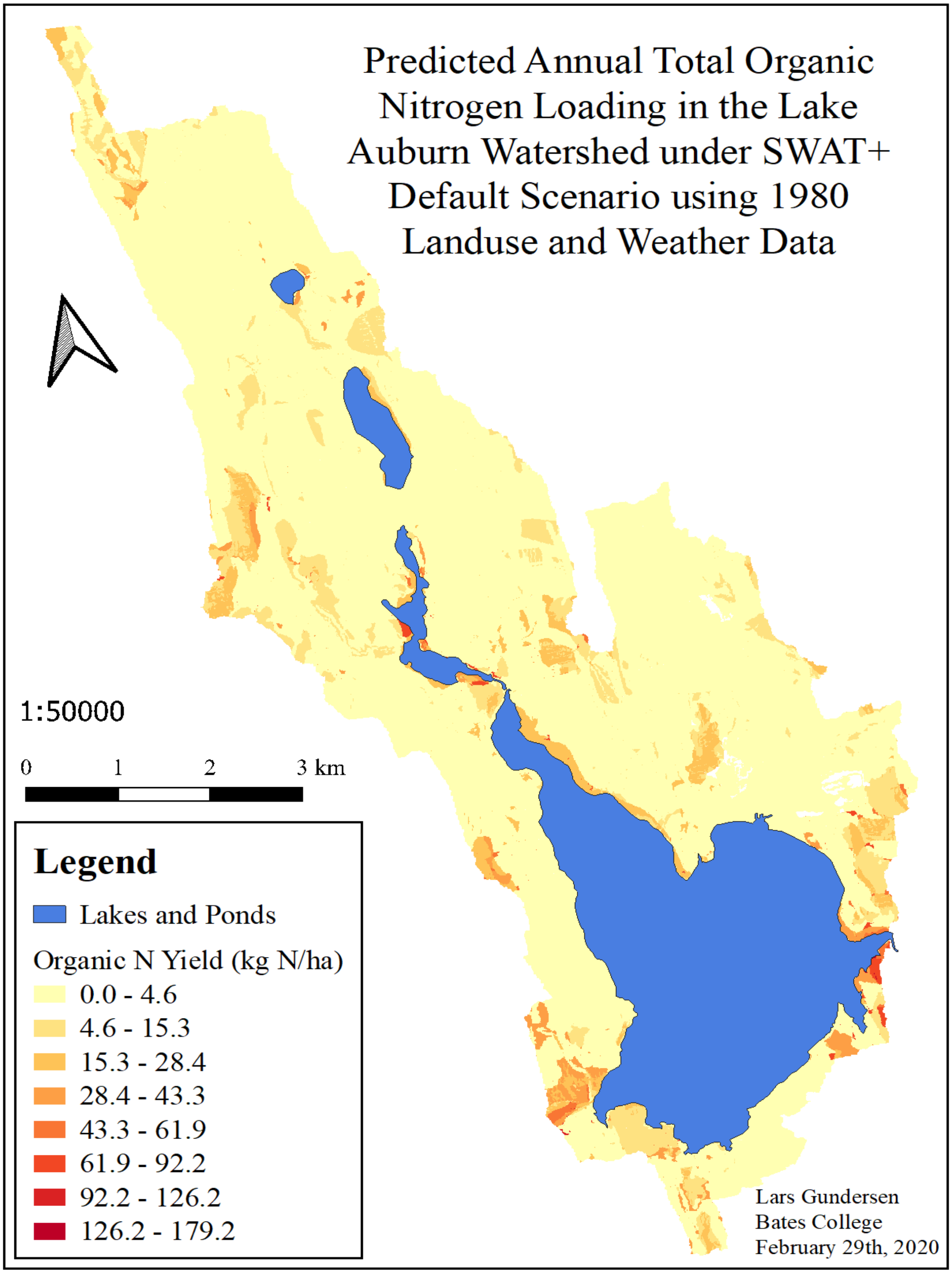
SM 7b: Predicted HRU-level annual total P moving from labile mineral to active mineral pool in the Lake Auburn watershed under SWAT+ default scenario



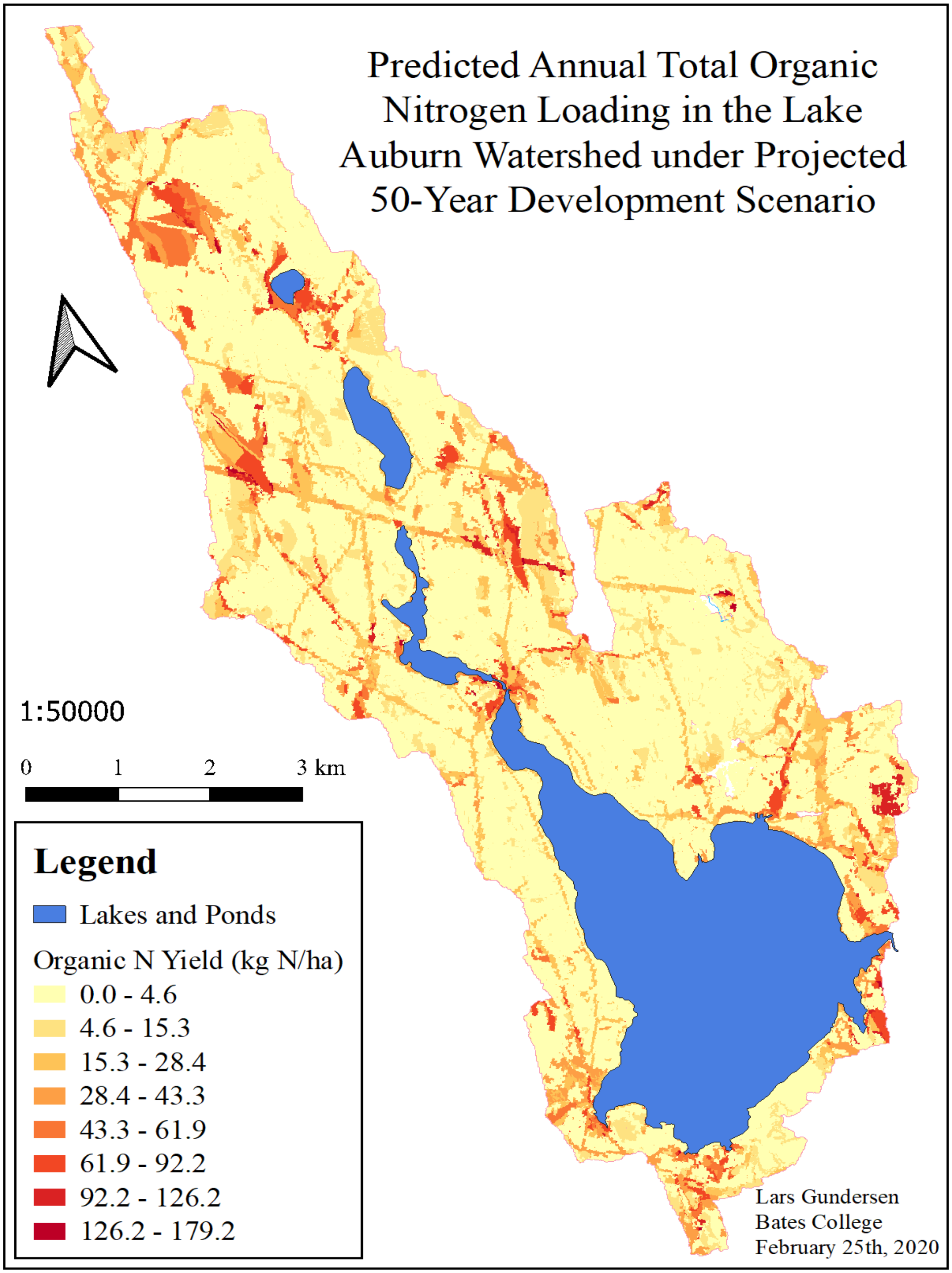
SM 7c: Predicted HRU-level annual total P