Local and landscape drivers of aquatic biodiversity in urban stormwater management facilities

by

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Author’s Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

Stormwater management facilities (SWMFs) are often constructed in urban areas to provide flood mitigation and runoff control, and thus replace some wetland functions. Yet, their suitability as habitat for wetland biota is contested. My research objective was to identify the important local- and landscape-level factors most likely influencing habitat suitability of SWMFs in Brampton, Ontario for two widely-used wetland bioindicators: fish and anurans. I discovered that fish were prevalent (60% of sites), though richness was low (average 1 species per site, 6 species total). Low richness likely reflected poor habitat and/or water quality (shrubs and chloride ions), as well as habitat isolation (low area of open water, and high area of roads and impervious cover). Fish communities were composed of native warm-water tolerant, benthivorous species (e.g., fathead minnows), but 20% of sites also included relatively abundant invasive omnivorous goldfish. Anurans were also prevalent (95% of sites) in SWMFs, with low richness (average 1.5 per site, 6 species total). Both anuran composition and species richness were related to the extent of robust emergent vegetation (e.g., cattail) and to the concentration of chloride in water. When I compared the anuran community composition and species richness in SWMFs to natural wetlands in the surrounding area, I discovered surprising similarity. SWMFs support lower diversity, however; spring peepers and wood frogs are present in natural wetlands but not in SWMFs. Despite lower richness of anurans, SWMFs do support American bullfrogs that were not detected in natural wetlands. I conclude that SWMFs do provide important ecological habitat to aquatic biota in urban areas and that promoting robust emergent vegetation and improving water quality in SWMFs could increase their habitat value. I contend that SWMFs should be considered novel ecosystems and incorporated into regional planning for the conservation of urban biodiversity.

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List of Abbreviations

AIC – Akaike information criterion

Cl^- - Chloride ion concentration

Chla – Chlorophyll-a concentration

NMDS – Non-metric multidimensional scaling

NO_2^- - Nitrite concentration

NO_3^- - Nitrate concentration

NH_3 – Unionized ammonia concentration

PCA – Principal component analysis

PerMANOVA – Permutational analysis of variance

PO_4^{3-} - Orthophosphate concentration

RobustVeg – Extend of robust emergent vegetation (e.g. cattail)

SD – Standard deviation

SWMF – Stormwater management facility

TRCA – Toronto and Region Conservation Authority

1.0 Introduction and literature review

The adverse effects of urbanization on the environment include habitat loss (Fahrig, 2003; Pimm and Raven, 2000) and alterations to local climate (Howard, 1833; Li et al., 2015; Qiu et al., 2020; Ward et al., 2016). These well-documented effects are often attributed to the dominance of impervious surfaces in urban landscapes (Arnold and Gibbons, 1996), which are also linked with increased frequency and severity of flooding events that can cause severe damage to urban infrastructure (Karamouz et al., 2010; Zhou et al., 2019). Stormwater management facilities (SWMF) are constructed ponds built in urban and peri-urban areas that provide critical flood mitigation services, and are increasingly seen as important in climate change adaptation (e.g., Alexander et al., 2019). Stormwater management facilities may also provide the only aquatic habitat in highly developed landscapes, where rates of stream and wetland habitat loss are usually very high (Kozlowski and Bondallaz, 2013). Indeed, in a study of monitoring data from seven rapidly-developing municipalities in Southern Ontario, Birch et al. (2022) found that wetland loss was matched by SWMF creation, resulting in indirect replacement of wetland habitat with SWMF ponds.

Because they are human-created and not natural ecosystems, stormwater ponds can be classified as novel ecosystems, which are ecosystems permanently altered by interactions with humans (Hobbs et al., 2006). As such, they may differ fundamentally from natural habitat analogues like wetlands and ponds (e.g., in supporting novel assemblages of biota or unique ecosystem functioning), but as Hobbs et al., (2006) argue, their potential value should not be entirely discounted simply because of the role of human agency in their creation and development. In this thesis, I aim to quantify the importance of several local (e.g., water quality) and landscape variables on community composition of two key taxonomic groups that use stormwater management ponds for habitat: chorusing amphibians and fish.

1.1 Urban ecology

In 2018, it was estimated that over 55% of the world population lived in urban areas, and this figure is projected to rise to 68% by 2050 (UN, 2018). The rapid increase in urbanization has caused urban ecology to become a critical field in modern science.

A major consequence of urbanization is the destruction of natural green (e.g., parks, forests, meadows) or blue (e.g., waterbodies, wetlands, rivers) areas, which are replaced by impervious cover such as roads, buildings, and sidewalks. Consequently, the function of the urbanizing ecosystem can be altered, often in negative ways. Habitat destruction is the leading cause of species extinction nationwide (Pimm and Raven, 2000), and losses due to urbanization are ongoing. Impervious surfaces are artificial structures where water is unable to penetrate through to the soil and they have long been used as an index of urbanization (Arnold and Gibbons, 1996), and impervious cover presents an inhospitable area for most biota (Arnold and Gibbons, 1996; Pimm and Raven, 2000). Urbanization results in the fragmentation of green and blue 'natural' habitats, leaving patches of remnant natural landcover that are functionally isolated from one another, and which make it difficult for organisms to disperse through urban areas (Fahrig, 2003). Roads and housing complexes are common causes of habitat fragmentation: they create a barrier that many terrestrial organisms are unable to cross (Hamer and Parris, 2011). Connecting natural areas with corridors can mitigate fragmentation and its effects and allow easier dispersal for organisms, though this is not always implemented in urban planning (Fahrig, 2003). As land continues to be developed, fewer corridors of connectivity are present, resulting in fragmented habitats where organisms will face challenges surviving.

Impervious cover can also be directly linked to habitat degradation, as water cannot infiltrate the soil, instead becoming surface runoff (Arnold and Gibbons, 1996; Guo, 2008). This runoff can carry toxins such as road salts, oils, and pesticides from developed areas into natural systems, resulting in harsh living conditions for aquatic flora and fauna (Lewellyn et al., 2016; Walsh et al., 2012). Degradation of habitat can also occur through temperature anomalies, where impervious surfaces cause cities to be much warmer than their surrounding natural counterparts, which can have drastic effects on human populations in summer months (Ziter et al., 2019). Studying urban environments is relatively new in the field of ecology, where the predominant perception of these systems was that they are unworthy of study, and only land without human impact had ecological value (Grimm et al., 2008; Wu, 2014). Only in the past few decades has urban ecology begun to advance with an emphasis on ecosystem services and their potential to positively influence human well-being (Grimm et al., 2008; Soga et al., 2014; Wu, 2014). The shift towards urban conservation can help facilitate a sustainable future for cities that benefit both humans and the environment (Wu, 2014).

While there is increased focus on efforts to conserve urban biodiversity, methods used to manage urban ecosystems are often debated (Knapp et al., 2021). Land sharing and land sparing are two opposing philosophies of conservation that can be applied to urban ecosystems (Soga et al., 2014). Land sharing involves a distribution of natural lands throughout urban areas, often in proximity to residential areas. Alternatively, land sparing involves isolating a large portion of natural area that is safeguarded against urban development (Lin and Fuller, 2013; Soga et al., 2014). Both strategies offer advantages. For example, land sparing has been shown to offer higher species diversity than land sharing and may be the best way to conserve sensitive native species (Edwards et al., 2015; Soga et al., 2014). Land sharing, however, can accommodate a larger extent of conservation lands as they can be distributed throughout urban areas. The proximity to cities means that ecosystem services provided by land sharing can benefit more people while still providing some habitat to local biota (Kremen, 2015). Encouraging low impact development practices that maximize green and blue cover in urban areas can yield valuable ecosystem services and habitat for flora and fauna.

