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The Stability of Temperate Lakes Under the Changing Climate

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Abstract

There is a collective prediction among ecologists that climate change will enhance phytoplankton biomass in temperate lakes. Yet there is noteworthy variation in the structure and regulating functions of lakes to make this statement challengeable and, perhaps, inaccurate. To generate a common understanding on the trophic transition of lakes, I examined the interactive effects of climate change and landscape properties on phytoplankton biomass in 12,644 lakes located in relatively intact forested landscapes. Chlorophyll-a (Chl-a) concentration was used as a proxy for phytoplankton biomass. Chl-a concentration was obtained via analyzing Landsat satellite imagery data over a 28-year period (1984-2011) and using regression modelling. The most common lake trophic state was oligotrophic (median Chl-a < 2.6 μg L^{-1}), while the least common was hyper-eutrophic (median Chl-a > 56 μg L^{-1}). Lake volume was the most important factor in determining the present trophic state of the lakes. The majority of the lakes (91.6%) did not show a change in trophic state over an almost 3-decade long sampling period; only 4.0% of the lakes became more eutrophic, and 4.4% of the lakes became more oligotrophic. Lakes with smaller volumes were further responsive to temperature (warmer lakes were more eutrophic), while lakes with larger volumes were more responsive to precipitation (wetter lakes were more oligotrophic). Early warning indicators of change in trophic state were examined in the patterns of the residuals of the time series of Chl-a once non-stationary and stationary trends were removed. Remarkably, the majority (56.5%) of the lakes showed patterns in the residuals that were not defined by a single trophic metric but fluctuated among different trophic states. There was an unexpected instability among some lakes as they switched between oligotrophic and eutrophic states (12.5%) or were transitioning from eutrophic towards oligotrophic states (23.4%), or from oligotrophic towards eutrophic states (20.6%). The complex responses of phytoplankton biomass to climate change suggests that our ability to predict the future trophic state of lakes will be limited but enhanced if we recognize that lakes and their catchments will be both impacted by climate change.

Keywords

Lake, phytoplankton, chlorophyll-a, trophic state, stable states, instability, oligotrophication, eutrophication, climate change.

Summary for Lay Audience

The impact that climate change will have on Canadian temperate lakes remains poorly understood. There are several reasons for our lack of confidence in describing the effect of climate change. First, many lakes that are near human populations are also impacted by the anthropogenic pressures – direct or indirect use of surface waters for consumption and the use of lakes for the intended or unintended deposits of wastes. Second, our ability to predict the changes in lake ecology is hampered by our slow observance of changes that are currently taking place. It appears that lakes may be changing from clear water states to turbid productive water states with an increased incidence of potentially harmful algal blooms. Although undesirable, these changes can be either gradual (i.e., linear), or small and non-linear, and the latter is much harder to identify. Finally, since lakes are of many shapes and sizes (i.e., they have different morphometry), they will not be impacted by climate change equally. Thus, reports on climate change about the functioning of lakes might be too general. This thesis attempts to avoid these problems by studying over 12,000 lakes in the temperate forest region of Canada. Using satellite records of lake chlorophyll-a (a proxy measure of algal biomass in lakes) over 28 years, I have determined that climate change affects ~44 % of the lakes, with ~21 % of the lakes becoming more productive and ~23 % of the lakes becoming less productive. The remaining lakes either do not respond to the changing climate or oscillate between low and high productivity. The trends documented in this thesis indicate how the lakes might look like in the future (as climate change continues) and if they can be used as a healthy water supply for the next generations.

Co-Authorship Statement

The PhD thesis contains three manuscripts. In each of these manuscripts, Aleksey Paltsev will be first author, as he conducted pre-processing of Landsat products (Chapter 2) and statistical analysis; he contributed to design of the research, interpretation of the research results and writing of the manuscripts. In each of these manuscripts, Dr. Irena Creed will be second author, as she contributed to the definition of the research problem, interpretation of results, synthesis of ideas and editing of the manuscripts.

The PhD thesis is extension of Aleksey Paltsev's Master of Science research: "Exploration of spatial and temporal changes in trophic status of lakes in the Northern Temperate Forest Biome using remote sensing" published as a Master's thesis in 2015. Although in this PhD thesis, Aleksey Paltsev adopted research approach and some ideas initially developed in his Master's research, all pre-processing steps of Landsat products and statistical analyses (including the development of the regression model) were substantially modified, which resulted in a different set of modelled data. Furthermore, more in-depth analyses were performed, for example the space-time kriging and the Mann-Kendall test. The Master's thesis is cited throughout the PhD thesis where it is appropriate.

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List of Abbreviations

AMO	Atlantic Multidecadal Oscillation
ANOVA	Analysis of variance
ALT	Altitude
B1	Landsat band 1 (blue)
B2	Landsat band 2 (green)
B3	Landsat band 3 (red)
B4	Landsat band 4 (near infrared)
B5	Landsat band 5 (infrared)
CDOM	Colored dissolved organic matter
Chl-a	Chlorophyll-*a*
Chl-a_{obs}	Ground-based measurements of Chl-a
Chl-a_{mod}	Modeled Chl-a
Chl-a yr^{-1}	Rate of change (slope) of Chl-a
CV	Coefficient of variation
DEM	Digital Elevation Model
DIC	Dissolved inorganic carbon
DOC	Dissolved organic carbon
DOM	Dissolved organic matter
DN	Digital numbers
DON	Dissolved organic nitrogen
DIN	Dissolved inorganic nitrogen
DR	Dynamic ratio
ETM+	Enhanced TM (ETM+) sensor
IKONOS	The name of the satellite, from Greek *eikōn* "An image”

LAT	Latitude
LZ	Littoral zone (in lakes)
Max	Maximum
MEI	Multivariate El Niño Southern Oscillation Index
m a.s.l.	Meters above mean sea level
MERIS	Medium Resolution Imaging Spectrometer
MODIS	Moderate Resolution Imaging Spectrometer
N	Nitrogen
NO_3^-	Nitrate
NAO	Northern Atlantic Oscillation
P	Phosphorus
PDO	Pacific Decadal Oscillation
Pr	Precipitation
Pr yr^{-1}	Rate of change (slope) of precipitation
r	Pearson correlation coefficient
r^2	Coefficient of determination
RMSE	Root means squire error
SD	Standard deviation
SD_{mv}	Standard deviation moving window (of residuals from Chl-a)
SD_5	5-year standard deviation moving window (of residuals from Chl-a)
SPOT	The name of the satellite, from French *Satellite Pour l'Observation de la Terre* "Satellite for observation of Earth"
TDP	Total dissolved phosphorus
TM	Landsat Thematic Mapper sensor
T_{max}	Maximum air temperature

T_{max} yr^{-1}	Rate of change (slope) of maximum air temperature
TOA radiance	Top of atmosphere radiance
TOA reflectance	Top of atmosphere reflectance
TP	Total phosphorus
TSS	Total suspended solids
V	Lake volume
W%	Wetland cover (in catchments)

1 Introduction

1.1 Problem statement

Despite continued efforts to understand drivers of phytoplankton biomass in freshwater ecosystems, a more complete understanding of their nature remains challenging (Baines *et al.*, 2000; Kosten *et al.*, 2012; de Senerpont Domis *et al.*, 2013). Recently, climate change (climate warming, in particular) has been implicated in the increase in phytoplankton biomass, changes in lake trophic state and production of algal blooms, especially in remote lakes located on relatively pristine landscapes with no history of direct discharge of chemical fertilizers (Scheffer & Van Nes, 2007; Capon & Bunn, 2015; Randsalu-Wendrup *et al*., 2016; Sinha *et al.*, 2017). Climate is a temporally dynamic mixture of non-stationary patterns (trends) and stationary signals (cycles). As a result, understanding climatic controls on lake ecosystems is challenging (Capon *et al.*, 2015). Further, climate changes in terms of rising air temperature and changing precipitation patterns provides little explanation to why some lakes from the same geographical area experience an increase in phytoplankton biomass while others do not (Oliver *et al.*, 2017; Richardson *et al.*, 2018). Landscape features–catchment and morphometry of lake basins–affect the source, storage and transport of water and nutrients (Baines *et al.*, 2000; Staehr *et al.*, 2012) that are essential for phytoplankton growth (Wetzel, 2001). However, catchment heterogeneity make it difficult to understand the interactive impacts of sources and sinks of nutrients (Fraterrigo & Downing, 2008; Anderson, 2014; Hipsey *et al*., 2015; Capon & Bunn, 2015). Thus, there is a need for detailed analyses of long-term time series (decades) of phytoplankton (or chlorophyll-*a* as a proxy for phytoplankton biomass) in conjunction with time series of climatic drivers (temperature and precipitation) and landscape features to better understand interactions and feedbacks among these environmental variables. Better understanding of these interactions and feedbacks will help shed light on phytoplankton development now and in the future under the reality of climate change.

1.2 Scientific rationale

The shape, productivity and trophic functioning of lakes have changed rapidly in the last 50 years–more rapidly than at any other time in human history (Hipsey *et al.*, 2015). In general, these changes could hardly be called positive as they often lead to an emergence of harmful algal blooms (O'Neil *et al.*, 2012). The frequency and duration of harmful algal blooms is increasing globally (Svrcek & Smith, 2004; Carey *et al*., 2012) as well as within the temperate forest biome of North America (Winter *et al*., 2011). This is possible evidence of eutrophication of lakes, and the shifts towards nutrient-rich condition. While eutrophication has long been ascribed to either direct discharge of waste products and chemical fertilizers into surface waters (Glibert *et al*., 2005) or land cover changes (e.g., deforestation, wetland drainage; Foley *et al*., 2005), there is incomplete understanding of the factors leading to algal blooms in lakes that have never recorded eutrophic conditions (Winter *et al*., 2011; Carey *et al*., 2012). Newly eutrophied lakes are located on relatively undisturbed landscapes at considerable distances from urban areas and agricultural lands–such as those within the temperate forest biome in central Ontario (Winter *et al*., 2011). Further, the temperate forest biome rests on phosphorus-poor Precambrian rocks of the Canadian Shield (Ontario Geological Survey, 2003)–the landscape that should not have the natural capacity to support lakes with a high trophic conditions. Thus, it is becoming clear that algal blooms are no longer a strict anthropogenic eutrophication (nutrient enrichment) problem (Paerl & Huisman, 2008; Posch *et al.*, 2012). Existing conceptual models that attempt to describe the factors regulating the trophic state of lakes and drivers contributing to a change in phytoplankton biomass are insufficient to explain eutrophication in these remote temperate lakes.

With the absence of direct anthropogenic activities on relatively undisturbed landscape of the temperate forest biome in central Ontario, the recent reports on algal blooms may be partly explained by climate-associated temporal and landscape-related spatial factors. Although both direct (e.g., Blenckner, 2005; Adrian *et al.*, 2009; Posch *et al.*, 2012; Richardson *et al*., 2018) and indirect (via regional hydrology influencing water and nutrient transport; e.g., George *et al*., 2008; Whitehead *et al*., 2009) effects of climate change on lake ecosystems have been widely described, the role of climate change as a regulator of phytoplankton biomass and eutrophication remains poorly understood (Scheffer & Van Nes, 2007). For example, although some studies suggest that climate warming might promote the turbid (eutrophic) state in temperate lakes (Jeppesen *et al.,* 2003; Mooij *et al*., 2007), there is evidence that it might favor the clear state (Rooney & Kalff, 2000; Lottig *et al*., 2014). Some recent studies suggest that although increasing air temperature should be taken into account, changes in precipitation patterns might be more important in driving eutrophication (Sinha *et al.,* 2017).

The characteristics of the contributing source areas of water and nutrients (i.e., catchment size, topography, presence of wetlands) affect the source, storage and transport of water and nutrients to lakes (Blenckner, 2005; Staehr *et al*., 2012). In addition, the characteristics of the receiving waters (e.g., lake depth, volume, size of littoral zone) affect the fate of the nutrients within lakes (Søndergaard *et al.*, 2005; Nõges, 2009; Orihel *et al*., 2017). Only a few studies have assessed the coupled terrestrial-aquatic systems upon which lake ecosystems depends (e.g., Anderson, 2014; Hipsey *et al*., 2015). From these studies, there is evidence that higher proportions of wetlands in lake catchments contribute to maintaining either turbid or clear state in lakes depending on the location of the wetlands (e.g., upstream or downstream) and lake basin morphometry (Cobbaert *et al.,* 2015). Further, despite the fact that the lake's littoral zone is known to be an important sink for allochthonous nitrogen (N) and phosphorus (P) (Klimaszyk *et al.*, 2015), there is incomplete understanding of the role of this zone in providing a source of nutrients to phytoplankton (Kornijów *et al.,* 2016).

Eutrophication and an increased frequency of algal blooms might also indicate that temperate lakes are experiencing functional changes in their inherent properties. The concept of ecological resilience developed by Holling in 1973 (Holling, 1973) describes conditions in which an ecosystem loses its resilience, becomes unstable, and shifts into another regime of behavior (or stability domain), therefore implying that the ecosystem can have at least two stable states separated by unstable or transitional state(s). Holling (1973) defined resilience as the amount of disturbance that a system can withstand while keeping the same structure and function before it shifts into an alternative stable state. Ecosystem stability can be defined as the ability of a system to remain relatively unchanged under perturbation, and to return to the initial state quickly once the perturbation is over (Angeler & Allen, 2016). Over time, the structure and function of an ecosystem with high resilience remain relatively stable. However, gradually changing external conditions (i.e., an enduring pressure such as increasing air temperature) can lead to a gradual loss of resilience up to a point where even a small disturbance can push the system into a new stability domain, where the system reorganizes into a new stable (often radically different) state (Scheffer *et al.*, 2012)**.** Once in a new stable state, the system is maintained by internal feedback dynamics (e.g., prevalence of buoyant cyanobacteria), making the recovery to a previous state difficult (Scheffer *et al*., 2001; Scheffer *et al*., 2012).

Changes in the biomass of the phytoplankton and the associated lake trophic states are assessed using chlorophyll-*a* concentration (Chl-a)–the proxy for phytoplankton biomass (Thiemann & Kaufmann, 2000). However, ecological time series (such as time series of Chl-a) are typically too short and noisy to draw robust statistical measures, especially when analyzing resilience and stability of system states (Carpenter & Brock, 2006; Lenton *et al.*, 2012; Boettiger *et al.*, 2013). Therefore, it is important to use long-term (decades) time series and filter signals resulting from intrinsic ecosystem dynamics from various kinds of environmental noise including non-stationary and stationary signals (Lenton *et al.*, 2012; Arnoldi *et al.*, 2016).

Here a multi-scale approach (space: lake-catchment-region, and time: from one year to year 28) and statistical techniques are used to explore spatial and temporal patterns in phytoplankton biomass (measured as Chl-a) in thousands of small (< 10,000 ha) lakes in a large area of the temperate forest biome in Ontario, Canada. Although, the temperate forest biome is relatively pristine as a whole, there are some areas (e.g., some lowlands along Lake Huron, and the Greater Sudbury region) that have a greater intensity of anthropogenic activities (e.g., forest management, mining); these areas were still included in the analysis for comparison purposes. The temperate forest biome shows noticeable annual climate variability and landscape spatial heterogeneity in terms of topography (from lowland areas along the Great Lakes to Algoma and Madawaska Highlands), morphometry of lake basins (e.g., maximum depth ranges from 1 m to 59 m), and lake trophic states (from oligotrophic to hyper-eutrophic). The 30-year climatic record (McKenney *et al.*, 2011) indicates that, since 1984, average air temperature has increased by 2°C in the temperate forest biome, while annual mean precipitation has decreased by almost 20 mm in the central-northern areas and increased by 10 mm in the southern areas of the region.

1.3 Necessitated techniques

Traditional field sampling and therefore monitoring of lake phytoplankton biomass (or Chl-a) in lakes is often logistically limited (especially for remote northern areas). Furthermore, even in logistically accessible areas (e.g., southern regions of the temperate forest biome), representativeness, spatial and temporal coverage, and frequency of filed measurements are usually inadequate (Palmer *at al.*, 2015). Satellite missions and the availability of satellite imagery data since the 1970s (e.g., data provided by Landsat 1), however, allow for the estimation of phytoplankton biomass over large spatial extents and over long periods of time at relatively low costs.

Remote sensing methods rely on the measurement of radiation received from the surface of Earth in particular areas (i.e., bands) of the electromagnetic spectrum (Matthews, 2011). Satellite sensors detect the fraction of incoming solar irradiance reflected by a subject (e.g., water body) or a constituent (e.g., Chl-a), which is defined as reflectance (Dall'Olmo *et al.*, 2003). Phytoplankton detection is possible because all phytoplanktonic organisms have spectrally active photosynthetic pigments, such as Chl-a. Chl-a concentration is the most common parameter derived in remote sensing of inland waters that is used as an indicator of the abundance of phytoplankton in water and a proxy of lake trophic condition (Han & Jordan, 2005; Matthews, 2011). The absorption and reflectance characteristics of Chl-a are: strong absorption between 400–500 nm (blue) and at near-680 nm (red), and reflectance maximums at near-550 nm (green) and 700 nm (near-infrared: NIR) (Figure 1.1; Han & Jordan, 2005). However, due to the optical complexity of inland waters, these characteristics may differ from lake to lake; for example, the Chl-a reflectance maximum can shift to the longer wavelength (red) in turbid lakes owing to the presence of particulate material in the water column (Spitzer & Dirks, 1986). Satellite sensors do not directly measure Chl-a concentration; instead it is usually estimated through empirical models based on correlations of band reflectance values with near-simultaneous ground-based measurements (Gitelson *et al.*, 2000).

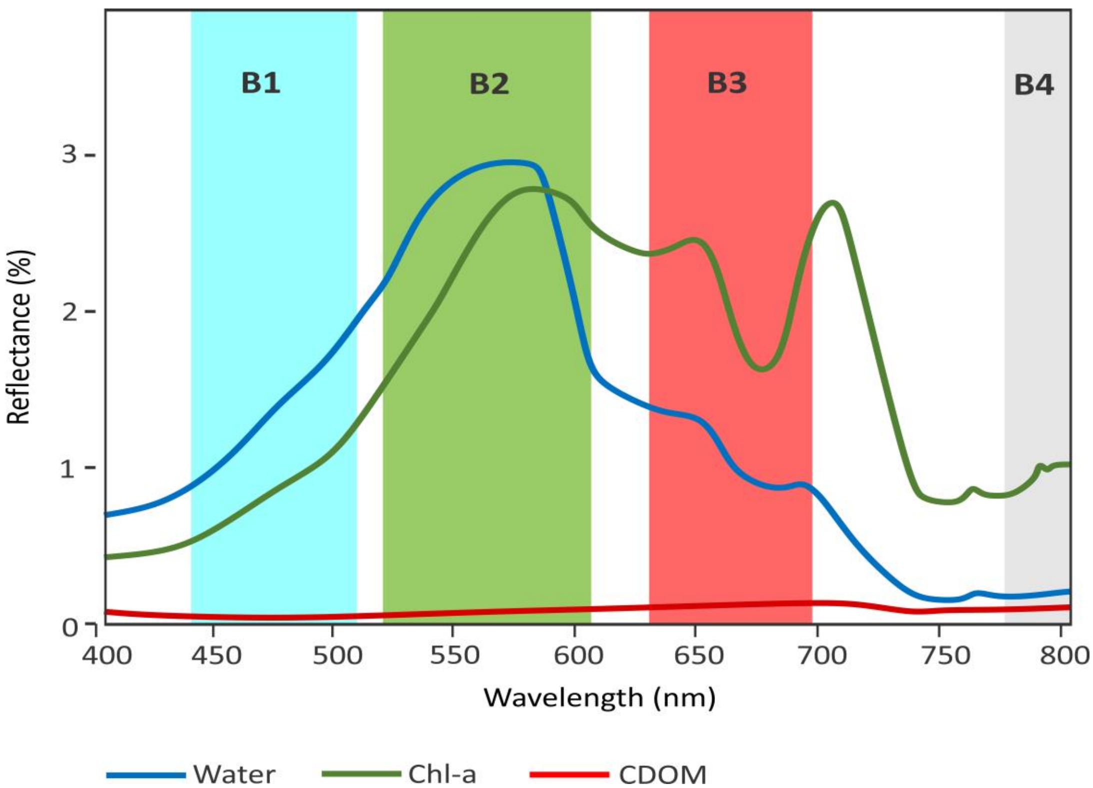

Figure 1.1 Reflectance spectra of water, Chl-a and CDOM. Colors symbolize Landsat TM/ETM+ bands: blue–B1, green–B2, red–B3, and gray–B4 (near-infrared) (modified from Olmanson *et al.*, 2016).

These days there are numerous satellites acquiring imagery at various spatial resolutions (e.g., Landsat, SPOT, MERIS, MODIS, IKONOS, etc.). However, most have relatively coarse spatial resolution (usually between 250 m and 1000 m) that does not allow for modelling Chl-a in small inland waters. Additionally, these satellites have only recently started to operate (e.g., MODIS and IKONOS were launched in 1999); therefore, they cannot be used for long-term (decades) monitoring of lake phytoplankton biomass (or Chl-a). Landsat Thematic Mapper (TM) and Enhanced TM (ETM+) sensors, on the other hand, have adequate spatial resolution (30 m) and provide a continuous record of satellite-based data between 1984 and 2011. Additionally, Landsat images and all associated data are available free of charge and can be uploaded directly from the U.S. Geological Survey (USGS) official website upon request.

Despite the benefits, there are several challenges in using Landsat sensors for Chl-a retrieval in inland waters that should be accounted for. First, Landsat sensors are primarily designed for terrestrial landscapes; therefore, application of atmospheric correction methods associated with these sensors (e.g., dark object subtraction methods – DOS and COST) over lakes may impact the performance of Chl-a retrieval algorithms (Palmer *at al*., 2015). One possible solution to this problem is a partial atmospheric correction that does not require the selection of the dark objects (Guanter *et al.*, 2010; Keith *et al*., 2018). Second, Dekker *et al*. (2002) pointed out that owing to Landsat's coarse spectral resolution, its sensitivity to spectral differences is relatively low, which can lead to more severe adjacency effects. The majority of inland waters are small and relatively shallow (Wetzel, 2001); therefore, there is a possibility of erroneous reflectance values originated from pixels adjacent to shorelines (littoral zones with abundant aquatic vegetation) and sediments (shallow areas). In this case, careful selection of lakes in terms of their size (e.g., using a criterion of minimal lake size) and depth, as well as removal of littoral zones from lakes, is strongly recommended (Verpoorter *et al.*, 2012).

Finally, reflectance and absorption spectra of Chl-a and colored dissolved organic matter (CDOM) overlap on the electromagnetic spectrum (Figure 1.1). Landsat's course spectral bands cannot resolve Chl-a narrow reflectance peaks only (Matthews, 2011). Therefore, there is a concern that covarying effects of CDOM and some other surface water constituents (e.g., total suspended solids: TSS) can hamper the interpretation of reflectance values associated with Chl-a (Dekker *et al*., 2002; Brezonik *et al*., 2005). CDOM is predominantly comprised of humic and fulvic acids originated from decomposition of plant material in soils and wetlands (Brezonik *et al*., 2005). Humic components absorb strongly in the blue band, turning the water brown; therefore, they might be a significant contributor to water color, especially if the concentration of these components is high (Matthews, 2011). One possible solution to minimize the effect of these constituents is using empirical approaches such as band ratios or band algorithms (especially three band algorithms) instead of using single bands (Östlund *et al*., 2001; Vincent *et al*., 2004; Keith *et al*., 2018). These algorithms can also eliminate some residual errors of atmospheric correction (Stumpf *et al.*, 2016). This is because while dividing one reflection by another, surface reflection and atmospheric influence are likely to be constant with wavelength being removed, and therefore the ratio is primary impacted by the water leaving radiance (Strömbeck & Pierson, 2001).

1.4 Research foundation

The research foundation for this thesis was study done by Paltsev (2015). The author used Landsat TM/ETM+ imagery to analyze natural variation in modelled Chl-a concentration in more than 6,000 temperate lakes. In this study, I adopted several approaches initially developed by Paltsev (2015). For example, I also used remote sensing (Landsat series in particular), a linear regression for Chl-a modelling, and the analysis of variance (ANOVA) to decompose the variation in Chl-a into three components (i.e., space, time and space×time interaction) according to Wiley *et al.* (1997). However, in this study, not only different methods were used for pre-processing of Landsat images but also more attention was payed to the selection of the Landsat spectral algorithm that would be more appropriate for Chl-a modelling, considering spectral properties of Landsat series and possible interference of Chl-a with other water constituents (e.g., CDOM).

Full atmospheric correction methods such as those used in Paltsev (2015; COST) are primarily designed for land applications (see Palmer *at al*., 2015). A partial atmospheric correction, on the other hand, is thought to be more appropriate when dealing with optically complex inland waters (Guanter *et al.*, 2010); hence, a partial atmospheric correction was applied in this study. Although single bands were used for Chl-a retrieval in the past (e.g., Sass *et al*., 2007), recently developed band ratio (or band algorithm) and semi-analytical methods are found to produce more accurate models, which take into account covarying effects of CDOM and TSS (Odermatt *et al.*, 2012; Keith *et al*., 2018). Therefore, a band algorithm was used in this study as compared to a single band (band 3: red) used in Paltsev (2015). Furthermore, Paltsev (2015) did not solve the problem of missing Chl-a values, which were a result of erroneous reflectances caused by haze and clouds on Landsat images. This decreased the number of lakes used for temporal analysis to around 6,000, and may have produced biased results while analyzing spatial patterns in Chl-a (i.e., averaged values over time). In this study, this flaw was corrected by applying a kriging technique (Cressie & Wikle, 2011) that resulted in more than 12,000 lakes with continuous Chl-a time series. Finally, I performed more in-depth analyses of temporal patterns in Chl-a by applying the non-parametric Mann-Kendall test on individual lakes. This allowed me to identify lakes with significant trends in Chl-a over 28 period, and describe spatial patterns that these “trending lakes” had (e.g., the proximity to Sudbury, cottage regions, and the Great Lakes).

1.5 Thesis goal, objectives and hypotheses

The goal of the thesis was to improve understanding of the interactive effects of climate changes and landscape properties on phytoplankton biomass in lakes located in intact forested landscapes in the temperate forest biome.

The following objectives were completed and associated hypotheses and predictions were assessed to reach this goal.

Objective 1. Describe the spatial and temporal patterns in lake Chl-a and determine the total variation in the Chl-a in space and time.

I hypothesized that there are temporal (trends) and spatial (clusters) patterns in Chl-a and associated trophic states in lakes of the study region. I predicted, however, that most of the variation in Chl-a will be due to lake-specific factors (e.g., lake morphometry) which will not produce any visible “broad-scale” patterns.

Objective 2. Explore the role of climate factors and landscape characteristics on lake Chl-a.

I hypothesized that there is a relationship between Chl-a (and associated trophic states) and landscape properties that cause different patterns in nutrient loading into lakes and nutrient availability within lakes. I predicted that lakes with similar landscape properties will respond coherently to increasing temperature and changing precipitation.

Objective 3. Assess ecosystem stability of lakes and determine lakes experiencing regime shifts in lake trophic states.

I hypothesized that alternative stable states exist in the lakes of the study region. I predicted that there will be two stable stables–oligotrophic and eutrophic–and several transitional (e.g., eutrophying and/or oligotrophying) and/or unstable state(s).

Objective 4. Explore the role of climate in contributing to lake instability and the rationale between changing trophic state in some lakes (eutrophying or oligotrophying lakes) with lakes expressing a stable state.

I hypothesized that there is relationship between climate (in terms of increasing temperatures and changing precipitation patterns), landscape properties and lakes that are eutrophying or oligotrophying. I predicted that increasing temperatures are driving the eutrophication of some lakes, while increasing precipitation and associated increased runoff is driving the oligotrophication of other lakes.

1.6 Thesis organization

This thesis has been prepared in the integrated article format and is comprised of three manuscripts (the first manuscript is related to Objective 1, the second manuscript is related to Objective 2, and the third manuscript is related to Objectives 3 and 4). The introduction (Chapter 1) provides an overview of the theoretical approach, the research problem, questions, hypotheses and objectives that form the basis of the thesis. The first manuscript (Chapter 2) presents a method to estimate Chl-a concentration from remote sensing imagery and applies the method to estimate Chl-a from archived Landsat imagery from 1984 to 2011 for thousands of temperate lakes. This chapter also characterizes the variation in Chl-a by using a statistical approach for decomposing variance into spatial, temporal, and space×time interaction components. The second manuscript (Chapter 3) examines relationship between Chl-a of 275 representative lakes and climate drives (air temperature and precipitation) and landscape (catchment and lake morphometric features) properties. Detailed descriptions of “typical” landscape features identified for each lake trophic state are provided in this chapter. The third manuscript (Chapter 4) applies an analytical framework to identify indicators of changes in lake ecosystem stability (instability) and regime shifts by analyzing anomalies; i.e., differences in behavior of residuals from Chl-a time series of 12,644 lakes. Additionally, this manuscript investigates the role of climate (in terms of air temperature and precipitation) and landscape characteristics as drivers of changes of lake stability in a subset of 78 lakes experiencing transitional states. The final chapter (Chapter 5) summarizes the major conclusions of the study, discusses the anticipated significance, and presents future research directions.

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2 Understanding patterns in remotely-sensed Chlorophyll-*a* in temperate lakes: spatial and temporal perspectives

2.1 Introduction

Biological communities in lakes can be sensitive to environmental changes (Adrian *et al.*, 2009). Small and shallow lakes are especially sensitive to these changes (Choi, 1998); those located in remote areas can provide abundant information about the particular effects of these changes in the absence of confounding anthropogenic land cover signals. Of various "lake sentinels", phytoplankton is of special interest because this group of photosynthetic organisms can respond rapidly to environmental changes (Williamson *et al.*, 2009), especially if these changes lead to eutrophication. The most notorious response of phytoplankton to eutrophication is the increasing emergence of harmful algal blooms (HABs) (O'Neil et al., 2012) which are frequently comprised of toxin-producing cyanobacteria (cyanoHABs) (Havens, 2008).

In contrast with natural eutrophication processes that occur over hundreds or thousands of years (Wetzel, 2001), anthropogenic eutrophication can happen within much shorter periods of time (e.g., decades or years). Anthropogenic eutrophication is usually ascribed to either direct discharge of waste products and chemical fertilizers into surface waters (Glibert *et al.*, 2005) or land cover changes (e.g., deforestation, wetland drainage) (Foley *et al.*, 2005). However, these explanations provide no insight into the emergence of lake eutrophication in areas that have never recorded eutrophic conditions (i.e., areas located on relatively undisturbed landscapes at considerable distances from urban areas and agricultural lands). Stoddard *et al.* (2016) found a dramatic reduction in the number of oligotrophic lakes in the United States located within relatively undisturbed catchments, possibly as a result of increased atmospheric deposition of phosphorus. Climate change (and, in particular, climate warming) also causes the raising of surface water temperatures and strengthening of the vertical stratification of lakes that are advantageous to many bloom-forming cyanobacteria species (Paerl & Huisman, 2008).

Given these phenomena, it is important to gain understanding of the relative contributions of spatial and temporal factors to lake eutrophication. This may be achieved only through long-term surveys (time series) of phytoplankton biomass covering large regional scales. These are not directly available due in part to the laborious nature of systematic sampling in difficult-to-access locations. The routine availability of remote sensing imagery since the 1980s, however, can allow for the estimation of phytoplankton biomass and, by extension, lake trophic state over large spatial extents over extended time periods. Phytoplankton detection is possible through the optical properties of a spectrally active pigment of phytoplankton: Chlorophyll-*a* (Chl-a). Chl-a has been widely used as a proxy of phytoplankton biomass (e.g., Wynne *et al.* 2013; Kudela *et al.*, 2015) and is often denoted as a biological indicator of lake trophic state (e.g., Thiemann & Kaufmann, 2000; Song *et al.*, 2013).

A number of satellite sensors have been used for Chl-a quantification in inland waters (reviewed by Matthews, 2011). Some sensors (e.g., MERIS) have fine spectral resolution that allows for measuring distinguishing features of wavelength absorption of the phycocyanin pigment, allowing for the detection of cyanoHABs. Their coarse spatial resolution, however, does not allow for mapping Chl-a concentration in small lakes. Moreover, the relatively short spans of records from these sensors do not allow for construction of long-term surveys. Landsat Thematic Mapper (TM) and Enhanced TM (ETM+) sensors, on the other hand, provide a continuous record of satellite-based observations from 1984-2011 at moderate (30 m) spatial resolution that have been used to successfully quantify phytoplankton Chl-a in lakes (e.g., Lathrop *et al.*, 1991; Svab *et al.*, 2005; Karakaya *et al.*, 2011; McCullough *et al.*, 2012; Tebbs *et al.*, 2013; Giardino *et al.*, 2014).

While Landsat TM/ETM+ data have the potential for estimating Chl-a in lakes, only a handful of studies have used the data to quantify Chl-a concentration in large numbers of lakes covering a range of trophic states (e.g., Allan *et al.*, 2011; Torbick *et al.*, 2013). Even fewer studies have used time series of Landsat TM/ETM+ data to map long-term Chl-a patterns (e.g., Sass *et al.*, 2007; Allan *et al.*, 2015). The most common approach used in applications of Landsat TM/ETM+ to estimate Chl-a is to develop empirical models that involve statistical association of satellite reflectance from wavelength band(s) with near-simultaneous ground-based Chl-a measurements (Gitelson *et al.*, 2000).

However, since Landsat TM/ETM+ are not initially designed for Chl-a retrieval, their spectral bands do not exactly correspond to Chl-a absorption and reflectance peaks; this often limits the ability of this satellite series to accurately model Chl-a concentration (Palmer *et al.*, 2015).

One possible solution is to use band ratios or band algorithms instead of single bands (Odermatt *et al.*, 2012). Band ratios and band algorithms can also offset some residual errors in atmospheric correction (Stumpf *et al.*, 2016) and reduce the effects of other optically active components on reflectance such as colored dissolved organic matter (CDOM) and total suspended solids (TSS) (Brivio *et al*., 2001; Keith *et al.*, 2018). In optically-complex waters (e.g., lakes), the components can seriously complicate the interpretation of reflectance values associated with Chl-a.

In this study, a 28-year (1984-2011) times series of Landsat TM/ETM+ satellite products was used to estimate annual Chl-a concentrations in thousands of small (< 10,000 ha) lakes in a large area of the temperate forest biome in Ontario, Canada. This extensive spatial and temporal dataset was used to determine the relative influences of spatial and temporal factors leading to variations of Chl-a concentration. The specific objectives were as follows: (1) to develop a regression model relating lake Chl-a concentration to TM and ETM+ optical reflectance using a band ratio algorithm; (2) to apply the model to estimate annual Chl-a concentrations in thousands of lakes over a continuous 28-year period, and (3) to decompose the total variation in annual Chl-a concentration into space, time and (space×time) interaction domains. The results of the study will help identify factors associated with increasing phytoplankton biomass and eutrophication and allow researchers to target geographical areas where lakes are more susceptible to eutrophication for future monitoring efforts.

2.2 Study region

The study region is located between 44.44 °N and 48.38 °N in the temperate forest biome within the Boreal (Canadian) Shield in Ontario, Canada (Figure 2.1). Climate in the region is humid continental; precipitation is influenced by the Great Lakes (Baldwin *et al.*, 2000).

Figure 2.1 Study region (the temperate forest biome) and locations of sampled Ontario and Alberta lakes.

Mean annual air temperature in the study region for the period of 1984-2011 was +5.1°C, ranging between +7.4°C in the south-east and +2.8°C in the north. Mean annual total precipitation for the same period was 960 mm yr^{-1}, ranging from 740 mm yr^{-1} in the southern areas of the region to 1180 mm yr^{-1} in the north-west (McKenney *et al.*, 2011).

Mean annual July-October (i.e., months that are under consideration in the study – see Chapter 3 for details) maximum temperatures increased significantly at a mean rate of 0.046°C yr^{-1} over the 1984–2011 period ($p < 0.05$). Trends in mean annual July-October total precipitation for the same period are less clear; the precipitation was variable from year to year with decreasing trends (a mean rate of -0.24 mm yr^{-1}) in western areas and increasing trends (a mean rate of 0.17 mm yr^{-1}) in central and south-eastern areas of the study region. The frost-free period extends from April to November in the warmer and more humid southern portions of the region, and from May to September in the northern portions (Baldwin *et al.*, 2000).

Bedrock geology of the study region is primarily composed of silicate greenstone with outcrops of more felsic igneous rocks of the Precambrian origin (Ontario Geological Survey, 2003). These rocks are covered with glacio-fluvial outwash (average depth is 1-2 m), which consists of sandy loam ablation till with river and deltaic deposits and a compacted lower slit loam basal till (Ontario Geological Survey, 2003; Appendix A: Figure A.1). Organic (Holocene) deposits are frequent in depressions and wetlands near rivers and lakes, and are predominantly comprised of peat and muck. Elevations range from 150 to 555 m a.s.l.; topography varies from flats and depressions along the shore of the Great Lakes to uplands (e.g., Algoma Highlands). Soils are thin brunisols in the southern portions of the region, and thick and differentiated orthic ferro-humic podzols in central and northern portions (Canada Soil Survey Committee, 1978). Wetlands cover from a small (< 3%) to a substantial part (25%) of lake catchments (average wetland cover is 12%; Eimers *et al.*, 2009) and are generally comprised of ferric humisols with highly humified organic deposits (Canada Soil Survey Committee, 1978). Forests in the region belong to the Great Lakes-St. Lawrence Forest Biome and lie in a transitional zone between deciduous and coniferous, with the latter being more prevalent in the northern areas.

2.3 Materials and Methods

2.3.1 Ground-based Chlorophyll-*a* and DOC measurements

Ontario lakes

Ground-based measurements of Chl-a concentration (Chl-a_{obs}) were made throughout the ice-free season (May-October) in 26 lakes of the study region located in the Algoma Highlands in Ontario (hereafter referred to as Ontario lakes) during a three-year (2009 to 2011) field campaign conducted by Ryan Sorichetti (Western University) (Sorichetti *et al.*, 2014; Appendix A: Table A.1, Figure A.1). Of 26 sample lakes, 9 lakes were sampled in more than one year, making the total sample size equal to 35. As a whole, the Algoma

Highlands have similar physiographic characteristics (i.e., geology, topography, soil and forest type) as the entire study region. Surficial geology of the Algoma Highlands is represented by double layered glacial till (sandy loam ablation till with overlying a compacted lower silt loam basal till). Soils are orthic ferro-humic podzols with dispersed pockets of ferric humisols. Topographic relief of this region ranges from hills (with gentle to steep hillslopes) to flats and depressions containing mineral or organic soils, which are often saturated. Forests are comprised of deciduous and coniferous species with the former being more prevalent (Mengistu *et al.,* 2014).

The sampled lakes are predominantly oligotrophic and mesotrophic with respect to Chl-a concentration (Carlson & Simpson, 1996), with maximum depth between 1.3 m to 42.7 m (average maximum depth is 7.6 m) and surface area between 16.5 ha to 1033.0 ha (average area is 143.6 m), thermally stratified during summer and mixed during spring snowmelt and fall storms (Appendix A: Table A.1, see also Sorichetti, 2014). Water samples integrated to 1.0 m depth were collected at lake centers (which were assumed to be the deepest part of lakes) outside of a phytoplankton bloom if present (Sorichetti, 2014). The samples were collected in 500 mL pre-rinsed polyethylene bottles, stored on ice, filtered onto Whatman GF/F filters, and analyzed for Chl-a concentration ($\mu g\ L^{-1}$) using a Turner 10-AU field fluorometer (excitation at 436 nm, emission at 680 nm) (Arar & Collins, 1997).

Alberta lakes

Chl-a_{obs} collected in mid-August of 1999, 2001 and 2002 in 54 lakes of the Utikuma Uplands located in the Boreal Plain ecozone of northern Alberta by Sass *et al.* (2007) was also used in this study (hereafter referred to as Alberta lakes; Appendix A: Table A.1, Figure A.2). Of 54 lakes, 8 lakes were sampled in more than one year, making the total sample size equal to 69. The Utikuma Uplands are considered relatively intact from human activities. The region is primary comprised of various glacial landforms ranging from moraine forms (hummocky regions with silt and clay) to outwash plain (with sand), to lacustrine plain (flats with clay) (Sass *et al.*, 2007; Figure A.2). The sample lakes ranged from oligotrophic to hypereutrophic (with respect to Chl-a concentration; Carlson & Simpson, 1996) with mean depth between 0.5 m to 2.0 m and surface area between 4.5

ha to 275.0 ha (average area is 30.5 m) (Table A.1; also see Sass, 2006). Water samples were collected at 0.2-0.4 m depth at lake centers (outside of algal blooms if present), filtered, frozen, and extracted with acetone. The extract was then analyzed for Chl-a concentration (μg L^{-1}) using a spectrophotometer at 750, 665 and 649 nm wavelengths according to EPA Method 446.0 (Bergmann & Peters, 1980).

Dissolved organic carbon (DOC) measurements

Dissolved organic carbon (DOC) is often used as a proxy for measuring CDOM in lakes (which is the photo-active component of DOC) (Brezonik *et al*., 2005; Zhu *et al.*, 2014). DOC was measured in both Ontario and Alberta lakes using a standard 0.45 μm filter from a sub-sample of the water collected for Chl-a determination. DOC concentration (mg L^{-1}) was determined using infrared detection (Shimadzu TOC 5000A, detection limit of 4 ppb). There were 23 DOC samples from Ontario lakes and 51 DOC samples from Alberta lakes. Lakes of the Boreal Plain are known to have elevated concentration of DOC (Bayley & Prather, 2003). However, Sass *et al.* (2007) did find any statistically significant correlation between Landsat band 3 (B3 – the band the authors used for Chl-a retrieval) and DOC; therefore, the authors concluded that DOC from sample lakes did not have a detectable influence on B3 and Chl-a retrieval procedure.

Details about concentration of Chl-a and DOC for Ontario and Alberta lakes are presented in Appendix A: Table A.1.

2.3.2 Landsat data acquisition and processing

1,067 Landsat TM (1984-2011) and 159 ETM+ (1999-2003) images intersecting the locations of ground-based measurements and containing less than 50% cloud or haze cover were acquired from US Geological Survey archives for the period from August to October (the period of the peak phytoplankton biomass known for the study region (Winter *et al.*, 2011). Several Landsat processing steps were undertaken, as follows:

(1) Bands 1-5 of Landsat images (blue: 0.45-0.52 μm; green: 0.52-0.60 μm; red: 0.63-0.69 μm; near infrared: 0.76-0.90 μm wavelengths; shortwave infrared: 1.547-1.749 μm) were radiometrically normalized (i.e., made radiometrically comparable) by converting 8-

bit scaled and offset stored digital numbers (DN) to top of atmosphere (TOA) radiance values (L_{sat}):

$$L_{sat} = (DN - B)/G \quad [2.1]$$

where B and G are published post-launch image gain and bias provided in image metadata.

(2) Since the full atmospheric correction over inland waters could result in erroneous reflectance values (due to high uncertainty in accounting for aerosol scattering; Guanter *et al.*, 2010; Lobo *et al.*, 2015), a partial atmospheric correction was applied in this study. This included calculation and subsequent subtraction of the Rayleigh scattering radiance from the total TOA radiance (L_{sat}).

Rayleigh scattering contribution was calculated for each Landsat band using the formula given by Gilabert *et al.* (1994):

$$L_r = \left\{\frac{(E_0 * \cos\theta_0 * P_r)}{4\pi(\cos\theta_0 + \cos\theta_v)}\right\} * \left\{1 - \exp\left[-\tau_r\left(\frac{1}{\cos\theta_0} + \frac{1}{\cos\theta_v}\right)\right]\right\} * t_{oz} \quad [2.2]$$

where:

L_r is atmospheric radiance due to Rayleigh scattering;

E_o is solar irradiance at the top of the atmosphere;

P_r is Rayleigh scattering phase function;

Θ_0 is the solar zenith angle and Θ_v is the view zenith angle;

τ_r is Rayleigh optical thickness; and

t_{oz} is ozone transmittance.

The solar zenith angle was obtained from image metadata, while the view zenith angle was equal to scattering angle (Ω), which is: 180–Θ_0 (Gilabert *et al.*, 1994). The Rayleigh scattering phase function was calculated following Chandrasekhar (1960):

$$P_r = \frac{3}{4} * \frac{1-\gamma}{1+2\gamma}(1 + \cos^2\Omega) + \frac{3\gamma}{1+2\gamma} \quad [2.3]$$

where γ is a term used to account for the depolarization factor p_n following Bucholtz (1995):

$$\gamma = p_n/(2 - p_n) \quad [2.4]$$

p_n was obtained from Bucholtz (1995) for band 1–4.

The Rayleigh optical thickness was calculated using an approximate expression provided by Gilabert *et al.* (1994):

$$\tau_r = 0.008569\lambda^{-4}\,(1 + 0.0113\lambda^{-2} + 0.00013\lambda^{-4}) \qquad [2.5]$$

where λ is the central wavelength of each band (bands 1–4).

Ozone transmittance was calculated according to Bird & Riordan (1986):

$$t_{oz} = \exp(-A * 0_3 * M) \qquad [2.6]$$

where A is the ozone absorption coefficient, O_3 is the ozone amount, and M_o is the ozone mass. O_3 was assumed to be 0.3 atm cm^{-1} (Jorge *et al.*, 2017), while A was obtained from Bird & Riordan (1986), and M was calculated following the same authors:

$$M = \left(1 + \frac{h}{6370}\right)\left(\cos^2\Theta_0 + \frac{2h}{6370}\right)^{0.5} \qquad [2.7]$$

where h is the height of the maximum ozone concentration, assumed as 22 km (Bird & Riordan, 1986).

