# Statistical Relationships Between Observational Water Quality and Catchment Agricultural Intensity in Rural Maine

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ACADEMIC ABSTRACT

Anthropogenic eutrophication of freshwater lakes due to land use change is a growing global problem with economic consequences, such as a reduction in shoreline property value. Managing eutrophication is of utmost importance in Maine, USA due to the large number of inland fresh waterbodies and their economic importance for fisheries, recreation, and real estate. This thesis investigates the relationships between water quality and catchment land use. Agricultural land use is a large driver of excess nutrient export to lakes, including in Maine, and can result in toxic cyanobacterial blooms, decreased water clarity, and fish kills. I developed a statistical relationship to quantitatively link agricultural intensity in the catchment and resultant water quality outcomes in Maine lakes. I observe a strong statistical relationship between water quality and anthropogenic activity in the catchment, as expected. Interestingly, I found that the effects of anthropogenic activity were most closely related to a five-year lag in water quality, which is between 0.8 to 4.71 years longer than the lake residence times. My results suggest that changes in land use may have long-term effects on water quality that last for far longer than would be expected. The analysis presented in this paper is novel for directly linking long term observational agricultural and biological datasets and presents a new way to quantitatively link water quality and anthropogenic intensity in the catchment area.

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PUBLIC ABSTRACT

Worsening water quality of freshwater lakes due to land use change is a growing global problem with economic consequences, such as a reduction in shoreline property value. Managing water quality is of utmost importance in Maine, USA due to the large number of inland fresh waterbodies and their economic importance for fisheries, recreation, and real estate. This thesis investigates the relationships between water quality and lakeside land use. Agricultural land use is a large driver of lowered water quality in lakes, including in Maine. I developed a statistical relationship to mathematically link agriculture near the lake and resultant water quality outcomes in Maine. I observe a strong statistical relationship between water quality and human activity in the catchment, as expected. Interestingly, I found that the effects of human activity were most closely related to a five-year lag in water quality, which is between 0.8 to 4.71 years longer than the lake residence times. My results suggest that changes in land use may have long-term effects on water quality that last for far longer than would be expected.

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**Table of Contents**

**List of Tables**



**List of Figures**

## I. Introduction

Anthropogenic eutrophication of freshwater lakes due to land use change is a common and growing global problem (Carpenter et al. 2011; Dodds et al. 2013). The reduction in water quality alters the ecological structure and function of freshwaters (Carpenter, Bolgrien, et al. 1998; Dodds et al. 2000). This multifaceted issue has negative effects for fisheries, recreation, industry, agriculture, drinking water, and has substantial economic consequences (Carpenter, Caraco, et al. 1998). For example, a reduction in water clarity from eutrophication reduces the value of shoreline properties (Boyle et al. 1998; Gibbs et al. 2002).

Lakes are an integral part of Maine's economy as they add tremendous value to the state through recreation, property values, drinking water, and commercial use. In 2010, direct expenditures on lakes accounted for 5% of the state's GDP ($2.5 billion dollars) and a 2006 study found that lakes support over 52,000 jobs statewide (CEPG 2010; MDEP 2006). Reductions in water clarity impacts a lake's aesthetic benefit, diminishes user satisfaction, and lowers the net economic value of lakes (Boyle et al. 2003; Boyle et al. 1997; Michael et al. 1996). The substantial impact of lakes on Maine's economy makes the changing conditions of lake and pond ecosystems a major concern to the state and the region.

Besides their economic importance, lakes in Maine are an important area of study as Maine has substantial inland surface water (5910 $km^2$), the majority being freshwater lakes. Most lake substrates have a relatively low amount of micronutrients which leads to most lakes in Maine being oligotrophic (Davis et al. 1978; Ferwerda 1997). Any observed eutrophication is usually due to anthropogenic activities in the lake catchment. Obvious symptoms of pollution and severe anthropogenic eutrophication exist only in a small number of Maine lakes, but subtle influences of anthropogenic nutrient loading are widespread (CEPG 2010; MDEP 2012).

Phosphorus (P) is a fundamental driver of primary production in freshwater lakes and excessive P input can lead to eutrophic degradation of lakes (Dillon et al. 1974; Schindler et al. 1971; Vollenweider 1968). The interdependence of land use and P export from a variety of human modified landscapes has been documented (Beaulac et al. 1982; Frink 1991), and studies have linked changes in lake water quality to land-use practices within catchments due to nonpoint source P pollution (Jones et al. 2004; Soranno et al. 1996). Efforts to improve or protect lake water quality have focused on reducing P input (Sharpley et al. 1994), but nonpoint source P loading in lakes remains a serious concern for water quality (Schindler et al. 2016). It is important to note that nitrogen (N) is also a key contributor to freshwater eutrophication (Elser et al. 2007; Lewis et al. 2008; Schindler et al. 2016), but N data are often not available in long-term monitoring records, hindering the study of N export and water quality relationships.

Trophic status models show lake response to changes in nonpoint source pollution of P (Vollenweider et al. 1980; Vollenweider 1970, 1975), and many empirical studies in the literature focus on anthropogenic eutrophication (Leone et al. 1993; Silvino et al. 2015; Taranu et al. 2008). Comparatively few models examine lake response to reduced nutrient loading, and the majority of these have focused on biological response without characterizing the nature of nonpoint source reductions (Anderson et al. 2005; Dokulil et al. 2005; Jeppesen et al. 2005). Existing empirical trophic status models, for both eutrophication and oligotrophication, often lack either quantitative detail of anthropogenic activity or temporal analysis (Kloiber et al. 2002b; Rast et al. 1983; Silvino et al. 2015). Economic models of water clarity often consider detailed anthropogenic activity, but approach water quality in the reverse of limnology, linking socio-demographic responses to changes in water clarity (Boyle et al. 2003; Boyle et al. 1997; Gibbs et al. 2002; Michael et al. 1996).

There is interest among social and natural scientists to document patterns of land use and understand the effects of these changes on people and natural biota (Matthews et al. 2007; Moore et al. 1996; Sisk 1998), and recognition of the need for interdisciplinary studies on the linkages between land use, ecology, and economics (Ahn et al. 2002). Similarly, there is a need for further modeling of lake oligotrophication with long-term datasets, as these studies as fundamental to understanding the future response of lakes to the confounding effects of climate, reduced nutrient loading, and trophic interactions (Anderson et al. 2005; Schindler 2012). The analysis in this paper fills the recognized need by quantification of the relationships between oligotrophication and the changing patterns of anthropogenic influence, both agricultural and demographic.

This study builds upon previous models by linking observational water quality and anthropogenic activity datasets to examine temporal trends in water quality with decreasing nonpoint source pollution. The lakes selected for this study are currently mesotrophic or oligotrophic and have decreasing agricultural activity in their respective catchments (Day 1963; MDEP 2010; Plantinga et al. 1999; Wescott et al. 1988). This study presents a multi-decadal statistical analysis (1982-2012) of observational water quality and catchment land use data to examine lake recovery. This unique approach integrates water column observations, as an indicator of change in recent lake history, and sociodemographic data that highlights changes in these watersheds over time. Using census records, the analysis is framed through historical conditions of water quality in two lake districts in rural Maine with decreasing agricultural activity. The analysis is refined through individual agricultural census reports to examine relationships between changing land use and water quality over time.