1.2 Stormwater management

Stormwater management is a growing issue that arose with urbanization (Holman-Dodds, 2007). In areas of low development and population density, precipitation that runs off fully saturated soil and impervious surfaces becomes overland flow as a natural part of the hydrological cycle. However, when the extent and prevalence of impervious surfaces increases due to urbanization, flooding and stream erosion increase, leading to changes in lotic environments collectively termed the urban stream syndrome (Holman-Dodds, 2007). The increase in human activity is also directly linked with an increase in pollution, which is carried by stormwater runoff into natural aquatic systems during flood events (Dhalla, 2012). Strategies to control both the quality and quantity of stormwater in urban areas most commonly involve stormwater management facilities.

Stormwater management facilities are primarily designed for flood prevention and control of runoff. During rain events, water runs off impervious surfaces and saturated soils and into strategically placed catch basins on streets that connect to SWMFs via underground pipelines (Dhalla, 2012). The water filters through pipes into the forebay, where most of the contaminants settle to the sediment before the water reaches the retention pond. The forebay is generally not directly linked to the retention pond via surface water, though during peak storm events they are designed to overflow and become one pond. As water levels in the retention ponds rise, water is released into connected steams either through surface or bottom draw discharge (Dhalla, 2012; Van Seters and Dougherty, 2019). This SWMF design is often referred to as a wet pond, as there is water in the retention pond year-round (Dhalla, 2012).

Though SWMF creation in urbanizing areas means that these ponds are abundant, the quality of SWMF habitat is the subject of debate (reviewed in Clevenot et al., 2018; Oertli and Parris, 2019). Much of the available research has focused on dragonflies. For example, Canadian SWMFs are known to promote dragonfly diversity in urban areas (e.g., Holtmann et al., 2018), and dragonfly diversity in SWMFs can be similar to that observed in natural rural wetlands and ponds (Perron and Pick, 2020). The habitat value for odonates (dragonflies) depends on the plant community (Perron and Pick, 2020) and the plant community appears to depend on habitat size and heterogeneity (e.g., Holtmann et al., 2019).

Studies of anurans in SWMFs have yielded less consistent results. In France, (Conan et al. 2022), Australia (Sievers et al. 2018, 2019), Canada, (Bishop et al. 2000) and the USA, (Gallagher et al., 2014) researchers suggest that SWMFs are ecological traps for anurans due to toxicants in the water. Contaminants can be correlated with land use patterns (e.g., higher in SWMFs with urban or rural land use context) (Clevenot et al., 2018), however some American studies suggest that pollution-tolerant species make good use of SWMF habitat, even in highly disturbed landscapes (Bateman, 2014; McCarthy and Lathrop, 2011). Research in Australia conducted by Hamer et al. (2012) and Hamer and Parris (2011) details that removal of predatory fish, as well as increases in aquatic vegetation and connectivity, can enhance the value of SWMFs as breeding sites for anurans. These results also align with McCarthy and Lathrop (2011), who found that connectivity, via increased natural cover surrounding sites, can make SWMFs within the USA a more suitable habitat. SWMFs are commonly populated with invasive species in high abundances. Compared to pristine natural wetlands, where invasive species are rare and native biodiversity is high, SWMFs then have comparatively low quality habitat (Beninde et al., 2015; Dearborn and Kark, 2010). Synthesizing somewhat inconsistent results from these studies, it is clear that water quality and landscape context are important drivers of SWMF habitat value for anurans, but it seems unlikely they can fully replace the habitat value lost when natural wetlands are destroyed.

Stormwater management facilities are clearly inhabited by a variety of wildlife and thus can be considered ecological habitat. However, in Ontario, SWMFs are not legally considered natural habitat by many Conservation Authorities (i.e., are not designed, managed or monitored in ways that increase biodiversity) due to the sink habitat characteristics and destructive maintenance (e.g., dredging) which could impair biodiversity. Since they are not legally considered habitat, they are not managed or monitored for their habitat value which in turn creates a feedback cycle that is likely to further impair their biodiversity value. Additional research is needed to identify SWMF attributes or landscape context features that could be leveraged to enhance their habitat value and expand the portfolio of ecosystem services that they provide to support a diversity of native species and to enhance landscape connectivity for aquatic and semi-aquatic organisms.

1.3 Naturalizing SWMFs

Over the last three decades, there has been a shift towards more environmentally sensitive development, including a move from traditional stormwater ponds with deep water, turfgrass catchments, hardened or steep-sided shores and short hydrologic residence times toward naturalized stormwater management ponds with native grasses planted in the catchments and gentler shoreline slopes or shelves to promote emergent vegetation in the water and longer residence times for water (Ross et al., 2018). This change was principally motivated to improve water quality (Vincent et al., 2014), and discourage nuisance species like Canadian geese (Smith, 2006). It has resulted in increased property and amenity values in municipalities (Heller, 2020), and over the life cycle of the development, reduced costs compared to traditional stormwater ponds (e.g., https://cnla.ca/uploads/pdf/LCCA-Stormwater-Report.pdf). Another key benefit that materialized due to naturalization is the increased habitat value of stormwater management facilities (e.g., Wilson et al., 2013). Naturalized SWMFs are designed to mimic natural wetlands, with expanded vegetation zones consisting of wet meadow, shallow marsh and deep marsh plants, and reduced extent of open water compared to traditional SWMFs. These naturalized SWMFs are designed to have shallower slopes and higher nutrient permeable soils to facilitate plant growth (Ross et al., 2018). Although naturalized SWMFs do not function exactly like naturally occurring wetlands (e.g., Rooney et al., 2015b), increasing the extent of wetland vegetation can shift SWMFs from pond-like to more wetland-like ecological function.

The naturalization of SWMFs may provide a higher value of biodiversity and ecosystem services in urban areas. Increases in wetland vegetation can promote biodiversity, as they act as a key driver of trophic interactions through habitat refuge, and as a food source for grazers (Bobbink et al., 2006). The addition of wetland plants also supports the removal of contaminants in SWMFs, which is critical considering the typical influx of pollutants to these waterbodies. Road salts, heavy metals, oils, and inorganic nutrients are all anthropogenic contaminants that can not only degrade the biodiversity within SWMFs but also within streams via discharge (Ross et al., 2018; Sałata et al., 2019). Emergent vegetation and a natural canopy surrounding SWMFs can also provide cooling effects to the waterbody through shading and evapotranspiration, which can reduce surface water temperature (Cronk and Fennessy, 2009). This service can help offset thermal pollution on streams from discharge which can have drastic effects on stream ecosystems (Abdi and Endreny, 2019; Wilby et al., 2015). Stormwater management facilities also offer recreational and other social values to humans, due their accessibility as urban blue spaces. They have been shown to be held in high regard by the public in terms of their aesthetics (Rooney et al., 2015b). As SWMFs may be the only wetland-equivalent ecosystems present in urban areas, the aesthetics of naturalized SWMFs may promote a healthier relationship between humans and the environment (Macdonald, 2016).

1.4.2 Anurans in urban SWMFs

Anurans and other amphibians are widely used as bioindicators in urban wetland systems, due to their susceptibility to environmental change. This sensitivity has caused widespread declines of anurans, and up to one third of known species are considered threatened globally (Holtmann et al., 2017; Hutto and Barrett, 2021). Anurans are highly sensitive to landscape changes from urbanization given their small home range and limited dispersal capabilities (Hutto and Barrett, 2021). On a local level, anurans are responsive to water quality changes due to the permeability of their skin which cannot filter out toxins (Sievers et al., 2019). Changes in vegetation may also affect availability of forage for juvenile stages as well as availability of refuges from predation (Scheffers and Paszkowski, 2013). Despite poor water quality and low vegetation cover in SWMFs, anurans may inhabit these systems, and are an ideal taxon to study the localized and landscape urbanization effects on these ponds. It is therefore no wonder that much research has been published about anurans in SWMFs.

Results of studies on anuran use of SWMFs have yielded conflicting results and the habitat value for SWMFs for anurans remains contested. A large portion of urban anuran studies suggest that SWMFs are poor habitats and may be acting as ecological traps (i.e., biota mistakenly prefer low quality habitats over other available habitat, reducing the fitness of the population) (Sievers, Parris, et al., 2018). These studies focus primarily on contaminant loading from runoff in the form of heavy metals, road salts, and pesticides (Bishop et al., 2000; Ficken and Byrne, 2013; Gallagher et al., 2014; Ramesh et al., 2017). Other researchers have found that SWMFs can support a variety of different anuran species, and that occupancy is affected by vegetation (Scheffers and Paszkowski, 2013; Sievers et al., 2019; Sievers, Hale, et al., 2018). Surrounding landcover, specifically road density, is also a key determinant in habitat occupancy (Hutto and Barrett, 2021).