The Rayleigh-corrected radiance values ($L_{sat\text{-}r}$) were then calculated as:

$$L_{sat-r} = L_{sat} - L_r \qquad [2.8]$$

(3) $L_{sat\text{-}r}$ were converted to TOA unitless reflectances (ρ_ρ), following Chander *et al.* (2009):

$$\rho_\rho = \frac{\pi * L_{sat-r} * d^2}{E_{sun} * \cos\Theta_0} \qquad [2.9]$$

where d is the earth-sun distance in astronomical units (taken from lookup tables according to image capture date); E_{sun} is an exoatmospheric solar constant (taken from lookup tables according to satellite sensor).

(4) Besides the effects of Rayleigh scattering, the sun-glint effect was considered for deriving accurate TOA radiances. According to Mustard *et al.* (2002) and Dekker & Hestir (2012), sun-glint effects are avoided if solar zenith angles are constrained to angles between 30° and 60°. Therefore, all Landsat images were manually checked to identify those corresponding to solar zenith angles falling beyond the 30° to 60° constraint. Although no images with solar zenith angles less than 30° were found, there were four images (all taken on October 30 or 31) that had solar zenith angles greater than 60°. These images were discarded from further analysis. These four images fell in the years

that had two more available images in each year for the August-October time period, so the removal of the former from the dataset did not result in missing data.

2.3.3 Lake identification

Pixels accounting for surface water in each image were identified from the local minimum in the bimodal histogram distribution of band 5 DNs; as shortwave infrared radiation is strongly absorbed by water, numbers below the minimum were classified as surface water (Frazier *et al.*, 2003). Contiguous water pixels were then converted to polygons. Because water polygons accounted for not only lakes but also for other water features such as rivers and streams, non-lake polygons were manually removed.

A software package Fmask 3.2 was used to generate cloud and cloud shadow masks from DNs (Zhu & Woodcock, 2012). In this package, the physical properties of clouds (e.g., temperature, brightness) and the darkening effect of cloud shadows in band 4 are used to classify cloud and cloud shadow pixels. Fmask was also used to detect pixels representing snow; this was important for analyzing images captured in late October in the norther areas of the region. Those lake polygons that overlapped or intersected any pixels classified as cloud, cloud shadow or snow were removed.

2.3.4 Lake selection for regression modeling

Several authors (e.g., Kloiber *et al.*, 2002**;** Verpoorter *et al.*, 2012) have highlighted potential errors in the prediction of lake parameters (e.g., Chl-a) in small waterbodies because of potential errors (i.e., mixed reflectance values due to adjacency effects) where mixed reflectances appear in pixels adjacent to shorelines (littoral zones – areas with shallow water and/or areas with aquatic vegetation). In order to reduce this problem, a minimum lake area criterion of 4.5 ha (30 m × 30 m pixels) was applied; lake polygons with a smaller area were discarded. The remaining lake polygons were buffered inside to a distance of 15 m (1/2 pixel distance); this further minimized the potential effects of mixed reflectance pixels from lake shorelines.

Further, potential errors can also arise from mixed reflectance pixels from sediments or emergent aquatic vegetation in shallow lakes, or in deep lakes with patchy phytoplankton.

Therefore, I applied an additional criterion by calculating the standard deviation (SD) of band 5 TOA radiance values in each buffered lake polygon (Sass *et al.*, 2007) and selecting those polygons with SD lower than or equal to the median SD of all lakes in an image (the remaining lake polygons were discarded). This criterion is based on the assumption that reflectance in band 5 is minimal in deep and clear water bodies due to strong absorption in this band (Smith & Baker, 1981) and hence should have relatively lower heterogeneity that can be expressed as low SD (Sass *et al.*, 2007).

Of 104 lake samples (both Ontario and Alberta lake datasets combined), 53 samples were matched with their lake polygons, meaning that 51 samples were either (1) cloud covered at the time of image capture, or (2) smaller than 4.5 ha, or (3) had high SD of radiance in band 5 (Table A.2). Mean TOA reflectance values for each band 1-4 were extracted within each buffered lake polygon (Table A.3).

2.3.5 Chlorophyll-*a* modeling

Pearson correlation was used to relate values of Chl-a_{obs} and natural log transformed Chl-a_{obs} (hereafter ln Chl-a_{obs}) to mean lake TOA reflectance in each band 1-4 as well as six band ratios/band algorithms from 53 samples. Band ratios/band algorithms were chosen on the basis of a review of studies conducted for inland waters (reviewed by Ho *et al.*, 2017; also Brivio *et al.*, 2001; Brezonik *et al.*, 2005; Keith *et al.*, 2018). Pearson correlation coefficients (r) were calculated to quantify the strength of linear relationships between Chl-a_{obs} and ln Chl-a_{obs} and lake reflectance values. The highest r coefficients (representing the strongest correlation) were analyzed. The same procedure was conducted on DOC and natural log transformed DOC (i.e., ln DOC) for the samples that had DOC data (i.e., of 53 samples with Chl-a_{obs}, 31 samples had DOC data).

For model development, outliers identified as data points where Cook's distance of ln Chl-a_{obs} versus mean lake reflectance were greater than 3/n in three iterations were removed. The reaming dataset was divided into two independent subsets, one for model development (80%) and the other for model validation (20%); ln Chl-a_{obs} in each subset was compared to produce approximately even ranges (Tables 2.1, 2.2). A linear regression model was developed with ln Chl-a_{obs} as the dependent variable and reflectance of the selected band algorithm as the independent variable. The resulting equation was applied to mean lake reflectance in the validation dataset; linear regression of ln Chl-a_{obs} versus modeled ln Chl-a (ln Chl-a_{mod}) was applied to test whether predicted values fit to observed values in a 1:1 line. Reflectance was averaged for each year in each lake polygon (resulting in time series of annual reflectance values). The model equation was then applied to 20,930 lake polygons to generate annual ln Chl-a_{mod} (hereafter ln Chl-a_{mod}). To generate a continuous 28-year (1984-2011) ln Chl-a_{mod} time series for each lake, missing ln Chl-a_{mod} values in lakes were interpolated using the space-time kriging method (see Appendix B for details). This resulted in the continuous time series of ln Chl-a_{mod} for 12,644 lakes, which was used for further analysis.

Table 2.1 Description of water chemistry and morphometry of Ontario lakes selected for the regression model. (Note: observations identified as outliers are not shown in this table).

Lake ID	Sample Name/ID	Longitude	Latitude	Lake maximum depth (m)	Lake area (ha)	Sample date	Trophic state	Chl-a (µg L^{-1})	DOC (mg L^{-1})	TP (µg L^{-1})	Secchi depth (m)	(B1−B3)⁄B2 reflectance	Used in the regression (R) or validation (V)
25on	Caysee2	-84.66	47.18	1.3	16.5	16-Jun-10	Oligotorphic	2.5	8.3	23.2	1.3	0.3655	R
7on	Little Turkey	-84.41	47.04	7.3	18.9	16-May-10	Oligotorphic	0.5	16.4	3.2	7.3	0.6262	R
17on	Appleby1	-83.35	46.43	5.1	24.3	26-Jun-09	Mesotrophic	4.6	-	10.8	2.3	0.4080	R
1on	Negick2	-84.49	47.21	5.3	26.6	16-Jun-10	Oligotorphic	2.3	2.6	12.8	3.5	0.3733	R
23on	Twin	-83.93	46.23	3.8	30.0	27-Jul-11	Eutrophic	7.4	-	10.3	1.7	0.4197	R
10on	Sill	-84.25	46.77	7.4	41.7	20-Jun-10	Oligotorphic	0.9	-	10.8	7.2	0.5579	R
19on	Woodrow2	-83.33	46.41	2.0	48.8	24-Aug-10	Oligotorphic	0.6	7.1	3.8	2.0	0.6616	R
5on	Big Turkey	-84.42	47.05	42.7	51.8	16-May-10	Oligotorphic	1.4	3.8	5.0	5.6	0.4563	R
22on	Eaket1	-83.25	46.35	4.5	56.7	26-Jun-09	Mesotrophic	2.8	-	9.2	2.9	0.3627	R
12on	Reception1	-83.25	46.48	2.8	88.7	26-Jun-09	Eutrophic	10.0	-	14.7	1.3	0.2840	R
26on	Carp	-84.56	46.97	1.5	112.1	16-Jun-10	Mesotrophic	3.7	5.6	17.2	1..5	0.2966	R
16on	Constance1	-83.23	46.43	7.8	120.1	26-Jun-09	Oligotorphic	1.1	-	6.9	4.7	0.5154	R
20on	Round	-83.83	46.39	3.2	128.4	24-Jun-09	Mesotrophic	3.3	-	19.5	2.7	0.4195	R
2on	Upper Griffin	-84.40	47.09	7.8	155.3	16-Jun-10	Oligotorphic	0.7	3.8	7.2	7.7	0.5779	V
8on	Upper Tilley2	-84.39	47.02	6.1	163.1	15-May-10	Oligotorphic	2.1	4.8	9.2	2.9	0.4804	V
24on	Dean2	-83.18	46.23	14.9	219.5	25-Jul-11	Mesotrophic	3.2	-	-	6.0	0.4652	R
14on	Cloudy	-83.93	46.44	7.4	248.8	24-Jun-09	Oligotorphic	0.5	-	9.4	4.8	0.5497	R
13on	Rock	-83.77	46.43	1.9	1033.2	24-Jun-09	Mesotrophic	3.7	-	13.4	1.8	0.4150	R

Table 2.2 Description of water chemistry and morphometry of Alberta lakes selected for the regression model. (Note: observations identified as outliers are not shown in this table).

Lake ID	Sample Name/ID	Longitude	Latitude	Lake mean depth (m)	Lake area (ha)	Sample date	Trophic state	Chl-a (µg L^{-1})	DOC (mg L^{-1})	TP (µg L^{-1})	Secchi depth (m)	(B1−B3)/B2 reflectance	Used in the regression (R) or validation (V)
68ab	61	-113.91	55.92	2	20.4	12-Aug-01	Oligotorphic	2.0	22.1	68.0	1.3	0.5069	R
98ab	16	-115.55	56.11	0.9	36.7	13-Aug-02	Eutrophic	12.0	22.5	68.5	0.9	0.2101	R
53ab	201	-115.71	56.12	1.2	35.1	13-Aug-01	Eutrophic	30.3	23.2	46.3	0.6	0.2950	R
78ab	27	-115.52	56.07	0.6	4.5	11-Aug-01	Eutrophic	12.4	25.9	25.9	48.4	0.4037	R
53ab	2012	-115.71	56.12	1.2	34.6	13-Aug-02	Eutrophic	13.0	27.1	58.5	0.8	0.3166	R
24ab	12	-115.88	56.10	1.3	4.6	11-Aug-01	Eutrophic	15.9	27.9	58.2	1.3	0.4123	R
16ab	101	-114.75	56.31	1.8	39.2	13-Aug-01	Oligotorphic	2.0	38.6	17.9	1.8	0.5722	R
2ab	42	-115.16	56.30	1.1	7.4	11-Aug-01	Eutrophic	40.8	40.9	117.0	0.6	0.3035	R
58ab	111	-115.43	56.03	0.6	5.0	14-Aug-01	Mesotrophic	2.8	48.8	39.2	0.6	0.4722	R
70ab	1211	-115.35	56.01	0.7	6.8	15-Aug-01	Mesotrophic	3.5	50.3	58.8	0.7	0.5302	V
5ab	7	-115.63	56.29	0.8	15.6	11-Aug-01	Mesotrophic	4.4	56.7	43.5	0.8	0.2975	R
37ab	1681	-115.20	55.99	0.7	11.2	15-Aug-01	Mesotrophic	6.4	58.3	102.4	0.7	0.3609	R
70ab	1212	-115.35	56.01	0.7	6.1	13-Aug-02	Eutrophic	12.1	58.5	105.9	0.5	0.2942	R
7ab	4	-115.68	56.42	0.6	6.4	11-Aug-01	Mesotrophic	2.8	59.9	233.9	0.6	0.4042	R
108ab	75	-114.85	55.96	0.9	31.4	12-Aug-01	Eutrophic	34.2	60.2	118.6	0.5	0.1629	R
101ab	5992	-115.38	56.07	1.6	19.7	14-Aug-02	Eutrophic	28.6	60.3	100.8	0.3	0.2212	R
80ab	55	-114.16	56.32	1.1	7.4	12-Aug-01	Eutrophic	61.5	60.7	246.4	0.4	0.0700	R
92ab	1223	-115.35	56.01	0.7	5.9	12-Aug-02	Eutrophic	31.4	68.9	123.0	0.3	0.2829	R
37ab	1682	-115.20	55.99	0.7	10.6	12-Aug-02	Mesotrophic	3.8	74.9	120.8	0.7	0.2878	V
75ab	87	-115.12	55.73	0.5	9.1	12-Aug-01	Eutrophic	7.4	79.6	57.2	0.5	0.4055	V
70ab	121	-115.35	56.01	0.7	6.4	15-Aug-99	Eutrophic	46.0	-	150.8	0.7	0.1435	V
92ab	122	-115.35	56.01	0.7	6.9	15-Aug-99	Hypereutrophic	58.0	-	77.7	0.6	0.1377	R
38ab	171	-115.19	55.98	0.6	8.5	15-Aug-99	Eutrophic	47.1	-	421.7	0.6	0.2145	R
46ab	165	-115.26	55.96	-	8.5	19-Aug-99	Hypereutrophic	63.4	-	178.6	0.5	0.2163	R
28ab	57	-115.39	56.08	0.6	9.8	15-Aug-99	Eutrophic	8.7	-	119.3	0.0	0.3494	V
55ab	81	-115.56	56.03	-	19.9	15-Aug-99	Eutrophic	9.2	-	54.4	0.5	0.3140	R
45ab	131	-115.60	55.96	-	27.1	15-Aug-99	Eutrophic	18.5	-	135.8	0.3	0.2519	V
53ab	2011	-115.71	56.12	1.2	34.4	15-Aug-99	Eutrophic	20.2	-	64.9	0.5	0.3693	R
67ab	127	-115.18	56.01	-	201.8	19-Aug-99	Hypereutrophic	57.2	-	212.4	0.8	0.0790	V
102ab	88	-115.50	56.04	1.1	274.7	15-Aug-99	Mesotrophic	3.7	-	30.2	0.7	0.4832	R
56ab	89	-115.51	56.02	-	311.9	15-Aug-99	Mesotrophic	3.5	-	66.7	0.4	0.3927	V

2.3.6 Decomposition of variation in Chlorophyll-*a*

In the framework used in this study (Wiley *et al.*, 1997), a two-way ANOVA decomposes variation into three components: space, time and space×time interaction, which is statistically expressed as the sum of squares in space (SS_{space}), time (SS_{time}), and space×time ($SS_{space \times time}$) (Wiley *et al.*, 1997). The space component reflects broad-scale landscape characteristics such as topography or soil types, while the time component reflects changes over time such as climate change or land cover development. The space×time interaction component reflects lake-specific biological and morphological attributes that may influence Chl-a concentration (see Sass *et al.*, 2007; Figure 2.2). Additionally, space×time interaction component reflects an error term (Wiley *et al.*, 1997), which may be a result of the absence of replicates in ground sampled lakes or errors and uncertainties in reflectance values (Sass *et al.*, 2007). Because it is impossible to separate lake-specific factors from the error term within the space×time interaction component, I acknowledged that the proportion constituting the error term may be considerable and caution should be exercised while analyzing the space×time interaction component.

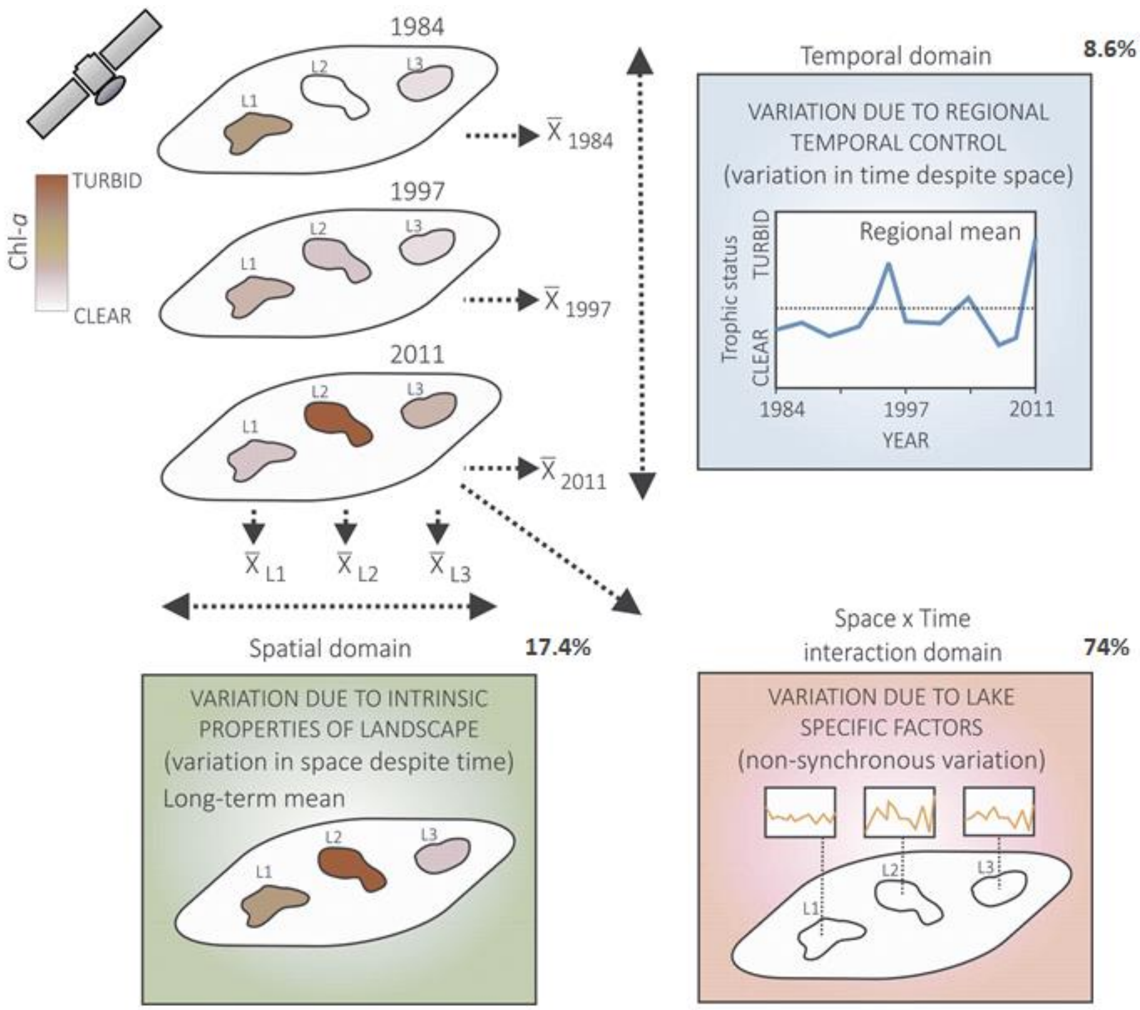

Figure 2.2 Schematic model of variation in lake ln Chl-a_{mod} (modified from Sass *et al.*, 2007). In bold: sources of variation in ln Chl-a_{mod} (in %) estimated by the two-way ANOVA in this study.

ANOVAs were calculated on matrices of Chl-a_{mod}, in which years were represented by columns and lakes were represented by rows (Paltsev, 2015). For a given lake: the spatial component was the difference between the 28-year average Chl-a_{mod} and the 28-year average of all lakes; the temporal component was the difference between the average Chl-a_{mod} of all lakes for a specific year and the 28-year average of all lakes; and the space×time interaction component was the difference between the total variation and the sum of variation in the spatial and temporal components (i.e., $SS_{space\times time} = SS_{total} - SS_{space} - SS_{time}$) (Sass *et al.*, 2007).

2.3.7 Analysis of spatial and temporal patterns in Chlorophyll-*a*

For an analysis of spatial patterns, ln Chl-a_{mod} was back-transformed using an exponential function and median Chl-a_{mod} values were calculated for each lake. Then, the lakes were classified to trophic states following Carlson & Simpson (1996): oligotrophic (< 2.6 μg L^{-1}); mesotrophic (2.6-7.3 μg L^{-1}); eutrophic (7.3-56.0 μg L^{-1}); and hyper-eutrophic (> 56.0 μg L^{-1}). To identify trends in ln Chl-a_{mod}, a non-parametric Mann-Kendall test (Kendall, 1975) was conducted on individual lakes using MATLAB (R2013b, the WathWorks Inc). Trends were considered significant at $p < 0.05$.

2.4 Results

2.4.1 Chlorophyll-*a* modeling

Chl-a_{obs} versus reflectance in visual bands (bands 1–3) showed an increase in correlation with increasing wavelength; ln-transformed Chl-a_{obs} (i.e., ln Chl-a_{obs}) versus reflectance yielded better results with the strongest correlation between ln Chl-a_{obs} and band 3 ($r = 0.45$; Table 2.3).

Correlation between Chl-a_{obs} or ln Chl-a_{obs} and reflectance from various band ratios/algorithms performed much better (i.e., higher r) than for single bands. Correlation between ln Chl-a_{obs} and the ratio of band 3 to band 1 (B3/B1) showed the strongest relationship ($r = 0.88$; $p < 0.0001$); correlation with three-band algorithm of (B1-B3)/B2 was almost as strong ($r = -0.85$; $p < 0.0001$).

The results of Pearson correlation showed that, of all bands, band 3 showed the strongest correlation with both ln Chl-a_{obs} and ln DOC (Table 2.3). Covariance between Chl-a_{obs} and DOC was also relatively high ($r = 0.34$). Therefore, it was important to select a band ratio or algorithm where the effect of ln DOC was minimal, while correlation with ln Chl-a_{obs} was still high. The (B1-B3)/B2 algorithm produced poor correlation with ln DOC ($r = -0.29$); in contrast, B3/B1 yielded stronger and significant correlation ($r = 0.47$; $p < 0.05$). The (B1-B3)/B2 was also recommended by Matthews (2011) as the most appropriate algorithm for Chl-a retrieval in sensors with broad spectral resolutions (i.e., Landsat TM/ETM+). The (B1-B3)/B2 was chosen for the model development.

From the 53 sample lake dataset, four observations were identified as outliers in the relationship between ln $Chl\text{-}a_{obs}$ and (B1-B3)/B2 reflectance and removed, leaving a dataset of 49 lake samples (see Tables 2.1, 2.2 for Ontario and Alberta lakes). $Chl\text{-}a_{obs}$ in 39 lakes selected for model development ranged from 0.45 to 63.4 $\mu g\ L^{-1}$ with a mean of 14.4 $\mu g\ L^{-1}$ and median of 4.6 $\mu g\ L^{-1}$; $Chl\text{-}a_{obs}$ in 10 lake samples selected for model validation ranged from 0.65 to 57.2 $\mu g\ L^{-1}$ with a mean of 15.1 $\mu g\ L^{-1}$ and median of 5.6 $\mu g\ L^{-1}$.

A linear regression model developed from ln $Chl\text{-}a_{obs}$ and (B1-B3)/B2 reflectance values in the 39-lake model development dataset explained 76% of variation in ln $Chl\text{-}a_{mod}$ ($r^2 = 0.76$, $p < 0.01$; Figure 2.3a). There was a strong and significant correlation between ln $Chl\text{-}a_{obs}$ and ln $Chl\text{-}a_{mod}$ in the model validation dataset ($r^2 = 0.85$, $p < 0.01$; Figure 2.3b). The slope of the best-fit function ln $Chl\text{-}a_{mod}$ *versus* ln $Chl\text{-}a_{obs}$ was 0.867 (not significantly different from 1, $p = 0.71$); the intercept was not significantly different from zero ($p = 0.32$). The root means square error (RMSE) of $Chl\text{-}a_{mod}$ prediction was 0.55.

In the 39-lake model development dataset, 23 lakes had in-situ DOC measurements; hence, I conducted a linear regression between ln DOC and (B1-B3)/B2 reflectance values to see if the former can be predicted from the latter. DOC can seriously interfere with Chl-a reflectance in optically complex inland waters (Brezonik *et al.*, 2005). (B1-B3)/B2 produced a relatively high correlation with ln Chl-a (the second highest after the B3/B1), but the correlation with ln DOC was poor and insignificant ($r^2 = 0.14$, $p = 0.09$; Figure 2.4). The latter indicates that DOC is unlikely to have a substantial effect on (B1-B3)/B2 reflectance in lakes in the study region. Appendix C provides analytical support with extended lake datasets that confirm that modeled Chl-a results in this study are not influenced by DOC. Appendix D provides analyses with extended lake datasets to determine if lake phosphorus (P) can be used in future work as a proxy for lake Chl-a to validate $Chl\text{-}a_{mod}$ over time series.

Table 2.3 Pearson correlation coefficients (r) between optically-related variables (Chl-a_{obs} and DOC) and Landsat TM/ETM+ bands and commonly used band combinations/band algorithms (* $p < 0.05$; ** $p < 0.0001$). ln indicates natural log transformed values.

	DOC	**ln DOC**	**B1**	**B2**	**B3**	**B4**	**B2/B1**	**B2/B3**	**B3/B1**	**B3/B4**	**(B1−B3)/B2**	**(1/B1-1/B2) B4**
Chl-a_{obs}, n = 53	0.34		0.04	0.27*	0.41*	0.04	0.48*	-0.57*	0.79**	0.45*	-0.74**	0.49*
ln Chl-a_{obs}, n = 53		0.59*	0.06	0.31*	0.45*	0.03	0.56**	-0.67**	0.88**	0.53**	-0.85**	0.54**
DOC, n = 31			0.38*	0.40*	0.47*	0.41	0.06	-0.61*	0.48*	0.15	-0.31	0.06
ln DOC, n = 31			0.48*	0.49*	0.55*	0.43*	0.05	-0.63**	0.47*	0.26	-0.32	0.04

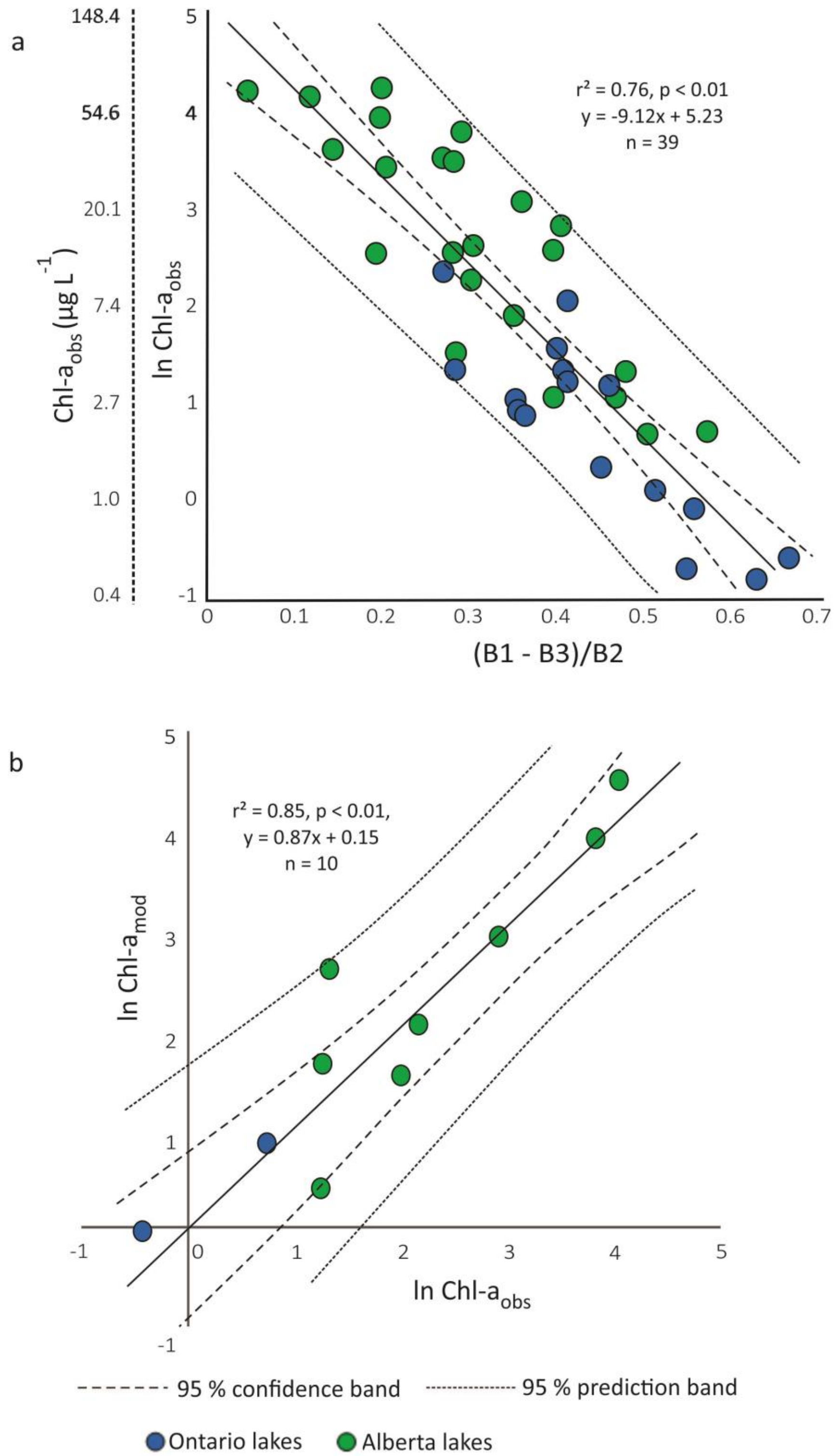

Figure 2.3 (a) Scatterplot of reflectance from (B1-B3)/B2 band algorithm regressed against ln Chl-a_{obs}; (b) comparison of ln Chl-a_{obs} (validation dataset) and ln Chl-a_{mod}. The solid line represents the 1:1 line.

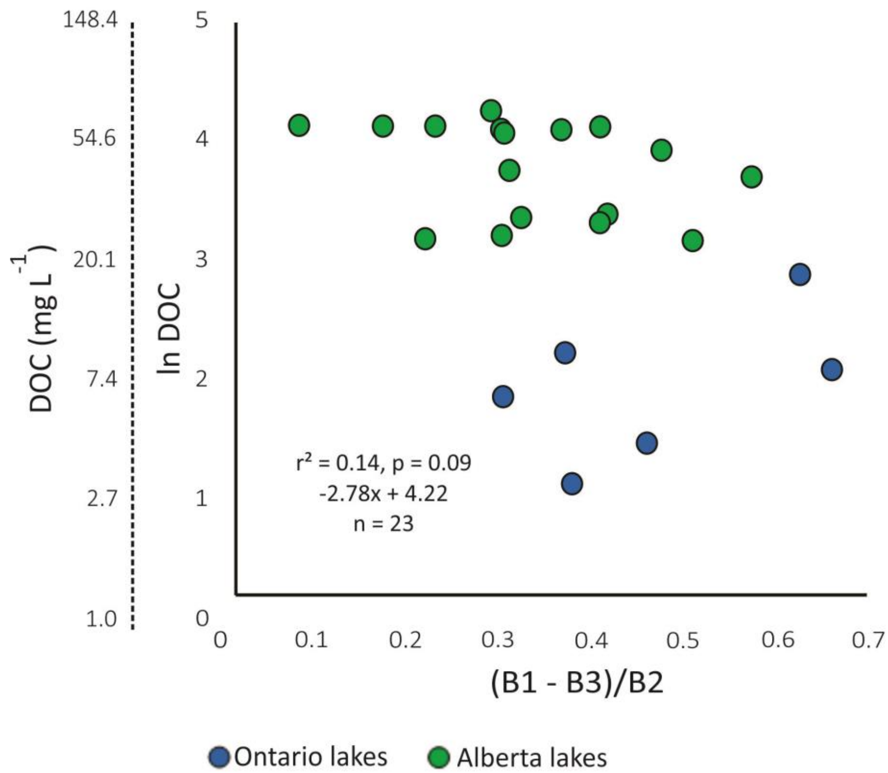

Figure 2.4 Scatterplot of reflectance from (B1-B3)/B2 band algorithm regressed against ln DOC.

2.4.2 Decomposition of Chlorophyll-*a* variation into space, time and space×time interaction components

Decomposition of variation in ln Chl-a_{mod} revealed that the space component explained 17.4%, the time component explained 8.6%, while the space×time interaction component explained 74.0% of the variation (Figure 2.2).

2.4.3 Spatial patterns in modeled Chlorophyll-*a*

Two distinct spatial patterns can be observed in a map of median Chl-a_{mod} (Figure 2.5). First, the highest density of oligotrophic lakes was found along the topographic divides between regional (Great Lakes, St. Lawrence River and Hudson Bay) drainage basins forming an “oligotrophic belt” (Figure 2.5b). Second, most eutrophic and hypereutrophic lakes were found to form clusters. In the north, these lakes tended to be located in the proximity to the Great Lakes, while in the south, they tended to be concentrated near the southernmost tip of the study region (Figure 2.5d).

2.4.4 Temporal patterns in modeled Chlorophyll-*a*

The Mann-Kendall tests conducted on individual lakes found a number of significant positive and negative trends ($p < 0.05$) in ln Chl-a_{mod}: 500 lakes displayed positive trends, and 561 lakes displayed negative trends (Figure 2.6). Positive trending lakes tended to be located along the southern boundary of the study region; there was also a large cluster of these lakes in the relatively populated industrial and mining areas surrounding Sudbury. No positive trending lakes were found within the Hudson Bay Basin. Negative trending lakes seemed to concentrate in the northern portion of the study region; there was also a small cluster of these lakes in the south-eastern tip of the region.

Trends in ln Chl-a_{mod} also showed approximately the same distribution of slopes (i.e., rate of change/year; Figure 2.6b and c). However, there were more negative trending lakes with higher rates of change in ln Chl-a_{mod}, than positive trending lakes. For example, there were 16 negative trending lakes with slopes < -0.15, and 6 positive trending lakes with slopes > 0.15.

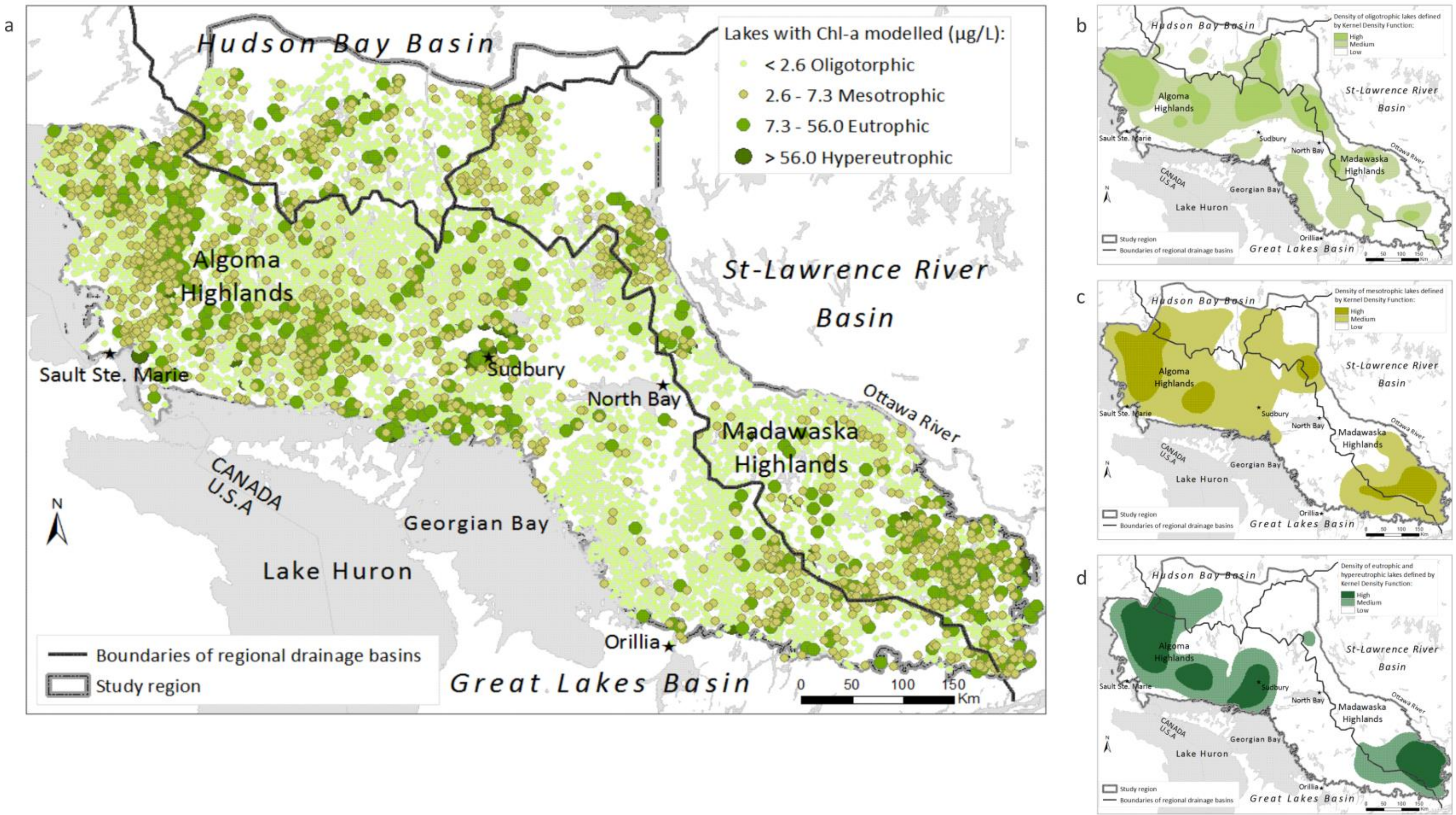

Figure 2.5 (a) Spatial distribution of lakes (based on lake trophic state calculated as median Chl-a$_{mod}$ over the 1984–2011 period), and Kernel density of: (b) oligotrophic, (c) mesotrophic, and (d) eutrophic and hypereutrophic lakes. In Kernel density, the default search radius was based on the number of lakes.

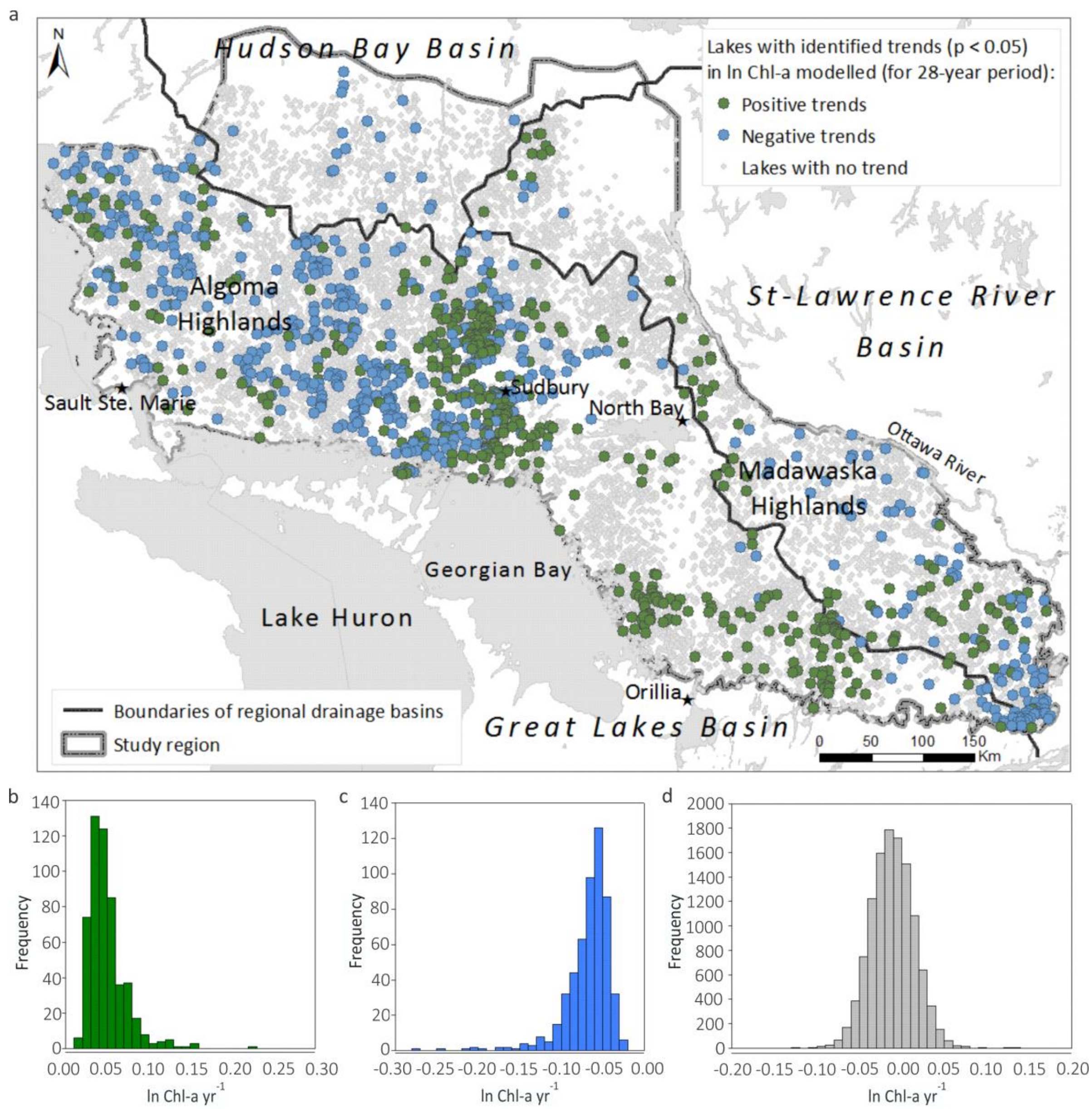

Figure 2.6 (a) Distribution of trends ($p < 0.05$) in ln Chl-a$_{mod}$ over 28 years, and frequency distribution of slopes (i.e., ln Chl-a yr^{-1}) in ln Chl-a$_{mod}$ for lakes with: (b) positive ln Chl-a yr^{-1}, (c) negative ln Chl-a yr^{-1}, and (d) no significant ($p > 0.05$) ln Chl-a yr^{-1}.

2.5 Discussion

Remote sensing of Chl-a

In this study, I used a three-band reflectance model based on reflectance values from Landsat TM and ETM+ sensors to model Chl-a concentration in thousands of temperate lakes. Several studies show that, to date, three-band algorithms have produced some of the most promising results for accurate modeling of Chl-a concentration using Landsat imagery in inland waters (see Brivio *et al.*, 2001; Allan *et al.*, 2011; Stumpf *et al.*, 2016; Keith *et al.*, 2018).

(B1-B3)/B2 reflectance related the peaks of Chl-a absorptions in both blue (at around 441 nm) and red bands to the peak of reflectance in the green band. (B1-B3)/B2 has been also used to reduce the effects of TSS and DOC on the reflectance of Chl-a (Mayo *et al.*, 1995, Brivio *et al.*, 2001). The effect of TSS might be especially pronounced in lakes with low Chl-a concentration (i.e., oligotrophic), which often show an increase in reflectance with increasing wavelengths caused by backscattering of TSS (Odermatt *et al.*, 2012). The subtraction of reflectance in band 3 from the reflectance in band 1 is assumed to correct for this additional "TSS-produced" reflectance (Brivio *et al.*, 2001).

Poor correlation between Chl-a and band 1 reflectance was likely due to a reflectance minimum of Chl-a near 440 nm (near the edge of band 1; Gitelson *et al.*, 2000) or the presence of carotenoids that have a reflectance minimum at 490 nm and could mask increasing Chl-a capacity towards band 2 (i.e., green wavelength; Yacobi *et al.*, 1995). Chl-a has a green reflectance peak at ~550 nm (near the center of band 2), hence correlation is stronger in this band.

Although Chl-a has a second reflectance minimum at 670 nm, I found a degree of correlation between Chl-a and band 3 reflectance. The correlation in the red part of spectrum was described by other studies (e.g., Sass *et al.*, 2007; Allan *et al.*, 2015), and might be attributed to the fact that light scattering by cell walls offsets this Chl-a absorbance, especially in situations with high algal density (Gitelson *et al.*, 2000). Poor correlation in band 4 might be explained by the assumption that reflectance in the NIR channel is close to zero due to strong absorption by water (Yacobi *et al.*, 1995).

Patterns in modeled Chl-a

The contribution of temporal (climatic drivers) and spatial (landscape drivers) factors to phytoplankton biomass has not been clearly understood. This is partly because these factors operate at different spatial and temporal scales which can be difficult to evaluate and incorporate into a single model. In this study, I identified spatial and temporal patterns of ln Chl-a_{mod} (as a proxy for phytoplankton biomass) in 12,644 lakes that covered a large (> 150,000 km^2) region of the temperate forest biome in North America over a 28-year period (1984-2011).

I did not intend to quantify specific drivers of ln Chl-a_{mod}. Rather, I intended to identify the relative contribution of temporal and spatial factors to variation in ln Chl-a_{mod}. This contribution was determined using a two-way ANOVA.