## II. Background

Quantitative models are frequently used in limnology to estimate the responses of inland water bodies to changes in nutrient loading, especially changes due to anthropogenic activities. Empirical modeling of changes in external nutrient loading to lakes is necessary for elucidating the substantial effect of anthropogenic influence on eutrophication. Although current models have various forms and varying degrees of complexity, two basic types of mathematical models have been developed for lake trophic status modeling: statistical and dynamic.

Dynamic models consist of a series of interrelated differential equations which attempt to describe the biological, chemical, and physical reactions and interactions governing aquatic plant growth, including nutrient loads and other driving forces, such as light and temperature. Alternatively, an empirical eutrophication model is a statistical regression that quantifies a basic correlation and does not attempt to account for or characterize every component involved in the eutrophication process. For example, plotting the value of the chosen measure of effect (e.g., algal production) as a function of the factor exerting primary control over the effect (e.g., P load) and determining the line of best fit through the individual points (Rast et al. 1983). Empirical models have been studied in a variety of contexts (Jones et al. 1982; Lee et al. 1978; Rast et al. 1978; Reckhow 1979; Vollenweider 1968, 1975, 1976). Such models can be viewed as statistical approaches relating nutrient inputs and resultant eutrophication related water quality responses to these inputs. They will not provide a detailed description of in-lake nutrients like dynamic models, but rather, are formulated to describe the steady-state or equilibrium response.

A number of empirical studies have been conducted on north temperate lakes using statistical eutrophication models and biological data over time. Kloiber et al. (2002a) developed a regression model that linked satellite brightness data and Secchi disk transparency to relate historical land use to remote sensing data. Using that model, Kloiber et al. (2002b) collected Landsat imagery of approximately 500 lakes in Minnesota to assess spatial patterns and temporal trends in lake clarity. Similarly, a statewide assessment of over 10,000 lakes in Minnesota using Landsat imagery examined spatial and temporal trends in lake clarity across eco-regions (Olmanson et al. 2008). Peckham and Lillesand (2006) identified spatial trends over time by ecoregion and hypothesized changing clarity to be related to changes in land use, zoning, or stream and lake vegetation buffers in Wisconsin. Bruhn (2005) conducted a long-term study of 71 lakes in Michigan using Secchi depth observations data and found a correlation between increased water clarity and residential land use within 100 meters of lakes in the study area and decreased water clarity when wetland cover is within 500 meters of lakes. Doubek et al. (2015) differs by sampling from a single year of data using the 2007 U.S. Environmental Protection Agency's (EPA) National Lakes Assessment dataset to study 236 lakes across the U.S. and examines how lakeshore land use is associated with total phytoplankton and cyanobacterial biovolume.

Comparable statistical models have been conducted in Maine to investigate the relationship between land use and changes in lake water quality. For example, Carter (2012) conducted a study of 25 years of satellite imagery to monitor temporal changes in water quality variables in east-central Maine. The study investigated whether water quality changes are associated with temporal patterns of land use, but found no significant effects in a six-year time series. Other studies of Maine lakes include the prolific work conducted at Colby College, a private liberal arts college located in Waterville, Maine, close to the northern lake district of this study. The Colby Environmental Assessment Team (CEAT) creates an annual report on the ecology of a segment of the Belgrade Lakes system. Past studies have focused two study lakes for this paper; Long Pond and Messalonskee Lake (CEAT 2013; CEAT et al. 1994, 1995; CEAT et al. 1997; CEAT et al. 2006, 2007). The CEAT papers use nutrient loading models to establish present P levels and identify land use types that contribute significant amounts of P to a given study lake for a given study year. The models suggested that the majority of phosphorus loading in the Belgrade system comes from atmospheric inputs, point sources, camp roads, agricultural land, successional land, and shoreline septic use and development.

## III. Methods and Procedures

*Study lakes and lake response variables*

I assembled water clarity data for eight north-temperate lakes in Maine to track changes in their water quality over time (Figure 1, Table 1). Lake water quality data came from databases maintained by the Maine Volunteer Lake Monitoring Program (VLMP) for the study years of 1982 to 2012 at five year increments (VLMP 2014b). The sampling program by the VLMP provided Secchi depth data that follows standard methods with a viewing scope and quality assurance/quality control protocols (Williams 2013). Only data collected in the summer stratified period (June to August) were included to reduce the effects of seasonal variability. Observations were only included from a single sampling point at the deepest spot in each lake to ensure consistency across lakes. For lakes with more than one observation in each year, only the minimum Secchi depth value was used. Secchi depth observations were imputed in missing study years using a partial mean matching univariate imputation (Royston 2004; Young et al. 2010).

**Figure 1.** The study focuses on eight lakes grouped into two lake districts as shown in the south and north panels. In the south lake district, the four lakes are located across three counties (Androscoggin, Cumberland, and Oxford) and in the north lake district, the lakes are located across two counties (Franklin and Kennebec). Catchment boundaries for each of the lakes are denoted by red lines; lake district areas are shown in tan; county lines are shown by black.