Relatively few studies have looked at both landscape and local variables in combination in terms of their effects on anuran communities in SWMFs. Those that do (e.g., Hamer et al., 2008; Hamer and Parris, 2011; Holtmann et al., 2017; McCarthy and Lathrop, 2011) stress that local and landscape variables can have synergistic effects on anurans. Interestingly, the study by Holtmann et al. (2017) found that anuran community composition in SWMFs was similar to that in natural wetlands, though it was noted that some of the "wetlands" were actually artificial ponds. The majority of studies on anurans in urban environments were conducted outside of Ontario, where the species pool is considerably different. To my knowledge only one study has looked at anurans in SWMFs in Ontario (Bishop et al., 2000), and it focused on water quality. Bishop et al. (2000) found higher anuran species richness and abundance in a natural wetland compared to SWMFs, but considered only one natural wetland site and so would not have captured the range of variation typical among natural wetlands. Inconsistent results regarding the effects of landcover and local effects on SWMF anuran communities, as well as the lack of research conducted in Ontario, calls for further research on this topic, including a comparison between the anuran communities in natural wetlands in urbanized Ontario and those using SWMFs.

1.4.3 Urban fish ecology

Fish are important organisms in aquatic ecosystems, as they have major impacts on food webs through predation and nutrient cycling (McIntyre et al., 2008). As with anurans, fish can also be used as bioindicators of contaminants. The low dispersal ability of fish may leave them sensitive to water quality and landscape changes within SWMFs, though this has not been investigated in Ontario. There is limited research on fish in urban areas, let alone in stormwater management facilities (Hassall, 2014). Most research involving SWMFs focuses on the adverse effects of water discharge on fish in receiving streams (Bliss et al., 2015; Brix et al., 2010; Stearman and Lynch, 2013), or mesocosm water toxicology experiments (Young et al., 2018). There is also a growing literature supporting the presence of goldfish, *Carassius auratus*, a species that has invaded many aquatic systems in Canada via pet release (Campbell, 2021; TRCA, 2016).

The presence of fish in SWMFs may influence the occupancy of anurans. Some predatory fish species feed both on juvenile and adult stages of some anuran species in urban wetland systems (Ficetola and Bernardi, 2004; Hamer et al., 2012). It is therefore critical to sample these organisms together to distinguish the effects of local or landscape features of the pond and its context from interactions between anurans and fish. The presence of fish in SWMFs, for example, could confound effects of water quality or habitat isolation, if not controlled for. More, anurans and fish are both highly charismatic organisms that the public generally cares about and thus both taxa could serve a flagship species for urban conservation. Strategies to research and protect charismatic anuran and fish species may help SWMF design improvements obtain public support, providing another rationale for their selection as study organisms.

1.10 Thesis Objectives

Although the value of SWMFs in flood mitigation and runoff control is clearly recognized and the explicit purpose for building them, they may provide many other, unquantified, services, including providing habitat for native species, enhanced landscape connectivity for aquatic and semi-aquatic organisms and water purification services. Stormwater management facilities can be inhabited by fish and anurans, but the quality of the habitat is contested. My thesis will address gaps in our knowledge about the services provided by SWMFs through two data chapters.

In chapter 2 of my thesis, I explore which (if any) species of fish use 20 SWMFs. I further explored the life history of the species detected and evaluated local and landscape predictors of species richness to formulate hypotheses for future research. I predict that the fish occupying SWMFs will be disturbance-tolerant species capable of persisting under low oxygen levels due to the harsh water quality conditions in some SWMFs. I expected that species richness of fish would be predicted by local characteristics of the ponds (e.g., vegetation cover and water quality) more so than landscape characteristics (e.g., the extent of impervious surface, canopy cover, and water cover in a buffer around each pond). On the local scale, I found species richness of fish was associated with water quality and vegetation. The addition of landscape variables did not substantially reduce the quality of the model but also did not improve it.

In chapter 3 of my thesis, I evaluate richness and community composition of chorusing anurans in 21 SWMFs in relationship to both local (e.g., vegetation and water quality) and landscape factors (e.g., canopy cover, open water cover), and I contrast

chorusing anuran communities in SWMFs with those in nearby natural wetlands. I predicted that chorusing anuran species richness would be determined by a combination of local- and landscape-level variables, reflecting the greater sensitivity of anurans to habitat connectivity. I found that chorusing anuran species richness was positively correlated with emergent vegetation. I further predicted that the community composition of chorusing anurans using SWMFs would be a nested subset of species from natural sites and would include fewer species of conservation concern because SWMFs receive contaminated stormwater and may experience water quality issues that limit the survival of chorusing anurans. I found that the chorusing anuran community composition of natural sites was significantly distinct from that of SWMFs. Additionally, I found that chorusing anuran communities in SWMFs were more homogenous in composition (lower beta diversity) and comprised mostly a subset of species also occurring in natural wetlands, with one important exception. I found that the American bullfrog, *Lithobates catesbeianus*, a species of conservation concern, occurred in several of the SWMF sites but in none of the natural wetlands.

2.0 Fish species present in stormwater management facilities in urban areas

2.1 Introduction

Urbanization is currently predicted to be one of the largest drivers of biodiversity loss globally (Kondratyeva et al., 2020; Mcdonald et al., 2013). As human populations grow, the demand to develop natural areas increases, causing the destruction of many freshwater aquatic habitats (Arnold and Gibbons, 1996; Pimm and Raven, 2000). A common feature of urban development is the creation of stormwater management facilities (SWMFs) to mitigate flooding and to detain and treat runoff from impervious surfaces that dominate urban landscapes (Birch et al. 2022). These stormwater ponds do not provide the same range of ecological functions as natural wetlands (Rooney et al., 2015); however, they may comprise some of the only aquatic habitat available in urbanized areas and are often inhabited by a variety of aquatic organisms, including fish (Bishop et al., 2000; Huang et al., 2021), invertebrates (Hassall and Anderson, 2015; Perron and Pick, 2020), anurans (Clevenot et al., 2018), birds (Blackwell et al., 2008) among others (Oertli and Parris, 2019).

Fish are one of the most prevalent organisms in freshwater ecosystems and can be found at high abundances in urban streams and ponds (Hassall, 2014; Oertli and Parris, 2019). Fish are both predators and prey in aquatic food webs, and affect nutrient cycling (McIntyre et al., 2008). Being fully aquatic, the ability of fish disperse to other regions is highly dependent on aquatic connectivity or the movement of other organisms (e.g., egg transfer from birds) (Hirsch et al., 2018). Urban areas generally have fewer corridors and overall less biodiversity than more natural areas, which can leave fish dispersal limited in many sites (Arnold and Gibbons, 1996; Oertli and Parris, 2019). Fish in urban wetlands, ponds, and streams may also be subject to harsh conditions, such as elevated concentrations of chloride from road salts (e.g., van Meter and Swan, 2014), eutrophication issues or elevated concentrations of nitrate or ammonia (Holzer, 2014), heavy metals (e.g., Campbell and Johns, 1994), and other emerging contaminants (e.g., Gillis et al., 2022; Liu et al., 2019), which are common in urban aquatic systems (Hassall, 2014; Oertli and Parris, 2019a). More recently the movement of invasive fish, namely the goldfish (*Carassius auratus*), by humans through pet release has created issues in urban ponds where this species outcompetes native species and is able to survive in a wide variety of water conditions (Campbell, 2021; TRCA, 2016). Their sensitivity to water quality and habitat structure coupled with their low dispersal ability and the introduction of invasive goldfish make fish an ideal taxa for studies in isolated SWMFs.