The effect of climate on phytoplankton has been widely described (e.g., Smol & Cumming, 2000; Paul, 2008; Whitehead *et al.*, 2009; Posch *et al.*, 2012; Rigosi *et al.*, 2014; Sinha *et al.*, 2017). Significant trends found in the 28-year time series of ln Chl-a_{mod} might be driven by increases in air temperature (O'Reilly *et al.*, 2015) or changes (increases or decreases) in precipitation patterns over the same period (Yeung *et al.*, 2018). However, these trends were found only in 8.4% of all lakes (Figure 2.6). Similar results have been described by other authors. For example, in a recent study conducted on 2,913 lakes located in the Northeastern United States, Oliver *et al.* (2017) found that only a minority (~22%) of lakes had significant trends in Chl-a for the period from 1990 to 2013.

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Paltsev (2015) used single Landsat band (band 3) to model Chl-a in lakes of the temperate forest biome, and did not find any significant trends in time series of Chl-a. This might be due to the fact the author: (1) used much smaller lake dataset (the number of lakes with continuous Chl-a concentration was 6,384 *versus* 12,644 lakes used in this study), which might not have included lakes that had significant trends, and (2) analyzed average Chl-a (over all 6,384 lakes), while in this study I performed time series analysis (the Mann-Kendall trend test) on individual lakes; hence trends in Chl-a of each lake were captured (as opposed to averaged Chl-a values of all lakes). The latter indicates the importance of analyzing individual times series as opposed to time series of average or medium values (i.e., data that were aggregated over entire datasets).

The direct correlation between climate and phytoplankton may be weak as both landscape and lake properties filter or modify climate signals differently (Magnuson *et al.*, 1990). The climate signals are filtered (and hence modified) through spatially heterogeneous landscape control elements (i.e., space component) and inherent lake-specific features (i.e., space×time interaction component) influencing nutrient supply to lakes (Anderson, 2014). I found that both positive and negative trending lakes were often in close proximity to each other. The clustering pattern of lakes with either positive or negative trends (i.e., coherent behavior of these lakes) suggests that landscape and lake properties make some lakes more sensitive to changes in temperature while other lakes are more sensitive to changes in precipitation.

There were fewer positive trending lakes than negative trending lakes. One might expect that increasing temperatures in the region (Bush *et al.*, 2014, Yeung *et al.*, 2018) would lead to higher phytoplankton biomass (O'Neil *et al.*, 2012), higher frequency of algal blooms reports (Winter *et al.*, 2011), both indicative of widespread eutrophication. However, I found that there were fewer positive trending lakes than negative trending lakes (Figure 2.6). This contradicts findings of Oliver *et al.* (2017) in their study region of the Northeastern United States where positive trending lakes dominated.

It is also important to make certain that changes in ln Chl-a_{mod} are ecologically meaningful. For example, the annual rate of change in Chl-a_{mod} of 1 μg L^{-1} (i.e., ln Chl-a yr^{-1} is ~ 0.04) would show a 10 μg L^{-1} change in Chl-a_{mod} per decade. This is large enough to move lakes from an oligotrophic to a eutrophic state or *vice versa* (Carlson & Simpson, 1996). My results show that 237 (out of 500) positive trending lakes and 313 (out of 561) negative trending lakes had annual rate of change > 1 μg L^{-1}. These lakes indeed experienced ecologically meaningful changes that might be referred to as eutrophication (for lakes with positive trends) and oligotrophication (for lakes with negative trends).

Although landscape controls such as forest cover and soil type (Sand-Jensen & Staehr, 2007; Klimaszyk & Rzymski, 2011) have been referred as important contributors to variations in phytoplankton biomass, the contribution of the space component in explaining variation in Chl-a was relatively small (Figure 2.2). However, some "broad-scale" spatial patterns in Chl-a_{mod} can still be observed. For example, spatial factors most likely contribute to "lake effect" observed in this study (i.e., lakes of eutrophic and hyper-eutrophic states tend to be located near the Great Lakes) (Figure 2.5d).

The interaction of lakes with the catchments is implicit and is carried out via nutrient loading from catchments to lakes. This loading, in turn, depends on spatially variable controls (i.e., the space component) such as elevation, presence of wetlands, soils and forest type (Blenckner *et al.*, 2007; Staehr *et al.*, 2012). Therefore, the lakes located on the lower reaches of the local rivers near the Great Lakes may be affected not so much by the neighboring Great Lakes but rather by water flows bringing increasing organic matter and nutrients from upstream areas. Lakes located on the lower reaches of the local rivers or lowlands are likely to receive more nutrients than those located on uplands. Nõges (2009) analyzed chemistry and morphometry of 1,337 European lakes and found that concentration of organic matter, N, P, and Chl-a was much higher in lowland lakes than in upland lakes. Catchments with lower elevation (lowland catchments) generally have more gentle slopes than upland catchments. The proportion of wetlands and riparian zones is also greater in catchments with lower relief. This influences the residence time that water and associated nutrients spend in catchments before being flushed to receiving lakes (McGuire *et al.*, 2005); residence times are usually longer in catchments with gentle slopes and higher proportion of wetlands (Harms *et al.*, 2016). Longer water residence times and as result slower flows increase the opportunity of higher flux of dissolved organic carbon, dissolved organic nitrogen and P to receiving waters (Creed *et al.*, 2008; Mengistu *et al.*, 2014).

In support of the importance of elevation in regulating ln Chl-a_{mod} in study lakes, there is another important pattern found in the study region. My results show that there is a high concentration of oligotrophic lakes (i.e., lakes with lower ln Chl-a_{mod}) in the central portions of the study region (Figure 2.5b) that somewhat corresponds to the natural topographical and hydrological boundaries between main drainage basins–the region where nutrients are less likely to accumulate but are instead washed out to downstream areas via surface and groundwater flows.

Although temperate forest biome is a relatively intact region, there are some areas of greater anthropogenic development. Greater intensity of anthropogenic activities (including mining and forest management activities; Carleton, 2000) in areas along the Lake Huron and near Sudbury may also contribute to elevated Chl-a_{mod} in lowland lakes. Logging practices increase erosion and lead to changes in nutrient composition in local soils and wetlands (Kreutzweiser *et al.*, 2008) and subsequent increased nutrient accumulations in downstream lakes and increased phytoplankton biomass (Devito *et al.*, 2005). This may explain the fact that many reports of HABs are described for lowland areas along the Great Lakes and near the Sudbury eutrophic lake cluster (Figure 2.5; Winter *et al.*, 2011).

The space×time interaction component accounted for variation in ln Chl-a_{mod} that represented interactive effects of spatial and temporal factors on phytoplankton (Wiley *et al.*, 1997) rather than exclusively spatial or temporal patterns (Sass *et al.*, 2007). I found that the space×time interaction component explained the majority of variation in ln Chl-a_{mod}. This suggests that phytoplankton biomass in these lakes were influenced by lake-specific physical, chemical and biological factors. The majority of variation explained by the space×time interaction was also found by Wiley *et al.* (1997) for two insect species (77% and 44% accordingly) and Sass *et al.* (2007) for lake trophic states (measured as Chl-a concentration; 43%) in the boreal lakes in Alberta.

These lake-specific factors may be classed into two groups: top-down factors, such as zooplankton grazing on phytoplankton (Baines *et al.*, 2000); and bottom-up factors such as lake-specific physical properties (e.g., depth, volume, the extent of littoral zone) that influence water mixing, dilution of nutrients in water column and sedimentation, which in turn control nutrient concentrations that affect primary production (Hakanson, 2005; Nõges, 2009; Orihel *et al.*, 2017).

Future perspectives on the improvement of the Chl-a retrieval algorithm using remote sensing

In this study, my intention was to make use of long-term time series provided by Landsat TM and ETM+ and simple but robust linear regression model to estimate Chl-a concentration in a large region within the same climatic and eco-zone. The launch of new type of satellites with better spectral and spatial resolutions continue to enhance Chl-a retrieval models in optically-complex inland waters. For example, Sentinel-2 satellite launched in 2015 and operated by EU Copernicus Programme has the Multi-Spectral Instrument sensor with 13 spectral bands from 443 to 2190 nm with spatial resolution of 10, 20 and 60 m. This fine spectral resolution also allows for identification of cyanobacteria by distinguishing the accessory pigment phycocyanin. For instance, MERIS satellite imagery has already been successfully used for quantification of cyanobacteria blooms in relatively large lakes (Simis *et al.*, 2005; Lunetta, *et al.*, 2015; Tomlinson *et al.*, 2016).

Furthermore, significant progress has been made on the improvement of semi-analytical methods (including band ratio algorithms) that provide new solutions to the problem of co-varying effect between Chl-a and CDOM. Keith *et al.* (2018) used three-band algorithm comprised of reflectances from bands 1, 3 and 5 (near-infrared band: NIR) of the Landsat-8 Operational Land Imager (OLI) sensor to accurately estimate Chl-a concentration in several CDOM-dominated productive lakes in Rhode Island. More sophisticated artificial neural network techniques have also showed promising results in Chl-a estimation in lakes, and particularly in separation of Chl-a reflectances from reflectances of other water constituents (e.g. for application of neural network for Landsat see Sudheer *et al.*, 2006). Although these state of art methods still require more *in-situ* data for validation of results (especially for temporally and spatially dynamic environments such as the temperate forest biome), they nevertheless may be used for the improvement of the model developed in this study in future.

2.6 Conclusions

Unprecedented increases in the eutrophication of lakes in North America established a need to understand the spatial and temporal factors that may be contributing to this phenomenon. Developing such understanding requires large datasets of spatially and temporally extensive information on lake Chl-a concentration generated through remote sensing products and modeling. To the best of my knowledge, this is the first time when Chl-a concentration was modelled in 12,644 lakes located in a large (> 150,000 km^2) portion of the northern temperate zone and for an extended period of time (almost three decades) by using satellite imagery (archived Landsat TM/ETM+ products).

The observed spatial and temporal patterns in modelled lake Chl-a and associated lake trophic states were analyzed for the whole region. By analysing different factors that might be contributing to variation in Chl-a, I found that space×time interaction (i.e., lakes-specific) factors were the most important (74 % of total variation in Chl-a$_{mod}$). My results also provide evidence that, although not as common as expected, there were lakes which Chl-a$_{mod}$ concentration was changing over a 28-year period, and sometimes these lakes altered their trophic states (i.e., the change in Chl-a$_{mod}$ was ecologically meaningful). I observed that there was no "unidirectional" trend in the change in lake trophic state; some lakes were becoming more eutrophic, whereas other lakes were shifting to be more oligotrophic. In contrast to conventional wisdom, more lakes were experiencing oligotrophication than eutrophication. Future work will focus on quantifying drivers of ln Chl-a$_{mod}$ including climate, contributing catchment properties (e.g., presence of wetlands) and lake-specific properties such as lake volume and depth.

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3 Changes in phytoplankton biomass as seen through the prism of lake morphometry and catchments characteristics

3.1 Introduction

Temperate ecosystems are being affected by global environmental changes (Kirtman *et al.*, 2013). Increasing air temperatures and changing precipitation patterns with associated hydrological intensification are leading to fundamental changes in land-aquatic biogeochemical linkages. These effects can be best studied and, in fact, have been already observed in lake ecosystems, as lakes integrate atmospheric, terrestrial and aquatic processes (Williamson *et al.*, 2009). Signs of changes in phytoplankton composition, eutrophication and algal blooms have been found even in remote temperate lakes located far from any human activities (O'Neil *et al.*, 2012; de Senerpont Domis *et al.*, 2013).

Phytoplankton biomass is a product of a complex interaction between external forces and internal processes (Baines *et al.*, 2000; Blenckner *et al.*, 2005). External forces are broad-scale climate-related variables such as air temperature and precipitation (Hollert *et al.*, 2018). Although the direct effect of temperature on phytoplankton biomass has been a major focus of research in recent years, the results of the studies are contradictory. For example, a temperature increase was found to cause an increase (Jeppesen *et al.*, 2009; Kraemer *et al.*, 2017) or a decrease in phytoplankton biomass (Tadonléké, 2010), or to have no significant effect (Kosten *et al.*, 2012; Rasconi *et al.*, 2017). However, there is consensus that the growth rates of many cyanobacteria generally increase with temperature (O'Neil *et al.*, 2012). Some recent studies suggest that although increasing temperature should be taken into account, changes in precipitation (and runoff) patterns might be more important in influencing phytoplankton biomass in lakes (Klimaszyk & Rzymski, 2011; Sinha *et al.*, 2017). This is because precipitation influences nutrient delivery and composition in lake catchments via runoff. Given that phytoplankton depends on nutrient availability, even a slight change in precipitation patterns might trigger a change in phytoplankton biomass (Scheffer & Van Nes, 2007; O'Neil *et al.*, 2012).

The response of lakes to climatic factors may be expected to be synchronous in a region with similar climate patterns (e.g., Magnuson *et al.*, 1990; Baines *et al.*, 2000). However, catchment- and lake-specific properties modify regional climate signals (Blenckner, 2005; Palmer *et al.*, 2014; Hollert *et al.*, 2018). Previous studies have shown that lakes with more similar catchment properties or lake morphometry have the highest synchrony in the response to climate (e.g., see Blenckner, 2005). Catchment morphological properties (i.e., catchment size, topography, presence of wetlands, forest and soil type) affect the source, storage and transport of water and nutrients (e.g., dissolved organic matter (DOM), nitrogen (N) and phosphorus (P)) to receiving waters (Nõges, 2009; Staehr *et al.*, 2012). For example, several studies have shown strong relationships between catchment topography and amount and composition of nutrient export (Devito *et al.*, 2000; Klimaszyk & Rzymski, 2011; Harms *et al.*, 2016). Wetlands in lake catchments have been found to be sinks of inorganic solutes (e.g., nitrate–NO_3^-) and sources of organic solutes, especially dissolved organic carbon (DOC), dissolved organic nitrogen (DON) and P (Mengistu *et al.*, 2014; Li *et al.*, 2015; Harms *et al.*, 2016), while deciduous forests are generally associated with higher export of N and P than coniferous forest (Klimaszyk & Rzymski, 2011). Properties of receiving lake basins (e.g., lake depth, volume) affect the fate of the nutrients within lakes (Søndergaard *et al.*, 2005; Søndergaard, 2007). Smaller-volume and shallower lakes usually have shorter P retention time in the sediments than larger-volume lakes because P can be easily re-suspended due to wind disturbance or fluctuations of water level (Nõges, 2009). Similarly, lakes with well-developed littoral zones (i.e., wide and with established communities of macrophytes) retain more nutrients and organic matter in the surface waters than lakes with small littoral zones (Vadeboncoeur *et al.*, 2002; Kornijów *et al.*, 2016).

In contrast to the many studies where the properties of the lake were used to explore lake physics and chemistry (Magnuson *et al.*, 1990; Palmer *et al.*, 2014; Orihel *et al.*, 2017; Sharma *et al.*, 2019), there are relatively few studies where the properties of the coupled terrestrial-aquatic system (catchments + receiving lakes) were studied (e.g., Staehr *et al.*, 2012; Read *et al.*, 2015). Even fewer studies used the coupled terrestrial-aquatic system in a large number of lakes to assess phytoplankton biomass (or Chlorophyll-*a*: Chl-a) concentration as a proxy for phytoplankton biomass) (Nõges, 2009; Stomp *et al.*, 2011;

Kosten *et al.*, 2012). One of the explanations is that phytoplankton biomass in lakes is not expected to show synchronous behavior in response to climate change, as the biomass is principally determined by processes that affect the availability of nutrients (especially growth-limiting nutrients such as P and N), the composition of phytoplankton (e.g., presence of N-fixing cyanobacteria taxa), and the presence of grazing zooplankton (Baines *et al.*, 2000). It is difficult to separate the direct response of phytoplankton biomass to climate from “indirect” direct responses (i.e., signals) that have been modified by the landscape properties (Palmer *et al.*, 2014).

The goal of this study was to explore the effect of climate and the properties of the coupled terrestrial-aquatic system on Chl-a in 275 lakes in the temperate forest biome of Canada. The study region was selected on the premise that it had minimal local human activities, so that lakes can be analyzed in terms of the natural response to climate with minimal anthropogenically-caused nutrient discharge. I hypothesized that lake Chl-a is regulated by the combined effects of climatic factors and landscape characteristics, where the latter modify the response of individual lakes to regional climate. My predictions were: (1) Chl-a will increase with increasing air temperature and decreasing precipitation, lake volume and lake depth; (2) lakes with the lowest Chl-a receive relatively more precipitation and have the largest volumes and depths (nutrient dilution); and (3) lakes with the highest Chl-a receive relatively less precipitation and have catchments with a large proportion of wetlands (nutrient sources) and lakes with well-developed littoral zones (nutrient deposition areas that are accessible by phytoplankton in the surface waters). This study builds on Chapter 2, where remote sensing techniques were used to model Chl-a in the lakes of the temperate forest biome over a 28-year period.

3.2 Study region and Sites

The study region is the temperate forest biome located between 44.44°N and 48.38°N in central Ontario, Canada, within the Great Lakes–St. Lawrence forest region (Figure 3.1a). Climate in the region is humid continental. Mean annual air temperature was +5.1°C, ranging between +7.4°C in the south-east and +2.6°C in the north (based on the period from 1984 to 2011). Mean annual precipitation for the same period was 960 mm, ranging from 740 mm in the southern areas of the region to 1180 mm in the north-west

(McKenney *et al.*, 2011). Precipitation is influenced by lake effects from the Great Lakes and local orographic effects in areas of relatively high relief (Baldwin *et al.*, 2000).

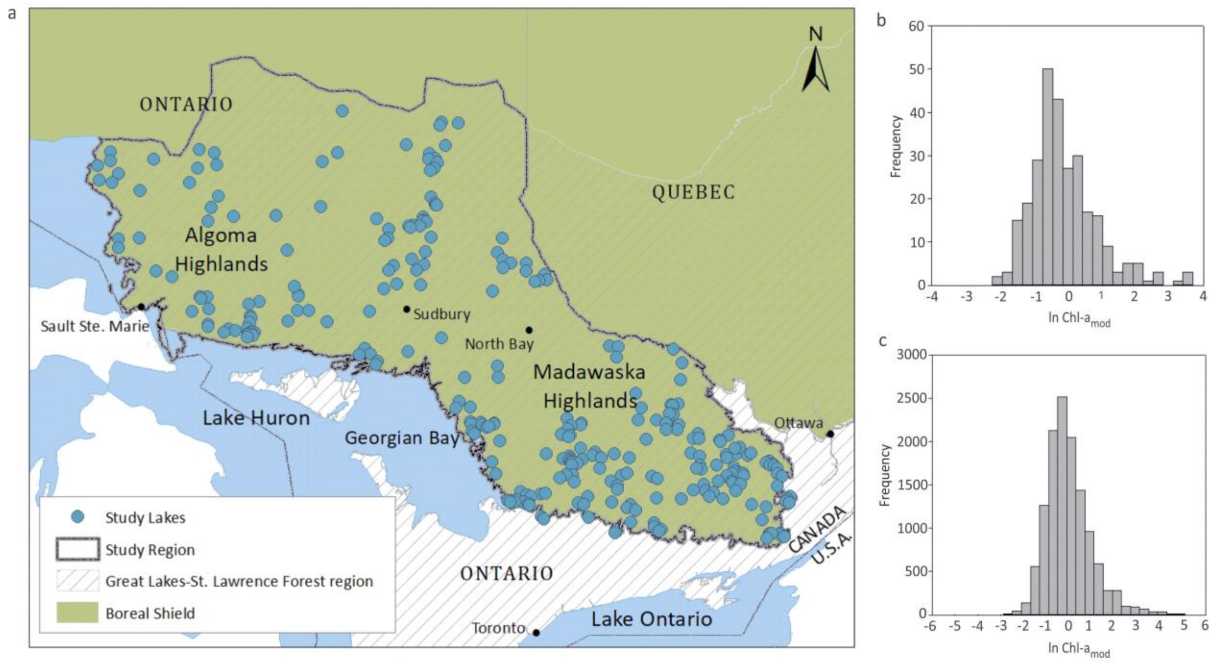

Figure 3.1 (a) Map showing location of the study region (temperate forest biome) and 275 lakes selected for the analysis; and distribution of Chl-a (ln-transformed modelled Chl-a: ln Chl-a$_{mod}$) in (b) 275 lakes (subset of lakes selected for a landscape analysis), and (c) all 12,644 lakes.

The region rests on the Precambrian rocks (primarily comprised of silicate greenstone) of the Boreal (Canadian) Shield (Ontario Geological Survey, 2003). Topography varies from flats and depressions along the shore of the Great Lakes to hills and uplands (Algoma and Madawaska Highlands). Soils are thin and undifferentiated brunisols (in the south), and thick and differentiated orthic ferro-humic podzols (in central areas and in the north). Forests in the region lie in the transitional zone between deciduous and coniferous forests, with the latter being more prevalent in the northern areas (Baldwin *et al.*, 2000).

A subset of lakes (n = 275) were selected from a large (n = 12,644) dataset of temperate lakes with modeled Chl-a (Chl-a$_{mod}$, see Chapter 2). Lakes were selected on the basis of the following factors: (1) lake trophic state (covering the natural range of trophic conditions found in the region, which was based on Chl-a concentration in accordance with Carlson & Simpson, 1996); (2) lake maximum depth and lake area showing approximately the same distribution as in lakes of the dataset used for the model development (Ontario dataset, Appendix E, also see Chapter 2); and (3) the availability of lake bathymetric data. The subset of 275 lakes represented approximately the same distribution of trophic states (in %) (Table 3.1) and Chl-a (Table 3.2, Figure 3.1b) as dataset of 12,644 lakes. Eutrophic and hyper-eutrophic lakes were merged into one group and called "eutrophic" for simplicity. Of 275 lakes, 229 lakes were oligotrophic, 34 lakes were mesotrophic, and 12 lakes were eutrophic (Carlson & Simpson, 1996). The lakes ranged in maximum depth from very shallow (0.6 m) to deep (45.0 m), and ranged in lake area from 5.2 ha to 1394.2 ha (Table 3.2, also Table E.1).

Table 3.1 Number and proportion of lakes according to the trophic state for all lakes (n = 12,644) and a subset of lakes (n = 275) used for climate and landscape control analysis.

		Oligotrophic lakes	**Mesotrophic lakes**	**Eutrophic lakes**	**All lakes**
All lakes	n	10,105	2,000	539	12,644
	%	80	16	4	100
Lakes (subset)	n	229	34	12	275
	%	83	12	4	100

Table 3.2 General descriptive statistics of Chl-a_{mod}, maximum depth and surface area of lakes used in this study (n = 275) and for each trophic state.

	Statistics	**Oligotrophic lakes (n = 229)**	**Mesotrophic lakes (n = 34)**	**Eutrophic lakes (n = 12)**	**All lakes (n = 275)**
Chl-a ($\mu g\ L^{-1}$)	Mean	0.8	3.6	17.1	1.9
	Median	0.6	3.0	12.5	0.7
	Min	0.1	2.2	7.3	0.1
	Max	2.3	7.3	41.3	41.3
	SD	0.5	1.6	12.4	4.7
Lake maximum depth (m)	Mean	11.4	9.7	4.5	10.6
	Median	9.8	7.6	1.5	9.1
	Min	1.3	1.5	0.6	0.6
	Max	45.0	30.5	15.0	45.0
	SD	7.0	7.5	5.0	6.9
Lake surface area (ha)	Mean	109.9	98.1	129.4	99.6
	Median	52.1	36.0	20.5	46.5
	Min	5.2	6.3	7.3	5.2
	Max	1394.2	660.1	1258.8	1394.2
	SD	170.7	162.7	356.8	161.4

3.3 Materials and Methods

3.3.1 Modeled Chlorophyll-*a* time series

Chl-a was modeled for the lakes of the temperate forest biome using remote sensing techniques (see Chapter 2). In brief, 1,067 Landsat 4-5 TM (1984-2011) and 159 Landsat 7 ETM+ (1999-2003) 30-meter images for the period of August to October were acquired over a 152,231 km^2 area (Figure 3.1) from United States Geological Survey archives. Lake locations and boundaries in each image were determined by reclassifying pixels below the local minimum in the bimodal distribution of band 5 images (shortwave infrared) as surface water. To avoid the problem of mixed reflectance due to adjacency near or along lake shorelines (areas with very shallow water and abundant aquatic vegetation), lakes with area less than 4.5 ha (i.e., 30-meter pixels) and high standard deviation of reflectance in band 5 were discarded. The remaining lakes were buffered inside to a distance of 15 meters (1/2 pixel distance). A partial atmospheric correction was conducted, which included subtraction of the Rayleigh scattering radiance from top of atmosphere (TOA) radiance. TOA radiance values were then converted to TOA unitless reflectance. An algorithm based on TOA reflectance values of three Landsat bands [(Band 1-Band 3)/Band 2)] was developed, and derived reflectance values were used in a regression model. The regression model was performed with mean lake reflectance values as the predictor variable and natural log transformed Chl-a (ln Chl-a) observations in 39 sampled lakes as the response variable. The regression model equation was then applied to all mean lake reflectance values in the Landsat archive. A time series of modeled ln Chl-a_{mod} of 12,644 lakes was generated. Universal space-time kriging was used to interpolate missing ln Chl-a_{mod} data found in some lakes as a result of clouds obscuring lake pixels. Median values of Chl-a_{mod} for the 28-year period were calculated for each lake. In this study, a subset of 275 lakes with available lake morphometry was used.

3.3.2 Extraction of temperature and precipitation values

Summer peak Chl-a occurs between August and October in the study region. Annual mean July-October maximum air temperature (T_{max}) and annual mean July-October total precipitation (Pr) data for 1984-2011 were extracted from 300 arc-second resolution monthly grid climate data (McKenney *et al.*, 2011) for each of 275 lakes. Median values were then calculated for the 28-year period. The period of July–October was chosen over August–October (that would correspond to the peak Chl-a period) because of a delay in phytoplankton response to climate factors (e.g., temperature and precipitation); one month is considered a reasonable period of time for phytoplankton to respond to changes in climate conditions during the summer period (Wetzel, 2001).

3.3.3 Landscape data acquisition and processing

Lake bathymetric data for 60 lakes were obtained from Ontario Ministry of Natural Resources and Forestry (OMNR) (available upon request at: www.ontario.ca/data/ministry-natural-resources-and-forestry-topographic-map) in a vector polyline format (i.e., the lakes were digitized). Lake bathymetric data for the remaining 215 lakes were obtained as analog maps from the Western University Map and Date Centre (www.lib.uwo.ca/madgic) and these bathymetric data were georeferenced and converted to a digitized polyline format. Lake polylines were interpolated to raster bathymetry grids at 20 m resolution for each lake.

Catchment topographic data were derived from the 20 m resolution Ontario Digital Elevation Model (DEM) (version 2.0.0; obtained from Scholars Geoportal at http://geo2.scholarsportal.info/#r/details/_uri@= 658779033). DEM hydrological conditioning was performed using the “Fill Depressions” and “Flow Direction” tools in ArcGIS (version 10.2). The raster bathymetry grids of the lakes were added to the DEMs and catchment boundaries for each lake were delineated from the DEMs.

Catchment wetland data were derived from the Ontario Ministry of Natural Resources Ontario Wetland Inventory Database (revised 2015 version; www.ontario.ca/data/wetlands) and added to the DEMs.

3.3.4 Landscape controls of Chlorophyll-*a*

Eighteen landscape variables (Table 3.3) were assumed to act as metrics of landscape controls of lake Chl-a_{mod} on the basis of their potential contribution to Chl-a_{mod}

concentration and lake trophic states (Nõges, 2009; Staehr *et al.*, 2012). Figure 3.2 shows the conceptual model of hypothesized climatic (direct and indirect) and landscape controls of Chl-a_{mod} of temperate lakes.

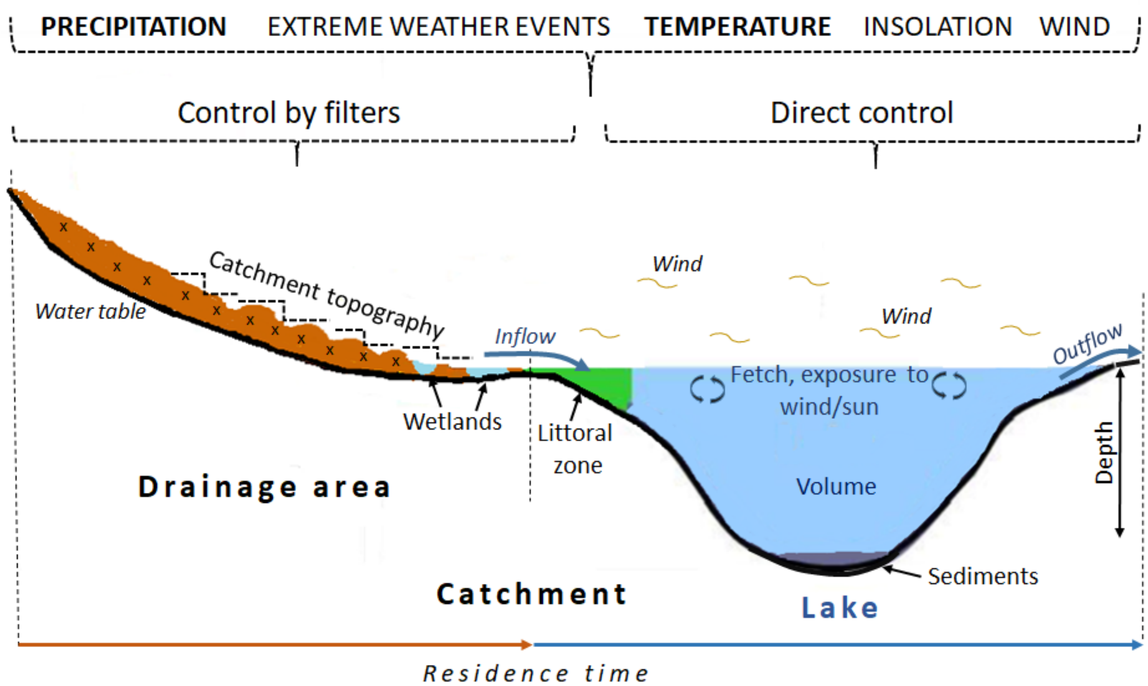

Figure 3.2 Conceptual model of hypothesized climatic (direct and indirect) and landscape controls on Chl-a_{mod} of temperate lakes (modified from Hollert *et al.*, 2018). Climatic controls analyzed in the current study are in bold.

Simple variables (e.g., lake depth) were automatically calculated in ArcGIS, while compound variables (e.g., dynamic ratio) were calculated separately in Excel. The littoral zone of a lake is defined as the area adjacent to lake shore with depth of ≤ 2 m and bathymetric slope $\leq 2°$. Some very shallow study lakes had mean depth ≤ 2 m or less; therefore, in this case the area of the littoral zone was equal to lake surface area of these lakes.

Correlations among landscape variables were determined using the Pearson correlation test. The variables that did not meet the assumption of normality were ln-transformed (all variables except latitude, altitude, wetland cover, relative depth, development shoreline index, sphericity, and dynamic ratio). Variables found to have significant correlations ($p < 0.05$) with each other (i.e., have collinearity) were excluded from further analysis.

Table 3.3 Hypothesized landscape controls, their possible effects on lakes and proposed metrics to describe the controls.

Hypothesized landscape controls	Examples of hypothesized effects (with examples from literature)	Metric	Indicators of source areas	Indicators of hydrological flushing	Indicators of nutrients' fate
Location	The duration of ice-free period, spring/autumn runoff (Stomp *et al*., 2011; Kosten *et al*., 2012)	Latitude		✓	
Catchment size, topography and wetlands	Larger catchments, more complex topography and/or more wetlands indicate that nutrients spend more time in catchment, more input of nutrients (e.g., DOC, DON, P) (Creed & Band, 1998; Verhoeven *et al*., 2006; Mengistu *et al*., 2014)	Altitude, catchment area and length, catchment slope, wetland cover, drainage area to lake area ratio	✓	✓	✓
Lake surface area and mean depth, fetch	Higher dynamic ratios or larger fetches indicate increased probability of wind-driven sediment resuspension, more exposure to the sun/wind (Bachmann *et al*., 2000; Hakanson, 2005)	Lake surface area, fetch (length of longest lake axis; Wetzel, 2001), mean depth, dynamic ratio (square root of lake surface area divided by mean depth; Hakanson, 2005)	✓		✓
Lake volume and depth	Lakes with larger volumes lead to higher nutrient dilution in water, enhanced stratification (Nõges, 2009; Staehr *et al*., 2012).	Lake volume, mean depth, max depth, relative depth (Hutchinson, 1957)	✓		✓
Development of littoral zones (describes the littoral effect on lake and lake connection to surrounding landscape)	Wider littoral zones lead to development of rooted aquatic plants; increase the potential for enhanced nutrient loading (Kornijow *et al*., 2016)	Littoral zone, sphericity (Hutchinson, 1957), shoreline development (Hutchinson, 1957), bathymetric slope	✓		✓

3.3.5 Statistical analysis

Regression tree analysis was used to investigate environmental controls of ln Chl-a_{mod}. Regression tree analysis is a nonparametric recursive method that iteratively partitions data into mutually exclusive homogeneous groups with the smallest within-group variance for the response variable (De'ath & Fabricius, 2000). The analysis is particularly suited for ecological and environmental data, because these data often exhibit complex nonlinear relationships among predictor variables (De'ath & Fabricius, 2000; De'ath, 2002). Regression tree analysis was performed using landscape metrics and two climatic controls (i.e., T_{max} and Pr) as predictor variables to investigate environmental controls of ln Chl-a_{mod}. The regression tree was pruned at the branch where the complexity parameter minimized the cross-validation error (De'ath & Fabricius, 2000).

Since regression tree analyses may produce unstable models (Breiman, 2001; De'ath, 2002), the random forests analysis was also performed. Random forests overcome this flaw by producing numerous (up to 1,000 and more) trees, and then aggregating the predictions of each individual tree to a single model prediction (Breiman, 2001). This prediction can be presented on the "variable importance plot" showing the most important variables in an increasing order of importance. The "importance" can be based on two measures – either mean square-error ("%*IncMSE"*) or node purity ("*IncNodePurity*") (De'ath, 2002). Random forest analysis was performed with setting the forest size (number of trees) at 1,000 (Breiman, 2001). %*IncMSE* was used to calculate the importance of predictor variables; the variable having the lowest absolute value (i.e., lowest %*IncMSE*) was considered "unimportant", as suggested by Strobl *et al.* (2009).

Both regression tree and random forest analyses were performed in R (R Core Team, 2018) using "rpart", "party", and "randomForest" (for the random forests) packages.

3.4 Results

3.4.1 Correlation analysis

Pearson correlation analysis between various landscape variables showed that larger lakes (in terms of the surface area) had larger volumes, deeper depths, and larger catchments (Table 3.4). Both wetland cover and littoral zone area were positively correlated with catchment area and lake surface area but negatively correlated with altitude, lake depth, and catchment and bathymetric slopes. Littoral zone area was also strongly positively correlated with drainage area/lake area ratio. Dynamic ratio was poorly (but significantly) positively correlated with catchment area, catchment length and lake surface area, but negatively correlated with catchment and lake slopes and lake depth.

Pearson correlation analysis also revealed that most landscape variables were correlated with each other (i.e., collinearity of independent variables was present), which could potentially affect the results of the regression analysis (Kosten *et al.*, 2012). Therefore, I chose only those variables that did not have significant correlations with other variables, and at the same time could explain almost as much variation in ln Chl-a_{mod} as all variables.

Five variables representing various catchment and lake controls of ln Chl-a_{mod} were selected: lake volume (V), dynamic ratio (DR), littoral zone area (LZ), wetland cover (W%), and latitude (LAT). I chose V over lake surface area and lake depth because: (1) it can be a proxy of the other two lake properties; and (2) it is indicative of amount (i.e., volume) of water, and therefore, more suitable for description of processes associated with nutrients such as mixing and dilution in water column. I chose W% over other catchment properties because of the important role wetlands play in the storage and delivery of nutrients from catchments to receiving waters.

Table 3.4 Pearson correlation coefficients (r) between various landscape (catchment and lake) variables. All variables except latitude, altitude, wetland cover, relative depth, development shoreline, sphericity, and dynamic ratio were ln-transformed. Coefficients in bold indicate significant ($p < 0.05$) correlation.

	Altitude (m)	**Catchment area (ha)**	**Catchment length (m)**	**Catchment slope (°)**	**Wetland cover (%)**	**Drainage area/lake area**	**Lake surface area (ha)**	**Lake volume (m^3)**	**Lake fetch (m)**	**Lake mean depth (m)**	**Lake max depth (m)**	**Lake relative depth**	**Littoral zone area (ha)**	**Development shoreline**	**Sphericity**	**Bathymetric slope (°)**	**Dynamic ratio**
Latitude	**0.49**	-0.00	0.00	**0.19**	0.05	-0.06	-0.06	-0.02	-0.02	-0.02	0.02	0.05	-0.01	0.02	-0.05	0.03	-0.01
Altitude (m)		**-0.18**	**-0.19**	**0.38**	**-0.41**	**-0.14**	-0.10	-0.03	-0.10	0.10	**0.16**	**0.18**	**-0.19**	-0.07	0.04	**0.17**	**-0.17**
Catchment area (ha)			**0.98**	-0.09	**0.15**	**0.75**	**0.69**	**0.56**	**0.66**	0.12	**0.21**	**-0.34**	**0.28**	**0.36**	**-0.25**	**-0.15**	**0.14**
Catchment length (m)				-0.08	**0.15**	**0.75**	**0.68**	**0.55**	**0.66**	0.11	**0.20**	**-0.37**	**0.29**	**0.39**	**-0.26**	**0.15**	**0.14**
Catchment slope (°)					**-0.60**	-0.03	-0.09	0.07	0.11	**0.29**	**0.26**	**0.21**	**-0.16**	-0.10	-0.03	**0.40**	**-0.34**
Wetland cover (%)						0.03	**0.18**	0.02	**0.16**	**-0.25**	**-0.26**	**-0.28**	0.14	0.08	0.00	**-0.45**	0.13
Drainage area/lake area							0.04	-0.02	0.11	**-0.12**	0.08	-0.15	**0.72**	0.06	-0.10	-0.07	0.07
Lake surface area (ha)								**0.88**	**0.92**	**0.31**	**0.40**	**-0.42**	**0.56**	**0.49**	**-0.26**	**-0.15**	**0.14**
Lake volume (m^3)									**0.81**	**0.70**	**0.72**	-0.05	0.14	**0.44**	**-0.20**	**0.26**	0.13
Lake fetch (m)										**0.26**	**0.37**	**-0.42**	**0.56**	**0.67**	**-0.56**	-0.06	0.11
Lake mean depth (m)											**0.89**	**0.54**	**-0.26**	0.09	-0.04	**0.76**	**-0.70**
Lake max depth (m)												**0.54**	-0.01	**0.24**	**-0.12**	**0.68**	**-0.54**
Lake relative depth													**-0.52**	**-0.21**	**0.15**	**0.69**	**-0.47**
Littoral zone area (ha)														**0.41**	**-0.24**	**-0.47**	0.13
Development shoreline															**-0.53**	0.05	0.07
Sphericity																**-0.18**	0.02
Bathymetric slope (°)																	**-0.73**

3.4.2 Regression tree analysis

Regression tree analysis revealed complex interactions between climatic controls, landscape metrics and ln Chl-a_{mod} (Figure 3.3). Eleven groups were identified in the model. V was the most important predictor of Chl-a, making the first split in the tree at 13 ln m^3 (442×10^3 m^3), and clearly differentiating lakes with large (> 13 ln m^3) from lakes with small volumes (< 13 ln m^3). After V, Pr was the next most important predictor of ln Chl-a_{mod} in larger-volume lakes, while T_{max} was the next most important predictor of ln Chl-a_{mod} in lakes with smaller (< 13 ln m^3) volumes. Landscape metrics (DR, W% and LZ) explained part of ln Chl-a_{mod} but appeared in the regression tree only for the larger-volume lakes. LAT did not appear in the tree and was the least significant variable reported by the random tree analysis (Figure 3.3b). The response variable (ln Chl-a_{mod}) in each group showed an overall increase from the first group (median Chl-a_{mod} = 0.9 ln μg L^{-1}) to the last group (median Chl-a_{mod} = 2.6 ln μg L^{-1}).

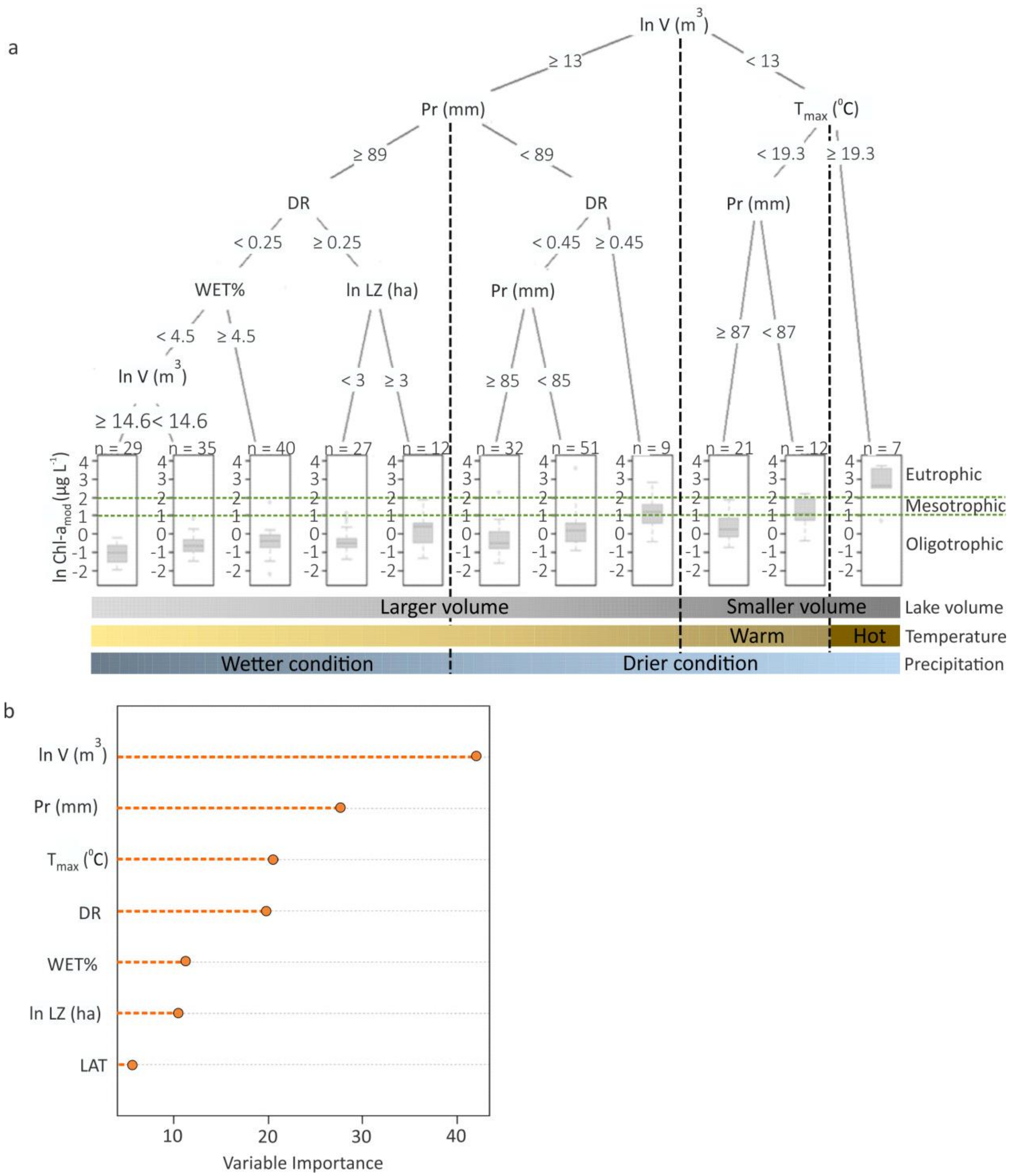

Figure 3.3 (a) Regression tree, and (b) results of the random forests analysis depicting climate and landscape determinants of ln Chl-a_{mod}. Panels with colors below the regression tree depict general patterns (increase or decrease) of the most important determinants of ln Chl-a_{mod} over an increase ln Chl-a_{mod} from left to right. Abbreviations: Pr–precipitation, T_{max}–maximum air temperature, V–lake volume, DR–dynamic ratio, WET%–wetland cover, LZ–littoral zone, LAT–latitude; ln indicates ln-transformed values.

3.4.3 Conceptual panels on lake trophic states

Conceptual panels describing "typical landscape features" (i.e., median values based on landscape metrics identified in the regression tree analysis) for each lake trophic state were generated (Figure 3.4). Median values of T_{max} and Pr were also provided for each trophic state.

