Spring Fever? Climate Change And Water Quality In Our Local Lakes

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Abstract

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Eutrophication, a state in which a body of water becomes over-enriched with nutrients, remains one of the biggest water quality problems both in the United States and globally. Although nutrient pollution that causes eutrophication is regulated in the United States under the Clean Water Act (CWA), eutrophication remains a significant threat. Continued land-use change, internal regeneration of nutrients, and climate change all continue to contribute to nutrient loads that lead to eutrophication. This thesis is comprised of two chapters that aim to address the interplay between external and internal nutrient sources in lakes in the face of climate change. In the first chapter I examine how climate change impacts the internal nutrient cycling in two neighboring lakes in western Washington (USA). I found that climate variability can drive the internal regeneration of phosphorus in lakes that exhibit summer anoxia. This implies that climate change has the potential to increase nutrient loads in highly productive lakes, thereby increasing the risk of eutrophication in these systems in spite of control on external nutrient inputs. In the second chapter, I review the Clean Water Act and the policy tools implemented under it by which we manage external nutrient inputs. I found that the Clean Water Act is very good at addressing easy-to-identify point sources of nutrient pollution, but is much less effective at addressing diffuse, nonpoint sources. These results suggest that in order to maintain acceptable water quality in the future, we may need to be more aggressive with nonpoint source pollution that we currently are, particularly for productive lakes that exhibit summer anoxia. This information may also help managers allocate resources for nutrient control and identify lakes that are particularly at risk of eutrophication into the future.

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Overview

Eutrophication, a state in which a body of water becomes over-enriched with nutrients, remains one of the biggest water quality problems both in the United States and globally. Eutrophication impacts aquatic systems in a number of undesirable ways, such as decreasing species diversity, increasing the risk of toxic algae blooms, and decreasing fish habitat availability (Smith & Schindler 2009). These ecological impacts also have human implications, as the side-effects of eutrophication can decrease the aesthetic, recreational, and economic value of aquatic systems (Carpenter et al. 1998).

In the United States, nutrient pollution that causes eutrophication is regulated under the Clean Water Act (CWA). Since the implementation of the CWA, we have been able to reduce external nutrient loading in lakes and other bodies of water and significantly improve water quality (Conley et al. 2009). However, eutrophication remains a significant threat. Continued land-use change, internal regeneration of nutrients, and climate change all continue to contribute to nutrient loads that can lead to eutrophication (Jeppesen et al. 2009).

This thesis is comprised of two chapters that aim to address the interplay between external and internal nutrient sources in lakes in the face of climate change. In the first chapter I examine how climate change impacts the internal nutrient cycling in two neighboring lakes in western Washington (USA). I found that climate variability can drive the internal regeneration of phosphorus in lakes that exhibit summer anoxia. This implies that climate change has the potential to increase nutrient loads in highly productive lakes, thereby increasing the risk of eutrophication in these systems in spite of control on external nutrient inputs. In the second chapter, I review the Clean Water Act and the policy tools implemented under it by which we manage external nutrient inputs. I found that the Clean Water Act is very good at addressing easy-to-identify point sources of nutrient pollution, but is much less effective at addressing diffuse, nonpoint sources. These results suggest that in order to maintain acceptable water quality in the future, we may need to be more aggressive with nonpoint source pollution that we currently are, particularly for productive lakes that exhibit summer anoxia. This information may also help managers allocate resources for nutrient control and identify lakes that are particularly at risk of eutrophication into the future.

Chapter 1:

Trophic Status Regulates the Internal Response of Oxygen and Phosphorus to Climate Change in Lakes

Abstract

While the phenological impacts of climate change in lakes are well described, the biogeochemical implications of changing phenology continue to be poorly understood. Because hypolimnetic oxygen depletion enhances internal regeneration of phosphorus (P) and is controlled by trophic state, we hypothesized that oxygen depletion and internal P loading would be more sensitive to climate change in productive lakes than in those with lower nutrient loads. Using a 20-year dataset, we investigated the relationships between the timing of spring stratification and subsequent summer oxygen and P concentrations in the hypolimnia of two monomictic lakes in western Washington, USA. In productive Lake Sammamish, early onset of stratification led to early onset and longer duration of hypolimnetic anoxia, as well as higher P concentrations by late summer. In slightly less productive Lake Washington, hypolimnetic oxygen and P concentrations were not affected by the timing or duration of thermal stratification. Our results suggest that water quality in eutrophic lakes will likely be more sensitive to the cumulative impacts of climate warming than lakes with more moderate nutrient loads, and that water quality management efforts may require additional measures in a warmer future.

Introduction

Ecosystem response to climate change depends critically on the interactions between global climatological drivers and the local physical, chemical, and biological features of ecosystems. Changing local temperature and precipitation patterns caused by global climate regimes interact with more localized human-driven environmental stressors such as eutrophication, habitat degradation, flow modification, resource exploitation, and species invasions to affect the specific responses of ecosystems to changes in the environment (Woodward et al. 2010; IPCC 2007). It is reasonable to expect that climate change confounds or amplifies local perturbations to ecological structures and function, though these interactions remain poorly understood for most ecosystems.

The cumulative effects of globally driven climate changes- and local-scale human-driven change are particularly prevalent in lakes and reservoirs, as these systems are highly responsive to watershed and atmospheric inputs (Schindler 2009). Physical features such temperature, precipitation, and wind regulate chemical and biological processes in lakes, so changes to these physical forcing variables can substantially alter community structure and ecosystem functioning. Climate change been shown to impact nutrient loading by changing hydrology (Jeppesen et al. 2009; Nõges et al. 2011), food web dynamics (Kosten et al. 2012; Hill & Magnuson 1990; Elliott et al. 2006), and the timing and duration of thermal stratification (Hambright et al. 1994; Winder & Schindler 2004b).

Arguably the most thoroughly described responses of lakes to changing climate regimes has been in the phenology of a variety of ecological processes (Meis et al. 2009; Winder & Schindler 2004a; Winder & Schindler 2004b; Thackeray et al. 2010; Walters et al. 2013; Feuchtmayr et al. 2012). In particular, it is generally appreciated that spring thermal stratification is occurring earlier in the year due to warmer winter and spring air temperatures (Peeters et al. 2007; Winder & Schindler 2004b; Adrian et al. 2009; Straile 2002).

Previous studies of changes in the phenology of stratification patterns have emphasized effects on the timing of plankton blooms (Meis et al. 2009; Feuchtmayr et al. 2010), on de-coupling of predator prey relationships (Winder & Schindler 2004a), and on changes in hypolimnetic oxygen dynamics in lakes (Wilhelm & Adrian 2008; North et al. 2014; Foley et al. 2012). However, the extent to which changes in thermal stratification phenology translate into changes in biogeochemical cycling remains poorly understood.

Oxygen depletion regulates the internal regeneration of phosphorus from sediments through its control on redox potential, which can limit the effectiveness of water quality measures focused on limiting external phosphorus loads (Filbrun et al. 2013). Phosphorus is the major limiting nutrient in many lakes, and managing the concentrations of this nutrient is critical to maintaining water quality (Søndergaard et al. 2003; Nürnberg 1984; Schindler 1977; Smith & Schindler 2009). Functional linkages between the climate-driven dynamics of dissolved oxygen and phosphorus regeneration from sediments are poorly described but are a likely mechanism that produces important ecosystem responses to climate change in some lakes.

We quantified how variability in the timing of spring stratification controls biogeochemical cycling in two large urban lakes with differing trophic status. In particular, our goals were to describe the interannual variation in the timing of spring thermal stratification, and the association of stratification timing with hypolimnetic temperatures, summer oxygen concentrations, and total phosphorus concentrations in two lakes. One lake has a sufficiently high phosphorus load that it experiences summer anoxia while the other does not. We expect that the more productive lake will develop extended periods of anoxia in years when stratification establishes earlier in the spring, thereby producing a longer period of time during which phosphorus can be regenerated from hypolimnetic sediments; phosphorus regeneration the lakes that does not develop anoxia should be relatively insensitive to the timing of thermal stratification.

Materials and methods

Study area

Lake Sammamish and Lake Washington are large, mesotrophic lakes located in the greater Seattle area of Washington, USA (47°36′35″N, 122°19′59″W). Lake Sammamish is located less than 12 km east of Lake Washington. Neither lakes freeze; the water columns of both lakes are fully homogenous during the winter but develop strong thermal stratification during the summer. During the stratified period, the hypolimnion of Lake Sammamish becomes anoxic in response to a higher phosphorus load and shallower depths, which allows for greater interaction at the sediment-water interface (Welch et al. 1980). Lake Washington generally does not exhibit hypolimnetic oxygen concentrations below 6 mg/L (Cerco et al. 2006).

