An Examination of Wetland Conversion and Resulting Effects on Landscape Connectivity in Southern Ontario Municipalities.

by

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Author's Declaration

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Statement of Contributions

The two manuscripts contained within this thesis are co-authored, editorial contributions were given by Dr. Michael Drescher, Dr. Rebecca Rooney, and Dr. Jeremy Pittman. The extent of contribution is indicated by the author order in each manuscript. Editorial contributions for the entire thesis were also given by Dr. Michael Drescher and Dr. Rebecca Rooney.

Data for this thesis was obtained from Land Information Ontario, the City of Cambridge, City of Kitchener, City of London, City of Markham, City of Vaughan, City of Waterloo, and Town of Whitby.

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Abstract

The few wetlands that remain in the human-dominated landscape of southern Ontario are affected by degradation and conversion to other land use and land cover types. Conversion has negative impacts on wetland-provisioned ecosystem services, such as aquatic species habitat, water filtration and flood prevention. Impacts on the latter services are especially concerning, given the increase of flood events that likely will be exacerbated by a changing climate. Stormwater management (SWM) ponds are constructed to control urban runoff, but do not have the same form and function as wetlands. This study examined recent (2002-2011) trends and drivers of wetland conversion (i.e. wetland loss and SWM pond gain) in seven southern Ontario municipalities. Following this, a Markov model was constructed to project future conversion given specific land use and land cover types. Network analytical approaches were then used to investigate effects of conversion on landscape connectivity. Results show that most wetlands lost were smaller than 2 hectares. While the total area of SWM ponds gained was greater than that of wetlands lost, the size of the average SWM pond gained was less than the size of the average wetland lost. Wetland conversion is projected to continue under all examined land use and land cover types, with losses particularly high in extractive and urban land uses. Overall, wetland conversion corresponded with decreased connectivity. Wetlands appeared to be more connected over the landscape compared to SWM ponds. However, SWM ponds likely acted as stepping-stones between wetlands and compensated somewhat for connectivity losses. The results provide further evidence for the need to halt wetland losses, especially for small wetlands, while showing the potential for connectivity improvements by SWM ponds. By conserving wetlands, policy makers can help to protect human life and property that rely on the critical ecosystem services provided by wetlands.

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List of Abbreviations

GIS – Geographic Information Systems

GTA – Greater Toronto Area

ha – Hectares

LID – Low impact development

LIO – Land Information Ontario

m – Metres

NC – Number of Components

NL – Number of Links

OME – Ontario Ministry of the Environment

OMNRF – Ontario Ministry of Natural Resources and Forestry

OWES – Ontario Wetland Evaluation System

PPS – Provincial Policy Statement

PC – Probability of Connectivity

shp – Shapefile

SOLRIS – Southern Ontario Land Resource Inventory System

SWM – Stormwater Management

SWOOP – South Western Ontario Orthophotography Project

1.0 General Introduction

This chapter serves as an introduction to the overarching concepts that guide this thesis. The concepts are further explained in the introduction sections of each data chapter. The first sub-section explains the problem context in terms of the significance of wetlands, trends and drivers of their loss, and the importance of landscape connectivity. This is followed by a brief explanation of this thesis' purpose in relation to the broader problem context. Lastly, there is an overview of the subsequent sections of this thesis, which include the literature review, general methodology, two data chapters (manuscripts), and a synthesis chapter.

1.1 Problem Context

Wetlands are unique ecosystems that provide our populations with critical services that are not directly replicable by human infrastructure, such as stormwater management (SWM) ponds (Rooney et al., 2014; Tixier, Rochfort, Grapentine, Marsalek & Lafont, 2012; Moore, Hunt, Burchell & Hathaway, 2011). These services include carbon sequestration and flood prevention, meaning that the presence of wetland ecosystems is socio-economically important, especially as we progress into a climatically turbulent future (Moudrak, Hutter & Feltmate, 2017). Wetlands also provide habitat for a multitude of species that include Ontario's freshwater turtles and amphibians, many of which at considered Species-At-Risk (Government of Ontario, 2018b). The swamps, marshes, bogs, and fens of Ontario comprise approximately twenty-five per cent of Canada's wetlands, and six per cent of global wetlands (OMNRF, 2017). As such, these wetlands are valuable and unique ecosystems that must be maintained in order to protect our communities and valued environmental services.

Despite their now-evident importance, wetlands were not always understood as being vital and beautiful natural features. Rather, they were seen as dirty areas that were sources of disease, and their drainage was encouraged (Wiebusch & Lant, 2017). Most conversion is estimated to have occurred in northern countries over the first half of the twentieth century, and this has resulted in the conversion of over 72% of Ontario's pre-European Settlement wetland extent (Ducks Unlimited Canada, 2010; Zedler & Kercher, 2005). Agriculture has been a main driver of wetland conversion, as these areas have rich and moist soils that can provide optimal conditions for crop growth (Wiebusch & Lant, 2017; Snell, 1987). More recent drivers of wetland loss are urbanization and urban sprawl, which has contributed to continued wetland loss, despite policy shifts towards the protection of these natural areas (Ducks Unlimited Canada, 2010; Schulte-Hostedde, Walters, Powell & Shrubsole, 2007).

The remaining wetlands in southern Ontario exist in human-dominated landscapes where they may be isolated, which can be problematic as wetland species and processes rely on the connectivity of these ecosystems (Thorslund et al., 2017; Baxter-Gilbert, Riley, Neufeld, Litzgus & Lesbarrères, 2015; Mackinnon, Moore & Brooks, 2005; Haxton, 2000). In the context of wetlands, connectivity is vital for both biological and hydrological reasons. Without connectivity, species cannot disperse to the habitats that they require throughout their lives, and declines can occur due to a subsequent lack of genetic diversity, which is needed for the maintenance of viable populations (Haxton, 2000; Reh & Seitz, 1990). When species attempt to disperse, which facilitates gene flow, additional species declines result due to the high mortality that is associated with dispersal across human infrastructure such as roads (Baxter-Gilbert, Riley, Lesbarrères & Litzgus, 2015; Mackinnon et al., 2005). Connectivity also plays a key role in climate change adaptation for biodiversity, since the species that we rely on for critical ecosystem services need to be able to disperse as their ranges shift to higher latitudes (Opdam & Wascher, 2004; Humphries, Thomas & Speakman, 2002)

The hydrological connectivity of wetlands is also vital since these ecosystems generally do not exist in isolation, but rather, are part of an interconnected system that regulates water balances within a hydrologic catchment (Thorslund et al., 2017). The regulation of water balances can help prevent flooding, which is a common issue where wetlands have been fragmented by less pervious surfaces, and is especially important given the increased threat of flooding that climate change brings (Moudrak et al., 2017; Thorslund et al., 2017). An additional result of hydrologic wetland connectivity is the maintenance of nutrient balances, as isolated wetlands are more likely to accumulate organic matter that can damage water quality (Racchetti et al., 2011). Planning for connectivity is critical for the biological and hydrological reasons mentioned, especially as climate change has the ability to further fragment sensitive wetland ecosystems via the alteration of hydrological regimes (Werner, Johnson & Guntenspergen, 2013; Zedler & Kercher, 2005).

1.2 Research Purpose

While Ontario is one of the few provinces for which multiple comprehensive estimates of wetland loss exist, none have examined loss more recent than 2002, or for wetlands smaller than 10 hectares (Ducks Unlimited Canada, 2010; Snell, 1987). Additionally, while we know that SWM ponds do not fulfill the same habitat provision role as wetlands, little to no knowledge exists on how they may contribute to landscape connectivity (Tixier et al., 2012; Moore et al., 2011). This study will fill these gaps in the literature by examining trends and drivers of wetland loss from 2002-2011 and for wetlands as small as 0.5 hectares. This study also examines changes in landscape connectivity over time, and how SWM ponds influence connectivity. This work will inform better land use planning through an increased understanding of the state of wetlands and how to improve connectivity. The presence and function of wetlands should be of critical importance to decision makers, as these issues have broad-ranging impacts on ecosystem services, such as flood mitigation, that human communities rely on now and will continue to even as climate change continues to worsen (Moudrak, Hutter & Feltmate, 2017).

1.3 Thesis Structure

This is a manuscript-style thesis that includes two manuscripts, or data chapters. Following this introductory section, this thesis is structured as follows:

Chapter 2: Literature Review

This chapter contains a comprehensive overview of literature relevant to this thesis, which is sectioned into an overview of wetland ecosystems, conversion of wetlands, and ecological modelling. Conclusions are presented and lead to a discussion of the research questions that guide this thesis.

Chapter 3: Methodology

This chapter contains an explanation of the methodology used to prepare datasets for further analysis in each data chapter. The overall research approach is first introduced and discussed, this is followed by information on the study area and site selection within this area. Data management is then explained in terms of how data collection and cleaning occurred.

Chapter 4: Manuscript 1 – Trends and Predictors of Wetland Conversion in Southern Ontario Municipalities.

This chapter presents research on the trends and drivers of recent wetland conversion in southern Ontario. In addition to an examination of historical trends, a Markov model is used to predict future wetland change given different land use and land cover types.

Chapter 5: Manuscript 2 - Connectivity Contributions of Wetlands and Stormwater Management Ponds in Urbanized Landscapes.

This chapter presents research on the contributions of wetlands and SWM ponds to landscape connectivity, and how connectivity has changed with recent wetland conversion in southern Ontario.

Chapter 6: Synthesis

This chapter acts as an overarching discussion of results brought forth in each data chapter. The significance and links between these results are first discussed. Then, limitations of this study, recommendations of beneficial areas for future research, and areas for change in wetland policy are discussed.

2.0 Literature Review

This chapter contains a literature review that lays the foundations for this thesis by examining common themes and issues among the relevant bodies of work. This review begins with an introduction, which includes the purpose and type of the review, as well as details on the methods used to complete it. Subsequent sub-sections contain the key findings, which touch on pertinent issues and methods used by existing studies. Following this is a brief summary of the most pertinent findings of these. Lastly are the research questions that emerged from this review.

2.1 Introduction

2.1.1 Purpose and Type

According to Creswell (2013), there are typically four types of literature review, with each taking one of the following purposes: to integrate what other studies have done and found, criticize other studies, build bridges between related topics, and/or identify the central issues in a field. Most theses integrate past works (1), then organize the literature into related topics (3), and summarize it by identifying central issues (4). This study will use quantitative methods, for which literature tends to be summarized at the beginning of the study as a separate section, which gives rise to the research questions or hypotheses. This section gives the study context, which is revisited in the discussion, when results of the study are compared with the existing literature. This is done in each of the data chapter's discussion sections (Sections 4.5 & 5.5), as well as in the "Synthesis" chapter (Section 6.0)

For a quantitative study, the review should include sections related to the major independent and dependent variables, and consider studies that compare these variables. The final review should generally be composed of five parts, which include: the introduction, the independent variable topics, the dependent variable topic, studies that address both the variables, and a summary. If there are several variables of study, subsections can be used, or focus can be placed on the most important variable. When looking for studies that compare the variables, it is possible that no literature will exist, identifying a gap where the study can contribute. Lastly, the summary should highlight key studies, themes, and suggest the need for study (Creswell, 2013)

2.1.2 Methodology

To begin this literature review, I initially synthesized my ideas into three main topics, which formed my main search areas. These topics were: land use and land cover change, ecological modeling, and habitat creation. These topics were later modified and expanded upon to form the various sub-headings found in this review. Broad searches were performed for each of these topics and I consulted some of the most highly cited and recent literature for key terms and common sources. I compiled lists of key terms for each of these topics, an example being that for the sub-subtopic of stormwater management ponds, key terms include: *constructed wetlands, stormwater, wet ponds, ecosystem services, wetland health, wetland services,* and *ecological engineering*. I then used these terms to guide further literature searches, and garnered further sources from these.

Often, the first articles I examined were secondary sources, or reviews, which helped me to find the primary literature. This helped me to ensure that I was reading work that is found relevant by experts in the field. Additionally, I found different databases and search engines to be more useful for different goals. For example, Google Scholar and the University of Waterloo Library website were very helpful for quickly finding sources when I already had a citation from another piece of literature. However, when I wanted to search in more detail, I mostly focused on the use of in-depth databases, such as Scopus, while Google Scholar was often also adequate for this purpose.

As I selected studies I wished to include, I briefly summarized them, and sectioned my summary into the categories suggested by Creswell (2013), which include: context, purpose, methodology, results, and critiques. I used a spreadsheet to organize these summaries for each of the main themes and their subthemes, which are represented by the subsequent headings.

2.2 Wetland Ecosystems

2.2.1 Stormwater Management Ponds

Modern landscapes are less conducive to water retention than their natural counterparts, as agricultural fields and urbanized areas do not allow for the same amount of water infiltration as wetlands and forests do (Dietz & Clausen, 2005). In the face of climate change, there is an increased importance of stormwater management (SWM), and engineered pond facilities are seen as best management practices to remove nutrients and pollutants from water runoff (Tixier et al., 2012; Moore et al., 2011). The use of SWM ponds for this purpose is generally preferred over using existing wetlands, as stormwater can cause damage to wetlands through hydrologic changes, increased sedimentation, and introduction of chemical contaminants (Schulte-Hostedde et al., 2007). These facilities have also been presented as a means of increasing available habitat for wildlife and might be able to increase connection of the wetland system (Thorslund et al., 2017; Tixier et al., 2012). However, they do not function exactly as wetlands do, and there is potential for them to have deleterious ecological effects (Tixier et al., 2012).

To examine the ecological risks of such facilities, Moore et al. (2011) compared effluent organic nitrogen (ON) concentrations and ON to Total Nitrogen (TN) ratios against untreated influent, as well as reference ON data from a singular wetland in Northern Carolina. SWM ponds are known to provide phosphorus and nitrogen removal, but ON has been found to persist in effluent from these facilities, which is also the case in wetlands that can be an ON source for receiving streams (Hathaway & Hunt, 2010; Pellerin et al., 2004). This persistence of ON points towards the possibility of there being an irreducible concentration of ON, due to generation by wetland vegetation (Hathaway & Hunt, 2010). Some forms of organic nitrogen, including urea and free amino acids, are known to have effects on phytoplankton and bacterial communities, which could lead to shifts in their composition (Berman & Bronk, 2003; Berg, Glibert, Jorgensen, Balode & Purina, 2001). The Moore et al. (2011) study found that organic nitrogen levels were reduced after SWM pond treatment when compared to urban influent, and that these features may help to re-establish the balance between organic and inorganic nitrogen forms. This study suggests that to examine the biological effects of organic nitrogen effluent, further research on the bioavailability of these nutrients is required (Moore et al., 2011).

Tixier et al. (2012) focused on habitat quality by analyzing a Toronto SWM pond’s chemical composition, sediment toxicity, and composition of the benthic community. The benthic community test focused on oligochaetes, which are good bioindicators of aquatic ecosystem sediment quality and tend to be pollution tolerant. Heavy metal and polycyclic aromatic hydrocarbons are known to accumulate in SWM pond sediment (Kamalakkannan, Zettel, Goubatchev, Stead-Dexter & Ward, 2004). These compounds were found in high levels in the ponds studied by Tixier et al. (2012). Additionally, seasonal concentrations of heavy metals in the pond water and chloride accumulation in the sediment pore water likely influenced the benthic community’s composition most. This study suggests biomonitoring, especially that which includes oligochaetes, as a useful method to better manage these facilities that may be too far from natural systems and poor in habitat quality to host biodiverse communities (Tixier et al., 2012).

Due to issues surrounding pollutants, and other biotic and abiotic factors, it is possible that SWM ponds may function as ecological traps, which are habitats that species prefer but can lower their fitness (Sievers et al, 2018). Clevenot, Carre & Pech (2018) performed a review of the factors that may lead to SWM ponds being ecological traps or high-quality breeding habitat for amphibians, using 25 publications that looked at the colonization of SWM ponds by amphibians in urban or highway areas. They found the main factors that influence ecological viability of SWM ponds to be: the shape of ponds, biotic factors (i.e., vegetation), abiotic factors (i.e., water level), and water pollutants. However, these authors also determined that due to a low number of available publications, more research is needed to be able to draw stronger conclusions about the status of SWM ponds as ecological traps (Clevenot, Carre & Pech (2018). Further, Sievers et al. (2018) found the first empirical evidence of SWM ponds acting as ecological traps for frogs, as tadpoles showed lower survival and less response to predator olfactory cues when they were raised in more polluted SWM ponds. They also state that more information is needed to determine how SWM ponds act as habitats, so that management decisions can be made to mitigate their associated ecological costs (Sievers et al., 2018).

As discussed, constructed SWM ponds have some similarities to natural wetlands, but are generally not equal from a biophysical perspective. There are several types of SWM pond, which are outlined by the Ontario Ministry of the Environment (OME) (OME, 2003). Those determined to be visually similar enough to a natural wetland were included in this study, and include wet ponds, stormwater wetlands and hybrid ponds. Excluded types include dry ponds, which are designed to only hold water for up to 24-hours. Wet ponds have the majority of their volume comprised by deep water zones, with aquatic plants in approximately 20% of the surface area, in perimeter shallow zones. Conversely, stormwater wetlands have the majority (70%) of their area comprised of the shallow zones, and hybrid ponds combine these two in a series, with at least 50% of the pond's volume occurring in deep water areas (OME, 2003).

2.2.2 Wetland-Dependent Species

Inland wetlands provide critical habitat for a variety of species, which include herpetofauna, avifauna, and flora (Quesnelle, Fahrig & Lindsay, 2013; Ashley & Robinson, 1996). Conversion of wetland habitats and the resulting fragmented landscapes have concomitant effects on these species, and can lead to biodiversity decline (Quesnelle et al., 2013). In addition to the loss of the physical wetland, these species are impacted by management of adjacent land, which could contain additional critical habitats that are required for activities like nesting or foraging. Deleterious uses on adjacent land can also have a large impact, as is seen in the case of road mortality for species that attempt to disperse across a developed landscape (Quesnelle et al., 2013). Given their increasing rarity in an urbanizing and agricultural landscape, Southern Ontario wetlands hold unique wetland-species relationships for a host of at-risk-species (Findlay & Houlahan, 1997; Snell, 1987).

Herpetofauna

Ontario is home to a variety of at-risk herpetofauna, including reptiles such as snakes and freshwater turtles; and amphibians such as salamanders, skinks, and frogs (Government of Ontario, 2018b). Federally, all eight of Ontario's native freshwater turtle species are listed as at-risk, with four species of special concern, one threatened species and three endangered species, while seven of eight species are listed as at-risk provincially (COSEWIC, 2018; Government of Ontario, 2018b). According to Cushman (2006) amphibians are the most threatened with extinction of all vertebrates in the current anthropogenic-driven extinction event. This highlights the vulnerability of Ontario's seven endangered amphibian species (Government of Ontario, 2018b). Further, among land use-related drivers of extinction, road mortality is a prevalent driver of decline among herpetofauna species, especially as it pertains to turtles, who have long life histories and tend to be drawn to roadsides for nesting (Thompson, 2015; Patrick, Gibbs, Popescu & Nelson, 2012; Steen & Gibbs, 2004; Haxton, 2000; Ashley & Robinson, 1996).

Ashley & Robinson (1996) studied road mortality of amphibians, reptiles, mammals, and birds on the Long Point causeway, which is a two-lane paved road adjacent to a Southern Ontario protected area. They found temporal variations in mortality, with peaks in the spring and fall that are consistent with life history events, such as reproduction and dispersal. Similar results were found by Mackinnon et al. (2005) in their study of reptile road mortality near Georgian Bay, a major water body in central Ontario. They observed seasonal mortality peaks for both turtles and snakes, which corresponded with events such as nesting. Mortality was highest closer to the major water body and away from driveways, where human development is dense and there is little habitat available. Crossing and mortality patterns were also explained by the proximity to adjacent wetland habitat and water crossings, road intersections that increased the road area, and buildings that were often located close to water (Mackinnon et al., 2005).

To prevent road mortality, exclusion and connectivity structures that divert species under or over a roadway are now commonly installed along major roadways (Baxter-Gilbert, Riley, Neufeld, et al., 2015). Baxter-Gilbert, Riley, Lesbarrères, et al. (2015) studied the effectiveness of these structures for turtle mortality mitigation along a major highway expansion in Central Ontario, and found no difference in the abundance of turtles on the road between areas with and without mitigation. Part of this failure was due to mitigation structures not providing effective exclusion, and also being poorly located in relation to existing dispersal corridors. In order for these structures to be more effective, they must be located in a manner that is conducive to the spatial ecology of species, and should be made of enduring materials that fully exclude the species, such as concrete gravity walls (Baxter-Gilbert, Riley, Neufeld, et al., 2015).

Avifauna

Wetlands in Canada provide significant habitat for migratory bird species of continental and international importance (Dahl & Watmough, 2007). Relatedly, the Ramsar Convention (Section 2.3.2 "*Policy"*) was initiated by parties concerned about habitat for migratory birds, who rely on wetlands such as those in Ontario that are designated as internationally important (OMNRF, 2017). Quesnelle et al. (2013) assessed the independent effects of landscape factors that are thought to contribute to wetland bird and turtle declines, such as wetland area, configuration, and landscape matrix composition. They found that the amount of wetland at a landscape-scale was the most important variable for bird species and as such, wetland loss is the primary landscape variable associated with the decline of wetland birds (Quesnelle et al., 2013).

With the creation of wetlands comes the question of species recruitment, especially in fragmented landscapes such as those found in Southern Ontario. Pynenburg, Moore & Quinn (2017) examined the recruitment of common terns to restored habitat with the use of call playbacks and decoy birds. This habitat included artificial islands for nesting purposes. Although birds were recruited and successfully nested in the new habitat, Pynenburg et al. (2017) did not find evidence of these social attractants being effective, and the main driver for recruitment was likely other tern individuals that had nested in the wetland previously. Although this study focused on a restored coastal wetland, it suggests that the availability of good-quality artificial nesting habitat may be sufficient to re-establish wetland-dependent bird species (Pynenburg et al., 2017).

Flora

Plant communities are fundamental to the function of wetlands and the ecosystem services that they provide, such as carbon sequestration (Houlahan, Keddy, Makkay & Findlay, 2006; Zedler & Kercher, 2005). The composition of plant communities and their persistence can be dependent on wetland type and adjacent land uses. Houlahan et al. (2006) examined the effects of adjacent land uses that include forest cover, road density and building density on the species richness and community composition of wetland plants. They found wetland size to be the most important predictor of species richness, especially for forest species and rare native species, while size had little bearing on the relationship with invasive species. A strong relationship existed between land uses 250-300 m from the wetland and species richness, as seed propagule sources further than this are likely limited in terms on their ability to impact wetland plant communities. However, wetlands also can receive seed sources via migrating waterfowl, which Farmer et al. (2017) found to be able to disperse weed seeds up to a potential 2900 km of the source, meaning that land uses on areas beyond adjacent lands can impact wetland plant communities. Due to a great deal of variation occurring in species composition between wetlands, there is a need for a diversity of these ecosystems in order to conserve a landscape's diversity in full (Houlahan et al., 2006). This connects to the importance of small wetlands, which Houlahan et al. (2006) found to host the least frequently occurring species, indicating that species diversity cannot successfully be conserved by only protecting large wetlands.

2.2.3 Wetland Connectivity

It is widely accepted that ecosystem connectivity is critical for biological conservation, as without dispersal of animals and plants, breeding and subsequent gene flow among populations is greatly reduced (McRae & Beier, 2007; Slatkin, 1987). Isolated populations will eventually lose reproductive fitness as they become inbred and these populations become more susceptible to deleterious stochastic events like disease (Saccheri et al., 1998). For this reason, maintaining and re-creating landscape connectivity is a conservation priority in many habitats, including wetlands (McRae & Beier, 2007; Environment Canada, 2013; Kininmonth, Bergsten & Bodin, 2015). Unfortunately, many environmental management decisions are focused on wetland ecosystems at a local level, without considering these as a landscape-wide system (Thorslund et al., 2017).

Importance of wetland connectivity extends beyond species movement at the landscape level, for the hydrologic movement within these ecosystems must also be considered (Thorslund et al., 2017). Thorslund et al. (2017) refer to a hydrologically connected system of wetlands and their entire catchment as a “wetlandscape” and argue for the need to consider wetland function at this large scale. A key reason for this concept is that hydrologic functions such as groundwater and evapotranspiration occur within such wetlandscapes, and impacts on one component can be felt across the watershed. To evaluate such effects, they examined functional differences between individual wetlands and wetlandscapes, performed an expert survey, a hydro-climatic change analysis, and general wetland literature review. They generally found a mismatch between the scale of wetland research and management, which tends to be too narrow when compared to the reality of the broader hydrologic connectivity in these wetlandscape systems (Thorslund et al., 2017).

Preston & Bedford (1988) first emphasized the need to examine landscape-scale impacts when considering projects that affect a single wetland, due to the potential for individual decisions to result in cumulative effects across the wetland system (Thorslund et al., 2017). Although the idea of cumulative effects is relatively simple, it is scientifically difficult to evaluate. In order to do so, it is necessary to look at effects on spatial scales that encompass the wetland function and extend over a temporal scale that is conducive to the disturbances on these (Preston & Bedford, 1988). For example, the habitat support provided by wetlands is based on the spatial scale of a species' range, so the effects of wetland change on habitat connectivity would best be evaluated across this species range (Preston & Bedford, 1988).

2.3 Wetland Conversion

2.3.1 Trends

Although it is known that a great deal of the world's wetlands have been drained, their loss has not been well documented (Dahl, 2004; Schulte-Hostedde et al., 2007). Modern mapping and Geographic Information System (GIS) methods have allowed for wetland evaluation and change analysis to be performed retroactively; however, the lack of a standardized approach means that global and regional estimates of loss tend to vary (Dahl & Watmough, 2007; Ducks Unlimited Canada, 2010; Zoltai & Vitt, 1995). These inconsistent global estimates range anywhere from 5.3 to 12.8 million km^2 of wetlands lost worldwide with one estimate stating that half of the world's wetlands as lost (Zedler & Kercher, 2005). The majority of this loss is estimated to have occurred in northern countries during the first half of the twentieth century, while conversion since the 1950s has increased in tropical and subtropical areas. It is also worth noting that even when wetlands have not been converted, a great deal are often degraded in terms of ecohydrological functions (Zedler & Kercher, 2005).

Estimates of wetland loss in the United States were generated by Dahl (1990), who compared the pre-colonial wetland extent with the 1980 extent. Pre-colonial estimates were based on colonial or state records, land use records that traced the conversion of lands by use categories, drainage statistics, the extent of hydric soils, and historical wetland acreage data.

National Wetland Inventory data were used for 1980s wetland extent, and although these were more complete than colonial estimates, they were fragmented, as some states had incomplete data. Findings show that in colonial times, the land currently occupied by the United States contained approximately 392 million acres of wetlands, with over half of these (221 million acres) in the lower forty-eight states, and just under half (170 million acres) in Alaska (Dahl, 1990). Over this study's two-hundred-year period, the lower forty-eight states lost approximately 53% of these wetlands, while Alaska lost only a fraction of 1% (Dahl, 1990). Overall, twenty-two states lost 50% or more of their original wetland extent, with the highest percentage loss in California (91%) and highest loss by area in Florida (9.3 million acres) (Dahl, 1990).

Canada is currently home to around a quarter of the world's wetland area, yet it is estimated that approximately 65-85% of the pre-European settlement wetland area has been lost (Asselen, Verburg, Vermaat & Janse, 2013). Despite the existence of these prior estimates, work is underway to complete a national wetland inventory (Ducks Unlimited Canada, 2019). Provinces have created more detailed inventories, which generally include the classification criteria put forth by the Canadian National Wetlands Classification system to determine what constitutes a wetland (Zoltai & Vitt, 1995). Amani et al. (2019) independently used this system to classify Canada's wetlands with the use of Landsat-8 imagery and image processing techniques within the Google Earth Engine. With an overall accuracy of 71%, they estimated that 36% of Canada is covered by wetlands (Amani et al, 2019).

Dahl & Watmough (2007) stated that there was a lack of comprehensive and scientifically sound data on the status and conversion trends of Canadian wetlands, aside from data put forth by several independent, region-specific studies. Since a shared concern for North America's wetlands exists among Canada and the United States, this lacking data is not only problematic for Canada, and it is suggested that a more collaborative cross-border approach needs to be taken (Dahl & Watmough, 2007). The Canadian Wildlife Service has implemented a monitoring program on the status and trends of wetlands in Canada's Prairie Ecozone, which found an annual loss rate of 0.31% of this area's wetlands between 1985 and 2001 (Watmough & Schmoll, 2007). Work is underway to complete a more comprehensive Canada Wetlands Inventory, which will be an important information source that can inform efforts to sustain both Canada's and North America's wetlands as a whole (Dahl & Watmough, 2007). Some parts of this inventory have now been completed for the Boreal and Prairie ecozones and Quebec's Saint Lawrence Lowlands; it is expected that this inventory will be completed nation-wide within five years after additional funding is secured (Ducks Unlimited Canada, 2019).

Consistent with Zoltai & Vitt's (1995) statement that more detailed wetland inventories exist provincially than do federally, two comprehensive studies exist of wetland conversion in Southern Ontario, by Snell (1987) and Ducks Unlimited Canada (2010). Snell's (1987) study, used soil data, land use data, and supplementary information to map wetlands and conversion on 125 map sheets. Subsequently, Ducks Unlimited Canada (2010) replicated this study using GIS techniques but extended their study period up to 2002. Although differences between these studies are minor, they are likely based on the somewhat different data sources. An example is the use of country soil surveys and quaternary geological data by Ducks Unlimited Canada (2010) instead of Canada Land Inventory agricultural capability maps and the National Topographic System (Ducks Unlimited Canada, 2010).