Although fish are widely studied, their diversity and community patterns are seldom investigated in urban ponds (Huang et al., 2021). In a review of urban pond biodiversity conducted by Oertli and Parris in 2019, only 4% of papers had fish as a key word, and these studies often focused on fish as an explanatory variable rather than a response variable. These studies investigated how fish influence biodiversity from predation on macroinvertebrates, plants, and amphibians (Ficetola and de Bernardi, 2004; Hamer et al., 2008; Hamer and Parris, 2011; McCarthy and Lathrop, 2011) or competition from invasive ornamental species (Copp et al., 2008; Pyke, 2008; Scheffer et al., 1993; van Kleef et al., 2008). The studies that do investigate fish communities in relation to SWMFs are currently more focused on the downstream effects of these ponds via the discharge of contaminants (Bliss et al., 2015; Brix et al., 2010; Stearman and Lynch, 2013). This lack of literature leaves a gap in our knowledge of fish community dynamics within SWMFs, which to my knowledge has only been addressed in two studies (Canada: Bishop et al., 2000; Taiwan: Huang et al., 2021). Huang et al. (2021)

suggest that SWMFs are a suitable habitat for native fish, although they are outcompeted over time by invasive fish species, while Bishop et al. (2000) concluded that SWMFs were toxic systems where high contaminant loading negatively affected fish. This inconsistency in results, along with a general lack of urban fish community studies, calls for an assessment of fish in SWMFs and the drivers that influence their composition and species richness.

Due to the very limited information available about fish use of SWMFs in urban environments, there is limited empirical ground for generating hypotheses about the determinants of fish community structure and dynamics in SWMFs. To address this gap, I investigated whether fish were occupying SWMFs and, if so, what species occurred. Fish species richness was also investigated. The SWMFs studied were located in the City of Brampton, Ontario. I particularly sought to determine whether any species of conservation concern might occur in the SWMFs. I further explored whether environmental characteristics about the SWMFs or their immediate surroundings were associated with fish species richness in SWMFs to generate hypotheses about possible environmental drivers of fish community structure and dynamics in SWMFs. Considering SWMFs as aquatic "islands" of habitat in an "ocean" of impervious land covers, I applied an island biogeography theory lens (*sensu* MacArthur and Wilson, 1967). Due to the dispersal constraints on isolated SWMFs and the poor habitat quality, I anticipated that fish diversity would be low in general. However, SWMFs with more aquatic habitat surrounding them might support higher fish richness through increased colonization rates and SWMFs with better water quality might have higher fish richness through reduced local extinction rates. Consequently, I predicted that a combination of local- and landscape-level habitat factors in SWMFs would be correlated with fish species richness. I further expected that any fish species I did detect using SWMFs would have generalist and opportunistic life history and ecological traits, including a tolerance for warm, low-oxygen, turbid or otherwise poor water quality.

2.2 Methods

2.2.1 Study region and SWMF selection

The field work for this project took place in 2021 in the City of Brampton, in southwestern Ontario (Figure 2-1). Brampton is the fastest growing urban centre in Canada; its population increased by 10.6% over the last decade to roughly 650,000 people (Frisque, 2022). The city ranges from low density residential neighborhoods in the north to high density residential, industrial, and commercial developments closer to Lake Ontario. Despite this north-south gradient in urbanization, the entire city has pockets of green and blue space including parks, forests, wetlands, and agricultural areas. Many of these natural areas line riparian habitats bordering streams that discharge into Etobicoke and Humber Creek (Fig. 2-1).

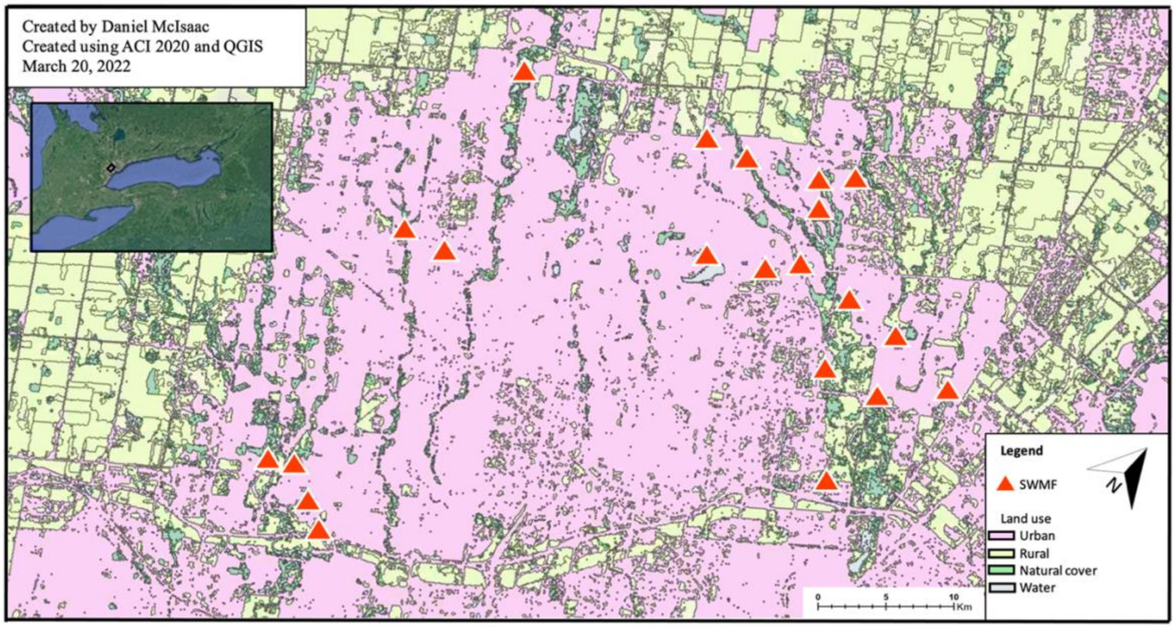


Figure 2- 1. A map of the 20 SWMFs sampled for fish in the 2021 field season. Land use was visualized using the annual crop inventory (ACI) database in QGIS and is colour coded based on urban (pink), rural (yellow), natural cover (green) and water (blue).

I selected 20 sites from the population of 194 mapped SWMFs in Brampton in a stratified random manner. I excluded any sites over 4000 m^2 and under 1000 m^2 in area due to the general positive relationship between fish diversity and area (Barbour and Brown, 2015). I also excluded any sites that were constructed or dredged within the last 10 years due to the diversity-age relationship of metacommunities (Sferra et al., 2017). Lastly, I ensured that all selected SWMFs were a minimum of 600 m apart to control for spatial dependence, especially with buffer widths for landcover data extraction (for which I used a 300 m radius buffer) (Ewers and Didham, 2006). I selected sites from the remaining population of eligible SWMFs to capture a gradient in urbanization intensity (indicated by the percentage of impervious cover within 300 m radius buffer around each SWMF), and an independent gradient in habitat connectivity (indicated by the percent cover of open water – lotic and lentic - within 300 m radius buffers around each SWMF) (Li et al., 2018; Semlitsch and Bodie, 2003). The average SWMF size among the 20 selected sites was 2238 m^2 (minimum 1001 m^2, maximum 3685 m^2, SD± 859.6 m^2). The 20 sites ranged from 10.9%-55.2% impervious cover, and 0%-7.6% open water area within 300 m (See Appendix 3 for full range of landcover at the sampled sites).

2.2.2 Field collections

To assess the fish communities in SWMFs I used minnow traps that were deployed in September and October 2021. I set up to six traps, but no less than four, in each site in the evening, and collected them the following morning after 16 hours. This trapping was conducted under the animal use protocol AUP-41452, registered at the University of Waterloo. I deployed four traps within the littoral zone of each site among vegetated areas of the pond, with two additional traps deployed in the deeper limnetic zone at a subsample of 14 of the sites. I baited each trap with similar amounts of cat food (dry and wet), bread, tin foil, and Cheetos ® by Frito-Lay. Upon collection, I identified each fish in the field using Holm et al. (2009). During identification, I photographed three individuals from each taxa (dorsal and lateral) and euthanized any injured or invasive fish. Identifications were confirmed in the laboratory from euthanized and photographed specimens. During the second sampling period in October, I measured all caught fishes fork length before releasing them back into the SWMF.