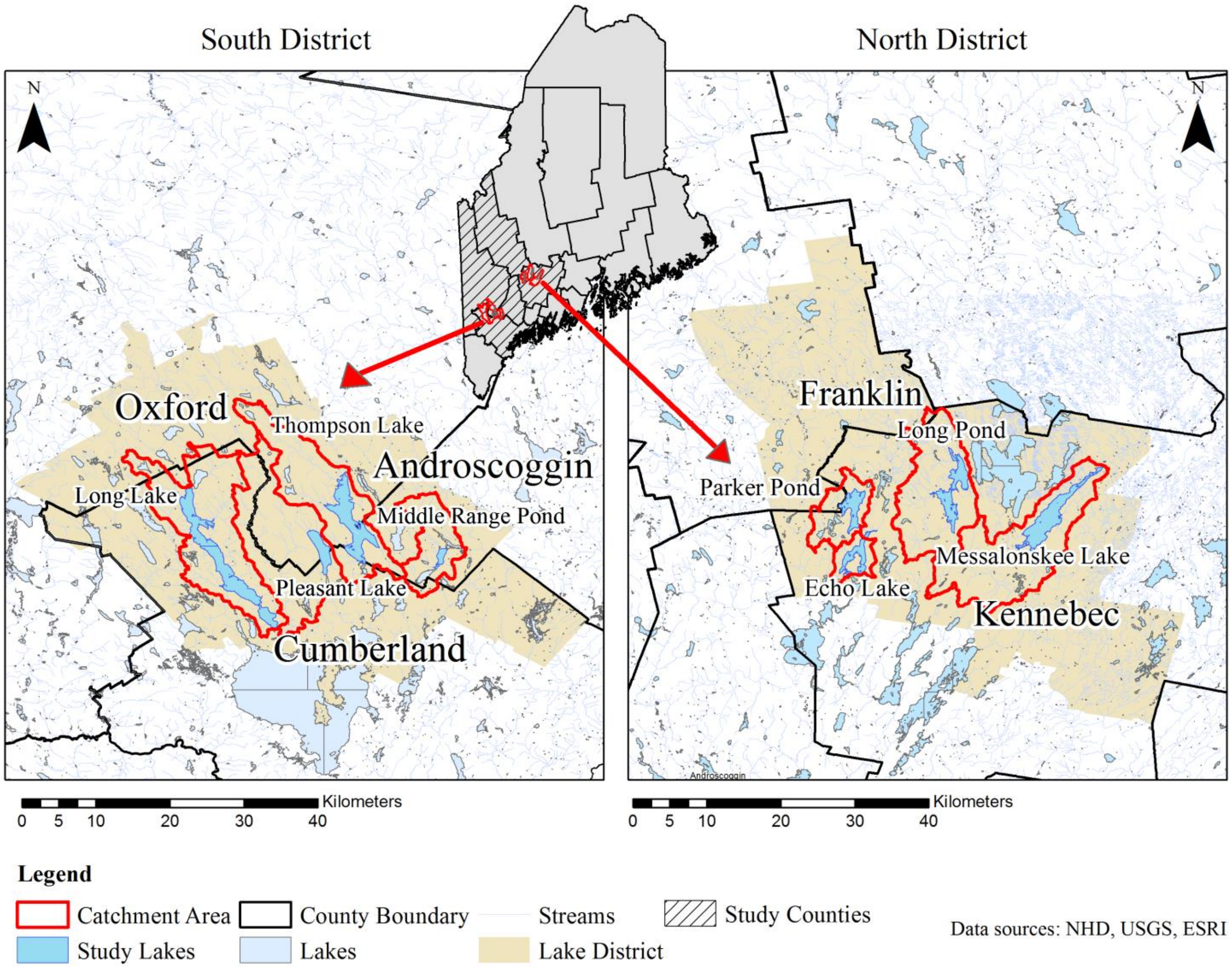


**Table 1.** Summary information for the eight study lakes from the Maine Volunteer Lake Monitoring Program, ordered within districts by residence time.

| Lake Name | Area ($km^2$) | Maximum depth (m) | Total Drainage Area ($km^2$) | Hydraulic Residence Time, years | Trophic Category | Latitude | Longitude | Town(s) |
|---|---|---|---|---|---|---|---|---|
| *North Lake District* | | | | | | | | |
| Long Pond | 10.35 | 32 | 313.4 | 0.3 | mesotrophic | 44.50556 | -69.9142 | Belgrade, Mount Vernon, Rome |
| Echo Lake | 4.49 | 36 | 108.5 | 0.5 | mesotrophic | 44.43977 | -70.0218 | Fayette, Mount Vernon, Readfield |
| Messalonskee Lake | 14.94 | 34 | 458.4 | 0.6 | mesotrophic | 44.47917 | -69.7892 | Belgrade, Oakland, Sidney |
| Parker Pond | 6.17 | 23 | 39.6 | 3.3 | mesotrophic | 44.49422 | -70.0315 | Fayette, Vienna |
| *South Lake District* | | | | | | | | |
| Middle Range Pond | 1.55 | 20 | 23.02 | 1.1 | mesotrophic | 44.02665 | -70.3795 | Poland |
| Long Lake | 21.43 | 18 | 295.3 | 1.1 | mesotrophic | 44.03717 | -70.6554 | Bridgton, Harrison, Naples |
| Thompson Lake | 17.88 | 37 | 122.8 | 3.4 | oligotrophic | 44.06914 | -70.4901 | Poland, Casco, Otisfield, Oxford |
| Pleasant Pond | 5.39 | 19 | 25.5 | 4.2 | oligotrophic | 44.03729 | -70.5225 | Casco, Otisfield |

*Anthropogenic data*

The anthropogenic variables consist of both agricultural and demographic datasets. As agricultural activities are considered to be a major source of nonpoint P input to lakes (Carpenter, Caraco, et al. 1998; Daniel et al. 1994; Sharpley et al. 1999), the anthropogenic variables focus on agricultural intensity in the catchment. Agricultural data came from the U.S. Agricultural Census and consisted of two primary parts: county records from 1790 to 2012 (Haines 2010; NASS 2009, 2014) and farm reports from 1982 to 2012 (DoC 1984, 1989, 1994; NASS 1999, 2004, 2009, 2014). Demographic data was included as an indicator of general watershed activity and came from the county level U.S. Decennial Census (Haines 2010) and the National Bureau of Economic Research's population estimates for intercensal years (Roth 2016).

The anthropogenic data was aggregated at two spatial scales; the lake district scale and the county scale. The lake districts were formed by the natural geographic grouping of lakes (Figure 2). The lake district area extends beyond the catchment areas as it contains any zip code that intersects the lake drainage area. Expanding from the catchment area to the lake district was necessary as the agricultural data are reported at the zip code level. The farm reports included in each lake district were weighted according to the geographic weight of the zip code in the lake catchments and aggregated to create a total value for each metric in each lake district for each study year. The county level data have a similar geographic weighting, according to the proportion of the lake catchments in each county. In the north lake district, the Franklin agricultural metrics are weighted at 3% and Kennebec is weighted as 97%. In the south lake district, Androscoggin is weighted at 13.6%, Cumberland at 50%, and Oxford as 36.4%.

The collected data were slightly modified from the raw reports to correct for missing data and non-response to the Census. For example, prior to 1860, some study counties did not exist so the geographic weighting was adjusted for an approximation of the county area at the time of the census. Farmer non-response to the agricultural Census was corrected for using the weights reported in the census publications for each year. Tabulations using individual farmer reports are subjected to the same rigorous standards for disclosure as the Agricultural Census publications, which created gaps in some metrics (NASS 2014). Missing observations were imputed using a modified linear imputation.

## IV. Empirical Model

*Lake and anthropogenic driver variables*

Lake variables included the physical characteristics of the lake and catchment area that are invariant through time: total drainage area, maximum depth, and hydraulic residence time. These variables have been identified as important to lake trophic status in the literature (Carpenter, Caraco, et al. 1998; Noges 2009; Taranu et al. 2008). An indicator variable was included for the different lake districts to control for unobserved physical or chemical differences in the two watershed groups. The control variables also include an interaction term of lake residence time and the north district indicator. Residence time is an important factor for lake recovery (Bartram et al. 1996), and the north district has statistically different and lower residence times than the southern district so the interaction term shows the differential effect between the two districts.