Both of these change studies found that the pre-European settlement extent of wetlands in Southern Ontario was roughly 2 million hectares, with the exact area found to be 2.38 million hectares (ha) by Snell (1987), and 2.03 million ha by Ducks Unlimited Canada (2010). Snell

(1987) found an overall reduction in wetland area equal to 61% of the pre-settlement area by 1982, with only 0.92 million ha remaining at this time. Meanwhile, Ducks Unlimited Canada (2010) found a 72% loss of pre-settlement wetland extent by 2002, equal to 1.4 million ha of wetland lost. Although the rate of loss has slowed down in recent history, there were still 70,854 ha of wetland lost from 1982 to 2002, an average of 3,543 ha per year, or 354 large (10 ha) wetlands per year. It is also important to note that the detection limits of these studies mean that they provide conservative estimates of wetland loss, for example, Ducks Unlimited Canada's (2010) study only included wetlands that were 10 ha or more in area. This means that the conversion of smaller wetlands, which still have eco-hydrological significance, were likely missed.

From an overall land use and land cover change perspective, Cheng & Lee (2008) examined change in Ontario’s Greenbelt from 1993-2007; this is an area that now contains protected green space, but has undergone extensive conversion. Cheng & Lee (2008) used Landsat imagery to assess this change, which is an efficient method of surveying landscapes, but may have resulted in an underestimation of the actual amount of land use conversion due to the imagery’s coarse resolution (28.5 m). Results show prevalent land conversions that include urbanization, followed by the creation of golf courses and stone quarries. However, little wetland or forest conversion was found, which may be due to the period of study, the location within Ontario’s protected Greenbelt, or the resolution-based limitations of LANDSAT data (Section 2.4.1) (Cheng & Lee, 2008).

2.3.2 Drivers

Management

A variety of anthropogenic actions related to the management of land use are the driving factors for wetland conversion and loss, and include activities related to agriculture, resource extraction, and urbanization (Schulte-Hostedde et al., 2007; Dahl, 2004). These activities are regulated by the policies discussed in Section 2.3.2 ("*Policy")* and the various agencies that are tasked with implementing and regulating their use, such as the Ontario Ministry of Natural Resources, and the Province's thirty-six Conservation Authorities (OMNRF, 2017). Wetlands were previously thought to be sources of disease and barriers for development, and as such, their conversion was encouraged in many countries, including the United States (Wiebusch & Lant, 2017). With scientific advances has come a shift in the attitudes towards these natural features, and a subsequent shift in management with the Canadian focus now resting on having no net loss of wetlands (Dahl & Watmough, 2007). However, this does not necessarily mean that wetlands are not still drained, and they are still threatened with loss and conversion (Ducks Unlimited Canada, 2010).

The drainage of wetlands for agricultural purposes has been the main driver of wetland conversion globally, as they are productive, nutrient-rich environments. Wiebusch & Lant (2017) performed an economic analysis of wetland conversion given crop prices in the United States, based on the hypothesis that the crop prices must be high enough to offset the cost of drainage, which is a time and resource-intensive process. They also examined two drainage-related programs, the Agricultural Conservation Program, which existed in the mid-1900s and decreased the cost of drainage through subsidies; and the Wetland Reserve Program, which existed after 1991 and is a lump-sum payment program for wetland conservation. It was found that this shift towards conservation-oriented programs was effective at reducing wetland conversion, while short-term increases in crop prices that even occurred during the conservation program period could be responsible for an observed net loss of wetlands. This study illustrates that in order to prevent agricultural conversions of wetlands, policy makers must generally be able to offer a benefit that offsets any gain a farmer may receive from converting a wetland (Wiebusch & Lant, 2017).

One resource industry in Southern Ontario is the aggregate industry, which Cheng & Lee (2008) found to account for 13% of all general land conversion from 1993 to 2007 within what is now Ontario's Greenbelt, via the creation of gravel pits and quarries. At a much smaller percentage than for overall land conversion, Ducks Unlimited Canada (2010) found that resource activities only accounted for a total of 1.4% of all wetland conversions from the pre-European settlement wetland extent to 2002. Of this 1.4%, 1.3% of this was attributable to tree plantations, leaving only 0.1% of wetland conversions attributed to extractive industries like the aggregate industry. On the other hand, aggregate operations can also result in wetland construction during the rehabilitation phase. Santoul, Gaujard, Angélibert, Mastrorillo & Céréghino (2009) found these types of constructed wetlands to support water birds and increase ecological connectivity by acting as intermediate steps between wetlands.

Urbanization is a prominent process that Cheng & Lee (2008) found to account for 68% of all land conversion in their study, and can generally be a key driver of wetland loss (Schulte-Hostedde et al., 2007). Although Ducks Unlimited Canada (2010) only found built-up areas to account for 4.2% of converted wetlands of Southern Ontario, more than 50% of these conversions occurred in the Metropolitan areas of Toronto and Peele, which are within the rapidly growing Golden Horseshoe Region. This study focused on large wetlands, so it is very possible that smaller wetlands are being lost at a higher rate in urbanizing areas (Ducks Unlimited Canada, 2010). While wetlands are lost through urbanization, this process also leads to the implementation of SWM, with the primary objective of controlling sedimentation and erosion during construction in order to manage the quality of downstream waters afterwards (Schulte-Hostedde et al., 2007). Due to the potential negative effects of stormwater quantity and contamination on wetlands, they are generally not permitted for this use in Ontario, and the management of SWM ponds is implemented through specific sub-watershed, site and subdivision, and SWM plans. With OME's (2003) SWM design guidelines, the focus of SWM transitioned from being purely flood-control-oriented, to water-quality-oriented, including design guidelines for naturalized features (OME, 2003; Schulte-Hostedde et al., 2007).

Policy

Globally, wetland conservation is governed by the Ramsar convention, which has 170 contracting parties member states and 2,331 wetland sites of international importance, this amounts to a total area of 249,591,447 hectares (Ramsar Convention Secretariat, 2018a, 2018b). These wetlands of international importance are designated based on standardized criteria, including if the wetland contains a rare, representative, or unique example of a wetland type within the biographic region; or, meets criteria to conserve biological diversity based on ecological communities, water birds, fish, or other taxa (Ramsar Convention Secretariat, 2014). Canada signed the Ramsar Convention in 1981, forming an agreement to use wetland resources sustainably, designate internationally important wetlands, and conserve them (Schulte-Hostedde et al., 2007). Although there are thirty-six Ramsar sites in Canada, Schulte-Hostedde et al., (2007) state that these sites are only representative of a small portion of Canada's wetlands, for which Environment Canada is responsible, while the rest fall under the jurisdiction of provincial and municipal governments (Dahl & Watmough, 2007).

Canadian policy takes a "no net loss" approach to wetland conservation, which allows for offsetting of wetland losses through the creation of constructed wetlands elsewhere on the landscape (Dahl & Watmough, 2007). This is also the approach that was historically taken in the United States (Dahl & Watmough, 2007). However, in 2004, wetland policy in the United States changed to include a desire for increased wetland quantity and quality as well as retention of wetland function. Dahl & Watmough (2007) found that these policies were effective in promoting a net wetland gain from 1998-2004.

Policies that focus on this type of wetland offsetting are criticized for a variety of reasons, including that they simplify wetlands to their area alone, allowing them to be replaced with features that are inadequate in terms of their biophysical functionality (Bendor, 2009). More specific to Canada, the Federal Policy on Wetland Conservation aims to "promote the conservation of Canada's wetlands to sustain their ecological and socio-economic functions, now and in the future" (Government of Canada, 1991). Within this policy, offsetting of wetland function is facilitated by the goal to have "no net loss of wetland functions on all federal lands and waters" (Government of Canada, 1991). However, in their discussion of restoration, creation, and recovery of U.S. wetlands, Kentula (n.d.) states that the functional replacement of wetlands has generally not been demonstrated, and that restoration of damaged or destroyed wetlands is more likely to be successful than wetland offsetting.

In the Province of Ontario, wetland management is governed by a variety of legislative tools, which began to be developed in 1981 (Schulte-Hostedde et al., 2007). The initial system had three phases to its completion, which began with the 1984 establishment of the Ontario

Wetland Evaluation System (OWES); a 1981 policy under the *Planning Act*; and lastly, a host of incentive-, education-, and strategic direction-focused programs and partnerships. Through their analysis of Ontario Wetland Policy, Schulte-Hostedde et al. (2007) found that there has been incremental improvement of wetland protection, but the OWES fails to include wetlands smaller than 2 ha in most instances, despite these wetlands being important from a landscape-perspective (Thorslund et al., 2017). Today, wetlands in Ontario are governed by over twenty pieces of legislation that are implemented by agencies that include the federal and provincial governments, conservation authorities, and municipalities. A wetland conservation strategy was proposed for the province by OMNRF (2017), but there is no one provincial policy currently covering wetlands (Warren, 2014).

The main provincial legislation for wetlands on private land comes through the 2014 Provincial Policy Statement, under the *Planning Act* (Government of Ontario, 2019d). This policy protects wetlands designated by the OWES as "Provincially Significant" from both development and site alteration based on their location (OMNRF, 2017). Such locations include Ecoregions 5E, 6E, and 7E, coastal wetlands, and other locations where alteration is only permitted if there will be no negative impacts on the wetland or is ecological function, such as in the Canadian Shield north of Ecoregions 5E, 6E, and 7E (Government of Ontario, 2019d). However, these provisions in the Provincial Policy Statement only lead to the protection of approximately a third of Ontario's wetlands (Warren, 2014). Another provincial policy that regulates wetland conservation in Ontario is section 28 of the *Conservation Authorities Act*, which gives power to the Province's thirty-six Conservation Authorities to "prohibit, regulate, or require permission" for development that may interfere with wetlands (Government of Ontario, 2019a; Rich, 2014).

As discussed (Section 2.3.2 *"Management"*), drainage for agriculture is a predominant driver of wetland loss, and is regulated provincially in Canada (Walters & Shrubsole, 2005). In Ontario, drainage is dictated by legislation that includes the *Drainage Act,* which facilitates the creation of drainage works, but does not specifically acknowledge wetlands (Ducks Unlimited Canada, Earthroots, EcoJustice & Ontario Nature, 2012; Government of Ontario, 2018a). This act allows for removal of natural features like wetlands if the removal leads to an "improvement" of land, such as increased crop production. However, some protection may come through the mandate that Conservation Authorities be informed of drainage works (Walters & Shrubsole, 2005). Review of the *Drainage Act* occurred in 1972 and recommendations involved there being a need to incorporate environmental impact statements and cost-benefit analyses to improve outcomes in terms of wetland conservation. However, these recommendations were replaced with a mechanism that involves referral to wetland stakeholders, including Conservation Authorities, who have regulatory and bargaining power. Although this process was implemented, such regulation through the *Drainage Act* has had a limited influence on wetland conservation, and losses have continued on private land (Walters & Shrubsole, 2005).

Regional plans also play a role for wetland protection in Ontario, with relevant instruments including: the *Niagara Escarpment Planning and Development Act and Plan,* the *Oak Ridges Moraine Conservation Act and Plan*, the *Greenbelt Act and Plan*, *Places to Grow Act*, the *Growth Plan for the Greater Golden Horseshoe*, the *Lake Simcoe Protection Act and Plan*, and Municipal Official Plans (Ducks Unlimited Canada et al., 2012). As one of the largest examples of these plans, the Greenbelt Plan spans a total area of 720,000 ha and surrounds a great deal of Ontario's highly populated Golden Horseshoe Region, including lands previously designated with the Oak Ridges Moraine and Niagara Escarpment Plans. This plan was primarily generated to protect sensitive environmental and agricultural land from urban sprawl and takes a systems-based approach to planning in order to restore and reconnect natural features in order to retain their ecosystem services. Ducks Unlimited Canada et al. (2012) examined the protection of wetlands within the Greenbelt, and found that land use policies in this region are more effective at protecting wetlands from development than in other areas of the Province.

As the policy approach to wetland management in Ontario is somewhat complex and fragmented, other pieces of legislation do exist that somewhat indirectly impact the conservation of wetlands (Ducks Unlimited Canada et al., 2012). These include the *Endangered Species Act, Environmental Assessment Act, Aggregate Resources Act,* and *Ontario Water Resources Act.* An example of this indirect approach to conservation exists in the case of the *Endangered Species Act's* prohibition for the damage or destruction of species-at-risk habitat, which includes wetland-dependent species (Section 2.2.2) (Ducks Unlimited Canada et al., 2012). Additional wetland protection also comes in the form of stewardship programs for private land that include the Conservation Lands Tax Incentive Program, which offers 100% property tax exemptions for eligible natural heritage features on private property (Government of Ontario, 2019b). Eligible features include areas of natural and scientific interest, Niagara Escarpment Natural Areas, endangered species habitat, provincially significant wetlands and community conservation lands (Government of Ontario, 2019b).

Climate Change

In addition to anthropogenic drivers of loss, there is evidence that wetland ecosystems are likely to be vulnerable to the effects of climate change, especially due to the alterations of water volume, which impacts wetland area and integrity (Zedler & Kercher, 2005). Although they are not the focus of this thesis, coastal wetlands are likely to be particularly affected by sea level rises from climate change, which can reduce shoreline vegetation and subsequently impact fisheries habitat and the ability of shorelines to further mediate sea level rises. Coastline subsidence may even affect upland freshwater wetlands as rising saltwater mixes with them, making it unlikely for freshwater species to persist in affected areas (Zedler & Kercher, 2005).

Werner, Johnson & Guntenspergen (2013) examined the effects of recent warming on wetlands in the Prairie Pothole Region of North America, using a hindcast approach between two time periods (1946-1975 and 1976-2005). They used the “wetlandscape” model to simulate historic wetland conditions, including wetland surface water, groundwater and vegetation dynamics, and then determined if this warming has been able to impact wetland function. In this model, the interaction between climate and wetland function was evaluated as wetland productivity through the cover cycle index. This index was based on two equally weighted variables: the proportion of time spent in a hemi-marsh stage, and the mean number of cover cycle state changes. Results of this model showed that recent warming was sufficient to shift trends towards shortened hydroperiods and less dynamic vegetation cycling, leading to lowered wetland productivity in some parts of the Prairie Pothole Region, including the Canadian prairies. As a result of climatic shifts, a 7% increase in low-productivity wetlands was observed in the middle of the study area (Werner et al., 2013).

This link between climate change and wetland loss is mirrored by findings by Opdam & Wascher (2004), who related climate change to ecological fragmentation, which is defined as the breaking apart of habitat (Moore, 1962; Curtis, 1956). They found that an increased frequency of extreme weather events was likely to broaden landscape gaps and restrict ecological ranges, especially in generally fragmented landscapes (Opdam & Wascher, 2004). These findings are relevant for wetlands in the Prairie Pothole Region, which have been degraded through conversion for agricultural and urban land uses (Werner et al., 2013). If this degradation had not occurred, the observed shift of favourable climatic conditions to these more degraded regions could have led to increased wetland productivity that may have offset the observed productivity decreases (Werner et al., 2013).

It is important to note that while climate change is likely to have negative impacts on species, which could be due to decreases in connectivity, Fahrig (2019) cautions against the automatic assumption of fragmentation as being detrimental to species. This is especially important as the observation of fragmentation has been used as a rationale to only conserve large, contiguous habitats, which ignores the value of small habitat patches (Fahrig, 2019). A key flaw of the fragmentation concept is that it has been largely based on the extrapolation of patch-scale patterns and island biogeography theory to landscape effects (Fahrig 2019; Fahrig et al., 2018). Fahrig et al. (2018) state that more landscape-scale empirical studies of the effects of fragmentation are required to determine what the actual effects of fragmentation are on biodiversity. In general, there is no substantiated evidence that groups of many small habitat patches have lower ecological value than fewer large, contiguous patches (Fahrig et al., 2019).

2.4 Ecological Modelling

Ecological models are a representation of reality that help us to interpret ecological processes and predict how they may change in the future (Whittingham, Stephens, Bradbury & Freckleton, 2006; DeAngelis & Waterhouse, 1987). Data are generally collected in an observational manner and it is difficult to determine what factors explain the data, so description of the system is limited to models that are most consistent with observations (Whittingham et al., 2006). One of the most common methods of ecological modelling can be found in multiple regression, a general linear model with multiple predictors. The most ideal models tend to be those that are most parsimonious and generalizable, meaning that they are simplified in terms of the number of variables, while still agreeing with the data. A general method of achieving such parsimony in multiple regression is via stepwise multiple regression, which is a widespread approach to variable selection that is supported by many statistical packages. Whittingham et al. (2006) found that approximately half of studies that applied multiple regression used stepwise multiple regression, despite downsides that include: biased parameter estimation, inconsistent model selection algorithms, and an inappropriate reliance on a single model. Although there are many approaches to modelling, this example demonstrates that statistical methods should be approached with caution and alternate approaches should be considered and tested (Whittingham et al., 2006).

2.4.1 Scale

The issue of scale in ecological research is multifaceted, as ecological phenomena generally occur over a wide spatiotemporal scale, while studies can often only measure much narrower scales, and an issue exists when attempting to generalize patterns measured at these scales to the broader reality (Schneider, 2001). Despite this, a great deal of research still models phenomena such as population dynamics as though they are closed systems, while it is known that populations tend to interact over the landscape, in networks known as metacommunities (Leibold et al., 2004). Similarly, it is well understood that wetland dynamics extend far beyond the scale of individual wetlands, as these systems are connected to large-scale water fluxes and influence landscape functionality from an ecological perspective (Thorslund et al., 2017). This is especially problematic when it comes to wetland management, as decisions tend to be made at the scale of individual wetlands, without considering if such solutions may be effective on a landscape scale. However, there is potential to expand the scale at which wetland management operates by combining ground-based measurement, modelling and statistics, and remote sensing and GIS, to a scale that Thorslund et al. (2017) refer to as the “wetlandscape” (Section 2.2.3).

A consideration of spatial scale specific to modelling and GIS occurs when it comes to the resolution-based limitations of data used for wetland classification, which can have a significant effect on the accuracy of wetland delineation, and this is especially true for inland freshwater wetlands (Klemas, 2011). Advances in remote sensing technologies have made the delineation of such wetlands more practical, such as the advent of high-resolution imagers to map small and patchy upstream wetlands. Additional technologies exist to enhance remote sensing, such as Synthetic Aperture Radars that can help to distinguish forested wetlands from upland forests. Klemas (2011) reviewed uses of remote sensing technologies for wetland delineation, and found that the most effective method of determining long-term trends and short-term changes in wetland vegetation and hydrology comes through a combination of satellite and aircraft imagery combined with fieldwork. However, it should be noted that the use of high resolution imagery for large wetland areas or entire watersheds can be infeasible from a cost perspective, and it may be better to map these areas at medium-resolution and have critical areas examined at a higher resolution (Klemas, 2011).

2.4.2 State and Transition Models

State and transition models include a suite of models that are able to describe ecological processes in terms of alternative states and the transitions that occur between these. These models are able to represent the complex nature of reality that is not grasped by models that simplify ecosystems by assuming that there are linear processes and climax communities (Briske, Fuhlendorf & Smeins, 2005). An initial ecological application of state and transition models occurred in the case of range dynamics by Westoby, Walker & Noy-Meir (1989), who discussed such models and their ability to describe rangelands. Transitions could be prompted by natural processes such as climatic events like fire, as well as management events like grazing. Under this model, there was no long-term, permanent equilibrium, but rather, a continual process that preferred favourable circumstances and avoided unfavourable ones (Westoby et al., 1989).

Markov models are one approach to state and transition modelling of stochastic processes that incorporate the probabilities of a variable staying in one state or moving to another state after one time step (Klein, Berg & Dial, 2005). These models are said to be memory-less, meaning that the probability of a system being in a particular state at time t depends only on the state of the system at time t-1, and not on previous states, which is known as the Markov property (Otto & Day, 2007b). A Markov model is described by its full set of transition probabilities, which are the probabilities that dictate if a system will be in a given state at some time in the future, given that the system was in some other state one time-step prior. An important feature of Markov models is that they can have absorbing states, which are states that cannot be left once they are reached. An example of this would be death in a model that describes progression of a disease (Otto & Day, 2007b).

There exist several examples of Markov models being applied to studies of landscape change, including those that focus particularly on wetland ecosystems. One of these is by Klein et al. (2005), who examined change via climatic landscape drying in the Kenai Lowlands of South-Central Alaska. They studied several spatial scales, and used a Markov model for a regional analysis of overarching drying trends by classifying locations at randomly sampled point locations into one of four wetland states: “water”, “wet”, “open”, or “wooded”, with a time step of 50 years. Their situation technically violated the Markov property of memorylessness, the assumption that transition to another state only depends on the previous state, and an assumption that there is no influence of spatial distribution due to ecological succession indeed being spatially dependent. Although the underlying assumptions of Markov models (i.e., memorylessness and spatial independence) present theoretical limitations to the applicability of these models, they have successfully been used in ecological studies including this one, which developed a transition matrix that showed how past drying trends could manifest into the future (Klein et al., 2005). An approach to address the suitability of Markov models for specific applications is discussed by Muller and Middleton (1994) in their study of land use changes in Ontario's Niagara region. These authors emphasize the need to determine that land use changes are not random prior to applying Markov models, which can be achieved with the use of the Chi-squared statistic to test if changes are independent from those of previous or subsequent years (Muller & Middleton, 1994).

Another example of the application of Markov models for wetland change studies is Zhang et al.'s (2011) study of wetland change in China's arid Yinchuan Plain. This study used wetland distribution maps from 1991 and 1999 to construct a transition probability matrix that included the natural wetland states of "river wetland" and "lake wetland", artificial wetland states of "pond wetland" and "paddy wetland", and the remaining land cover classified as "non-wetland". They also used a chi-square test to test the accuracy of the model based on the actual and predicted wetland area for 2006, and found the model to be an accurate predictor of wetland change. As this model can be used to predict future wetland cover according to current management practices, they recommend these models as a means of technical support for wetland management decisions (Zhang et al., 2011).

2.4.3 Connectivity Models

Several methods exist that allow for computational landscape connectivity analysis, which include network and circuit-based analyses. Network-based approaches involve a graph-based representation of the landscape as a collection of nodes (i.e. habitat patches) and the links between them, which represent a potential path for organism dispersal between two nodes (Saura, Estreguil, Mouton & Rodríguez-Freire, 2011; Pascual-Hortal & Saura, 2006). Circuit-based models can evaluate multiple dispersal corridors simultaneously, which allows them to calculate a landscape's overall resistance to wildlife dispersal (Koen et al., 2010; McRae & Beier, 2007). Each of these broad approaches to connectivity modelling comes with their own advantages and disadvantages, and differing applicability exists for each.

Network connectivity analyses provide a balance between the amount of input data required and the amount of detail they can provide in terms of connectivity results, making them a good tool for application-based connectivity analysis without intensive data requirements (Saura et al., 2011). These graph theory based models are also computationally powerful and can overcome the limitations that may occur when analyses are attempted on large datasets (Pascual-Hortal & Saura, 2006). Within network-based approaches, there are multiple indices available for analysis, and each has differing characteristics, complexity, and limitations (Saura et al., 2011).

As a variety of graph-based connectivity indices exist, Pascual-Hortal & Saura (2006) systematically reviewed ten, in order to better understand the behaviour of each. This included how sensitive each index is to spatial changes, such as habitat node and dispersal corridor loss, as well as how effective each is for identifying the landscape elements that are vital for the overall conservation of habitat. They used their critique of existing indices to present a new index, the integral index of connectivity. This index is most balanced in terms of the elements they analysed, with one consideration being the defined and bounded range (0 to 1). They also suggest that connectivity should be better considered in terms of habitat availability, where habitat nodes are considered as a place where connectivity exists, and both the area and connectivity of habitat patches are considered in the same analysis (Pascual-Hortal & Saura, 2006).

In their study of seascape connectivity in a Great Barrier Reef reserve, Engelhard et al. (2017) calculated the network-based Probability of Connectivity (PC) index, using Conefor 2.6 software (Saura & Torné, 2012). PC is seen as one of the more comprehensive and robust indices available, as it considers connectivity at the node-level while using the concept of habitat availability (Engelhard et al., 2017; Saura et al., 2011). This index considers connectivity both within and among nodes, and the connectivity value of each individual node (dPC) is the change in PC when the node is removed from the analysis (Engelhard et al., 2017). The results of this study show that the Probability of Connectivity (dPC) explained 51-60% of species diversity for fish with intermediate home ranges, while species diversity of fish with small home ranges was best explained within nodes, by the dPCintra component of PC. This study provides an example of the feasible use of network analysis, applied to the planning of more functionally effective conservation areas (Engelhard et al., 2017).

Circuit theory has traditionally been used for connectivity analyses of neural, social and other networks, and has more recently been used for gene flow modelling (McRae et al., 2008). This theory is based on electrical networks, where nodes are connected by resistors and lower resistance results in higher current flow (Klein & Randić, 1993; McRae et al., 2008). In these networks, connectivity increases with the number of pathways available, as is the case in landscape ecology (McRae et al., 2008). These models integrate random walk theory, and predict dispersal success by random walkers, the organism of interest (McRae et al., 2008). However, there are of course some limitations to this method, including that it does not represent movement that occurs only in one direction, while actual movement may be biased in direction. Despite existing limitations, circuit-theory-based models still have great potential when applied to problems of landscape ecology (Koen et al., 2010; McRae et al., 2008).

One study that used circuit theory for an ecological application is by McRae & Beier (2007), who evaluated the effectiveness of a circuit-theory-based model when applied to threatened mammal and tree species. They found that this model consistently achieved a better fit than more traditional methods, especially for the mammal species, wolverines. Results were also improved by incorporating the shape of a species' range, which has not been considered by most other studies, but ignoring this factor creates the potential for connectivity predictions to be biased. Lastly, they also found that barriers to dispersal may be exaggerated by connectivity analyses, which can again be improved by including a species' range shape (McRae & Beier, 2007).

In their landscape-level study, Koen, Garroway, Wilson & Bowman (2010) focused soley on the use of a circuit-theory-based connectivity model, with CIRCUITSCAPE 3.5 software (McRae, Shah & Mohapatra, 2015). This study was focused on the issue of map boundaries when employing these methods, as there is potential to create an artificial barrier to dispersal, causing sites at the boundary to be represented as less connected than in the interior. Koen, Garroway, Wilson & Bowman (2010) found that landscape resistance was increased for maps with set boundaries, but decreased for those with a buffer composed of actual or randomized landscape data. Their study does show the applicability of circuit theory to spatial data, but illustrates a key consideration for the user, artificial boundary mitigation. Overall, circuit theory is shown to be a method with high applicability for studies of landscape connectivity (Koen et al., 2010).

2.5 Summary of Findings

Key findings have emerged under each of the major topics of this review (wetland ecosystems, wetland conversion, and ecological modelling) and these are summarized in State and transition models (i.e. Markov models) quantify the probability of transitions occurring between different states (Zhang et al., 2011; Otto & Day, 2007b; Briske et al., 2005; Klein et al., 2005). The Markov property states that transitions depend only on the state one-time step previous (Otto & Day, 2007b; Klein et al., 2005). However, this assumption is not usually tested, but rather, treated as a potential limitation of the study in case the assumption might not be met (Klein et al., 2005).

Connectivity models allow for computational analyses of movement across a given landscape. Graph-theory based approaches represent this landscape as a set of nodes (i.e. habitat patches) and edges, which represent dispersal paths between a given set of nodes (Saura et al., 2011; Pascual-Hortal & Saura, 2006). These models are generally informative for the purposes of ecological management, with multiple indices available for analysis, and without the heavy computational demands of other models (Saura et al., 2011; Pascual-Hortal & Saura, 2006). Meanwhile, circuit-theory based approaches can be used to examine connectivity in greater detail, by calculating the landscape's resistance to dispersal (Koen et al., 2010; McRae & Beier, 2007). Overall, the state and transition approach to modelling is used to examine research question 1 through Chapter 4.0, and connectivity models are used to address research question 2 through Chapter 5.

Table 2-1. In the following paragraphs, these findings are both summarized, and discussed in terms of their relationship to the research questions (Section 2.6).

Firstly, for the topic of wetland ecosystems, it was found that SWM ponds are constructed to mitigate flood risk in urban and peri-urban areas, which is a service that lost wetlands would normally offer (Tixier et al., 2012; Moore et al., 2011; Schulte-Hostedde et al., 2007). However, SWM ponds are not biophysically equal to wetlands, as they have a different form, contain contaminants, and may function as ecological traps (Clevenot, Carre & Pech, 2018; Sievers et al., 2018; Tixier et al., 2012; Moore et al., 2011; Kamalakkannan et al., 2004). While literature is available on the biophysical function of SWM ponds, it is somewhat limited, especially in terms of their function as ecological traps (Clevenot, Carre & Pech, 2018; Sievers et al., 2018). This means that the impact of SWM ponds on wetland-dependent species like Ontario's at-risk herpetofauna is somewhat unclear. There was very little to no literature available on the landscape-level function of SWM ponds, and it is unclear how they impact connectivity, which is an important factor for the normal functioning of ecological and hydrological process (Thorslund et al., 2017; Kinninmonth et al., 2015; McRae & Beier, 2007; Saccheri et al., 1998; Slatkin, 1987). There was also little information available on the connectivity of southern Ontario's remaining wetlands, warranting the examination of the connectivity of both SWM ponds and wetlands (see research question 2).