Both lakes have undergone several ecological changes in the last century due to human activities. During the 1940s – 1960s, phosphorus levels were at a peak due to secondary sewage effluent, which caused eutrophication in both lakes (Edmondson 1994; Welch et al. 1980). The effluent was diverted in 1968 and by 1975 Lake Washington was considered fully recovered and has since exhibited stable nutrient inputs (Edmondson 1994). Lake Sammamish did not recover as quickly; the delayed response to restoration in this lake is well documented and has been attributed to internal regeneration of phosphorus (Welch et al. 1980; Shuster et al. 1986; Welch 1985) though differences in land-use between the two watersheds may translate into a higher P load to Lake Sammamish. The major morphometric characteristics of both lakes are summarized in Table 1.

Field methods

Physical, chemical, and biological data on Lake Sammamish and Lake Washington used in this analysis were collected by the Water and Land Resources Division of King County (KC) as part of their Major Lakes monitoring program. KC has been collecting data on Lake Washington and Lake Sammamish since 1974, however only since the early 1990s were data collected regularly throughout the year (Frodge 2005). For this reason, only data from 1993 through 2012 were used for this study.

KC monitoring efforts involve monthly sampling throughout the year and often bimonthly sampling between the months of March and October to coincide with the thermally stratified season. In each lake, temperature and dissolved oxygen (DO) profiles were recorded at specific depths using a Hydrolab Datasonde, and water samples were collected throughout the water column to characterize nutrient conditions. Water samples were analyzed for total phosphorus (TP) using the automated ascorbic acid method after manual persulfate digestion (Frodge 2005). In Lake Washington, data were collected every 10 meters between November and March and every 5 meters between April and October. In Lake

Sammamish, data were collected at 1m, 10m, and 25m between November and March and at every meter between April and October. Date for this study were drawn from Lake Washington KC site 0852 (47°38’N, 122°16’W) and Lake Sammamish KC site 0612 (47°35’N, 122°05’W). These two sites were chosen based on their placement in the deepest basin of each lake, thereby limiting the effects of inflow and mixing from shoreline waves.

Analysis

We defined the hypolimnion as the bottom 30% of each lake by volume (30-60m in Lake Washington and 15-25m in Lake Sammamish). Daily averages for temperature and DO were calculated through linear interpolation between sampling dates. The onset date of stratification was defined as the first day of a temperature difference $\geq$ 1°C between the average temperatures of the epilimnion (surface to 10.5m) and the hypolimnion.

The onset date of anoxia was defined as the first day at which the hypolimnetic volume-weighted mean (VWM) DO concentration was $\leq$ 2mg/L. The duration of summer anoxia was defined as the number of consecutive days of anoxia. Annual summer maximum hypolimnetic TP concentrations were identified using the VWM TP for the hypolimnion of each lake.

TP data were assessed for normality with the Shapiro-Wilkes test and log-transformed. Visual residual diagnostics were used to remove outliers. Least squares linear regression was used to examine the relationship between the date of onset of anoxia, duration of anoxia, and maximum summer TP concentrations in each lake as a function of stratification date. We also used least squares linear regression to test whether the variation in summer TP values was a function of the date and duration of anoxia in Lake Sammamish.

Given the uncertainty in calculating the onset dates of stratification and anoxia through interpolation, we tested the robustness of our analyses via a Monte Carlo approach. The two sampling dates between which stratification and onset of anoxia occurred were set as the range of potential dates from which data could be drawn. We used a moderately informative beta probability distribution to randomly generate 10,000 datasets of the possible date of initial stratification for each year. We re-parameterized the beta distribution using a mode parameter (m), expressed as the calculated date of stratification per year, and a K parameter, which represents the level of confidence that the estimated date is the true date of stratification (Figure A2). The standard shape parameters for the beta probability distribution, α and β, are calculated based on the K-value and m for each year:

$$\alpha = m(K-2) + 1 \qquad \beta = (1-m)(K-2) + 1$$

Using a beta probability distribution allowed for a higher probability that dates close to the dates calculated via linear interpolation would be selected from the range of potential dates for each year in the Monte Carlo process (Figure A1).

We used a conservative K value of 5 to generate pseudo datasets from the beta probability distribution. This K values allowed for a high probability that values close to the calculated date of stratification would be selected, while still allowing the potential for any date within the interval to be selected (Figure A2). For each of the pseudo datasets, we fit least-squares linear regressions for the date of onset of anoxia, duration of anoxia, and maximum summer TP concentration. We then compared the distribution of estimated slope and r^2 values.

Results

Lake Sammamish is a smaller, shallower lake than Lake Washington, and also exhibits much more seasonal variability in key limnological parameters (Figure 1, Table 2). For all years but one, Lake Sammamish stratified before Lake Washington (Figure 2). During the twenty year study period, both lakes had a 41-day range of stratification onset dates – in Lake Washington, stratification occurred between March 21st and May 1st, with a mean stratification date of April 9th (sd = 11.3 days); in Lake Sammamish, stratification occurred between March 14th and April 24th, with a mean stratification date of April 2nd (sd=14.3 days). The mean stratification date of Lake Sammamish was significantly earlier than that of Lake Washington (t-test, p=0.02). Over the course of the study period, both lakes exhibited a trend towards later stratification. The onset of thermal stratification in Lake Washington was delayed 1.25

days/year (r^2 = 0.43, p = 0.002), and onset of stratification in Lake Sammamish was delayed 1.06 days/year (r^2 = 0.33, p = 0.007) between 1993 and 2012.

Hypolimnetic DO concentrations were more variable in Lake Sammamish than in Lake Washington (Figure 1b). In Lake Sammamish, by June, hypolimnetic DO was less than 7 mg/L (a level unsuitable for cold water fish) and in September and October the hypolimnion was anoxic (<2 mg/L). For all years of the study, once the DO fell below 2mg/L, it remained anoxic until autumn turnover. The date of onset of anoxia in Lake Sammamish ranged between July 29 and September 19, with a mean onset date of August 27. This date was weakly but positively correlated to the date of stratification – earlier stratification led to earlier onset of anoxia (linear regression, slope=0.60, p=0.05, r^2=0.19) (Figure3a). The duration of anoxia ranged between 36 and 102 days (mean = 68 days). This number was negatively correlated to the date of stratification – earlier stratification led to a longer period of anoxia (linear regression, slope=-0.88, p = 0.02, r^2 = 0.27) (Figure 3b). Lake Washington DO concentrations only dropped below 7 mg/L very briefly, and never dropped below 6 mg/L (Figure 1b).

For much of the year (November – August), hypolimnetic total phosphorus levels in Lake Washington and Lake Sammamish were fairly similar, averaging 17.8 µg/L in Lake Washington and 17.40 µg/L in Lake Sammamish. In autumn however, the TP in Lake Sammamish showed substantial interannual variation (Figure 1c). September/October mean hypolimnetic TP (25.96 µg/L) was significantly higher (t-test, p= 0.001) than during the rest of the year, as well as significantly higher (p=0.003) than September/October mean hypolimnetic TP in Lake Washington (mean = 19.06 µg/L). In Lake Washington, there was no difference between mean hypolimnetic TP in September/October compared to the rest of the year (t-test, p=0.34).

The seasonal maximum hypolimnetic TP in Lake Washington was not sensitive to date of thermal stratification (linear regression, p=0.24) (Figure 4a). In Lake Sammamish, however, seasonal maximum hypolimnetic TP was significantly higher with earlier onset of stratification (linear regression, slope = -011, p=0.005, r^2=0.37) (Figure 4b). The Monte Carlo analyses for the relationship of maximum TP to stratification date were nearly identical to standard linear regression results, demonstrating that this relationship was robust to uncertainties in assigning a precise date to these phenological events (Figure 5).

Earlier onset of anoxia in Lake Sammamish led to higher hypolimnetic TP values in late summer (linear regression, $p = 0.02$, $r^2 = 0.26$) (Figure 6a). Similarly, maximum summer hypolimnetic TP was positively correlated to the duration of the anoxic period – longer periods of anoxia led to higher TP values (linear regression, $p = 0.02$, $r^2 = 0.26$) (Figure 6b). These relationships were logarithmic such that longer periods of anoxia lead to exponentially higher TP concentrations than shorter periods.