The anthropogenic variables include indicators of landscape use that can potentially influence lake water clarity (Table 2). Besides the primary regression, a second model employed a lag on the anthropogenic variables to test for time differences between anthropogenic observations and resultant water quality. A reduction in phosphate load can in some cases lead to a rather quick response in water quality (Edmondson et al. 1981; Schindler 1974), but changes in anthropogenic variables may not be immediately evident in lake water quality due to delays in flushing in lakes with long residence times and internal P loading from sediments (Carey et al. 2011; Sondergaard et al. 2001).

The primary concern for external nutrient loading in the study area is from nonpoint pollution from both agricultural and urban sources. Phosphorus loading from agricultural land depends on the water delivery ratio, the water body sensitivity, the land management, and on characteristics of the land (Goetz et al. 2000), but can be modeled through agricultural intensity variables (Schilling et al. 2007; Silvino et al. 2015). Nonpoint source pollution from agricultural lands are generally from excessive fertilizer use on cropland and high-density livestock operations (Carpenter, Caraco, et al. 1998). Poultry and cattle are both common livestock operations in Maine and can have net phosphorus loss through overfeeding of P (Sharpley et al. 1999). Cropland can have detrimental effects on water clarity as farmers tend to apply fertilizer above agronomic rates (Delgado and Bausch 2005; Delgado, Khosla, et al. 2005; Sheriff 2005). The proportion of cropland over total area in the catchment is an indicator for nonpoint source nutrient loss from agricultural watersheds (Jones et al. 2004). Demographic activity can also be influential, even in rural watersheds, so human impact is modeled through total population in the counties intersecting the watersheds, similar to other approaches in the literature (Leone et al. 1993).

**Table 2**. Summary statistics are reported in the table for the anthropogenic variables. Lagged data are reported as a separate group as the data are assumed to have a time trend and removing the oldest year of data should change the reported means. The means of each variable are statistically different across lake districts.

| | mean | count | mean | count | mean | count |
|---|---|---|---|---|---|---|
| *Anthropogenic Variables* | | | *North Lake District* | | *South Lake District* | |
| Total population[++] | 136,835.2<br>(25,207.66) | 1000 | 113,904.6<br>(3802.103) | 514 | 161,087<br>(12,121.9) | 486 |
| Proportion of cropland harvested[+] | 4.081<br>(1.630) | 1000 | 2.626<br>(0.542) | 514 | 5.619<br>(0.740) | 486 |
| Cattle[+] | 259.538<br>(145.643) | 1000 | 359.786<br>(137.226) | 514 | 153.514<br>(43.118) | 486 |
| Poultry[++] | 337,508.7<br>(290,446.9) | 1000 | 208,272.4<br>(358,449.4) | 514 | 474,190.7<br>(37,923.13) | 486 |
| *Anthropogenic Variables, five-year lag* | | | | | | |
| Total population[++] | 135,122.2<br>(23,963.39) | 860 | 113,105.9<br>(3,501.752) | 441 | 158,294.4<br>(10,842.26) | 419 |
| Proportion of cropland harvested[+] | 4.067<br>(1.475) | 860 | 2.756<br>(0.466) | 441 | 5.447<br>(0.720) | 419 |
| Cattle[+] | 272.023<br>(153.100) | 860 | 383.754<br>(134.359) | 441 | 154.426<br>(46.189) | 419 |
| Poultry[++] | 351,682.1<br>(291,604.4) | 860 | 233,458.3<br>(368,351.1) | 441 | 476,113.3<br>(40,685.81) | 419 |

[+]lake district spatial scale, [++]county spatial scale
Data from the USDA Agricultural Census, U.S. Decennial Census

*Statistical Analysis*

The lake response variable (Secchi depth) was modeled with the corresponding lake district to create eight lake series paired in two districts. The statistical model quantifies the anthropogenic variables and explains observed variation in lake water clarity through an ordinary least squares regression model (Robinson 1991; Wooldridge 2015). The model is linear in predictors as not enough data ($N$ = 56) were available to elucidate non-linearity; the sample size was too small to overcome the collinearity of higher order and interaction terms. Each included predictor depicts whether and how the independent variable affects Secchi depth for any of the study lakes. While the variables are aggregated at the lake district, changes in water quality are monitored at the lake level and the model results will show the predictor's impact on the lake. The lake variables (maximum depth, residence time, and drainage area) have problems with collinearity but none could be removed using an F-test for exclusion. The agricultural variables have similar collinearity, but also could not be removed from the model. Additionally, throughout the model building process, after including each independent variable, a likelihood ratio test was used to determine whether the more complex model was significantly better than the simpler model. The loss of precision from the collinear variables was adjusted for as the sample was artificially inflated (to $N$ = 1000) to increase statistical efficiency (Freedman 1981).

The model took a cross-sectional form as the limitations of the sample did not allow for a time series model. Below is the general cross-sectional form of the model, using a vector for classes of variables, one anthropogenic and one for control variables (lake characteristics, years into the study, and lake district):

$$Y = \beta_0 + \beta X + \Gamma C + \varepsilon, \tag{1}$$

where $Y$ is the observed Secchi depth reading for each lake; $\beta_0$ is the intercept across all lakes; $\beta$ and $\Gamma$ are both vectors of coefficients which show the relationship between independent variables and Secchi depth; $X$ is a vector of the anthropogenic variables in the model and $C$ a vector for any control variable; and $\varepsilon$ is the error, where $\varepsilon \sim N(0, \sigma^2)$, and $\sigma^2$ represents the variance in the dependent variable controlling for the variables.

A similar second model, listed below, tested for lagged effects of anthropogenic variables on lake water quality:

$$Y^t = \beta_0^{t-5} + \beta X^{t-5} + \Gamma C + \varepsilon, \qquad (2)$$

where the only difference from the previous model is $X^{t-5}$ is a vector of lagged anthropogenic variables. The model is constructed with five year intervals for anthropogenic data, so in the second model $X^{t-5}$ refers to data five years prior to the Secchi depth reading, where $Y^t$ is a current year reading for Secchi depth.

## V. Results

*Historical Results*

Since the late 1800s agricultural land use has undergone distinct phases of expansion and retraction, reaching a peak of land cleared for agriculture in 1880 with 476,400 acres in the north district counties and 464,700 acres in the southern district counties (Figure 2). The acres in farmland declined gradually for both districts from 1880 through 1945, by only 25% to 360,000 acres in the north and at a slightly higher rate, 40%, to 275,000 acres in the south. Agricultural land fell sharply in both districts for the next few decades from 1945 to the 1970s. In the north, agricultural land decreased by 66% by 1974 to 120,000 acres, but continued a slower decrease to reach 77,000 acres in 2012. In the south, the decline is similar, reaching a leveling point in 1974, but the rate of decrease is larger at 75% and declines to a lower total acreage of 70,000 acres.

The south district also continues to decline in agricultural acreage, but maintains both lower acreage and a lower rate of decrease, reaching 66,000 acres in 2012.