Secondly, for the topic of wetland conversion, it was found that while two estimates of wetland conversion exist for southern Ontario, by Ducks Unlimited Canada (2010) and Snell (1987), wetland conversion estimates are generally inconsistent and incomplete (Ducks Unlimited Canada, 2010; Dahl & Watmough, 2007; Zedler & Kercher, 2005; Dahl, 2004; Dahl, 1990). For example, while Ducks Unlimited Canada (2010) found that less than 28% of

Ontario's pre-European settlement wetland extent still existed in 2002, this only included loss of wetlands that were at least 10 ha. As such, an investigation of the conversion of small wetlands is included in research question 1. An existing lack of coordinated policy to protect small wetlands in Ontario, as discussed by OMNRF (2017), Schulte-Hostedde et al. (2007), and Ducks Unlimited Canada et al. (2012), makes it probable that both continued wetland loss and a high magnitude of small wetland loss will be observed. It is also clear that key drivers of wetland conversion in North America have generally included agriculture, resource extraction, and urbanization, which is why spatial factors were included in research question 1 (Wiebusch & Lant, 2017; Ducks Unlimited Canada, 2010; Schulte-Hostedde et al., 2007; Dahl, 2004). Lastly, while not explicitly related to the research questions, the potential of climate change to further exacerbate wetland loss, as suggested by Werner et al. (2013), Zelder & Kercher (2005), and Opdam & Wascher (2004), highlights the importance of conserving wetlands and the irreplaceable ecosystem services that they offer.

Lastly, the topic of ecological modelling guided how the research questions are explored, which is discussed further in the methods sections of Chapter 4.0 and 5.0. Ecological models are a deductive approach to explaining real-world processes and predicting how they may change in the future, which limits their generalizability and ability to account for real-world factors such as economics and stochastic events (Whittingham, Stephens, Bradbury & Freckleton, 2006; Schneider, 2001; DeAngelis & Waterhouse, 1987). State and transition models (i.e. Markov models) quantify the probability of transitions occurring between different states (Zhang et al., 2011; Otto & Day, 2007b; Briske et al., 2005; Klein et al., 2005). The Markov property states that transitions depend only on the state one-time step previous (Otto & Day, 2007b; Klein et al., 2005). However, this assumption is not usually tested, but rather, treated as a potential limitation of the study in case the assumption might not be met (Klein et al., 2005).

Connectivity models allow for computational analyses of movement across a given landscape. Graph-theory based approaches represent this landscape as a set of nodes (i.e. habitat patches) and edges, which represent dispersal paths between a given set of nodes (Saura et al., 2011; Pascual-Hortal & Saura, 2006). These models are generally informative for the purposes of ecological management, with multiple indices available for analysis, and without the heavy computational demands of other models (Saura et al., 2011; Pascual-Hortal & Saura, 2006). Meanwhile, circuit-theory based approaches can be used to examine connectivity in greater detail, by calculating the landscape's resistance to dispersal (Koen et al., 2010; McRae & Beier, 2007). Overall, the state and transition approach to modelling is used to examine research question 1 through Chapter 4.0, and connectivity models are used to address research question 2 through Chapter 5.

Table 2-1. Summary of literature review findings.

Topic	Key Findings
Wetland Ecosystems	1) SWM ponds are constructed to compensate for the flood mitigation function that lost wetlands can no longer offer, but are not biophysically equal to wetlands. 2) Southern Ontario is home to a variety of wetland dependent species, including herpetofauna and avifauna that are at-risk from habitat loss. 3) Wetland connectivity is imperative for the ecohydrological function at the landscape-scale, the loss of wetlands can lead to cumulative effects at this scale.
Wetland Conversion	4) Estimates of wetland conversion are generally conservative, but approximately two-thirds of Southern Ontario's pre-European settlement wetlands have been lost. 5) Land management activities that lead to wetland conversion include agriculture, resource extraction, and urbanization. 6) Wetland conservation policies are relatively recent instruments in Ontario and wetlands are governed by a suite of over twenty pieces of legislation. 7) Climate change is likely to further drive wetland loss, especially in regions that are already degraded.
Ecological Modelling	8) Ecological models are a representation of reality that can be used to explain processes, but should be approached with caution. 9) State and transition models (i.e. Markov models), are used to quantify the probability of transitions occuring between given states and can be used to model wetland conversion. 10) Connectivity models can simulate potential movement across the landscape, main modes of connectivity modelling include graph- and circuit-flow-based approaches.

2.6 Research Questions

As discussed in the previous literature summary, the following questions emerged from this literature review:

1) How has the composition and abundance of wetlands in Southern Ontario changed. Has the creation of SWM ponds had any effect on their presence? As discussed, Ontario has experienced extensive wetland conversion; however, little work has examined this change specifically for smaller wetlands and SWM ponds. I expect to find loss of wetlands that is similar to or greater than known recent trends, and that this loss will be greater than the creation of SWM ponds. I also expect that losses will continue to be likely, and potentially impacted by spatial factors such as wetland size or proximity to an urban centre.

2) How does the current wetland landscape function from an ecological connectivity perspective? Do SWM ponds have any influence on this connectivity? As discussed, many wetlands have been lost to urbanization and agriculture, while SWM ponds have been created in urbanizing regions. I expect that these SWM ponds have increased landscape connectivity, but not to a level that can compensate for the wetlands that have been lost. I also expect that there will be some density of SWM ponds required for there to be a positive influence on connectivity, alongside a threshold of wetland loss at which the landscape becomes disconnected.

3.0 Methodology

This chapter contains an explanation of the methodology used to prepare datasets that are further analyzed in both of the data chapters (Section 4.0 & 5.0). The overall research approach is first introduced, and briefly discussed. This is then followed by information on the broad study area, and site selection within this area. Then, information on data management is presented, including data collection and cleaning. Specific methods used for relevant modelling and analysis are contained within each data chapter's methods section (Section 4.3 & 5.3).

3.1 Research Approach

This study uses quantitative methodology, which is one of the three general approaches to research, in addition to qualitative and mixed-methods research (Creswell, 2013). This approach generally examines the relationship between variables that can be measured in a numerical fashion. Throughout this process, the quantitative researcher tests theory in a deductive manner, with it broken down into hypotheses or research questions, variables used to test the hypotheses, and statistical analyses of results to determine if the hypotheses are supported by the data or are not. Additionally, quantitative research generally attempts to produce results that can be generalized at a greater spatiotemporal scale, and these results should be replicable (Creswell, 2013). The overarching research questions for this thesis are outlined in Section 2.6, while more specific questions are contained within each data chapter's introduction (Section 4.2 & 5.2).

3.1.1 Study Area

This study is focused within Southern Ontario and municipalities within this broad area. This is the same general area that was examined by Ducks Unlimited Canada's (2010) and Snell's (1987) studies of wetland conversion, as it has been subject to a great deal of developmental and agricultural pressures. These pressures have led to an estimated 72% of the pre-European settlement wetland extent being lost as of 2002 (Ducks Unlimited Canada, 2010).

Assessments of data availability indicated that stormwater management (SWM) pond datasets were the limiting factor to what specific study areas could be examined, as these were available at the municipal level and distributed by the municipalities themselves. Therefore, municipalities were included in the study only if they had publicly available SWM pond data that contained the pond type and year of construction. This data was successfully gathered for a total of seven municipalities, which include the City of Cambridge, City of Kitchener, City of London, City of Markham, City of Vaughan, City of Waterloo, and Town of Whitby (Figure 3-1 & Table 3-1). Other relevant datasets (Table 3-2), including: wetlands, wetland change, and aerial imagery were available for all of Southern Ontario.

Table 3-1. Population and land-based statistics for the study area. Area is based on lower-tier municipal boundaries, obtained from Land Information Ontario (2017) and calculated in ArcGIS. All other statistics were obtained from Statistics Canada (2019).

Municipality	**Area (ha)**	**Population**			**Population density/km^2 (2011)**
		2001	**2011**	**% change**	
City of Cambridge	11588.30	110,372	126,750	13.61	1,121.7
City of Kitchener	13821.13	190,399	219,153	14.05	1,602.1
City of London	42320.21	336,539	366,151	8.40	870.6
City of Markham	21268.42	208,615	301,709	35.59	1,419.3
City of Vaughan	27425.23	182,022	288,301	44.49	1,054.0
City of Waterloo	6517.51	86,543	98,780	12.55	1,542.9
Town of Whitby	14882.89	87,413	122,022	31.13	832.7

Figure 3-1. Map of the study area including the municipalities, most current wetland extent, and waterbodies, where the mixedwood plains ecozone encompasses all of southern Ontario.

3.2 Data Management

3.2.1 Collection

Open-source data (Table 3-2) were used for this study, which were collected from sources that included Land Information Ontario (LIO)'s Metadata Management Tool, Scholars GeoPortal, municipal open data portals, and direct communication with municipalities (LIO, 2017; OCUL, 2018). When considering the completeness of this data, it is important to note that minimum spatial detection limits exist for remotely sensed data, below which features can not be reliably detected. As an example, the minimum mappable unit of the wetland change inventory (Table 3-2, SOLRIS 2.0) is 0.5 ha, meaning that changes in wetlands, or sections of wetlands smaller than 0.5 ha, may have been omitted (OMNRF, 2015). Additionally, there may be a slight discrepancy between the features detected by the change inventory and the wetland and SWM pond datasets. This is because the wetland and SWM pond datasets are based on a combination of field and remote data, which would allow for the detection of smaller features than remote data alone does.

Although active maintenance is ongoing for the wetland dataset, prior to 2011 this dataset only included wetlands that had been evaluated by the Ontario Wetland Evaluation System to determine if they were considered "provincially significant" (Government of Ontario, 2014). Criteria that determine provincial significance include biophysical characteristics, such as vegetation communities; as well as the wetland's size, as wetlands smaller than 2 ha are generally not evaluated (Government of Ontario, 2014). Management of this provincial wetland dataset is ongoing, and despite recent efforts to make it more complete with the 2011 Wetland Consolidation project, it may not include every wetland (LIO, 2017).

Table 3-2. Metadata for the data used in this study, including the source, format, resolution, and date of collection, where applicable.

Category	Dataset	Creator	Source	Form	Resolution	Date
SWM Ponds	Various	Municipalities	Municipal Open Data	Vector (shp)	n/a	to 2010
Wetlands	Wetland	OMNRF Provincial Mapping Unit	LIO	Vector (shp)	n/a	2013
Wetland Change	Southern Ontario Land Resource Information System (SOLRIS) v2.0	OMNRF	LIO	Vector (shp)	n/a	2011
Aerial Imagery	Greater Toronto Area (GTA) Orthophotography Project 2013	OMNRF Spatial Data Support Unit	Scholars GeoPortal	Raster (JPG 2000)	20cm	Apr/ May 2013
	Southwestern Ontario Orthophotography Project (SWOOP) 2015	OMNRF	Scholars GeoPortal	Raster (JPG 2000)	20cm	Apr/ May 2015
Land Cover	SOLRIS v1.2	OMNRF	LIO	Vector (shp)	n/a	2000-2002
	SOLRIS v2.0	MNRF	LIO		n/a	2009-2011

3.2.2 Cleaning

Change Periods

The time periods used in this study correspond to the detection and change periods used by the creators of the SOLRIS 2.0 change inventory and a categorization of these dates alongside the included SWM pond construction dates can be found in Table 3-3. Detection dates correspond to approximately 2002 (t=0), 2005/2006 (t=1), and 2011 (t=2). Within the SOLRIS dataset, the change from t=0 to t=1 corresponded to change period 1, while the change from t=1 to t=2 corresponded to change period 2. Specific dates differed somewhat by municipality, as the imagery used for the SOLRIS change inventory differed by region. This is the case at t=1, when some municipalities fell within the boundaries of imagery from 2005, and others within that from 2006 (Table 3-3). The SWM pond datasets that were chosen for this study had to include construction dates for the majority of ponds, and these dates were used to include SWM ponds at the different time steps, where t=0 includes construction dates up to 2002, t=1 up to 2004/2005 (depending on the municipality), and t=2 up to 2010. However, in few instances, individual ponds within a dataset did not have a construction date, and these were excluded from the study.

Table 3-3. Time steps with included SOLRIS detection dates and included SWM pond construction dates, by municipality.

Dataset	t = 0	t = 1	t = 2	Municipalities
SWM Ponds (Construction Dates)	Before and Including 2002	2003-2006	2007-2010	London, Kitchener, Cambridge, Waterloo
		2003-2005	2006-2010	Vaughan, Markham, Whitby
SOLRIS Change Inventory (Image/Detection Dates)	Aug 10, 2002	Aug 2006, Sept Cloud Replacement	June 2011	London, Kitchener, Cambridge, Waterloo
	Aug 2002, Sept 1999 Cloud Replacement	Aug 2005	Sept 2011	Vaughan, Markham, Whitby

Stormwater Pond Type

In addition to the construction date requirement, SWM ponds were only included if they were deemed to be similar enough to a wetland. As an example, “dry” ponds were excluded from this study, as they are designed to only hold water for a maximum of twenty-four hours following a storm (OME, 2003). In addition to “dry” SWM ponds, other excluded types include:

“Sedimentation/Forebay”, “LID”, “Structure”, “Flood Control”, “Infiltration/Exfiltration”, “Oil & Grit Separator”, and “Open Channel”.

Stormwater Pond Area Correction

Upon visual inspection of the SWM pond datasets, it became clear that stormwater pond boundaries were generally an overrepresentation of the actual stormwater pond area, and often included features such as lawns and walkways. To correct their area, all ponds were re-digitized using a heads-up digitization method with 20cm resolution aerial imagery from the spring (Table 3-2), when the ponds were assumed to likely be inundated with water. The heads-up digitization method involves manually drawing features that are visible in an imagery dataset. This was done in ArcGIS 10.6, using the freehand polygon tool. The pond boundary was generally identified as the land-water boundary, except in cases with dense emergent vegetation. When vegetation was observed, it was included within the pond boundary, as the steep physical bank edges were generally visible around the vegetation.

Wetland Extent

The “clip” geoprocessing tool was used in ArcGIS 10.6 to identify if the wetland change inventory and wetland datasets overlapped, and twenty-six instances (out of a total 114 change events) of overlap were found. This was a relatively negligible number of overlaps, given that the wetland dataset includes over seven thousand features within the study area municipalities. However, under the assumption that these change events represent true change and that their removal from the wetland dataset was simply overlooked, it was determined that they should be removed for the purposes of this project.

To remove these overlaps, the “clip” tool was employed within an edit session in ArcGIS 10.6, to clip all change inventory overlaps from the provincial wetland dataset. The resulting dataset represented the post-study-period wetland extent, at t=2. To obtain the pre-conversion wetland extent, which would have included the lost wetlands, these change events were added to this wetlands dataset from t = 2. This was done using the merge tool in ArcGIS 10.6, and occurred in two stages. Firstly, the losses that occurred in period 2 (t=1 to t=2) were added to the t=2 wetland dataset, to result in the intermediate wetland extent, at t=1. Secondly, the losses that occurred from in period 1 (t=0 to t=1) were added to the t=1 wetland extent, to result in the pre-study wetland extent, at t=0.

4.0 Manuscript 1 - Trends and Predictors of Wetland Conversion in Southern Ontario Municipalities.

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4.1 Abstract

Wetlands provide critical ecosystem services like flood mitigation, yet in southern Ontario's urban and peri-urban areas, most have been destroyed and indirectly replaced with stormwater management (SWM) ponds to manage flooding and contaminants. Given projections of climate change driven flooding, wetland loss is especially concerning. We examine the loss of wetlands and gain of SWM ponds within eight southern Ontario municipalities from 2002-2011. We apply a Markov model to project the future extent of wetlands and SWM ponds, both over the landscape as a whole and within specific land use and land cover types. We find that most wetlands lost were smaller than 2 ha. Although the total area of wetland loss was compensated for by the creation of SWM ponds, SWM ponds appeared to be smaller than wetlands. Losses of wetlands and gains of SWM ponds are projected to continue into the future under all examined land use and land cover types, which include extractive and urban land uses. We show that more stringent wetland protection policies are needed to conserve the small wetlands that remain in southern Ontario municipalities, to ensure continued provision of wetland-related ecosystem services and to protect communities from climatically-exacerbated flooding.

Key Words: *Wetland, Land Use and Land Cover Change, Wetland Policy, Ontario*

4.2 Introduction

Across North America, wetlands were historically seen as unpleasant areas that were sources of disease, and their drainage was encouraged (Wiebusch & Lant, 2017). This drainage predominantly occurred for agricultural purposes, as converted wetlands can provide nutrient-rich and damp soils that are optimal for crop growth (Snell, 1987; Wiebusch & Lant, 2017). As such, less than 28% of southern Ontario's pre-European settlement wetland extent now remains (Ducks Unlimited Canada, 2010). We now know that wetlands provide ecosystem services that are critical for the resilience of our communities, and some policy shifts have occurred to avoid further losses of wetland area (Schulte-Hostedde, Walters, Powell & Shrubsole, 2007). However, while these changes have slowed loss, they have not been sufficient to halt it (Dahl & Watmough, 2007; Ducks Unlimited Canada, 2010).

Development of Ontario's wetland policies began in 1981, and these now consist of a somewhat uncoordinated collection of legislative tools, including the *Planning Act, Greenbelt Act* and *Conservation Authorities Act,* and their associated regulations (Government of Ontario, 2005a, 2019a, 2019d; Schulte-Hostedde et al., 2007). It is also problematic that wetland management tends to occur at the site-level, despite the potential for singular impacts to be echoed through cumulative effects on the landscape (Thorslund et al., 2017). Section 2.1 of the Provincial Policy Statement, 2005, precludes development and site alteration within "provincially significant wetlands." The *Planning Act* allows for the designation of wetlands as provincially significant if they meet criteria put forth by the Ontario Wetland Evaluation System (Government of Ontario, 2014).

Assessment of wetland significance is based on biophysical factors such as vegetation community type and wetland size, as wetlands smaller than two hectares are generally not evaluated (Schulte-Hostedde et al., 2007). Despite their size, small wetlands are important for a variety of reasons, including their tendency to host the least frequently occurring plant species, and ability to increase connectivity by acting as stepping stones between other wetland habitats (Houlahan, Keddy, Makkay & Findlay, 2006; Keitt, Urban & Milne, 1997). This lack of protection for small wetlands is just one example of the inadequacy of policy, and the Ontario Wetland Evaluation System in particular, to protect what is left of Ontario's wetlands (Schulte-Hostedde et al., 2007). Recognising the threats to Ontario's wetlands, OMNRF (2017) proposed a wetland strategy for 2017-2030 to create a common focus for wetland conservation in the province. This strategy advocates for more stringent wetland protections, with the main targets aiming for a halt of loss where it has been the greatest, and a gain in wetland area and function (OMNRF, 2017). These targets speak directly to the maintenance and enhancement of the ecosystem services that are critical for our human and wildlife communities.

With regard to human life and property, flood mitigation is perhaps one of the most visible services offered by wetlands, and is especially pertinent given the potentially detrimental damage that will come with storms as climate change continues to worsen (Moudrak, Hutter & Feltmate, 2017). As human populations continue to migrate to urban areas that have prevalent impervious surfaces, a lack of wetlands to absorb stormwater runoff has become a common issue, especially given the monetary implications of flooding (Dietz & Clausen, 2005). In one urban test site, Moudrak, Hutter & Feltmate (2017) estimated that flood-related costs could have been up to 38 percent lower, or CAD $51.1 million less, if wetlands had been maintained.

While the majority of wetlands have been lost or degraded, stormwater management (SWM) ponds have been installed to manage urban runoff (Ducks Unlimited Canada, 2010; Tixier, Rochfort, Grapentine, Marsalek & Lafont, 2012). This process has likely led to a conversion of natural wetlands into SWM ponds in urban and peri-urban areas. SWM ponds are designed to hold water for long enough to allow for sediment removal and to prevent flooding, thereby protecting wetlands from human-introduced contaminants (OME, 2003). Newer designs of SWM ponds usually incorporate habitat features described as "naturalized" to better mimic wetlands and increase their habitat value for wetland fauna (Tixier et al., 2012). Importantly, even naturalized SWM ponds do not fully replace the habitat provisioning service that wetlands provide, due to issues that include toxicity and a form that includes steep bank edges (Moore, Hunt, Burchell & Hathaway, 2011; Rooney et al., 2014; Tixier et al., 2012).

Our study investigates recent (2002-2011) wetland loss and SWM pond gain trends and their potential drivers in seven urban/peri-urban municipalities in Southern Ontario. We investigate wetlands and SWM ponds of all sizes, with wetland loss likely limited by a minimum mapping unit of 0.5 ha. In our investigation, we are building on existing knowledge about large (> 10 ha) wetland loss (Ducks Unlimited Canada, 2010; Snell, 1987). First, we examine trends in the loss of wetlands and gain of SWM ponds. Second, we combine these trends and drivers to project what the future of Ontario's wetland landscape may be under the status quo. Third, we incorporate land use factors to determine what may act as drivers of projected changes.

As a whole, we aim to inform land use planning that will prevent further loss of wetlands and their associated ecosystem services. We present several questions and corresponding expectations related to management and policy, which are as follows:

(1) Has the extent of lost wetlands been fully compensated for by the creation of SWM ponds, from an area-based perspective? As Ontario's wetland policy is somewhat uncoordinated, and decisions about wetland management are generally made at the site-level, we expect that net loss of wetland area will be observed (OMNRF, 2017; Schulte-Hostedde et al., 2007; Thorslund et al., 2017).

(2) Has there been a high prevalence of wetland loss among small wetlands? Due to the Ontario Wetland Evaluation System's focus on protecting wetlands larger than 2 ha, we expect a disproportionate amount of loss amongst wetlands smaller than 2 ha (Government of Ontario, 2014).

(3) Will wetland loss be more probable in areas affected by urbanization and other human-dominated land uses than in land uses that may be less intensive? Agriculture is a historically dominant driver of wetlands loss, and Ducks Unlimited Canada (2012) found built-up lands to be a dominant predictor of wetland loss within urban and peri-urban area. As such, we expect human-dominated land uses to correspond with greater projected wetland loss (Ducks Unlimited Canada, Earthroots, EcoJustice & Ontario Nature, 2012; Snell, 1987; Zedler, 2000).

4.3 Methods

Study Area and Spatial Data

This study uses data on provincial wetland extent and change (Southern Ontario Land Resource Information System - SOLRIS 2.0), and SWM pond extent for seven municipalities in Southern Ontario (Figure 4-1). All GIS operations are carried out using ArcMap 10.6. These municipalities were chosen because of the availability of SWM pond datasets that contain year of construction and pond type. The pond type was an important consideration, as not all SWM ponds are similar enough to a wetland to constitute comparison to or compensation for wetland losses. "Dry" ponds provide one example of this, as they are only designed to hold water for up to twenty-four hours after a storm (OME, 2003). As such, only ponds labelled as "wet", "wetland", hybrid", or "natural" were included. The distinction between wet, wetland, and hybrid ponds is in the depth of these ponds. Wet ponds are deep and contain shallow aquatic plant zones around their perimeter, while wetland-type ponds are dominated by shallow zones, and hybrid ponds combine the characteristics of wet and wetland-type ponds (OME, 2003). The "natural" pond designation only occurred twice in the dataset, and although wetlands are not permitted for use as stormwater ponds under OME (2003), inclusion of these "natural" ponds in municipal SWM data may mean that they were previously modified for this purpose. As such, these two ponds were retained in the analyses.

Visual inspection found that recorded SWM pond boundaries were generally an overestimation of the actual pond area. To remedy this, boundaries were redrawn using a heads-up digitization method, which involved manually re-drawing ponds with 20 cm resolution imagery from the spring. It is assumed that ponds are inundated with water at this time, which was important as the land-water boundary was chosen to be the SWM pond boundary. The land-water boundary included emergent vegetation ones, such as cattail or bulrush. Additionally, these ponds have steep banks that tended to be visible in the imagery, and this aided in boundary delimitation when dense emergent vegetation masked the land-water boundary.

Figure 4-1. Map of South-Eastern Ontario with the study area municipalities labelled. Inset shows the position of study area municipalities relative to the Great Lakes. Basemap imagery source: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community.

Temporal Scale

The temporal scale of this study extends from 2002 to 2011, which is based on detection periods of the SOLRIS 2.0 wetland change dataset (OMNRF, 2015). These years were translated into time steps, where t = 0 corresponds to the landscape in the year 2002 and includes SWM ponds constructed up to this date. Due to the availability of aerial imagery, the next time step, t = 1 corresponds to the landscape in 2005 in the City of London, City of Kitchener, City of Cambridge, and City of Waterloo; and 2006 in the City of Vaughan, City of Markham, and Town of Whitby. In this time step (t =1), we included SWM ponds built from 2003 to 2005/06, depending on the municipality. The final time step, t = 2 includes change that occurred after t =

1, to detection in 2011. The detection in 2011 occurred in June in some municipalities, which was early in comparison to the detection at other time steps that occurred in August and September. As such, we included only SWM ponds built from 2006/07 up until 2010 are included in this time step (t = 2).

Conditional Probabilities

Conditional probabilities are used to determine how factors like the time period and wetland size may be related to the observed probabilities of wetland loss or SWM pond gain. These probabilities describe the relationship between outcomes, where the probability of observing outcome A, given that outcome B has happened is denoted as P(A|B). This is calculated as the fraction of cases where outcome A also occurs when outcome B occurs, out of all occurrences of outcome B (Otto & Day, 2007a).

State and Transition Model

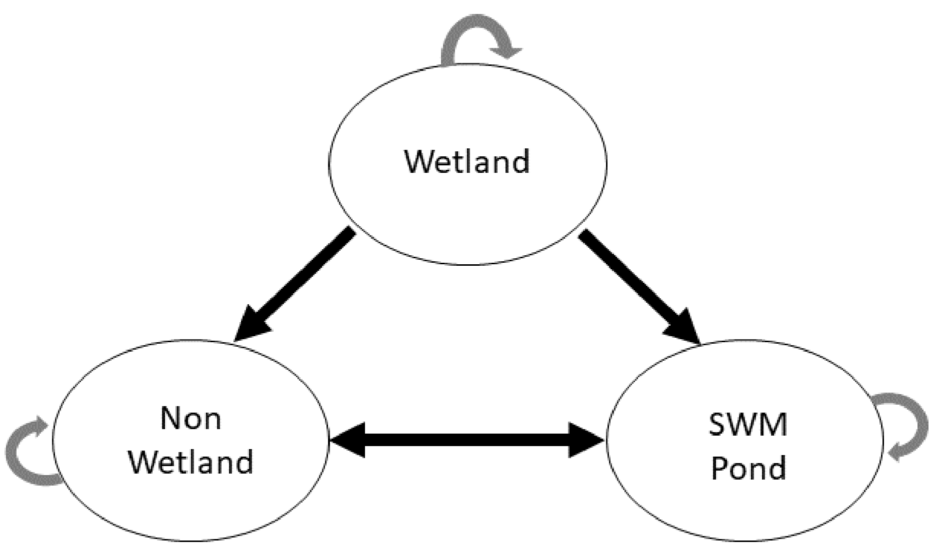

Figure 4-2. Possible wetland states and transitions used in the Markov model. States are represented by ovals and transitions are represented by arrows, where straight black arrows represent transitions between states and curled grey arrows represent no state change.

A Markov model was used to quantify transition probabilities between three states: “wetland”, “non-wetland”, and “SWM pond” (Figure 4-2), using the “msm” package for continuous-time Markov modelling by Jackson (2018) in R version 3.5.2 (2018-12-20). Markov models quantify the probability of transitions between the given states (Otto & Day, 2007b). The Markov property dictates that these models are “memory-less” and transitions only rely on the state one-time step previously (Otto & Day, 2007b). We did not test whether this assumption holds true for the present study, but following Klein, Berg & Dial (2005), we assumed that the Markov property was met. Making this assumption means that our results might be only approximately correct (Klein, Berg & Dial, 2005).