Discussion

Shifts in the phenology of thermal stratification have considerable impacts on the duration of hypoxia and the internal regeneration of phosphorus in productive lakes, but the magnitude of these impacts may vary depending on a lake's trophic status and morphometry. Summer TP in the hypolimnion of Lake Sammamish was highly sensitive to both timing of the anoxic period of the lake and onset date of stratification indicating that a) the period of anoxia in productive lakes responds to climate changes and b) it is through this mechanism that climate change may affect internal phosphorus loading of lakes. The lack of the sensitivity in the hypolimnetic TP of Lake Washington to changes in stratification suggests that internal P loading in lakes without hypolimnetic oxygen depletion are much less sensitive to climate-driven changes in thermal stratification.

During our study period, both lakes exhibited a range of stratification onset dates with over one month of variability. However, both lakes demonstrated the same overall trend of later stratification date over the twenty year period. This was somewhat surprising, as previous studies have shown that Lake Washington and other lakes around the word are experiencing earlier onset of stratification (Hambright et al. 1994; Winder & Schindler 2004a; Adrian et al. 1995). However, surface lake temperatures demonstrate responses to local meteorological phenomena (George et al. 2000; Arhonditsis et al. 2004; Livingstone 2008). Our current understanding of large scale climate patterns suggests that the PDO shifted towards a cool phase in 1999 or 2000 (Trenberth & Fasullo 2013). This may be an explanation for the observed trend in stratification, as surface water temperatures and thus spring mixing patterns would reflect this regime shift.

Winder and Schindler (2004b) found that over the course of a 40-year period (1962 – 2002), spring stratification in Lake Washington shifted earlier by about 16 days. This trend was dominated by a regime shift in the PDO that occurred in the middle of the study period. Because this study used only a twenty-year dataset, short term and local weather patterns are likely to drive the variability in stratification more than long term, global climate regimes.

A current theory in climate and lake interactions suggests that physical processes in lakes are ultimately driven by large-scale climate forcing, and that variability in lake response to these signals is a function of “local noise” (Livingstone 2008). We interpret this somewhat differently whereby large-scale climate forcing is expressed through local ‘filtering’ that is a function of watershed and lake characteristics. George *et. al* (2000) found this to be true in a study of five lakes varying in size, productivity levels and flushing rates, but subject to the same local weather patterns: the physical responses to climate variability across the lakes was highly coherent, while chemical response were less coherent. The variability in the chemical responses of the lakes was attributed to variability in nutrient loads between catchments (George et al. 2000). Similarly, in our study we observed the same trend in the onset of stratification (a physical response to climate drivers) between Lake Washington and Lake Sammamish, but varying sensitivity in the internal nutrients (chemical response to climate drivers).

TP concentrations in the hypolimnion of Lake Sammamish increase with depth, and the positive relationship between duration of anoxic and yearly summer maximum hypolimnetic TP both suggest substantial internal regeneration of P from sediments. We demonstrated here that physical drivers of climate change can control the internal loading of TP via oxygen depletion. However, previous studies have shown that oxygen depletion rates have in turn been correlated to annual mean TP and driven by productivity in lakes (Rippey & McSorley 2009; Matzinger et al. 2010). These results suggest that exacerbating internal P release can create a positive feedback loop: increased nutrient loads lead to greater productivity, thereby increasing the consumption of deep water oxygen and thus incurring further P release. Such increases in internal P loading may have important consequences for algae blooms in late summer and early autumn. The P cycle of Lake Sammamish may be trapped in an alternate stable state that is maintained by anoxia and P regeneration from sediments. Our results suggest that ongoing climate change may further stabilize this feedback and further delay recovery from eutrophication.

Our results imply that more highly productive lakes will respond to climate impacts in their biogeochemical cycles, resulting in increased internal nutrient loading and increased risk of eutrophication. Climate change is already exacerbating eutrophication and causing other problems in lakes worldwide via other pathways. Warmer temperatures are shifting food webs and increasing external nutrient loads from increased winter rainfall (Moss 2012; Jeppesen et al. 2009). They are linked to greater instances of toxic cyanobacteria blooms (Posch et al. 2012), and the combined effects of warmer surface temperatures and increased deep water anoxia reduce fish habitat availability (Arend et al. 2011). The link between changing phenology of thermal stratification, hypolimnetic anoxia, and P loading from hypolimnetic sediments should be integrated into the growing list of climate-sensitive processes in lakes.

Many lake management policies rely on controls of external nutrient loading to maintain lake water quality. External loads originating at point sources are easy to control and regulate; however, loads entering bodies of water from non-point sources remain a management challenge. Future climate conditions may further compromise regulatory efforts to maintain lake water quality, or require longer periods of time for lakes to fully recover from historically high P loads.

Tables and Figures

Table 1 – Physical characteristics of Lake Washington and Lake Sammamish (King County, 2012)

	Lake Washington	**Lake Sammamish**
Length	35 km	12.9 km
Mean depth	32.9 m	17.7 m
Maximum depth	65.2 m	32 m
Flushing rate	2.33 years	1.79 years
Lake area	87.6 km^2	19.8 km^2
Lake volume	$2.9x10^9$ m^3	$3.5x10^8$ m^3
Drainage basin area	1,448 km^2	255 km^2

Table 2 - Summary of hypolimnetic temperature, dissolved oxygen, and total phosphorus concentrations in Lake Sammamish and Lake Washington. Yearly variability describes the range of monthly means. Summer is defined as June – September.

	Temperature (°C)		**DO (mg/L)**			**TP (µg/L)**		
	yearly range	summer average	yearly range	range of yearly summer minima	average summer minimum	yearly range	range of yearly maxima	average summer maximum
Lake Sammamish	6.40 – 9.60	9.41 (sd= 0.91)	1.01-11.22	0.51 - 1.29	0.81 (sd = 0.20)	10-40	20-179	50 (sd = 30)
Lake Washington	7.11 – 9.44	8.76 (sd = 0.68)	6.42 – 11.07	5.68 - 6.87	6.31 (sd = 0.40)	10-20	20-60	30 (sd = 10)

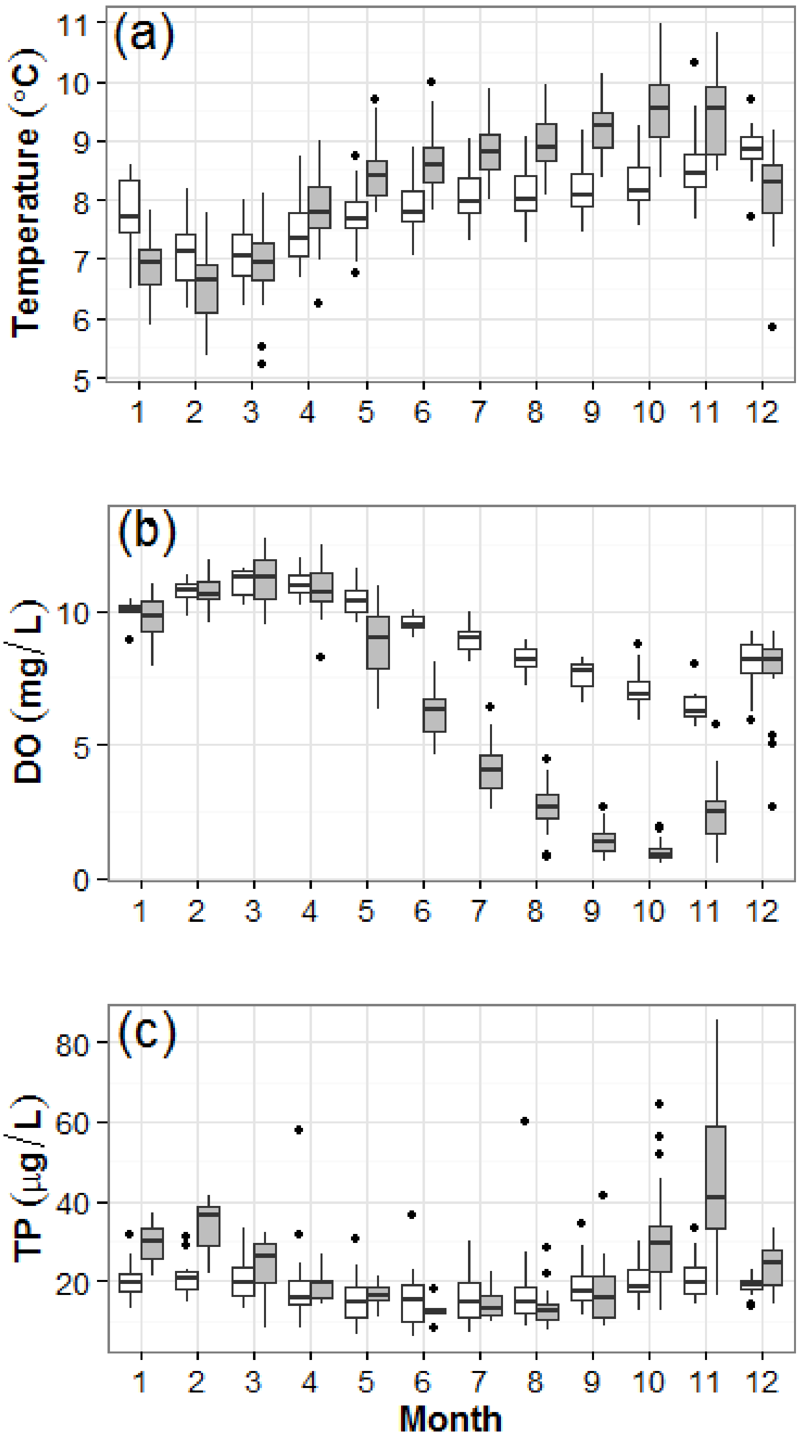


Figure 1 – Seasonal variation in hypolimnetic (a) temperature, (b) dissolved oxygen, and (c) total phosphorus in Lake Washington (white) and Lake Sammamish (gray). Dark bars indicated median values, and boxes indicate the interquartile range.