To assess the emergent vegetation zones at each site, I took measurements (m^2) of pond area and patches of plants that were emerging out of the water using an SX Blue II GNSS receiver with real time sub-meter accuracy (Geneq Inc., Montreal, QC). I characterized patches of vegetation based on their physical structure as either: 1) robust emergent, 2) narrow leaf emergent, 3) shrubs, or 4) downed trees, as defined in the Ontario Wetland Evaluation System southern manual v. 3.2 (OWES, 2013) and the Marsh Monitoring Program habitat description form (MMP Amphibians, 2009). Briefly, I characterized robust emergent plants as having leaves with flat broad surfaces (e.g., cattails and European common reed), while narrow leaf emergent plants were characterized by smaller, more narrow leaf structures (e.g., rice cut grass and reed canary grass). Shrubs were characterized as any woody vegetation with a height below ~2 m, that was either emerging out of the water, or hanging over the water's edge creating a shadow. Downed trees were measured as any fallen tree that was in the water and not entirely submersed. I also recorded the area of the pond at the water's edge to account for the differences in area amongst the SWMFs. All GPS points were stitched together into polygons in QGIS v3.10 (QGIS Development Team, 2022).

I sampled water quality at each of the 20 sites twice, once in June after a wet period and another in July after an algae bloom in the hottest portion of the summer. At each site I took water temperature (°C) and conductivity (mS cm^-) measurements at three points in the pond at roughly 50 cm depth. The measurements were averaged across each site and sampling period. Additionally, I took 1 L samples of water from three equidistant points at each pond and stored on ice in a dark cooler to be brought to the laboratory for fluorometric determination of chlorophyll-a (mg L^-) and phaeophytin (mg L^-), gravimetric determination of total suspended solids (TSS mg), and measurement of the chloride ion concentration (Cl^- mg L^-). At the same three points, I took separate samples to measure the concentrations of bioavailable forms of nitrogen and phosphorus (mg L^-), in the forms of nitrate (NO_3^-), nitrite (NO_2^-), ammonia (NH_3), and orthophosphate (PO_4^{3-}) in a composite 250 mL sample.

2.2.3 Laboratory methods

For chlorophyll-a and phaeophytin analysis, I rinsed GF/F filters with 50 mL of de-ionized water before filtering a known amount of sample water over the filter and storing it in the freezer. Within three days, each filter was placed in 90% acetone for 24 hours. I then analyzed the samples using a Turner Designs TD 10AU fluorometer.

For TSS, I filtered a measured amount of sample water through a pre-rinsed and weighed GF/F filter, which I oven dried at 100 deg C and subsequently re-weighed.

I measured chloride ion concentration using an ion-selective electrode (Model: ISECl18101, Intellical) immersed in 25 mL of sample of water mixed continuously with a stir plate, using a chloride ionic strength adjustment powder pillow (Method 10255, HACH)

The composite nutrient samples were analysed by ALS Environmental (Waterloo, Ontario) within one week of collection during both sampling periods. Nitrate and nitrite inorganic ions were analysed using ion chromatography with conductivity and UV detection. Orthophosphate was determined colourimetrically on samples filtered through a 0.45 mm membrane filter. Ammonia analysis was carried out using sulfuric acid preserved methods following a modified procedure from Watson et al. (2005).

2.2.4 Statistical analysis

Prior to analysis, I converted fish counts to catch per unit effort (CPUE) through the following formula adopted from Maunder et al. (2006).

$$\text{CPUE} = C/T/S/24$$

Where C is the total catch of a species at one site, T is the number of minnow traps deployed, S is the number of sampling periods and 24 is a constant to represent the hours within a day. This equation assumes that all traps were deployed for exactly 16 hours, though there may be some slight discrepancies in time deployed per site (+/- 20 min per trap). Raw fish data values between sampling periods are listed in Appendix 1.

I used step-wise Akaike's Information Criteria, corrected for small sample size (AIC_c; Burnham and Anderson, 1998) model competitions to identify the combination of 1) local, 2) landscape, and 3) local + landscape variables that best predicted the species richness of fish in SWMFs (n = 20). First, I assessed collinearity among the local and landscape variable sets using a Pearson correlation coefficient (See list of variables in Appendix 2). Variables with an $r > 0.6$, I considered collinear, and I selected only one of the pair to retain in the modeling based on the ecological explanation. Following this I removed variables that did not exceed the tolerance envelope for any of the species detected.

Due to the zero inflation of fish richness, I used generalized linear models with a Negative binomial distribution to perform backwards step-wise AIC model competition. I determined the appropriate theta value using the 'glm.nb' function in the *stats* package in base R version 4.1.3 and used the recommended link function (R Core Team, 2021). I preformed model competition on the three subsets of variables: local variables only, landscape variables only, and local and landscape variables combined. To predict fish species richness, I used a penalty of two degrees of freedom for removing variables for a maximum of 1000 steps to achieve the final model. To carry out this analysis I used the 'StepAIC' function in the *MASS* package in R version 4.1.3 (Ripley et al., 2022), with AIC as the criterion for determining the order in which effects enter and leave at each step.

Next, I compared the best model that used only local variables to the best model using only landscape variables and the best model using both local and landscape variables, based on their respective AIC_c (Burnham and Anderson, 1998). Models were ranked using ΔAIC_c, where

$\Delta AIC_c = AIC_{ci} - AIC_{cmin}$ when AIC_{ci} represents the i^{th} model from the candidates. The akaike weights (W_{AICC}) were calculated to assess the likelihood that each model was the best from the range of candidate models. I carried out this analysis using the 'aictab' function in the *AICcmodavg* package (Mazerolle, 2020) in Rstudio (R Core Team, 2021).

2.3 Results

Fish were unexpectedly common in the SWMFs, with only 8 of 20 ponds having no fish trapped (Table 2-1). A total of six different species of fish were detected in the 20 SWMFs I surveyed, of which five were native to southern Ontario. The non-native goldfish (*C. auratus*) was the sixth species. Goldfish are opportunistic feeders that consume invertebrates and plant matter and are able to survive periods of anoxia (Walker and Johansen, 1977). The five native species detected were predominately opportunistic benthivores which were all tolerant to a variety of disturbances and high water temperatures (Table 2-2).

Table 2- 1. The average CPUE per trap adjusted to a 24 hr period as well as median and standard deviation CPUE values for each detected fish species across the 20 SWMFs, sampled in August and September of 2021.

Site	Goldfish	Fathead minnow	Pumpkinseed sunfish	Brook stickleback	Brown bullhead	Creek chub
1	0	0	0	0	0	0
3	0.27	0.0034	0	0	0	0
4	0.41	0	0	0	0	0
5	0.55	0	0	0	0	0
6	0	2.04	0.75	0.17	0	0
7	0	0	0	0	0	0
8	0.0034	6.65	0	0	0	0
9	0	0	0	0	0	0
10	0	0	2.57	0	0	0
11	0	0	0	0	0	0
12	0.0034	5.54	0	0	0	0
13	0	0	0	0	0	0
14	0	10.04	0	0	0	0
15	0	0	0.31	0	0.13	0.04
16	0	0	0	0	0	0
17	0	0	0	0.14	0	0
18	0	0	0	0	0	0
19	0.29	0	0	0	0	0
20	0	10.44	0	0	0	0
21	0	0	0	0	0	0
Median	0.28	6.09	0.75	0.15	0.13	0.04
SD	0.17	3.47	0.59	0.05	0.03	0.01