Within the msm package, transition intensities represent the instantaneous risk of a feature moving from state *r* to *s*, for a given pair of states and time (Jackson, 2018):

$$q_{rs}\big(t, z(t)\big) = \lim_{\delta t \to 0} \frac{P(s(t+\delta t) = s | S(t) = r)}{\delta t}$$

The transition intensities for each set of transitions form a matrix, Q, where the rows sum to zero and the diagonals are defined by $q_{rr} = -\sum_{s \neq r} q_{rs}$. To calculate transition intensities, a Qt matrix, with the allowed (q = 1) and prohibited (q = 0) transitions specified, was input to the msm package. The “gen.inits = TRUE” option was specified during model fitting to automate the calculation of transition intensities (Jackson, 2018):

$$Q_t = \begin{bmatrix} q_{11} & q_{12} & q_{13} \\ q_{21} & q_{22} & q_{23} \\ q_{31} & q_{32} & q_{33} \end{bmatrix} = \begin{bmatrix} -(q_{12}+q_{13}) & q_{12} & q_{13} \\ 0 & -q_{23} & q_{23} \\ 0 & q_{32} & -q_{32} \end{bmatrix} = \begin{bmatrix} 1 & 1 & 1 \\ 0 & 1 & 1 \\ 0 & 1 & 1 \end{bmatrix}$$

Meanwhile, within msm, the transition probability matrix, P(t), gives the likelihood of a transition occurring within a given time (Jackson, 2018). This assumes that Q is constant within the given time interval. P(t) can be calculated as the matrix exponential of the scaled transition intensity matrix, where $P(t) = Exp(tQ)$. However, it is more reliable and faster to calculate P(t) analytically for simpler models, including the three-state model used in this study (Jackson, 2018). For this study, time *t* transition probabilities that correspond to the given Q are given by:

$$p_{11}(t) = e^{-(q_{12}+q_{13})t}$$

$$p_{12}(t) = \begin{cases} \dfrac{q_{12}}{q_{12}+q_{13}-q_{23}}\left(e^{-q_{23}t} - e^{-(q_{12}+q_{13})t}\right) & (q_{12}+q_{13} \neq q_{23}) \\ q_{12}te^{(-(q_{12}+q_{13})t} & (q_{12}+q_{13} = q_{23}) \end{cases}$$

$$p_{13}(t) = \begin{cases} 1 - e^{-(q_{12}+q_{13})t} - \dfrac{q_{12}}{q_{12}+q_{13}-q_{23}}\left(e^{-q_{23}t} - e^{-(q_{12}+q_{13})t}\right) & (q_{12}+q_{13} \neq q_{23}) \\ (-1 + e^{(q_{12}+q_{13})t} - q_{12}t)e^{-(q_{12}+q_{13})t} & (q_{12}+q_{13} = q_{23}) \end{cases}$$

$$p_{21}(t) = 0$$

$$p_{22}(t) = e^{-q_{23}t}$$

$$p_{23}(t) = 1 - e^{-q_{23}t}$$

$$p_{31}(t) = 0$$

$$p_{32}(t) = 1 - e^{-q_{33}t}$$

$$p_{33}(t) = e^{-q_{33}t}$$

(Jackson, 2018)

To prepare data for Markov modelling, landscapes had to be classified under each of the three states of interest (wetland, non-wetland, or SWM pond). To do so, the extent of wetlands and SWM ponds were merged, and all other areas within the municipal boundary were designated as non-wetland. Then, the loss of wetlands for each time period was subtracted and gain of SWM ponds was added. The resulting three datasets were rasterized using the “polygon to raster” tool, with the maximum combined area option selected and a 50 m cell size resolution (OMNRF, 2015).

To estimate how probabilities of conversion may be affected by different land use and land cover types, SOLRIS land classification data were incorporated into the Markov model as covariates. This is possible by modelling transition intensities as a function of the covariate variable, $z(t)$ (Jackson, 2018). To incorporate covariates, SOLRIS version 1.2 ($t = 0$) and 2.0 ($t = 2$) land use data were used, alongside the SOLRIS 2.0 change data. This change data was split into two time periods, with the first representing change up until $t = 1$, which was used to construct the appropriate landscape at $t = 1$ by combining the change data up to this point with SOLRIS 1.2 data. To achieve this, SOLRIS 1.2 data was first converted to vector format using the "raster to polygon tool" with the original geometry preserved. This dataset was then merged with the change inventory, using the "FIRST" merge rule, so that data from the change inventory was written over the SOLRIS 1.2 data. Following this merge, the data were re-rasterized at the original 15 m cell-size-resolution using the "maximum combined area" option and a raster value that corresponded to the SOLRIS 1.2 land use categories. This re-rasterized data was then resampled to the 50 m cell size resolution using the "majority" option, to maintain consistency among the three time-steps. Using the field calculator, land use categories were then standardized across all time steps for the land uses and land cover types of interest. These land use and land cover types are: forest; extraction; built (impervious and pervious); transportation; and the combined tilled and undifferentiated SOLRIS classes, which contain agricultural lands as well as others (see Supplementary Information) (OMNRF, 2015; Schulte-Hostedde et al., 2007).

Lastly, the "extract multi values to points" tool was used to extract the state and land use/land cover covariate value at the raster cell centres, for each of the time steps. These data were then merged, sorted by point ID and time, and input to the msm model in R. During this process, approximately 15 improbable transitions from the non-wetland and SWM pond states to the wetland state were found. As this is a small number of data points when compared to the dataset (>1.6 million data points) and these are assumed to be the result of a rasterization error, they were manually removed. To remove these points, the misclassified state was changed to be the one that prevailed over the majority of time steps. This meant that if a cell were classified as a wetland at t = 1 and t = 2, but as a non-wetland at t = 0, t = 0 was assumed to be an error and the classification was changed to wetland at all time steps.

Additionally, no transition from SWM pond to the non-wetland state was observed, but this is assumed to be due to the relatively brief time span of the study period itself, rather than this state being an absorbing state that can not be exited. To train the Markov model to not recognize SWM pond as an absorbing state, one entry was added that showed an additional cell transitioning from SWM pond (t = 0), to SWM pond (t = 1), to non-wetland (t = 2). After these modifications were made, transition probability matrices were then extracted for each of the given land uses at t = 1 and t = 6. These matrices are then used to project the future proportional land cover given the effect of each land use, using a standard value of each time step as equal to four years, which is the average of all time periods for the input data. This means that t = 6 corresponds to approximately the year 2026, given that the starting time is 2002.

4.4 Results

Historical Wetland Conversion

For all municipalities combined, the observed number (i.e. frequency) of SWM ponds gained from 2002-2011 is 1.6 times greater than the number of wetlands lost (Table 4-1). This higher frequency of SWM pond gain remains the case in six of seven municipalities, with the exception of the City of London, which has a higher frequency of wetland loss. Additionally, SWM pond gain is most prevalent of all municipalities in Kitchener, where four times more SWM ponds were gained than wetlands were lost. Conversely, 2.7 times more wetlands were lost in London than SWM ponds gained.

In total, between 2002 and 2011, the seven study area municipalities lost 95.45 ha of wetlands. Over the same period, 111.64 ha of SWM ponds were created, which resulted in a net transition from wetlands to SWM ponds of 1.17 times the total area of wetlands lost (Figure 4-3). Given that the total study area included 137,823.69 ha of land (Table 4-1), this represents 0.7 % of the landscape being converted from wetlands, and 0.8% of the landscape being converted to SWM ponds. This remains true in four of the six study area municipalities that experienced wetland loss, namely the City of Kitchener, City of Markham, City of Vaughan, and Town of Whitby. In these municipalities, the total area of SWM pond gained ranges from being 3.48 times greater than the area lost (City of Vaughan), to being 1.49 times greater than the area lost (City of Whitby). Municipalities that do not mirror this trend include the City of Cambridge and City of London, where the total area of SWM pond gain is less than that of wetland loss. This trend is more muted in the City of Cambridge, where the total area of wetland loss is only 1.31 times greater than the area gained, while the City of London experienced wetland loss that is 2.1 times the total area of SWM pond gained.

Although the seven municipalities experienced a cumulatively greater gain of SWM ponds than loss of wetlands by area (Figure 4-3), the average individual SWM pond created was smaller than the size of lost wetlands (Figure 4-4). Importantly, both lost wetlands and gained SWM ponds were still small (<2 ha), with an average lost wetland being 0.8 ha and an average gained SWM pond being 0.6 ha. However, this trend is only mirrored in three of the seven study area municipalities individually (Figure 4-4). Overall, 95.89 % of wetlands lost and SWM ponds gained were smaller than 2 ha (Figure 4-5).

Table 4-1. Observed number of wetland losses and SWM pond gains from 2002-2011 for the study area municipalities. Area is based on lower-tier municipal boundaries, obtained from Land Information Ontario (2017) and calculated in ArcGIS.

Municipality	Transition		Municipal Area (ha)
	Wetland Loss	SWM Pond Gain	
Combined	114	178	137823.69
City of Cambridge	9	17	11588.30
City of Kitchener	10	40	13821.13
City of London	60	22	42320.21
City of Markham	15	37	21268.42
City of Vaughan	12	33	27425.23
City of Waterloo	0	6	6517.51
Town of Whitby	8	23	14882.89

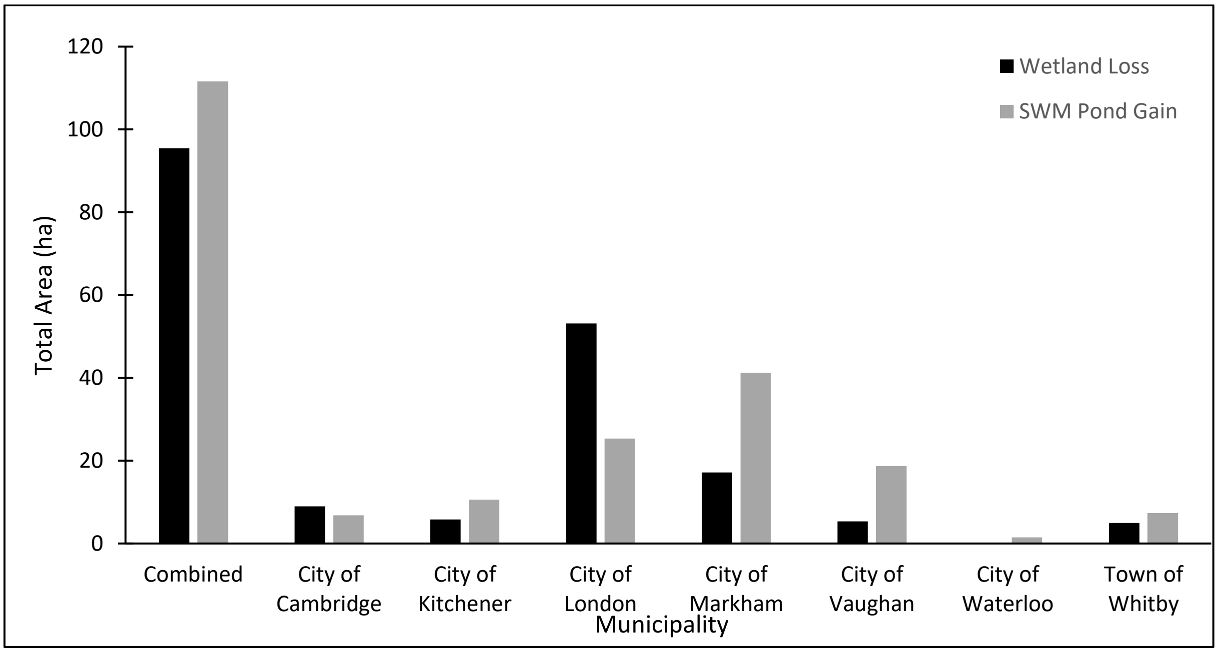

Figure 4-3. Cumulative extent (ha) of observed wetland loss and SWM pond gain over the period from 2002-2011 for each of the seven municipalities and their combined total.

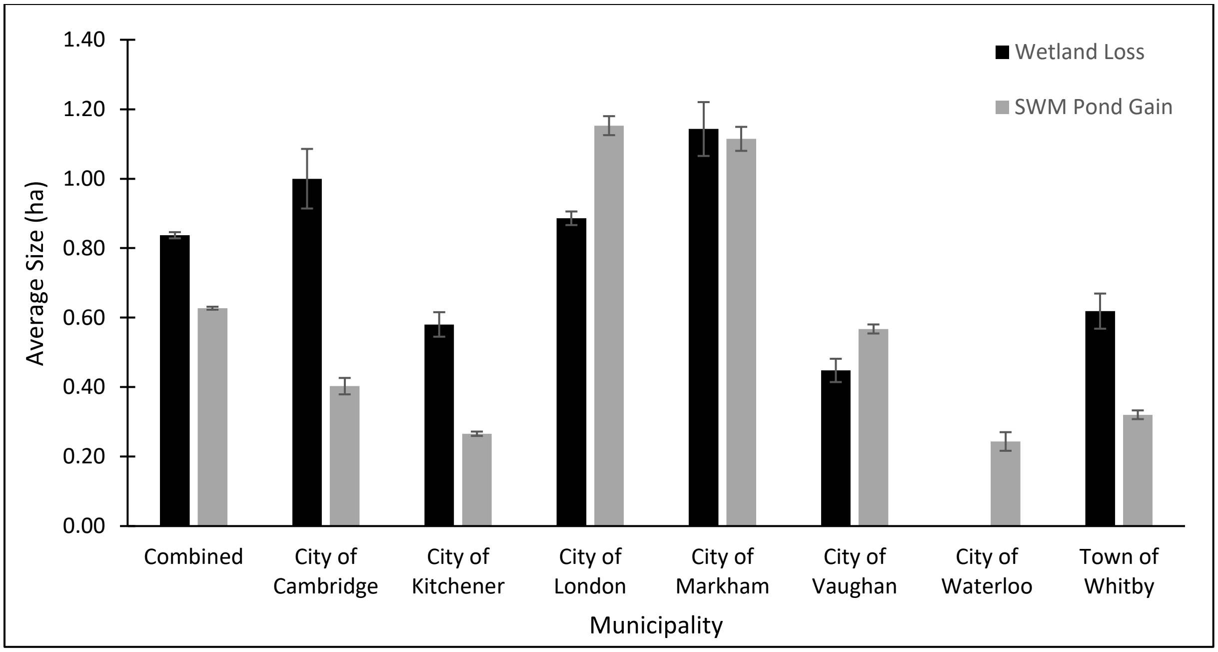

Figure 4-4. Average size (ha) of destroyed wetlands and created SWM ponds over the period from 2002-2011 for each municipality individually and their combined total. Error bars represent standard errors, see Table 4-1 for sample sizes.

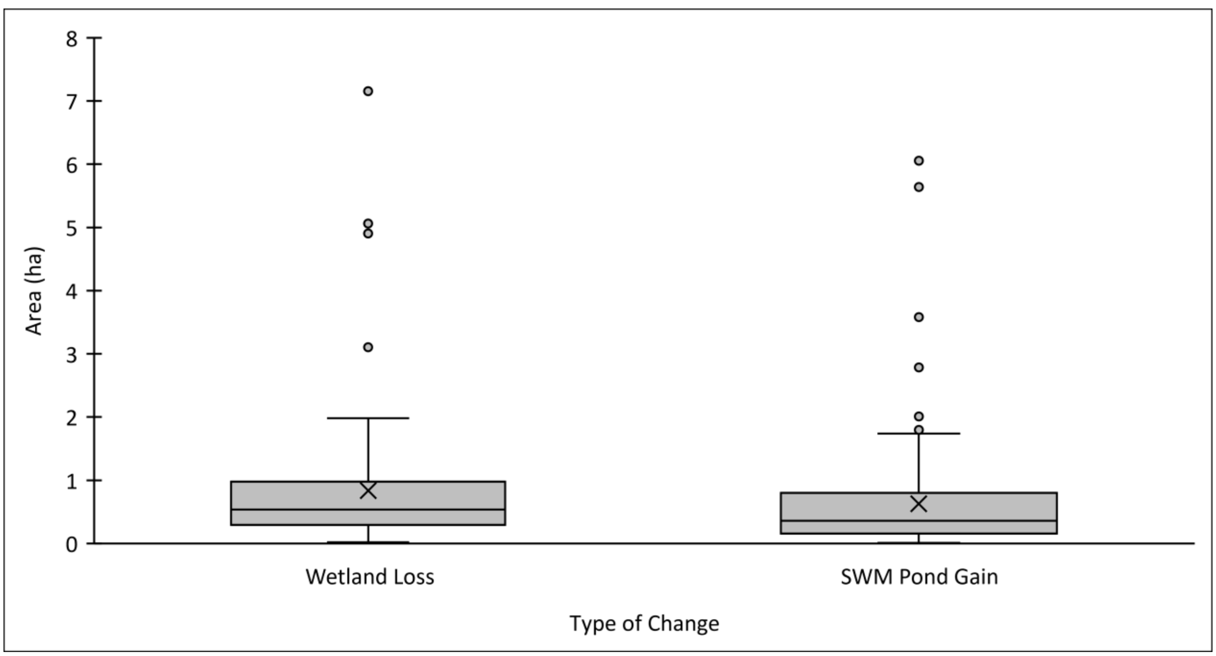

Figure 4-5. Box and whisker plot of the sizes of wetland loss and SWM pond gain over the period from 2002-2011, for all study area municipalities combined.

The majority of loss (> 89%) occurred among small wetlands (< 2 ha, Figure 4-6), for all municipalities combined, and for each municipality individually except for the City of Waterloo, where no wetland loss was observed during the study period (2002-2011). Furthermore, in three of the seven municipalities no wetlands > 2 ha were lost.

The continued loss of small wetlands is also evident in the overall probabilities of state change given wetland size (Figure 4-7). The gain of small (< 2 ha) SWM ponds is incrementally more probable than the loss of similarly sized wetlands, while this incremental difference is mirrored in the slightly greater probability of large (> 2 ha) wetlands being destroyed than the probability of large SWM ponds being created. These probabilities change only slightly throughout the study period, with a decreased probability of SWM pond gain and increased probability of wetland loss during the second half of the study period (Figure 4-8).

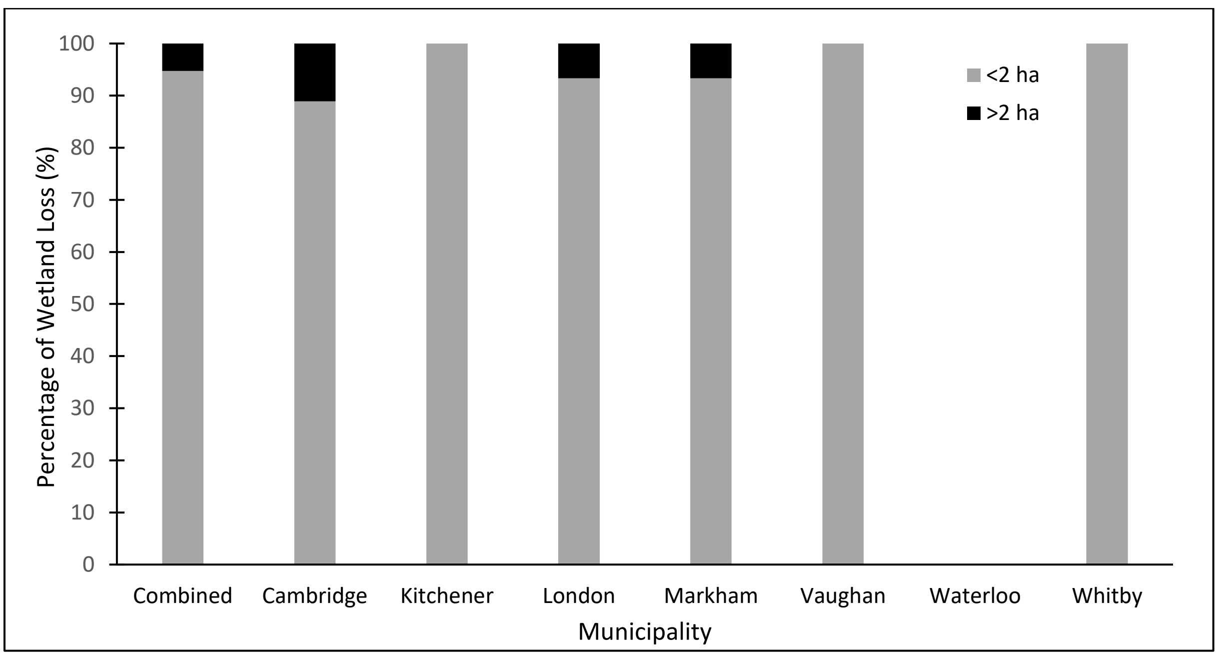

Figure 4-6. Percentage of wetlands lost over the period from 2002-2011 by wetland size (less than or greater than 2 ha) for individual municipalities and all municipalities combined by averaging. Note that no wetlands were lost in the Municipality of Waterloo during this period. See Table 4-1 for sample sizes.

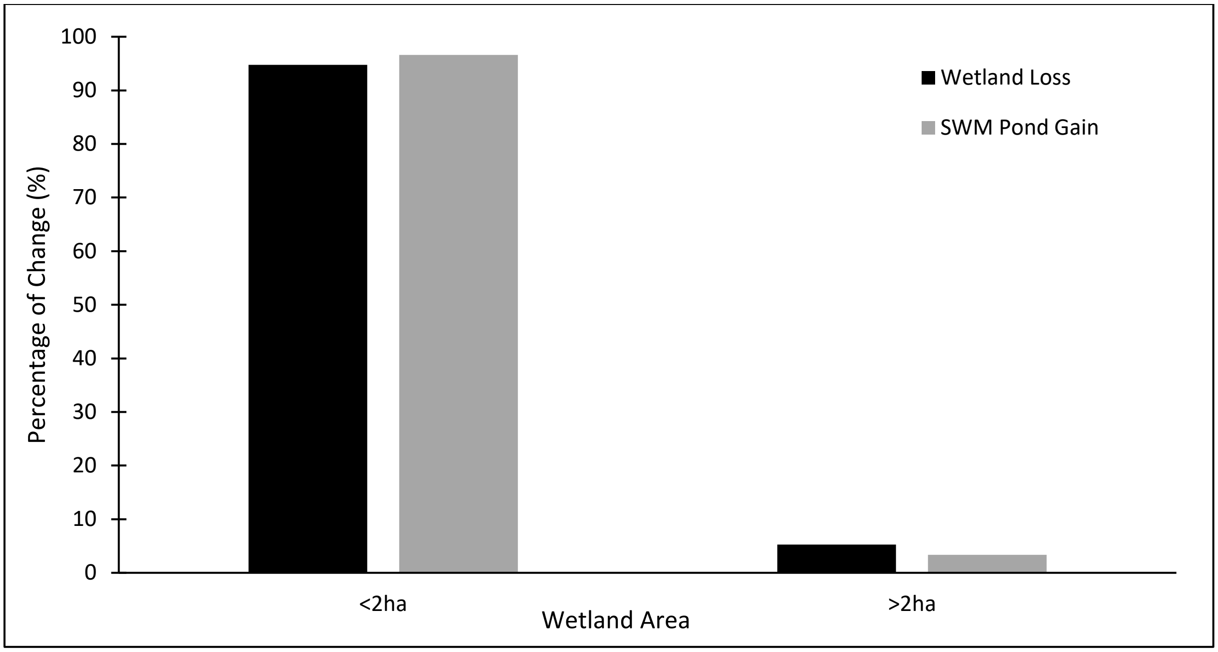

Figure 4-7. Wetland loss and SWM pond gain separated by the percentages of change that occurred among small (< 2 ha) and large (> 2 ha) features that were converted. Conversion occurred over the period from 2002-2011 and is for the wetland loss and SWM pond gain that occurred among all study area municipalities combined. See Table 4-1 for sample sizes.

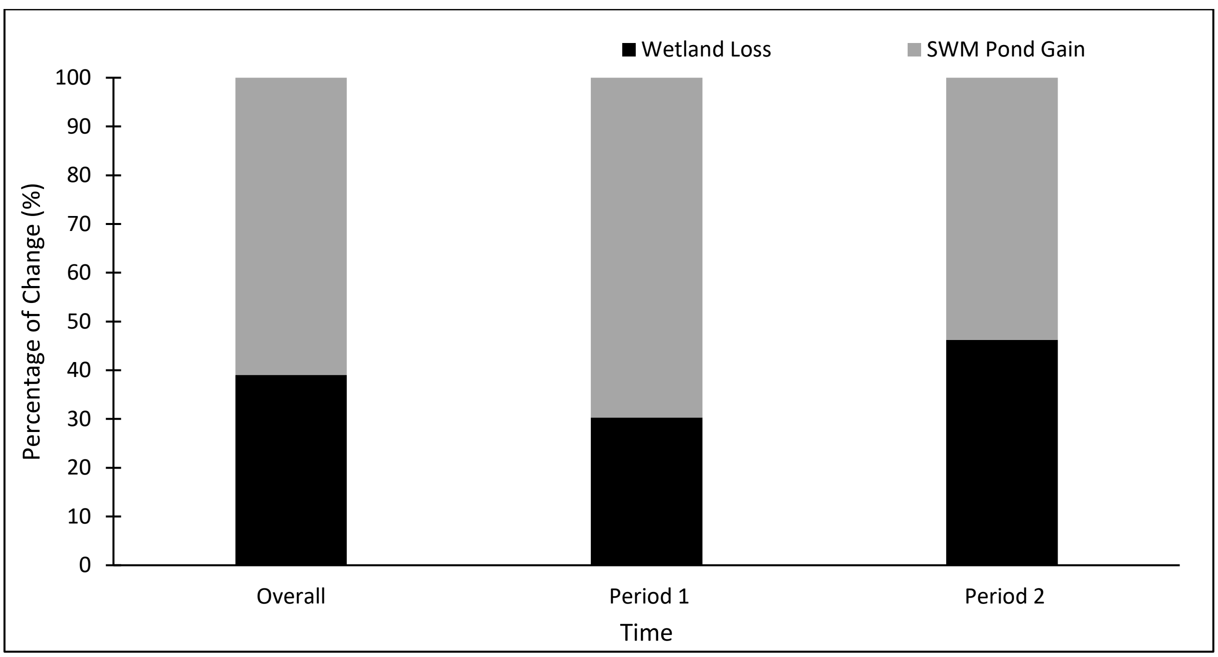

Figure 4-8. Wetland loss and SWM pond gain as a percent of the overall change during the period from 2002-2011, in addition to conditional probabilities given the time period, for all municipalities combined. Period 1 = 2002-2005/2006, and period 2 = 2005/2006-2011.

Projected Wetland Conversion

A Pearson-type test for the goodness of fit for the Markov model reveals that this model fits the data well, as indicated by non-significant differences between the observed and expected number of transitions between each set of states, and small deviance values for these transitions (Table 4-2). An initially generated simple Markov model (i.e. without the individual consideration of land use and land cover covariates) suggests that wetland area will likely continue to be lost in the future (Table 4-3). Under these conditions, by 2020-2032 there is a 4.66% projected probability of any wetland having been converted to non-wetland (Table 4-3). Meanwhile, the probability of SWM pond generation is expected to increase the total area of SWM ponds by 2.94 times their original area (Table 4-3).

When land use and land cover covariates are incorporated into the model, differences are seen in terms of the expected area of wetlands and SWM ponds by t = 6, or approximately 2026 (Figure 4-9). Wetland area will likely decrease the most if the effect of the "Extraction" land use was applied across the landscape, while all other covariates show slightly less wetland loss than when all land use and land covers are considered. Meanwhile, SWM pond gain is expected to be the greatest if the effect of the "Undifferentiated & Tilled" land use were applied across the landscape. SWM pond gain is expected to be of lower magnitude when the individual effects of all other land use and land cover covariates are considered, compared to the aggregated effect of all land use and land covers. The smallest projected area of SWM ponds is expected if the effect of "Extraction" and "Transportation" land uses were applied across the landscape.

Table 4-2. Pearson-type statistics, which give the goodness-of-fit for the general model without land use covariates included. “Obs.” is the observed number of transitions, “Exp.” is the expected number of transitions, and “Dev.” is the deviance between observed and expected values.

Initial State	Present State	Time = 1 Obs.	Time = 1 Exp.	Time = 1 Dev.	Time = 2 Obs.	Time = 2 Exp.	Time = 2 Dev.
Wetland	Wetland	22054	22062.24	-0.003	21887	21878.71	0.003
	Non-Wetland	185	176.227	0.437	166	174.761	-0.439
	SWM pond	0	0.531	-0.531	1	0.526	0.427
Non-Wetland	Non-Wetland	533240	528373.1	< -0.001	528370	528367.1	<0.001
	SWM pond	191	187.947	0.050	185	187.945	-0.046
SWM pond	Non-Wetland	0	0.417	-0.417	1	0.582	0.300
	SWM pond	484	483.583	<0.001	674	674.418	<0.001
Statistic	2.653						
P	0.617						
DF Lower	4						
DF Upper	8						

Table 4-3. Total wetland area (ha*100) at the beginning of the study period (2002), versus the projected wetland area in approximately 2026 (t=6), where one time-step equals approximately 4 years, based on Markov model-projected transition probabilities and the original wetland extent.

		Wetlands	SWM Ponds
Area (ha)	**Original (2002)**	9370.72	151.06
	Projected (2026)	8932.63	444.82

Table 4-4. Projected proportion of land use and land cover change from the initial state in 2002 (t = 0) to the present/projected state in 2005/06 (t = 1) and approximately 2026 (t = 6). Projected land use and land cover change is shown for all cases with and without the consideration of land use and land cover covariates.