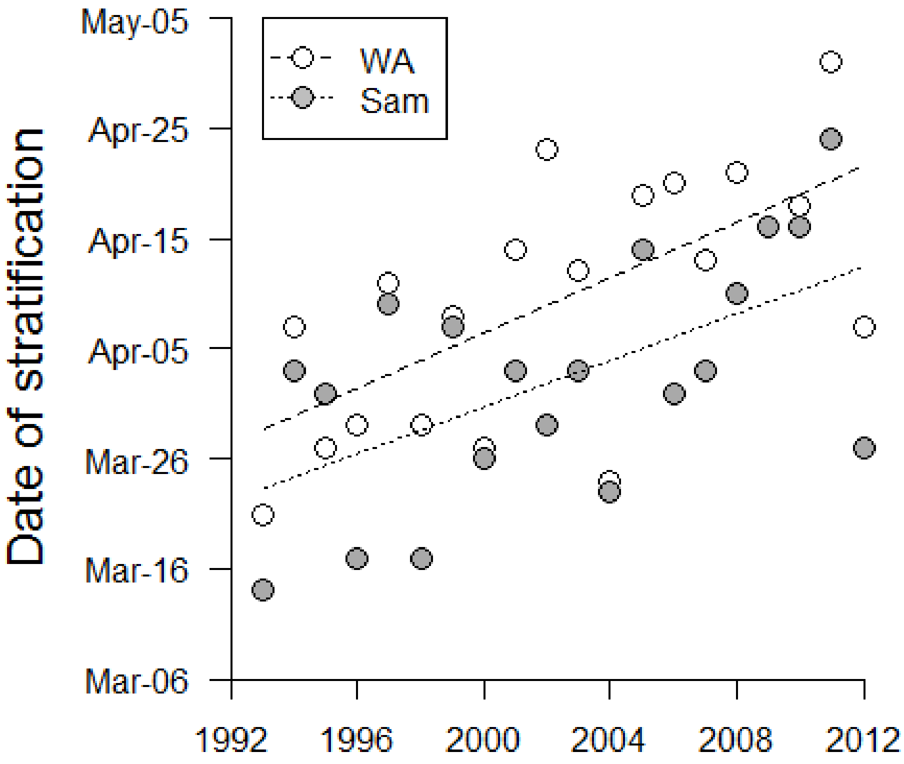


Figure 2 – Estimated date of stratification for Lake Washington (white) and Lake Sammamish (gray) from 1993-2012. Both lakes exhibited a trend towards later stratification over the study period (Least squares regressions: Lake Washington slope = 1.25, p = 0.002, r^2 = 0.43; Lake Sammamish slope = 1.06, p = 0.007, r^2 = 0.33). In all years but one, Lake Sammamish stratified prior to Lake Washington.

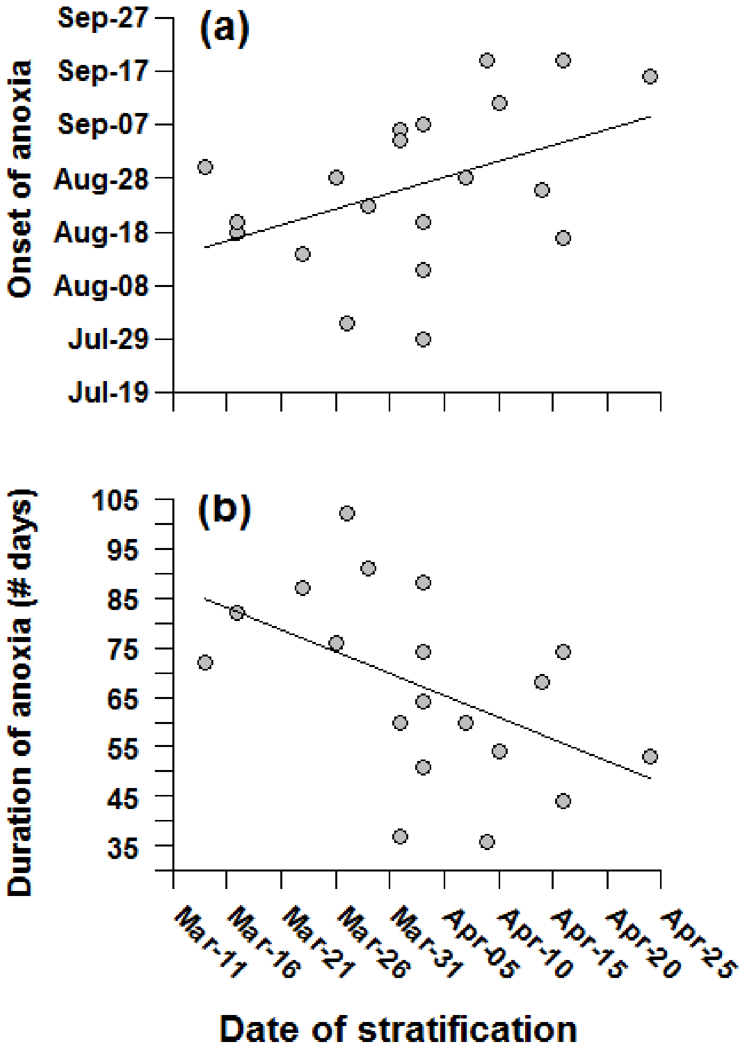


Figure 3 – (a) Onset date of anoxia as a function of date of stratification in Lake Sammamish. The earlier the date of stratification, the earlier the onset date of anoxia (slope = 0.60, $p = 0.05$, $r^2 = 0.19$). (b) Duration of summer anoxia as a function of date of stratification in Lake Sammamish. The earlier the date of stratification, the longer the duration of summer anoxia (slope = -0.88, $p = 0.02$, $r^2 = 0.27$).