Compared to the consistent decline of agricultural land use, agricultural intensity has an unsteady pattern of increase and decrease. Prior to 1940 poultry was relatively stable in Maine but the broiler industry grew exponentially from 1940 to 1945, with an increase of 80,000 broilers in the north district and 50,000 in the south (Figure 2). By the early 1950s, over 20 million broilers were processed in the state each year. Despite the boom in poultry, Maine followed general agricultural trends of decline in the post-war era. The broiler industry saw collapse in the early 1980s and between 1982 and 2012 the northern district counties went from an estimated 770,000 to 2,000 broilers and layers. In the southern district counties, the change was just as sharp, moving from 650,000 birds to 3,000 in thirty years (Figure 2).

Similar to poultry, Maine dairies both declined and centralized into fewer higher density farming operations, reducing both the number of farms and number of cattle in the lake district counties. In the early 1950s, Maine dairy farms had an average of twenty milk cows, by the mid-1980s it had over eighty. Federal farm programs helped insulate Maine dairy during the postwar era, but their removal ushered in an era of decline, albeit at a much slower rate than poultry (Wescott et al. 1988). Between 1954 and 2007, dairy operations in the north lake district counties have had a steady decline from 11,000 cows to 6,000. The change is similar in the south district counties from 6,600 to 2,700 (Figure 3).

**Figure 2.** Historic agricultural metrics from the USDA agricultural census. The data are aggregated at the county level and weighted according to the geographic weight of each county in the lake catchments.

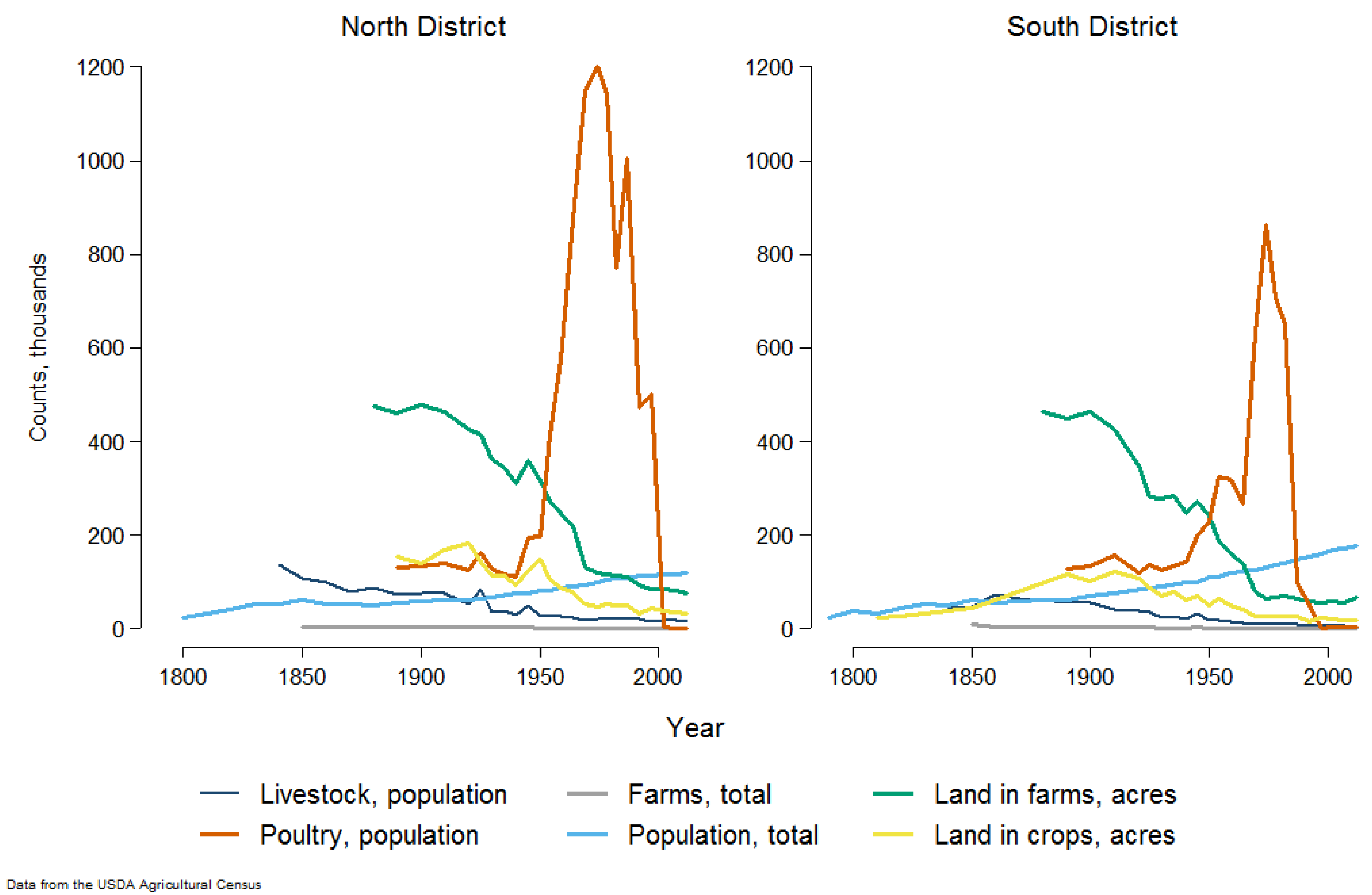
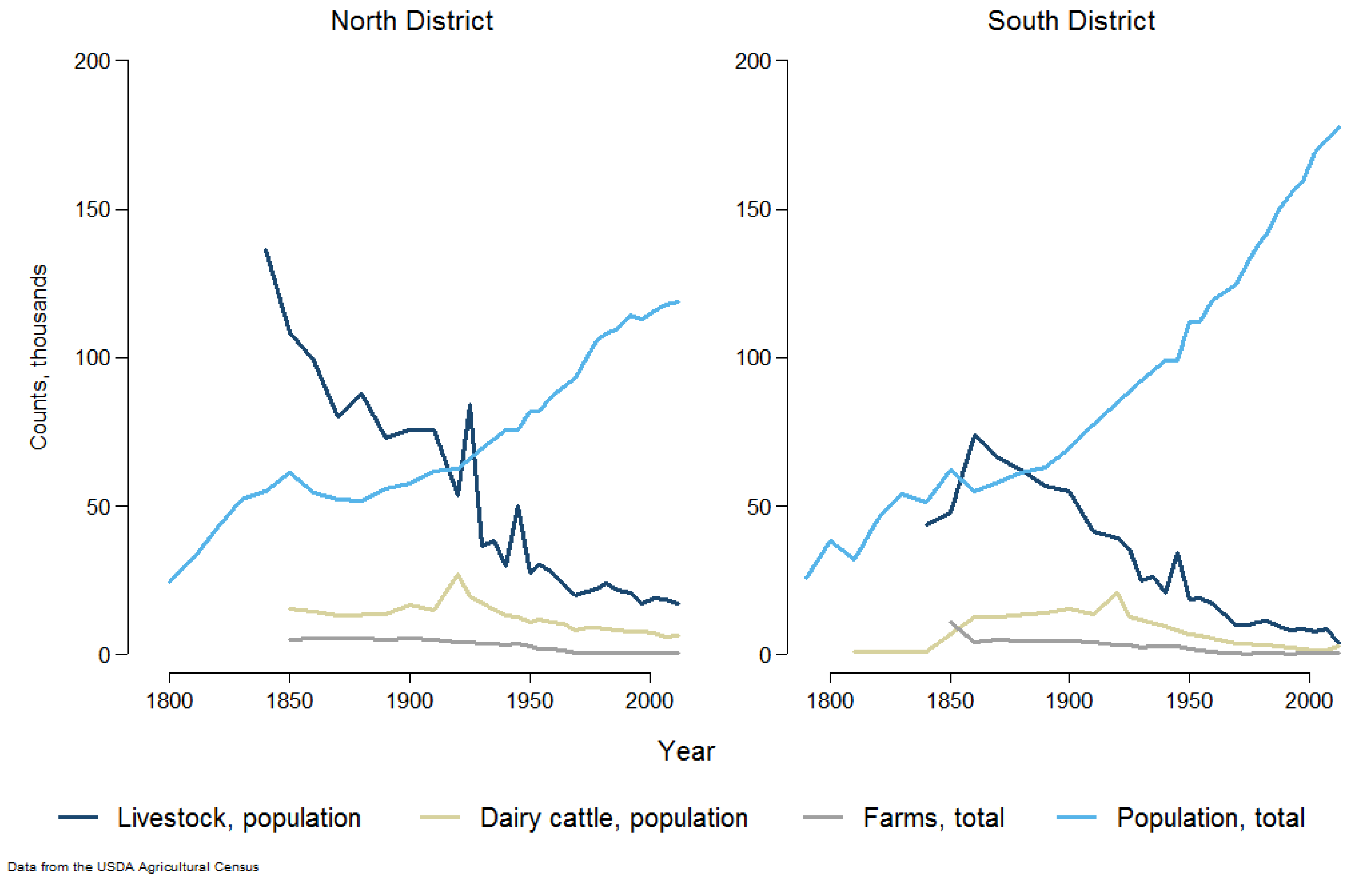