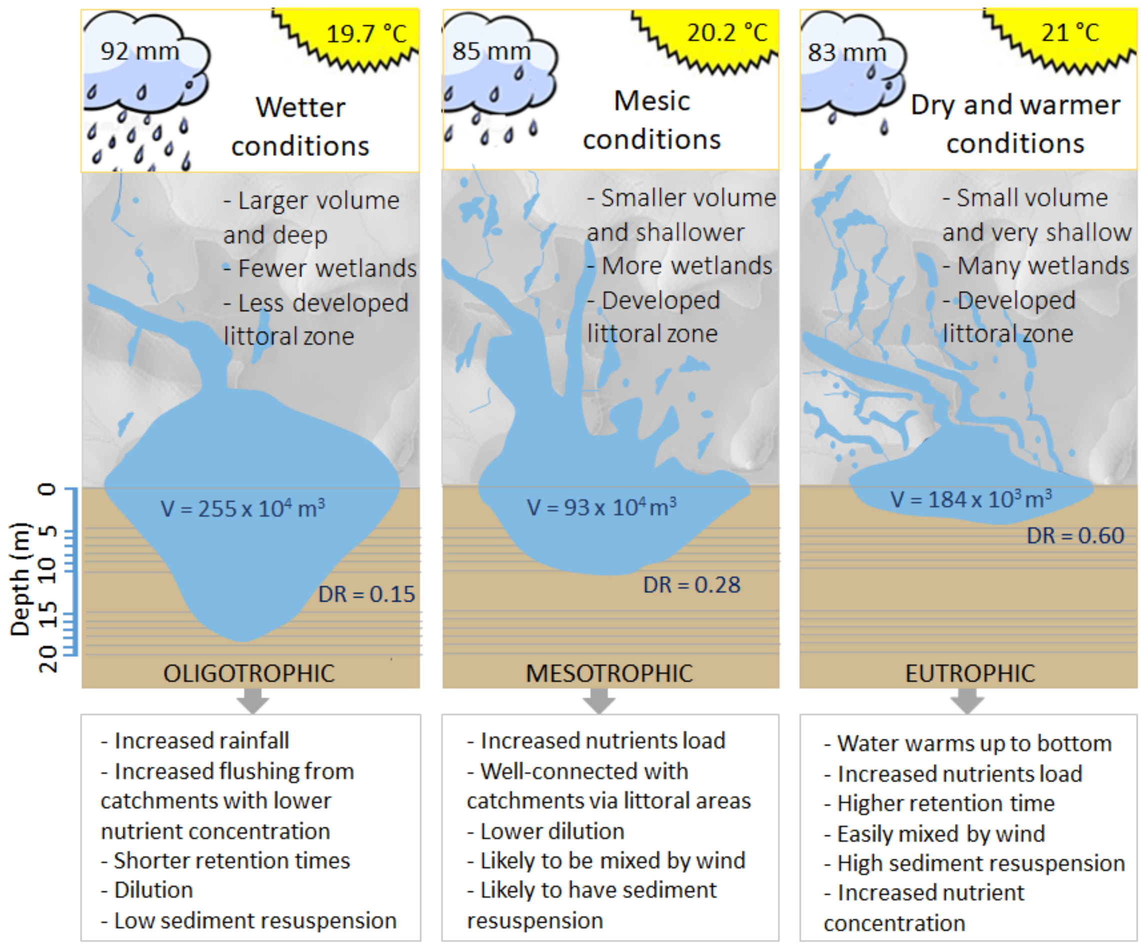

Figure 3.4 Conceptual panels depicting "typical landscape features" for each lake trophic state. Values of climate and landscape determinants of ln Chl-a_{mod} are median (except for max depth, for which values are mean). Abbreviations are metrics developed to describe the landscape determinants: V–volume, DR–dynamic ratio.

T_{max} increased from oligotrophic (19.7°C) and mesotrophic (20.2°C) to eutrophic (21°C) lakes. In contrast, Pr decreased following the same order as T_{max} (Pr for oligotrophic lakes = 92 mm, mesotrophic lakes = 85 mm and eutrophic lakes = 83 mm).

V was largest in oligotrophic lakes (255×10^4 m^3), followed by mesotrophic (93×10^4 m^3) and eutrophic (184×10^3 m^3) lakes. Eutrophic lakes had the highest DR (0.60) with max DR reaching 3.60; mesotrophic lakes had the second highest DR (0.28) followed by oligotrophic lakes (0.15). W% increased from oligotrophic (4.4%) to mesotrophic (5%) and eutrophic (9.9%) lakes. Maximum W% was also found in eutrophic lakes where it reached 64% (in comparison to maximum W% of oligotrophic and mesotrophic lakes where it was 22% and 38% accordingly). Oligotrophic lakes had the smallest median LZ (1.9 ha), followed by mesotrophic lakes (3.7 ha) and eutrophic lakes (15.7 ha).

3.5 Discussion

I examined the interactive effects of climate (air temperature and precipitation) and landscape properties on phytoplankton biomass and associated trophic condition. Based on the analysis of dataset of 275 temperate lakes covering the whole range of lake trophic states (from oligotrophic to mesotrophic to eutrophic), I found a relationship between T_{max} and Pr and ln Chl-a_{mod}. However, I also found a relationship between several landscape metrics (i.e., V, DR, W%, LZ and Lat) and ln Chl-a_{mod}, providing evidence that both climate and landscape properties are important predictors of Chl-a in lakes.

Climate controls

Climate was an important factor in determining Chl-a, but in unexpected ways. Pr was an important predictor of ln Chl-a_{mod} in larger-volume lakes, where ln Chl-a_{mod} decreased as Pr increased (Figure 3.3a). In contrast to Pr, T_{max} predicted ln Chl-a_{mod} only in smaller-volume lakes. Further, even in smaller-volume lakes, T_{max} was the sole climatic predictor of ln Chl-a_{mod} in only seven lakes; in all other smaller-volume lakes Pr came into play. This finding indicates that Pr but not T_{max} is the main driver of phytoplankton biomass in the lakes of the study region.

The relationship between Pr and phytoplankton biomass remains unclear (O'Neil *et al.*, 2012; de Senerpont Domis *et al.*, 2013). While more precipitation mobilizes nutrients on land, potentially leading to increasing nutrient enrichment of receiving waters in some regions (Paerl & Huisman, 2008), the relatively undisturbed region in this study is known to have naturally low P in soils (Jeffries & Snyder, 1983), and to have experienced steadily decreasing deposition of P (Eimers *et al.*, 2009) and total N for at least the last 20 years (Mengistu *et al.*, 2014; Geddes & Martin, 2017). Therefore, it seems unlikely that high precipitation (and associated runoff) significantly contributes to increased nutrients and by extension phytoplankton biomass in the study lakes, at least at present. Further, larger-volume lakes can mitigate the impact of nutrient loading by the effect of dilution (de Senerpont Domis *et al.*, 2013; De Sousa Barroso *et al.*, 2016). Larger-volume lakes generally have "more water" resulting in shorter water residence times, lower concentration of nutrients and, therefore, lower phytoplankton biomass (Staehr *et al.*, 2012).

To my knowledge, there are very few studies that describe the effects of natural dilution of nutrients in lakes caused by precipitation (e.g., Abongwa & Atekwana, 2018). Most of studies on lake dilution address "artificial" dilution as a generally successful (but nevertheless expensive) technique to control algal blooms and minimize agricultural/industrial eutrophication (so-called "lake restoration measure"; see Welch *et al.*, 1972). In these studies, the effect of artificial dilution on lake water column refers to reducing nutrient concentration to growth-limiting levels of algae and decreasing water residence time, which lead to slowing down the growth rate of the algae and eventually to a decrease in algal biomass (Welch *et al.*, 1972; Shinohara *et al.*, 2008). Although described as a restoration measure for human-induced eutrophication in lakes, the same chain of events might happen in a natural system (especially in a larger-volume lake) as well, where high precipitation reduces residence times, facilitating water exchange, and diluting nutrients in a water column (Tang *et al.*, 2019).

The relationship between temperature and phytoplankton biomass also remains unclear. The idea that temperature (and increasing air temperature in particular) drives phytoplankton biomass and contributes to eutrophication has a little support in the field studies and laboratory experiments. Most of the studies on temperature-phytoplankton relationship show that this relationship is very complex and depends upon nutrient availability, the rate of temperature increase, changes in precipitation patterns, structure of phytoplankton and zooplankton, etc. (Gerten & Adrian, 2002; Moss *et al.*, 2003;

Blenckner *et al.*, 2007; Striebel *et al.*, 2016; Richardson *et al.*, 2018). Therefore, depending on these and many other factors, an increase in temperature can have a positive, negative or no effect on phytoplankton biomass (Jeppesen *et al.*, 2009; Tadonléké, 2010; Kraemer *et al.*, 2017; Rasconi *et al.*, 2017). Further, many studies (e.g. Gerten & Adrian, 2002; Jöhnk *et al.*, 2008; Mosley, 2015) examined temperature-phytoplankton relationship during prolonged periods of droughts and/or exceptionally warm years or seasons, hence emphasizing phytoplankton behavior under extreme conditions but not under regular (e.g., yearly averaged) or gradually increasing temperature.

Of all lakes, the seven lakes for which T_{max} was the sole climatic predictor of ln Chl-a_{mod} had the smallest V and the smallest depth (median max depth = 1.5 m). This might indicate that direct physical factors associated with temperature (e.g., heat distribution through the water column) play a major role in driving phytoplankton biomass in these lakes. Indeed, the water column of smaller-volume lakes generally warms up faster and deeper (often all the way down to lake bottom) than that of larger-volume lakes (Johnson *et al.*, 2014; Sharma *et al.*, 2019). My findings also suggest that phytoplankton in smaller-volume lakes is more susceptible to drier/dry conditions (i.e., with reduced precipitation). This interpretation is consistent with many other studies demonstrating that smaller-volume lakes are generally more sensitive to broad scale climate stressors (e.g., temperature and precipitation) due to the lower capacity to buffer these stressors (Choi, 1998; Whitehead *et al.*, 2009; Sharma *et al.*, 2019).

The interactive effects of higher temperatures accompanied by reduced precipitation or prolonged droughts can lead to longer water residence times (Zwart *et al.*, 2017), lower dilution potential, and increasing nutrient levels in water column (Mosley, 2015) – all of which lead to higher ln Chl-a_{mod}. These climatic factors can cause a significant decrease in water level of smaller-volume lakes, resulting in sediments being in a direct contact with the trophogenic layer (the upper photosynthetically active layer of the lake) (Søndergaard, 2007; Nõges, 2009). In this case, lake sediments might be easily disturbed by winds (especially in lakes with relative large surface areas), leading to intensified internal nutrient loading. Nõges *et al.* (2007) analyzed internal nutrient loading of two

Estonian lakes and found that shallower lakes have lower P retention in the sediments; hence, P availability in the water column of these lakes was found to be higher than in deeper lakes. The intensified internal nutrient loading can also happen in relatively deeper lakes (without direct sediment contact with the trophogenic layer) caused by anoxia. This condition is often met when high temperatures and reduced precipitation lengthen the period of thermal stratification, leading to reduced vertical mixing of water column (Winder & Sommer, 2012) and as a result promoting lower levels of dissolved oxygen and anoxic conditions (Nürnberg, 2009; Dittrich *et al.*, 2013). However, it has been demonstrated by previous studies that oligotrophic and mesotrophic lakes of the temperate forest biome (the Muskoka region in particular) tend to have relatively low rates of internal P loading (Nürnberg & LaZerte, 2004; Orihel *et al.*, 2017).

While climatic factors contribute selectively to the growth of eukaryotic phytoplankton (depending on species; Striebel *et al.*, 2016), they favor most of the cyanobacteria taxa (Jöhnk *et al.*, 2008; Kosten *et al.*, 2012). Cyanobacteria have growth optima at higher temperatures than other phytoplankton (Carey *et al.*, 2012; Lürling *et al.*, 2013) and they have the ability to regulate buoyancy, especially during periods of stratification (O'Neil *et al.*, 2012). These abilities together with an elevated nutrient concentration of internal or external origin create a perfect environment for cyanobacteria to thrive and develop algal blooms (Paerl & Huisman, 2008), resulting in increased overall phytoplankton biomass. Although Kosten *et al.* (2012), who studied 143 lakes from subarctic Europe to southern South America, did not determine a significant relationship between higher temperatures and higher overall phytoplankton biomass, they found that the proportion of cyanobacteria in phytoplankton communities of shallow lakes significantly increased with temperature.

Landscape controls

Landscape was also an important factor in determining Chl-a. Landscape properties are known to be key controls on nutrient accumulation, transformation and transport within a coupled catchment-lake system (Magnuson *et al.*, 1990; Baines *et al.*, 2000). Lake volume (V) was found to be the most important predictor ln Chl-a_{mod} (Figure 3.3). As a surrogate of lake surface area and depth, V describes lake features such as exposure to the sun and wind, residence time, and mixing of water and nutrients within lakes (Nõges, 2009). Therefore, V can be used for assessing the relative importance of both direct atmospheric and indirect landscape-filtered controls on lakes (Magnuson *et al.*, 1990). My finding that Chl-a increases with decreasing lake volume and depth is in good agreement with other studies (e.g., Duarte & Kalff, 1989; Stomp *et al.*, 2011; Staehr *et al.*, 2012). Stomp *et al.* (2011) analyzed Chl-a and phytoplankton composition in 540 lakes throughout the continental USA and found that shallow lakes generally had higher Chl-a concentration. The fact that lake volume (but not climatic factors) makes the first split in the regression tree (Figure 3.3a) might indicate that indirect controls (i.e., filtered by landscape characteristics) are more important in regulating Chl-a in the study lakes.

Increases in DR, W% and LZ were correlated to ln Chl-a_{mod}. DR was initially developed to identify the relative area of lake bottom influenced by wind-driven resuspension and has often been used for an assessment of lake bottom dynamics and the intensity of wave disturbance (Qin *et al.*, 2004; Hakanson, 2005; Zhu *et al.*, 2015). Higher DR represents a higher risk for wind-induced sediment resuspension events (Bachmann *et al.*, 2000; Hakanson, 2005) and the influx of nutrients from sediments that promote phytoplankton growth (Søndergaard *et al.*, 2001). In smaller-volume lakes, DR may serve as an indicator of a nutrient source, while in larger-volume lakes (i.e., where sediments are less influenced by wave-induced sediment resuspension), DR may serve as an indicator of a nutrient sink (i.e., nutrient sedimentation and accumulation at the bottom of the lake). The residence time of nutrients is generally longer and their content is more uniform (i.e., less disturbed) in sediments of larger lakes than in sediments of small or shallow lakes (Søndergaard, 2007). Although DR was the third most important predictor of ln Chl-a_{mod} in larger-volume lakes, it did not appear in the smaller-volume lake section of the regression tree.

Increases in W% were related to increases in ln Chl-a_{mod}. Wetlands typically covered a small portion of catchments (e.g., median proportion of wetlands in catchments of oligotrophic lakes in the study region is only 4.4%). Despite this, their contribution to the nutrient loading to lakes and streams is extremely important (Mitsch & Gossilink, 2000;

Verhoeven *et al.*, 2006). This is due to their position in catchment and biogeochemical processes occurring within them (Marton *et al.*, 2015). Wetlands are often located in hydrologically-connected depressions, areas with low relief and near lakes, and therefore a large proportion of catchment discharge waters pass through them on their way to receiving waters (Creed & Band, 1998). Wetlands also affect water residence time in catchments (McGuire *et al.*, 2005; Pelster *et al.*, 2008). Generally, water residence time increases with increasing wetland cover, as wetlands slow flows on the way to discharge waters (McGuire *et al.*, 2005). While intercepted in wetlands, water chemistry is changed–nutrients are retained, removed or transformed to different forms until they are flushed out with the next runoff (Harms *et al.*, 2016).

Increases in LZ were related to increases in ln Chl-a_{mod}. Littoral zone area can indicate both fate and source of nutrients in lakes (Kornijów *et al., 2*016; Orihel *et al.*, 2017). However, in this study, I could not separate one role from another because I did not differentiate LZ from the pelagic part of lake–in this sense, lake V and lake LZ were interconnected. However, some patterns describing the effect of LZ on ln Chl-a_{mod} still can be observed in the regression tree. For example, an increase in ln Chl-a_{mod} with increasing LZ is likely due to the fact that more developed LZ (i.e., larger littoral areas) means a closer connection of lakes with their catchments. This in turn reflects the potential for enhanced external nutrients loading (Vadeboncoeur *et al.*, 2002) and as a result increased ln Chl-a_{mod}.

Further, it is reasonable to assume that LZ in some of my lakes might act as a buffer where external material (i.e., sediments and nutrients) accumulate and/or are taken up by macrophytes and attached algae. For example, oligotrophic and mesotrophic lakes that are under mesic condition (i.e., relatively wet; Figure 3.3a) receive a reasonable amount of water and nutrients from runoff. Lake volumes (V) may not be large enough for the high rate of dilution, while the depth may not be small enough to initiate sediment resuspension and/or allow water column to warm down to the bottom. However, the lakes still might have some reasonably shallow areas for the development of LZ with communities of rooted macrophytes, which act as the buffer, and presumably protect these lakes from shifting to more eutrophic condition. Some studies on regime shifts in lakes exemplify this sequence of events (e.g., Scheffer & Van Nes, 2007; Winder & Sommer, 2012).

Conceptual model

My findings suggest that landscape properties influenced ln Chl-a_{mod} in temperate lakes and its response to climatic factors (T_{max} and Pr) by modifying these climatic factors directly (e.g., temperature *versus* phytoplankton biomass of smaller-volume lakes) and indirectly (e.g., through surface water and nutrients discharge). Based on these observations, I developed a simple conceptual model based on different climate conditions (in relation with T_{max} and Pr) and landscape properties for each lake trophic state (oligotrophic, mesotrophic and eutrophic; Figure 3.4) within the temperate forest biome.

Oligotrophic lakes received the highest Pr and hence higher runoff (for July–October). Increased inflow of water results in shorter water residence times (Cardille *et al.*, 2004; Nõges, 2009). Low W% of surrounding landscapes indicates relatively low nutrient concentration in the runoff. Additionally, oligotrophic lakes have reduced connections with their catchments due to small LZ. Since oligotrophic lakes have the largest V and increased Pr, they are prone to nutrient dilution. Further, these lakes are not susceptible to wind-driven sediment resuspension, which is indicated by the low DR index and by the fact that oligotrophic lakes are also the deepest (i.e., sediments are unlikely to be disturbed by wind especially under high Pr).

Mesotrophic lakes received less Pr, but are characterized by higher external nutrient loading than oligotrophic lakes. This is because these lakes have more wetlands (i.e., larger W%), which are well-connected to the lakes by more developed LZ. Since mesotrophic lakes have smaller V, the process of nutrient dilution is less pronounced there, while their moderate depth, higher DR and more developed LZ indicate higher rate of resuspension from lake sediments. Additionally, LZ of these lakes are likely to be occupied by communities of rooted macrophytes (Wetzel, 2001; Kornijów *et al.*, 2016).

Eutrophic lakes received the least Pr and were warmer (they have the highest T_{max}) than oligotrophic and mesotrophic lakes. Although eutrophic lakes have the smallest V indicating shorter water residence time, reduced water inflow makes water "stay longer" in the lakes, therefore extending the residence time. Further, because of high T_{max} and shallow depth, water column of eutrophic lakes might warm down to the bottom, leading to reduced concentration of dissolved oxygen and enhanced internal nutrients loading from sediments (Nõges, 2009). This is supported by large LZ (which indicates extensive shallowness) and very high DR. The latter also means eutrophic lakes have large fetches and hence are susceptible to wind disturbance. Finally, these lakes have high loading of allochthonous material because of extensive W%.

3.6 Conclusions

Phytoplankton biomass in the lakes of the temperate forest biome is influenced by air temperature and precipitation and filtered through catchment and lake characteristics. Of all metrics developed do describe landscape controls on ln $Chl\text{-}a_{mod}$, V is found to be the most important, as it regulates water mixing, and nutrient dilution and resuspension. I found that lakes with lower ln $Chl\text{-}a_{mod}$ had larger V, were sensitive to Pr and were mostly oligotrophic (lakes with minimum sediment resuspension, less-developed LZ and water discharged from catchments with small W%). On the other hand, lakes with the higher ln $Chl\text{-}a_{mod}$ had smaller V, were sensitive to T_{max} and were either mesotrophic or eutrophic (lakes that were very sensitive to wind-driven sediment resuspension with developed LZ, which were better connected to their catchments with higher W%).

3.7 References

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4 Ecological stability in trophic state of temperate lakes

4.1 Introduction

Increased reports of potentially harmful algal blooms in northern intact landscapes (Carey *et al.*, 2008; Winter *et al.*, 2011) may be evidence of lakes experiencing regime shifts between alternative stable states (Scheffer & Van Nes, 2007). While anthropogenic eutrophication is undoubtedly an important driver of the shift from oligotrophic towards eutrophic states, there is mounting evidence that these regime shifts may also be a consequence of the impaired stability of lake ecosystems caused by long-term effects of climate change (Scheffer, 2001; Scheffer & Van Nes 2007; de Senerpont Domis *et al.*, 2013; Wagner & Adrian 2009; Dakos *et al.* 2014). For example, potentially harmful algal blooms have been reported in temperate lakes in relatively pristine landscapes at large distances from urban areas or agricultural lands (Winter *et al.*, 2011).

Climate change may have little apparent immediate effect on an ecosystem but can still undermine its stability and cause loss of resilience over time (Scheffer *et al.* 2001). Ecological resilience theory suggests that an ecosystem has at least two stable states that are separated by an unstable state(s) (Holling, 1973). An ecosystem becomes increasingly unstable until a bifurcation point is passed, at which the ecosystem shifts to a new stable state (Andersen *et al.*, 2009; Scheffer *et al.*, 2012). In colloquial terms, this is referred to as “the tipping point.” As an unstable ecosystem approaches the bifurcation point, its response to small perturbations slows–a phenomenon called “critical slowing down” (Andersen *et al.*, 2009).

Indicators of the critical slowing down phenomenon in time series–and therefore instability and regime shift–include abrupt rises in short-term autocorrelation and variance (Carpenter & Brock, 2006; Dakos *et al.*, 2008; Wang *et al.*, 2012) and a shift to lower variance frequencies of ecosystem variables (Kleinen *et al.*, 2003). Temporal variance in time series (expressed in standard deviations: SD) is the most commonly used indicator due to its comparative ease of measurement (Scheffer *et al.*, 2009; Lindegren *et al.* 2012; Boettiger *et al.*, 2013). Temporal variance in time series generally increases as an ecosystem accumulates the effects of shocks from small perturbations and approaches the bifurcation point (Kuehn, 2011).

Detection of critical slowing down indicators can fail or, in the worst case, result in false alarms (Lenton *et al.*, 2012; Boettiger *et al.*, 2013). False alarms can arise because indicator measurements may be due to extrinsic rather than to intrinsic factors influencing ecosystem dynamics (Carpenter & Brock, 2006). It is therefore important to isolate variance resulting from intrinsic factors from extrinsic factors that include non-stationary (e.g., annual temperature increase as a constituent of climate change) and stationary (i.e., climate oscillations) signals. De-trending is already commonly used to remove non-stationary signals from ecological time series (Lenton *et al.*, 2012). Sophisticated spectral methods (e.g., wavelet analysis) have since emerged to identify stationary signals in time series (Sabo & Post, 2008; Mengistu *et al.*, 2013a; Ruhí *et al.*, 2015). After non-stationary and stationary signals are removed from a time series, residuals in the distribution of the remaining signals represent variance due to intrinsic ecosystem dynamics (Fung *et al.*, 2013). Low and high variation in SDs of residuals in sliding time series windows indicate ecosystem stability and instability respectively, while increasing or decreasing trends in SDs of residuals over time indicate movement towards new stable states (Dakos *et al.*, 2014; Arnoldi *et al.*, 2016).

In the face of reports of eutrophication and algal blooms in temperate forest regions, I posed three questions: (1) How stable are trophic states of temperate lakes? (2) Are regime shifts in lake trophic states occurring? And (3) if temperate lakes are losing their resilience and experiencing any change in their stability (i.e., become unstable), is this driven by climate change in terms of rising temperatures and changing precipitation patterns? To respond to these questions, I removed non-stationary and stationary signals from a 28-year (1984-2011) time series of chlorophyll-*a* concentration (Chl-a) (as a proxy of phytoplankton biomass) in 12,644 lakes in a relatively undisturbed temperate forest biome in central Ontario, Canada, and used SDs of the residuals as an indicator of lake stability. I used the conceptual ("stability landscapes") model developed by Scheffer *et al.* (2001) as a template to classify patterns in the SDs of the residuals of Chl-a times series into lake stability classes which I compared to lake trophic conditions (i.e., trophic states). This conceptual model is illustrated in Figure 4.1 with two alternative stable states and several transitional/unstable states of temperate lakes embedded into the stability landscapes of a relatively intact landscape. Over time, the structure and function of a lake ecosystem with high resilience (e.g., lake with an oligotrophic condition) remain relatively stable. However, changing external conditions (e.g., increasing air temperature) can lead to a gradual loss of resilience up to a point where even a small disturbance can push the ecosystem into a new stability domain, where the system reorganizes into a new and radically different (e.g., oligotrophic *versus* eutrophic) stable state (Scheffer *et al.*, 2012). Once in a new stable state, the lake ecosystem is kept there by internal feedback dynamics of that state (e.g., prevalence of buoyant cyanobacteria), making the recovery to a previous state difficult (Scheffer *et al.*, 1993; Scheffer *et al.*, 2012). Lakes found to experience changes in trophic states (transitional lakes) were then related to changes in temperature and precipitation to determine how these climate factors can drive lake instability and regime shifts towards new trophic states.

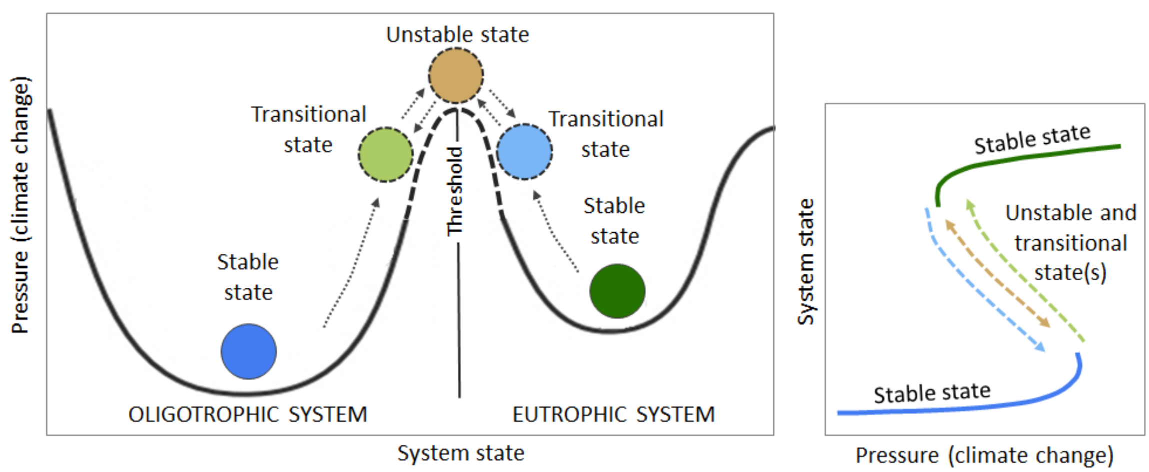

Figure 4.1 "Multiple stable states" concept depicted using "stability landscapes" (as exemplified by freshwater lakes). Valleys represent stability domains, in which a stable system, represented by the ball, is kept by internal feedback mechanisms until an external pressure is long and "stressful" enough to move the ball into a new stability domain, where the system reorganizes into a new stable state (modifed from Scheffer *et al.*, 2001).

4.2 Study region

The study region is the temperate forest biome located within the Great Lakes–St. Lawrence Forest region in Ontario, Canada (Figure 4.2). Climate in the region is humid continental, with precipitation influenced by lake effects from the Great Lakes and local orographic effects in areas of high relief (Baldwin *et al.*, 2000). Mean annual air temperature in the study region for the period of 1984-2011 was +5.1°C, ranging between +7.4°C in the south-east and +2.6°C in the north. Mean annual precipitation for the same period was 960 mm, ranging from 740 mm in the southern areas of the region to 1180 mm in the north-west (McKenney *et al.*, 2011). Geology is the Precambrian rocks comprised of silicate greenstone of the Boreal (Canadian) Shield (Ontario Geological Survey, 2003). Topography varies from flats and depressions along the shore of the Great Lakes to hills and uplands (Algoma and Madawaska Highlands). Soils are thin and undifferentiated brunisols (in the south), and thick and differentiated orthic ferro-humic podzols (in central areas and in the north) (Canada Soil Survey Committee, 1978). Forests in the region lie in a transitional zone between deciduous and coniferous with the latter being more prevalent in the northern areas (Baldwin *et al.*, 2000).

Climate is changing in the temperate forest biome. Simple linear regression analysis shows that annual mean July-October (i.e., months that are under consideration in the study) maximum daily temperatures increased significantly at a mean rate of 0.045°C yr^{-1} over the 1984–2011 period ($p < 0.05$). Trends in mean annual July-October total precipitation for the same period are less clear; the precipitation was variable from year to year with decreasing trends (a mean rate of -0.24 mm yr^{-1}) in western areas and increasing trends (a mean rate of 0.17 mm yr^{-1}) in central and south-eastern areas of the study region.

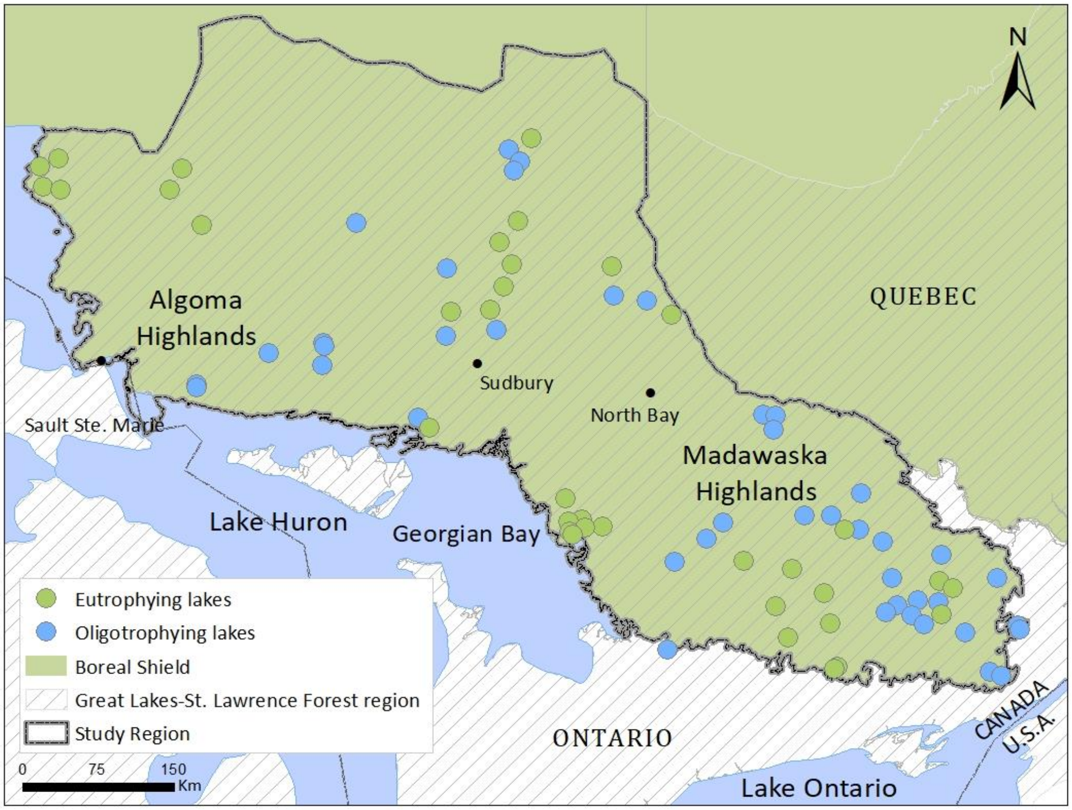

Figure 4.2 Map showing location of the study region (the temperate forest biome) and eutrophying (n = 36) and oligotrophying (n = 42) lakes used for the analysis of environmental controls of transitional lakes.

4.3 Materials and Methods

4.3.1 Modeled Chlorophyll-*a* time series

Landsat 4-5 TM (1984-2011) (1,067 images) and Landsat 7 ETM+ (1999-2003) (159 images) 30 m products for the period of August to October were acquired from the United States Geological Survey archives covering a 152,231 km^2 area (Figure 4.2; see Chapter 2). Lake locations and boundaries in each image were determined by reclassifying pixels below the local minimum in the bimodal distribution of band 5 images (shortwave infrared) as surface water. To avoid the problem of mixed reflectance due to adjacency near or along lake shorelines (areas with very shallow water and abundant aquatic vegetation), lakes with area less than 4.5 ha (i.e., 30 m pixels) and high SD of reflectance in band 5 were discarded, and the remaining lakes were buffered inside to a distance of 15 meters (1/2 the pixel distance). Atmospheric correction was conducted by subtracting the Rayleigh scattering radiance from top of atmosphere (TOA) radiance. The atmospherically corrected TOA radiance values were then converted to TOA unitless reflectances. An algorithm based on TOA reflectance values from three Landsat bands [(band 1-band 3)/band 2)] was developed and related to natural log transformed Chl-a (ln Chl-a) observed in lakes. This relationship was then applied to all mean lake reflectance values in the Landsat archive, generating a times series (28 years from 1984 to 2011) of modeled annual August-October Chl-a (ln Chl-a_{mod}) for 12,644 lakes. See Chapter 2 for further details.

4.3.2 Climate and landscape variables

Climate variables include time series of annual mean July to October maximum air temperature (T_{max}) and annual mean July to October total precipitation (Pr) grids for 1984-2011 that were calculated from 300 arc-second resolution monthly grid climate data (McKenney *et al.*, 2011). The period of July to October was chosen over August to October (the period of ln Chl-a_{mod} measurements) because phytoplankton biomass generally delays in responding to environmental factors (e.g., temperature and precipitation) (Wetzel, 2001). T_{max} and Pr grid values were extracted at lake centroids and annual rates of change were calculated as the slope parameters.

Landscape variables were selected as proxies of potential controls of ln Chl-a and lake stability (see Chapter 3 for sources of data and derivation of metrics). These variables included lake volume (V), dynamic ratio (DR), wetland cover (W%), littoral zone area (LZ), and geographical latitude (LAT). The landscape metrics were computed for 36 eutrophying and 42 oligotrophying lakes. Calculations were implemented in ArcGIS (ArcMap, version 10.2) and Microsoft Excel.

4.3.3 Non-stationary and stationary signals in Chlorophyll-*a* time series

Non-stationary and stationary signals in the time series of ln Chl-a_{mod} were removed following the steps presented in Figure 4.3 using MATLAB (R2013b, the WathWorks Inc).

Non-stationary signals (trends) in the ln Chl-a_{mod} time series for each lake were identified using the Mann-Kendall non-parametric trend test (Kendall, 1975). Time series with significant ($p < 0.05$) trends were de-trended by subtracting linearly regressed ln Chl-a_{mod} (i.e., ln Chl-a_{mod} regressed to year as the predictor variable) from ln Chl-a_{mod}.

Stationary signals (oscillations) in the ln Chl-a_{mod} time series for each lake were identified and sequentially removed using wavelet analysis. Wavelets are defined as small “groups” of waves with specific frequencies that approach zero at both ends.

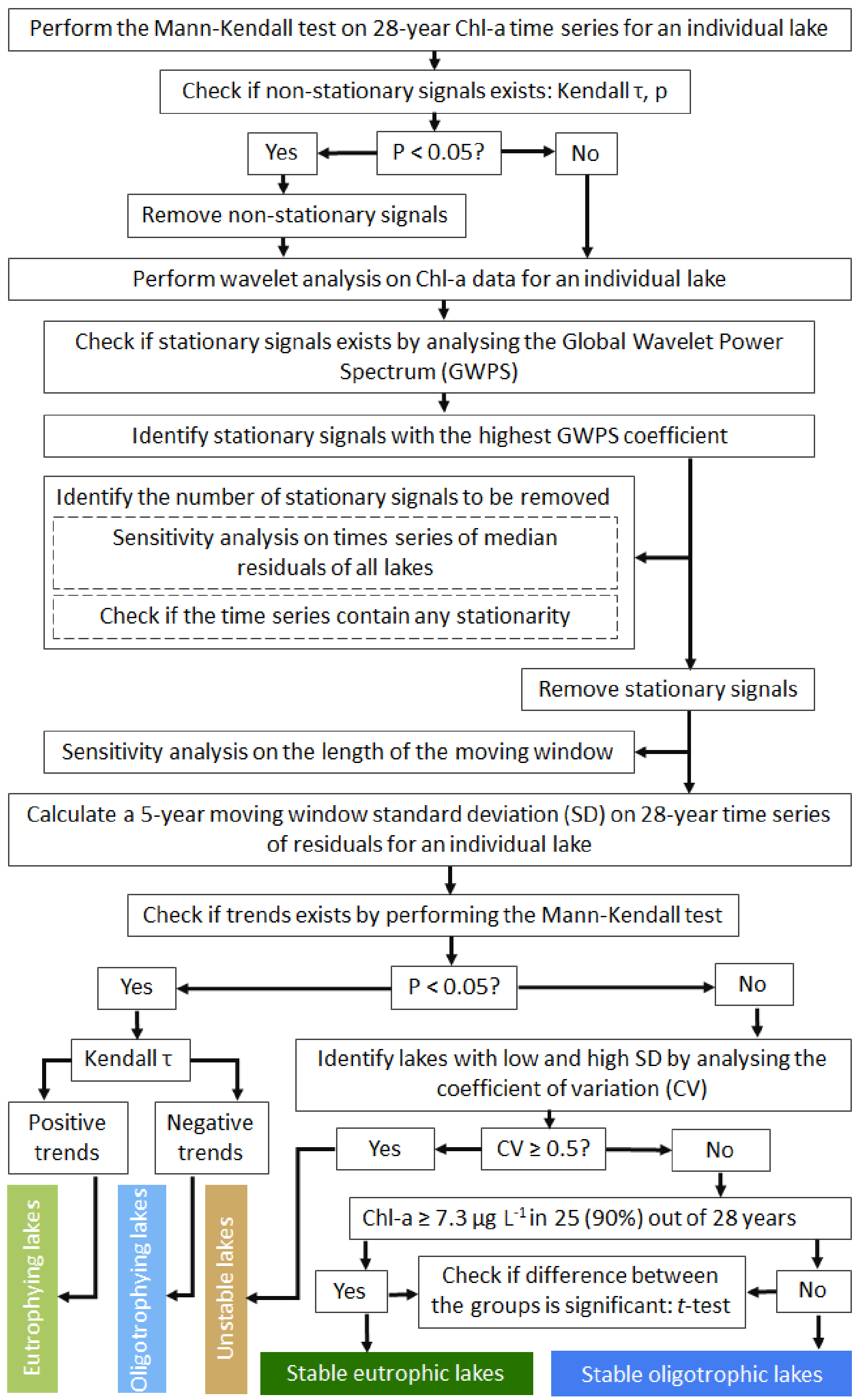

Figure 4.3 Flowchart summarizing steps for removing non-stationary and stationary signals from ln Chl-a_{mod} time series and identification lake trophic stability classes.

Wavelet analysis allows for decomposition of a time series into a time-frequency domain where dominant periodicities can be detected (Torrence & Compo, 1998; Labat, 2010). Wavelet power spectrums in ln Chl-a_{mod} time series (both de-trended time series and original time series with no significant non-stationary signals, hereafter called “time series”) were obtained for each lake. Wavelet power spectrums were computed by convoluting the time series with a scaled version of a transforming wavelet function. The continuous Morlet wavelet was applied because it provides a good time/frequency resolution compared to other wavelet types (Labat, 2010), and it has been successfully used in many analyses conducted on ecological, climatological and hydrological time series (e.g., Cazelles *et al.*, 2008; Kogovšek *et al.*, 2010; Santos & de Morais, 2013; Mengistu *et al.*, 2013b).

Global wavelet power spectrum (GWPS) coefficients were computed by time-averaging wavelet spectrum values over the local spectra. Scales with large GWPS coefficients were assumed to contribute more and significant spectral energy, while scales with small GWPS coefficients were assumed to contribute small or insignificant spectral energy (Mengistu *et al.*, 2013a). Stationary signals with the largest GWPS coefficients were identified and sequentially removed from the time series of ln Chl-a_{mod} in each lake. A sensitivity analysis was conducted to identify the number of stationary signals to be removed. After removal of each stationary signal (i.e., at the end of each step), the median values of the residuals from all 12,644 lakes were calculated and plotted, and the point at which a polynomial line of the residuals leveled off was considered the maximum number of stationary signals that should be removed.

4.3.4 Lake stability classification

A moving window of the time series of SDs of residuals of ln Chl-a_{mod} were extracted in 3-, 5-, 7- and 10-year lengths in 2,000 randomly selected lakes. Significance (p-value) of Kendall rank correlation coefficients (τ) in the time series of SDs of residuals from linear trends within the moving windows were calculated using Mann-Kendall non-parametric trend tests to determine the window length with the most significant (smallest p-value) correlation in the majority of lakes. Time series of SDs of residuals (SD_{mv}) were then calculated in all lakes using the window length found to be most significantly correlated in 95% of 2,000 randomly selected lakes.

Lake stability classes were identified. First, to identify lakes experiencing gradual changes in stability (i.e., transitional lakes), trends in SD of residuals were evaluated in each lake by calculating τ in time series of SD_{mw} using Mann-Kendall non-parametric trend tests. Lakes with significant ($p < 0.05$) and positive τ were classified as eutrophying, while lakes with significant ($p < 0.05$) and negative τ were classified as oligotrophying. Second, to identify lakes experiencing "instability", the coefficient of variations (CV) of SD_{mw} was calculated for each remaining lake (Lindegren *et al.*, 2012). Lakes having $CV_{mw} \geq 0.5$ in more than 90% of time series of SD_{mw} were classified as unstable. Third, the remaining lakes were classified as stable eutrophic if their Chl-a_{mod} concentration was ≥ 7.3 µg L^{-1} (minimum Chl-a concentration for eutrophic lakes; Carlson & Simpson, 1996) in more than 90% of 28 years, and differences in mean SD_{mw} between these lakes and remaining lakes (i.e., stable oligotrophic) was significant ($p < 0.05$). Significance was determined by applying a *t*-test. Those lakes that were not identified as eutrophic were identified as oligotrophic.

The statistical significance of differences among lake stability classes (except for stable oligotrophic *versus* stable eutrophic) was examined using one-way analysis of variance on ranks (ANOVA on ranks). The statistical significance of differences between the classes was assessed by pair-wise comparison test (Dunn's test). Kernel density surfaces were generated from lake centroids in each classification to illustrate spatial patterns.

4.3.5 Analysis of non-stationary and stationary signals

Pearson correlation tests were performed to evaluate relationships between rates of change in ln Chl-a_{mod} (µg L^{-1} yr^{-1}) and rates of change in Tmax (ºC yr^{-1}) and in Pr (mm yr^{-1}) in lakes where significant ($p < 0.05$) trends in ln Chl-a_{mod} (µg L^{-1}) were found from 1984-2011. The tests were performed separately on lakes with positive and negative trends in ln Chl-a_{mod}. Pearson correlation tests were also performed to identify relationship between stationary signals and the global climate oscillations. In this study, the Multivariate El Niño Southern Oscillation Index (MEI), Atlantic Multidecadal

Oscillation (AMO), Northern Atlantic Oscillation (NAO) and Pacific Decadal Oscillation (PDO) indices were selected due to their pronounced effect on global climate and lake ecosystems (Blenckner *et al.*, 2007). Global climate oscillation indices data were obtained from the National Center for Atmospheric Research (open resource: www.cgd.ucar.edu). Pearson correlation tests were performed on each stationary signal; the correlation was considered significant at $p < 0.1$.

4.3.6 Analysis of environmental controls of transitional lake classes

Classification tree and random forest analyses were performed to investigate if environmental controls (climate and landscape) were related to eutrophying and oligotrophying lakes. The random forest analysis was performed using 1,000 trees (Breiman, 2001) and the node purity ("*IncNodePurity*") was used to determine the importance of predictor variables (De'ath, 2002); the variable having the lowest absolute value of *IncNodePurity* was considered "unimportant" (Strobl *et al.*, 2009). The rates of change in temperature (T_{max} yr^{-1}) and precipitation (P yr^{-1}), and five landscape properties – V, DR, W%, LZ and LAT – were used as predictor variables, while the lake stability class (eutrophying and oligotrophying) was used as the response variable in the classification tree and random forests analyses.

4.4 Results

4.4.1 Non-stationary signals in Chlorophyll-*a* time series

Mann-Kendall non-parametric trend tests revealed significant ($p < 0.05$) trends ln Chl-a_{mod} from 1984 to 2011 in 1,061 lakes (500 lakes had positive trends and 561 lakes had negative trends; Appendix F: Figure F.1).

The correlation analysis between ln Chl-a yr^{-1} and T_{max} yr^{-1} did not reveal any significant relationship ($r < 0.01$, $p = 0.87$), while there was a significant but weak correlation between ln Chl-a yr^{-1} and P yr^{-1} ($r = 0.15$, $p < 0.0001$). However, when I performed Pearson correlation on positive trending and negative trending lakes separately, I found that ln Chl-a yr^{-1} of positive trending lakes demonstrated significant correlation with both

T_{max} yr^{-1} and P yr^{-1} (Figure 4.4). Ln Chl-a yr^{-1} increased with increasing T_{max} yr^{-1} ($r = 0.25$, $p < 0.0001$) and decreasing P yr^{-1} ($r = -0.15$, $p = 0.0007$). I also found that ln Chl-a yr^{-1} of negative trending lakes did not have significant correlation with T_{max} yr^{-1} ($r = -0.03$, $p = 0.46$), while there was a significant positive correlation between ln Chl-a yr^{-1} and P yr^{-1} ($r = 0.19$, $p < 0.0001$).