Initial State (t=0) →		**Wetland**			**Non-Wetland**		**SWM pond**	
Present State (t=1 or t=6) →		**Wetland**	**Non-Wetland**	**SWM pond**	**Non-Wetland**	**SWM pond**	**Non-Wetland**	**SWM pond**
All Land Use and Land Cover Covariates	P(t=1)	0.992	0.008	<0.001	~1.000	<0.001	0.001	0.999
	P(t=6)	0.953	0.047	<0.001	0.998	0.002	0.005	0.995
Forest	P(t=1)	0.996	0.004	<0.001	~1.000	<0.001	<0.001	~1.000
	P(t=6)	0.977	0.023	<0.001	0.999	0.001	<0.001	~1.000
Transportation	P(t=1)	0.996	0.004	<0.001	~1.000	<0.001	0.001	0.999
	P(t=6)	0.976	0.024	<0.001	~1.000	<0.001	0.004	0.996
Built	P(t=1)	0.996	0.004	<0.001	~1.000	<0.001	<0.001	~1.000
	P(t=6)	0.979	0.021	<0.001	0.999	0.001	<0.001	~1.000
Pervious	P(t=1)	0.997	0.003	<0.001	~1.000	<0.000	<0.001	~1.000
	P(t=6)	0.984	0.016	<0.001	0.999	0.001	<0.001	~1.000
Impervious	P(t=1)	0.995	0.005	<0.001	~1.000	<0.001	<0.001	~1.000
	P(t=6)	0.967	0.032	<0.001	0.999	0.001	<0.001	~1.000
Extraction	P(t=1)	0.968	0.032	<0.001	~1.000	<0.001	<0.001	~1.000
	P(t=6)	0.821	0.179	<0.001	~1.000	<0.001	0.001	0.999
Undifferentiated & Tilled	P(t=1)	0.998	0.002	<0.001	0.999	0.001	<0.001	~1.000
	P(t=6)	0.986	0.014	<0.001	0.997	0.003	<0.001	~1.000

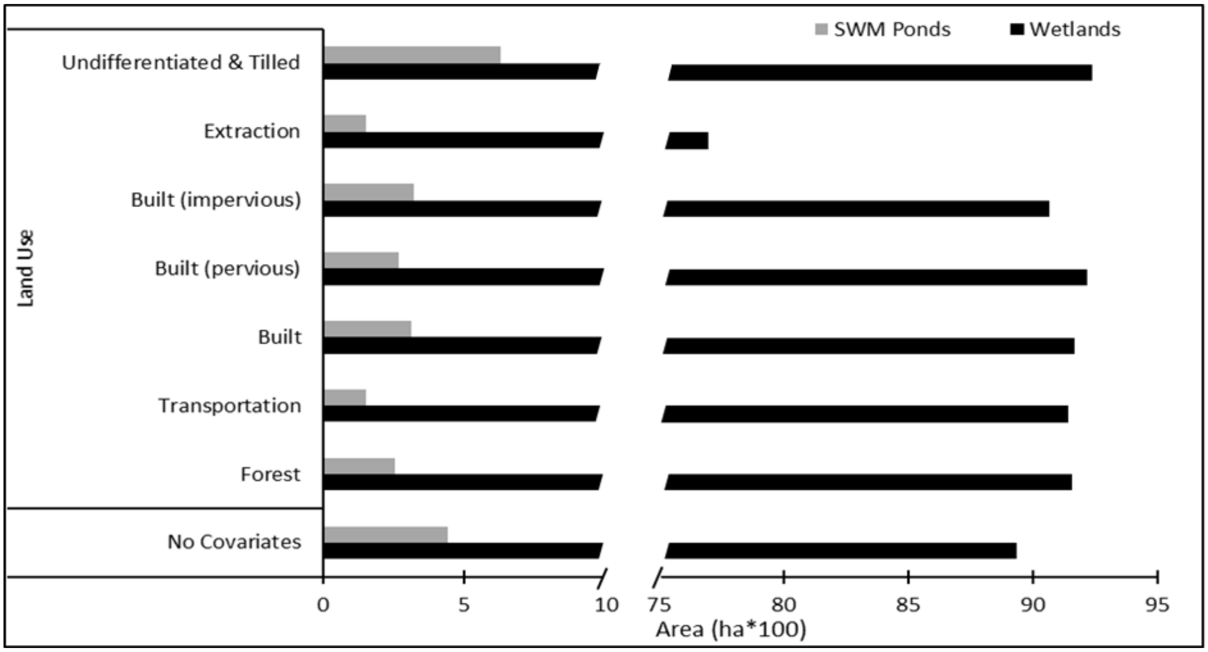

Figure 4-9. Projected total area (ha*100) of wetlands and SWM ponds in approximately 2026 (t=6, where one time-step equals approximately 4 years and the starting year is 2002), based on Markov-model calculated transition probabilities. Projections are calculated both with and without the effect of land use and land cover covariates considered.

4.5 Discussion

First, we examined trends in the loss of wetlands and gain of SWM ponds in seven southern Ontario municipalities between 2002-2011. We found a total of 95.45 ha of wetland loss, most of which was concentrated among small wetlands (< 2 ha). Under the Ontario Wetland Evaluation System, these small wetlands are generally not considered for Provincially Significant Wetland status, and are therefore not protected under the *Planning Act* (Government of Ontario, 2014, 2019; Schulte-Hostedde et al., 2007). While this loss occurred, there were also 111.64 ha of SWM ponds created, which indicates that no net loss of area was observed for combined wetlands and SWM ponds. Although this trend of greater area of SWM ponds gained than wetlands lost was observed in most municipalities, the created SWM ponds tended to be smaller than the lost wetlands. Also important to note is that SWM ponds are generally not equal to wetlands in ecohydrological function (see "*Historical Wetland Conversion"*).

Second, we combined the trends of wetland loss and SWM pond creation to predict what the future of Ontario's wetland landscape may be under the status quo. Assuming current trends of conversion persist, we expect that by 2026, 438.09 ha of wetlands will be lost and 293.76 ha of SWM ponds will be created. These estimates are based on probabilities of conversion and the original (2002) wetland, SWM pond, and non-wetland extent. The greater total wetland loss than SWM pond gain is expected because these estimates also include conversion to and from non-wetland land uses. Given the observed trends, it is expected that the majority of this change will continue to occur among small wetlands and SWM ponds (< 2 ha).

Third, we incorporated land use and land cover types to determine what may act as drivers of the projected wetland loss and SWM pond gain. The greatest of wetland loss is expected within the "Extraction" land use. Although this land classification may include both aggregate and peat extraction, Cheng & Lee (2008) found aggregate extraction (pits/quarries) to be a significant cause for land use conversion in Southern Ontario's Greenbelt (OMNRF, 2015). Given that Ducks Unlimited Canada (2010) found that only 28% of Southern Ontario's wetlands remain, it is unlikely that significant peat extraction is taking place within the study area, meaning that the projected conversion can likely be attributed to aggregate extraction. After the "Extraction" land use, the next greatest wetland losses are projected when the effect of the "Built (impervious)" land use is considered. Conversely, the greatest SWM pond gain is projected to occur within the "Undifferentiated & Tilled" land use, which includes agricultural lands, urban brownfields, and others.

Historical Wetland Conversion

First, the total area of SWM ponds gained was greater than the total area of wetlands lost. This was unexpected finding because of the uncoordinated wetland policy approach in Ontario that does not amount to an overarching strategy in response to wetland losses (Ducks Unlimited Canada et al., 2012). In fact, there is no evidence that the gain of SWM ponds was directly connected to losses of wetlands. Additionally, while in most municipalities SWM pond gains were higher than wetland losses, this was not the case in London.

Schulte-Hostedde et al. (2007) found that the implementation of protection was difficult for wetlands designated as "Locally Significant" in London, partially as most of these wetlands were zoned as Agriculture. This finding indicates a link between wetland conversion and municipal-level policies, which were not directly examined in this study. Further, Section 2.1.7 of the Provincial Policy Statement (2005) stated that "nothing in Policy 2.1 is intended to limit the ability of existing agricultural uses to continue." This supports the strength of agricultural land zoning as it predominates over local significance designation for wetlands, especially in agriculturally dominant municipalities like London (Government of Ontario, 2005b).

Second, and as expected, wetland loss was concentrated among small wetlands, likely due to the failure of the Ontario Wetland Evaluation System to protect wetlands smaller than 2 ha (Government of Ontario, 2014; Schulte-Hostedde et al., 2007). This finding is concerning both ecologically and hydrologically. From an ecological perspective, small wetlands are valuable for the maintenance of biodiversity, as discussed by Semlitsch & Bodie (1998) and supported by Houlahan et al.'s (2006) finding that these wetlands tend to host the least frequently occurring plant species. Further, Keitt et al. (1997) found that small habitat patches (i.e. wetlands) show large per-area contributions to connectivity, meaning that they may be able to act as stepping stones between larger wetlands, thereby playing an important role in ecologically-important landscape connectivity (Saura, Bodin & Fortin, 2014).

From a hydrologic perspective, small and geographically isolated wetlands are important for water retention and infiltration, and have a strong influence on downstream water quality (Marton et al., 2015; McLaughlin, Kaplan & Cohen, 2014). The retention and infiltration capacity of small wetlands is supported by McLaughlin et al.'s (2014) simulation of water dynamics in geographically isolated wetlands, which highlights these wetland's ability to reduce variability in the water table and in base flow. Through a survey of literature, Marton et al. (2015) found that these wetlands support disproportionately high rates of biogeochemical processing, given the size of their perimeters relative to area. This finding highlights the importance of small and isolated wetlands to reduce loads of nutrients, pollutants, and sediment to downstream waters, and the overall need to preserve these ecosystems (Marton et al., 2015).

The observed loss of small wetlands aligns with critiques of the Ontario Wetland Evaluation System that were discussed by Schulte-Hostedde et al. (2007). These include the failure to protect small wetlands, provide regular monitoring, the presence of an unclear rating system for wetland significance, and the fact that there are wetlands that have yet to be evaluated (Schulte-Hostedde et al., 2007). This trend of small wetland loss, and sparse protection for such wetlands aligns with broader trends across North America (Goldberg & Reiss, 2016; Serran & Creed, 2015; Semlitsch & Bodie, 1998). As such, Semlitsch & Bodie (1998) argue for the need to preserve wetlands as small as 0.2 ha, until more information is available on the biological implications of their loss. This recommendation aligns with Creed et al. (2017), who advocate for a default protection strategy for regions with high historic loss of vulnerable waters, as is the case for southern Ontario's wetlands (Ducks Unlimited Canada, 2010; Snell, 1987).

It also appears that these lost small wetlands are being replaced by even smaller SWM ponds, as indicated by the lower average area of gained SWM ponds. However, this observation could be an artefact, partially due to the minimum mapping unit constraint that exists for the wetland datasets (see "*limitations*") (OMNRF, 2015). Regardless, this trend towards the increased presence of small SWM ponds could have implications for biodiversity, as small and large habitats (i.e. wetlands) are known to hold different ecological roles (Keitt et al., 1997; Uden, Hellman, Angeler & Allen, 2014). More specifically, the increased occurrence of smaller SWM ponds may be the result of landscape fragmentation. Curtis (1956) and Moore (1962) defined fragmentation as the breaking apart of habitat, and this concept has generally been accepted to be a factor in species decline (Fahrig, 2019; Baxter-Gilbert, Riley, Lesbarrères & Litzgus, 2015; Keitt et al., 1997). Goldberg & Reiss (2016) discuss landscape fragmentation and overall re-organization of the landscape as a concerning result of no-net-loss policies, such as those proposed for Ontario by OMNRF (2017).

Despite the discussed concerns about fragmentation, Fahrig (2019) cautions that fragmentation is not unequivocally detrimental to species. In fact, they argue that fragmentation often has weak, positive effects on species, and that small habitats that exist in fragmented landscapes remain beneficial for conservation (Fahrig, 2019). Fahrig's (2019) argument bolsters support for the need to conserve small natural wetlands. Nevertheless, the observed trends of wetland loss and the creation of SWM ponds still likely indicate a potential negative trend for species, due to the lower biophysical quality of SWM ponds compared to wetlands (Clevenot, Carre & Pech. 2018; Sievers et al., 2018; Tixier et al., 2012; Moore et al., 2011).

Emphasizing the importance of small habitat patches, these features are known to have a potentially important role as stepping-stones to enhance landscape connectivity (Saura et al., 2014). It is possible that both small wetlands and SWM ponds could fulfill this stepping-stone role (Saura et al., 2014). However, it is unclear if small conserved wetlands or created SWM ponds would act as stepping-stones in a landscape that may not have larger wetlands remaining for them to connect (Ducks Unlimited Canada, 2010; Snell, 1987)

Overall, the results from the current study on past wetland conversion points towards several areas for future research. As there is no evidence to support a direct link between the gain of SWM ponds and loss of wetlands, an examination on documentation of such decisions may be warranted. Since research by Schulte-Hostedde et al. (2007) found a link between wetland protection difficulties and zoning, and the current study found municipal-scale differences in wetland conversion, these trends may be best investigated through an analysis of Zoning By-Laws and Official Plans. To capture more accurate rates of conversion, a greater variety of constructed ponds could be examined. This includes ponds on agricultural fields, such as the agricultural irrigation reuse pits examined by Uden et al. (2014) in their study of connectivity. There is also an opportunity to examine if small wetlands and SWM ponds fulfill the potential stepping-stone role that they may hold, given that there may be few large wetlands remaining for them to connect (Saura et al., 2014; Ducks Unlimited Canada, 2010; Snell 1987).

Projected Wetland Conversion

The increased wetland conversion expected under extractive and urban land uses supports the expectation that wetland loss would be more probable in human-dominated land uses. This also aligns with Ducks Unlimited Canada’s (2010) finding of built-up lands as a significant factor for wetland loss within a highly urban region of southern Ontario (Golden Horseshoe Region). The dominance of urbanization within our study area is supported by Cheng & Lee’s (2008) finding that change to urban/built-up land uses was the most significant land conversion in Ontario’s Greenbelt from 1993 to 2007, including conversion from high-value agricultural land.

As aggregate extraction is considered by the *Provincial Policy Statement* to be an interim land use for which remediation/rehabilitation should occur, the relatively low magnitude of projected SWM pond gain in connection to extractive land uses is somewhat surprising (Government of Ontario, 1997). This surprising result might be addressed by revisiting the fairly restrictive inclusion criteria for constructed pond types in the current study. If inclusion criteria would have been broader and additional pond types were considered in this analysis, more pond creation may have been observed. Further research in this direction could better examine the efficacy of extractive rehabilitation in terms of the preservation of wetland area, as it is known that aggregate extraction sites that are adequately rehabilitated as constructed wetlands can help to preserve biodiversity (Santoul, Gaujard, Angélibert, Mastrorillo & Céréghino, 2009). Given trends found by Cheng & Lee (2008) of urbanization in southern Ontario, aggregate extraction is likely to continue alongside demand for building materials, meaning that the effect of this land use on wetland loss is likely to continue. This emphasizes the importance of work to preserve and improve rehabilitation of biodiversity-supporting wetlands in post-extractive areas.

The trend towards urbanization in Southern Ontario may also explain why a relatively low magnitude of wetland loss is projected for the “Undifferentiated & Tilled” land use, which includes agriculture. Given findings by Ducks Unlimited Canada (2010) and Snell (1987) that agriculture has acted as a significant driver of Ontario’s wetland loss in the past, it would be expected that wetland losses would be greater if the individual effect of agricultural land uses could be examined. However, the individual examination of agricultural land uses was not possible as the “Undifferentiated” land use also includes urban brownfields, power line corridors, the edge of transportation corridors, and forest clearings, while the “Tilled” class was not available at all time steps (Ducks Unlimited Canada, 2010; OMNRF, 2015). Regardless, it is possible that these loss projections are lower than expected as a majority of wetlands have already been lost to agriculture in Southern Ontario (Ducks Unlimited Canada, 2010; Snell, 1987). It is also possible that as a result of urbanization, agriculturally-induced wetland loss may be occurring further outside of historical limits, as conversion to agricultural lands is no longer predominant within the study area (Cheng & Lee, 2008). This potential side-effect of urban sprawl may warrant further study and action to prevent the continued loss of wetlands in areas that have historically not seen as much wetland conversion as Southern Ontario.

The lower magnitude of loss projected among the “Built (pervious)”, and “Transportation” land uses fail to support expectations to the same degree as the discussed land uses. As the “Undifferentiated” class includes the edge of transportation corridors, it is possible that wetland losses caused by construction of roadways are being accounted for outside of the “Transportation” class (Ducks Unlimited Canada, 2010; OMNRF, 2015). Other studies, including Ducks Unlimited Canada (2010) did not examine the “Transportation” SOLRIS class specifically, and overall, results for this land use are somewhat inconclusive. Despite the “Built” land use having high projected loss, pervious built lands have less projected loss than impervious built lands. A possible reason for the relative preservation of wetlands within these pervious urban areas could be their recognized utility for water infiltration in an urban environment, despite the direct use of wetlands for SWM purposes being prohibited in Ontario (OME, 2003; Schulte-Hostedde et al., 2007).

Although the magnitude of projected SWM pond creation among the “Undifferentiated & Tilled” land use was unexpected, such lands may provide an opportunity for SWM pond creation, especially given the need for SWM with the predominant conversion of agricultural lands for urban land uses that was found by Cheng & Lee (2008). The opportunistic nature of certain lands for SWM pond creation may also provide an explanation for the relatively high projected creation of SWM ponds when the effect of the "Forest" land cover is considered. It is possible that conversion is occurring in unprotected forests for the creation of SWM ponds, which is another concerning potential effect of urban sprawl that may warrant further study.

It is again possible that an even higher rate of SWM pond creation would be observed in the "Undifferentiated & Tilled" land use if additional pond types were considered. The relatively high magnitude of SWM pond generation when the effects of all "Built" land uses are considered appears to be reasonable, as SWM is implemented to counteract effects of urban runoff (Schulte-Hostedde et al., 2007). Meanwhile, the low magnitude of SWM pond creation when the effect of the "Transportation" land use is considered may simply be due to this being an inappropriate location for an SWM pond. Overall, the projected wetland conversion results from the Markov model are informative in terms of where wetland losses and SWM pond gains may continue to be the greatest, and where prioritization may be necessary to preserve wetland area.

Limitations

In addition to those discussed previously, there are limitations that are inherent to remote sensing and GIS methods used in our study, which can generally be improved with on-ground verification (Dahl, 2004). Such limitations are evident in the minimum mapping unit (MMU) of data used that includes the SOLRIS 2.0 change inventory, which could not reliably detect land use and land cover changes smaller than 0.5 ha (OMNRF, 2015). This is an improvement in comparison to the 10 ha minimum detection limit in Ducks Unlimited Canada's (2010), and Snell's (1987) studies of wetland loss. However, given that nearly all loss occurred in wetlands smaller than 2 ha and only change greater than 0.5 ha could be reliably detected, our results are still likely underestimates of actual change. This is a concern mirrored by Cheng & Lee (2008) and Ducks Unlimited Canada (2010). The conservative nature of wetland loss estimates should be a key consideration for practitioners, who should not use them to substantiate further wetland loss.

Additionally, the predictive approach we took to modelling projected wetland loss and SWM pond gain entails some uncertainty. In general, models are a simplified version of reality that help us to interpret processes and predict how they may change under future scenarios (DeAngelis & Waterhouse, 1987; Whittingham, Stephens, Bradbury & Freckleton, 2006). While the Markov model used in this study is informative, a variety of factors beyond its scope, such as policy, economics, and climate, have the ability to modify future patterns of wetland conversion (Dahl & Watmough, 2007; Werner, Johnson & Guntenspergen, 2013; Wiebusch & Lant, 2017). Further, our assumption that the Markov property was true for our data potentially rendered the results from the current study only approximately correct. However, it is common for this assumption to not be tested and instead be treated as a model limitation, as discussed by Koen et al. (2010).

Further, some variation is present in the time steps used to calculate projections, which we calculated based on a transition probability matrix for six time-steps from the original (2002). The average time step of the input data was approximately four years, but this varied based on the availability of imagery for the SOLRIS 2.0 change inventory, and thus, projections do not correspond exactly to the year 2026 (OMNRF, 2015). While the Markov model may be informative for applications like the prioritization of land uses within which wetland conservation is needed, these limitations should be kept in mind, and results should not be used to justify any further wetland conversion.

4.6 Conclusion

Our results show a continued loss of wetlands, with trends towards their replacement by SWM ponds. Losses are concentrated among wetlands that are generally not protected by provincial policies because of their small size, and even then, this is likely an underestimate due to technical limitations preventing the detection of change events smaller than 0.5 ha. Loss of wetlands is likely to continue, and may be most likely in areas with extractive and urban land uses. Conversely, SWM pond gain appears to be most likely in a class of land uses that include agriculture, urban areas, and forested areas. We project continued loss of wetlands, which is especially problematic given the small fraction of historical wetlands that remain in southern Ontario and the critical role these ecosystems play in flood retention, a key consideration in the context of climate change adaptation planning. To protect human life and property from pressing issues that include the increased likelihood of flooding as climate change progresses, policy and decision makers should prioritize the protection of all wetlands, including small wetlands, in addition to constructing SWM ponds to manage urban runoff.

5.0 Manuscript 2 - Connectivity Contributions of Wetlands and Stormwater Management Ponds in Urbanized Landscapes.

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5.1 Abstract

Wetlands in human-dominated landscapes are affected by degradation and destruction, which threatens the critical ecosystem services that they provide, such as water retention and filtration, and may lead to reduced connectivity of these important ecosystems. While stormwater management (SWM) ponds are designed to control urban runoff and contaminants, they do not replace the full complement of ecosystem services and function of wetlands. Since landscape connectivity influences wetland function and the subsequent ecosystem services provisioned, we used a graph-theory-based approach to analyze connectivity of wetlands and SWM ponds in seven southern Ontario municipalities. We considered changes in connectivity through time, in addition to the effect of SWM ponds on connectivity. We calculated the number of links and number of components at the landscape-level, and the probability of connection and two of its components (dPCflux and dPCconnector) at the wetland-level. Results suggest that connectivity has decreased with wetland loss, while SWM pond construction has improved connectivity. Wetlands appear to be more connected over the landscape as a whole, while SWM ponds may act as stepping-stones between wetlands. Our results point towards the need to preserve wetlands in order to protect the critical ecosystem services they provide, while it may be possible for improvements in connectivity to be achieved through strategic placement of SWM ponds.

Key Words: Landscape Connectivity, Wetland Ecosystems, Wetland Conversion.

5.2 Introduction

Globally, landscapes are now dominated by human populations and associated land uses. It is estimated that half of global wetlands have been lost, which is mostly attributable to wetland conversion in the global north over the first half of the twentieth century (Zedler & Kercher, 2005). Wetland loss can lead to a loss of connectivity at the landscape level, which prevents species from being able to move between the habitats they may require for different functions and life stages, such as foraging or nesting habitat (Thorslund et al., 2017; Haxton, 2000). Without connectivity, populations can become isolated, especially due to the high mortality associated with dispersal (Baxter-Gilbert, Riley, Lesbarrères & Litzgus, 2015; Mackinnon et al., 2005; Steen & Gibbs, 2004). Decline of isolated populations can also follow due to a lack of gene flow and subsequent reductions in genetic diversity (Reh & Seitz, 1990).

In southern Ontario, wetlands are ecosystems for which connectivity loss is relevant, as there are few remaining (Ducks Unlimited Canada, 2010; Snell, 1987). As wetland loss has been poorly documented, estimates generally vary and are often conservative in nature (Dahl, 1990; Ducks Unlimited Canada, 2010; Snell, 1987; Zedler & Kercher, 2005). However, at least 72% of southern Ontario's pre-European settlement wetlands have been lost (Ducks Unlimited Canada, 2010). This loss has been very deliberate, and has generally occurred via drainage for agricultural purposes, as well as more recent urbanization and resource extraction (Dahl, 2006; Schulte-Hostedde, Walters, Powell & Shrubsole, 2007).

Wetlands are also vulnerable to the effects of climate change, which can lead to loss of wetland area and connectivity (Werner, Johnson & Guntenspergen, 2013; Zedler & Kercher, 2005). These losses are likely to result from the alteration of water volumes that will occur with increases in extreme weather events such as drought and flooding, which may ultimately reduce wetland area and integrity (Werner, Johnson & Guntenspergen, 2013; Wright, 2010; Zedler &

Kercher, 2005). Meanwhile, large-scale wetland connectivity is critical to allow species to adapt to climatic change as their ranges expand towards higher latitudes (Root & Scheider, 2006). More locally, connectivity allows populations to re-establish following disturbances that are anticipated to increase in frequency and magnitude (Humphries, Thomas & Speakman, 2002; Opdam & Wascher, 2004).

In addition to the potential for wetland loss to result in connectivity loss, the replacement of natural ecosystems with less pervious land cover and land use types (e.g., agricultural row crops or residential areas) tends to lead to reduced infiltration and retention of stormwater (Bronstert, Niehoff & Gerd, 2002). The replacement of these flood-preventing natural ecosystems is especially concerning for the resilience of urban communities, given that storm events are expected to increase in frequency and magnitude as climate change progresses (Erwin, 2009; Opdam & Wascher, 2004). To manage flooding in urbanized areas, stormwater management (SWM) ponds are often included or mandated in urban site planning (Schulte-Hostedde et al., 2007).

Although SWM ponds are effective at retaining urban stormwater, and newer ponds tend to include habitat features that are described as "naturalized", even these ponds are not able to fully to replace all functions provided by wetlands (Tixier, Rochfort, Grapentine, Marsalek & Lafont, 2012). In addition to water retention during storms, a key function of SWM ponds is to allow for contaminant and sediment removal from urban runoff (Tixier et al., 2012; Moore, Hunt, Burchell & Hathaway, 2011). This means that SWM ponds tend to accumulate contaminants and excess nutrients (Tixier et al., 2012; Moore, Hunt, Burchell & Hathaway, 2011). The presence of pollutants can affect the ecological quality of these ponds, in addition to other biotic and abiotic factors that include pond shape, which has led some authors to question if SWM ponds may function as ecological traps (Clevenot, Carre & Pech, 2018; Sievers et al., 2018). Sievers et al. (2018) found evidence that SWM ponds can act as ecological traps for tadpoles, as they had lower survival, and were less responsive to predator olfactory cues when raised in more polluted SWM ponds. This finding indicates the need to mitigate the potential ecological costs of SWM ponds, which will require more research on their function as ecological traps (Sievers et al., 2018).

In the context of their hydrologic function, to prevent contaminants from entering groundwater, SWM ponds are generally equipped with an underlying impervious liner (OME, 2003). Although this is a necessary protective measure, it also means that SWM ponds do not contribute to groundwater recharge. Conversely, this is a hydrologic function that wetlands fulfill, which further indicates that SWM ponds are not biophysically equal to wetlands (Rooney et al., 2014).

While SWM tends to occur at the municipal level, there are a host of policies at various levels of government that regulate the protection of wetlands in Ontario. Part of this policy system is the Ontario Wetland Evaluation System, which is used to determine if a wetland is eligible for designation as "Provincially Significant". Wetlands that are evaluated and qualify for provincial significance are awarded protection under the *Planning Act*'s Provincial Policy Statement (Government of Ontario, 2019c; Schulte-Hostedde et al., 2007). Also under the Ontario Wetland Evaluation System, wetlands that meet additional criteria can be considered part of a wetland complex if they are within a maximum distance of 750 m from one another (Government of Ontario, 2014; Schulte-Hostedde et al., 2007). The formation of a wetland complex may be one indicator of connectivity, as these are groupings of wetlands that are commonly related and have similarities in their biological, social and/or hydrological function

(Government of Ontario, 2014). While the Government of Ontario (2014) did not justify the chosen 750 m distance threshold for a wetland complex with empirical data, it is used as a basis for management decisions, which makes it a useful threshold distance for our analyses.

Though it is known that SWM ponds are not of the same habitat quality as wetlands, and do not exactly mimic all of their ecological functions, they may contribute to wetland connectivity at a landscape level (Moore et al., 2011; Tixier et al., 2012; Uden, Hellman, Angeler & Allen, 2014). However, little work has been performed to examine the function of SWM ponds from a wetland connectivity perspective, despite the potential impacts of connectivity on wetland-provisioned ecosystem services (Moore et al., 2011; Rooney et al., 2014; Tixier et al., 2012; Uden, Hellman, Angeler & Allen, 2014). To close this knowledge gap, we apply ecological connectivity models to determine changes in wetland connectivity at the landscape level over time, both for wetlands alone, and with the inclusion of SWM ponds. Our work is guided by the following questions:

(1) Has loss of wetlands over the recent past led to decreases in wetland connectivity at the landscape level? Habitat loss is a known driver of connectivity loss, and Ducks Unlimited Canada (2010) found that wetlands continue to be lost in Ontario (Cushman, 2006; Haxton, 2000; Keitt et al., 1997). As such, we expect connectivity to decrease over the study period from 2002-2011.

(2) Are SWM ponds less connected to other SWM ponds and wetlands, relative to how connected wetlands are to other wetlands and SWM ponds? Little work has examined the connectivity contributions of constructed ponds, though it is known that SWM ponds are of lesser habitat quality than wetlands (Moore et al., 2011; Rooney et al., 2014; Tixier et al., 2012;

Uden et al., 2014). Following this trend, we expect that SWM ponds will be less connected than wetlands.

(3) Has creation of SWM ponds over the recent past led to an increase in wetland connectivity at the landscape level, when both wetlands and SWM ponds are considered? Uden et al. (2014) found that connectivity increased when agricultural reuse pits were included in their analysis, and we expect the same with the inclusion of SWM ponds in our study.

5.3 Methods

Study Area

This study examines wetland change among seven Southern Ontario municipalities, including the City of Cambridge, City of Kitchener, City of London, City of Markham, City of Vaughan, City of Waterloo, and Town of Whitby (Figure 5-1). These municipalities were chosen based on the availability of SWM pond datasets. The broad study area of southern Ontario is densely populated, which has caused a great deal of developmental and agricultural pressures for natural systems (Cheng & Lee, 2008; Ducks Unlimited Canada, 2010). These pressures have led to the loss of over 72 % of the pre-European settlement (prior to 1800) wetland area (Ducks Unlimited Canada, 2010; Snell, 1987). Municipalities within this broad study area were chosen based on the availability of SWM pond datasets that contained sufficient information for further analysis, as explained in "*Spatial Data"*.

Figure 5-1. Map of the study area municipalities within the broader study area of southern Ontario. Study area municipalities are those for which SWM pond datasets with the year of construction and pond type were available. Basemap source: Esri, Garmin, GEBCO, NOAA, NGDC, and other contributors.