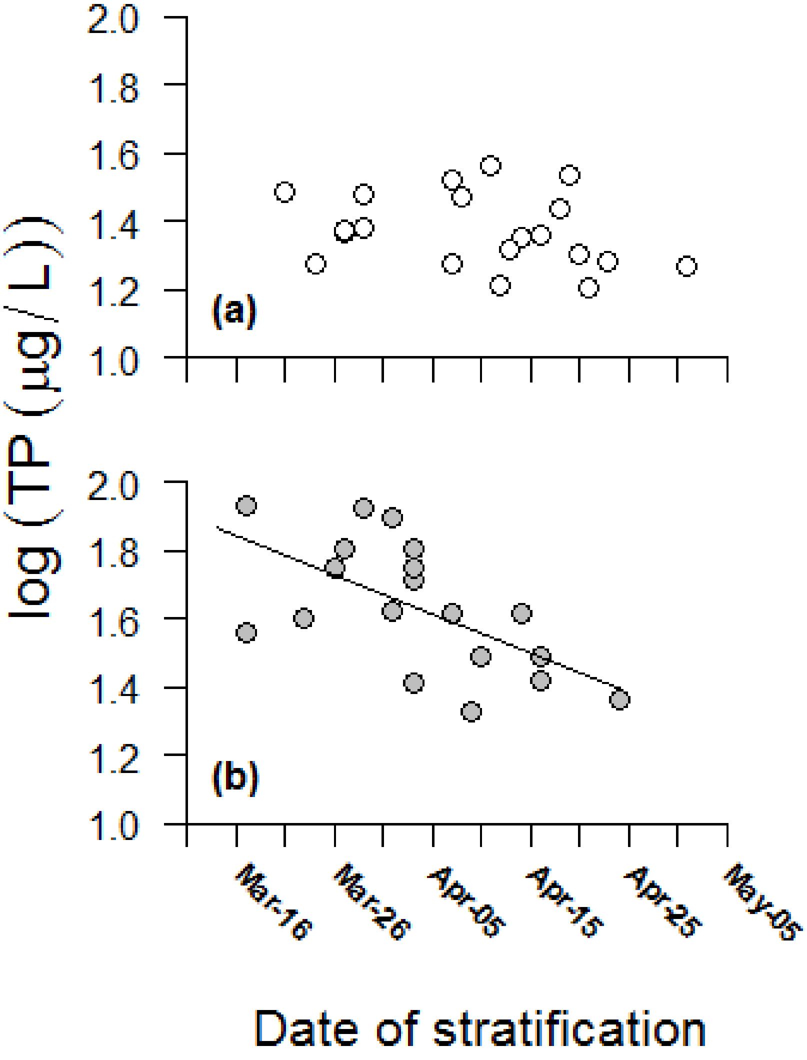


Figure 4 – Summer maximum volume-weighted hypolimnetic TP as a function of stratification date for (a) Lake Washington ($r^2 = 0.07$, $p = 0.24$) and (b) Lake Sammamish (slope = -0.01, $p = 0.005$, $r^2 = 0.37$). Earlier stratification dates resulted in higher concentrations of TP.

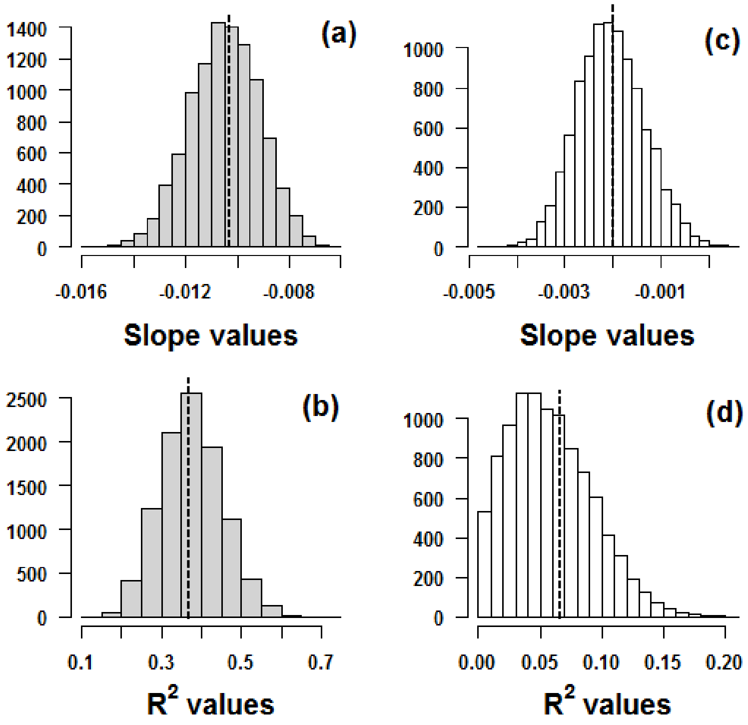


Figure 5 – Results for Monte Carlo analysis linear regression of log-transformed, volume-weighted summer maximum TP as a function of stratification date in each lake. Based on a moderately conservative beta-distribution model, it is predicted that the most likely relationship between stratification date and hypolimnetic TP in Lake Sammamish (a, b) is negative. In Lake Washington (c, d), predicted slopes are less steep, the fit of the regress is less tight, and fewer relationships are significant. Dashed lines represent slope and r^2 values for the corresponding linear regressions (Figure 6).

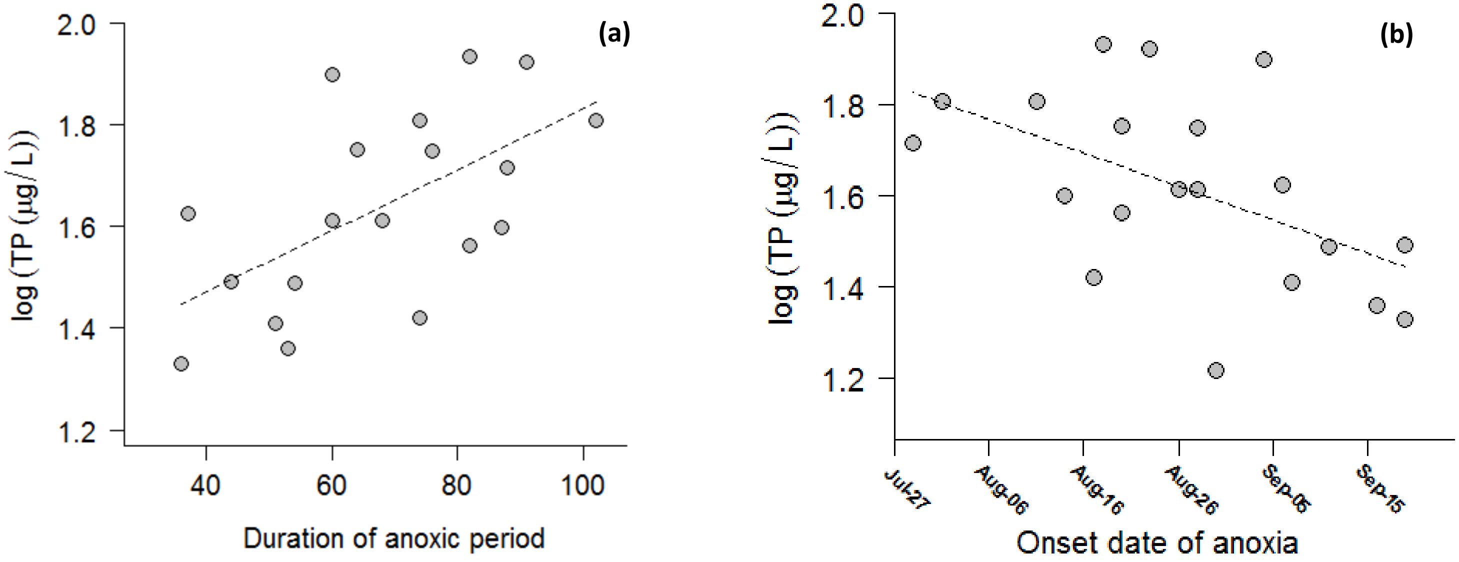


Figure 6 – (a) Lake Sammamish yearly summer maximum volume-weighted TP as a function of the onset date of anoxia for each year of the study period (slope = -0.01 µg/L/day, p = 0.02, r^2 = 0.28). The earlier the onset date of anoxia, the higher summer TP concentrations. (b) Lake Sammamish yearly summer maximum volume-weighted TP as a function of the duration of anoxia for each year of the study period (slope = 0.01, p = 0.02, r^2 = 0.26). Longer periods of anoxia result in greater summer TP concentrations.

Appendix A: The effect of varying mode and K parameters on the probability density function of the beta distribution

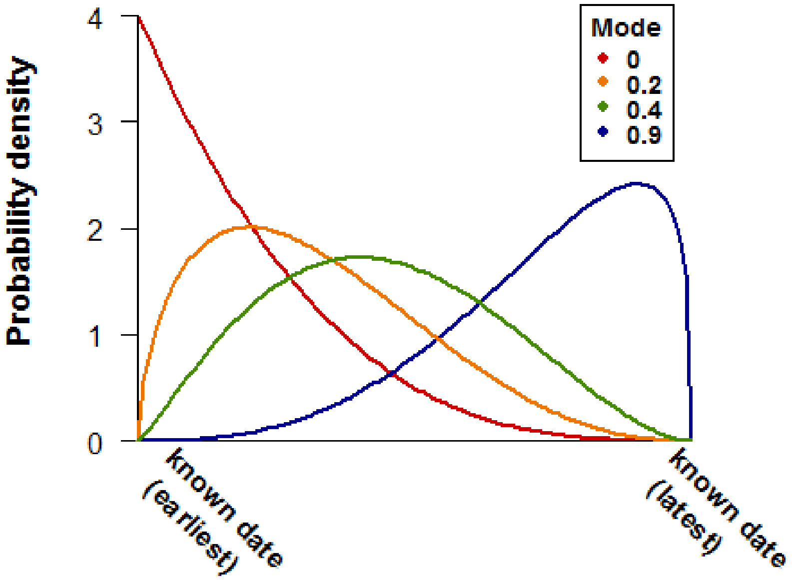


Figure A1 – Possible beta probability distributions using different mode parameter values. In this parameterization, the mode represents where the calculated date of stratification falls between two sampling days. A mode of 0 or 1 would indicate a estimated stratification date on the early or late sampling date, respectively. An estimated stratification date that fell exactly between the two sampling days would have a mode of 0.5. Using a conservative k value (k=5), the values generated by the beta distribution are more likely to be near the mode (i.e., estimated date of stratification), however all values between the two sampling dates are in the range of possibility