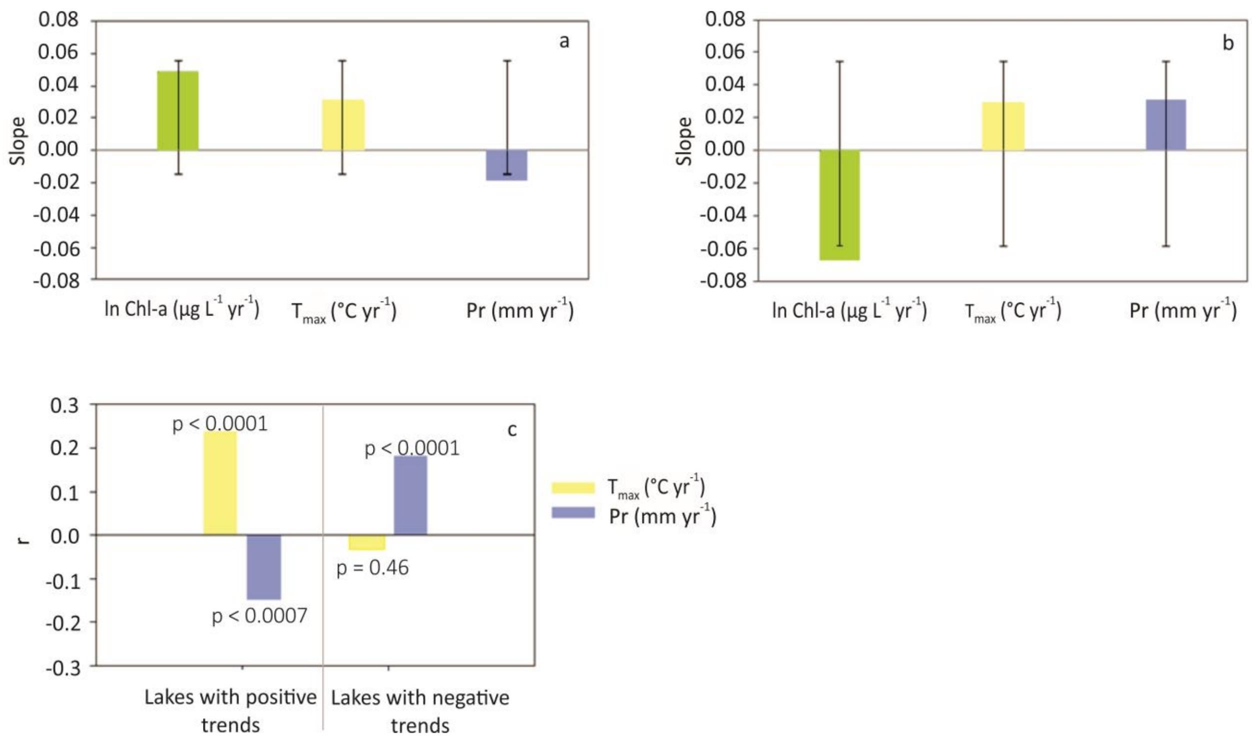

Figure 4.4 Rates of change (slopes) of Chl-a_{mod} (ln Chl-a yr^{-1}), max air temperature (T_{max} yr^{-1}), precipitation (Pr yr^{-1}) for (a) positive trending lakes ($n = 500$) and (b) negative trending lakes ($n = 561$); and (c) Pearson correlation coefficients (r) between ln Chl-a yr^{-1} for positive and negative trending lakes and climate forces (i.e., rates of change in max air temperature – T_{max} yr^{-1} and precipitation – P yr^{-1}). Whiskers depict standard deviation.

4.4.2 Stationary signals in chlorophyll-*a* time series

There were multiple stationary signals in ln Chl-a_{mod} time series. By analyzing the results of sensitivity analysis (Figure 4.5) and time series of median residuals of ln Chl-a_{mod} for 12,644 lakes (Figure 4.6) I determined that removal of six signals was appropriate to eliminate most detectable stationary patterns from the time series. The correlation between residuals and global climate oscillation indices revealed that the first and the second stationary signals had significant correlations ($p < 0.1$) with the AMO index (Table 4.1).

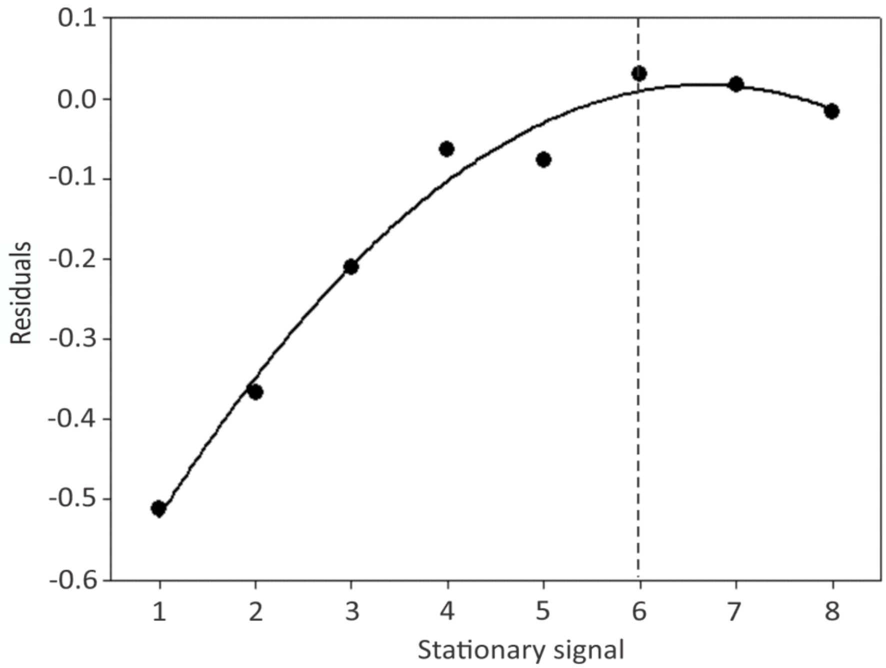

Figure 4.5 Sensitivity analysis to identify a threshold when removal of stationary signals should be stopped. Vertical dashed line indicates the threshold (the sixth stationary signal).

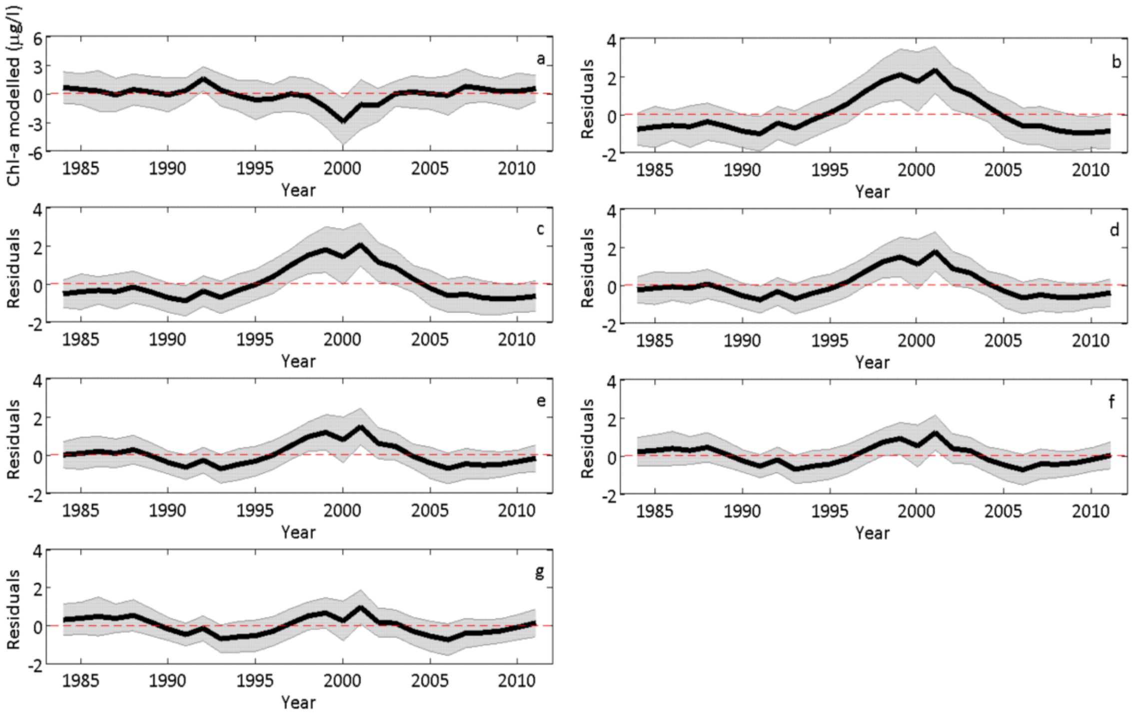

Figure 4.6 (a) Original (de-trended) ln Chl-a$_{mod}$ time series (median over 12,644 lakes), and (b–g) stationary signals identified in the time series. Six stationary signals are depicted for all 12,644 lakes (median ln Chl-a$_{mod}$ residuals) in accordance with letter from b (first signal) to g (sixth signal).

Table 4.1 Correlation between identified stationary signals and climatic indices. Values in bold show significant correlation at $p < 0.1$.

Stationary signal	Climate index	r	p value
1	MEI	-0.04	0.83
	NAO	-0.06	0.74
	PDO	-0.05	0.79
	AMO	0.36	**0.04**
2	MEI	-0.06	0.75
	NAO	-0.05	0.78
	PDO	-0.06	0.76
	AMO	0.33	**0.08**
3	MEI	-0.09	0.61
	NAO	-0.03	0.85
	PDO	-0.07	0.72
	AMO	0.27	0.16
4	MEI	-0.14	0.47
	NAO	-0.01	0.94
	PDO	-0.07	0.75
	AMO	0.17	0.37
5	MEI	-0.19	0.57
	NAO	-0.00	0.96
	PDO	-0.08	0.89
	AMO	0.19	0.39
6	MEI	-0.21	0.58
	NAO	-0.00	0.96
	PDO	-0.08	0.90
	AMO	0.22	0.40

4.4.3 Lake trophic stability classes

Time series of SDs of residuals within 5-year moving windows (SD_5) were observed to be most significantly correlated in 95% of the randomly selected lakes (Appendix G Figure G.1); therefore, time series of 24 SD_5 were calculated in all lakes. Stable, unstable and transitional (eutrophying and oligotrophying) lake stability classes were identified by analyzing the time series of SD_5. I identified 5,344 lakes (42.3%) as stable oligotrophic (low SD_5 and low Chl-a in concentration in ln Chl-a_{mod} across time), 146 lakes (1.2%) as stable eutrophic (low SD_5 and high Chl-a concentration in ln Chl-a_{mod}), 1,586 lakes (12.5%) as unstable (high SD_5 across time), 2,605 lakes (20.6%) as eutrophying (SD_5 increasing over time), and 2,963 lakes (23.4%) as oligotrophying (SD_5 decreasing over time). Significant differences ($p < 0.05$) in the mean SD_5 between all stability classes were observed.

I assessed the relation of lake stability classes to lake trophic condition (i.e., trophic state: oligotrophic, mesotrophic, eutrophic/hyper-eutrophic). For this, Chl-a_{mod} of individual lakes in each lake stability class was averaged across 5-year intervals from 1985 to 2009 and plotted as a box-plot (Figure 4.7). Both stable oligotrophic and stable eutrophic lakes did not show any clear change across 5-year intervals from 1985 to 2009 (i.e., they were stable). However, eutrophying lakes were shifting from oligotrophic to mesotrophic and eutrophic states, while oligotrophying lakes were shifting from eutrophic to mesotrophic and oligotrophic states. Unstable lakes were largely mesotrophic throughout the 1985-2009 period; however, these lakes still showed "short-term" (within several years) patterns where the lakes "switched" from oligotrophic to eutrophic and back to oligotrophic states.

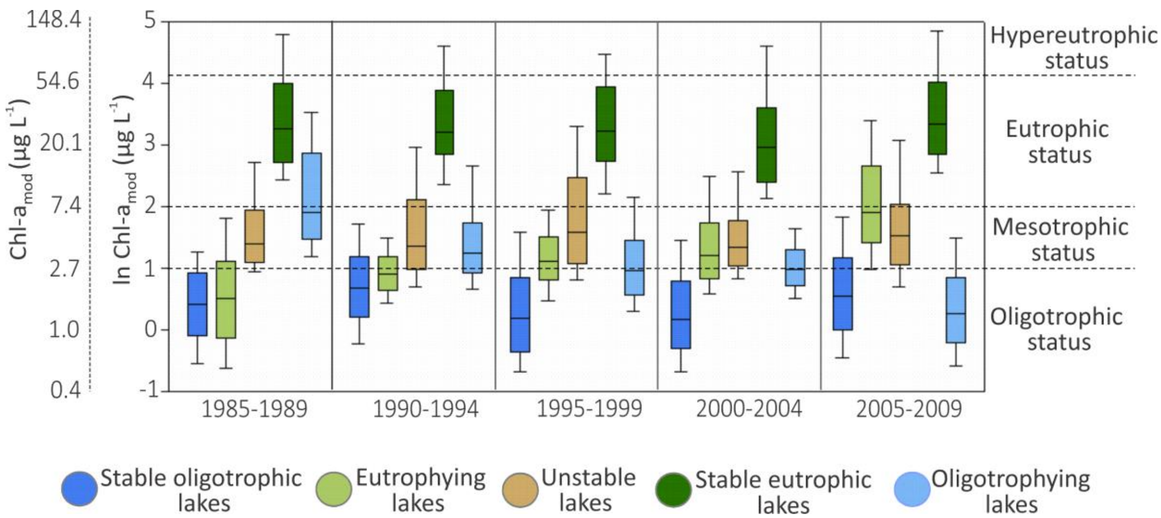

Figure 4.7 Temporal distribution of lake stability classes (based on time series of SD_5 of residuals from ln Chl-a_{mod}) in relation to trophic states (based on ln Chl-a_{mod} concentration).

I found no discernible pattern in the spatial distribution of lakes of stable oligotrophic class across the study region (Figure 4.8a), but I did observe clusters in the distribution of lakes in the other stability classes (Figures 4.8b, c, d, e).

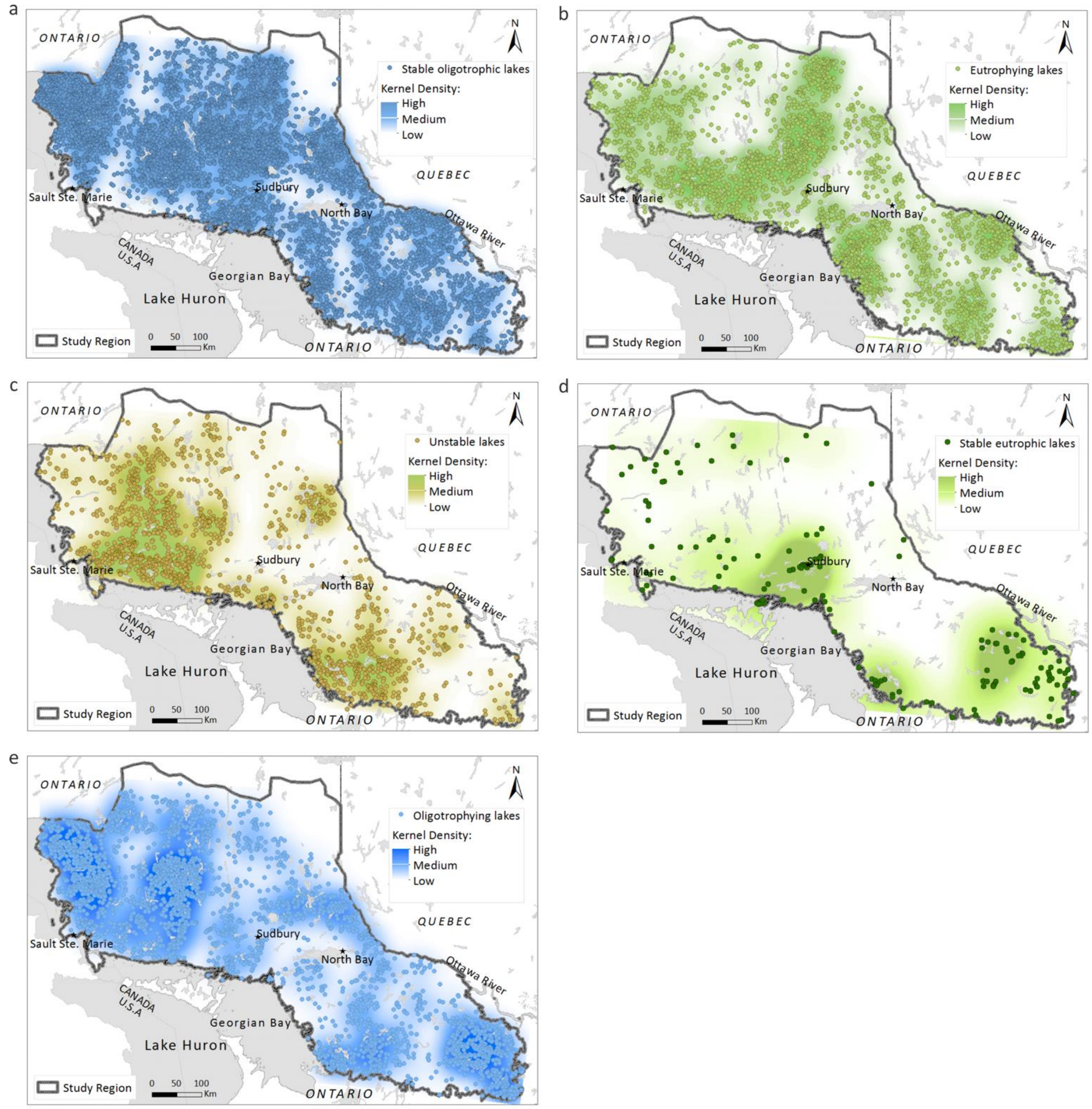

Figure 4.8 Spatial distribution and Kernel density of lakes for various lake stability classes (based on time series of SD_5 of residuals from ln Chl-a_{mod}): (a) stable oligotrophic (n = 5,344), (b) eutrophying (n = 2,605), (c) unstable (n = 1,586), (d) stable eutrophic (n = 146), and (e) oligotrophying (n = 2,963) classes. In Kernel density, the default search radius was based on the number of lakes.

4.4.4 Environmental controls of transitional lakes

Pearson correlation tests indicated that the selected landscape variables did not significantly correlate with each other (i.e., there was no co-linearity found between variables; see Chapter 3). I related two climate factors (i.e., rates of change in temperature and precipitation: T_{max} yr^{-1} and P yr^{-1}) and five landscape factors (i.e., V, DR, W%, LZ and LAT) to the transitional lakes in the classification tree and random forest models to determine if these variables are associated with eutrophying or oligotrophying lakes. Six lake groups were identified in the classification tree (Figure 4.9a). The rate of change in precipitation (P yr^{-1}) was the most significant predictor of transitional lakes, followed by lake volume (V). In the larger-volume lakes, W% was the next most important variable, whereas in smaller-volume lakes, DR and LZ were the next most important variables. T_{max} yr^{-1} and LAT did not appear in the classification tree and were the least significant variables in the random forests (Figure 4.9b). Overall, the percentage of oligotrophying lakes increased with more precipitation (i.e., a low rate of precipitation decrease or increasing precipitation) and larger W% and LZ but with smaller V and DR. On the other hand, the percentage of eutrophying lakes increased with less precipitation (i.e., a high rate of precipitation decrease) and smaller W% and LZ but with larger V and DR.

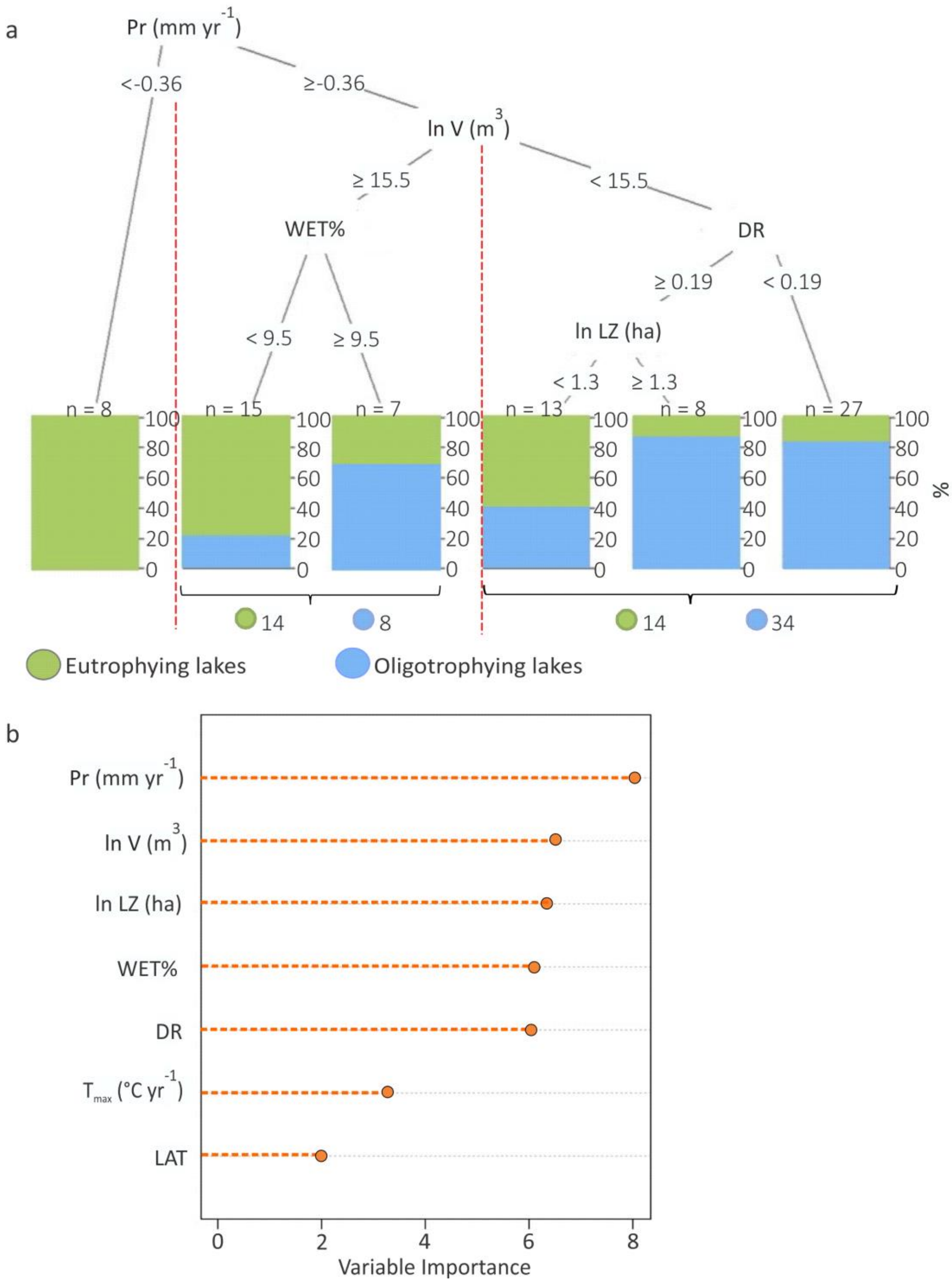

Figure 4.9 (a) Correlation tree, and (b) results of the random forests showing the environmental (climatic and landscape) controls of transitional lakes (n = 78). Climatic controls: rate of change of air max temperature and precipitation (T_{max} yr^{-1} and Pr yr^{-1}), landscape controls: V–volume, DR–dynamic ratio, W%–wetland cover, LZ–littoral zone, LAT–latitude; ln indicates ln-transformed values. Red dashed lines on the tree indicates larger- and smaller-volume sections of the tree (with Pr yr^{-1} ≥ -0.36 mm).

4.5 Discussion

The goal of this study was to investigate whether temperate lakes in a relatively undisturbed forested region undergo any noticeable changes in ecosystem stability in response to climate change. In an earlier study (Chapter 2), I observed that only 8.4% of lakes in the study region had a significant trend in the ln Chl-a_{mod} time series, of which 4.0% were eutrophying and 4.4% were oligotrophying. Climate modification through landscape filters that regulated water residence time and nutrient sources, transport and fates were important determinants of whether lakes showed eutrophying or oligotrophying trends (Chapter 3). However, when examining more subtle signals of ecosystem stability such as the standard deviation (variance) in the residuals of the time series once non-stationary and stationary signals were removed, I observed that 56.5% of my study lakes showed ecosystem instability, with 12.5% of the lakes unstable (or switching between oligotrophic and eutrophic), 20.6% eutrophying and 23.4% oligotrophying. Of the remaining lakes, 42.3% were stable oligotrophic and 1.3% were stable eutrophic.

Lake instability

A significant increasing trend in variance over time indicated that 20.6% of lakes in the region are losing their resilience and shifting towards a new stable eutrophic state (eutrophying). Contrary to my expectations and studies showing significant increases in phytoplankton biomass (Winter *et al.*, 2011) and P (as a principal limiting nutrient of phytoplankton growth) (Stoddard *et al.*, 2016, but see Eimers *et al.*, 2009), a larger percentage of lakes (23.4%) were found to show a significant decreasing trend in SD of residuals over time, i.e., shifting towards a new stable oligotrophic state (oligotrophying). Spatial and temporal patterns of eutrophying lakes appear to mirror those of oligotrophying lakes (i.e., spatial clusters of eutrophying lakes appear in areas of low oligotrophying lake density and *vice versa* (Figures 4.8b and e)), and 5-year average annual ln Chl-a_{mod} in eutrophying lakes increases proportionately with decreases in oligotrophying lakes over time (Figure 4.7). The temporal mirroring suggests that both eutrophying and oligotrophying lakes are responding to the same environmental driver(s) but that the different direction of response is likely due to some intrinsic characteristics of lakes in each class. The relatively monotonic nature of trends in variance of eutrophying and oligotrophying lakes implies that these trends caused by long-lasting (over years) environmental drivers but not short-term factors (e.g., extreme weather events). The fact that around 50% of eutrophying and around 75% of oligotrophying lakes crossed the mesotrophic-eutrophic and mesotrophic-oligotrophic boundary implies that regime shifts to a new stable state (either eutrophic or oligotrophic) might have occurred in these lakes, while it may occur in the remaining lakes (50% eutrophying and 35% oligotrophying) in the future.

Correlation tree analysis revealed complex interactions between precipitation (rate of change – Pr yr^{-1}), landscape controls, and the direction of transitional lakes (i.e., eutrophying or oligotrophying). The correlation tree analysis indicated that higher rates of precipitation decrease were related to eutrophying lakes (Figure 4.9). The role of precipitation as a driver of phytoplankton growth is less known than that of temperature (Sinha *et al.*, 2017). While more precipitation mobilizes nutrients on land, potentially leading to increasing nutrient enrichment of receiving waters and thereby promoting eutrophication (Adrian *et al.*, 2013; Paerl & Huisman, 2008), the study region is known to have naturally low soil P (Jeffries & Snyder, 1983) and to have experienced steadily decreasing deposition of P (Eimers *et al.*, 2009) and total N for at least the last 20 years (Mengistu *et al.*, 2013b; Geddes & Martin, 2017). There is evidence that less precipitation can promote eutrophication as well (Cobbaert *et al.*, 2015). Decreasing precipitation can lead to less volume for nutrient dilution in a water column and lower concentration of dissolved oxygen (Whitehead *et al.*, 2009; Xia *et al.*, 2016). In addition, more intense precipitation is likely to lead to higher levels of soil saturation, hindering water infiltration and, therefore, the ability of water to flush nutrients out from the soils in catchments (Knapp *et al.*, 2008).

While the rates of change in Pr were variable, ranging from negative (-0.24 mm yr^{-1}) to positive (0.17 mm yr^{-1}), all areas of the region experienced an increase in T_{max} (mean rate = 0.045°C yr^{-1}). Correlation tree analysis revealed that temperature (rate of change of maximum air temperature–T_{max} yr^{-1}) did not appear to have a significant influence on transitional lakes (Figure 4.9a). The low "variable importance" for T_{max} yr^{-1} revealed by the random forests analysis supports this observation (Figure 4.9b). However, the mean annual air temperature for 1984-2011 in the region was only +5.1°C, while mean annual July-October maximum air temperature for the same period was 17.2°C, suggesting that the temperatures remain below what is optimal for phytoplankton growth.

The presence of landscape factors in the classification tree indicates that surrounding catchment and lake-specific characteristics influence transitional lakes. While V, DR and LZ might affect the fate of the nutrients within lakes (Vadeboncoeur *et al.,* 2002; Hakanson, 2005; Nõges, 2009), wetland cover (W%) might affect the source, storage, and transport of water and nutrients to lakes (Mitsch & Gossilink, 2000; Harms *et al.,* 2016). Contrary to expectations, lakes with smaller volume do not seem to favor eutrophication; in fact, the number of eutrophying lakes is the same in both larger- and smaller-volume sections of the classification tree (n = 14) (Figure 4.9a). I do not have an explanation for this discrepancy, but one of the possible reasons might be the differences in lakes depth. The presence of DR in the smaller-volume section of the classification tree partly supports this suggestion. The great majority of lakes (~81%) with low DR (< 0.19) are oligotrophying. Lakes with low DR are less prone to wind-driven sediment resuspension (Bachmann *et al.,* 2000; Hakanson, 2005). These lakes are generally deep with relatively small fetches; therefore, the sediments are unlikely to get in direct contact with a trophogenic layer or to get disturbed by wind activity. This condition might be favored by increased water fluxes caused by increasing precipitation. In contrast, higher DR indicates higher rates of sediment resuspension that might bring nutrients (especially P) from sediments back into water column (Nõges, 2009) and therefore promote eutrophication.

Interestingly, larger littoral zones areas (LZ) seemed to favor oligotrophying lakes. Besides being indicative of very shallow depths within lakes and lake connections to catchments (Vadeboncoeur *et al.,* 2002; Kornijów *et al.,* 2016), extended littoral zones also increase the probability of development of communities of rooted aquatic plants (macrophytes; Kornijów *et al.,* 2016). These fringing communities are known to act as the buffer, where external material and associated nutrients accumulate and are quickly taken up by the macrophytes and attached algae (Schindler & Scheuerell, 2002). Additionally, littoral zones provide refuges for zooplankton; hence, zooplankton abundance is generally larger in lakes with developed macrophyte P (Kornijów *et al.,* 2016). In this case, due to grazing pressure of zooplankton on phytoplankton, the lakes might be more prone to oligotrophication than eutrophication or have cyclic behavior (i.e., vegetated *versus* phytoplankton states; Van Nes *et al.*, 2007; Scheffer & Van Nes, 2007).

In lakes with larger volumes, wetlands (W%) became an important predictor of transitional lakes. Higher proportions of wetland area may contribute to the maintenance of oligotrophic states in lakes (Cobbaert *et al.,* 2015) due to the ability of wetlands to remove and retain N (primarily in the form of NO_3^-; Verhoeven *et al.*, 2006). However, other studies showed that wetlands in fact may act as a large source of P, dissolved organic nitrogen (DON), and DOM (Mengistu *et al.*, 2014; Harms *et al.*, 2016), and therefore they might contribute to eutrophication. In respect with the study region, in Chapter 3 I found that lakes with higher Chl-a concentration generally had higher wetland cover in the catchments. This seems to contradict my finding that higher W% mostly drives oligotrophying lakes. However, it is important to consider the temporal factor; in the current study I analyzed a long-term change in a system, while in Chapter 3 the static condition of lakes was considered (i.e., median precipitation and median Chl-a). Therefore, a lake that is changing over time (i.e., eutrophying or oligotrophying) might be with low, median or high Chl-a concentration (i.e., be oligotrophic, mesotrophic or eutrophic) at any given period of time. Similarly, wetlands might be a source of nutrients at any given time, but they might also act as sinks for nutrients over a long period of time.

The mechanism behind the “switching behavior” of the lakes with unstable states may belong to “slow-fast cyclic transitions” proposed by Rinaldi & Scheffer (2000) in which, after lakes shift to a new regime, a negative feedback starts to “pull” environmental conditions back until a shift to the previous regime occurs (Dakos *et al.*, 2014). This phenomenon has been described for cyclic shifts between “vegetated” and “barren” states in shallow Dutch lakes caused by the build-up of organic matter and resulting anaerobic conditions at the bottom of the lakes (Van Nes *et al.,* 2007), and between vegetated and phytoplankton states in Canadian Boreal Plain lakes caused by differences in water level (Cobbaert *et al.*, 2015) and harsh winter conditions (Bayley *et al.*, 2007).

Lake stability

The dominance of stable oligotrophic lakes *versus* stable eutrophic lakes (42.3% *versus* 1.3% of all lakes) in the relatively undisturbed temperate forest study region may be explained by naturally low soil P levels (the major limiting nutrient of phytoplankton growth; Smith, 2003) in the Precambrian Canadian Shield underlying the region (*see* Jeffries & Snyder, 1983). Three spatial clusters of stable eutrophic lakes (Figure 4.8d) are present in areas of relatively greater anthropogenic development: two in “cottage country” areas in the south of the study region, and one surrounding the Greater Sudbury urban region where intense mining practices accompanied by land clearing and logging in the 1960s and 1970s may have led to intensified soil erosion and, as a result, increased P leaching (Pearson *et al.*, 2002). Further, during the same time period (1970s to 1980s), many lakes of the Sudbury region were found to have heavy metal content (e.g., Ni, Cu, and Zn) caused by direct runoff from mining sites and atmospheric deposition of metallic dust (Semkin & Kramer, 1976; Pearson *et al.,* 2002). This contamination created toxic conditions for local fishes and zooplankton (Spry & Weiner, 1991; Pearson *et al.*, 2002), possibly leading to serious modification in and even collapse of food chains (Carpenter *et al.*, 2014; Dakos *et al.*, 2014). In the absence of pressure from phytoplankton feeders, regime shifts to the lakes with a phytoplankton-dominated eutrophic state might have occurred. Since the 1990s, some recovery of fish and invertebrate communities in many Sudbury lakes has been observed (Valois *et al.*, 2011; Keller *et al.*, 2018), and more time might be needed for a return in their ecosystems to pre-disturbance state (Carpenter *et al.*, 2014). Only 30% of stable eutrophic lakes were found outside these clusters where regional-scale anthropogenic development is not apparent (although local-scale development may have occurred).

4.6 Conclusions

More than half of the almost 13,000 lakes experienced changes in their ecological stability during the study period. Contrary to expectations, the dominant trend was towards oligotrophication rather than eutrophication. Both transitions occurred in response to an external climate factor (precipitation) and the extent of change was modified by landscape properties. Changes in precipitation patterns seemed to be more important in altering lake stability in lakes. Higher rates of precipitation decrease tended to drive eutrophication in lakes, while smaller rates of precipitation decrease or precipitation increases tended to result in oligotrophying lakes. In the absence of confounding land use, the changes in precipitation patterns anticipated as consequences of climate change can be used to understand regional patterns of eutrophication and oligotrophication in the temperate lakes; however, these patterns will still be largely depended on catchment and lake-specific properties.

4.7 References

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5 Summary

The major contributions of this thesis include: (1) modeling Chl-a concentration (as a proxy of lake phytoplankton biomass) in lakes in the temperate forest biome of central Ontario through space and time; (2) finding spatial and temporal trends in Chl-a and developing an understanding of the role that climatic and landscape factors play in lake Chl-a concentration in the study area; (3) developing a framework for classifying lake trophic stability over time (and for identifying possible signs of regime shifts in lake trophic state); and (5) developing an understanding of the importance of climate in driving lake instability to understand why some lakes change over time while others remain in a stable state (oligotrophic or eutrophic).

5.1 Research findings

The lakes of the study area are located in the relatively intact region of the temperate forest biome within the Boreal (Canadian) Shield in central Ontario. The study region was selected on the assumption that human activities and anthropogenically-driven nutrient discharges in the area are minimal, so that changes in lake Chl-a concentration can be considered as natural responses to climate changes. The study lakes differ in volume, size, depth, and trophic state (from oligotrophic to hyper-eutrophic, based on modeled Chl-a).

Remote sensing was employed to model Chl-a concentration using archived Landsat TM/ETM+ satellite products obtained from August to October (the peak phytoplankton biomass for the temperate regions of North America) from 1984 to 2011. Reflectance values from the archived images and sample lake Chl-a concentration measurements were used to develop a regression model to estimate Chl-a in 12,644 lakes over a 28-year period. A two-way ANOVA showed that the temporal (variation in climate) and spatial (regional landscape controls) components accounted for 26% of the total variation in Chl-a concentration, while the interaction (lake-specific controls) component accounted for the remainder of the variation (74 %). A high density of oligotrophic lakes were found in a belt formed by topographic divides while clusters of eutrophic and hyper-eutrophic lakes were found near the Great Lakes and Sudbury, and at the southernmost parts of the study region. However, correlations between Chl-a concentration in different lake trophic states and topography were weak. Temporal trends in Chl-a concentration were found to correlate with climatic controls, where lakes with increasing Chl-a concentration were more correlated with increasing air temperature ($r = 0.25$, $p < 0.0001$) and lakes with decreasing Chl-a concentration were more correlated with increasing precipitation ($r = 0.19$, $p < 0.0001$).

Air temperature, precipitation, and various landscape properties (lake volume, dynamic ratio, wetland cover in catchment, littoral zone area, and latitude) were used in the regression tree model to explain median Chl-a concentration in a subset of 275 lakes. Lakes with the highest Chl-a concentration had smaller volumes ($< 442 \times 10^3$ m^3) and were more sensitive to temperature change. On the other hand, lakes with lower Chl-a concentration had larger volumes ($> 442 \times 10^3$ m^3), were more sensitive to precipitation change, and were mostly oligotrophic (but also mesotrophic). These findings indicate that: (1) lakes with smaller volumes were more responsive to climate change and that this response was more "typical" and more "direct" (higher temperatures = higher Chl-a); and (2) lakes with larger volumes were generally less responsive to climate change, but shallower large-volume lakes (and with larger littoral zones) behaved similarly to lakes with smaller volumes.

Non-stationary and stationary trends were removed from the time series of lake Chl-a concentration, and trends in the standard deviations of residuals within a moving time window were categorized into five classes of lake trophic stability. Two of these classes were characterized as stable (either oligotrophic or eutrophic), one as unstable, and two as transitional (either eutrophying or oligotrophying). The majority of lakes (42.3%) were stable oligotrophic, and the minority (1.2%) were stable eutrophic, while 12.5% of lakes were unstable. There were more lakes experiencing oligotrophication (23.4%) compared to those experiencing eutrophication (20.6%). This indicates that despite the fact that both eutrophication and oligotrophication are occurring simultaneously in the region, eutrophication is still not as ubiquitous as one might think. Additionally, the fact that stable oligotrophic is still a dominating state in the region indicates that many lakes exhibit a large degree of resilience to the environmental changes observed in the northern temperate ecosystems. Classification tree models and random forests analysis showed that both transitional lake classes are driven by changes in precipitation patterns (high rate of decrease in precipitation or an increase in precipitation) but not by temperature increase. Precipitation is likely to drive changes in lake stability via changes in nutrient loading patterns that are manifested through catchment (i.e., wetland coverage) and lake morphometric characteristics.

5.2 Research significance

This thesis provides a valuable contribution to understanding of the controls of Chl-a concentration in temperate lakes. To my knowledge, this study is the first meta-analysis comparing the importance of climate, catchment and lake controls on phytoplankton biomass and lake trophic stability.

The thesis provides an insight into how phytoplankton from lakes with different morphometry responds to changing climate (Chapter 3). The finding that lakes with different volumes respond differently to precipitation controls may help inform development of different methods of lake protection under changing climate scenarios (Steffen *et al.*, 2018).

This thesis provides a framework for identifying the trophic stability of lakes over time (Chapter 4). Variance of residuals has been long used as an indicator of ecosystem stability and regime shifts (Dakos *et al.*, 2014). However, to my knowledge, this is the first time when variance of residuals has been used to identify stability classes in accordance with the conceptual model developed by Scheffer *et al.* (2001), where some lakes exhibit change over time while others remain in a stable state. This framework can be applied as a template for assessing the trophic stability of lakes located in different regions and under different climatic or environmental conditions.

5.3 Future research needs

Understanding the processes regulating phytoplankton biomass have proven to be challenging; complicated not only by changing patterns of temperature and precipitation but also by variability in catchment characteristics and lake morphometry acting on different spatial and temporal scales. This thesis examined the processes and patterns that help to explain the heterogeneity in phytoplankton biomass (estimated as Chl-a concentration). However, there are two main areas that need to be addressed in further research, which include:

1. Explaining trends in limiting to algal growth nutrients, as they may affect changes in lake Chl-a concentration. Sufficient long-term measurements of lake P, N in areas within (or near) the study region may be particularly useful. For example, Eimers *et al.* (2009) had temporally extensive (around 20 years) measurements of P in eleven catchments draining into three lakes in Boreal Shield (and therefore encompassing the study region). In this study, however, I did not intend to compare long-term trends in nutrients with trends in Chl-a; my intention was to explain general relation between climate, landscape factors and Chl-a, and offer possible explanation on how these factors might affect nutrient loading into lakes and hence Chl-a concentration within these lakes;
2. Estimating the effect of brownification in the study lakes. This phenomenon was found to be caused by increasing runoff of terrestrially derived DOM to receiving lakes, which can result in reduced primary productivity and nutritionally poorer lake food webs (Creed *et al.*, 2018). Brownification might potentially explain trends with decreasing Chl-a found in some study lakes. Lake instability might also be partly influenced by this phenomenon under the condition of gradually increasing supply of DOM to the study lakes.

5.4 References

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Appendices

Appendix A: Description of lake samples used for the development of the regression model in Chapter 2.