Spatial Data

To address wetland change, this study combined publicly available data on the provincial wetland extent (provincially-amalgamated data from a variety of sources, including Southern Ontario Land Resource Inventory System - SOLRIS 2.0), and land use changes (SOLRIS 2.0) (OMNRF, 2015). To incorporate the connectivity contributions of SWM ponds, SWM pond data that included information on pond type (e.g., wet SWM pond, or dry SWM pond) and year of pond construction were obtained from the seven study municipalities in southern Ontario (Figure 5-1). The SWM pond type is an important factor because not all SWM ponds are similar enough to a wetland to be compared to them. For example, dry SWM ponds are only designed to hold water for up to twenty-four hours after a storm (OME, 2003). As such, only ponds labelled as "wet", "wetland", hybrid", or "natural" were included. The distinction between wet, wetland, and hybrid ponds is in their depth, as wet ponds are deep and there are shallow aquatic plant zones around the perimeter, while wetland ponds are dominated by shallow zones, and hybrid ponds combine the two types (OME, 2003). The "natural" pond designation only occurred twice in the dataset, and although wetlands are not permitted for use as stormwater ponds under OME (2003), inclusion of these "natural" ponds in municipal SWM data may mean that they were previously modified for this purpose. As such, these two ponds were included in the analyses.

The minimum mappable unit for SOLRIS 2.0 is 0.5 ha, meaning that features below this size could not be reliably detected (OMNRF, 2015). Upon visual inspection, recorded SWM pond polygons were found to generally be overestimations of the actual pond area, and it was necessary to correct the area of SWM ponds. This correction was achieved via a heads-up digitization method that made use of aerial imagery (SWOOP and GTA Orthophotography Project) from spring months, when the ponds were assumed to be inundated with water. During this process, the Town of Whitby's dataset was also digitized into polygons, as it came in point format.

Temporal Period

Overall, the temporal period of this study extends from 2002 to 2011, which is based on the study period of the SOLRIS 2.0 land use change inventory and includes three time steps (OMNRF, 2015). Spatial data layers were created (Figure 5-2) that represent spatiotemporal changes in the combined system of wetlands and SWM ponds (Figure 5-3) for each time step in the study period. These time steps are such that $t = 0$ corresponds to the landscape in the year 2002 and includes SWM ponds constructed up to this date. Due to the available aerial imagery, the next time step, t = 1 corresponds to the landscape in 2005 in the City of London, City of Kitchener, City of Cambridge, and City of Waterloo; and 2006 in the City of Vaughan, City of Markham, and Town of Whitby. This time step (t =1) includes SWM ponds built from 2003 to 2005/06, depending on the municipality. The final time step, t = 2 includes change that occurred after t = 1, to detection in 2011. The wetland change detection in 2011 occurred in June in some municipalities, unlike the wetland detection at other time steps, which occurred in August and September. The June detection means that a great deal of additional wetland change could still have occurred in 2011, and as such, t =2 was only assumed to capture wetland change up to and including 2010, which was detected in 2011 and thus represents the landscape at this time. To remain consistent with this detection, only SWM ponds built from 2006/07 until 2010 were included in this time step (t = 2).

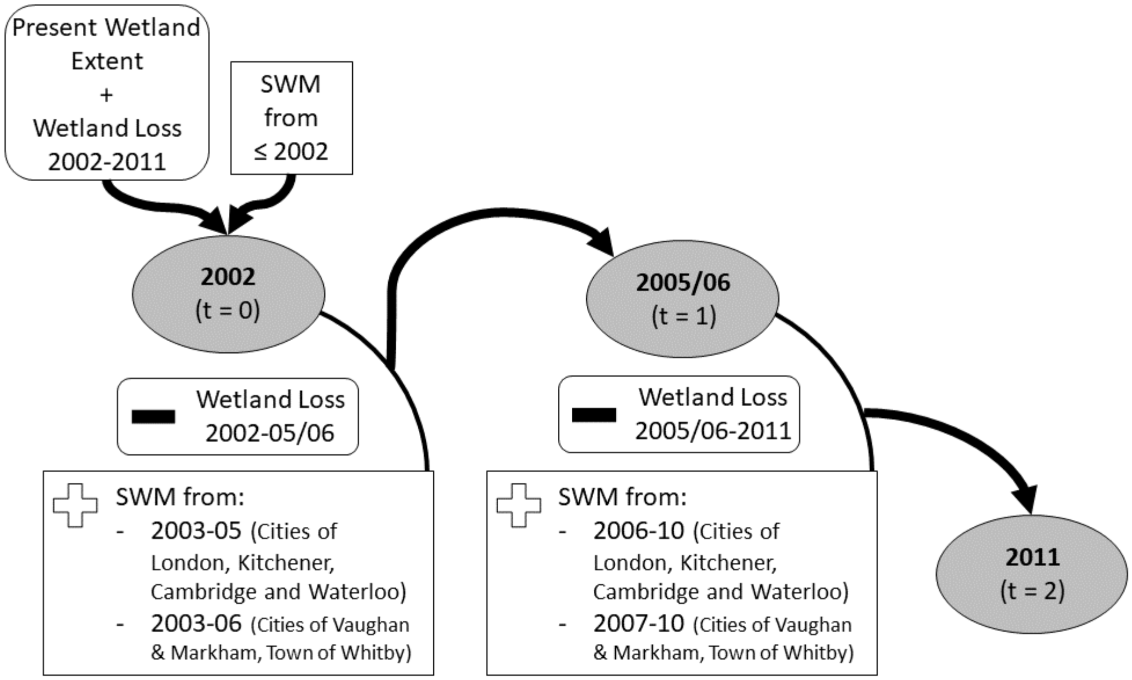

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Figure 5-2. Illustrated workflow of spatial data layer creation for each time step, including the subtraction of wetland loss and addition of SWM ponds by construction year, depending on the municipality.

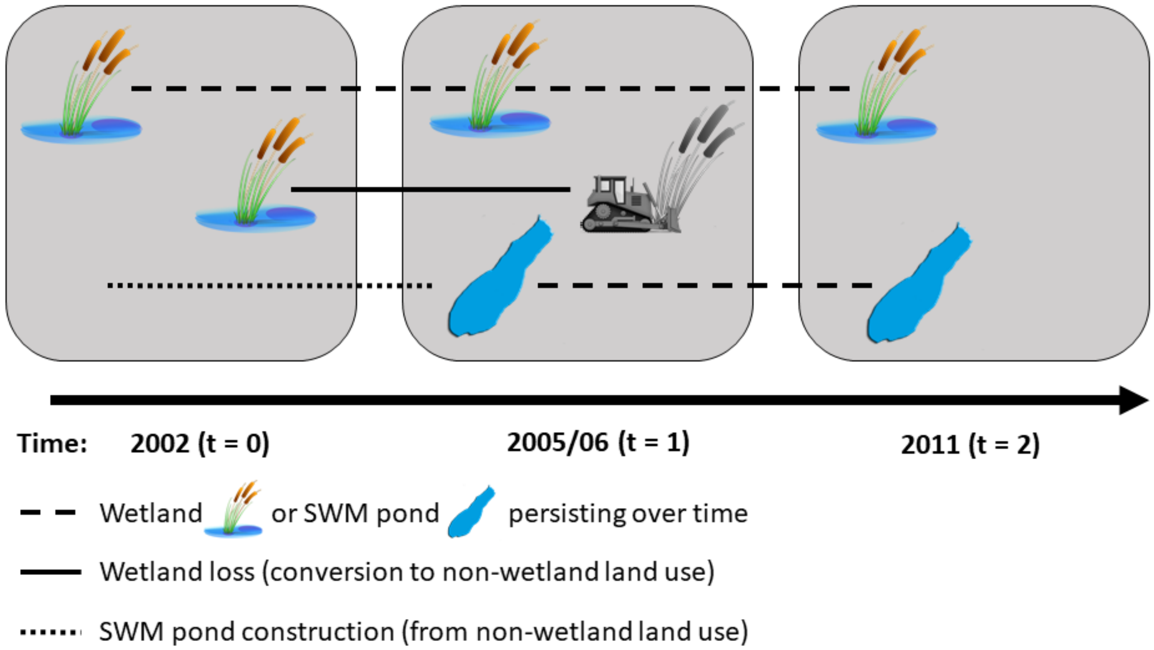

Figure 5-3. Illustration of observed spatiotemporal changes that occurred within the combined system of wetlands and SWM ponds over the study period.

Network Analysis

The graph-theory-based software Conefor 2.6 was used for all connectivity analysis in this study (Saura & Torné, 2012). Graph-theoretic models represent the landscape as a set of nodes and edges, which are the potential paths that an organism may take to disperse between a set of nodes (Saura, Estreguil, Mouton & Rodríguez-Freire, 2011; Pascual-Hortal & Saura, 2006; Urban & Keitt, 2001). This method is computationally powerful and also balances the required amount of input data and the detail provided by their results (Saura et al., 2011; Pascual-Hortal & Saura, 2006). As such, graph-theory based approaches are an effective means for connectivity modelling without the intensive data requirements that may come with other methods (Urban & Keitt, 2001).

To prepare data for this analysis, the “ID within distance” tool by Jenness (2016) was employed in ArcGIS 10.6 (ESRI, 2017). Within this tool, a maximum distance threshold of 750 m was used, with the "calculate from feature edges" option selected. This distance was chosen as it is how far wetlands may be from each other in order to be considered a functionally related "wetland complex" by the Ontario Wetland Evaluation System (Government of Ontario, 2014). These analyses were completed for the wetlands, and for the combined wetland-SWM pond systems at each of the three time-steps. Outputs of this analysis are node and distance files, where nodes represent habitat patches (i.e. wetlands), and distances represent the possible connections between them.

These outputs were then used to calculate landscape-level and wetland-level connectivity indices. Two landscape-level indices were calculated (Figure 5-4), the number of links (NL), and number of components (NC), again using the previously mentioned distance threshold of 750 metres. For the landscape-level indices, greater connection is shown by a greater NL value, which represents a larger total number of links between wetland nodes (Saura & Pascual-Hortal, 2007). Conversely, increasing wetland connectivity at the landscape level corresponds to a decreasing NC value. Within this index, a component is a set of wetland nodes where a path exists between every wetland pair, while an isolated wetland will make up its own component (Saura & Pascual-Hortal, 2007). As such, the most connected wetlands at the landscape-level will have NC = 1 (Saura & Pascual-Hortal, 2007).

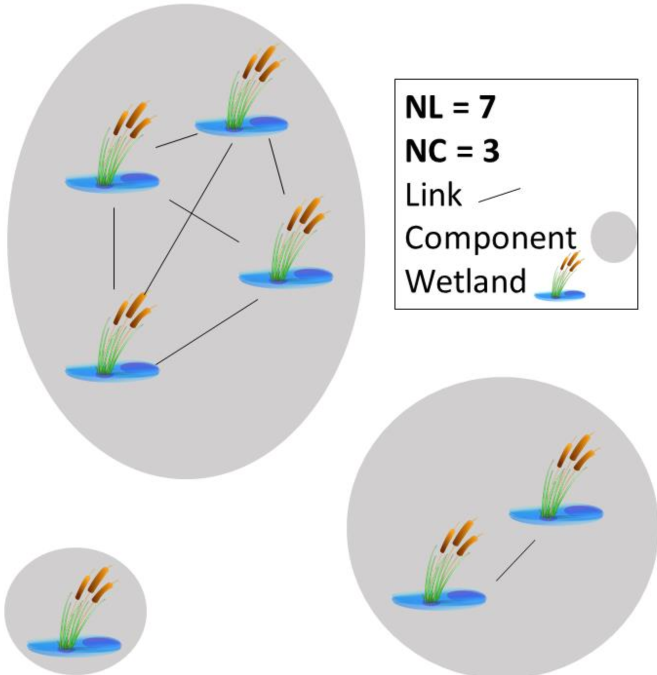

Figure 5-4. Illustration of the links and components formed between and among wetlands, and the resulting NL and NC indices.

At the wetland-level, the probability of connection (PC) index, and two of its components (dPCflux and dPCconnector) were calculated for the combined system of wetlands and SWM ponds at each time step. The results of these analyses were separated by wetlands and SWM ponds by joining the resulting table of PC values with the original wetland attributes (i.e. wetland type) in ArcGIS. The PC index is focused on individual wetlands, and is defined as the probability that two randomly placed points will fall within an interconnected habitat area, based on the specified distance threshold (Saura & Rubio, 2010). Values for this index range from zero to one, where high values indicate a more connected wetland and are given by:

$$PC = \frac{\sum_{i=1}^{n}\sum_{j=1}^{n} a_i\, a_j\, p_{ij}^{*}}{A_L^2}$$ (Saura & Pascual-Hortal, 2007)

In the context of this study, n is the total number of wetland nodes in landscape, a_i and a_j are area of wetlands i and j, A_L is the total area of the analysed landscape, and p^*_{ij} is the maximum product probability of all paths between patches *i* and *j* (i.e. the best path). If wetland nodes are close enough to one another, the maximum probability path is the direct step between the wetlands, but if they are further away, the maximum probability path will likely be comprised of several paths through stepping-stone wetlands (Figure 5-5). When two wetlands are fully isolated from one another, $p^*ij = 0$ (Saura & Pascual-Hortal, 2007).

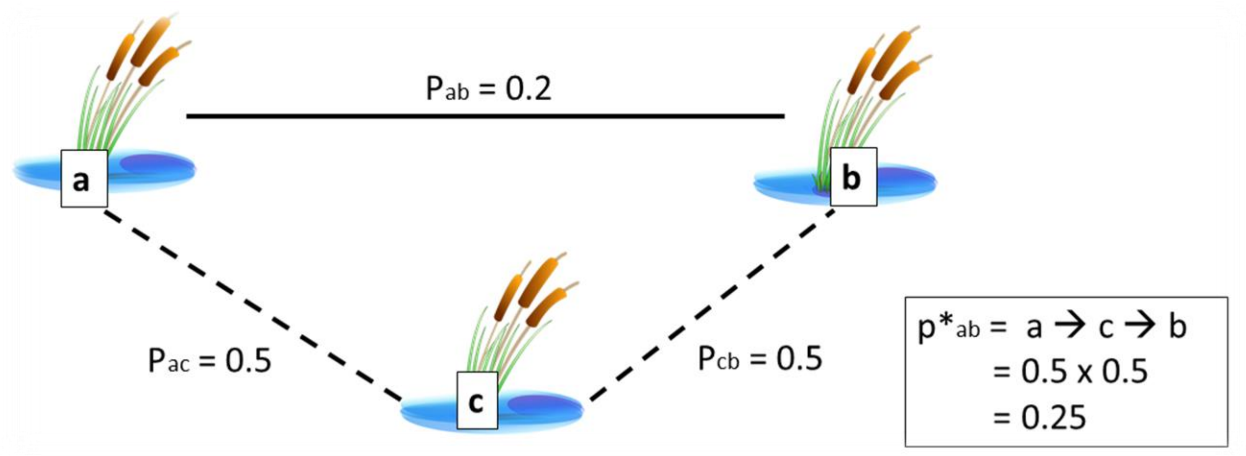

Figure 5-5. Conceptual illustration of possible connections between wetlands, and the calculation of p^*_{ab}, which is the maximum product probability (i.e. best path) between wetlands *a* and *b*. In this example, p^*_{ab} results in a greater probability of dispersal with the use of wetland *c* as a stepping stone between *a* and *b*, than does the direct dispersal path between *a* and *b* (p_{ab}) (Saura & Pascual-Hortal, 2007).

Further, the PC index is comprised of three components: dPCintra, dPCflux, and dPCconnector, where $dPC = dPCintra + dPCflux + dPCconnector$ (Saura & Torné, 2012). The PC index ranges in value from zero to one, with increasing values representing improved connectivity (Saura & Pascual-Hortal, 2007). The dPCflux and dPCconnector components are further analysed in this study, as they are focused on the way a habitat patch (i.e. wetland) functions across the landscape (Saura & Rubio, 2010). dPCintra focuses on connectivity within a single wetland, which is not the focus of this study, and therefore is not further investigated. The dPCflux component represents how connected a given wetland is to other wetlands across the landscape, but does not quantify the importance of that wetland for the maintenance of connectivity between others. This index varies based on the area of the given wetland, and its position on the landscape. The dPCconnector index represents how important a given wetland is as a connector for others, and is not size-dependent (Saura & Rubio, 2010).

Computation of the PC index is more demanding than the landscape-level indices, and the entire study area contained too many data points to make this analysis feasible at this scale (Saura & Pascual-Hortal, 2007). As such, these calculations were completed only for the City of Markham, which is closest to the average area of all the municipalities (mean municipal area = 19,689.10 ha, City of Markham area = 21,268.42 ha). PC calculations were completed using the previously mentioned 750 m maximum connection distance. The probability value of direct dispersal was set equal to 0.5, as is common practice (Herrera, Sabatino, Jaimes & Saura, 2017; Saura, Estreguil, Mouton & Rodríguez-Freire, 2011).

Using ArcMap 10.6, symbology of dPC index results was displayed as quantities with graduated symbols, calculated based on geometrical intervals with three classes, such that the same value categories applied to both wetlands and SWM ponds (ESRI, 2017). The geometrical interval classification scheme in ArcGIS generates breaks between the classes using a geometric series, meaning that a constant coefficient is multiplied to each value (ESRI, 2018). Intervals are calculated by subtracting the minimum from maximum values, and the geometric coefficient is calculated by dividing the previous interval by the current interval, while this coefficient can change to its inverse to optimize class ranges (ESRI, 2008). The geometrical interval method was chosen as it is designed for continuous data and ensures that classes remain consistent by minimizing the square sum of elements per class (ESRI, 2018). This ensures that each class contains approximately the same number of values, and maintains a consistent change between each class interval (ESRI, 2018). The classified dPC values were then labelled such that the lowest category (dPC = 0.000001 – 0.028627) represented wetlands that were defined as being the "least connected" and highest category (dPC = 0.255194 – 2.048408) represented wetlands that were the "most connected" (Figure 5-12).

Statistics

All statistical tests were performed in R statistical software version 3.5.2, using an alpha level of 0.05 (R Core Team, 2018). Tests were performed to examine differences in connectivity between wetlands and the combined wetland-SWM pond system, as well as over the three time-steps for each wetland system. The Pearson's Chi-Square test of independence was used to test for significant differences in the frequency-based NC and NL indices. The probability-based dPC, dPCflux, and dPCconnector indices, and the area of wetlands and SWM ponds in the City of Markham produced numerical values that did not follow a normal distribution, which was determined using the Shapiro-Wilks test of normality. As such, non-parametric tests were used. The Wilcoxon rank sum test was used to test for a difference between the mean probability indices of the two wetland types (wetlands and SWM ponds), as well as for differences between the mean area of each wetland type. The Kruskal-Wallis test was used to test for differences in the PC index components over time for each of the two wetland types.

5.4 Results

Over the nine-year study period from 2002-2011, wetland losses resulted in a 1.51 % (n = 114) decrease in the number of nodes (i.e., wetlands) for the wetland system (Table 5-1). Conversely, over the same period, SWM pond gains resulted in a 73.55 % (n = 178) increase in the number of nodes (i.e. SWM ponds) for the SWM pond system (Table 5-1). For the wetland-

SWM pond system, an overall increase of 0.82 % (n = 64) was observed in the number of nodes (Table 5-1).

Table 5-1. Observed number of nodes (i.e., wetlands) for wetlands, SWM ponds, and the combined wetlands and SWM ponds. Numbers are given for three times over the study period (2002-2011), for the aggregate of study area municipalities in southern Ontario.

Year	**Number of Nodes**		
	Wetlands	**SWM Ponds**	**Wetlands and SWM Ponds**
2002	7556	242	7798
2005/06	7482	334	7816
2011	7442	420	7862

Landscape Connectivity

Over the nine-year study period, there was a significant decline in NL (i.e., number of links) of 1.04 % for the wetland system (X^2 (2, n = 3) = 8.124, p = 0.017) (Figure 5-6, Table 5-2). A non-significant increase of 0.60 % in NL occurred for the combined system of wetlands and SWM ponds (X^2 (2, n = 3) = 2.724, p = 0.256) (Figure 5-6, Table 5-2). Additionally, a significant difference (X^2 (2, n = 6) = 10.218, p = 0.006) was found between the decreasing NL of the wetland system and the increasing NL of the combined system (Table 5-3).

The NC (i.e., number of components) of the wetland system decreased by 17.31 % from the start to the end of the study period. However, this decrease was non-significant (X^2 (2, n = 3) = 1.705, p = 0.426) (Figure 5-7, Table 5-2). For the combined system of wetlands and SWM ponds, a significant increase of 52.03% in NC was found (X^2 (2, n = 3) = 31.764, p < 0.001) (Figure 5-7, Table 5-2). Additionally, a significant difference existed between the non-significant decrease in NC for wetlands and the significant increase in NC for the combined wetlands and SWM ponds (X^2 (2, n = 6) = 14.040, p = 0.001) (Table 5-3).

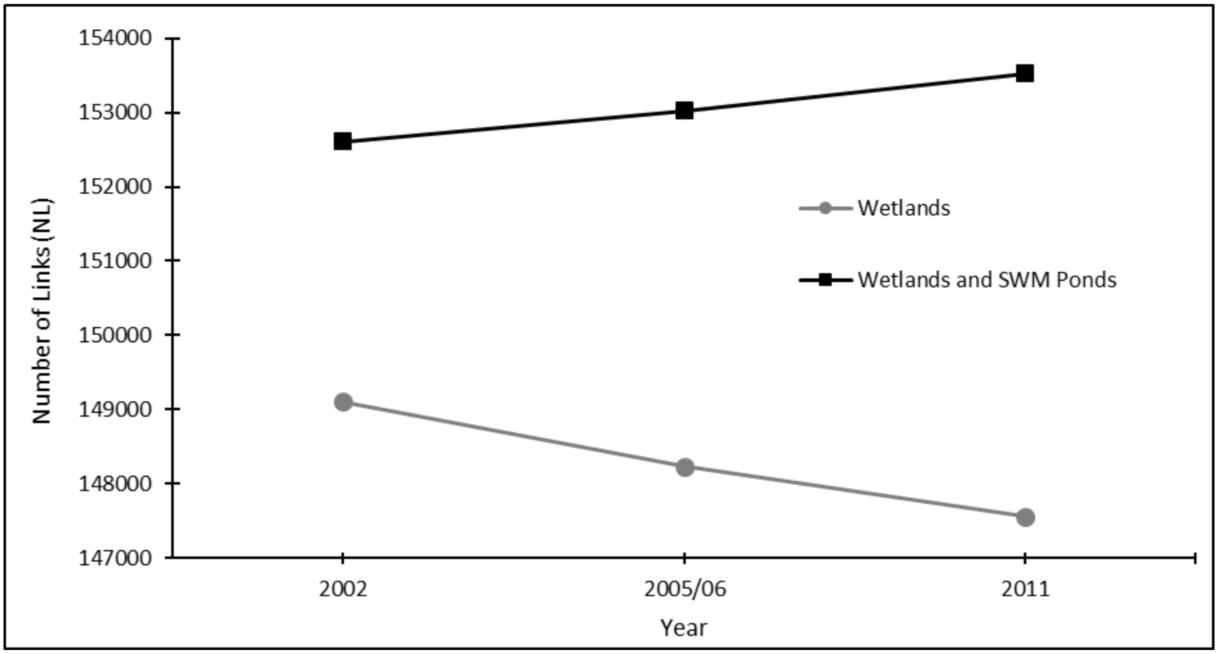

Figure 5-6. Change in the number of links (NL index) versus time for wetlands, and the combination of wetlands and SWM ponds from 2002-2011. Results are for the entire study area, which is comprised of seven municipalities in Southern Ontario, and a higher NL index indicates a more connected landscape.

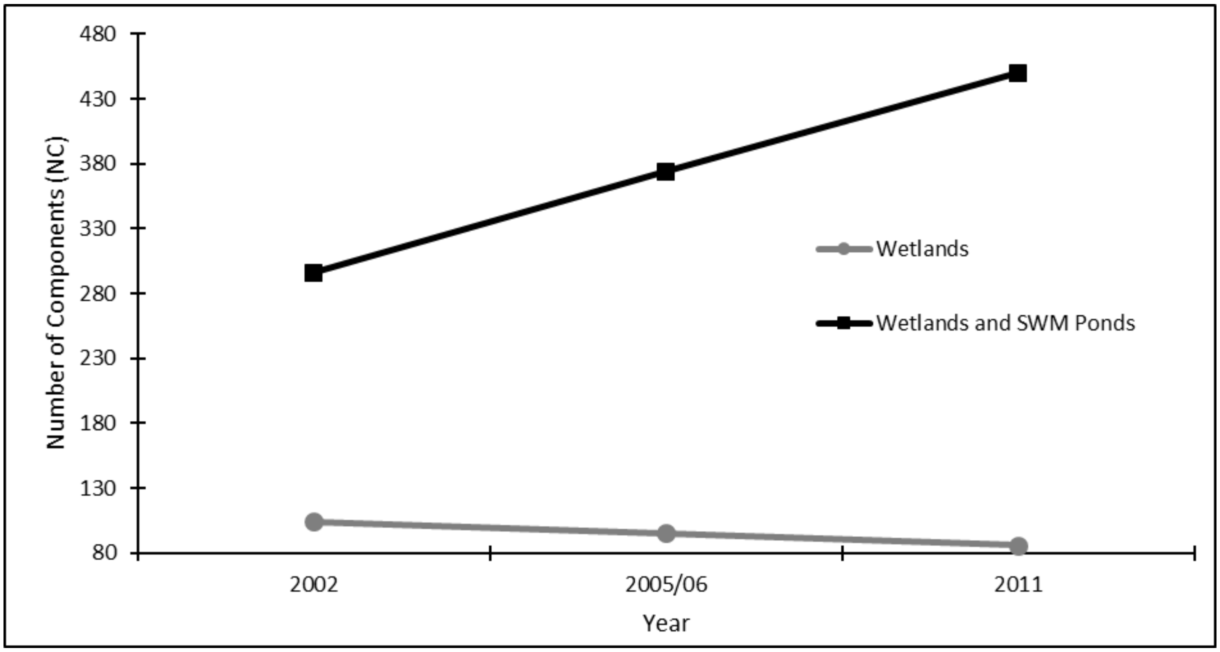

Figure 5-7. Change in the number of components (NC index) versus time for wetlands and the combination of wetlands and SWM ponds from 2002-2011. Results are for the entire study area, which is comprised of seven municipalities in Southern Ontario, and a lower NC index indicates a more connected landscape.

Table 5-2. Chi-square test results for differences between the number of links (NL), and number of components (NC) over three time-steps for wetlands and the combined system of wetlands and SWM ponds. Results are for the entire study area, which is comprised of seven municipalities in Southern Ontario from 2002-2011. A higher NL and lower NC index represent a more connected landscape, ***** indicates statistical significance at $p < 0.05$, ******* indicates statistical significance at $p < 0.001$ and ^{ns} indicates a non-significant result.

	Wetlands			**Wetlands and SWM Ponds**		
	2002	**2005/06**	**2011**	**2002**	**2005/06**	**2011**
NL	149109	148235	147561	152609	153025	153521
X^2		8.124			2.724	
D.F.		2			2	
P		**0.017***			0.256^{ns}	
NC	104	95	86	296	374	450
X^2		1.705			31.764	
D.F.		2			2	
P		0.426^{ns}			**<0.001*****	

Table 5-3. Chi-square test results for differences in number of links (NL), and number of components (NC), between wetlands and the combination of wetlands and SWM ponds. Results are for the entire study area, which is comprised of seven municipalities in Southern Ontario from 2002-2011. A higher NL and lower NC index represent a more connected landscape, ***** indicates statistical significance at $p < 0.05$.

		2002	**2005/06**	**2011**
NL	**Wetlands**	149109	148235	147561
	Wetlands and SWM Ponds	152609	153025	153521
	X^2		10.218	
	D.F.		2	
	P		**0.006***	
NC	**Wetlands**	104	95	86
	Wetlands and SWM Ponds	296	374	450
	X^2		14.040	
	D.F.		2	
	P		**0.001***	

Wetland Connectivity

In the City of Markham, SWM pond size was clustered around a mean of 1.02 hectares (ha) (Table 5-4, Figure 5-8). This was significantly larger than wetlands, which had a mean size of 0.72 ha (W (n = 1672) = 72210, $p < 0.001$). Despite being smaller on average, wetlands had a greater number of larger area outlier wetlands than SWM ponds did.

Results for the PC index and its components should be considered in terms of how a given wetland (i.e. wetland or SWM pond) was connected to all other wetlands across the landscape. No statistically significant differences were found when the mean dPC index and its components were compared over time for wetlands and SWM ponds (Table 5-5). Although this was the case, the dPC index appeared to decline somewhat over time for both wetland types (Figure 5-9). This overall decrease was very minimal for wetlands, while it was more pronounced for SWM ponds. The dPCconnector index remained relatively similar for wetlands, but decreased for SWM ponds (Figure 5-10). The dPCflux index remained at a fairly stable value for wetlands, while this index increased over time for SWM ponds, opposing the trends seen for other indices (Figure 5-11).