**Figure 3.** Agricultural intensity variables from the Agricultural Census by district. The data are aggregated at the county level and weighted according to the geographic weight of each county in the lake catchments.



*Secchi Depth Trends*

Lakes in each district exhibit differing trends of increasing and decreasing water quality during the study period (Figure 4). The lakes in the northern district have mixed patterns as Parker and Echo appear to have increasing quality while Messalonskee seems to have a decreasing trend. But, the only statistically significant univariate time trend in the north district is Echo Lake's increase in water clarity (Table 4). In the south district Thompson Lake, Pleasant Pond, and Middle Range Pond appear to have a decreasing trend in water clarity whereas Long Lake appears to have an increasing trend in water quality, but none of the linear time trends are statistically significant.

Splitting the lakes intro groups of trending water quality shows relatively similar traits across lakes with increasing and lakes with decreasing Secchi Depth. The lake characteristics of maximum depth, total drainage area, and residence time have similar spreads between the two groupings, with the only major difference in trophic status. Two of the lakes with decreasing water quality are both oligotrophic, as opposed to the remainder with mesotrophic water quality. The two oligotrophic lakes, Thompson and Pleasant, also have the longest residence time of the lakes, some of the deepest Secchi depth readings, and are both in the southern lake district.

**Figure 4.** Secchi depth values used for statistical analysis from the Maine Volunteer Lake Monitoring Program dataset. The line of best fit through each lake panel shows the relationship between water quality in each lake and time.

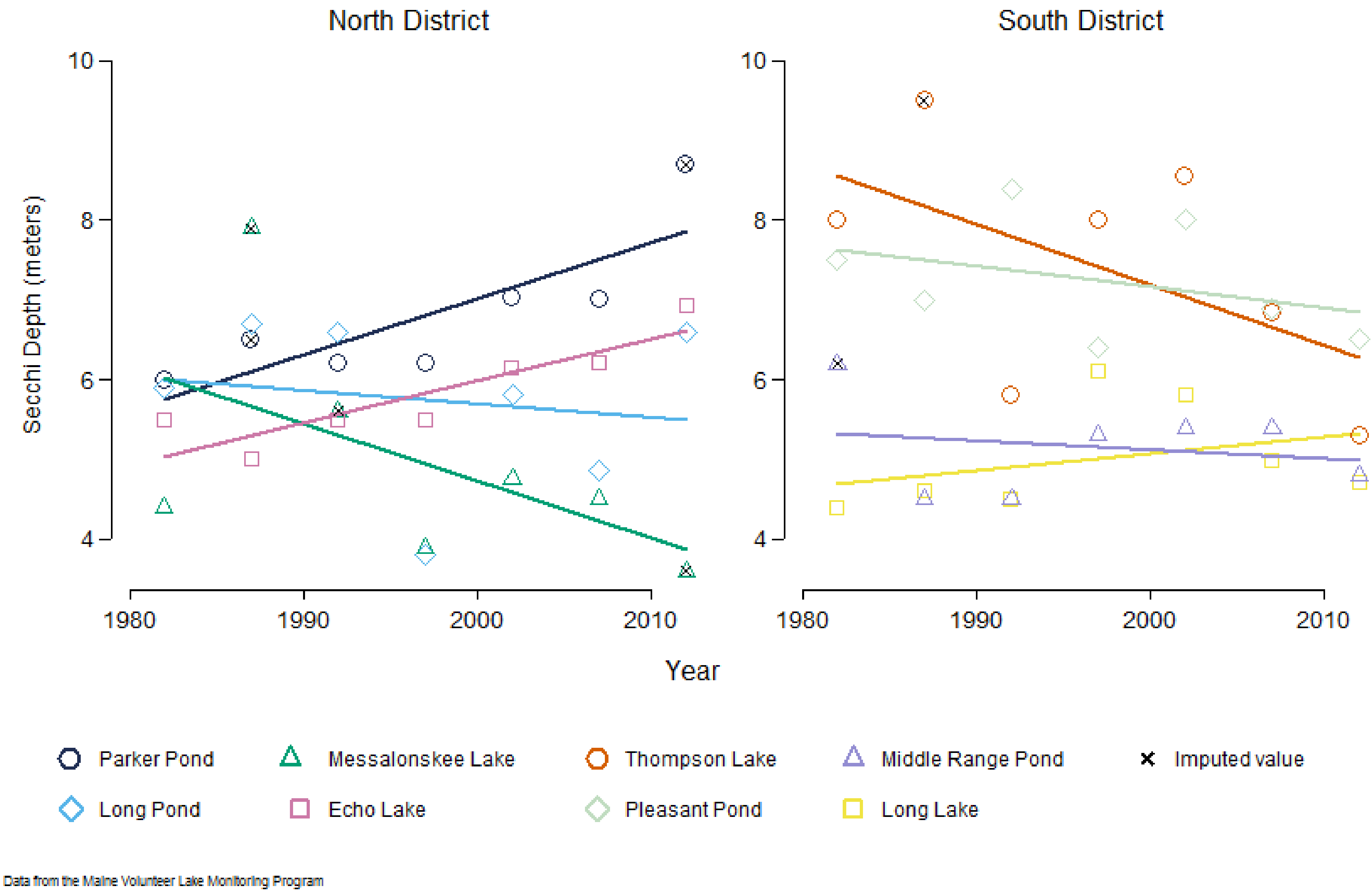


**Table 3.** Coefficient estimates from the univariate line of best fit for the minimum Secchi depth values, separated by district.