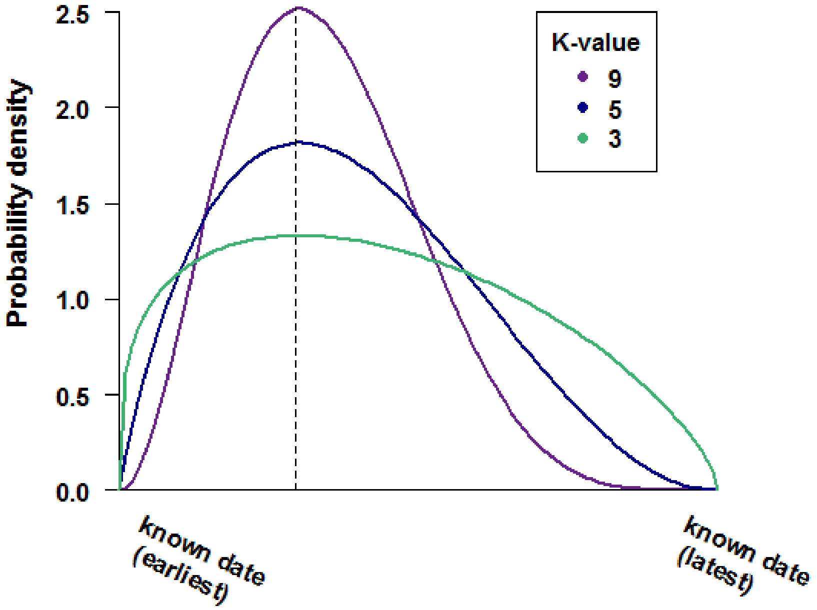


Figure A2 – Beta probability distributions using different K-values. Varying the K-value changes the weight given to the mode in the distribution. A higher K-value reflects greater confidence in the calculated mode. The mode is set to 0.3 in this example.

Chapter 2:

Policy Tools For The Regulation Of Lake Pollution: From The Federal To Local Levels

Introduction

Nutrient pollution is one of the largest threats to lake health throughout the United States (Environmental Protection Agency 2012c). People have been increasing the amount of phosphorus in our aquatic ecosystems through runoff from fertilizers use, animal feed lots, and municipal and industrial wastewater (Bennett, Carpenter, and Caraco 2001). Phosphorus (P) is a mineral that is essential for all life, but under natural conditions it is relatively scarce in the environment. In aquatic ecosystems, and especially lakes, phosphorus acts as a limiting nutrient, regulating the growth of algae and aquatic plants (Minnesota Pollution Control Agency 2007). Humans have been increasing the amount of phosphorus in our aquatic ecosystems through runoff from fertilizers use, animal feed lots, and municipal and industrial wastewater (Bennett, Carpenter, and Caraco 2001). This over-enrichment of phosphorus causes excessive algae growth in lakes – eutrophication – which negatively impacts water quality and water clarity. Eutrophication decreases water clarity, reduces fish habitat, and increases the likelihood of toxic and nuisance algae growth. These biological impacts have significant human impacts as well: decrease in recreational water use, decrease in the value of waterfront property, spending on water quality treatment and mitigation measures – a 2008 economic analysis estimated that the United States spends about $2.2 billion per year due to eutrophic waters (Dodds et al. 2009).

Phosphorus pollution enters bodies of water through point sources – single identifiable discharge points such as sewage treatment outflow pipes – or nonpoint sources. Nonpoint sources of pollution are those that originate from multiple sources over a large area, such as leaking septic systems, urban stormwater runoff, forestry activities, and non-commercial agriculture (Environmental Protection Agency 2012a; Department of Ecology 2005). Point source pollution is quite easy identify and therefore regulate, however because nonpoint source pollution is more diffuse and because the origin of it is difficult to identify, management of nonpoint source pollution is quite difficult. Not only is nonpoint source pollution one of the most significant water quality problems in the county, it is the primary source of pollution in Washington’s freshwaters (Environmental Protection Agency 2012b; Department of Ecology 2005).

In addition to nutrient pollution, lakes and other bodies of water throughout our nation are subject to a number of other environmental stressors, such as habitat degradation, flow modification, resource exploitation, and species invasions. All of these stressors are exacerbated by climate change (Woodward et al. 2010; Dudgeon et al. 2006). In King County, climate change is expected to increase winter discharge – leading to higher flood risks and the possibility of sewage overflow into floodwaters, and decrease summer discharge – leading to water temperature exceedances and increased concentrations of pollutants (King County 2007).

Lake ecosystems are particularly sensitive to climate warming because metabolism and growth rates of phytoplankton and macrophytes are temperature-controlled (Whitehead et al. 2009). Evidence of climate-related temperature change has already been observed in lakes of the region (King County 2012; Winder & Schindler 2004b). For example, in Lake Washington, spring water temperatures have increased over 1° C since 1962, and spring stratification of the lake now occurs 21 days earlier than it did forty years ago (Winder & Schindler 2004a). These changes have uncoupled phytoplankton and zooplankton blooms in the lake, causing declines the zooplankton *Daphnia* (Winder & Schindler 2004a). *Daphnia* are a major food source for juvenile fish and their presence contributes to water clarity because they prey on algae (Winder & Schindler 2004b). This shifting food web structure, combined with increased external nutrient loading and warmer temperatures, is predicted to increase lake productivity and promote algal blooms. Algal blooms decrease water clarity, may cause noxious odors, and can be toxic to humans and animals (Johnston & Jacoby 2003). Furthermore, algal blooms can deplete deep water oxygen levels. Oxygen depletion regulates the internal release of phosphorus stored in lake sediments (Filbrun et al. 2013). Between this internal P release and increasing external P loads from terrestrial sources, lakes are particularly at risk to climate change. Because management of nutrient pollution in lakes focuses primarily on limiting external nutrient sources, future climate regimes that exacerbates the internal release of stored P are likely to circumvent or inhibit lake management and restoration efforts.

Managing nutrient pollution and water quality in lakes is extremely challenging. Setting specific allowable nutrient levels for lakes within a jurisdiction is not particularly useful because the physical, chemical, and biological characteristics of a lake largely determine the nutrient load a lake can withstand. Every lake responds uniquely to environmental changes, and nutrient levels in any given lake may not necessarily indicate a problem. Furthermore, nonpoint sources are major contributors to nutrient pollution. Controlling these sources is difficult because they are by definition, diffuse and hard to identify.

In King County (Washington, United States), Lake Washington and Lake Sammamish are both in moderate to good condition, but the smaller lakes at the urban fringe of the region are experiencing severe eutrophication (Moore et al. 2003). In the 2012 Comprehensive Plan, King County stated that the region's aquatic habitats are considered important biological, social, and economic resources, and as such, has directed the protection and management of these resources:

- *Lakes should be protected through management of lake watersheds and shorelines. Lakes sensitive to nutrients shall be protected through the management of nutrients that stimulate potentially harmful algae blooms and aquatic plant growth (p. 236).*
- *King County shall provide a high level of protection to critical freshwater habitats due to the important ecological functions they provide (p. 319)*
- *King County shall protect ecological functions associated with critical freshwater habitat as necessary to assure no net loss from shoreline activities and associated changes (p. 319)* (King County 2012)

Because freshwater is so important to KC communities, polices are implemented at the federal, state, and local levels to maintain water quality and preserve fresh water for the current and future needs of people and the natural environment. These policies are largely regulatory, but the challenges of managing nutrient pollution necessities a broad base of tools. In this paper, I examine the effectiveness of the policy tools used to manage lake water quality in Washington State. Understanding these tools and their effectiveness helps us plan and adapt to future climate scenarios, and keep Washington lakes healthy into the future.

Status Quo

Management of water quality in Washington State originates at the national level, though the Clean Water Act (CWA) (33 U.S.C. § 1251 – 1376, P.L. 95-217. 1972). The CWA is the primary federal law governing water quality throughout the United States. It is a regulatory policy tool, employing management-based regulations, technology-based standards, and performance-based standards. The CWA arbitrates the adoption of water quality standards, regulates point source pollution through a permitting process, and mandates the management of nonpoint sources of pollution (Environmental Protection Agency 2014).