Statistically significant differences existed for the mean dPC, dPCconnector and dPCflux between wetlands and SWM ponds (Table 5-6). As no significant differences existed for these indices over time, they were compared at $t = 2$ only, which includes all wetland loss and SWM pond creation observed over the study period. At this time step, the average dPC index for wetlands was 26.64 % higher than for SWM ponds ($W (n = 1672) = 64512$, $p = 0.019$). This was similar to the difference observed in the dPCflux component, although of a greater magnitude, where the dPCflux of wetlands was double that of SWM ponds ($W (n = 1672) = 27578$, $p < 0.001$). An opposite difference existed for the dPCconnector component, where the value for wetlands was very close to zero, and that for SWM ponds was 99.16 % higher ($W (n = 69217) = 27578$, $p < 0.001$).

When the dPC index was plotted spatially, interesting patterns emerged (Figure 5-13). First, the most connected wetlands appeared to be concentrated in the north-west portion of the study area, which also appeared to be outside of the most urban portions of the municipality. Second, most of the “most connected” wetlands were wetlands, while the “least connected” wetlands were both wetlands and SWM ponds. These “most connected” wetlands comprised 11.54 % of wetlands and 8.70 % of SWM ponds, while the “least connected” wetlands comprised 40.30 % of wetlands and 55.07 % of SWM ponds. Third, it appeared that the SWM ponds were concentrated toward the central areas of the municipality, whereas wetlands more commonly occurred along the northern and eastern boundary of the municipality.

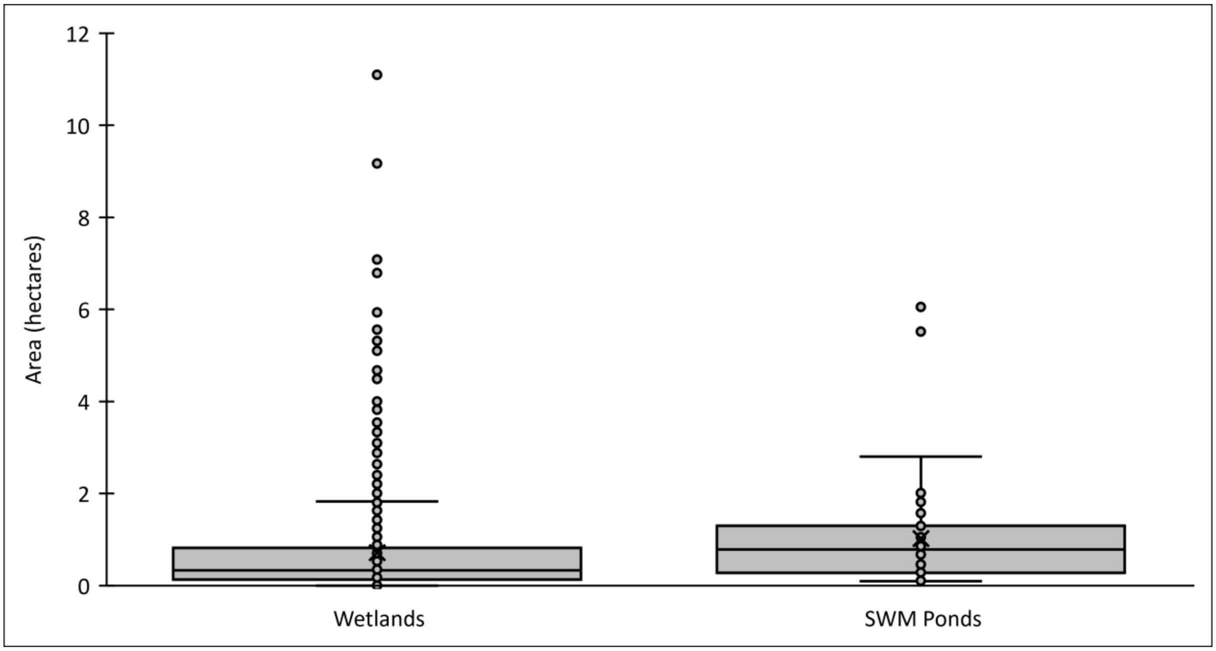

Figure 5-8. Boxplot of wetland area distribution within the municipal boundary of the City of Markham at t = 2 (2011). Area is given in hectares, $N_{wetlands} = 1603$, and $N_{SWM\ ponds} = 69$.

Table 5-4. Wilcoxon test results for the differences in mean area of wetlands and SWM ponds in the City of Markham at t = 2, which corresponds to the year 2011. *** indicates statistical significance at p < 0.001.

		Wetlands	**SWM ponds**
	Observations	1603	69
Area (ha)	Mean	0.7162	1.0174
	SD	1.1720	1.0874
	SE	0.0293	0.1309
	W	72210	
	P	**<0.001*** **	

Table 5-5. Kruskal-Wallis test results for the differences in mean dPC indices (dPC, dPCconnector, dPCflux) between three time-steps from 2002-2011 for wetlands and SWM ponds. Results are for the City of Markham, where higher values for each of the dPC indices represent a more connected wetland, [ns] indicates a non-significant test result.

		Wetlands			SWM Ponds		
		2002	**2005/06**	**2011**	**2002**	**2005/06**	**2011**
	Observations	1618	1610	1603	32	50	69
dPC	**Mean**	0.1224	0.1223	0.1216	0.1009	0.1039	0.0892
	SD	0.2340	0.2325	0.2296	0.4261	0.3638	0.1489
	SE	0.0058	0.0058	0.0057	0.0753	0.0515	0.0179
	X^2		0.3020			5.6927	
	D.F.		2			2	
	P		0.860 [ns]			0.058[ns]	
dPCconnector	**Mean**	0.0003	0.0003	0.0003	0.0738	0.0634	0.0314
	SD	0.0077	0.0078	0.0074	0.4004	0.3418	0.0984
	SE	0.0002	0.0002	0.0002	0.0708	0.0483	0.0118
	X^2		0.3046			5.5822	
	D.F.		2			2	
	P		0.8587 [ns]			0.0614 [ns]	
dPCflux	**Mean**	0.1213	0.1212	0.1205	0.0262	0.0394	0.0567
	SD	0.2314	0.2298	0.2273	0.0358	0.0579	0.0882
	SE	0.0058	0.0057	0.0057	0.0063	0.0082	0.0106
	X^2		0.9036			5.2526	
	D.F.		2			2	
	P		0.637 [ns]			0.072 [ns]	

Table 5-6. Wilcoxon test results for the differences in mean dPC indices (dPC, dPCconnector, dPCflux) between the wetland system and the SWM pond system at t = 2, which corresponds to the year 2011. Results are for the City of Markham, where higher values for each of the dPC indices represent a more connected wetland, *** indicates statistical significance at p<0.001 and [ns] indicates a non-significant result.

		Wetlands	SWM Ponds
	Observations	1603	69
dPC	**Mean**	0.1216	0.0892
	SD	0.2296	0.1489
	SE	0.0057	0.0179
	W	64512	
	P	**0.019***	
dPCconnector	**Mean**	0.0003	0.0314
	SD	0.0074	0.0984
	SE	0.0002	0.0118
	W	69217	
	P	**<0.001*****	
dPCflux	**Mean**	0.1205	0.0567
	SD	0.2273	0.0882
	SE	0.0057	0.0106
	W	27578	
	P	**< 0.001*****	

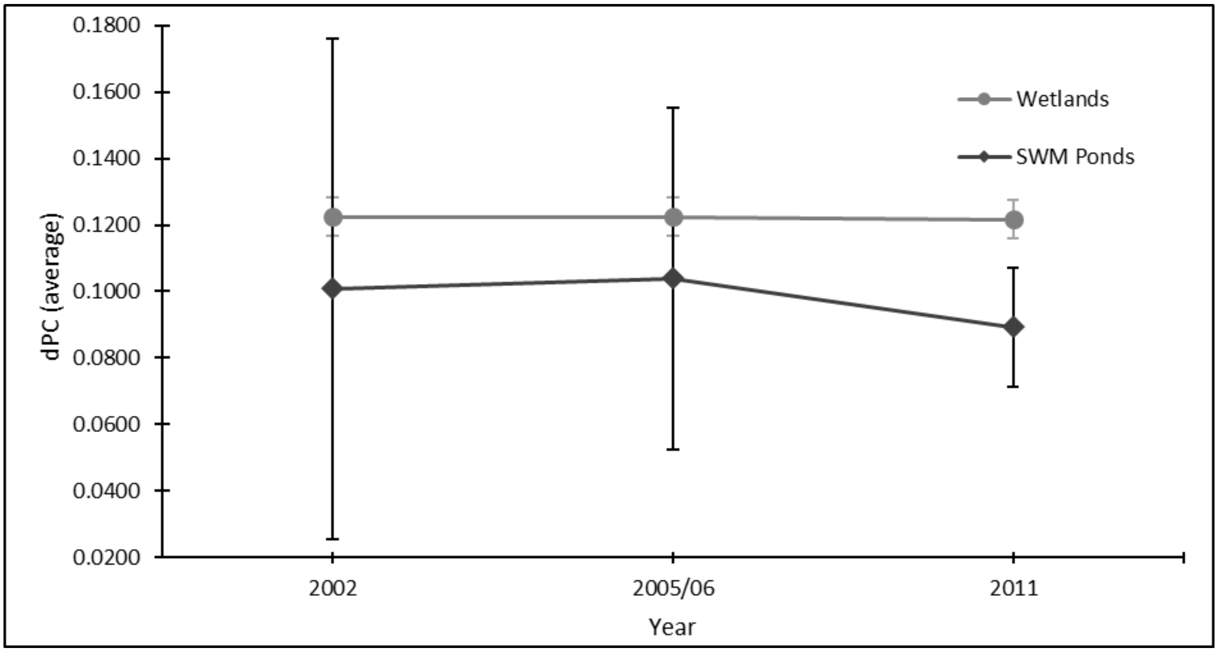

Figure 5-9. Average dPC index over time (2002-2011) and between wetlands and SWM ponds in the City of Markham, where the dPC index shows how connected a wetland is overall, and a higher value represents a more connected wetland. Error bars represent standard error, see Table 5-5 for sample sizes by wetland type and time step.

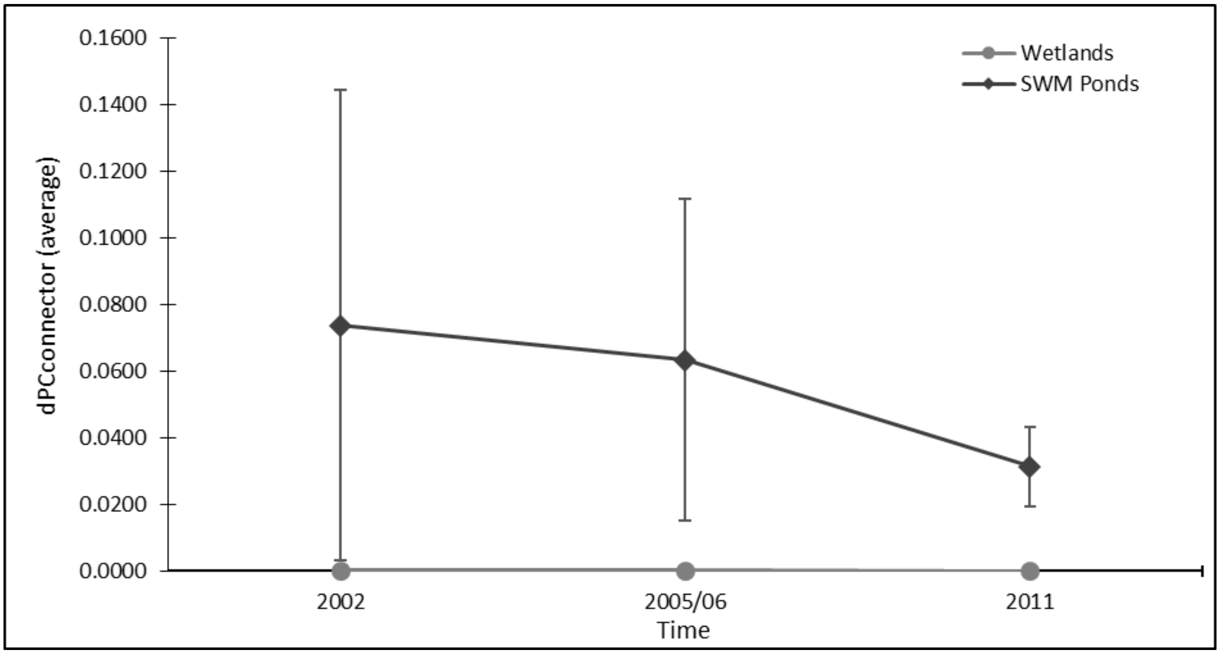

Figure 5-10. Average dPCconnector index over time (2002-2011) and between wetlands and SWM ponds in the City of Markham, where the dPCconnector index shows how well a wetland acts as a connector, or stepping-stone, for other wetlands and a higher value represents a more connected wetland. Error bars represent standard error, see Table 5-5 for sample sizes by wetland type and time step.

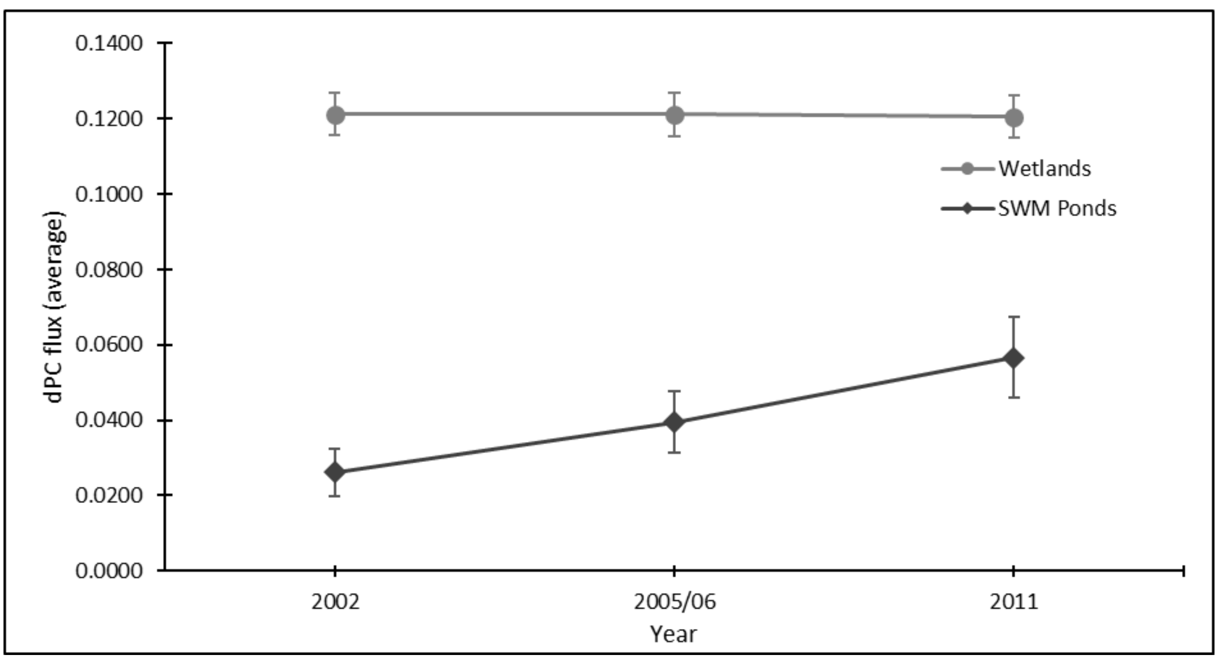

Figure 5-11. Average dPCflux index over time (2002-2011) and between wetlands and SWM ponds in the City of Markham, where the dPCflux index shows how connected a wetland is to others across the landscape and a higher value represents a more connected wetland. Error bars represent standard error, see Table 5-5 for sample sizes by wetland type and time step.

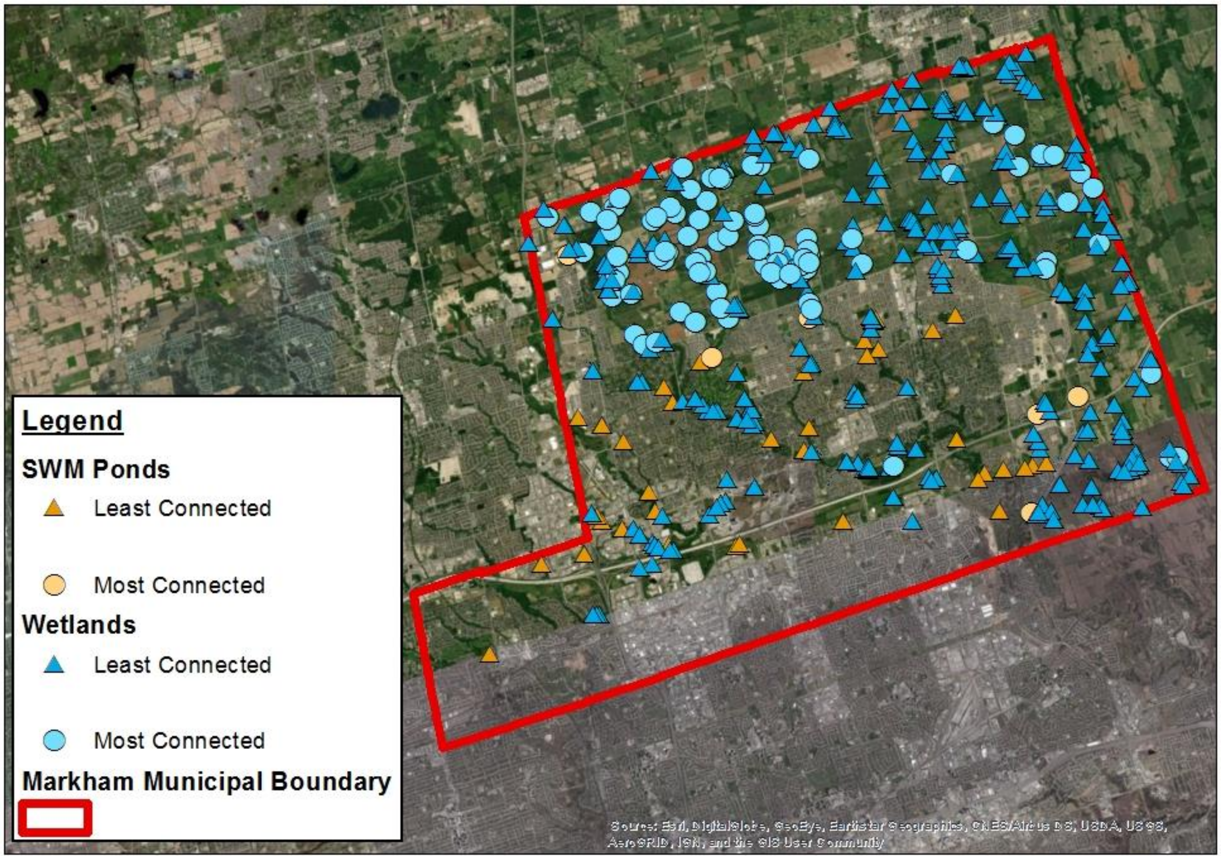

Figure 5-12. Spatial distribution of the PC index values for wetlands and SWM ponds in the City of Markham in 2011. PC values are categorized into three classes based on geometric groupings, where features with dPC values in the lowest class were considered "least connected" and those with dPC values in the highest class were considered "most connected", $N_{wetlands} = 1603$, and $N_{SWM\ ponds} = 69$.

5.5 Discussion

Overall, our results show that recent wetland loss has led to a decrease in wetland connectivity, as shown by the decrease in the number of links between wetlands (NL) over the study period. Across the landscape, SWM ponds were found to be less well connected than wetlands, as shown by the lower dPC and dPCflux values. Lastly, the creation of SWM ponds has increased connectivity for the combined network of wetlands and SWM ponds. This is shown by the higher number of links (NL) for the combined system of wetlands and SWM ponds than for wetlands alone, and by the higher dPCconnector values for SWM ponds, which indicates the ability of SWM ponds to act as stepping-stones between other wetlands.

Landscape Connectivity

Landscape connectivity loss was shown by a significant decrease in NL of the wetland system over the study period. This decrease in connectivity likely follows from the observed wetland losses, which has been linked to negative impacts on plant and animal populations (Fahrig, 2019; Cushman, 2006; Haxton, 2000; Keitt, Urban & Milne, 1997). Specific to wetland loss, Verheijen, Varner & Haukos (2018) studied connectivity in the Rainwater Basin of Nebraska (USA), where wetland numbers have decreased by more than 90% of the pre-European settlement wetland extent. They found that connectivity decreases resulted from this loss, as indicated by an increase of 150 % in the average distance between wetlands and an increase from 3.5 to 10 km in the overall distance needed to travel through the whole wetland network (Verheijen et al., 2018). Because Ducks Unlimited Canada (2010) found that recent rates of wetland loss in southern Ontario have declined, connectivity losses in the present study were likely not of the same magnitude as found by Verheijen et al. (2018). This is especially pertinent since our study only examines loss from 2002-2011, not from pre-European settlement to the present. A great deal of wetland loss already occurred prior to 2002, as found by Ducks Unlimited Canada (2010), meaning that connectivity declines in the past and over longer time scales may have been of a similar magnitude to those found by Verheijen et al. (2018).

When SWM ponds were considered together with wetlands, increased connectivity was shown by a significant increase in NL, relative to the wetland system alone. This improvement of connectivity due to SWM ponds is consistent with the work of Uden et al. (2014), who examined wetland connectivity in an agricultural region of Nebraska, USA. They found that wetland connectivity had decreased from historical conditions, but that present connectivity was maintained by agricultural irrigation reuse pits (Uden et al., 2014).

Despite the general connectivity improvements made by SWM ponds, results of the present study show that the increasing connectivity of the combined wetland system was non-significant over time. This finding is somewhat surprising, given the magnitude of urban expansion and sprawl found by Cheng & Lee (2008) in Ontario's Greenbelt from 1993-2007, which coincides with the present study's time period and overlaps spatially in many areas. Since SWM is implemented in site-planning, a higher rate of SWM pond creation may be expected to coincide with this growth (Schulte-Hostedde et al., 2007). While it is possible that SWM pond construction rates over a longer period may result in greater connectivity gains, this may not functionally make up for the high rates of wetland loss that correspond with urban land uses in southern Ontario (Ducks Unlimited Canada, 2010).

As a lower NC value indicates a connected landscape, the increased NC value for the combined wetland system may indicate a loss in connectivity when SWM ponds were considered, while the decreased value indicates a connectivity gain for wetlands (Saura & Pascual-Hortal, 2007). The results for NC were significant over time for the combined wetland system, and for the two systems in comparison to one another. The lower NC value for wetlands may be explained if wetland loss took place among wetlands that were previously isolated. Isolated wetlands would form their own components, and as such, their loss would lower the NC index (Saura & Pascual-Hortal, 2007). This is plausible as it is known that wetlands in southern Ontario are now less abundant, and the overall landscape is more human-dominated than previously, meaning that remaining wetlands are likely separated by other land uses (Cheng & Lee, 2008; Ducks Unlimited Canada, 2010; Snell, 1987).

Meanwhile, the implied connectivity loss with SWM pond addition, which was indicated by the increase in NC, could possibly be due to the spatial distribution of SWM ponds, as it appears that SWM ponds occurred closer to the urban centre than wetlands did. As such, these SWM ponds could be too separated from their wetland counterparts to form combined wetland complexes with them within the 750 m distance threshold. If this is the case, the results may suggest an increasingly disconnected wetland network due to the generation of independent wetland complexes, while improved connectivity may be seen within these complexes, as indicated by the results for NL.

Our finding of decreased connectivity with the addition of small habitat patches (i.e., SWM ponds) aligns with Fahrig's (2019) discussion of fragmentation as not necessarily being negative for species. In the context of our study, overall connectivity appeared to increase when SWM ponds were included in combined wetland complexes. This was despite more components being generated in a pattern that may be perceived as being fragmented.

Further, although we only considered connectivity from a distance-based perspective, the presence of impermeable barriers like roads leads to reduced connectivity for dispersing species (Baxter-Gilbert, Riley, Neufeld, Litzgus & Lesbarrères, 2015; Mackinnon et al. 2005). As such, wetlands located closer to the urban centres of our study area municipalities were likely less connected from the perspective of these dispersing species.

Wetland Connectivity

Results of this study show that wetlands and SWM ponds may play different roles in connectivity, as indicated by significant differences between each wetland type's dPC, dPCconnector, and dPCflux indices. From an overall perspective, wetlands appear to be more connected than SWM ponds, as they had higher dPC values (Saura & Torné, 2012; Saura & Pascual-Hortal, 2007). As there were 1603 wetland nodes, and only 69 SWM pond nodes, this finding may be attributable to a greater number of connection opportunities for wetlands. Additionally, visual inspection showed that wetlands appeared to be more tightly clustered together on the urban periphery, whereas SWM ponds appeared to be more centrally located and spaced out from one another. As our landscape-level results showed that connectivity increased with the addition of SWM ponds, this distribution could likely have a positive impact on species due to an overall increase of habitat and more uniform connectivity across the landscape.

The different ways in which wetlands and SWM ponds contribute to connectivity were indicated by the examined fractions of the PC index, dPCconnector and dPCflux. Wetlands tended to have a higher dPCflux value than SWM ponds, while SWM ponds tended to have a higher dPCconnector value. As dPCflux represents how well a wetland is connected to other wetlands in the landscape, this indicates that wetlands tend to be more connected on average than SWM ponds (Saura & Rubio, 2010). This finding may be explained by the higher proportion of wetlands, which appeared to be more tightly clustered together than SWM ponds were.

Meanwhile, as dPCconnector represents how important a wetland is as a connector for others, the higher value for SWM ponds indicates that they tend to act as linkages between other wetlands (Saura & Rubio, 2010). This finding may be explained by the tendency of SWM ponds to be located in the centre of the study area municipality, where they might form connections between wetlands at the periphery of the municipality. Further, the improvement of connectivity by SWM ponds is supported by Uden et al.'s (2014) work on connectivity of a wetland landscape in an agriculturally-intensive area of Nebraska, USA. They found agricultural irrigation reuse pits to maintain or improve connectivity in this landscape. In Uden et al.'s (2014) study, the reuse pits were most important as connectors at dispersal distances less than one kilometer. This finding could indicate that the connectivity value of SWM ponds may vary at distance thresholds other than the present one of 750 m, and species that disperse greater distances may rely less on SWM ponds as connectors.

The role of SWM ponds as stepping-stone connectors may also be explained by the work of Keitt, Urban & Milne (1997), who represented habitat mosaics from the southwestern USA as mathematical graphs, in order to quantify connectivity at multiple scales. They found that while large habitat patches (i.e. wetlands) had the greatest total contribution to landscape connectivity, a number of small patches showed similar contributions when considered per unit area, which may have indicated their ability to act as stepping-stones between larger patches (Keitt et al., 1997). Herrera et al. (2017) also emphasize the importance of small habitat patches as stepping-stone connectors in their study of grassland connectivity in Argentina, and they found these features to be most important for species that disperse long distances. In the present study, the average SWM pond was slightly larger than the average wetland, but some larger outlier wetlands existed. This finding, coupled with the high dPCconnector values for SWM ponds and the mentioned literature, may further support the potential role of SWM ponds to connect larger wetlands if they still exist on the landscape.

From a habitat standpoint, the potential stepping-stone function of these SWM ponds is likely positive for the maintenance of biodiversity. Saura, Bodin & Fortin (2014) created a generalised network model of habitat connectivity and found that stepping-stone habitat patches can be an important factor in species persistence across wide spatiotemporal scales. However, stepping-stone habitats must be of sufficient size or quality to be of conservation value, but SWM ponds are unlikely to provide optimal habitat for wetland-dependent species (Moore et al., 2011; Saura et al., 2014; Tixier et al., 2012; Uden et al., 2014). As such, while the results from the current study show that SWM ponds can increase wetland landscape connectivity, they do not indicate that SWM ponds are an adequate replacement for wetlands.

Limitations

Pascual-Hortal & Saura (2006) describe limitations of the NL and NC indices, which perform poorly in terms of correctly identifying the importance of lost patches (i.e. wetlands). A possible reason for this limitation could be the failure of these indices to consider patch area. Additionally, these indices are only indicative of landscape connectivity *among* habitat patches. The probabilistic indices (PC and its components) further examine connectivity of the patches themselves, and these are suited to prioritization of wetlands (habitat nodes) that may maintain the landscape's connectivity (Pascual-Hortal & Saura, 2006; Saura & Rubio, 2010). However, due to computational requirements that lead to the PC index being limited to a lower number of wetland node inputs, this index could only be computed for the City of Markham (Saura & Pascual-Hortal, 2007). The remotely sensed data used in this study come with limitations, and should be interpreted with care. As mentioned, the remotely sensed wetland change inventory has a minimum mapping unit of 0.5 ha, meaning that the loss of wetlands smaller than this could not be reliably detected (OMNRF, 2015). In general, larger minimum mapping units tend to mask sparse land cover classes and underestimate landscape diversity (Saura, 2002). It is possible that decreases in connectivity may have been greater than observed, had this mapping limitation not prevented the potential inclusion of wetlands smaller than 0.5 ha and corresponding loss. Additionally, as Keitt et al. (1997) found that smaller patches (i.e. wetlands) can generally act as stepping-stone connectors, it is possible that the contribution of wetlands in this role (indicated by dPCconnector) may have been greater than observed if smaller wetlands had been included. As such, the lack of significant results over time for some indices should not be interpreted as if there were no declines in connectivity, and do not provide justification for further wetland loss, especially since the majority of southern Ontario's wetlands have already been destroyed (Ducks Unlimited Canada, 2010; Snell, 1987).