*North Lake District*

| | (1) Parker | 2) Long Pond | (3) Messalonskee | (4) Echo |
|---|---|---|---|---|
| Year | 0.0710* | -0.0169 | -0.0718 | 0.0524** |
| | (3.29) | (-0.38) | (-1.42) | (4.22) |
| Constant | -135.0* | 39.42 | 148.3 | -98.87* |
| | (-3.13) | (0.45) | (1.47) | (-3.98) |
| N | 7 | 7 | 7 | 7 |
| $R^2$ | 0.684 | 0.0285 | 0.286 | 0.781 |

*South Lake District*

| | (1) Thompson | (2) Pleasant | (3) Middle Range | (4) Long Lake |
|---|---|---|---|---|
| Year | -0.0761 | -0.0257 | -0.0107 | 0.0213 |
| | (-1.45) | (-0.89) | (-0.43) | (0.81) |
| Constant | 159.3 | 58.59 | 26.55 | -37.49 |
| | (1.52) | (1.01) | (0.54) | (-0.72) |
| N | 7 | 7 | 7 | 7 |
| $R^2$ | 0.295 | 0.135 | 0.0362 | 0.117 |

*t* statistics in parentheses
$^{*}$ $p < 0.05$, $^{**}$ $p < 0.01$, $^{***}$ $p < 0.001$

*Statistical Results*

The overall results between the lagged model and simultaneous model are relatively similar, however, the model fit is much better for the lagged model. Within the models, there are differing patterns of positive and negative effects for the anthropogenic variables. All of the included anthropogenic variables included would be expected to have a negative effect on water clarity, but, the results only indicate that two of the model variables have a negative effect. The coefficients reported in Table 4 are the results of the statistical model and are the change in water clarity (in meters) given a one-unit change in the variable, holding all else equal. The coefficient results show the trends of water quality aggregated across all lakes and negative effects indicate a decrease in water clarity whereas positive results indicate increasing water clarity.

While the majority of anthropogenic metrics are decreasing over time, total population has increased. The result suggests that holding all else equal, an increase in population in the county area by 10,000 people would reduce Secchi depth transparency by 0.26 meters in the non-lagged model and by 0.76 meters for the lagged model. Contrastingly, the proportion of cropland harvested and poultry have been decreasing over time, but show positive effects on Secchi depth. The proportion of cropland harvested has a pronounced positive effect as an increase of one percentage point of the proportion of cropland harvested in the lake district would increase Secchi depth by 0.16 meters in the simultaneous model and by 0.40 meters in the five-year lag model. The poultry variable is similarly positive with an increase of 1,000,000 poultry in the weighted county area causing increase in Secchi depth by 1.3 meters for the simultaneous model and 0.86 meters for the lagged model. However, the magnitude of the poultry coefficient is small enough to be not meaningful ecologically.

The included control variables account for the differences between lakes beyond the changing anthropogenic metrics and. The control variables are only statistically significant for the lake characteristics, but two have relatively small effects, maximum depth and total drainage area. An increase in lake depth by one meter should increase lake clarity by 0.03 meters and an increase in the total drainage area by one $km^2$ should decrease water clarity by 0.001 meters.

While the model variables are statistically different across the lake districts, the lake district indicator variable is not statistically significant. However, the interaction term between the lake district and the lake residence time is significant and has a negative effect on Secchi depth. For lakes in the north district, the residence time variable will have an additional negative effect of -0.18 meters in the simultaneous model and -0.14 meters in the lagged model. The residence time variable is positive and significant for both models, but due to the interaction term will be slightly lower for the northern district but will still have a positive effect of 0.48 meters in the simultaneous model and 0.55 meters in the lagged model.

**Table 4.** Coefficient results from the statistical model with t-statistics in parentheses.

| | (1) No Lag | (2) 5 Year Lag[0] |
|---|---|---|
| *Anthropogenic Variables* | | |
| Weighted population in county, ten thousands | -0.264* (-2.51) | -0.764* (-2.54) |
| Proportion of cropland harvested[1] in lake district | 0.160** (3.03) | 0.389*** (4.80) |
| Cattle in lake district | -0.00432*** (-5.92) | -0.000356 (-0.23) |
| Poultry in county, millions | 1.297*** (5.97) | 0.861*** (4.50) |
| *Control Variables* | | |
| Year | 0.00266 (0.18) | 0.0666 (1.75) |
| Maximum Depth (m) | 0.0349*** (5.46) | 0.0309*** (4.04) |
| Total Drainage Area ($km^2$) | -0.00158*** (-6.54) | -0.00104*** (-3.75) |
| Hydraulic residence time, years | 0.656*** (28.00) | 0.685*** (22.84) |
| Northern lake district | 0.964 (1.67) | -1.791 (-1.06) |
| North district and residence time interaction | -0.179*** (-3.70) | -0.140* (-2.28) |
| Constant | 7.348*** (5.54) | 12.67*** (3.46) |
| N | 1000 | 861 |
| AIC | 2608.8 | 2328.4 |
| BIC | 2662.8 | 2380.8 |
| $R^2$ | 0.561 | 0.527 |

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

[0] 5-year lag refers to a lag in the anthropogenic variables

[1] Ratio of cropland harvested to total farmland for each farm in district

## VI. Discussion

In the anthropogenic variable results, the total population variable is of note for its ecological significance as the only anthropogenic variable with an increasing trend in the historical dataset. Increases in population show a decrease in Secchi depth, or a reduction in water clarity, consistent with the general literature (Leone et al. 1993; Silvino et al. 2015). Increases in human activity from an expanding population would likely include increases in septic use, human interaction with the watershed, and infrastructure use (roads and camps). However, forward projections using the model results are likely to be an overestimation of the effects of population as the model assumes that institutions remain constant throughout the analysis. Yet, there have been changes in lake management for nonpoint source pollution through Maine lakeside zoning laws, EPA rulings, and the presence of Maine lake associations (CEPG 2010; Dennis et al. 2014; Hardy et al. 2008; MDEP 2006). The extent to which these institutions manage external nutrient loading to lakes will determine the total impact of population in the future.