Species	Prevalence	Feeding behaviour	Habitat preference	Water quality tolerance	References
Brown bullhead	• 5 % • CPUE: 3.33	• Benthivore • Opportunistic omnivore Wide diet range	• Wide variety of slow-moving or lentic water bodies	• Disturbance-tolerant • Temperature envelope: 10 - 33°C • High salinity tolerance • Dissolved oxygen > 0.2 ppm	(Becker, 1983; Kline et al., 1996)
Creek chub	• 5 % • CPUE: 1	• Generalist carnivore • Invertebrates, fish, and anurans	• Creeks, small rivers, and lakes • Prefer habitats with vegetation and coarse substrate	• Disturbance tolerant • Temperature envelope: 1.7 - 32°C • Dissolved oxygen > 9.7 ppm	(Becker, 1983; Holm et al., 2009; Smiley et al., 2017)
Fathead minnow	• 30 % • CPUE median: 146.67	• Benthivore • Opportunistic omnivores (algae and protozoans)	• Lentic water (ditches, ponds, headwaters) • Highly disturbed ponds with low fish diversity	• Disturbance tolerant • Temperature envelope: 8.8 - 32°C • Salinity < 10000 ppm • Tolerant to low dissolved oxygen	(Chivers et al., 1996; Holm et al., 2009; Page and Burr, 2011)
Goldfish	• 20 % • CPUE median: 7.34	• Benthivore • Opportunistic omnivore • Wide diet range	• Small waterbodies • Dense vegetation (though not needed)	• Disturbance tolerant • Temperature envelope: • 0 - 41°C • Salinity < 17000 ppm • High turbidity • Tolerant to low dissolved oxygen	(Holm et al., 2009; Page and Burr, 2011)
Pumpkinseed sunfish	• 15 % • CPUE median: 23.33	• Benthivore • Omnivorous, primarily macro-invertebrates and plant debris • Pelagivore in larger waterbodies	• Small waterbodies • Dense vegetation and clear water	• Disturbance tolerant • Temperature envelope: 10°C - 33°C • Moderate salinity tolerance	(Holm et al., 2009; Jordan et al., 2009)

Brook stickleback	• 10 % • CPUE median: 6.35	• Benthivore • Carnivorous, mainly invertebrates	• Lakes and ponds with moderate to dense vegetation	• Disturbance tolerant • Temperature envelope: 4 - 31°C • Salinity < 15000 ppm • Tolerant to low dissolved oxygen	(Becker, 1983; Stewart et al., 2007)

Table 2- 2. Prevalence percentage and life history ecology of each of the six detected species along with their relative water quality tolerances.

Catch per unit effort was generally low (Table 2-1; Bishop et al., 2000), with the highest median CPUE for fathead minnow (*Pimephales promelas*) (147 fish per 24 hr per trap; Table 2-2). Fathead minnow were also the most common and abundant, being detected in 30% of SWMFs. Invasive goldfish (*C. auratus*) were less common, found in 20% of SWMFs, but where they occurred, they could reach high abundances (max CPUE = 0.55; Table 2-1). Pumpkinseed sunfish (*Lepomis gibbosus*) and brook stickleback (*Culaea inconstans*) were less common, being present in 15% and 10% of ponds respectively, and creek chubs (*Semotilus atromaculatus*) and brown bullheads (*Ameiurus nebulosus*) were the least common and were found in only 5% of ponds (i.e., one SWMF).

The average species richness of fish per site was 1 SD± 1, range: 0 - 3. Due to collinearity, Cl^- was chosen over conductivity due to its direct presence in road salts, which I predicted to be the main source of salinity. Cl^- was also highly correlated with $NO2^-$ and $NO3^-$, however Cl^- remained in the models due to levels on average (1,100 mg L^- SD ± 757) exceeding the Canadian Water Quality Guidelines (short term exposure 640 mg L^{-1}; long term exposure 120 mg L^{-1}; CCME, 2011), while $NO2^-$ and $NO3^-$ were generally low. NH_3 and PO_4^{3-} were also correlated, and NH_3 was included in the final model due to its average levels (0.20 mg L^{-1} SD ± 0.51) far exceeding the Canadian Water Quality Guidelines (0.019 mg L^{-1}) and therefore could have toxic effects on fish. I also found high correlations with shrubs and down trees and removed down trees from the models since shrubs can both produce habitat and shading effects for fish (Miranda and Hodges, 2000), and floating downed trees may not be representative of the amount of fully submerged trees at the sites. From the landscape variables, road area and area of impervious cover were highly correlated, and I removed impervious cover from the models due to roads likely being a better indicator for water quality due to catchment placement. I included NH_3, Cl^-, area of robust emergent vegetation, area of shrubs and pond area as local variables and canopy cover, road area and water area as landscape variables. These variables were also combined in a final step-AIC model for local and landscape variables in determining the best subset of variables for predicting species richness. The best subset of variables for the local and the local and landscape models included area of shrubs and Cl^- (Table 2-3; $\Delta AIC_c = 0.00$, $W_{AICC} = 0.78$, $r^2 = 0.30$), while the best subset for the landscape model only included canopy cover (Table 2-3; $\Delta AIC_c = 0.00$, $W_{AICC} = 0.82$, $r^2 = 0.03$). The best subset of these models was the local model (Table 2-4; $\Delta AIC_c = 0.00$, $W_{AICC} = 0.85$, $r^2 = 0.30$).

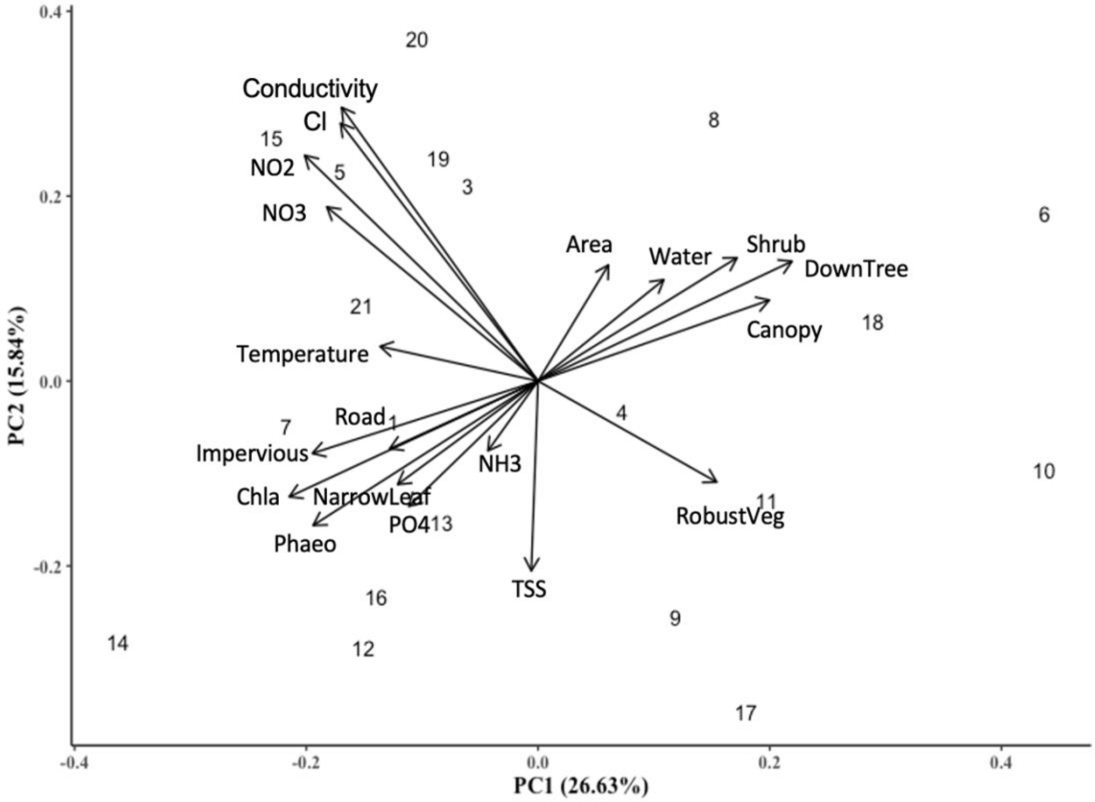


Figure 2- 2. A principal component analysis displaying all local and landscape environmental covariates measured in this study displayed on two axes. Sites are numbers in concordance with Table 2-1

Table 2- 3. AIC_c table for the top models in the Step-wise AIC for local and landscape, local and landscape models of fish species richness within two AIC from the top model. Direction of estimate is labeled as **(+)** for a positive correlation and **(-)** for a negative correlation. Local and local and landscape models produced the same top models

Model type	Model	K	AIC	AICc	ΔAICc	W_{AICC}	$W_{cum.}$	LL	r^2
	(+) Cl^- + **(+)** Shrub	4	51.54	58.97	0.00	0.78	0.78	-24.15	0.30
Local /	**(+)** Cl^- + **(+)** Shrub +	5	52.02	61.93	2.96	0.18.	0.95	-23.82	0.33
Local and	**(-)** NH_3	6	52.96	64.63	5.66	0.05	1.00	-23.09	0.37
landscape	**(+)** Cl^- + **(+)** Shrub + **(-)** NH_3 + **(+)** RobustVeg								
	(+) Canopy	3	54.58	62.43	0.00	0.82	0.82	-27.47	0.03
Landscape									
	(+) Canopy + **(+)**	4	56.42	65.42	2.99	0.18	0.18	-27.38	0.04
	Water								

Table 2- 4. AIC_c model selection table of only local, only landscape, and local and landscape variables following backwards step-wise selection using AIC_c.