Connectivity is one element of wetland function, but it should be kept in mind that there are many other functional variables, and the approach taken for this study is deductive in nature. One consideration is the 750 m dispersal threshold that was used, which is based on the Ontario Wetland Evaluation System's maximum distance within which a wetland complex can be designated (Government of Ontario, 2014a). This is the distance within which the interconnectedness of wetlands and their function is considered at the provincial level, which is why it was chosen as a general indicator of connectivity for these analyses. However, actual dispersal distances of species are known to be variable (Cushman, 2006; Saura & Rubio, 2010). Herrera et al. (2017) showed that connectivity, specifically the importance of small habitat patches, differed based on dispersal distance, which may indicate the benefit of examining dispersal distance in another manner (see "*Next Steps*").

Next Steps

As mentioned, differing results may have been observed if a wider range of distance thresholds had been analyzed, rather than the overarching 750 m wetland complex definition used in the present study (Government of Ontario, 2014; Keitt et al., 1997). Engelhard et al.'s (2017) work on seascape connectivity in Australia indicates another approach to dispersal distances that could be used for further research. In this approach, they used metrics that were based on two important native species that represented a range of other species (Engelhard et al., 2017). This method combines two others: connectivity metrics that are based on average spatial requirements of multiple species, and connectivity based on the requirements of an umbrella species (Engelhard et al., 2017; Minor & Lookingbill, 2010; Olds et al., 2014). In general, it would be helpful to compare a range of dispersal thresholds in a sensitivity analysis to determine how sensitive the connectivity results are to the choice of a specific dispersal threshold (Engelhard et al., 2017).

A mentioned limitation of this study was the calculation of the PC index, which was only able to be completed for one municipality, the City of Markham. While this City is similar to the other municipalities from an area-based perspective, a variety of factors, including municipal policy, impact wetland conversion and can therefore impact connectivity (Schulte-Hostedde et al., 2007). This indicates the need to expand the scale of PC analyses, in addition to the potential to also analyse these indices and others in relation to policy, to determine what may drive patterns in connectivity at different scales. Such an approach would be consistent with the work of Kininmonth, Bergsten & Bodin (2015), who discuss the importance of understanding governance structures in order to uphold wetland connectivity in socio-ecological systems. Further analyses of spatial connectivity patterns may also be informative, as the most connected wetlands appeared to be located in more peripheral parts of the municipality, while the most connected SWM ponds appeared to occur closer to the urban core.

Jurisdictional boundaries are prevalent in wetland planning and work related to connectivity, which is evident in the municipality-based approach taken in the present study, but are not conducive to the larger scale on which ecological processes function (Kininmonth, Bergsten & Bodin, 2015; Thorslund et al., 2017). Kininmonth, Bergsten & Bodin (2015) emphasize the need for cross-boundary connectivity management, and discuss the importance of coordinators for interjurisdictional wetland connectivity management. Such work further supports the need to expand the scale of connectivity analyses, and particularly of probabilistic indices, in order to identify priority sites for connectivity planning beyond municipal boundaries. The prioritization of connectivity at such a scale would require the coordination of stakeholders at multiple levels, but this approach is most conducive to the maintenance of biodiversity (Cui, Zhang & Lei, 2012; Engelhard et al., 2017; Kininmonth et al., 2015). As mentioned, connectivity is only indicative of one aspect of wetland function, and should not be used as a sole means of conservation planning, but is instead complementary to other functional criteria (Engelhard et al., 2017). Attum, Lee, Roe & Kingsbury (2008) studied connectivity in relation to the distribution of several Ontario herpetofauna species, and found that simple measures of connectivity, like distance, may be adequate when examining more common and less vagile species. However, additional considerations, such as corridor quality, are needed to conserve less common and more vagile species (Attum et al., 2008).

Study area boundaries are also an important consideration in connectivity analyses due to their potential to act as barriers to the modelled dispersal, especially for circuit-theory-based analyses (Koen et al., 2010). The potential impact of such barriers is unclear for our graph-theory-based study, but it is likely that the effect would be reduced if the study area was expanded to the watershed scale. Koen et al. (2010) recommend the use of randomized landscape buffers as a solution to the map boundary effect, which is another feasible method that could be taken in future research. To determine if the map boundary effect was impacting results, a sensitivity analysis could be completed using a variety of randomized buffers. Comparing connectivity results obtained with these different buffers would both help to determine if the map boundary effect was impacting results, and determine how best to mitigate these effects (Koen et al., 2010).

In order to take a more holistic approach to conservation, Nel, Reyers, Roux, Impson & Cowling (2011) recommend key considerations for the planning of freshwater biodiversity in their study of a water management area in South Africa. They suggest the inclusion of ecosystems with high ecological integrity, connectivity considerations, priority areas for population persistence, and mappable spatial components (Nel et al., 2011). As such, results of the present study may be indicative of the potential to incorporate connectivity indices in a broader decision-making framework for the conservation of wetlands and allocation of SWM ponds. Under such a framework, SWM ponds could be strategically placed to fulfill their potential to act as stepping-stones, while prioritizing the conservation of wetlands.

5.6 Conclusion

Overall, our results show that connectivity has decreased over the study period with continued wetland loss, while this decrease was somewhat remedied by SWM pond creation. Overall, wetlands were more connected than SWM ponds, which appear capable of acting as stepping-stones between existing wetlands. Declining connectivity is troublesome given that its maintenance is a key aspect of ecosystem resilience, which impacts the conservation of biodiversity that human populations rely on for critical ecosystem services. This is especially concerning given the progression of climate change, as it is only with connectivity that these ecosystem service provisioning species will be able to disperse as their ranges expand to higher latitudes. As mentioned, the consideration of trans-boundary connectivity is critical, as wetland species and processes do not operate only within the jurisdictional constraints within which wetland management tends to occur. Moving forward, wetland connectivity may be maintained or improved through the preservation of remaining wetlands, and strategic placement of SWM ponds, as necessary.

6.0 Synthesis

This chapter serves as a conclusion and synthesis of the two data chapters (i.e. manuscripts) that were presented in this thesis. The findings are first re-discussed in terms of their significance, limitations, and relation to one another. Then, future recommendations are discussed. These recommendations are made in terms of what future research could be beneficial, and where policy improvements could be made to result in better wetland conservation outcomes.

6.1 Principle Findings

Chapter 4 – Trends and Predictors of Recent Wetland Conversion in Southern Ontario Municipalities.

The objective of this chapter was to increase understanding of recent trends of wetland conversion (i.e. wetland loss and SWM pond gain) and to project how these trends may continue into the future. These projections were also considered in terms of the effects of land use and land cover types on wetland conversion, given the prevalence of wetland conversion in certain human-dominated land uses (Ducks Unlimited Canada, 2010; Zedler & Kercher, 2005). With several more specific questions, this chapter addressed the overarching research question (1): how has the composition and abundance of wetlands changed, and has the creation of SWM ponds had any significant effect on their presence? The chapter-specific research questions were related to wetland management and policy, and are as follows:

(1) Has the extent of lost wetlands been fully compensated for by the creation of SWM ponds, from an area-based perspective? As Ontario's wetland policy is somewhat uncoordinated, and decisions about wetland management are generally made at the site-level, it was expected

that net loss of wetlands would be observed (OMNRF, 2017; Schulte-Hostedde et al., 2007; Thorslund et al., 2017).

(2) Has there been a high prevalence of wetland loss among small wetlands? Due to the Ontario Wetland Evaluation System's focus on protecting wetlands larger than 2 ha, it was expected that a disproportionate amount of loss would occur amongst wetlands smaller than 2ha (Government of Ontario, 2014).

(3) Will wetland loss be more probable in areas affected by urbanization and other human-dominated land uses than in land uses that may be less intensive? Agriculture is a historically dominant driver of wetlands loss, and Ducks Unlimited Canada (2012) found built-up lands to be a dominant factor for wetland loss within urban and peri-urban area. As such, it was expected that human-dominated land uses would correspond with greater projected wetland loss (Ducks Unlimited Canada, Earthroots, EcoJustice & Ontario Nature, 2012; Snell, 1987; Zedler, 2000).

Loss of wetlands continued over the study period, while replacement by SWM ponds occurred from a total-area perspective. However, lost wetlands seemed to be of greater average area than the SWM ponds that replaced them. This finding could possibly be the result of a data artefact, due to detection limitations that prevented the observation of lost wetlands smaller than 0.5 ha. However, this finding may also indicate that wetlands were replaced by a greater number of relatively smaller SWM ponds.

Despite the detection limitation mentioned above, the present study was able to detect a greater size range of wetland loss events than Ducks Unlimited Canada (2010) and Snell (1987), who used a minimum mapping unit of 10 ha. Without being limited to the detection of large wetlands (equal to or larger than 10 ha), the present study found that the majority of lost wetlands were smaller than 2 ha. This result reinforces cautioning by Ducks Unlimited Canada (2010) that prior wetland loss estimates were conservative in nature and might actually be higher than estimated. A higher magnitude of total area wetland loss would likely be observed if it were technically possible to detect loss of smaller wetlands.

The concentration of wetland loss among wetlands smaller than 2 ha reinforces critiques of the Ontario Wetland Evaluation System that are discussed by Schulte-Hostedde et al. (2007) in their analysis of Ontario's wetland policy evolution. Despite their limited size, the loss of small wetlands is concerning, both because very few wetlands remain on the landscape, and because of the critical ecosystem services the remaining wetlands provide (Ducks Unlimited Canada, 2010; Houlahan et al., 2006; Snell, 1987). These small wetlands are particularly important because of their unique contributions to biodiversity (Houlahan et al., 2006; Semlitsch & Bodie, 1998). For example, Houlahan et al. (2006) found that small wetlands tend to host the least common species. Meanwhile, small wetlands also have the potential to act as stepping-stones between larger wetlands and improve overall wetland connectivity, a concept that was examined in Chapter 5 (Herrera et al., 2017; Keitt, Urban & Milne, 1997; Saura, Bodin & Fortin, 2014).

If wetland management continues under the current policy regime, which includes the Ontario Wetland Evaluation System, this study projects that wetland losses will continue into the future. Allowing loss to continue is directly contradictory to climate change adaptation and mitigation efforts, as wetlands play a critical role both in flood management and carbon sequestration (Moudrak, Hutter & Feltmate, 2017; Zedler & Kercher, 2005). Wetland loss is projected to be the most magnified in areas with extractive and urban land uses. The projections of greater loss in areas with urban land uses (i.e. “Built Impervious” and “Built Pervious”) are consistent with findings presented by Ducks Unlimited Canada (2010). Meanwhile, SWM pond gain is projected to be most magnified in areas with a class of land uses that includes agriculture (i.e. “Undifferentiated” and “Tilled”), urban land uses, and forested land cover. Further, it is possible that the construction of SWM ponds on potential agricultural lands and in forests still may be related to urban sprawl, given that SWM ponds are now routinely constructed as part of new residential developments (Schulte-Hostedde et al., 2007). Results from a study by Cheng & Lee (2008) suggest that urban sprawl is a pertinent concern in parts of the current study area, where urban sprawl was found to be a major driver of land conversions.

Overall, the results of the current study demonstrate that the present policy regime is likely insufficient to prevent loss of smaller wetlands that are critical to the ecology and hydrology of the area and provide human populations with important ecosystem services (Houlahan et al., 2006; Marton et al., 2015; McLaughlin et al., 2014; Semlitsch & Bodie, 1998). Greater attention needs to be paid to protecting wetlands of all sizes, but especially small wetlands. Although it appears that SWM ponds may currently be compensating for lost wetlands by total area, they are unlikely to compensate from a functionality perspective (Moore, Hunt, Burchell & Hathaway, 2011; Rooney et al., 2014; Tixier et al., 2012).

Chapter 5 – Connectivity Contributions of Wetlands and Stormwater Management Ponds in Urbanized Landscapes.

This data chapter built on the results of the previous data chapter by examining the functional implications of observed wetland conversion (i.e. wetland loss and SWM pond gain), in terms of effects on wetland connectivity across the landscape. The objective of this chapter was to increase understanding of how wetland connectivity has changed given recent wetland conversion, and if wetlands and SWM ponds contribute to connectivity in different manners. With several more specific questions, this study addressed research question (2); how does the current wetland landscape function from an ecological connectivity perspective, and do SWM ponds have any influence on connectivity? The chapter-specific research questions were as follows:

(1) Has loss of wetlands over the recent past led to decreases in wetland connectivity at the landscape level? Habitat loss is a known driver of connectivity loss, and Ducks Unlimited Canada (2010) found that wetlands continue to be lost in Ontario (Cushman, 2006; Haxton, 2000; Keitt et al., 1997). As such, connectivity was expected to decrease over the study period.

(2) Are SWM ponds less connected to other SWM ponds and wetlands, relative to how connected wetlands are to other wetlands and SWM ponds? Little work has examined the connectivity contributions of constructed ponds, though it is known that SWM ponds are of lesser habitat quality than wetlands (Moore et al., 2011; Rooney et al., 2014; Tixier et al., 2012; Uden et al., 2014). Following this trend, it was expected that SWM ponds would be less connected than wetlands.

(3) Has creation of SWM ponds over the recent past led to an increase in wetland connectivity at the landscape level, when both wetlands and SWM ponds are considered? Uden et al. (2014) found that connectivity increased when agricultural reuse pits were included in their analysis, and the same was expected with the inclusion of SWM ponds in this study.

The results of this study indicated that connectivity decreased with loss of wetlands, and that connectivity increased when SWM ponds were included in the analyses. These trends were indicated by a decrease in the NL index (i.e. Number of Links) over time for the wetland system, and increase in NL when SWM ponds were included in analyses (Saura & Pascual-Hortal, 2007). Results for the NC index (i.e. Number of Components) for wetlands indicated an increase in the number of components when SWM ponds were included in the analysis. Interpretation of the NC index in terms of its meaning for connectivity may require a closer examination of the data. For example, a decrease in the NC index (i.e., lower number of components) can indicate increasing connectivity among components, leading to a lower number of larger components; or it can simply indicate loss of components (Saura & Pascual-Hortal, 2007). Visual inspection of the spatial arrangement of SWM ponds helped to interpret these results for the current study. The findings suggest that SWM ponds were more centrally located within the study area than wetlands, which tended to be distributed along the municipal periphery. Given these differences in the locations of wetlands and SWM ponds, it is possible that SWM ponds tended to form their own, new components, while connectivity of wetlands decreased.

The PC (i.e. Probability of Connectivity) index was used to analyse connectivity at the wetland-level, and was more computationally-intensive than the previously discussed landscape-level indices (Saura & Pascual-Hortal, 2007). These computational limitations meant that this index could only be calculated for one municipality, and the results are preliminary indicators of the connectivity roles that wetlands and SWM ponds may play. The PC results suggest that the average wetland was more connected over the landscape than the average SWM pond. This difference in average connectivity can likely be explained in terms of the greater abundance of wetlands relative to SWM ponds, despite the abundance of wetlands being greatly reduced from pre-European settlement times (Ducks Unlimited Canada, 2010; Snell, 1987). However, despite being less connected on average, SWM ponds may act as stepping-stones and improve connectivity between wetlands.

Overall, the results of Chapter 5 indicate that losses in wetlands are leading to decreases in connectivity. However, the results also suggest that SWM ponds may provide benefits by acting as stepping-stones between wetlands. Meanwhile, wetlands appeared to be more connected as a whole, compared to SWM ponds. These findings continue to build on evidence that Chapter 4 gave for the need to halt wetland loss, but also indicate the potential to improve connectivity of remaining wetlands by the creation of SWM ponds.

6.1.1 Connection of Principle Findings

The wetland loss and SWM pond gain observed in Chapter 4 was connected to Chapter 5 through analysis of the functional implications of these changes. Other studies have generally found that small habitat patches (i.e. wetlands) can act as stepping stones to maintain landscape connectivity (Herrera, Sabatino, Jaimes & Saura, 2017; Keitt, Urban & Milne, 1997). The ability of small features to act as stepping-stones is supported by the finding from the present studies that SWM ponds, which were small relative to lost wetlands (Chapter 4), likely fulfilled this stepping-stone role (Chapter 5).

Chapter 4 demonstrated that the overall area of SWM pond gain was greater than that of wetland loss. As such, there may be some potential for these SWM ponds to be strategically placed to improve landscape connectivity in their role as stepping-stones. This potential is further supported by Uden, Hellman, Angeler & Allen (2014)'s work in an agriculturally-intensive area of Nebraska, USA, which found that connectivity improved when constructed agricultural irrigation reuse pits were included in the analysis. However, it is known that SWM ponds are not of equal habitat quality to wetlands, and it remains imperative to halt the disproportionate loss of small wetlands (< 2 ha) that was observed in Chapter 4 (Moore et al., 2011; Rooney et al., 2014; Tixier et al., 2012).

6.2 Recommendations

6.2.1 Limitations and Future Research

Several areas for future research have emerged from this thesis, including a need to expand the spatial extent of these analyses, examine a greater variety of constructed ponds (e.g. ponds on agricultural lands), and policy analysis. In Chapter 4, the effects of the limited spatial extent on the study results may be most evident in terms of the unexpectedly low magnitude of loss projected for areas of the "Undifferentiated & Tilled" class of land use and land cover types. This land use class included agriculture, which is a historically dominant driver of wetland loss, but the projected wetland losses in the current study were much smaller than expected (Snell, 1987; Wiebusch & Lant, 2017; Zedler & Kercher, 2005).

It is possible that agriculture is simply no longer a dominant force of wetland conversion in the study area, but may be acting as a driver elsewhere. This notion is supported by the results of Cheng & Lee (2008), who found that urban sprawl in southern Ontario has led to the conversion of prime agricultural land. This may suggest that urban sprawl may be taking over from agriculture as dominant driver of landscape change in southern Ontario. However, assuming that urban areas continue to expand into agricultural lands, the lost agricultural areas may have to be re-gained elsewhere, such as in northern Ontario (Caldwell & Marr, 2011). It is therefore possible that agriculture will lead to wetland loss further outside of past agricultural boundaries. The possible expansion of agricultural boundaries is a concern for wetlands in areas where the magnitude of wetland losses historically has been lower than found by the studies of Ducks Unlimited Canada (2010) and Snell (1987) for southern Ontario. Consequently, improved monitoring of wetland conversion should be enforced in areas of potential concern. This includes northern Ontario, where wetlands are also threatened by mining, energy development, and transportation infrastructure (OMNRF, 2017).

Ecological processes do not operate within the jurisdictional constraints they are generally managed within, which is relevant in the context of this municipal-boundary-based thesis (Kininmonth, Bergsten & Bodin, 2015; Thorslund et al., 2017). The issue of ecological scale was most evident in Chapter 5, as real-world wetland connectivity is not limited by jurisdictional boundaries. It is also possible that differing connectivity results might have been observed if additional municipalities had been analyzed using the PC indices, especially since there were municipal-level differences in wetland conversion found in Chapter 4. However, the effects of the spatial extent of the study area were also relevant in Chapter 4. Results in this chapter suggest that landscape-level wetland losses are mostly driven by urban land uses, while other factors may have been more prevalent outside of the boundaries of the largely urbanized study area.

To remedy the constraints that municipal boundaries may have placed on this thesis, further research could expand the spatial scale of the current analyses to take an inter-jurisdictional approach. This could possibly follow watershed boundaries or species ranges, which would be more conducive to the way that wetland connectivity operates across the landscape. This approach would also be consistent with a more holistic examination of wetland loss and could help to mitigate potential issues like the map boundary effect on connectivity analyses. Alternatively, map boundary effects could be further mitigated with the use of randomized buffers (Koen et al., 2010) Additionally, the consideration of connectivity across boundaries could improve adaptation management efforts for wetland-dependent species that will need to disperse as their ranges expand to higher latitudes, due to climate change (Humphries, Thomas & Speakman, 2002; Opdam & Wascher, 2004).

Spatial resolution is also a pertinent consideration when it comes to the detection of wetlands and their losses. The minimum mapping unit of wetland area (0.5 ha) for the data used in this study likely masked the loss of wetlands smaller than 0.5 ha, and likely led to the perception of a generally simplified landscape (OMNRF, 2015; Saura, 2002). This mapping limitation is prevalent in wetland management, and ultimately restricts how effectively wetlands can be conserved through evidence-based policies (see Section 4.2.2 – Policy) (Creed et al., 2017). Ultimately, better mapping and inventory of wetland resources is needed for effective protection. This is especially pertinent as Walters & Shrubsole (2005) highlight that Ontario's Wetland Evaluation System assumes that all wetlands are identified and delineated through the evaluation process. This assumption means that wetlands missed by mapping limitations would also miss a chance at being protected by the agencies that review these maps, such as conservation authorities and the Ministry of Natural Resources (Walters & Shrubsole, 2005).

While SWM ponds are a common form of constructed pond within urban and peri-urban areas, there is likely a need for further work to include a greater variety of constructed ponds within the analyses. Doing so would help to capture more accurate rates of pond creation. One example of potential pond types to examine is shown by Uden, Hellman, Angeler & Allen's (2014) work, which examined the connectivity contributions of agricultural irrigation reuse pits. If the extent of this study were expanded outside of urban and peri-urban areas, as previously recommended, the consideration of constructed ponds other than SWM ponds would be necessary.

While the trends observed in the present study were briefly compared and contrasted to policy, a comprehensive study of wetland conversion as it relates to policy is likely needed in order to improve evidence-based decision-making. In Chapter 4, municipal-level differences in wetland conversion were observed, which could be related to factors such as zoning and official plan policies. This is supported by Schulte-Hostedde et al.'s (2007) work, which found agricultural zoning in the City of London to be an inhibitor of the protection of Locally Significant Wetlands. As such, the loss of wetlands and gain of SWM ponds could be examined in relation to the policies that guide conversion. This approach would be pertinent as the present results do not indicate whether the observed trends of wetland loss and SWM pond gain are deliberate result of policy (see Section 4.2.2 – Policy). Lastly, the effects of policy at the municipal- and provincial-level should be examined in their relation to the resulting differences in wetland connectivity, to examine the functional implications of these policies.

6.2.2 Policy

The results of this study have revealed several wetland policy and management concerns. These concerns are mostly related to the failure of current policy regimes to protect small (< 2 ha) wetlands, and the subsequent need for future policy to be more oriented around the conservation of all wetlands, not just large ones. The failure of Ontario's wetland policy system, including the Ontario Wetland Evaluation System, to protect wetlands smaller than 2 ha was highlighted in Chapter 4. The disproportionate losses observed among small wetlands is concerning, given their importance for the maintenance of biodiversity and hydrological regimes (Marton et al., 2015; McLaughlin et al., 2014; Houlahan et al., 2006; Semlitsch & Bodie, 1998).

Current provincial policy will likely continue to allow the loss of small wetlands, as projected in Chapter 4. The "Wetland Conservation Strategy for Ontario 2017-2030" was released by OMNRF (2017) under the previous government, attempting to create a more unified approach to wetland management in the province. However, the provincial government changed in 2018, and it does not appear as if the new government will pursue stricter environmental legislation that could lead to stronger protection of Ontario wetlands. The Wetland Conservation Strategy contains some problematic language that could lead to further wetland loss if it is used to guide future policies. This language is contained in the key goals of the strategy, which aim for a halt in "net loss of wetland area and function where wetland loss has been the greatest" and "net gain in wetland area and function where wetland loss has been the greatest" (OMNRF, 2017). This type of approach, aiming for no net loss but for a net gain of biodiversity, is known as biodiversity offsetting (Bull, Suttle, Gordon, Singh & Milner-Gulland, 2013; BBOP, 2012).

Offsetting can occur in terms of the species composition, habitat structure, ecosystem function, and anthropogenic values connected to biodiversity (Bull et al., 2013; BBOP, 2012). However, by allowing offsetting to occur, this strategy does not adequately address the loss of wetlands. Instead, it reinforces the loss of wetlands in exchange for the creation of SWM ponds, which are not a functionally adequate replacement from an ecological or hydrological perspective (Rooney et al., 2014; Tixier et al., 2012; Moore et al., 2011). Offsetting strategies are controversial; although they may be economically reasonable, they allow for loss of ecosystems. The functions and services provided by these ecosystems are generally not fully understood and are exchanged for features that have uncertain benefits (Bull et al., 2013). However, what is certain is that the benefits provided by SWM ponds are lower compared to natural systems (Bull et al., 2013).

As such, the best approach for the conservation of remaining wetlands in southern Ontario is likely a complete protection-based strategy (Creed et al., 2017). This type of strategy would protect all wetlands, and is recommended by Creed et al. (2017) as a simple and effective means of preventing loss. The protection-based strategy is especially recommended where wetlands are not yet adequately mapped and evaluated. This appears to be the case in southern

Ontario, as mapping and evaluation is limited for small wetlands (Creed et al., 2017; Government of Ontario, 2014). If advancements were made in wetland mapping, such as a smaller minimum mapping unit and quantification of wetland function, additional strategies may be considered. This includes an effect-based strategy that protects wetlands that may have effects on downstream waters, or a function-based strategy that is based on quantified wetland function (Creed et al., 2017). However, since wetlands in the study area are already so sparse and degraded, evidence points towards the need for the protection-based strategy, to halt further wetland loss (Ducks Unlimited Canada, 2010; Snell, 1987).

Municipal-level differences in wetland conversion point towards the need for policy changes in municipalities with the greatest wetland losses, such as the City of London. These protections may be based in zoning and official planning, which would require municipal-level prioritization of wetland conservation. However, Schulte-Hostedde et al.'s (2007) finding, that protection of Locally Significant Wetlands in the City of London was largely ineffective in regions zoned as agriculture, also likely relates to provincial policy. This lack of protection may have been because Section 2.1.7 of the Provincial Policy Statement (2005), stated that “nothing in Policy 2.1 is intended to limit the ability of existing agricultural uses to continue” (Government of Ontario, 2005b). The Provincial Policy Statement (2014) retained this language in Section 2.1.9 (Government of Ontario, 2019c). This clause prioritizes common agricultural practices that include wetland drainage over the conservation of wetlands, and such policy should be reviewed, given the pertinent need to conserve what remains of southern Ontario’s wetlands. However, the OMMAH (2019) has proposed changes to the Provincial Policy Statement and this clause has been retained, which indicates that no increased protection of wetlands on agricultural lands is likely under the current, new government.

In Ontario, the *Drainage Act* deals with municipal outlet drainage, which includes agricultural drainage (Government of Ontario, 2018a; Walters & Shrubsole, 2005). Provincially significant wetlands are protected from urban development under the previously mentioned *Planning Act*'s Provincial Policy Statement (2014), but drainage is not considered development (Walters & Shrubsole, 2005). This means that even provincially significant wetlands are not provided official protection under the Drainage Act (Government of Ontario, 2018a; 2019c). Instead, to protect wetlands against drainage, this Act employs a referral system, which includes agencies like Conservation Authorities in the decision-making process regarding impacts of drainage works on wetlands (Walters & Shrubsole, 2005).

Walters & Shrubsole (2005) examined the assumption that this referral system is sufficient to prevent the loss of wetland area, and found that it generally was not. Recommendations given through the referral process tended not to be followed by agricultural operators and wetland loss continued in their study of Zorra Township. Since this Township was found to be perceived as progressive in wetland management, Walters & Shrubsole (2005) suggested that these issues are likely to be echoed province wide. Given that the referral process is the sole means of protection from drainage for wetlands evaluated under the Ontario Wetland Evaluation System, this is a concerning circumstance for wetlands of all sizes. This examination of the *Drainage Act* further indicates a need for more regulation in Ontario's wetland-related policy (Government of Ontario, 2018a; Walters & Shrubsole, 2005).

The need for no further loss of wetlands is especially pertinent, since losses of small wetlands can have cumulative effects. However, cumulative effects of wetland losses across large spatial extents can be difficult to quantify or predict accurately (Creed et al., 2017; Thorslund et al., 2017). Overall, it is recommended to:

(1) conduct a review of the Ontario Wetland Evaluation System and consider inclusion of wetlands smaller than 2 ha, and;

(2) move towards no-loss policies for wetlands that would follow a complete protection-based approach to wetland management (Creed et al., 2017).

The implementation of no loss policies is especially relevant given the general scarcity of wetlands in southern Ontario, as indicated by studies from Ducks Unlimited Canada (2010) and Snell (1987). While provincial action is needed, the observed municipal-level differences in wetland losses also indicate the potential for municipal action on wetland loss. This could be achieved by enacting policies that include the evaluation and protection of locally-significant wetlands, especially those not on agricultural lands that may not be addressed within the scope of the Provincial Policy Statement or the *Drainage Act* (Government of Ontario, 2019d; Schulte-Hostedde et al., 2007; Walters & Shrubsole, 2005). Positive municipal action on the management of non-provincially significant wetlands may be strengthened by responsible use of OMMAH's (2019) proposed changes to the provincial policy statement. This may be possible through a new clause (2.1.10), which states that "Municipalities may choose to manage wetlands not subject to policy 2.1.4 and 2.1.5, in accordance with guidelines developed by the Province."

Further, issues such as the *Drainage Act*'s recommendations-based referral system indicate the need for more regulation and oversight in wetland policy. The recommendations-based referral system is just one example but highlights the fragmented policy approach that currently addresses wetland conservation in Ontario. As a whole, strengthened wetland conservation measures are critical for the maintenance of biodiversity and hydrological regimes (Creed et al., 2017; Houlahan et al., 2006; Marton et al., 2015; Semlitsch & Bodie, 1998). These measures will also help communities adapt to the effects of climate change that include flooding risks, which should be a municipal priority (Moudrak et al., 2017).

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