As opposed to the demographic variable, the agricultural variables have all been decreasing over time, but the only variables with ecological significance are cropland and cattle. Aggregated across all lakes, the cropland variable shows a positive effect on water quality after accounting for the other agricultural and lake control variables, contradictory to the literature (Carpenter, Caraco, et al. 1998). An increase in row crops in the catchment would be expected to decrease water clarity from fertilizer runoff or lack of riparian buffers (Goetz et al. 2000). The positive coefficient in the model likely occurs because the variable does not capture alternative land uses. The CEAT et al. (2007) study shows that both land transitioning from farmland to woodland and residential development can have a largely negative impact on water clarity, and a greater impact than row crops, in the Belgrade lakes system. If cropland converts to either transitional land or into residential use, an increase in cropland could increase water clarity as it has lower nutrient export than the alternative uses. While poultry is not ecologically significant, it does have the opposite sign as would be expected from the literature (Edwards et al. 1992). The positive effect is likely due to the effects of spatial scale in the analysis as poultry is tabulated at the county level and changes in the larger area are not necessarily represented in the watershed.

The most interesting control variable result is the lake residence time as it affects Secchi depth opposite of its expected value. The hydraulic residence time variable has a positive sign and indicates that water clarity increases with a one-year increase in the residence time, contrary to the literature (Paerl et al. 2008). Increasing residence time should decrease water clarity as cyanobacteria favors higher residence time (Elliott 2010). The positive sign on residence time is partially driven by the difference in residence time between the two districts. The significant and negative interaction term of lake district and residence time reflects the relationship between the statistically shorter residence times in the northern district and the higher proportion of mesotrophic lakes. Similarly, the variable is likely also driven by the differences between lake residence times and trophic status. The only two oligotrophic lakes have the two longest residences times of any lake. Even though Thompson lake and Pleasant lake appear to have decreasing water quality, their most recent trophic status was still oligotrophic, compared to the rest of the lakes with a mesotrophic status (VLMP 2014a). The residence time effect is likely driven by the correlation between higher residence times and higher water quality in the study lakes.

Nutrient loading effects in lakes are strongly driven by hydraulic residence time (Anderson et al. 2005; Soendergaard et al. 2005), which supports a model with lagged anthropogenic variables. Nutrients entering a lake from anthropogenic land use can stay in the lake until they are flushed downstream. The lagged model has a better model fit than the simultaneous model considered in Table 5, consistent with expectations from the literature (Edmondson et al. 1981). However, the five-year lag is longer than the residence time for any of the study lakes, ranging from 0.29 years in Long Pond to 4.2 years in Pleasant Lake (Table 1). The lagged model fit is likely due to more than just residence time. Since the agricultural data is collected as an aggregate of all activity in the lake district, it does not contain information about how close the origin of the external nutrient load is to the lake. If farms in the watershed are far from the lake, there will be a time lag for sediment to move to the deepest part of the lake.

While some of the lakes have particularly long residence times, such as Parker Pond, Thompson Lake, and Pleasant Pond, the remainder have residence times of 0.3 years to slightly over one year. The spatial distribution of external nutrient loading in the catchment is unlikely to account for all of the difference between the lag and the residence time. The results suggest that changes in land use may have effects on water quality that last for far longer than would be expected. Sediment-water column interactions may cause a lake to maintain a lower state of water clarity even if external nutrient load decreases (Carey et al. 2011). Internal loading of phosphorus can delay lake recovery after a phosphorus loading reduction (Soendergaard et al. 2005). The lagged model fit reflects the necessary time for a lake to recover after a change in external nutrient load.

*Limitations*

Spatial homogeneity causes the majority of limitations in this analysis. Summing the weighted zip code measures gave an approximation of agricultural activity in the watershed but assumes perfectly homogenous agricultural activity across the watershed, which can lead to a skewed analysis. The model results indicate that spatial concerns are more prevalent as scale increases to the county level where homogeneity is a much less likely assumption to hold. However, the county level data could not have been refined further with the available data.

The spatial layout and yearly use patterns of residential homes in Maine can significantly impact lake water quality, though these were not included in this study. Lakeside residential homes can particularly exacerbate excess nutrient loads to lakes and year-round homes pose more of a threat to lakes through extended use of septic systems. When improperly designed or used, septic systems found at year-round or seasonal homes can be potentially large sources of nutrients (CEAT et al. 1998). The model does not differentiate between forms of human activity and instead aggregates all impacts into the total population variable. While this allows for a broader view of anthropogenic impact in the lake area, it neglects to differentiate the more influential changes in shoreline development. The homogeneity of spatial analysis likely impacted the results if lakes in each cluster have different shoreline development patterns.

The configuration of lakes within the district can influence nutrient loading as chained lakes influence those downstream, which was not taken into account here. The study examined land use in the Belgrade lakes chain, which includes Messalonskee Lake and Long Pond, with Messalonskee at the bottom of the Belgrade chain with Long Pond right above. Lakes at the end of a chain will have their respective phosphorus input and transparency influenced by the lakes above them. The preceding lake in the chain acts as a point-source pollutant that is not accounted for in the model. The influential watershed would give more detail into determining clarity rather than the direct drainage area. Aggregation of catchments accounts for the influential watershed in jointly chained lakes, like Long Pond and Messalonskee, but does not hold for the other chained lakes like Middle Range Pond.

## VII. Conclusions

This study is unique in linking long term observational datasets as existing temporal studies of water quality and anthropogenic land use generally focus on percent of impervious surface, but do not include intensity of land use, such as population or livestock (Carter Courville 2012; Kloiber et al. 2002b), which is important for analysis of nutrient input to a lake. Contrastingly, studies that examine intensity of land use often do not include temporal data (Matias et al. 2005; Silvino et al. 2015). The few studies that include both anthropogenic intensity and biological response in a temporal analysis often do not use observational data but rather employ simulations under policy changes or projections from current trends (Burkart et al. 2012; CEAT et al. 1997; CEAT et al. 2007). This study is unique in quantitatively linking anthropogenic intensity of land use and Secchi depth observations as a visual measure of water quality outcomes.

The specific relationships developed in this study between anthropogenic variables and lake response are largely consistent with expectations in the literature, however the lagged model gave unexpected results with its model fit. While lagged lake response models are used in the limnology literature (Edmondson et al. 1981), they have not been implemented in any linked anthropogenic land use and limnology papers. The second model, which employed a lag by correlating Secchi depth observations with anthropogenic activity five years prior, had a significantly better model fit than the model of same-year Secchi depth observations and anthropogenic activity. However, a five-year lag is significantly longer than the residence times for any of the study lakes and indicates that changes in land use may have long-term effects on water quality that last far longer than would be expected given the hydraulic residence time. Thus, the recovery period for lakes following a reduction in external loading will likely be longer than the residence time. Future work on oligotrophication will need to consider this expanded timeline for lakes to recover from external nutrient loads.

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