The power to implement the CWA and enforce surface water quality standards is given to the states, provided that a state's regulations are at least as strict as the standards set at the federal level (Scheberle 2004). The EPA sets standards for states which do not establish their own water quality standards or whose standards do not meet federal guidelines (Environmental Protection Agency 2014). In Washington State, the Department of Ecology (Ecology) serves are the regulatory hand of the CWA. It is through Ecology that point source discharge permits are issued, nonpoint source pollution control programs are implemented, and water quality standards are set for waters within the state. The water quality standards developed by Ecology have been approved by the EPA.

The CWA regulates point source pollution through a permitting system, the National Pollutant Discharge Elimination System (NPDES). NPDES permits must be issued to anyone who discharges pollutants through a point source into a body of water. Municipalities, water treatment plants, animal feed operations, and aquaculture facilities all may have NPDES permits. NPDES permits limit what a facility or municipality may discharge, and prescribe monitoring and reporting requirements (Environmental Protection Agency 2007). Storm water that is collected into municipal storm sewer systems and then discharged falls under regulation by the NPDES, and in this way, the NPDES program plays a role in the regulation of nonpoint source pollution (King County 2013a).

Ecology administers NPDES permits in the state of Washington. King County has a municipal stormwater permit, which regulates discharge from the county's municipal separate storm sewer system (MS4) (King County 2013a). This permit has ten components: legal authority, mapping, coordination, public involvement, runoff control, structural storm water control, source control, detection and elimination of illicit discharges, operations and maintenance, and education and outreach, which are implemented in the county's Stormwater Management Plan (SWMP) (King County 2013a). The plan describes how the county intends to address the requirements of the NPDES permit.

Most of the policy tools under this permit are regulatory design-standards through enforcement of best management practices (BMPs). However, there are some elements of collaborative management tied into the permit requirements. The NPDES permit requires the County to implement intra- and intergovernmental coordination agreements to encourage watershed-level stormwater management. Additionally, the public has the opportunity to review and provide feedback on the SWMP; with each annual update of the SWMP, KC must consider public comments in its revisions.

To fund the activities mandated by the NPDES permit, the county has implemented a surface water management fee to property owners. Residential property owners pay a flat fee, while non-residential property owners pay on a scale based on the degree to which their parcel is impervious (King County 2013b). Market-based incentives are used to encourage commercial property owners to implement storm water control methods: non-residential parcels are eligible for a fee-discount program which incentivizes storm water control methods and best management practices such as detention ponds, rain gardens, debris barriers, and energy dissipaters. To further incentivize good stormwater management practices, non-residential parcel owners are eligible for two types of grants to help reduce impervious surfaces on their property (King County 2013c). One grant is a cost-sharing program, in which funds are allocated to property owners who wish to construct alternatives to impervious surfaces, such as compost-amended lawns, native-vegetated landscapes, or grassed modular grid pavement. The other grant program is a credit-based program, in which the fee category for the parcel is adjusted to reflect installment of alternative impervious surfaces (King County 2013c).

The CWA relies upon a combination of performance-based and management-based regulation to address nonpoint source pollution. Management based regulation is directed at the activities, rather than performance, of an actor, in order to encourage environmentally beneficial behavior (Coglianese & Nash 2006). The baseline standards of the NPDES program are management-based: emphasis is placed on technology-based effluent limitations, in which the “best practicable control technology currently available” must be used (33 U.S.C § 1311). These technology-based standards establish the minimum level of protection for all facilities within a category, standardizing discharge rules throughout all states.

If technology based standards fail to maintain the water quality of a particular body of water, additional performance-based standards must be implemented (Craig 2007). Under the CWA, states are required to identify navigable waters that are not expected to maintain water quality standards without action to control of nonpoint source pollution, identify sources of nonpoint source pollutants that could contribute to degradation of these listed waters, and describe the processes and programs used to manage and control each identified pollutant (33 U.S.C. § 1329). States that submit approved nonpoint source pollution management programs to the EPA are eligible for CWA Section 319 federal grants. However, to be eligible for funding and to retain funding, states must also annually report on performance-based accomplishments of their program, such as load reductions and actual water quality improvements. In Washington State, Ecology has a Water Quality Management Plan to Control Nonpoint Source Pollution. In the past this plan has emphasized process-based measures and actions. However, to meet the reporting requirements of Section 319, focus has shifted to focus more heavily on outcome-based performance measures (Department of Ecology 2005).

The TMDL program unites the point and nonpoint source aspects of the CWA – it addresses bodies of water that fail to meet water quality standards despite NPDES permits and nonpoint source regulation (Craig 2007). When technology based effluent limitations fail to maintain water quality for a particular water body, a TMDL for a specific pollutant is set – a goal for the total amount of that pollutant that can enter that water body. In Washington State, TMDLs are established by Ecology and submitted to the EPA. In King County, three lakes currently have TMDLs set for total phosphorous. Other TMDLs in the county include fecal coliform bacteria, dissolved oxygen, temperature, and ammonia-N (Department of Ecology 2013).

In addition to the state level actions that are implemented to comply with the CWA, Washington State has passed two laws to further limiting phosphorus pollution in the state — RCW 15.54.500, which prohibits the sale and application of turf fertilizer that contains phosphorus, and RCW 70.95L.020, which prohibits the sale of any dishwashing or laundry detergent that contains more the 0.5% phosphorus by weight, excluding detergents used for commercial and industrial purposes.

Evaluation

Social regulation is implemented to address behaviors that threaten the public good, regulating problems created by externalities in the economic market (May 2002). It is an appropriate regulatory tool to address point source water quality issues – it controls the behavior of specific entities where their actions have the potential to impact a broad group of people.

According to May (2002), regulations have four components: rules, which set the expectations of behavior; standards, which serve as benchmarks for compliance; sanctions, a combination of penalties and rewards to stimulate compliance; and enforcement mechanisms, to detect noncompliance and invoke sanctions. For regulations to be effective, regulated entities must feel that the cost of complying is worthwhile – the penalties for non-compliance must be large enough to be a reasonable deterrent, and the benefits of compliance must be large enough incentives. Regulation is most successful when the rules are reasonable, regulated entities are willing and able to comply, and there are enough resources for regulatory bodies to implement enforcement (May 2002).

Reasonable Rules: Often, regulations are critiqued for being too restrictive (May 2002). Giving states the power to implement their own programs offers flexibility to the CWA, which helps ensure that rules are reasonable. By implementing their own programs, states are able to craft management plans appropriate and relevant to their own specific water quality challenges, making it more likely that regulated entities can and will comply. The use of a permitting system adds an additional level of flexibility to the CWA; because permits are site specific, through permitting the CWA regulations can be adapted to the specific needs of individual sites (Environmental Protection Agency 2007).

Enforcement: Effectively implementing the CWA in Washington State requires a process of enforcement to ensure compliance. The Department of Ecology uses three tools to gain compliance: educational programs, cooperation based programs and enforcement, and deterrent-based enforcement

(Department of Ecology 2010). Most enforcement is enacted directly by Ecology through civil rather than criminal action through the courts (Department of Ecology 2010). Ecology emphasizes its use of educational- and cooperation-based programs, and that the primary goal is voluntary compliance (Department of Ecology 2010) (Figure 7). Ecology encourages voluntary compliance by providing technical assistance to firms. Furthermore, in the first step in the penalty process, notices are issued. These inform a site owner that he/she is in violation of, or have the potential to violate an environmental law (Department of Ecology 2010).

Administrative orders are used to order violators to correct their violations, and civil penalties are issued when a violation has occurred. The purpose of these penalties is to correct the violation and ensure future compliance, not to act as punishment (Department of Ecology 2010). As such, the size of the penalty is based on the nature of the violation (severity), prior behavior of the violator, and degree of cooperation of the violator (Department of Ecology 2010). Penalties are given in the form of field tickets directly during inspection of sites. Penalty tickets are issued only after a prior inspection has resulted in a warning ticket, to ensure that regulated entities have the opportunity to correct a problem (Department of Ecology 2010). Furthermore, violators have the option of negotiating an innovative settlement agreement, which can reduce the penalty in exchange for submitting a Supplemental Environmental Project (SEP), which would result in environmental benefits relevant to the nature of the violation. For example, in 2007, Ocean Park Concrete was fined a $15,000 penalty for groundwater contamination that resulted from improperly sampling and treating wastewater from concrete processing (Department of Ecology 2008). This penalty was amended through an innovative settlement agreement in which $5,600 of the penalty fee would go towards the installation of an environmental education structure at Pacific Pines State Park and only $1,400 to the Department of Ecology (Department of Ecology 2014).

The enforcement structure for environmental regulations used by the Department of Ecology is set up to encourage compliance. The step-wise nature of violation notifications provides ample time for firms to address compliance concerns before receiving penalties, and firms are offered technical assistance to aid with compliance. The SEPs allow firms to reduce their penalties by taking environmentally beneficial actions.