Model	K	AICc	ΔAICc	W_{AICC}	$W_{cum.}$	LL	r^2
Local / Local and landscape Cl^- + Shrub	4	58.97	0.00	0.85	0.85	-24.15	0.30
Landscape Canopy	3	62.43	3.47	0.15	1.00	-27.47	0.03

2.4 Discussion

In this chapter, I explored the use of SWMFs by fish and aimed to determine if fish are present in these systems, particularly any species of conservation concern. The SWMFs contained high levels of contaminants such as Cl^-, which was positively correlated with conductivity, NO_2^- and NO_3^- (Figure 2-2). These variables were negatively correlated with robust emergent vegetation, which is not surprising considering vegetation can reduce contaminants in water through phytoremediation, however high concentrations of Cl^- can also destroy aquatic vegetation (Guesdon et al., 2016; Jesus et al., 2014; Schück and Greger, 2022). Interestingly these variables were not strongly correlated with area of roads, which would be the source of these contaminants, however it is likely that the storm drains, where the water would collect, would be more indicative of these contaminants. Roads were positively correlated with impervious cover, Temperature and NH_3, which is not surprising considering areas with higher development are generally hotter. These variables were all negatively correlated with area of shrubs as well as canopy and water cover within 300 m, and generally less development would lead to increases in natural cover. Canopy cover and open water area are both positively correlated which is not surprising considering watersheds within Ontario generally use forests as a buffer around main rivers, and any discrepancy between the variables would be from wetlands or other SWMFs in the area.

Despite grates and other structures in place to prevent colonization of SWMFs, fish were captured in minnow traps in 60% of the ponds I surveyed. Abundances were fairly low, and the species present were generalists; none are of conservation concern. Goldfish and fathead minnows were the most prevalent and abundant species in SWMFs; both are warmwater fish that are tolerant to low oxygen concentrations. These species were generally either the only species present at a particular site (Table 2-1), or found together in sites with high Cl^- (Figure 2-2), which is likely attributed to their ability to overtake ponds and outcompete other species of fish (Holm et al., 2009). The other three species were detected at <20% of sites, and included brown bullhead and pumpkinseed sunfish, which are also warmwater. Brook stickleback and Creek chub are less tolerant to warm water and were found in 10% and 5% of sites, respectively. These species were generally found in sites that contained no fathead minnows or goldfish (Table 2-1). These findings align with the only other SWMF study of fish within the southern Ontario region by Bishop et al. (2000). These authors found a total of eight species, with low richness across sites. Three species were found in Bishop et al. (2000) that were detected in the SWMFs in my study (pumpkinseed sunfish, brook stickleback and goldfish). Bishop et al (2000) found that pumpkinseed sunfish were the most common species – detected at 40% of their sites vs. 15% of my sites - while goldfish were only detected at 13% of their sites vs. 20% of my sites. It is possible that the difference in goldfish prevalence reflects an increase in invasions across southern Ontario during the 20 year interval between our studies. I further explored if environmental characteristics about the SWMFs or their immediate surroundings were associated with fish species richness to generate hypotheses about possible environmental drivers of fish community dynamics, which have been under-studied to date. Richness was low, with gamma diversity (total richness across the 20 sites) of six species, of which only five species were native. Alpha richness was also low, averaging only one species per SWMF. Using a biogeography theory lens, I hypothesized that better local water quality, as well as natural cover types surrounding the ponds would increase fish species richness due to decreased extinction events and increased colonization. I found that local water quality variables were the best predictors of fish species richness, suggesting that water quality is likely more important in limiting fish richness than landscape connectivity. This could be attributed to local variables already accounting for landscape metrics, due to their correlation from contaminants, or due to the cryptic underground connectivity of SWMFs and sewer drains which may not be represented by a buffer around the retention pond.

My results are a precursor to understanding the use of SWMFs by fish in Brampton, ON, and future work should consider investigating several hypotheses stemming from my research. Hypothesis 1: I hypothesize that species richness is limited by dissolved oxygen (DO) in the water. Oxygen could be limiting in the hottest part of summer when water temperatures could reach as high as 32 °C and high nutrient concentrations could result in substantial biological oxygen demand, but principally I expect oxygen is limiting in the winter under ice (Eaton et al., 2005). Although I did not measure winter DO my study, I found high abundances of goldfish and fathead minnows which are capable of surviving in anoxic conditions (Ankley and Villeneuve, 2006; Walker and Johansen, 1977). These species were also often found in in sites without the rarer species, such as creek chub, which are sensitive to warm temperatures and likely intolerant of low DO (Holm et al., 2009). Hypothesis 2: I hypothesize that fish species richness and CPUE of rare species will increase in SWMFs with increased aquatic vegetation. In my study I saw correlations between species richness and shrubs and CPUE of three species with robust emergent vegetation (Figure 2-2; Table 2-4). This was not surprising considering increased vegetation can increase DO (Miranda and Hodges, 2000), and has historically been associated with habitats that include pumpkinseed, brook stickleback and creek chub. Hypothesis 3: I hypothesize that most Cl^- and nutrients have small effects on urban generalist fish species. I found a positive correlation between Cl^- and species richness as well as CPUE of goldfish and fathead minnows, which is interesting considering Cl^- can decrease fish richness through direct effects on body metabolism (Boeuf and Payan, 2001) or indirectly through altered food webs (e.g., Hintz et al., 2017; Table 2-3). Since Cl^- was also correlated with the suite of other contaminants measured in my study (Figure 2-2), I interpret this to indicate that fathead minnows and goldfish are Cl^- tolerant and likely tolerant of other road run-off contaminants, at least to the range of concentrations I observed in my study. The only nutrient that I saw correlations with a reduction of richness was NH_3 (Table 2-3). Dissolved oxygen can decrease in aquatic systems from oxidizing NH_3 (Weon et al., 2004), and levels on average (0.20 mg L^{-1} SD ± 0.51) exceeded the Canadian Water Quality Guidelines for the Protection of Aquatic Life (0.019 mg L^{-}

[1]; CCME, 2010). However, future work should continue to look at the role of contaminants and fish in SWMFs. I recommend these studies use conductivity as a proxy for Cl^- due to their strong positive relationship (r = 0.98), and the decrease of cost and labour for sampling. Improvements in NH_3 and Cl^- might permit greater diversity of fish species to persist in SWMFs and I do not recommend we take measures to increase salinity in these systems. Hypothesis 4: I predicted that fish species richness and CPUE would be positively correlated with the extent of open streams in the surrounding landscape, due to increased connectivity for colonization. I found an association between canopy cover within 300 m and fish species richness (Table 2-3), as well as open water area and CPUE of pumpkinseed sunfish (Figure 2-2), but the mechanisms of dispersal are unknown. In Brampton, ON, forest cover is generally indicative of the Etobicoke or Humber River, which are systems that may promote fish dispersal better than wetlands, which were also incorporated in the open water metric. Considering that SWMFs in Brampton are isolated from direct surface connections to streams, I hypothesize that fish colonization – at least colonization unassisted by human introductions – most likely occurs through egg or juvenile dispersal, either during large flood events or by larger animals moving among ponds (Hirsch et al., 2018). However, some of the species I encountered are also common baitfish (e.g., creek chub, fathead minnow) and the goldfish are most likely introduced by pet owners. Further research should investigate age-specific dispersal rates and mechanism, including the role of humans in moving adult fish among SWMFs.