moving from organic to labile pool in the Lake Auburn watershed under SWAT+ default scenario



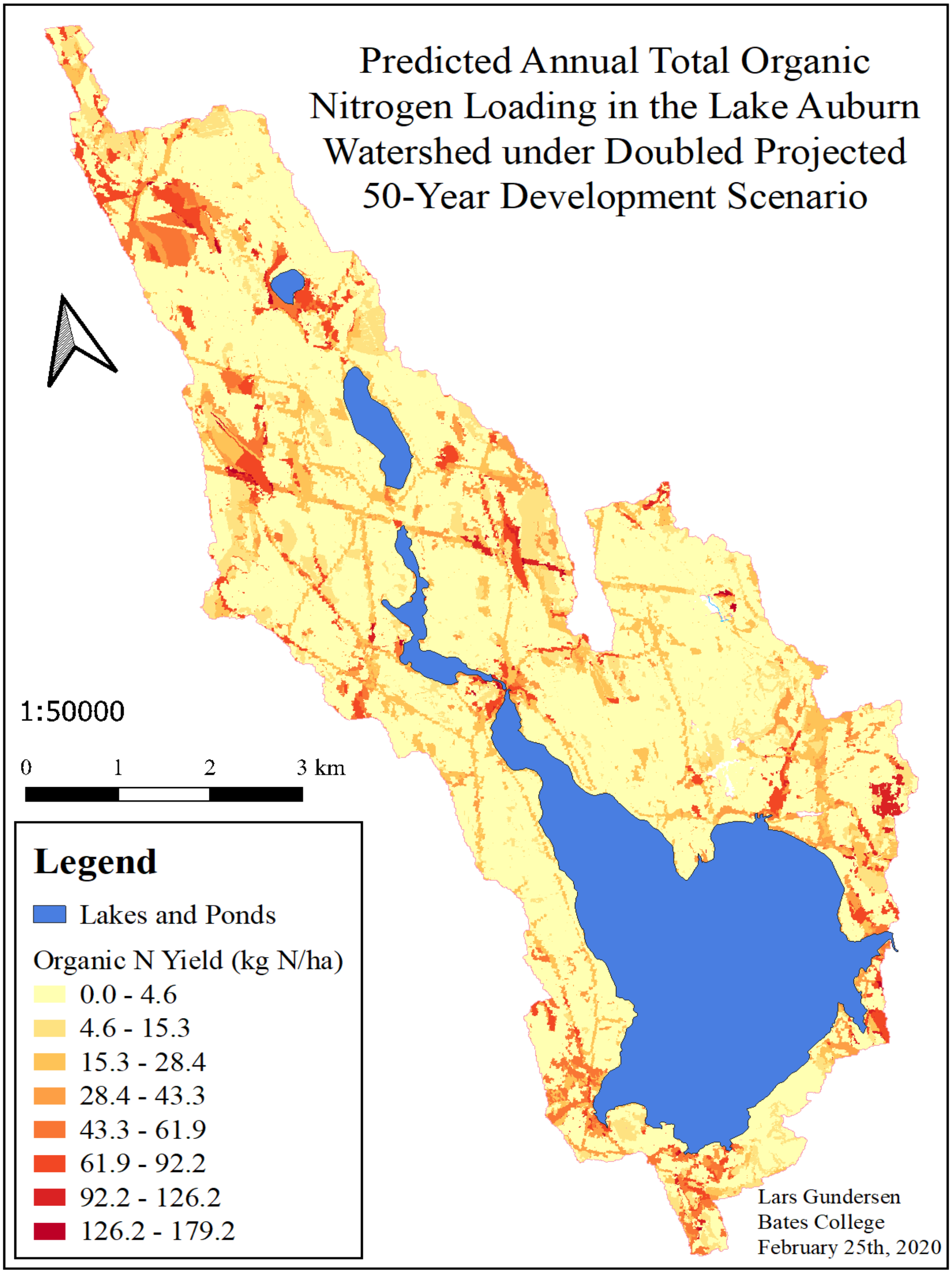
SM 7d: Predicted HRU-level annual total P moving from fresh organic residue to labile and organic pools in the Lake Auburn watershed under SWAT+ default scenario