Equity: One challenge of regulatory enforcement is implementing both consistent and appropriate sanctions to individual violators. The Department of Ecology has acknowledged that monetary penalties may not be the most effective way to achieve compliance from municipalities because these simply move funds from one government agency to another, and furthermore may limit an agency's operating budget for environmentally beneficial actions (Wrye 1999). Enforcement staff are therefore encouraged to use discretion in determining penalties or alternatives for municipal violators, and in this way Ecology distinguishes between industry and municipal violators.

Dividing violators into only municipal and industrial categories may not be the most effective way to decide on enforcement actions. If the overarching goal is to achieve compliance, penalties should be determined based on the capabilities and resources available to the violator. Small towns and small industries may be more similar than small towns and large towns or small industries and large industries in terms of access and availability of resources. For this reason, enforcement policies would be better structured based both on the size of the violating entity and the environmental impact of their violation (Wrye 1999). As consistent and fair enforcement policies encourage compliance (May 2002), more structure for determining penalties could improve the effectiveness of this regulatory tool.

Resources: It is in this area that the CWA is most lacking. By delegating implementation and enforcement to the states, the burden of developing management plans is relieved of the federal government, and the financial burden of implementation is largely born at the state level. While Section 319 does give states funding for nonpoint source pollution programs, nonpoint source pollution management programs are funded though the SWM fee.

In Washington State over 4,000 industrial and municipal facilities have NPDES permits, and each year Ecology inspects 25% of them (Department of Ecology 2010). In a 1999 review of Ecology's water quality enforcement program, it was noted that while the number of permittees had quadrupled in the previous ten years, but the size of the enforcement staff had remained the same, resulting in a 1:600 enforcement staff to permittee ratio (Wrye 1999). This is likely to decrease the effectiveness of the NPDES program, as inspectors may have less time to be thorough and firms may fall through the cracks.

Alternatives to command-and-control regulation: Noncompliance can be addressed in ways other than penalties. Market-based tools, production caps in particular, could be implemented to regulate industrial violators. These offer more flexibility for a firm to weigh the costs and benefits of environmental compliance. Additionally, creating social pressures by publicizing noncompliance may be a stronger deterrent to potential violators than monetary penalties. Currently Ecology only publicizes enforcement actions with penalties over $10,000 (Wrye 1999). However, as an information-based regulatory tool, making noncompliance information available to the public may change the behavior of both consumers and disclosers of information by addressing information asymmetries in the economic market (Fung et al. 2007).

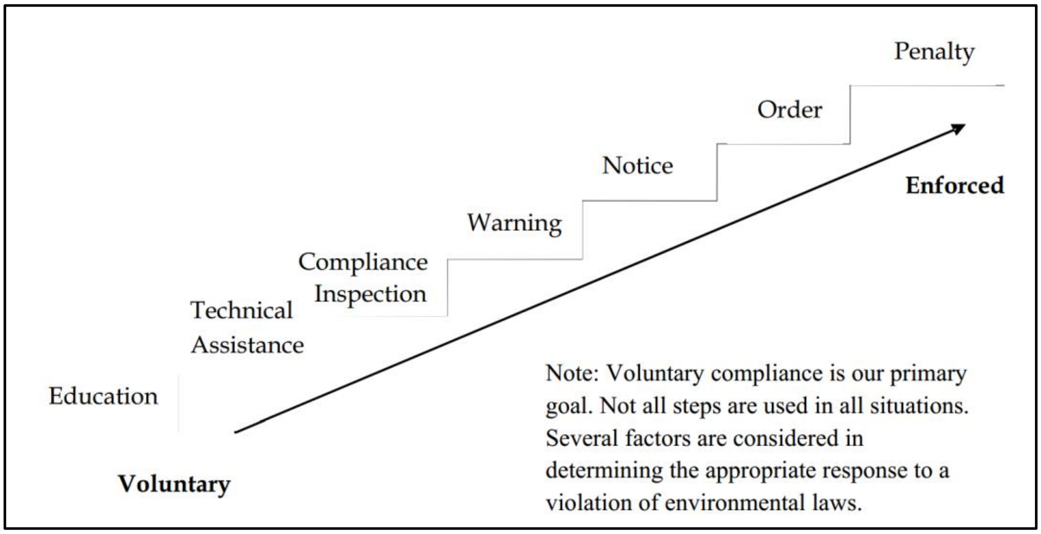
The NPDES program relies largely upon disincentives that discourage violations rather than incentives that encourage compliance. Setting regulatory standards also sets a bar at a minimum level of compliance but does not offer any incentives to surpass it or encourage innovation towards even higher standards. A common critique of regulatory policy tools, this problem can be addressed through voluntary environmental programs (VEPs) (May 2002; Prakash & Potoski 2007). VEPs are a management-based strategy that reward actors who adopt best practices that produce positive environmental benefits beyond minimum regulatory standards (Prakash & Potoski 2007). This policy tool could be used at multiple levels of water quality control: at a residential level, facility level, or municipal level. For example, property owners could become “pollution stewards” based on the number of BMPs they implement on their property (such as creating rain gardens, or installing permeable pavers), and could receive a sign they could display in their window, as well as gain recognition through KC announcements.

The SWM fee for private property is an excellent use of market-based governance tools rather than regulatory tools. Encouraging good water quality management practices by private landowners avoids the regulatory problem of inspection and enforcement on a large number of properties with constantly changing property owners. Furthermore, by encouraging the use of best management practices, the SWM fee addresses nonpoint source pollution without necessitating the identification and tracking of diffuse pollution sources – which is the main challenge of regulating nonpoint sources. Finally, traditional command and control regulation requires setting specific standards for both pollutants and polluters (Bryner 1999). As private property varies greatly in physical size, ecological traits, and use, both establishing and enforcing standards that are equitable for all cases would be extremely difficult.

The CWA offers few mechanisms to address trans-boundary water quality issues. The NPDES program acknowledges that coordination across within and across jurisdictions is important, and requires municipal permit holders to implement coordination mechanisms. KC's SWMP describes several regional forums that KC participates in: an outreach group whose purpose is to coordinate public education and outreach efforts across jurisdictions, a coordination group that addresses operations and maintenance requirements, a discussion group, a policy advisory group, and a storm water monitoring group (King County 2013a). These groups incorporate aspects of collaborative management – in particular, different groups of stakeholders are brought together to address a shared problem. However, a major component of collaborative management is that decision making power is shared between government actors and other stakeholders (Koontz et al. 2004). Because collaboration is enforced via the regulatory processes of the CWA, this governmental institution has a lot of power over the structure, resources, and outcomes of associated collaborative networks and over any decisions made within them. The effectiveness of these collaborative networks is therefore impaired by the regulatory system in which it exists. Command-and-control regulation and site-specific permitting is effective at addressing identifiable individual polluter behaviors. Collaborative management has the potential to more effectively address nonpoint source pollution, however this would require loosening the regulatory leash on those collaborative processes.

Conclusions

Creating effective policies to manage the nutrient inputs into lakes is challenging, both because lake ecosystems are complex, dynamic, and sensitive, and because nutrient inputs come for a very large variety of sources. The regulatory command-and-control policies of the CWA are effective for addressing point sources of pollution and enforcing the actions of identifiable actors. Devolution of authority to the states, and the use of permits allows for flexibility in the implementation of water quality standards. Market-based incentives help to address the number and diversity of private actors who contribute to nutrient pollution. Both Department of Ecology and King County encourage regulatory compliance through education and technical assistance. However, the challenges of implementing alternative polices outside of the regulatory process is highlighted by the strong federal regulatory oversight that the CWA holds on states and local jurisdictions. While command-and-control regulation is less effective at managing nonpoint source pollution, the overarching structure of regulatory governance can hinder the implementation of more innovative solutions for address nonpoint sources. Permit-based water quality management policies encourage actors to act independently of one another. Collaboration between different jurisdictions and stakeholders is constrained by regulatory requirements and government-held decision-making power. VEPs are one way that local agencies could encourage innovative nutrient management solutions outside of regulatory constraints, and true collaboration also has the potential for addressing more difficult, trans-boundary water quality problems. In a future of increasing ecological threats to our water systems, expanding our governance tools will be a critical component of sustaining the health of these important resources.



Note: Voluntary compliance is our primary goal. Not all steps are used in all situations. Several factors are considered in determining the appropriate response to a violation of environmental laws.

Figure 7 – Compliance structure for the CWA as implemented by the Washington State Department of Ecology (Department of Ecology 2010)

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