Table A.1 Description of lake samples (sample date, lake morphometry and chemistry)

Lake ID	Sample Name/ID	Location	Longitude	Latitude	Sample date	Lake mean depth (m)	Lake maximum depth (m)	Lake area (ha)	Chl-a (µg L^{-1})	DOC (mg L^{-1})	TP (µg L^{-1})	Secchi depth (m)	Turbidity (NTU)	Selected for final model (Yes/No)
10on	Sill	Ontario	-84.25	46.77	June 20, 2010	-	7.4	41.7	0.9	-	10.8	7.4	-	Yes
12on	Reception1	Ontario	-83.25	46.48	June 26, 2009	-	2.8	88.7	10.0	-	14.7	1.3	-	Yes
13on	Rock	Ontario	-83.77	46.43	June 24, 2009	-	1.9	1033.2	3.7	-	13.4	1.8	-	Yes
14on	Cloudy	Ontario	-83.93	46.44	June 24, 2009	-	7.4	248.8	0.5	-	9.4	4.8	-	Yes
16on	Constance1	Ontario	-83.23	46.43	June 26, 2009	-	7.8	120.1	1.1	-	6.9	4.7	-	Yes
17on	Appleby1	Ontario	-83.35	46.43	June 26, 2009	-	5.1	24.3	4.6	-	10.8	2.3	-	Yes
19on	Woodrow2	Ontario	-83.33	46.41	August 24, 2010	-	2.0	48.8	0.6	7.1	3.8	2.0	-	Yes
1on	Negick2	Ontario	-84.49	47.21	June 16, 2010	-	5.3	26.6	2.3	2.6	12.8	3.5	-	Yes
20on	Round	Ontario	-83.83	46.39	June 24, 2009	-	3.2	128.4	3.3	-	19.5	2.7	-	Yes
22on	Eaket1	Ontario	-83.25	46.35	June 26, 2009	-	4.5	56.7	2.8	-	9.2	2.9	-	Yes
23on	Twin	Ontario	-83.93	46.23	July 27, 2011	-	3.8	30.0	7.4	-	10.3	1.7	-	Yes
24on	Dean2	Ontario	-83.18	46.23	July 25, 2011	-	14.9	219.5	3.2	-	-	6.0	-	Yes
25on	Caysee2	Ontario	-84.66	47.18	June 16, 2010	-	1.3	16.5	2.5	8.3	23.2	1.3	-	Yes
26on	Carp	Ontario	-84.56	46.97	June 16, 2010	-	1.5	112.1	3.7	5.6	17.2	1..5	-	Yes
2on	Upper Griffin	Ontario	-84.40	47.09	June 16, 2010	-	7.8	155.3	0.7	3.8	7.2	7.8	-	Yes
5on	Big Turkey	Ontario	-84.42	47.05	May 16, 2010	-	42.7	51.8	1.4	3.8	5.0	5.6	-	Yes
7on	Little Turkey	Ontario	-84.41	47.04	May 16, 2010	-	7.3	18.9	0.5	16.4	3.2	7.3	-	Yes
8on	Upper Tilley2	Ontario	-84.39	47.02	May 15, 2010	-	6.1	163.1	2.1	4.8	9.2	2.9	-	Yes
101ab	5992	Alberta	-115.38	56.07	August 14, 2002	1.6	-	19.7	28.6	60.3	100.8	0.3	-	Yes
102ab	88	Alberta	-115.50	56.04	August 15, 1999	1.1	-	274.7	3.7	-	30.2	0.7	-	Yes
108ab	75	Alberta	-114.85	55.96	August 12, 2001	0.9	-	31.4	34.2	60.2	118.6	0.5	7.3	Yes
16ab	101	Alberta	-114.75	56.31	August 13, 2001	1.8	-	39.2	2.0	38.6	17.9	1.8	0.4	Yes
24ab	12	Alberta	-115.88	56.10	August 11, 2001	1.3	-	4.6	15.9	27.9	58.2	1.3	0.9	Yes
28ab	57	Alberta	-115.39	56.08	August 15, 1999	0.6	-	9.8	8.7	-	119.3	0.0	-	Yes
2ab	42	Alberta	-115.16	56.30	August 11, 2001	1.1	-	7.4	40.8	40.9	117.0	0.6	3.3	Yes
37ab	1681	Alberta	-115.20	55.99	August 15, 2001	0.7	-	11.2	6.4	58.3	102.4	0.7	1.2	Yes

Lake ID	Sample Name/ID	Location	Longitude	Latitude	Sample date	Lake mean depth (m)	Lake maximum depth (m)	Lake area (ha)	Chl-a (μg L^{-1})	DOC (mg L^{-1})	TP (μg L^{-1})	Secchi depth (m)	Turbidity (NTU)	Selected for final model (Yes/No)
37ab	1682	Alberta	-115.20	55.99	August 12, 2002	0.7	-	10.6	3.8	74.9	120.8	0.7	-	Yes
38ab	171	Alberta	-115.19	55.98	August 15, 1999	0.6	-	8.5	47.1	-	421.7	0.6	-	Yes
45ab	131	Alberta	-115.60	55.96	August 15, 1999	-	-	27.1	18.5	-	135.8	0.3	-	Yes
46ab	165	Alberta	-115.26	55.96	August 19, 1999	-	-	8.5	63.4	-	178.6	0.5	-	Yes
53ab	201	Alberta	-115.71	56.12	August 13, 2001	1.2	-	35.1	30.3	23.2	46.3	0.6	10.6	Yes
53ab	2012	Alberta	-115.71	56.12	August 13, 2002	1.2	-	34.6	13.0	27.1	58.5	0.8	20.0	Yes
53ab	2011	Alberta	-115.71	56.12	August 15, 1999	1.2	-	34.4	20.2	-	64.9	0.5	-	Yes
55ab	81	Alberta	-115.56	56.03	August 15, 1999	0.8	-	19.9	9.2	-	54.4	0.5	-	Yes
56ab	89	Alberta	-115.51	56.02	August 15, 1999	-	-	311.9	3.5	-	66.7	0.4	-	Yes
58ab	111	Alberta	-115.43	56.03	August 14, 2001	0.6	-	5.0	2.8	48.8	39.2	0.6	1.3	Yes
5ab	7	Alberta	-115.63	56.29	August 11, 2001	0.8	-	15.6	4.4	56.7	43.5	0.8	1.3	Yes
67ab	127	Alberta	-115.18	56.01	August 19, 1999	-	-	201.8	57.2	-	212.4	0.8	-	Yes
68ab	61	Alberta	-113.91	55.92	August 12, 2001	2.0	-	20.4	2.0	22.1	68.0	1.3	1.1	Yes
70ab	1211	Alberta	-115.35	56.01	August 15, 2001	0.7	-	6.8	3.5	50.3	58.8	0.7	0.7	Yes
70ab	1212	Alberta	-115.35	56.01	August 13, 2002	0.7	-	6.1	12.1	58.5	105.9	0.5	19.3	Yes
70ab	121	Alberta	-115.35	56.01	August 15, 1999	0.7	-	6.4	46.0	-	150.8	0.7	-	Yes
75ab	87	Alberta	-115.12	55.73	August 12, 2001	0.5	-	9.1	7.4	79.6	57.2	0.5	1.2	Yes
78ab	27	Alberta	-115.52	56.07	August 11, 2001	0.6	-	4.5	12.4	25.9	48.4	0.6	1.8	Yes
7ab	4	Alberta	-115.68	56.42	August 11, 2001	0.6	-	6.4	2.8	59.9	233.9	0.6	0.6	Yes
80ab	55	Alberta	-114.16	56.32	August 12, 2001	1.1	-	7.4	61.5	60.7	246.4	0.4	19.0	Yes
92ab	1223	Alberta	-115.35	56.01	August 12, 2002	0.7	-	5.9	31.4	68.9	123.0	0.3	-	Yes
92ab	122	Alberta	-115.35	56.01	August 15, 1999	0.7	-	6.9	58.0	-	77.7	0.6	-	Yes
98ab	16	Alberta	-115.55	56.11	August 13, 2002	0.9	-	36.7	12.0	22.5	68.5	0.9	-	Yes
58ab	1111	Alberta	-115.43	56.03	August 19, 1999	0.6	-	5.2	2.7	-	32.8	0.8	-	No/outlier
62ab	33	Alberta	-115.58	56.17	August 11, 2001	2.1	-	89.7	2.9	43.0	17.8	1.7	0.8	No/outlier
92ab	1222	Alberta	-115.35	56.01	August 15, 2001	0.7	-	7.0	78.8	50.0	126.4	0.6	4.6	No/outlier
9ab	47	Alberta	-114.85	56.49	August 12, 2001	1.8	-	9.4	2.0	41.3	23.7	1.7	0.4	No/outlier
11on	Echo Lake	Ontario	-83.98	46.56	June 20, 2010	-	5.3	1124.1	2.0	4.9	11.6	1.1	-	No
12on	Reception2	Ontario	-83.25	46.48	August 24, 2010	-	2.8	85.7	27.7	12.8	32.0	0.5	-	No
15on	Gordon	Ontario	-83.83	46.42	June 24, 2009	-	1.6	605.1	2.5	-	11.5	1.5	-	No
16on	Constance2	Ontario	-83.23	46.43	August 31, 2009	-	7.8	115.2	1.2	3.9	9.3	6.0	-	No
17on	Appleby2	Ontario	-83.35	46.43	August 31, 2009	-	5.1	21.7	8.6	9.4	18.1	1.8	-	No
18on	Desbarats	Ontario	-83.93	46.39	June 24, 2009	-	6.9	396.6	3.1	-	30.2	0.6	-	No
19on	Woodrow1	Ontario	-83.33	46.41	June 26, 2009	-	2.0	51.8	0.5	-	10.3	2.0	-	No
1on	Negick1	Ontario	-84.49	47.21	September 2, 2009	-	5.3	31.1	3.0	4.9	11.4	2.6	-	No
21on	Ottertail	Ontario	-83.75	46.38	May 17, 2009	-	2.5	424.0	4.7	7.9	23.7	0.3	-	No
22on	Eaket2	Ontario	-83.25	46.35	August 24, 2010	-	4.5	52.3	3.2	2.1	8.6	3.6	-	No
24on	Dean1	Ontario	-83.18	46.23	August 24, 2010	-	14.9	222.8	7.6	7.1	23.2	1.8	-	No

Lake ID	Sample Name/ID	Location	Longitude	Latitude	Sample date	Lake mean depth (m)	Lake maximum depth (m)	Lake area (ha)	Chl-a (µg L^{-1})	DOC (mg L^{-1})	TP (µg L^{-1})	Secchi depth (m)	Turbidity (NTU)	Selected for final model (Yes/No)
25on	Caysee1	Ontario	-84.66	47.18	September 2, 2009	-	1.6	16.5	4.3	8.7	16.3	1.6	-	No
3on	Lower Griffin	Ontario	-84.42	47.08	May 16, 2010	-	7.5	24.8	0.4	4.4	2.0	7.5	-	No
4on	Upper Batchawana	Ontario	-84.39	47.07	May 16, 2010	-	7.5	5.2	0.5	4.0	14.2	6.8	-	No
6on	Wishart	Ontario	-84.40	47.05	May 16, 2010	-	3.0	17.5	1.0	3.7	14.4	1.0	-	No
8on	Upper Tilley1	Ontario	-84.39	47.02	September 2, 2009	-	6.1	163.1	4.0	5.3	8.0	2.7	-	No
9on	Lower Tilley	Ontario	-84.39	47.00	May 15, 2010	-	1.6	143.6	2.5	3.5	18.4	1.6	-	No
100ab	62	Alberta	-115.28	56.07	August 19, 1999	-	-	13.6	10.1	-	72.9	0.6	-	No
101ab	59	Alberta	-115.38	56.07	August 14, 2001	1.6	-	22.1	86.1	43.6	264.2	0.5	4.4	No
101ab	5991	Alberta	-115.38	56.07	August 19, 1999	1.6	-	22.6	10.6	-	57.6	0.9	-	No
103ab	79	Alberta	-114.93	56.05	August 12, 2001	1.6	-	94.9	32.4	62.3	32.6	1.4	0.8	No
107ab	2051	Alberta	-115.16	55.96	August 12, 2002	-	-	13.0	6.0	77.0	61.8	-	-	No
107ab	205	Alberta	-115.16	55.96	August 19, 1999	-	-	17.4	41.9	-	134.3	0.7	-	No
109ab	17	Alberta	-116.04	55.84	August 11, 2001	1.5	-	5.4	18.5	34.6	354.9	0.9	1.2	No
12ab	5	Alberta	-115.56	56.33	August 11, 2001	0.7	-	21.1	1.9	51.8	175.7	0.7	0.9	No
14ab	52	Alberta	-114.32	56.40	August 12, 2001	1.1	-	7.1	28.2	23.7	98.1	0.6	1.8	No
17ab	53	Alberta	-114.32	56.38	August 12, 2001	0.9	-	4.5	22.9	24.8	66.5	0.6	0.7	No
19ab	29	Alberta	-115.66	55.75	August 11, 2001	1.0	-	6.2	20.7	23.5	65.7	0.7	1.3	No
33ab	31	Alberta	-115.50	56.07	August 13, 2002	-	-	6.7	7.2	71.2	68.3	0.3	-	No
34ab	80	Alberta	-115.04	56.07	August 12, 2001	0.9	-	11.3	9.1	56.4	46.9	0.9	0.6	No
37ab	168	Alberta	-115.20	55.99	August 15, 1999	0.7	-	11.1	31.0	-	248.3	1.0	-	No
38ab	1711	Alberta	-115.19	55.98	August 15, 2001	0.6	-	8.2	23.7	49.8	175.3	0.4	2.1	No
38ab	1712	Alberta	-115.19	55.98	August 12, 2002	0.6	-	7.6	3.4	65.8	79.3	0.4	-	No
43ab	95	Alberta	-114.56	56.14	August 13, 2001	1.6	-	9.2	5.9	68.3	22.3	1.4	0.9	No
47ab	599	Alberta	-113.84	56.04	August 12, 2001	1.1	-	16.5	55.2	35.4	281.3	0.7	2.0	No
52ab	39	Alberta	-115.17	56.12	August 11, 2001	2.0	-	93.2	12.8	20.0	38.3	1.6	1.7	No
54ab	71	Alberta	-113.96	55.95	August 12, 2001	1.3	-	24.2	136.4	37.9	264.8	0.4	7.9	No
58ab	1112	Alberta	-115.43	56.03	August 14, 2002	0.6	-	5.0	32.6	54.2	129.7	0.5	-	No
60ab	38	Alberta	-115.18	56.17	August 11, 2001	3.2	-	8.9	13.6	19.9	41.1	2.0	1.0	No
63ab	34	Alberta	-115.49	56.17	August 11, 2001	1.4	-	59.6	3.9	23.3	19.6	1.1	1.1	No
65ab	777	Alberta	-115.55	56.10	August 13, 2002	1.3	-	92.6	3.8	21.8	22.9	-	-	No
66ab	19	Alberta	-115.92	55.80	August 11, 2001	1.6	-	28.8	5.8	30.9	70.5	1.1	0.4	No
69ab	8	Alberta	-115.79	56.17	August 11, 2001	2.9	-	6.4	5.2	22.7	28.4	2.0	1.1	No
6ab	54	Alberta	-114.23	56.36	August 12, 2001	1.5	-	12.9	49.5	48.3	47.4	0.6	0.9	No
73ab	888	Alberta	-115.21	55.78	August 12, 2001	0.5	-	6.5	59.6	-	152.1	0.4	8.1	No

Lake ID	Sample Name/ID	Location	Longitude	Latitude	Sample date	Lake mean depth (m)	Lake maximum depth (m)	Lake area (ha)	Chl-a (µg L^{-1})	DOC (mg L^{-1})	TP (µg L^{-1})	Secchi depth (m)	Turbidity (NTU)	Selected for final model (Yes/No)
76ab	2	Alberta	-115.60	56.10	August 15, 1999	0.9	-	220.8	13.5	-	102.0	0.6	-	No
77ab	23	Alberta	-115.77	55.66	August 11, 2001	1.6	-	11.5	20.1	29.3	63.4	1.0	1.6	No
81ab	102	Alberta	-114.77	56.31	August 13, 2001	1.4	-	9.4	4.8	26.9	38.2	1.4	0.9	No
82ab	18	Alberta	-116.00	55.82	August 11, 2001	3.0	-	4.7	30.2	39.0	128.3	1.0	0.7	No
86ab	21	Alberta	-115.93	55.75	August 11, 2001	1.7	-	28.6	15.9	25.9	46.1	1.5	0.8	No
97ab	577	Alberta	-113.72	56.13	August 12, 2001	0.6	-	10.0	5.6	34.2	58.7	0.5	0.7	No

Table A.2 Correspondence of 53 lake samples with dates of Landsat image capture.

Lake ID	Sample Name/ID	Location	Longitude	Latitude	Sample date	Satellite overpass date	Difference in days
78ab	27	Alberta	-115.52	56.07	August 11, 2001	August 13, 2001	2
24ab	12	Alberta	-115.88	56.10	August 11, 2001	August 13, 2001	2
58ab	1111	Alberta	-115.43	56.03	August 19, 1999	August 15, 1999	4
58ab	111	Alberta	-115.43	56.03	August 14, 2001	August 13, 2001	1
7ab	4	Alberta	-115.68	56.42	August 11, 2001	August 13, 2001	2
70ab	1211	Alberta	-115.35	56.01	August 15, 2001	August 13, 2001	2
70ab	1212	Alberta	-115.35	56.01	August 13, 2002	August 8, 2002	5
70ab	121	Alberta	-115.35	56.01	August 15, 1999	August 15, 1999	0
92ab	1223	Alberta	-115.35	56.01	August 12, 2002	August 8, 2002	4
92ab	122	Alberta	-115.35	56.01	August 15, 1999	August 15, 1999	0
92ab	1222	Alberta	-115.35	56.01	August 15, 2001	August 13, 2001	2
2ab	42	Alberta	-115.16	56.30	August 11, 2001	August 13, 2001	2
80ab	55	Alberta	-114.16	56.32	August 12, 2001	August 13, 2001	1
38ab	171	Alberta	-115.19	55.98	August 15, 1999	August 15, 1999	0
46ab	165	Alberta	-115.26	55.96	August 19, 1999	August 15, 1999	4
75ab	87	Alberta	-115.12	55.73	August 12, 2001	August 13, 2001	1
9ab	47	Alberta	-114.85	56.49	August 12, 2001	August 13, 2001	1
28ab	57	Alberta	-115.39	56.08	August 15, 1999	August 15, 1999	0
37ab	1682	Alberta	-115.20	55.99	August 12, 2002	August 8, 2002	4
37ab	1681	Alberta	-115.20	55.99	August 15, 2001	August 13, 2001	2
5ab	7	Alberta	-115.63	56.29	August 11, 2001	August 13, 2001	2
25on	Caysee2	Ontario	-84.66	47.18	June 16, 2010	June 17, 2010	1
7on	Little Turkey	Ontario	-84.41	47.04	May 16, 2010	16--May-2010	0
101ab	5992	Alberta	-115.38	56.07	August 14, 2002	August 8, 2002	6
55ab	81	Alberta	-115.56	56.03	August 15, 1999	August 15, 1999	0
68ab	61	Alberta	-113.91	55.92	August 12, 2001	August 13, 2001	1
17on	Appleby1	Ontario	-83.35	46.43	June 26, 2009	June 23, 2009	3
1on	Negick2	Ontario	-84.49	47.21	June 16, 2010	June 17, 2010	1
45ab	131	Alberta	-115.60	55.96	August 15, 1999	August 15, 1999	0
23on	Twin	Ontario	-83.93	46.23	July 27, 2011	July 31, 2011	4
108ab	75	Alberta	-114.85	55.96	August 12, 2001	August 13, 2001	1
53ab	2012	Alberta	-115.71	56.12	August 13, 2002	August 8, 2002	5
53ab	2011	Alberta	-115.71	56.12	August 15, 1999	August 15, 1999	0
53ab	201	Alberta	-115.71	56.12	August 13, 2001	August 13, 2001	0
98ab	16	Alberta	-115.55	56.11	August 13, 2002	August 8, 2002	5
16ab	101	Alberta	-114.75	56.31	August 13, 2001	August 13, 2001	0
10on	Sill	Ontario	-84.25	46.77	June 20, 2010	June 17, 2010	3
19on	Woodrow2	Ontario	-83.33	46.41	August 24, 2010	August 29, 2010	5
5on	Big Turkey	Ontario	-84.42	47.05	May 16, 2010	16--May-2010	0

Lake ID	Sample Name/ID	Location	Longitude	Latitude	Sample date	Satellite overpass date	Difference in days
22on	Eaket1	Ontario	-83.25	46.35	June 26, 2009	June 23, 2009	3
12on	Reception1	Ontario	-83.25	46.48	June 26, 2009	June 23, 2009	3
62ab	33	Alberta	-115.58	56.17	August 11, 2001	August 13, 2001	2
26on	Carp	Ontario	-84.56	46.97	June 16, 2010	June 17, 2010	1
16on	Constance1	Ontario	-83.23	46.43	June 26, 2009	June 23, 2009	3
20on	Round	Ontario	-83.83	46.39	June 24, 2009	June 23, 2009	1
2on	Upper Griffin	Ontario	-84.40	47.09	June 16, 2010	June 17, 2010	1
8on	Upper Tilley2	Ontario	-84.39	47.02	May 15, 2010	16--May-2010	1
67ab	127	Alberta	-115.18	56.01	August 19, 1999	August 15, 1999	4
24on	Dean2	Ontario	-83.18	46.23	July 25, 2011	July 31, 2011	6
14on	Cloudy	Ontario	-83.93	46.44	June 24, 2009	June 23, 2009	1
102ab	88	Alberta	-115.50	56.04	August 15, 1999	August 15, 1999	0
56ab	89	Alberta	-115.51	56.02	August 15, 1999	August 15, 1999	0
13on	Rock	Ontario	-83.77	46.43	June 24, 2009	June 23, 2009	1

Table A.3 TOA radiance values for Landsat bands 1–5, standard deviation (SD) of radiance in band 5 and TOA reflectance values (with partial atmospheric correction) for Landsat bands 1–4 for 53 ground-sampled lakes. LT = Landsat 5; LE = Landsat 7.

Lake ID	Sample Name/ID	Landsat scene ID (LT or LE)	TOA radiance B1	TOA radiance B2	TOA radiance B3	TOA radiance B4	TOA radiance B5	Standard deviation (SD) B5	TOA reflectance B1	TOA reflectance B2	TOA reflectance B3	TOA reflectance B4
101ab	5992	LT05 044021 20020808	33.34	21.32	12.28	11.99	0.44	0.28	0.02	0.02	0.02	0.05
10on	Sill	LT50220272010168	40.85	24.92	12.58	12.32	0.70	0.43	0.03	0.02	0.01	0.04
102ab	88	LE07 045021 19990815	33.37	20.25	10.52	6.05	0.29	0.23	0.02	0.02	0.01	0.03
108ab	75	LE07 044021 20010813	44.51	30.10	18.01	12.18	0.54	0.21	0.05	0.04	0.04	0.05
1on	Negick2	LT50220272010168	41.71	26.17	14.61	13.86	1.07	0.48	0.03	0.03	0.02	0.05
12on	Reception1	LT50210282009174	47.32	32.62	18.40	17.46	0.75	0.43	0.04	0.04	0.03	0.06
13on	Rock	LT50210282009174	46.99	29.99	16.97	12.34	0.50	0.29	0.04	0.03	0.02	0.04
14on	Cloudy	LT50210282009174	46.76	29.29	15.26	13.19	0.60	0.37	0.04	0.03	0.02	0.05
16ab	101	LE07 044021 20010813	40.32	24.36	13.40	12.21	0.93	0.44	0.04	0.03	0.02	0.05
16on	Constance1	LT50210282009174	46.13	29.35	16.45	15.69	0.82	0.45	0.04	0.03	0.02	0.04
17on	Appleby1	LT50210282009174	45.49	28.70	16.33	16.49	0.85	0.46	0.03	0.03	0.02	0.06
19on	Woodrow2	LT50210282010241	34.87	20.44	10.22	7.95	0.42	0.27	0.02	0.02	0.01	0.03
20on	Round	LT50210282009174	47.62	31.60	16.84	14.43	0.64	0.36	0.04	0.04	0.02	0.05
2ab	42	LE07 044021 20010813	40.94	27.07	15.88	12.19	0.62	0.21	0.04	0.04	0.03	0.05
2on	Upper Griffin	LT50220272010168	41.34	24.74	12.87	10.73	0.62	0.37	0.03	0.02	0.01	0.04
22on	Eaket1	LT50210282009174	45.90	29.81	16.88	16.58	0.93	0.51	0.04	0.03	0.02	0.06
23on	Twin	LT50210282011212	45.54	28.90	16.61	13.96	0.83	0.22	0.04	0.03	0.03	0.05
24ab	12	LE07 044021 20010813	42.75	28.04	15.63	14.82	0.75	0.38	0.04	0.04	0.03	0.06
24on	Dean2	LT50210282011212	42.33	26.91	14.24	9.57	0.53	0.19	0.03	0.03	0.02	0.04
25on	Caysee2	LT50220272010168	41.35	25.76	14.54	13.12	0.87	0.49	0.03	0.02	0.02	0.05
26on	Carp	LT50220272010168	44.44	29.50	16.75	12.89	0.89	0.39	0.03	0.03	0.02	0.05
28ab	57	LE07 045021 19990815	33.47	20.54	11.39	17.28	0.53	0.2	0.02	0.02	0.02	0.07
37ab	1681	LE07 044021 20010813	44.41	29.90	17.05	18.13	0.79	0.29	0.05	0.04	0.03	0.08
37ab	1682	LT05 044021 20020808	34.30	21.76	12.39	10.00	0.47	0.17	0.03	0.02	0.02	0.04
38ab	171	LE07 045021 19990815	34.60	22.01	12.93	9.61	0.41	0.23	0.03	0.02	0.02	0.04
45ab	131	LE07 045021 19990815	34.46	22.54	12.40	8.01	0.36	0.26	0.03	0.02	0.02	0.03

Lake ID	Sample Name/ID	Landsat scene ID (LT or LE)	TOA radiance B1	TOA radiance B2	TOA radiance B3	TOA radiance B4	TOA radiance B5	Standard deviation (SD) B5	TOA reflectance B1	TOA reflectance B2	TOA reflectance B3	TOA reflectance B4
46ab	165	LE07 045021 19990815	33.89	20.54	12.62	7.66	0.41	0.26	0.02	0.02	0.02	0.03
5ab	7	LE07 044021 20010813	43.52	29.72	17.32	19.23	0.66	0.24	0.05	0.04	0.03	0.08
5on	Big Turkey	LT50220272010136	38.59	23.61	12.55	9.34	0.83	0.43	0.02	0.02	0.01	0.03
53ab	2011	LE07 045021 19990815	34.40	21.67	11.64	5.13	0.28	0.19	0.03	0.02	0.02	0.02
53ab	201	LE07 044021 20010813	44.41	30.99	17.74	10.10	0.51	0.25	0.05	0.04	0.03	0.04
53ab	2012	LT05 044021 20020808	33.18	20.92	11.49	5.48	0.29	0.23	0.02	0.02	0.02	0.02
55ab	81	LE07 045021 19990815	33.99	21.63	11.75	7.30	0.34	0.24	0.02	0.02	0.02	0.03
56ab	89	LE07 045021 19990815	34.04	21.72	11.16	6.69	0.24	0.18	0.02	0.02	0.02	0.03
58ab	1111	LE07 045021 19990815	32.63	19.02	9.71	7.54	0.43	0.22	0.02	0.02	0.01	0.03
58ab	111	LE07 044021 20010813	42.57	27.07	15.09	14.21	0.67	0.25	0.04	0.04	0.03	0.06
62ab	33	LE07 044021 20010813	42.04	25.23	13.76	10.38	0.53	0.27	0.04	0.03	0.02	0.04
67ab	127	LE07 045021 19990815	35.18	25.48	14.24	9.71	0.30	0.18	0.03	0.03	0.02	0.04
68ab	61	LE07 044021 20010813	44.03	27.51	15.63	14.76	0.72	0.33	0.05	0.04	0.03	0.06
70ab	121	LE07 045021 19990815	33.52	21.08	12.76	7.94	0.38	0.21	0.02	0.02	0.02	0.03
70ab	1211	LE07 044021 20010813	43.08	27.04	14.80	14.73	0.76	0.33	0.04	0.04	0.03	0.06
70ab	1212	LT05 044021 20020808	33.71	21.20	12.01	8.07	0.38	0.17	0.02	0.02	0.02	0.03
7ab	4	LE07 044021 20010813	43.14	28.40	15.93	25.53	1.04	0.47	0.05	0.04	0.03	0.11
7on	Little Turkey	LT50220272010136	37.41	22.05	11.78	8.76	0.73	0.43	0.02	0.02	0.01	0.03
75ab	87	LE07 044021 20010813	42.88	26.96	16.19	14.04	0.66	0.3	0.04	0.04	0.03	0.06
78ab	27	LE07 044021 20010813	43.36	27.93	16.26	17.24	1.09	0.3	0.05	0.04	0.03	0.07
80ab	55	LE07 044021 20010813	43.23	32.30	20.23	15.20	0.64	0.28	0.05	0.05	0.04	0.07
8on	Upper Tilley2	LT50220272010136	38.72	24.34	12.15	7.53	0.41	0.3	0.02	0.02	0.01	0.03

Lake ID	Sample Name/ID	Landsat scene ID (LT or LE)	TOA radiance B1	TOA radiance B2	TOA radiance B3	TOA radiance B4	TOA radiance B5	Standard deviation (SD) B5	TOA reflectance B1	TOA reflectance B2	TOA reflectance B3	TOA reflectance B4
9ab	47	LE07 044021 20010813	39.72	23.23	12.43	12.65	0.77	0.49	0.04	0.03	0.02	0.05
92ab	122	LE07 045021 19990815	34.29	23.61	13.12	8.78	0.38	0.24	0.03	0.03	0.02	0.04
92ab	1222	LE07 044021 20010813	45.37	32.48	18.99	15.16	0.66	0.25	0.05	0.05	0.04	0.06
92ab	1223	LT05 044021 20020808	33.93	21.43	12.22	9.35	0.45	0.26	0.02	0.02	0.02	0.04
98ab	16	LT05 044021 20020808	33.52	23.94	12.04	6.90	0.28	0.2	0.02	0.03	0.02	0.03

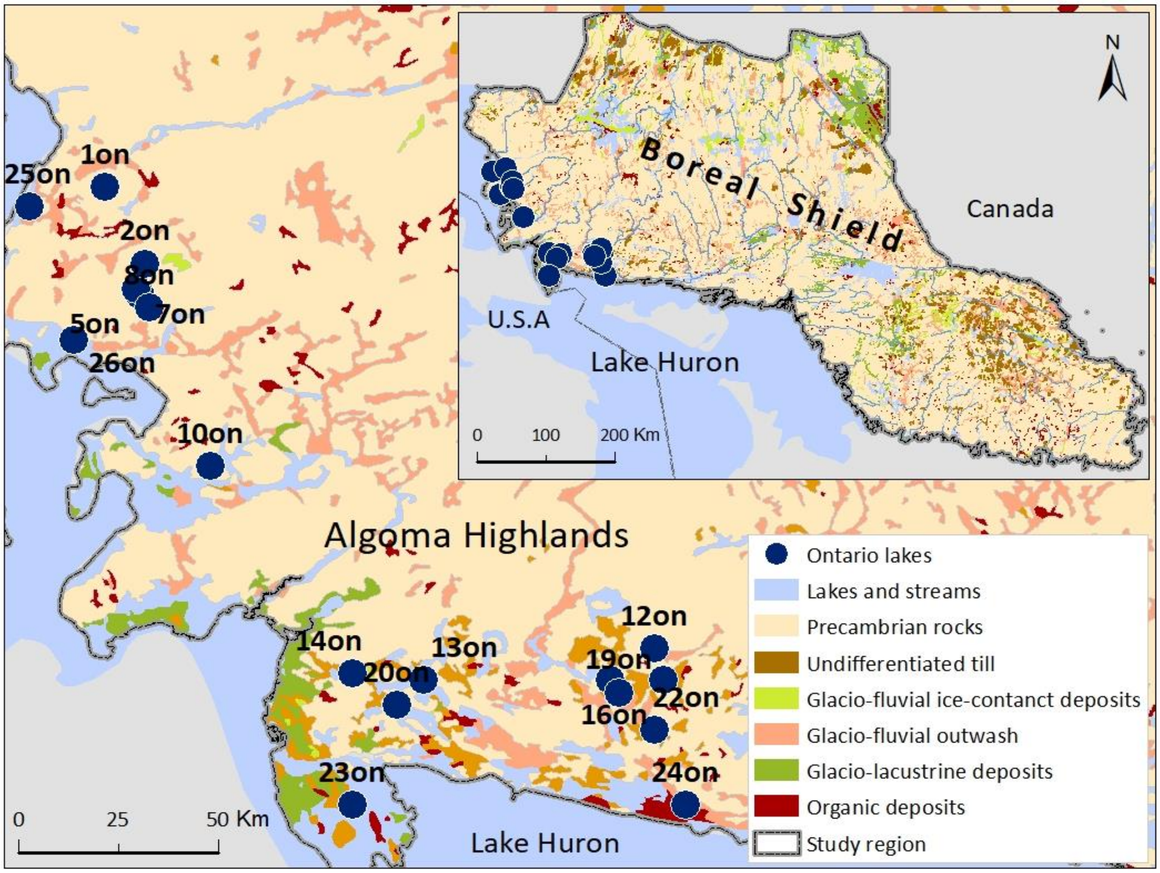

Figure A.1 Location of Ontario lakes within study region, and surficial geology of study region. Lake numbers on the map correspond to Lake IDs in Tables 2.1, A.1, A.2 and A.3.

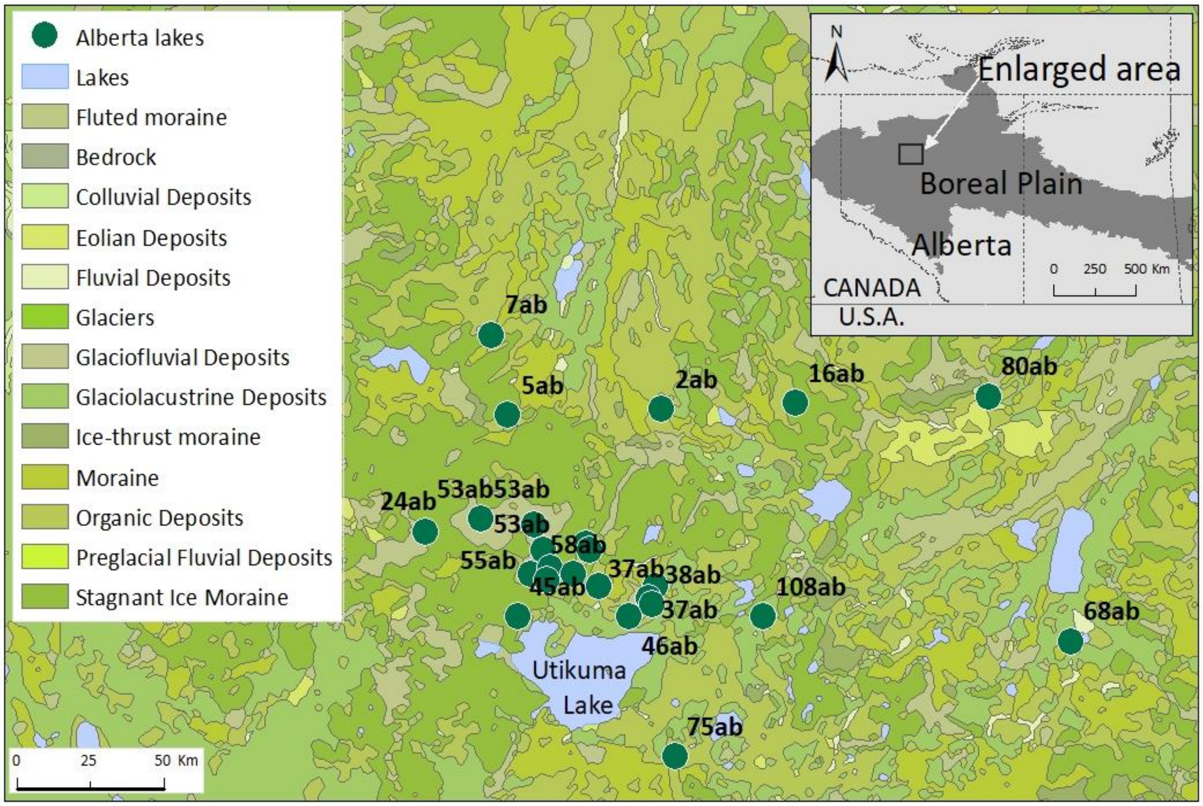

Figure A.2 Location of Alberta lakes, and surficial geology of the site (Utikuma Highlands). Lake numbers on the map correspond to Lake IDs in Tables 2.2, A.1, A.2 and A.3.

Appendix B: Interpolation of missing Chlorophyll-*a* values: kriging

Of 20,930 lakes in the study region, only 6,300 were found to have complete Chl-a_{mod} time series (i.e., for all 28 years). The spatial distribution of ln Chl-a_{mod} time series was uneven with large alternating north-south swathes of complete (where Landsat ground tracks overlap) and incomplete time series (see Figure B.1).

Considerable effort was undertaken to select an appropriate algorithm for interpolating gaps in the time series of ln Chl-a_{mod} values. Most existing methods consider cases when data are correlated either in time or space (Tobin *et al.*, 2011; Moreno *et al.*, 2014). Recently developed spatio-temporal kriging methods allow for correlation in both spatial and temporal dimensions (Cressie & Wikle, 2011). For this study, the universal space-time kriging (hereafter referred to as kriging) was adapted because it has been widely used for relatively unbiased prediction (Kilibarda *et al.*, 2014) of environmental variables (e.g., Heuvelink & Van Egmond, 2010; Wang *et al.*, 2015; Tonini *et al.*, 2016).

Several pre-processing steps were undertaken to prepare the time series of ln Chl-a_{mod} for kriging. Due to a large proportion of missing data (for some lakes it reached 89%), the first step was to identify the trade-off between the proportion of missing data acceptable for getting unbiased results and equal distribution of lakes throughout the study region. Lakes with more than five years missing were removed from the lake inventory and no interpolation efforts were applied. Remaining lakes accounted for 4.8% of missing data, within a 5% threshold considered to be acceptable for large datasets containing missing values (Schafer, 1999). Kriging was performed in R environment with using spacetime, gstat, and rgdal packages (R Core Team, 2013) following the procedure described by Tonini *et al.* (2016).

Consider a variable $z(s_i, t_i)$ that varies within a spatial domain S and a time interval T. Let z be observed at n space-time points (s_i, t_i), $i = 1 \dots n$. The idea thus is to predict $z(s_0, t_0)$ at a point (s_0, t_0) at which z was not observed (Heuvelink *et al.*, 2012). In kriging, predictions are obtained by analyzing spatio-temporal covariances between observed variables $z(s_i, t_i)$. This might be done by using a spatio-temporal sample variogram, which is empirically derived from the residuals of the data (i.e., Chl-a_{mod}). The variogram measures the average dissimilarity between data separated by a given spatial and temporal lag (h, u) defined as Equation A1 (Kilibarda *et al.*, 2014):

$$\gamma(\mathrm{h},\mathrm{u}) = \frac{1}{2\mathrm{n}(\mathrm{h},\mathrm{u})} \sum_{\mathrm{i}=1}^{\mathrm{n}(\mathrm{h},\mathrm{u})} [\mathrm{z}(\mathrm{s_i},\mathrm{t_i}) - \mathrm{z}(\mathrm{s_i}+\mathrm{h},\mathrm{t_i}+\mathrm{u})]^2 \quad \text{[B1]}$$

where h is the Euclidean distance and u is the time interval.

Once the sample variogram $\gamma(\mathrm{h},\mathrm{u})$ is obtained, a theoretical spatio-temporal variogram model may be fitted. Of the diverse range of models (e.g., metric model, product-sum model), the sum-metric model was applied in this study because it allows maximum flexibility between the spatial and temporal correlation domains (Kilibarda *et al.*, 2014). The sum-metric variogram structure is defined as Equation A2:

$$\gamma(\mathrm{h},\mathrm{u}) = \gamma_{\mathrm{S}}(\mathrm{h}) + \gamma_{\mathrm{T}}(\mathrm{u}) + \gamma_{\mathrm{ST}}\left(\sqrt{\mathrm{h}^2 + (\alpha - \mathrm{u})^2}\right) \quad \text{[B2]}$$

where γ(h, u) stands for the semivariance for h and u units of spatial and temporal distance, respectively. γ_{S} and γ_{T} describe the purely spatial and temporal components, while γ_{ST} space-time describes the interaction component; α is a parameter of the spatio-temporal anisotropy (Kilibarda *et al.*, 2014). The spatio-temporal anisotropy was calculated following Tonini *et al.* (2016), while other parameters (i.e., sill, nugget and range) were estimated by visual judgement of the sample variogram surface. The sum-metric model variogram was fitted using an exponential, Gaussian and spherical functions (Tonini *et al.*, 2016).

Interpolated ln Chl-a_{mod} values of the reconstructed time series were evaluated in terms of prediction accuracy. Two hundred lakes with continuous ln Chl-a_{mod} covering the entire range of Chl-a_{mod} (from minimum to maximum values) were selected. ln Chl-a_{mod} values were artificially removed with a different missing pattern from one to five years missing. Kriging was applied with one more iteration on all lakes including the new subset of 200 lakes. Both resulting interpolated ln Chl-a_{mod} values for 200 lakes and original ln Chl-a_{mod} values were averaged in each dataset, and then regressed against each other.

The constructed sample space-time variogram of residuals of ln Chl-a_{mod} (Figure B.2) indicated that these residuals were correlated in both space and time, and therefore kriging was applicable. Table B.1 summarizes the parameter estimates of the sum-metric variogram model. Kriging interpolated missing ln Chl-a_{mod} accurately ($r^2 = 0.99$, $p < 0.001$, RMSE = 0.16; Figure B.3), resulting in 12,644 lakes with continuous Chl-a_{mod} time series. Lakes that had more than five years of missing ln Chl-a_{mod} values were not used in further analyses.

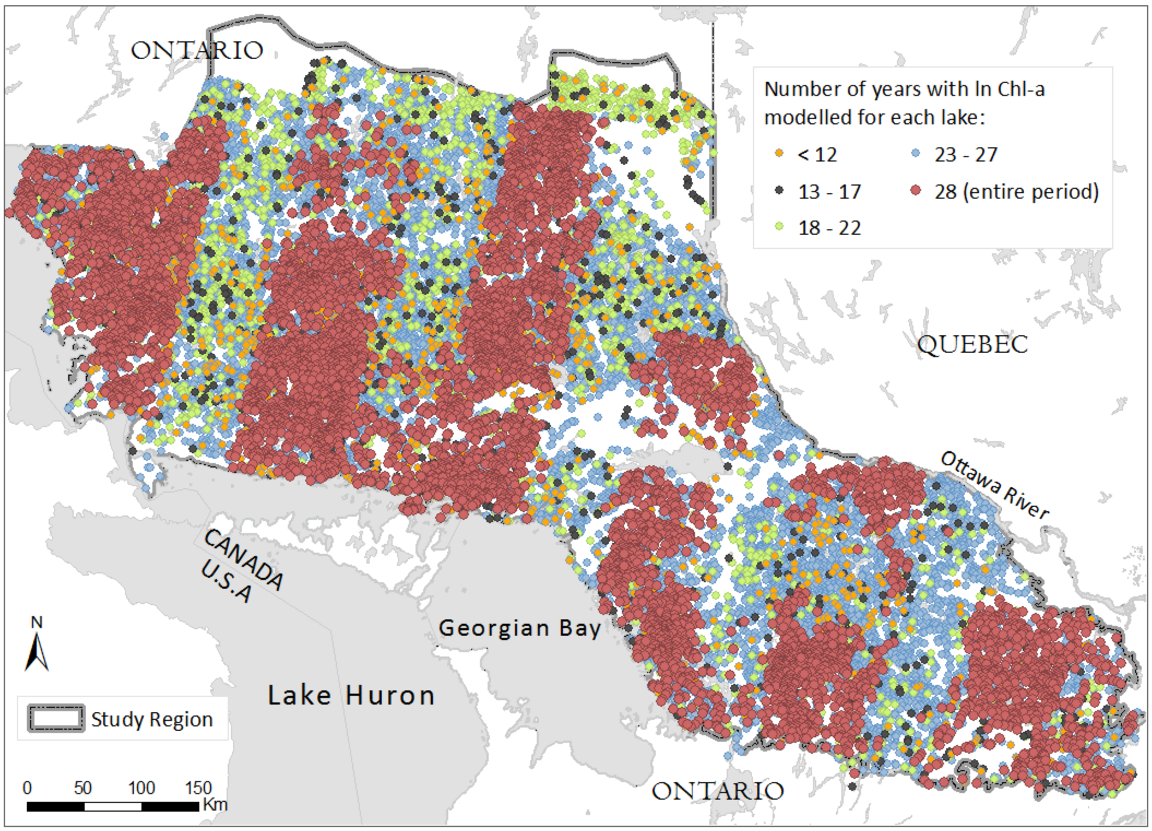

Figure B.1 Number of years (out of entire period: 28 years) of ln Chl-a_{mod} data by lake.

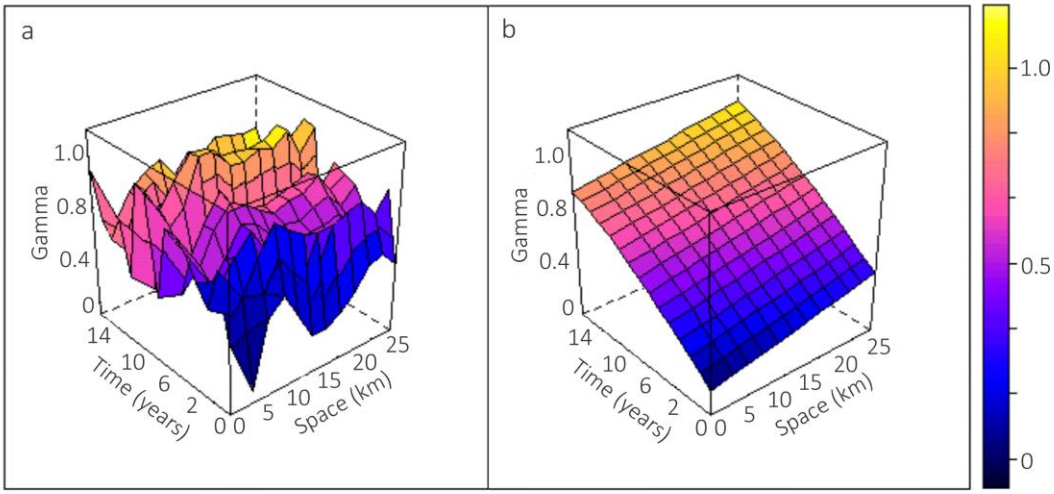

Figure B.2 (a) Sample space–time variogram of residuals from ln Chl-a_{mod}; (b) Fitted sum-metric model used in kriging.

Table B.1 Parameters of the fitted sum-metric variogram model for ln Chl-a_{mod} used in kriging.

Variogram component	Model	Nugget	Sill	Range	Anisotropy ratio
Space	Exponential	0.10	0.80	99.0 km	
Time	Gaussian	0	0.14	10.2 year	
Joint (space-time)	Spherical	0.14	0.64	150.0 km	4.51 km/year

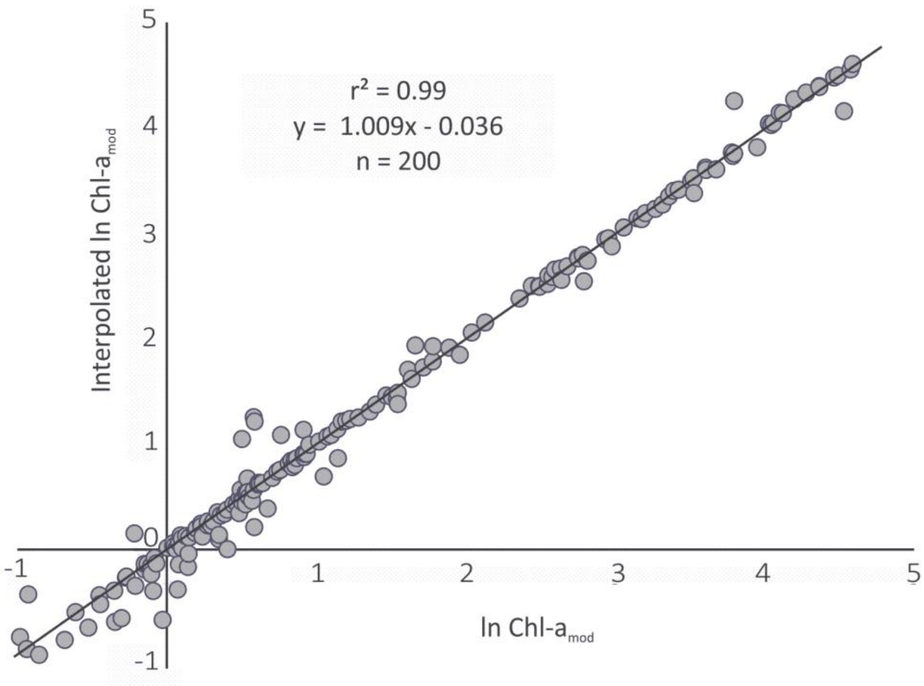

Figure B.3 Prediction accuracy of kriging preformed on ln Chl-a_{mod} from 200 randomly selected lakes to interpolate missing values. The solid line represents the 1:1 line. The root means square error (RMSE) of Chl-a interpolation was 0.16.