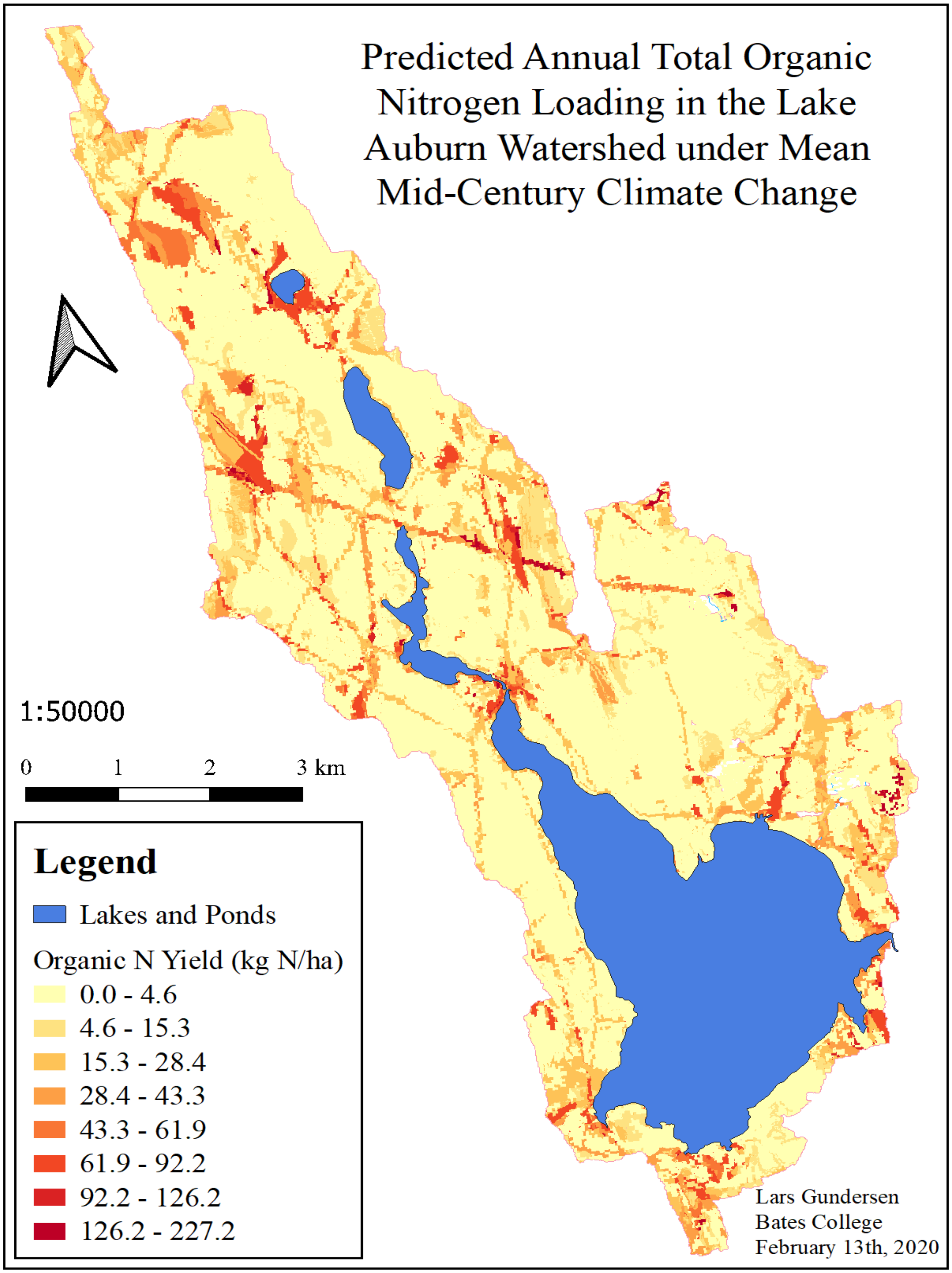
SM 8a: Predicted HRU-level annual total organic N loading in the Lake Auburn watershed under SWAT+ default scenario using 1980 land use and weather data



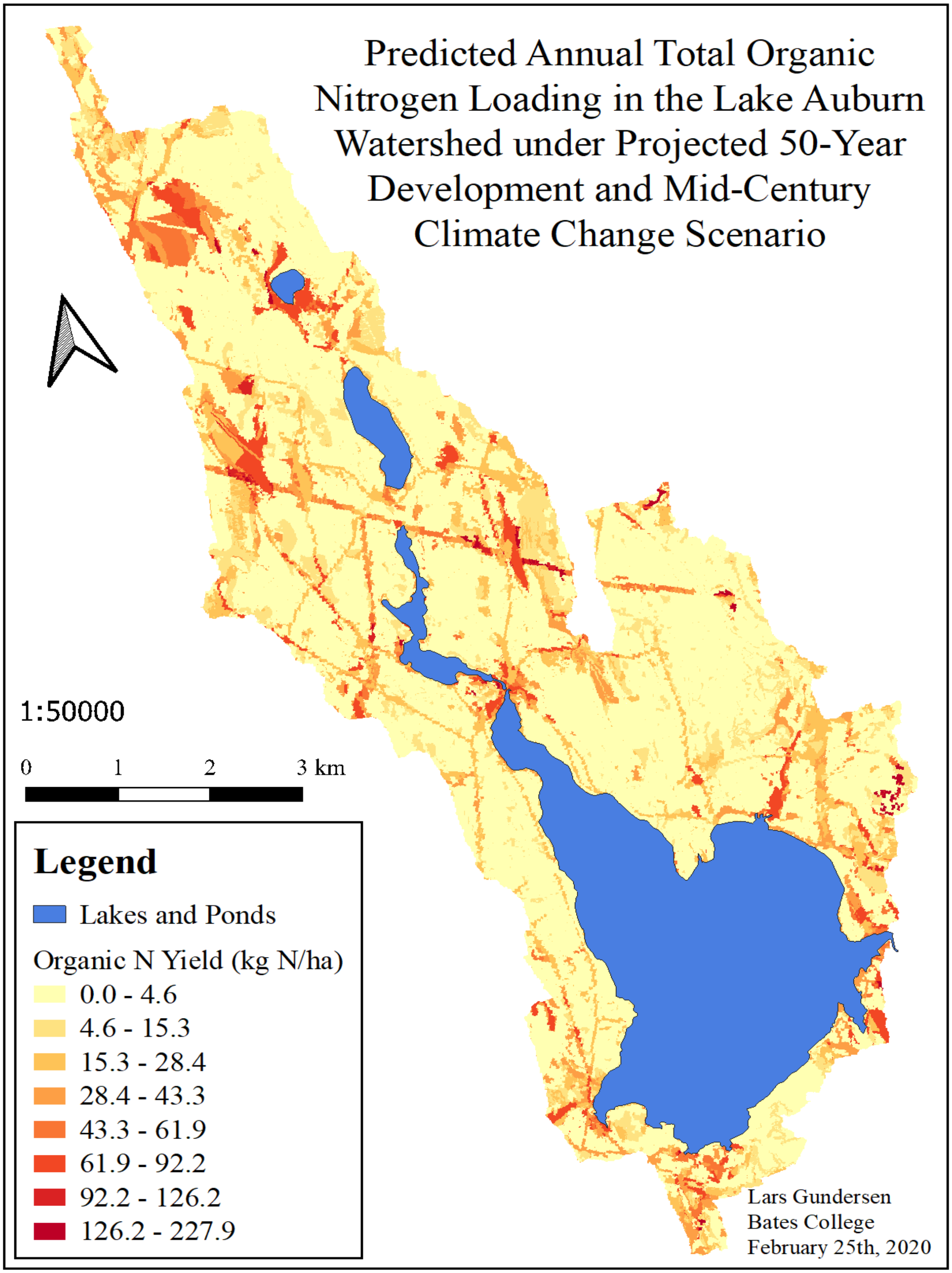
SM 8b: Predicted HRU-level annual total organic N loading in the Lake Auburn watershed under projected 50-year development scenario



SM 8c: Predicted HRU-level annual total organic N loading in the Lake Auburn watershed under doubled projected 50-year development scenario



SM 8d: Predicted HRU-level annual total organic N loading in the Lake Auburn watershed under mean mid-century climate change scenario



SM 8e: Predicted HRU-level annual total organic N loading in the Lake Auburn watershed under projected 50-year development and mid-century climate change scenario.