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Appendix C: Effect of lake DOC on modeling of lake Chlorophyll-*a* (Chapter 2)

The purpose of this appendix is to confirm that modeled Chl-a results in this study are not influenced by DOC. In Chapter 2, I found a weak and non-significant correlation (r^2 = 0.14, p = 0.09; Figure 2.4) between (B1-B3)/B2 reflectance values and ln DOC in 23 lakes in the 39-lake model development dataset, suggesting that ln DOC may not have a likely effect on (B1-B3)/B2 reflectance in lakes in the study region.

Given the strong correlation between observed and modeled Chl-a (r^2 = 0.76, $p < 0.01$; Figure 2.3a), I tested the difference between observed Chl-a and DOC in lakes in the study region to demonstrated that these parameters are not correlated and to validate the suggestion that lake DOC is not affecting modeled Chl-a results. To improve upon the small sample size in the model development and testing, I have searched for all potential in-situ lake datasets with Chl-a and DOC within the study region and have identified that only three have been validated, each obtained by members of Dr. Irena Creed's research team in the Department of Biology at Western University. The datasets collected by Ryan Sorichetti (Sorichetti dataset) for the years 2009-2011 and by Gabor Sass in Alberta lakes (Sass dataset) for the years 1999-2002 have already been used in the model development and validation. A third dataset was collected by Oscar Senar (Senar dataset) in 72 Ontario lakes for the years 2015-2016 (Table C.1).

Using all the lakes in each of the three datasets (Figure C.1), I found weak and non-significant correlations between observed lake Chl-a and DOC in the Sorichetti and Sass datasets consistent with the finding in Chapter 2 (Sorichetti: r^2 = 0.14, p = 0.367, n = 8; Sass: r^2 = 0.01, p = 0.741, n = 20). In contrast, however, there was a significant and stronger (but not strong) correlation between observed lake Chl-a and DOC in the Senar dataset (r^2 = 0.44, $p < 0.001$, n = 70). When all lakes in all three datasets are included in the same regression, the correlation between observed Chl-a and lake DOC remains weak but becomes significant (r^2 = 0.25, $p < 0.001$, n = 98; Figure C.2).

Given the limited range of observed lake DOC values in the Sorichetti and Senar datasets (max DOC = 16.40 and 12.40 mg L^{-1}, respectively), I consider it appropriate to include the Sass dataset in this comparison (similarly, the Sass dataset was included in the Chl-a model development dataset to compensate for the limited trophic range in the Sorichetti dataset). Therefore, I conclude that lake DOC is not strongly affecting modeled Chl-a results, and this conclusion is supported by a weak and non-significant correlation between modeled Chl-a and lake DOC in the Sorichetti and Sass datasets (Sorichetti: r^2 = 0.06, p = 0.556, n = 8; Sass: r^2 = 0.04, p = 0.379, n = 20; all lakes: r^2 = 0.11, p = 0.082, n = 28; Figure C.3). However, a coefficient of determination of 0.44 supports a degree of caution in the confidence of this conclusion.

Table C.1 Samples used to evaluate effect of lake DOC on modeled lake Chl-a.

Dataset	Sample Name/ID	Year	Latitude	Longitude	DOC (mg L^{-1})	Chl-a (µg L^{-1})
Sass	4	2001	56.42	-115.68	59.92	2.82
Sass	7	2001	56.29	-115.63	56.70	4.42
Sass	12	2001	56.10	-115.88	27.85	15.87
Sass	16	2002	56.11	-115.55	22.51	12.00
Sass	27	2001	56.07	-115.52	25.88	12.38
Sass	42	2001	56.30	-115.16	40.92	40.80
Sass	55	2001	56.32	-114.16	60.68	61.51
Sass	57	1999	56.08	-115.39	-	8.70
Sass	59	2002	56.07	-115.38	60.25	28.60
Sass	61	2001	55.92	-113.91	22.11	1.95
Sass	75	2001	55.96	-114.85	60.21	34.20
Sass	81	1999	56.03	-115.56	-	9.18
Sass	87	2001	55.73	-115.12	79.58	7.38
Sass	88	1999	56.04	-115.50	-	3.66
Sass	89	1999	56.02	-115.51	-	3.54
Sass	101	2001	56.31	-114.75	38.63	2.00
Sass	111	2001	56.03	-115.43	48.78	2.82
Sass	121	1999	56.01	-115.35	-	46.00
Sass	121	2001	56.01	-115.35	50.29	3.48
Sass	121	2002	56.01	-115.35	58.50	12.10
Sass	122	1999	56.01	-115.35	-	58.01
Sass	122	2002	56.01	-115.35	68.92	31.40
Sass	127	1999	56.01	-115.18	-	57.20
Sass	131	1999	55.96	-115.60	-	18.51
Sass	165	1999	55.96	-115.26	-	63.40
Sass	168	2001	55.99	-115.20	58.34	6.42
Sass	168	2002	55.99	-115.20	74.91	3.77
Sass	171	1999	55.98	-115.19	-	47.10
Sass	201	1999	56.12	-115.71	-	20.21
Sass	201	2001	56.12	-115.71	23.19	30.27
Sass	201	2002	56.12	-115.71	27.06	13.00
Sorichetti	Appleby	2009	46.43	-83.35	-	4.61
Sorichetti	Carp	2010	46.97	-84.56	5.64	3.72
Sorichetti	Caysee	2010	47.18	-84.66	8.27	2.48
Sorichetti	Cloudy	2009	46.44	-83.93	-	0.50
Sorichetti	Constance	2009	46.43	-83.23	-	1.11
Sorichetti	Dean	2011	46.23	-83.18	-	3.18

Dataset	Sample Name/ID	Year	Latitude	Longitude	DOC (mg L-1)	Chl-a (µg L-1)
Sorichetti	Eaket	2009	46.35	-83.25	-	2.76
Sorichetti	L4 (Little Turkey)	2010	47.04	-84.41	16.40	0.45
Sorichetti	L5 (Big Turkey)	2010	47.05	-84.42	3.76	1.39
Sorichetti	Negick	2010	47.21	-84.49	2.63	2.35
Sorichetti	Reception	2009	46.48	-83.25	-	9.99
Sorichetti	Rock	2009	46.43	-83.77	-	3.70
Sorichetti	Round	2009	46.39	-83.83	-	3.30
Sorichetti	Sill	2010	46.77	-84.25	-	0.92
Sorichetti	Twin	2011	46.23	-83.93	-	7.44
Sorichetti	Upper Griffin	2010	47.09	-84.40	3.81	0.65
Sorichetti	Upper Tilley	2010	47.02	-84.39	4.18	2.10
Sorichetti	Woodrow	2010	46.41	-83.33	7.15	0.56
Senar	Bass Lake	2016	44.68	-78.53	5.80	2.36
Senar	Bat Lake	2015	45.15	-78.63	4.50	2.57
Senar	Bearpaw Lake	2015	44.93	79.49	9.00	18.75
Senar	Bella Lake	2016	45.45	-79.02	2.60	2.00
Senar	Bigwind Lake	2015	45.06	-79.05	4.00	10.52
Senar	Boshkung Lake	2015	45.05	-78.72	3.80	2.50
Senar	Brandy Lake	2016	45.11	-79.52	9.50	7.89
Senar	Buck Lake (2)	2015	45.41	-79.39	7.20	2.48
Senar	Burrow's Lake	2016	44.84	-79.66	5.50	5.79
Senar	Cassels Lake	2015	47.07	-79.72	5.90	2.38
Senar	Chub Lake (2)	2015	45.21	-78.98	6.20	2.67
Senar	Cinder Lake	2016	45.06	-78.92	6.00	4.67
Senar	Clear Lake (1)	2015	46.10	-79.77	4.80	3.64
Senar	Clear Lake (2)	2015	45.45	-77.22	4.90	2.56
Senar	Couchiching, Lake	2016	44.65	-79.36	5.30	3.47
Senar	Crystal Lake	2016	44.76	-78.48	4.90	1.67
Senar	Dark Lake	2015	45.00	-79.59	4.20	3.90
Senar	Davis Lake	2016	44.79	-78.71	5.30	1.73
Senar	Deer Lake	2015	46.48	-80.22	8.80	13.27
Senar	Depensiers Lake	2016	46.31	-79.41	9.10	12.52
Senar	Devil's Lake	2016	44.87	-78.83	4.20	3.87
Senar	Dore, Lake	2015	45.63	-77.09	6.50	6.44
Senar	Dreany Lake	2016	46.29	-79.36	12.40	18.22
Senar	Eagle Lake (2)	2015	45.13	-78.50	4.10	2.52
Senar	Fawn Lake	2015	45.17	-79.26	9.00	8.04
Senar	Fletcher Lake	2015	45.35	-78.78	4.20	6.55
Senar	Fosters Lake	2015	45.25	-77.67	8.50	2.57
Senar	Four Mile Lake	2016	44.67	-78.74	5.60	-
Senar	Fox Lake	2016	45.39	-79.36	7.10	7.33
Senar	Gull Lake (1)	2015	44.59	-76.99	6.30	4.27
Senar	Gull Lake (2)	2016	44.84	-78.79	3.50	1.61
Senar	Head Lake	2016	44.74	-78.90	4.40	3.80
Senar	Kamaniskeg Lake	2015	45.39	-77.68	4.50	2.61
Senar	Kashagawigamog Lake	2016	44.99	-78.59	4.20	1.91
Senar	Koshlong Lake	2016	44.97	-78.49	3.80	2.38
Senar	Leggat Lake	2015	44.71	-76.73	3.90	4.91
Senar	Leonard Lake	2015	45.07	-79.44	4.80	2.96
Senar	Loom Lake	2016	44.75	-78.46	5.90	2.86
Senar	Loon Lake	2016	45.01	-78.38	5.20	3.57
Senar	MacLean Lake	2016	44.82	-79.66	7.40	11.45
Senar	Maple Lake	2016	45.10	-78.66	3.70	1.24
Senar	Mary Lake	2016	45.26	-79.24	5.10	2.15

Dataset	Sample Name/ID	Year	Latitude	Longitude	DOC (mg L^{-1})	Chl-a (µg L^{-1})
Senar	McKenzie Lake	2015	45.37	-78.01	5.30	3.24
Senar	Menominee Lake	2016	45.19	-79.14	7.10	3.44
Senar	Mink Lake	2016	46.18	-79.22	9.80	11.98
Senar	Moot Lake	2015	45.14	-79.17	6.40	4.61
Senar	Morrison Lake	2016	44.87	-79.45	5.50	2.91
Senar	Muskosung Lake	2015	46.49	-80.05	6.70	7.07
Senar	Norway Lake	2015	45.34	-76.71	8.70	2.31
Senar	Nosbonsing Lake	2016	46.20	-79.25	3.90	9.55
Senar	Otter Lake	2015	45.28	-78.87	4.10	3.36
Senar	Oxbow Lake	2016	45.44	-78.97	4.00	1.58
Senar	Paint Lake	2016	45.22	-78.95	3.90	3.62
Senar	Raven Lake	2016	45.21	-78.85	3.50	3.37
Senar	Red Chalk Lake	2015	45.19	-78.95	3.10	2.08
Senar	Red Squirrel Lake	2015	47.16	-80.02	3.80	2.03
Senar	Rib Lake	2015	47.22	-79.72	4.10	1.56
Senar	Ril Lake	2016	45.17	-79.01	4.10	4.04
Senar	Riley Lake	2016	44.84	-79.18	4.70	3.97
Senar	Rosseau, Lake	2015	45.24	-79.64	3.60	3.21
Senar	Shadow Lake	2015	44.73	-78.79	3.90	1.95
Senar	Siding Lake	2015	45.28	-79.32	8.00	6.77
Senar	Skeleton Lake	2016	45.24	-79.47	2.00	2.23
Senar	Skootamata Lake	2015	44.84	-77.23	5.80	3.27
Senar	Sparrow lake	2016	44.81	-79.38	4.90	2.63
Senar	St. John Lake	2016	44.69	-79.33	9.30	24.21
Senar	Tea Lake	2016	44.87	-79.65	6.30	3.89
Senar	Three Mile Lake	2016	45.17	-79.46	4.30	8.86
Senar	Twelve Mile Lake	2016	45.02	-78.71	3.20	-
Senar	Wasi Lake	2016	46.14	-79.23	7.90	6.31
Senar	Wicksteed Lake	2015	46.76	-79.69	7.80	4.85
Senar	Wood Lake	2016	45.02	-79.07	4.00	2.32

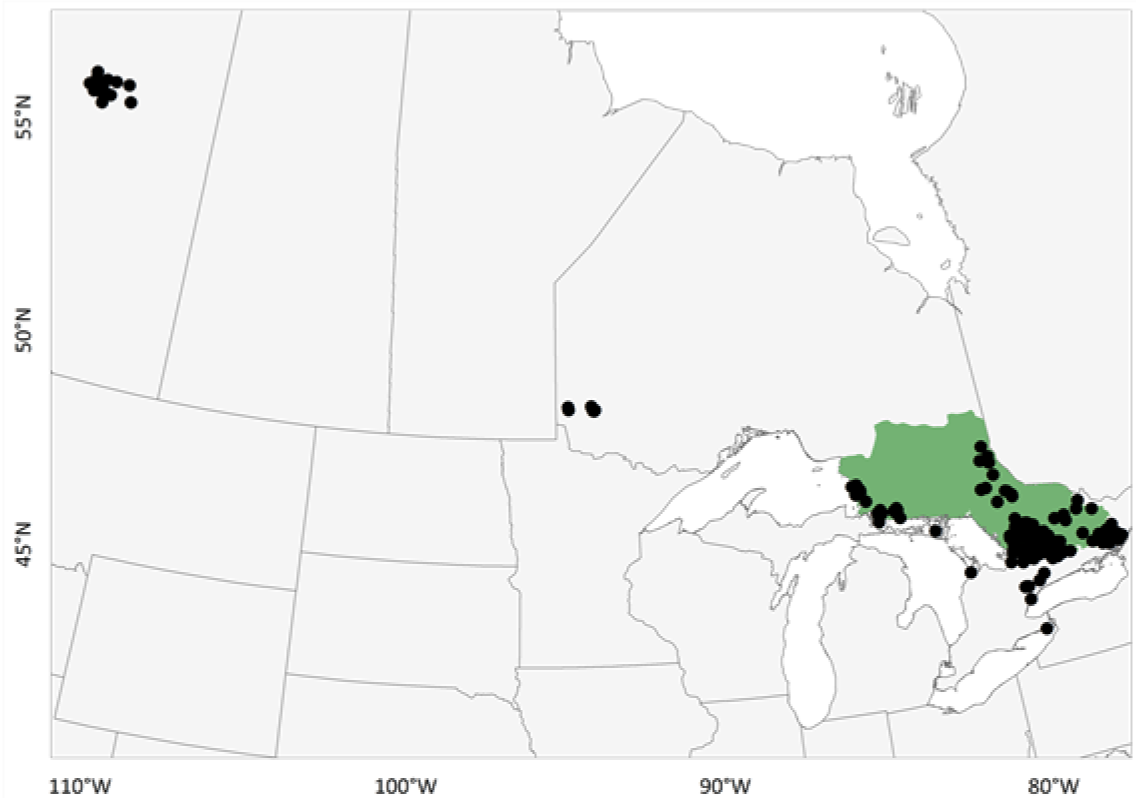

Figure C.1 Study region (green area) and location of in-situ Chl-a, TP and DOC data.

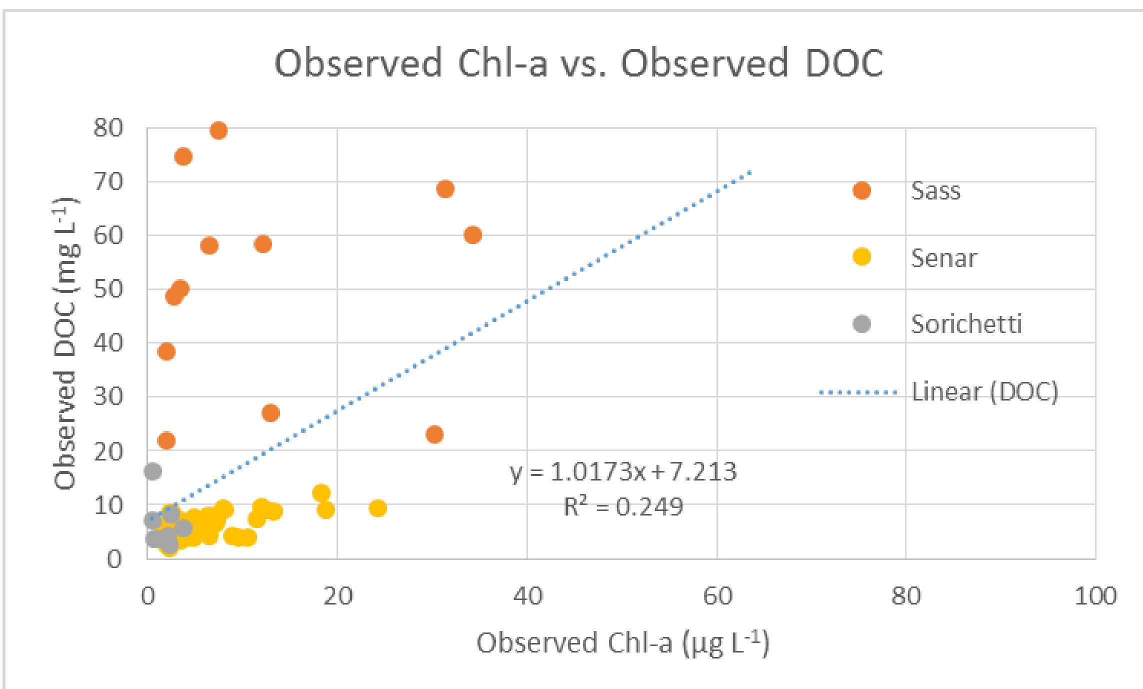

Figure C.2 Relationship between observed Chl-a and DOC.

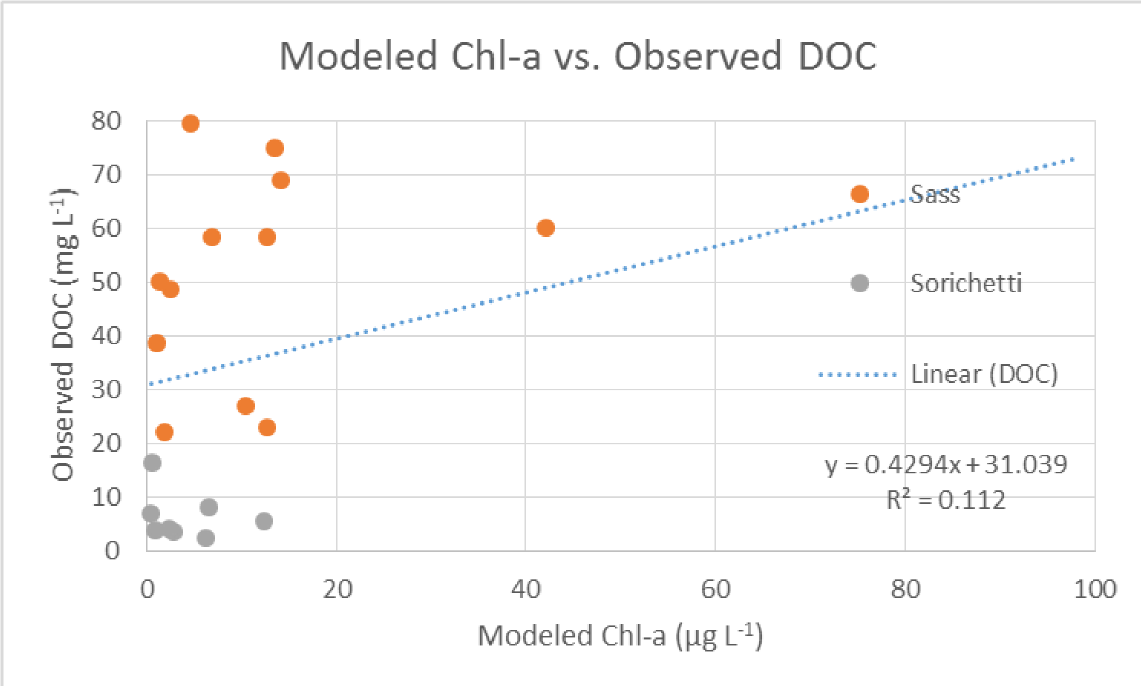

Figure C.3 Relationship between modeled Chl-a and observed DOC.

Appendix D: Using Total Phosphorus as a proxy for Chlorophyll-*a* (Chapter 2)

The purpose of this appendix is to provide support for future validation of modeled Chl-a temporal trend analyses as given in Chapter 2.4.4 and 4.5.3.

The datasets used for validation in Appendix C (Sass, Sorichetti and Senar datasets) do not contain time series of observed Chl-a and therefore do not support validation of the trend analyses presented in this study. Forthcoming validation of large time series datasets from Environment Canada, the Ontario Ministry of Natural Resources and Forestry, and the Ministry of the Environment, Conservation and Parks (Lake Partnership Program) may be expected in future work to help validate these results, but these datasets are largely limited to records of lake total phosphorus (TP) and DOC. Because P is the major limiting nutrient for phytoplankton growth in lakes, and because phytoplankton biomass is generally measured by Chl-a concentration, TP can potentially be used as a proxy for Chl-a.

I tested the viability of using this proxy with the Sorichetti, Senar and Sass lake datasets (Figure C.1; Table D.1) that include in-situ lake TP and Chl-a. After removing 22 outliers from a potential dataset of 117 matching observed lake TP and Chl-a (Cook's distance > 4/n), I found a strong ($r^2 = 0.72$) and significant ($p < 0.001$) correlation (Figure D.1). However, the strength of this correlation is the product of the correlation between observed lake TP and Chl-a in the Sass dataset ($r^2 = 0.83$, $p < 0.001$, $n = 18$); in contrast, the correlations in the Sorichetti and Senar datasets were weak ($r^2 = 0.24$ and 0.36, respectively) and less significant in the Sorichetti dataset ($p = 0.056$; Senar: $p < 0.001$). These differences may be the result of differences in collection times, sampling depths, and collection or analysis methods. Therefore, I do not conclude that we can be confident using multiple datasets of different origins for time series analysis or validation. Any records of observed lake Chl-a in the forthcoming datasets should be added to the comparison to test the validity of using lake TP as a proxy for lake Chl-a. In the future, I would recommend that the use of this proxy should be dependent on standardized datasets using consistent sampling depths and methods.

Table D.1 Samples used to evaluate correlation between lake TP and lake Chl-a.

Dataset	Lake Name/ID	Year	Latitude	Longitude	TP (µg L^{-1})	Chl-a (µg L^{-1})
Sass	4	2001	56.42	-115.68	233.90	2.82
Sass	7	2001	56.29	-115.63	43.50	4.42
Sass	12	2001	56.10	-115.88	58.20	15.87
Sass	16	2002	56.11	-115.55	68.50	12.00
Sass	27	2001	56.07	-115.52	48.40	12.38
Sass	42	2001	56.30	-115.16	117.00	40.80
Sass	55	2001	56.32	-114.16	246.40	61.51
Sass	57	1999	56.08	-115.39	119.30	8.70
Sass	59	2002	56.07	-115.38	100.80	28.60
Sass	61	2001	55.92	-113.91	68.00	1.95
Sass	75	2001	55.96	-114.85	118.60	34.20
Sass	81	1999	56.03	-115.56	54.40	9.18
Sass	87	2001	55.73	-115.12	57.20	7.38
Sass	88	1999	56.04	-115.50	30.20	3.66
Sass	89	1999	56.02	-115.51	66.70	3.54
Sass	101	2001	56.31	-114.75	17.90	2.00
Sass	111	2001	56.03	-115.43	39.20	2.82
Sass	121	1999	56.01	-115.35	150.80	46.00
Sass	121	2001	56.01	-115.35	58.80	3.48
Sass	121	2002	56.01	-115.35	105.90	12.10
Sass	122	1999	56.01	-115.35	77.70	58.01
Sass	122	2002	56.01	-115.35	123.00	31.40
Sass	127	1999	56.01	-115.18	212.40	57.20
Sass	131	1999	55.96	-115.60	135.80	18.51
Sass	165	1999	55.96	-115.26	178.60	63.40
Sass	168	2001	55.99	-115.20	102.40	6.42
Sass	168	2002	55.99	-115.20	120.80	3.77
Sass	171	1999	55.98	-115.19	421.70	47.10
Sass	201	1999	56.12	-115.71	64.90	20.21
Sass	201	2001	56.12	-115.71	46.30	30.27
Sass	201	2002	56.12	-115.71	58.50	13.00
Sorichetti	Appleby	2009	46.43	-83.35	10.83	4.61
Sorichetti	Carp	2010	46.97	-84.56	17.20	3.72
Sorichetti	Caysee	2010	47.18	-84.66	23.20	2.48
Sorichetti	Cloudy	2009	46.44	-83.93	9.10	0.50
Sorichetti	Constance	2009	46.43	-83.23	6.87	1.11
Sorichetti	Dean	2011	46.23	-83.18		3.18
Sorichetti	Eaket	2009	46.35	-83.25	9.20	2.76
Sorichetti	L4 (Little Turkey)	2010	47.04	-84.41	3.20	0.45
Sorichetti	L5 (Big Turkey)	2010	47.05	-84.42	5.00	1.39
Sorichetti	Negick	2010	47.21	-84.49	12.80	2.35
Sorichetti	Reception	2009	46.48	-83.25	14.70	9.99
Sorichetti	Rock	2009	46.43	-83.77	13.43	3.70
Sorichetti	Round	2009	46.39	-83.83	19.50	3.30
Sorichetti	Sill	2010	46.77	-84.25	10.80	0.92
Sorichetti	Twin	2011	46.23	-83.93		7.44
Sorichetti	Upper Griffin	2010	47.09	-84.40	7.20	0.65
Sorichetti	Upper Tilley	2010	47.02	-84.39	9.20	2.10
Sorichetti	Woodrow	2010	46.41	-83.33	3.80	0.56
Senar	Bass Lake	2016	44.68	-78.53	10.70	2.36
Senar	Bat Lake	2015	45.15	-78.63	14.40	2.57
Senar	Bearpaw Lake	2015	44.93	79.49	10.70	18.75
Senar	Bella Lake	2016	45.45	-79.02	6.50	2.00

Dataset	Lake Name/ID	Year	Latitude	Longitude	TP (µg L^{-1})	Chl-a (µg L^{-1})
Senar	Bigwind Lake	2015	45.06	-79.05	8.00	10.52
Senar	Boshkung Lake	2015	45.05	-78.72	7.70	2.50
Senar	Brandy Lake	2016	45.11	-79.52	48.50	7.89
Senar	Buck Lake (2)	2015	45.41	-79.39	9.20	2.48
Senar	Burrow's Lake	2016	44.84	-79.66	10.40	5.79
Senar	Cassels Lake	2015	47.07	-79.72	12.90	2.38
Senar	Chub Lake (2)	2015	45.21	-78.98	5.40	2.67
Senar	Cinder Lake	2016	45.06	-78.92	10.80	4.67
Senar	Clear Lake (1)	2015	46.10	-79.77	12.70	3.64
Senar	Clear Lake (2)	2015	45.45	-77.22	14.90	2.56
Senar	Couchiching, Lake	2016	44.65	-79.36	12.60	3.47
Senar	Crystal Lake	2016	44.76	-78.48	21.30	1.67
Senar	Dark Lake	2015	45.00	-79.59	5.80	3.90
Senar	Davis Lake	2016	44.79	-78.71	8.10	1.73
Senar	Deer Lake	2015	46.48	-80.22	19.60	13.27
Senar	Depensiers Lake	2016	46.31	-79.41	20.40	12.52
Senar	Devil's Lake	2016	44.87	-78.83	11.80	3.87
Senar	Dore, Lake	2015	45.63	-77.09	25.50	6.44
Senar	Dreany Lake	2016	46.29	-79.36	42.30	18.22
Senar	Eagle Lake (2)	2015	45.13	-78.50	10.70	2.52
Senar	Fawn Lake	2015	45.17	-79.26	21.90	8.04
Senar	Fletcher Lake	2015	45.35	-78.78	6.70	6.55
Senar	Fosters Lake	2015	45.25	-77.67	9.10	2.57
Senar	Four Mile Lake	2016	44.67	-78.74	6.80	
Senar	Fox Lake	2016	45.39	-79.36	11.10	7.33
Senar	Gull Lake (1)	2015	44.59	-76.99	8.80	4.27
Senar	Gull Lake (2)	2016	44.84	-78.79	5.10	1.61
Senar	Head Lake	2016	44.74	-78.90	11.30	3.80
Senar	Kamaniskeg Lake	2015	45.39	-77.68	4.60	2.61
Senar	Kashagawigamog Lake	2016	44.99	-78.59	7.80	1.91
Senar	Koshlong Lake	2016	44.97	-78.49	6.60	2.38
Senar	Leggat Lake	2015	44.71	-76.73	9.00	4.91
Senar	Leonard Lake	2015	45.07	-79.44	6.20	2.96
Senar	Loom Lake	2016	44.75	-78.46	7.40	2.86
Senar	Loon Lake	2016	45.01	-78.38	9.30	3.57
Senar	MacLean Lake	2016	44.82	-79.66	18.70	11.45
Senar	Maple Lake	2016	45.10	-78.66	7.50	1.24
Senar	Mary Lake	2016	45.26	-79.24	12.40	2.15
Senar	McKenzie Lake	2015	45.37	-78.01	6.80	3.24
Senar	Menominee Lake	2016	45.19	-79.14	11.40	3.44
Senar	Mink Lake	2016	46.18	-79.22	18.40	11.98
Senar	Moot Lake	2015	45.14	-79.17	28.90	4.61
Senar	Morrison Lake	2016	44.87	-79.45	9.00	2.91
Senar	Muskosung Lake	2015	46.49	-80.05	11.10	7.07
Senar	Norway Lake	2015	45.34	-76.71	25.60	2.31
Senar	Nosbonsing Lake	2016	46.20	-79.25	32.20	9.55
Senar	Otter Lake	2015	45.28	-78.87	7.30	3.36
Senar	Oxbow Lake	2016	45.44	-78.97	7.20	1.58
Senar	Paint Lake	2016	45.22	-78.95	10.40	3.62
Senar	Raven Lake	2016	45.21	-78.85	6.00	3.37
Senar	Red Chalk Lake	2015	45.19	-78.95	4.30	2.08
Senar	Red Squirrel Lake	2015	47.16	-80.02	7.10	2.03
Senar	Rib Lake	2015	47.22	-79.72	11.60	1.56
Senar	Ril Lake	2016	45.17	-79.01	10.90	4.04
Senar	Riley Lake	2016	44.84	-79.18	14.90	3.97

Dataset	Lake Name/ID	Year	Latitude	Longitude	TP (µg L^{-1})	Chl-a (µg L^{-1})
Senar	Rosseau, Lake	2015	45.24	-79.64	6.10	3.21
Senar	Shadow Lake	2015	44.73	-78.79	11.50	1.95
Senar	Siding Lake	2015	45.28	-79.32	29.70	6.77
Senar	Skeleton Lake	2016	45.24	-79.47	7.80	2.23
Senar	Skootamata Lake	2015	44.84	-77.23	6.80	3.27
Senar	Sparrow lake	2016	44.81	-79.38	13.40	2.63
Senar	St. John Lake	2016	44.69	-79.33	167.00	24.21
Senar	Tea Lake	2016	44.87	-79.65	7.50	3.89
Senar	Three Mile Lake	2016	45.17	-79.46	20.00	8.86
Senar	Twelve Mile Lake	2016	45.02	-78.71	8.10	
Senar	Wasi Lake	2016	46.14	-79.23	31.80	6.31
Senar	Wicksteed Lake	2015	46.76	-79.69	5.60	4.85
Senar	Wood Lake	2016	45.02	-79.07	10.70	2.32

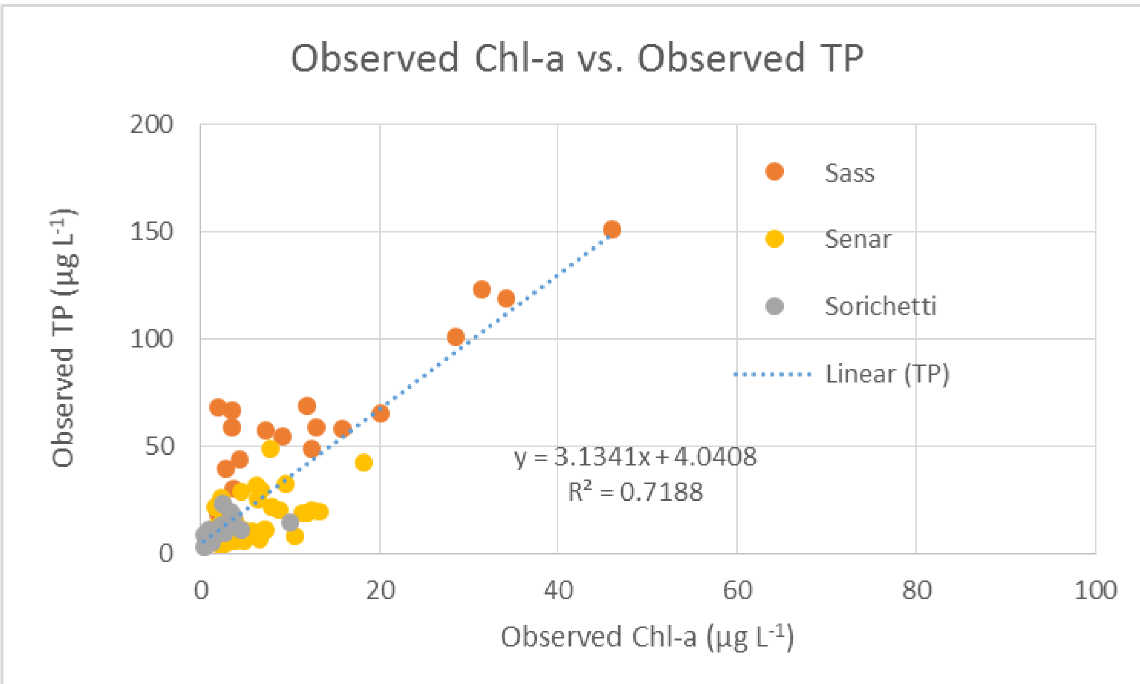

Figure D.1 Relationship between observed Chl-a and TP.

Appendix E: Morphological characteristics of lakes used in landscape analysis (Chapter 3) and trend analysis (Chapter 4)

Table E.1 Morphological characteristics of 275 lakes selected for landscape and trend analyses

Lake ID	Longitude	Latitude	Chl-a (µg L^{-1})	Lake area (ha)	Catchment area (ha)	Catchment slope (°)	Lake fetch (m)	Lake volume (m^3)	Lake mean depth (m)	Lake max depth (m)	Development shoreline	Bathymetry slope (°)	Wetland cover (% of catchment)	Littoral zone in lake (ha)	Dynamic ratio	Lakes used for environmental controls of transitional lakes in Chapter 4
17631	-81.82	48.17	6.6	7.3	659.0	1.1	485	1569.5	0.1	0.6	1.27	0.7	3.6	66545.5	2.70	No
47148	-76.37	45.12	38.1	12.9	101.0	2.3	646.5	4376.8	0.1	0.9	1.25	0.1	14.8	10115.3	3.59	No
47267	-76.31	45.05	41.3	20.9	167.0	0.8	748.8	85062.8	0.4	0.9	1.16	0.4	64.0	16576	1.14	No
47380	-76.37	44.96	30.9	25.5	102.3	1.8	991.4	185007.3	0.7	1.0	1.32	0.4	39.6	10251	0.72	No
47542	-76.25	44.84	12.1	12.3	34.3	2.6	807.4	79504.2	0.7	1.0	1.46	0.6	12.5	3417.4	0.50	No
34022	-79.55	44.82	0.5	47.4	239.2	2.4	1764	259945.2	0.6	1.3	2.42	1.2	14.7	22484.8	1.15	No
47995	-76.28	44.51	2.0	68.8	1159.2	3.5	1310.1	346710.2	0.5	1.5	1.83	1.0	5.0	116406.8	1.66	No
13626	-82.42	46.69	5.2	24.1	274.8	8.5	1021.5	158383.4	0.7	1.5	1.41	2.8	3.0	27331.5	0.70	No
33160	-78.81	45.20	1.7	6.9	41.1	6.6	551.1	45066	0.7	1.5	1.34	4.2	0.0	4107.2	0.38	No
46588	-76.90	45.29	7.2	20.0	149.2	7.4	807.8	183351.7	0.9	1.5	1.88	1.5	1.6	14910.4	0.50	Yes
47098	-76.46	45.14	12.8	37.4	531.6	4.2	939.1	338451.9	0.9	1.5	1.57	0.9	7.1	53403.4	0.68	No
36117	-78.37	46.18	0.3	43.3	1658.5	3.0	1062.5	520707.1	1.2	1.5	1.4	0.8	12.6	166724.9	0.55	Yes
18009	-80.60	48.06	1.2	29.2	1584.0	4.5	1839.3	250355.6	0.9	1.8	2.54	1.7	1.8	141111.1	0.60	No
46103	-77.60	45.50	2.7	15.2	99.1	6.3	755	123004.1	0.9	2.0	1.52	1.0	0.1	9636.4	0.43	No
45988	-77.88	45.54	2.3	19.3	146.6	8.0	1081.3	307822.2	1.8	2.0	1.77	4.1	0.0	7755.7	0.24	Yes
1681	-83.60	47.85	3.0	174.8	555.1	2.8	3013.2	3380434.6	1.8	2.0	2.83	0.8	4.8	29828.6	0.73	No
18726	-80.53	47.89	2.4	8.7	53.2	3.1	590.6	67715.7	0.8	2.0	1.33	1.6	0.0	4585.4	0.37	Yes
41377	-77.91	44.63	1.4	19.1	43.4	2.4	798.4	322882.3	1.8	2.1	1.39	1.4	29.1	2605.9	0.24	No
34150	-79.30	44.67	6.4	152.4	2994.4	1.0	2089.3	1404465.4	0.9	2.7	1.23	0.3	30.1	288235.3	1.37	No
34010	-79.75	44.83	1.0	50.1	517.9	1.1	2191.4	739876.6	1.5	2.9	2.79	1.1	26.5	34556.7	0.47	No
47942	-76.33	44.54	5.1	64.7	2016.0	4.2	2991.1	621702.6	1.0	3.0	3.4	2.4	14.4	170875	0.80	No
6115	-83.00	46.26	1.6	111.7	59801.8	5.9	2880.2	1923632	1.7	3.0	1.88	2.1	2.9	3424000	0.62	No
47078	-76.42	45.15	13.7	8.8	62.0	2.5	530	164623.8	2.1	3.0	1.25	2.7	14.5	3049	0.14	No
33942	-79.51	44.88	0.7	41.3	810.0	2.1	2339.2	732846.8	1.9	3.0	3.34	2.2	12.9	44549	0.34	No
33156	-78.97	45.20	2.1	7.0	33.5	8.3	501.2	147526.3	2.3	3.0	1.47	3.5	0.0	1485.7	0.12	No
32181	-79.52	46.82	0.7	135.3	1745.1	2.2	3552.1	3399080	1.6	3.0	4.11	2.1	12.4	110512.8	0.73	No
37919	-79.01	45.56	1.4	38.3	3543.9	6.8	2409.6	500154.2	1.4	3.0	2.67	2.0	5.0	251272.7	0.44	No

Lake ID	Longitude	Latitude	Chl-a (µg L^{-1})	Lake area (ha)	Catchment area (ha)	Catchment slope (°)	Lake fetch (m)	Lake volume (m^3)	Lake mean depth (m)	Lake max depth (m)	Development shoreline	Bathymetry slope (°)	Wetland cover (% of catchment)	Littoral zone in lake (ha)	Dynamic ratio	Lakes used for environmental controls of transitional lakes in Chapter 4
5977	-83.01	46.33	0.9	8.6	237.2	5.1	445.3	199767.8	2.3	3.3	1.26	4.4	1.7	9111.1	0.13	No
27339	-80.20	45.50	0.7	12.1	304.7	1.2	894.4	200634.4	1.8	3.4	1.81	2.5	21.2	15900	0.19	Yes
19040	-80.73	47.82	0.6	15.3	1301.6	5.0	1333.1	237554.1	1.7	3.7	2.2	2.2	0.3	77333.3	0.23	Yes
27368	-80.01	45.48	1.2	11.1	296.9	2.1	1172.7	197610.1	2.2	3.8	2.26	2.9	8.5	11351.4	0.15	No
16788	-81.53	46.16	0.5	32.6	286.9	5.1	2434.1	1130399.8	3.8	3.8	2.83	6.1	1.8	7539.8	0.15	Yes
36107	-78.47	46.18	1.3	10.5	39.6	3.1	535.8	137488.9	1.4	4.0	1.33	1.8	0.0	2988.7	0.23	Yes
5987	-83.24	46.33	0.4	46.8	684.8	4.3	1210.6	1157802	2.5	4.0	1.4	2.8	5.2	24235.3	0.27	No
46419	-77.35	45.38	3.0	8.1	36.9	9.9	557.4	113168.8	1.4	4.0	1.43	2.5	1.2	2690.9	0.20	No
10646	-83.45	47.36	0.3	53.8	1686.0	6.5	2264.6	1232736.8	2.4	4.0	2.91	2.7	9.8	92125	0.31	Yes
40863	-77.22	44.87	0.7	122.4	681.0	2.2	2278.1	4180612.2	3.1	4.5	2.45	1.5	15.3	26155.6	0.36	No
21993	-80.70	47.12	0.6	25.4	56744.2	5.3	1394.8	414793	1.8	4.7	1.99	5.7	10.6	0	0.28	Yes
46497	-77.08	45.35	1.9	28.4	256.3	5.6	856.6	1041147.5	3.5	4.8	1.78	2.4	3.1	7135.1	0.15	No
19586	-80.67	47.70	0.5	117.4	13740.1	4.0	2697.7	3974203.3	3.1	4.8	1.88	1.6	5.6	491555.6	0.35	No
5886	-82.49	46.40	0.6	20.8	8968.2	6.0	753.8	572491.9	2.3	5.0	1.5	2.8	4.2	398000	0.20	No
6091	-83.00	46.28	0.7	39.9	104.4	10.3	1726.6	1383587.4	3.6	5.0	1.8	4.3	2.2	2397.9	0.18	No
47235	-76.38	45.06	1.8	27.8	2431.5	2.5	1221.1	441009	1.5	5.0	1.55	1.7	12.2	185818.2	0.35	No
24266	-80.84	46.71	2.4	9.3	110.1	6.6	613.1	222274.6	2.7	5.0	2.03	4.7	0.3	5142.9	0.11	Yes
11035	-83.18	47.29	0.6	169.0	3852.6	6.2	4440.7	4535448.3	2.6	5.0	3.82	1.1	6.4	162909.1	0.50	No
23430	-81.16	46.86	0.5	111.7	1153.6	4.1	4778.5	3510041.7	2.7	5.0	4.74	2.8	10.3	56000	0.39	No
34042	-79.47	44.80	1.1	45.3	1101.7	1.5	2325.9	1756370.3	3.4	5.1	4.51	4.7	12.6	51122	0.20	No
6058	-82.98	46.31	0.5	264.2	1474.6	5.4	3861.1	9968518.3	3.6	5.4	2.21	2.4	6.5	31530.7	0.45	No
6100	-82.94	46.27	2.8	56.9	3747.6	5.6	1262.7	1256600.7	2.2	5.5	1.27	1.2	6.1	184266.7	0.34	No
6076	-83.00	46.29	1.0	45.9	714.2	4.6	1691.3	1535248.2	3.2	5.8	2.01	3.0	5.0	22875	0.21	No
27345	-79.90	45.47	1.9	1394.2	40626.2	3.2	8587.2	100440473	4.6	6.0	7.71	2.7	4.8	1845647	0.81	Yes
21469	-80.85	47.21	2.4	73.8	970.7	4.9	1539	2086297.1	2.7	6.0	1.65	1.3	2.2	36421.1	0.32	No
46204	-77.63	45.45	1.4	52.1	1237.0	6.9	1627.2	1991124.9	3.9	6.0	1.5	2.3	3.5	25904.8	0.19	Yes
37522	-78.07	45.64	6.6	68.9	58633.2	4.4	3181.6	1050672.6	1.4	6.1	2.88	3.3	9.6	5132000	0.59	No
10702	-82.08	47.37	0.3	104.3	1375.8	2.3	1699.8	1984055.4	1.9	6.1	1.73	0.9	9.9	81000	0.54	Yes
39605	-77.46	45.25	7.2	13.1	708.2	5.9	564.4	258666.1	2.0	6.1	1.13	2.9	2.7	39333.3	0.18	No
39677	-77.34	45.22	1.3	18.5	197.9	10.5	765.8	497630.7	2.7	6.1	1.31	3.0	1.7	8680.9	0.16	No
45915	-77.67	45.59	0.9	68.2	8042.8	4.6	2048.6	1899056.4	2.8	6.1	1.96	3.6	4.4	303500	0.29	No
16141	-82.00	46.39	0.5	44.2	726.8	5.3	921.5	1649260.5	3.7	6.1	1.18	1.7	15.3	21245.9	0.18	No
5916	-83.50	46.37	0.4	14.7	136.3	4.6	1372.8	506830.7	3.9	6.1	2.47	6.6	1.0	3703.7	0.10	Yes
40713	-77.27	44.92	0.7	7.7	232.0	3.8	692.7	165466.9	2.5	6.5	1.66	3.5	5.4	11151.5	0.11	No
30804	-79.82	47.10	0.8	32.9	695.2	3.6	1311	884378.9	2.8	6.5	1.69	2.9	6.0	31744.7	0.20	Yes

Lake ID	Longitude	Latitude	Chl-a (µg L^{-1})	Lake area (ha)	Catchment area (ha)	Catchment slope (°)	Lake fetch (m)	Lake volume (m^3)	Lake mean depth (m)	Lake max depth (m)	Development shoreline	Bathymetry slope (°)	Wetland cover (% of catchment)	Littoral zone in lake (ha)	Dynamic ratio	Lakes used for environmental controls of transitional lakes in Chapter 4
40405	-77.11	45.00	0.9	38.3	3233.9	4.1	1088.3	1220585.6	3.3	6.5	2.22	2.8	4.8	149000	0.19	Yes
40334	-77.94	45.05	0.8	38.1	281.6	7.3	1556.1	1279906.9	3.4	6.5	1.74	3.8	3.8	10459.3	0.18	Yes
41446	-77.82	44.57	0.6	265.1	82500.1	4.2	4190.1	11751726	3.8	6.5	2.81	2.3	8.1	2738667	0.43	Yes
44355	-77.67	46.16	0.3	74.1	1097.6	3.7	1818	2837129.5	3.9	6.5	2.52	3.1	6.2	33970.1	0.22	No
8912	-81.88	47.83	1.0	8.0	488.5	6.6	208.8	291921.3	4.0	6.5	1.51	6.4	1.4	19750	0.07	No
41472	-78.41	44.55	3.3	360.1	1808.9	2.1	4779.7	15687220	4.2	6.5	2.02	1.5	19.9	81226.1	0.45	No
32863	-78.84	45.29	0.4	58.9	890.1	4.9	1632.1	2580583.1	4.4	6.5	1.93	3.0	8.1	23939.4	0.17	No
33293	-78.87	45.16	1.0	23.6	148.3	6.7	1618.3	1050681.6	4.8	6.5	2.39	5.5	6.5	3320.8	0.10	No
27350	-80.05	45.46	1.0	219.7	2680.6	2.5	7945.5	12590324	5.1	6.5	5.66	5.6	9.8	86731.7	0.29	Yes
20623	-80.65	47.38	1.0	22.3	83.1	8.1	1181.1	1245451.1	5.9	6.5	1.82	5.5	2.1	1567.2	0.08	Yes
40711	-76.90	44.91	2.6	220.4	1637.7	4.3	3074.2	6044002.4	2.0	6.7	2.08	2.1	7.7	91703.7	0.74	Yes
33610	-79.09	45.03	0.3	21.2	207.2	5.1	849.6	585513.4	2.8	6.7	1.49	2.9	1.9	7456.3	0.16	No
32038	-79.80	46.85	0.7	15.6	50.2	3.7	661.4	513914.8	3.3	6.7	1.59	2.8	12.7	1602.6	0.12	No
41083	-78.21	44.79	0.5	40.0	401.2	3.8	1437.6	1252567	3.1	6.8	2.55	3.1	10.3	20560	0.20	No
27416	-80.16	45.42	0.5	38.2	167.5	3.0	1340.5	1033881.4	2.7	7.0	1.86	2.7	4.0	6193	0.23	Yes
34147	-79.33	44.69	3.6	656.8	5615.2	0.9	4005.6	33842622	4.2	7.0	1.84	0.7	34.1	133162.4	0.61	Yes
27140	-80.35	45.64	0.5	251.6	18514.8	3.0	10716	16417466	6.6	7.0	5.81	6.6	6.7	444000	0.24	No
10218	-83.36	47.48	0.3	231.5	10022.9	3.7	5165.3	5099200.4	2.1	7.0	4.71	2.3	7.4	583826.1	0.72	No
34083	-79.49	44.79	0.5	209.3	1081.7	1.5	5620.6	8217450.4	3.6	7.0	6.06	2.5	16.1	49927.5	0.40	No
40901	-77.06	44.85	1.5	10.7	56.7	4.7	649.8	660713.6	6.9	7.0	1.83	8.6	1.0	744.7	0.05	Yes
41453	-77.85	44.56	0.7	68.3	155.6	4.7	2161.2	2552545	3.6	7.2	2.28	2.7	4.4	5813.2	0.23	Yes
38119	-78.84	45.50	2.7	12.2	162.1	7.3	455.4	409258.3	3.5	7.2	1.21	5.5	3.8	7093.3	0.10	Yes
38410	-79.07	45.41	0.2	36.2	270.3	5.5	1039.9	2331283.2	6.5	7.2	1.73	4.2	2.0	8656.7	0.09	No
23121	-80.95	46.93	0.5	94.9	1414.6	5.6	2083.8	3871238.2	2.8	7.2	3.48	2.9	5.5	70388.1	0.35	No
32941	-78.65	45.26	0.5	56.0	246.5	6.2	2739.7	1438761.2	2.5	7.3	2.76	2.6	2.8	12722.5	0.30	Yes
5253	-82.36	46.61	0.7	57.1	265.2	7.0	1841.1	2378587.7	4.4	7.3	2.64	3.7	0.1	6325.6	0.17	Yes
40377	-76.94	45.00	3.3	12.5	93.8	6.8	885.8	259118.1	2.3	7.6	1.7	6.7	0.7	4992.5	0.15	No
45193	-77.58	45.89	0.4	31.9	384.6	5.5	1187.5	1175304.5	3.8	7.6	1.48	2.9	6.3	11373.5	0.15	No
33526	-78.93	45.06	0.6	89.7	1178.7	4.2	2851.8	5771834.4	6.1	7.8	4.07	6.0	6.6	40684.2	0.16	No
17902	-80.58	48.07	0.7	187.1	2930.1	3.4	3350	6667314.6	2.2	7.9	2.93	1.9	2.9	153125	0.62	No
27377	-79.99	45.48	0.8	165.3	2410.1	2.6	3671.9	6439957.5	2.7	7.9	4.36	2.9	4.3	90260.9	0.48	No
21639	-81.23	47.18	0.4	51.9	227.0	3.5	1530.8	1613129.5	3.2	7.9	1.99	2.4	4.4	6157.2	0.23	No
33800	-78.37	44.93	1.4	107.4	848.2	3.4	1973.3	6478490	5.6	8.0	2.19	3.2	2.4	41889.8	0.19	No
46418	-77.42	45.38	1.8	43.9	157.0	3.6	1099.5	707858.4	1.7	8.0	1.75	1.8	3.0	11785.7	0.39	Yes
40507	-76.83	44.97	2.7	8.6	83.6	4.8	486.9	274301.1	3.5	8.0	1.5	4.4	10.9	2796.1	0.08	No

Lake ID	Longitude	Latitude	Chl-a (µg L^{-1})	Lake area (ha)	Catchment area (ha)	Catchment slope (°)	Lake fetch (m)	Lake volume (m^3)	Lake mean depth (m)	Lake max depth (m)	Development shoreline	Bathymetry slope (°)	Wetland cover (% of catchment)	Littoral zone in lake (ha)	Dynamic ratio	Lakes used for environmental controls of transitional lakes in Chapter 4
33657	-78.45	45.00	1.2	74.1	7606.8	5.5	1441.9	2648708.4	3.6	8.0	1.45	1.8	5.6	260800	0.24	No
16914	-81.00	46.13	0.7	135.7	2369.6	2.5	4625	5892854.5	4.5	8.0	3.84	3.9	6.5	39719.3	0.26	No
17944	-80.36	48.08	1.1	99.7	3493.8	4.1	2720.4	7400642.3	7.2	8.0	1.8	4.3	7.6	89655.2	0.14	No
34016	-78.86	44.81	0.6	67.1	476.1	3.4	1513.8	4700390.4	4.8	8.0	1.96	4.2	4.9	6383	0.17	No
39929	-77.34	45.14	2.3	10.4	46.6	7.4	715.8	601847.6	6.1	8.0	1.6	7.5	1.9	1148	0.05	Yes
15840	-81.47	46.48	0.6	101.0	9163.0	6.2	1683.9	7318376.9	7.2	8.0	1.4	2.8	11.1	170181.8	0.14	No
31411	-79.88	46.98	0.9	53.2	1195.2	4.1	1904.5	2705628.6	5.2	8.0	3.2	4.4	4.7	46090.9	0.14	No
33993	-79.63	44.84	0.6	62.7	177.8	2.7	1672.7	1937156.9	2.9	8.1	3.23	2.8	25.3	9461.8	0.27	No
5680	-82.23	46.49	4.8	15.6	817.0	6.9	979.9	404749.3	2.8	8.2	2.05	7.6	3.3	28421.1	0.14	No
11017	-82.65	47.29	0.5	63.8	501.4	7.0	2195.7	2558950.5	3.7	8.2	3.25	3.0	4.8	24724.4	0.22	No
3897	-84.36	47.11	0.5	102.2	1090.8	7.3	2276.5	9655720.4	9.0	8.3	3.1	7.3	4.9	17702.1	0.11	No
40942	-77.57	44.85	1.3	89.7	1283.7	3.4	2462.8	2980322.6	3.1	8.5	1.96	2.0	2.1	51085.7	0.31	No
32410	-79.35	46.75	0.7	30.9	69.1	0.7	913.2	1081601.6	3.6	8.6	2	3.4	0.5	2514.5	0.15	No
3937	-84.62	47.11	0.9	105.8	217.5	6.4	1983.5	3706271.2	3.1	8.8	2.18	2.0	4.2	12262.8	0.33	No
19245	-80.73	47.77	1.0	48.2	10697.3	4.4	3055.1	1691409.9	3.4	8.8	3.54	5.2	7.2	351200	0.20	No
27477	-79.85	45.36	1.4	134.8	2247.5	3.3	2964.9	7581053	3.6	8.8	3.85	3.7	5.9	82800	0.32	No
33872	-79.58	44.91	0.5	21.1	153.2	3.7	1063.5	1050559.7	4.3	8.8	2	4.7	21.8	4927.5	0.11	No
31701	-79.80	46.92	0.5	37.8	1468.0	4.8	1733.3	1572897.2	4.4	8.8	2.42	5.0	2.2	46307.7	0.14	Yes
47102	-76.41	45.14	3.1	31.2	421.7	2.2	1222.1	1557132.6	4.8	8.8	2.31	4.6	5.2	13027	0.12	Yes
5389	-82.85	46.56	0.4	78.9	353.0	6.6	3048.4	3816405.7	5.0	8.8	3	5.7	1.4	8017.9	0.18	Yes
41196	-78.15	44.74	0.7	96.3	428.2	4.2	3452.2	5722949.6	5.0	8.8	3.89	4.7	9.9	14151.1	0.20	No
41380	-78.20	44.64	0.8	33.4	2430.5	3.2	1403.5	1731751.9	5.3	8.8	2.17	5.3	5.7	53428.6	0.11	No
20556	-80.66	47.40	0.4	38.2	2320.1	7.0	1067.7	2164772.6	5.5	8.8	1.4	3.5	1.4	83750	0.11	No
33931	-79.27	44.89	0.8	85.1	8460.2	3.0	3088	6191023.4	6.3	8.8	3.49	6.1	6.4	180800	0.15	No
18697	-81.01	47.89	0.7	13.0	138.5	3.4	841.4	806478.4	6.6	8.8	1.53	6.5	0.0	2978.7	0.05	No
20635	-80.63	47.38	0.6	85.8	1178.0	8.0	3071.6	6966029.7	6.7	8.8	2.58	4.6	2.0	23835.6	0.14	No
47049	-76.48	45.18	1.8	8.9	265.8	4.2	514.4	598414	7.1	8.8	1.29	6.5	1.6	2666.7	0.04	No
26759	-79.87	45.91	0.3	30.8	342.7	4.5	1079.9	2685162.9	8.9	8.8	1.49	5.6	9.3	5200	0.06	No
47515	-76.22	44.84	35.5	660.1	4327.4	1.9	7065.7	63110680	7.8	9.0	3.76	3.1	19.2	121699.3	0.33	Yes
31826	-79.71	46.90	0.9	83.4	750.1	6.1	2970.7	3294186	4.0	9.0	2.17	3.3	7.4	23603.6	0.23	No
38061	-78.82	45.51	0.5	16.6	445.8	7.8	736.4	1024359.7	5.4	9.0	1.55	5.4	2.1	11567.6	0.08	No
37133	-78.14	45.77	0.5	61.1	238.2	7.3	1382	3611446.2	5.8	9.0	1.32	5.6	4.4	6484.4	0.13	No
47589	-76.21	44.81	1.3	80.5	1244.2	2.8	2065.3	3414418.9	4.2	9.0	2.94	4.0	12.5	50400	0.21	Yes
21112	-80.82	47.27	0.8	240.1	5253.0	6.4	4770.6	17615197	7.2	9.0	3.14	3.5	5.3	133217.4	0.22	No
2008	-84.71	47.76	0.6	31.9	1077.4	7.0	1448	758896.9	2.4	9.1	3	4.3	3.4	61733.3	0.24	Yes

Lake ID	Longitude	Latitude	Chl-a (µg L^{-1})	Lake area (ha)	Catchment area (ha)	Catchment slope (°)	Lake fetch (m)	Lake volume (m^3)	Lake mean depth (m)	Lake max depth (m)	Development shoreline	Bathymetry slope (°)	Wetland cover (% of catchment)	Littoral zone in lake (ha)	Dynamic ratio	Lakes used for environmental controls of transitional lakes in Chapter 4
2625	-84.70	47.57	0.7	44.6	1561.5	7.7	1846.3	1046893.8	2.4	9.1	3.26	2.3	0.3	71034.5	0.28	Yes
5168	-83.51	46.61	0.6	33.2	1661.4	7.4	1306.8	772315.9	2.4	9.1	2.03	2.4	2.3	102200	0.24	No
45703	-77.69	45.67	1.9	47.8	41804.0	4.8	1274.6	1255046.5	2.4	9.1	1.58	2.7	7.9	2404000	0.29	No
19556	-80.69	47.69	1.4	21.6	68.2	8.3	748.7	547054.8	2.6	9.1	1.57	2.2	0.0	3343.8	0.18	Yes
45681	-77.70	45.67	2.9	79.5	40652.9	4.8	1519.2	2210047.1	2.6	9.1	1.55	2.7	7.9	1898000	0.34	No
46097	-77.62	45.50	1.7	10.8	1093.1	6.1	665.6	327378.1	2.8	9.1	1.44	2.9	9.3	38800	0.12	No
5947	-83.49	46.35	0.3	35.1	440.0	6.4	1260.1	1769685.6	5.2	9.1	1.69	6.9	7.8	7850	0.11	Yes
17078	-81.40	46.02	0.5	6.5	23.0	5.8	532.6	343482.5	6.8	9.1	1.76	9.4	2.2	70.9	0.04	No
21239	-80.81	47.25	1.2	86.8	11827.7	6.9	2084.2	2118124.8	2.5	9.2	2.59	1.6	4.4	521142.9	0.37	Yes
40718	-77.40	44.92	0.7	143.2	1933.8	4.0	4171.6	8110398.3	5.6	9.2	3.63	3.6	10.3	35729.7	0.21	Yes
40247	-76.79	45.06	1.0	264.9	1336.8	5.8	4861.1	16985254	6.1	9.3	2.89	6.4	4.0	43838.4	0.27	No
33274	-78.95	45.17	1.2	15.4	1583.8	6.6	885.5	430718.3	3.9	9.4	1.96	3.8	0.1	57600	0.10	No
36311	-78.39	46.09	0.2	239.2	4916.6	3.9	4743.7	15744140	6.5	9.4	3.02	3.3	11.5	139510.2	0.24	Yes
33751	-78.37	44.96	0.6	208.6	3317.1	3.9	2565.6	25043458	8.0	9.4	1.86	4.3	6.2	35111.1	0.18	Yes
1817	-84.71	47.84	0.3	155.4	1479.0	8.2	3614.3	12981142	8.4	9.4	2.14	4.5	3.1	18552.4	0.15	No
40561	-77.29	44.97	1.2	46.4	1231.8	3.8	1169.7	1530913	3.3	9.8	1.55	2.1	7.0	38000	0.21	Yes
6015	-83.15	46.31	0.6	88.9	2323.3	5.3	1300.6	1412819.5	2.3	10.0	1.3	1.9	6.8	90315.8	0.41	No
25288	-79.94	46.66	1.4	14.5	275.0	2.9	795.3	332732.8	2.4	10.0	2.06	3.2	17.7	16301.9	0.16	No
31842	-79.51	46.89	0.8	69.5	2585.7	2.9	2703.8	1887037.7	2.8	10.0	2.61	2.7	7.8	121037	0.30	Yes
21199	-80.79	47.26	0.8	25.9	5327.5	7.8	855.6	905150.3	3.6	10.0	1.55	2.6	4.6	176800	0.14	No
46215	-77.75	45.45	0.9	113.4	1077.4	6.9	1898	8188179.4	7.2	10.0	1.72	3.5	3.3	28533.3	0.15	Yes
21135	-81.38	47.27	0.3	6.0	14.6	1.3	358.4	488489.4	8.4	10.0	1.09	9.3	0.0	194.2	0.03	No
38272	-78.99	45.46	0.2	159.2	2309.2	4.9	3902.1	10945529	6.7	10.0	2.72	4.4	3.8	58898.6	0.19	No
27659	-79.95	45.17	1.5	137.1	6941.6	3.5	3625.5	3915161.2	2.7	10.0	3.19	2.0	9.9	249000	0.43	No
33098	-78.98	45.22	0.6	36.9	148.5	6.1	1393.2	2874352.7	7.8	10.0	2.23	7.1	0.0	3096.8	0.08	No
37920	-78.11	45.54	0.7	132.6	5904.9	6.6	3032.9	22521641	6.2	10.0	2.42	5.2	5.8	190545.5	0.19	Yes
2546	-83.73	47.57	0.8	15.2	42.2	2.9	837.2	862678.6	6.2	10.0	1.65	6.7	1.3	657.4	0.06	Yes
45666	-77.61	45.68	2.2	52.5	9838.8	5.1	2289.6	2153297.3	4.0	10.0	2.5	5.3	8.2	418400	0.18	Yes
47956	-76.37	44.52	2.2	240.5	4261.1	4.9	5089.3	18173317	5.7	10.0	3.29	3.7	11.3	112285.7	0.27	Yes
23409	-80.79	46.87	2.3	160.7	2041.8	5.9	2817.2	11078853	5.9	10.0	3.67	4.1	3.2	84911.4	0.21	No
5589	-83.49	46.50	0.4	33.1	168.6	4.6	1037.7	2052232	6.4	10.0	1.8	5.8	3.4	4285.7	0.09	No
33329	-78.94	45.15	0.7	11.8	88.6	10.6	779.4	523410.6	4.7	10.1	1.5	6.3	0.0	2406	0.07	No
34009	-78.44	44.82	0.4	177.3	663.9	3.1	2626.4	26114519	12.2	10.1	2.37	6.6	29.4	11191	0.11	No
33531	-79.01	45.06	0.3	14.6	823.8	3.7	779.5	714977.8	5.2	10.2	1.65	5.4	0.1	14888.9	0.07	No
27048	-80.37	45.70	0.4	339.9	2824.7	1.6	5719.8	14357568	3.7	10.2	6.51	2.9	8.9	134400	0.50	No

Lake ID	Longitude	Latitude	Chl-a (μg L^{-1})	Lake area (ha)	Catchment area (ha)	Catchment slope (°)	Lake fetch (m)	Lake volume (m^3)	Lake mean depth (m)	Lake max depth (m)	Development shoreline	Bathymetry slope (°)	Wetland cover (% of catchment)	Littoral zone in lake (ha)	Dynamic ratio	Lakes used for environmental controls of transitional lakes in Chapter 4
47566	-76.26	44.82	1.2	12.0	1642.1	1.6	590.9	495417.5	4.5	10.2	1.3	4.3	23.0	38857.1	0.08	No
19356	-80.63	47.75	0.5	27.5	9668.9	4.0	1244.7	1339076.5	5.0	10.2	1.8	5.2	7.9	224000	0.10	Yes
1729	-83.42	47.82	0.7	9.4	220.0	4.0	622	510572.2	5.9	10.2	1.69	7.1	3.1	4837.2	0.05	No
22160	-81.28	47.09	0.5	32.4	287.3	6.3	1023.3	1381966.9	6.4	10.2	1.44	5.1	1.5	3787.6	0.09	Yes
25967	-80.58	46.26	1.3	27.7	90.8	4.1	1361.1	1731824.8	6.6	10.2	2.42	7.8	2.9	1809.8	0.08	No
41159	-78.26	44.76	0.2	160.8	918.8	4.7	2489.1	14082426	6.6	10.2	2.1	3.6	5.6	12571.4	0.19	Yes
46464	-77.09	45.36	1.1	14.6	359.6	7.5	987.9	959792.9	6.8	10.2	1.74	7.4	3.8	10634.1	0.06	No
4273	-84.15	46.83	0.8	21.4	294.0	9.7	908.4	1477769.5	7.1	10.2	1.51	6.3	0.0	5972.6	0.07	No
32229	-79.29	46.80	2.3	25.5	156.9	3.2	926.2	2065196.2	8.2	10.2	1.46	5.6	3.9	3950.6	0.06	Yes
34149	-78.74	44.68	0.5	756.1	5086.7	1.8	6425.6	73450074	8.3	10.2	2.03	1.5	31.3	75516.8	0.33	No
16931	-81.59	46.11	0.4	73.1	427.5	5.7	2348.9	7901838.1	10.6	10.2	2.57	8.7	0.9	8046.8	0.08	No
20379	-80.71	47.45	1.8	159.3	14550.2	6.9	4225.9	6878379.2	4.4	10.2	3.78	3.2	3.0	764727.3	0.29	No
32439	-79.40	46.74	0.4	168.1	1391.9	0.8	2542.2	13975907	8.2	10.3	1.96	3.2	7.9	29487.6	0.16	No
34031	-79.65	44.82	9.3	112.3	1567.0	3.4	2792.7	5275134.7	4.7	10.4	2.46	2.1	16.9	43722.2	0.23	No
33548	-79.89	45.07	0.3	35.7	890.5	2.7	1212.6	1837465.5	5.2	10.4	1.97	3.8	3.9	21100	0.11	No
2125	-83.62	47.70	1.0	31.3	232.0	1.8	1058.4	2021681.6	6.8	10.5	2.1	6.2	8.1	3763	0.08	Yes
33255	-78.95	45.17	0.9	12.2	1435.1	6.5	600.3	432894	2.9	10.7	1.5	2.6	0.1	36888.9	0.12	No
33202	-78.97	45.19	0.7	17.4	124.6	8.5	911.6	552615	3.4	10.7	1.72	3.9	0.0	4086.3	0.12	No
38154	-78.96	45.49	0.3	33.5	227.0	4.3	1051.8	1140046.1	3.5	10.7	1.65	4.1	0.0	9687.1	0.17	No
39949	-76.91	45.12	8.7	23.2	335.8	4.2	1626.7	308161.3	1.5	11.0	2.75	3.0	4.6	26260.9	0.32	Yes
40060	-77.31	45.11	6.3	17.2	251.4	7.6	1473.4	377768.5	2.4	11.0	2.35	4.5	1.5	15768.1	0.17	No
24253	-81.13	46.71	1.1	6.4	122.7	10.5	588.4	196565.6	3.5	11.0	1.5	7.9	6.5	4000	0.07	No
40936	-77.88	44.85	1.0	16.4	139.2	4.0	707.5	758013.5	4.5	11.0	1.33	3.2	1.9	3593.2	0.09	Yes
33680	-78.91	45.00	0.5	47.8	269.6	3.8	1332.9	4273061.5	8.9	11.0	2.65	6.7	8.0	4610.2	0.08	No
5478	-82.98	46.53	5.6	22.3	241.3	8.3	3122.2	16094377	9.2	11.0	9.16	6.0	4.8	37376.3	0.05	No
2809	-84.35	47.50	0.6	86.9	537.9	5.8	1949.2	8404632.9	9.6	11.0	2.36	5.7	4.4	10543.2	0.10	No
11166	-83.49	47.25	0.3	31.6	245.9	6.6	1448.6	2144862.9	6.9	11.2	1.94	6.5	1.7	3907	0.08	No
26594	-79.89	46.00	2.0	65.3	3823.3	3.5	3764.8	2240619.4	3.1	11.3	4.94	5.0	10.4	145411.8	0.26	No
40124	-76.80	45.08	2.6	6.3	39.9	5.6	633.8	200566.5	3.5	11.3	1.61	6.5	0.6	1645.6	0.07	Yes
2189	-84.88	47.72	0.3	43.5	237.6	10.2	1457.8	3139591.3	7.2	11.4	2.1	5.2	0.0	6491.8	0.09	Yes
6047	-82.92	46.30	0.4	26.1	233.8	5.9	871.3	1220176.2	4.6	11.8	1.69	5.0	1.6	5535.7	0.11	No
33910	-79.33	44.89	0.9	26.3	1170.8	3.0	1412.7	1078475.2	4.4	12.0	3.98	5.2	6.7	57454.5	0.12	No
33031	-78.69	45.23	1.0	21.3	81.4	7.0	891.2	955565.7	4.6	12.0	1.62	4.8	2.1	2381.7	0.10	No
34047	-79.71	44.82	0.5	398.8	184783.7	1.7	4089.3	34141145	6.4	12.0	3.86	2.9	15.2	4746000	0.31	No
33073	-78.22	45.20	0.6	220.9	977.1	5.9	3496.4	15945319	7.2	12.0	2.65	3.8	2.0	17415.9	0.21	Yes

Lake ID	Longitude	Latitude	Chl-a (µg L^{-1})	Lake area (ha)	Catchment area (ha)	Catchment slope (°)	Lake fetch (m)	Lake volume (m^3)	Lake mean depth (m)	Lake max depth (m)	Development shoreline	Bathymetry slope (°)	Wetland cover (% of catchment)	Littoral zone in lake (ha)	Dynamic ratio	Lakes used for environmental controls of transitional lakes in Chapter 4
46582	-76.84	45.30	1.9	102.7	34888.0	4.7	2525.8	10907266	9.6	12.0	1.74	5.1	8.2	996000	0.11	No
44778	-77.56	46.04	0.7	62.1	187.2	4.2	1742.3	2692312.2	4.2	12.0	1.87	3.0	9.8	8108.4	0.19	No
22057	-81.24	47.11	0.5	13.2	271.2	5.1	751.2	1066645.9	8.3	12.0	1.89	11.3	2.9	4244.9	0.04	No
21288	-80.77	47.24	0.2	114.8	784.7	8.0	4089.8	12677917	11.2	12.0	2.6	6.8	0.6	10904.1	0.10	No
48016	-76.50	44.50	1.3	22.6	62.9	5.3	962.1	2722166.1	11.9	13.0	1.78	9.3	3.9	1822.2	0.04	No
47743	-76.56	44.66	1.3	95.1	28375.2	2.7	1924.7	6681891	7.1	13.0	2.13	3.9	8.5	842666.7	0.14	No
40101	-76.91	45.08	3.3	40.7	202.9	4.6	2208.9	822232.9	2.0	13.1	2.91	4.1	2.8	14328.4	0.32	No
27406	-80.19	45.44	0.6	180.4	1477.3	3.2	4427.8	9293776.5	3.2	13.1	3.37	3.6	7.7	50557.4	0.42	Yes
27323	-79.92	45.51	0.5	20.6	125.0	2.5	880.7	1118247.5	5.3	13.4	1.48	4.4	0.0	3757.6	0.09	No
2575	-84.86	47.59	0.3	27.6	1148.4	8.8	1354.3	1460162.1	5.5	13.7	1.77	5.1	0.0	28500	0.10	Yes
6049	-83.29	46.30	0.4	141.8	384.6	5.3	2636.1	8201145.9	5.7	13.7	1.75	3.0	0.0	9105.7	0.21	No
12560	-82.50	46.99	0.5	663.6	31747.0	4.0	8401	70006159	8.9	14.1	5.34	4.4	4.5	696190.5	0.29	No
5279	-83.59	46.59	0.2	128.7	2448.0	8.2	2408.9	13359060	10.9	14.8	1.77	4.2	1.5	4830.2	0.10	No
9255	-82.04	47.72	16.8	1258.8	6508.2	3.5	8225.9	50208806	2.9	15.0	2.87	0.8	5.3	238549.2	1.22	No
39380	-78.02	45.36	0.3	313.6	5343.7	5.1	4185.6	27400977	8.7	15.0	2.15	4.7	3.7	132067.8	0.20	No
46314	-77.36	45.43	1.2	87.7	1866.8	5.0	2484.3	6491663.8	7.1	15.6	2.15	5.3	5.4	61872.3	0.13	No
21605	-80.96	47.19	0.5	46.8	267.6	7.5	1839.8	2502527.3	5.4	16.0	1.85	4.3	0.0	7474.3	0.13	No
47591	-76.69	44.80	1.2	8.5	292.5	2.7	455.6	759131.1	9.8	16.0	1.22	8.6	6.1	1379.3	0.03	Yes
40444	-76.93	44.99	7.3	11.7	169.2	5.9	597.6	646681.8	5.9	16.0	1.7	8.7	4.9	3652.2	0.06	Yes
47066	-76.51	45.17	3.5	22.4	745.9	4.6	820.7	2677484.1	12.3	16.0	1.34	8.6	3.2	6666.7	0.04	No
4101	-84.62	47.03	0.2	148.5	537.9	11.6	2993.1	17451483	11.8	16.0	2.08	6.1	0.6	6188.4	0.10	No
5800	-82.97	46.44	0.2	85.9	3088.2	7.3	3065	5426888.7	6.5	16.2	2.22	7.9	4.3	34428.6	0.14	No
40067	-77.38	45.11	0.4	153.3	1682.1	5.6	2668.5	21099217	13.4	16.2	2.44	6.6	2.6	16615.4	0.09	No
32382	-79.25	46.75	1.3	10.2	233.5	2.1	577.2	536571.5	5.6	16.3	1.28	6.1	13.0	6232.6	0.06	No
19615	-80.71	47.68	0.5	15.5	62.4	5.6	915.5	737973.2	4.9	16.5	1.8	5.1	0.0	1879.5	0.08	No
9278	-83.39	47.72	0.2	12.4	74.9	1.7	590.6	659551.4	5.6	16.5	1.53	6.8	0.4	1939.4	0.06	No
26756	-80.28	45.91	0.4	15.8	268.1	1.2	739.7	1074894.7	7.1	16.5	1.43	6.6	5.6	4339	0.06	No
33967	-79.46	44.87	0.8	317.8	7000.1	2.7	2752.1	17199627	3.0	16.8	1.7	2.1	5.9	298755.6	0.59	No
47940	-76.47	44.54	0.5	42.1	804.5	4.1	1269.6	2982537.1	7.2	16.8	2.33	5.4	3.7	14384.6	0.09	Yes
5824	-82.95	46.42	0.1	91.6	454.0	8.2	1796.9	13571090	14.3	17.0	1.5	6.6	0.8	3465.3	0.07	No
15004	-81.29	46.67	0.5	43.3	137.6	6.8	1306.8	4673017.6	11.1	17.0	1.84	7.8	7.5	1460.3	0.06	Yes
27325	-80.08	45.51	0.5	71.5	2081.8	3.1	2038.7	5754173.3	6.8	17.0	2.8	7.1	8.8	80470.6	0.12	Yes
32957	-78.41	45.25	0.5	78.7	833.3	4.6	1534.8	11031808	13.8	17.0	1.79	8.1	4.2	14170.2	0.06	No
5910	-82.98	46.38	0.2	54.9	224.5	11.0	1599.6	5454021.4	10.1	17.2	1.78	8.5	1.4	2377	0.07	No
2383	-84.61	47.65	1.0	15.3	243.6	6.4	1337.4	485267.5	3.5	17.5	2.49	6.2	0.0	12190.5	0.11	No

Lake ID	Longitude	Latitude	Chl-a (µg L^{-1})	Lake area (ha)	Catchment area (ha)	Catchment slope (°)	Lake fetch (m)	Lake volume (m^3)	Lake mean depth (m)	Lake max depth (m)	Development shoreline	Bathymetry slope (°)	Wetland cover (% of catchment)	Littoral zone in lake (ha)	Dynamic ratio	Lakes used for environmental controls of transitional lakes in Chapter 4
33946	-78.74	44.87	1.2	49.5	303.4	5.5	1186.3	2302311.3	4.7	17.7	1.65	3.0	5.0	12736.2	0.15	No
21503	-80.99	47.21	0.6	98.7	1213.4	8.5	3149	6604114.7	6.8	18.0	2.87	4.2	4.3	37086.4	0.15	No
22773	-80.77	46.98	0.4	504.1	4170.7	5.0	4233.8	53160156	7.4	18.0	2.52	3.6	4.0	105586.8	0.30	Yes
39086	-78.23	45.07	0.7	221.9	5078.9	6.1	2928	33826241	15.3	18.0	1.75	5.2	4.1	40909.1	0.10	No
5642	-83.70	46.48	0.2	26.9	117.2	10.7	684.3	2943626.4	11.2	18.0	1.15	7.4	0.0	1659.4	0.05	No
19204	-80.65	47.77	0.7	121.9	26446.6	4.1	3615.5	14472315	12.0	18.0	2.87	7.8	6.3	436800	0.09	No
40074	-76.89	45.09	0.9	27.7	100.5	7.3	1388.8	2087089.5	8.3	18.5	2.25	8.9	0.4	797.1	0.06	No
23754	-81.24	46.82	0.6	157.4	1463.0	5.3	2914.6	13330161	6.4	18.5	3.39	4.9	10.0	45666.7	0.20	Yes
27309	-80.21	45.52	0.5	178.8	2187.9	1.2	4654.7	9768876.7	3.8	18.6	5.42	3.2	20.9	71756.1	0.35	No
47981	-76.48	44.52	1.2	32.9	1532.7	3.9	1577.3	2104655.3	5.6	18.9	2.59	7.4	9.7	56000	0.10	No
38471	-78.98	45.39	0.3	21.1	407.1	8.5	999.1	1379658.5	6.7	18.9	1.53	6.2	0.5	9923.1	0.07	Yes
33399	-78.59	45.12	1.0	113.9	24082.2	5.2	1812.6	8183235.3	7.2	19.0	1.3	2.4	5.8	360000	0.15	No
40754	-77.17	44.91	0.4	87.4	3326.8	4.0	1911	11103627	12.6	19.0	1.6	6.4	5.7	52153.8	0.07	Yes
40178	-76.93	45.07	0.6	111.9	713.6	5.0	2778.7	8191692.7	7.3	19.8	2.74	5.3	3.9	13936.3	0.14	No
21444	-80.92	47.21	0.4	96.3	453.6	7.2	2749	6366801.8	5.3	20.0	2.69	4.5	1.2	11226.4	0.19	No
23738	-80.89	46.83	0.6	16.1	56.4	4.2	670.2	1172663.5	7.4	20.0	1.58	6.8	0.0	1052.6	0.05	Yes
46406	-76.98	45.40	1.9	618.4	26427.4	4.8	5230	36125312	4.6	20.1	3.49	1.6	8.2	1310435	0.54	No
41423	-77.91	44.60	0.4	327.9	1216.6	2.9	3347.6	27111450	6.1	20.2	2.11	3.4	24.1	39807.4	0.30	No
1969	-84.17	47.77	0.7	173.7	1549.8	7.3	2960.2	9917117.4	4.0	20.5	3.93	4.0	4.3	58000	0.33	No
38414	-78.97	45.41	0.3	18.1	74.8	5.5	963.2	1097299.2	6.3	21.0	1.83	7.0	0.1	1933.9	0.07	No
39568	-76.89	45.24	1.8	5.2	91.2	6.9	439.4	485800.7	10.8	21.0	1.4	15.0	6.6	905.3	0.02	No
39604	-77.01	45.24	1.5	170.0	907.3	5.9	1397.2	4155734.9	12.2	21.0	0.8	9.2	1.9	2866.3	0.11	No
46636	-76.71	45.25	0.9	17.7	51.8	6.6	906.9	2061429.7	11.9	21.0	1.94	9.2	7.1	985.3	0.04	No
46080	-77.76	45.51	0.5	271.4	2681.2	7.2	2973.9	35712748	13.1	21.0	1.69	4.6	3.2	52356.4	0.13	No
34074	-78.80	44.80	0.7	189.6	110129.0	4.7	2524.6	13948270	7.2	21.9	2.61	3.5	4.7	2134000	0.19	No
5182	-82.37	46.63	0.3	46.5	331.8	7.3	1633	3901740.8	8.6	22.0	2.51	7.6	2.7	5514.3	0.08	Yes
39698	-77.00	45.21	0.7	133.6	906.2	6.8	2878.6	25856953	19.5	22.0	2.26	9.9	4.9	6775.5	0.06	No
33164	-79.05	45.20	0.4	74.3	120.9	6.3	1368.4	6717722.7	9.0	22.9	1.57	4.2	0.0	1928.3	0.10	No
47623	-76.51	44.78	1.2	53.2	207.9	5.9	1439	5963822.1	11.3	23.8	1.72	6.2	7.4	2843.8	0.06	No
5674	-82.38	46.49	0.3	24.9	693.4	5.2	973.7	2359911.8	9.4	24.0	2.14	4.0	6.4	17111.1	0.05	Yes
39595	-76.74	45.23	1.6	14.6	250.0	6.6	1020	700070	4.4	24.4	2.04	6.3	4.2	9310.3	0.09	No
5170	-83.59	46.60	0.2	30.5	123.7	10.7	1147	2681174.6	9.0	24.4	1.84	8.5	0.0	2292.7	0.06	No
4309	-83.95	46.78	0.6	196.7	384.7	13.9	2314.3	21490761	10.7	36.6	1.35	3.9	1.0	8641.9	0.13	No
40467	-77.88	45.02	0.7	42.7	374.5	5.7	1693.4	4852956.8	11.4	37.0	2.1	8.8	5.6	7754.4	0.06	No
40232	-77.01	45.05	0.6	34.9	939.0	6.8	1371.5	5543611.1	17.6	39.0	1.68	15.6	4.3	7135.1	0.03	No

Lake ID	Longitude	Latitude	Chl-a (μg L^{-1})	Lake area (ha)	Catchment area (ha)	Catchment slope (°)	Lake fetch (m)	Lake volume (m^3)	Lake mean depth (m)	Lake max depth (m)	Development shoreline	Bathymetry slope (°)	Wetland cover (% of catchment)	Littoral zone in lake (ha)	Dynamic ratio	Lakes used for environmental controls of transitional lakes in Chapter 4
27132	-80.23	45.65	0.3	235.9	3252.9	3.6	5216	35016148	14.1	40.5	0.88	8.1	4.8	37260.3	0.11	Yes
33542	-78.32	45.05	0.3	260.6	1738.3	6.6	3236.2	50626835	19.2	41.0	1.95	6.4	4.6	15040	0.08	No
16910	-81.43	46.10	1.6	874.0	10327.8	8.2	10386	169174344	11.7	45.0	8.97	8.8	3.6	255435.3	0.25	Yes

Appendix F: Non-stationary signals in time series of modeled Chlorophyll-*a*.

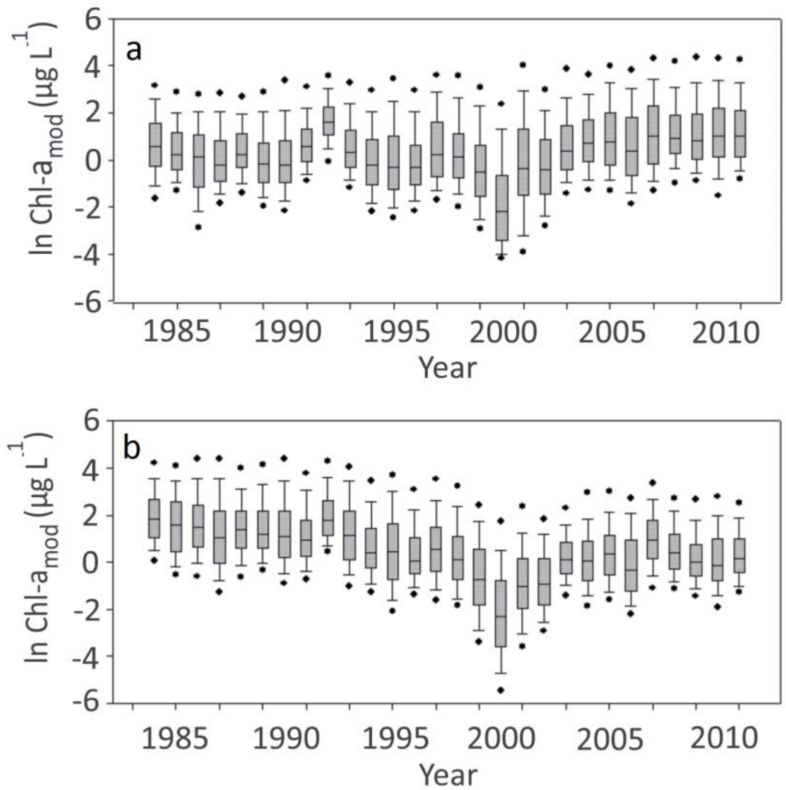

Figure F.1 Significant ($p < 0.05$) non-stationary signals in ln Chl-a$_{mod}$ time series (1984–2011) for: (a) positive trending lakes ($n = 500$), and (b) negative trending lakes ($n = 561$).

Appendix G: Sensitivity analysis to identify the most appropriate size of mowing window standard deviation.

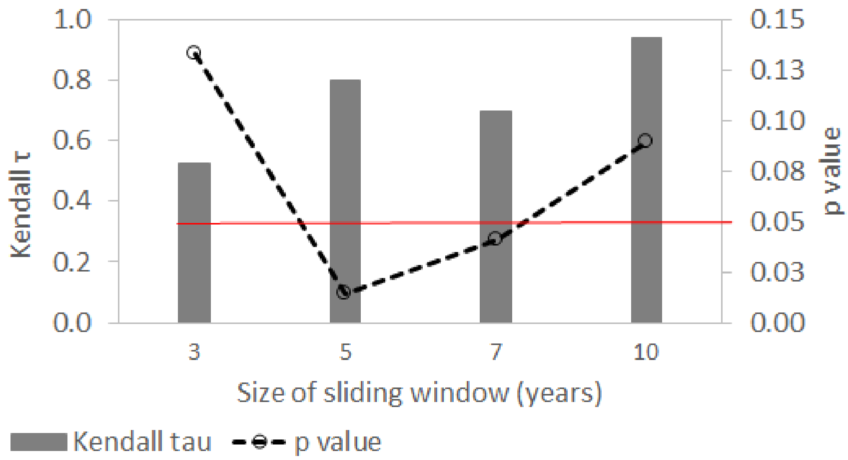

Figure G.1 Averaged results of the sensitivity analysis of 95% of 2,000 randomly selected lakes to identify the most appropriate size of mowing window standard deviation (SD_{mv}, years) measured by the Kendall's statistics (Kendall tau (τ) and p values).

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Yeung C.Y. A., **A. Paltsev**, A. Daigle, P. N. Duinker, I. F. Creed (2018) Atmospheric change as a driver of change in the Canadian boreal zone. Environmental Reviews. doi: org/10.1139/er-2018-0055

Presentations:

1. Association for Sciences in Limnology and Oceanography (ASLO)–summer meeting, Victoria, Canada, June 2018 (oral presentation).
2. The 2nd Interdisciplinary Freshwater Harmful Algal Blooms Workshop (IFHAB), Toronto, Canada, April 2018 (oral presentation).
3. American Geophysical Union (AGU)–fall meeting, New Orleans, USA, December 2017 (poster presentation).
4. The Great Lakes Research (IAGRL)–59th annual conference, Guelph, Canada, June 2016 (oral presentation).

5. The Water Initiative for the Future (WatIF): international graduate student conference, Kingston, Canada, July 2016 (oral presentation).
6. Symposium on Sustainable Agricultural Systems in the Great Lakes Basin, London, Canada, April 2016 (poster presentation).
7. American Geophysical Union (AGU)–fall meeting, San Francisco, USA, December 2015 (co-author, oral presentation).
8. Fallona Family Interdisciplinary Showcase, London, Canada, December 2015 (poster presentation).
9. Canadian Network for Aquatic Ecosystem Services (CNAES)–annual meeting, Sault Ste. Marie, Canada, April 2015 (oral presentation).
10. North American Lake Management Society (NALMS)–35th annual international symposium, Saratoga Springs, New York, USA, November 2015 (oral presentation).
11. Canadian Network for Aquatic Ecosystem Services (CNAES)–annual meeting, Montreal, Canada, April 2014 (oral presentation).
12. International Botanical Conference–12th annual meeting, S.-Petersburg, Russia, 2004 (co-author, oral presentation).
13. International Associations for Vegetation Science (IAVS)–45th symposium: Vegetation dynamics in space and time, Porto Alegre, Brazil, March 2002.