Table 2- 5. Variables highlighted as important for fish species richness and CPUE that require further analysis in future research. Mean, standard deviation and range of values are presented from the 20 SMWFs I surveyed in 2021. Hypotheses that I suggest requires further investigation in future research

Variable	Mean ± SD	Range	Hypothesis
Shrub vegetation (m^2)	1.43 ± 2.05	0 – 7.33	Positively correlated with CPUE and species richness
Cl^- (mg/L)	1100.06 ± 746.86	212 - 2679	Relatively small effect to urban generalist species
Canopy cover within 300 m (m^2)	51964 ± 40743	4785.11 – 137841.01	Positively correlated with CPUE of rare species, likely indicative of streams
NH_3 (mg/L)	0.20 ± 0.49	0.005 – 2.32	Negatively correlated with species richness
Open water area within 300 m (m^2)	16641.26 ± 6321.59	0 – 14746.30	Positively correlated with rare fish CPUE and species richness

This chapter comprises the first investigation of fish in southern Ontario SWMFs in over 20 years (Bishop et al., 2000). I found that fish abundance was generally low and fish communities included exclusively generalist species that are tolerant to urbanization. I outlined several environmental variables that should be looked at further in any ongoing research (Table 2-5), and highlight the probable importance of DO, which was not measured in this study. Future research should expand on this study by increasing the variation of gear used, such as fyke nets or electrofishing. Since fishing gear is known to introduce sampling bias (e.g., Fisher and Quish 2014) and my study was limited to minnow traps, there is potential that I missed larger-bodied species and likely underestimated fish species richness. The means of fish dispersal in SWMFs also needs to be quantified, and future studies could contrast fish communities in SWMFs to neighbouring streams, or contrast SWMFs that were designed to impede fish movement (like those I surveyed) with SWMFs that directly connect to a natural water source. Although more work is needed to gain an effective understanding of fish use in SWMFs, my study confirms that fish are using SWMFs and highlights the need for greater public education regarding invasive species release in aquatic habitats.

3.0 Anurans in stormwater management facilities are sensitive to local conditions but can resemble anuran communities in urban wetlands.

3.1 Introduction

The rapidly growing human population and urbanization have caused population declines of many animal species worldwide (Pimm and Raven, 2000). Urbanization leads to the conversion of natural heritage features to residential, commercial, and industrial land covers and an increase in road density, resulting in wildlife habitat loss and increased area of impervious surfaces (Arnold and Gibbons, 1996). Wetlands are especially prone to destruction through urban development, endangering their sensitive species (Alikhani et al., 2021). Urbanization not only removes wetlands, but results in the creation of stormwater management facilities (SWMFs) to protect infrastructure and buildings from flooding and to control and treat runoff (Rooney et al., 2015c). Consequently, there is indirect replacement of wetlands by SWMFs in urbanizing environments, though this is not a one-for-one replacement (e.g., Birch et al. 2022).

Although the indirect replacement of natural wetlands with SWMFs maintains flood prevention and runoff control functions in the urbanized landscape, this does not fully replace the portfolio of ecosystem services that natural wetlands provide (e.g., Rooney et al., 2015). The habitat value of SWMFs for wildlife is currently debated, though the bulk of current literature suggests they are poor habitats that are toxic ecological traps (i.e., a low quality habitat that is chosen over a high quality habitat in the area, where an organism has a resulted reduction in fitness; see Clevenot et al., 2018; Connor et al., 2012; Sievers et al., 2018). Pollution from contaminated runoff and habitat fragmentation reduce habitat quality for wildlife and result in higher rates of invasive species introductions (Alikhani et al., 2021; Ehrenfeld, 2000; Ravit et al., 2017). These stresses are likely higher in SWMFs than in natural wetlands situated in urban landscapes because SWMFs are designed to catch and treat runoff, and are not generally constructed to create high quality habitat for wildlife (Rooney et al., 2015c; Ross et al., 2018). In contrast, natural wetlands remaining in urban landscapes can play an important role in the conservation of biodiversity, even if they are degraded relative to wetlands in less urbanized landscapes (Alikhani et al., 2021).

The loss, fragmentation, and degradation of habitats associated with wetland destruction and SWMF creation negatively impact amphibian communities (Hamer et al., 2008). Chorusing anurans are known to use SWMFs as breeding sites and aquatic refugia (Hamer and Parris, 2011; Simon et al., 2009). Their porous skin, relatively small home ranges, and low dispersal capabilities leave them highly susceptible to poor water quality (Sievers et al., 2019), suboptimal habitat structure (Hamer et al., 2012), and alterations in landscape composition and connectivity (Hamer and Parris, 2011). These sensitivities have resulted in amphibians currently being the most at-risk group of vertebrates globally with roughly 34% of species listed on the IUCN red list worldwide (Hamer et al., 2008; IUCN, 2021). The sensitivity of anurans to urbanization makes them an ideal candidate as a bioindicator when evaluating the biodiversity and habitat value of SWMFs.

A variety of environmental conditions influence community composition and species richness of anurans in SWMFs. Locally, chloride ions (Cl^-) from road salt (Brand and Snodgrass, 2010; Brown et al., 2012; Collins and Russell, 2009; Gallagher et al., 2014), nitrogen pollution (Boone and Bridges, 2003; Massal et al., 2007), metal contamination (Bishop et al. 2000), and the presence of predatory fish (Ficetola and de Bernardi, 2004; Hamer et al., 2008; Hamer and Parris, 2011; McCarthy and Lathrop, 2011) are associated with anuran morbidity and extirpation, while the abundance of aquatic vegetation is associated with anuran diversity (McCarthy and Lathrop, 2011).

At the landscape scale, road density (Hutto and Barrett, 2021) and impervious cover (Simon et al., 2009) are negatively associated with amphibian occurrence and richness, while greater landscape connectivity through corridors of tree canopy, natural land cover, and cover of waterbodies are positively correlated with anuran richness and occurrence (Clevenot et al., 2018). A review by Hamer et al. (2008) discusses the need to assess environmental conditions both locally (within the pond) and in the surrounding landscape to accurately detect drivers of anuran community patterns in urban ponds as local and landscape drivers can combine unexpectedly. Yet, no studies to my knowledge have previously compared the influence of both local- and landscape- level factors as possible drivers of chorusing anuran occurrence patterns in Ontario SWMFs, or compared chorusing anurans in natural wetlands to those in SWMFs across a range of urbanization.

In this chapter, I describe a study I carried out to assess the local and landscape drivers of chorusing anuran richness and community composition in SWMFs in the City of Brampton, Ontario. These SWMFs ranged from low to high degrees (10-55%) of impervious cover, reflecting variation in the degree of urbanization present across the municipality. Further, I compared the community composition of chorusing anurans in these SWMFs to those in natural wetlands across the same range in urbanization. I contrasted three models predicting chorusing anuran richness in SWMFs. The first was based entirely on local characteristics, such as water quality and vegetation cover within the pond. The second was based entirely on landscape-level characteristics such as the extent of open water area, road density, or impervious cover surrounding each pond. The third combined both local- and landscape-level factors. Based on the review by Hamer et al (2008), I predicted this third model would best predict chorusing anuran richness in SWMFs. I further predicted that both local- and landscape-level factors would be strongly associated with variation in chorusing anuran community composition among my SWMFs. I also contrast the richness and community composition of chorusing anurans in SWMFs to those in natural wetlands across the same region. I predicted that although the landscape-level factors in SWMFs and natural wetlands in the municipality would be similar, the composition of chorusing anurans would differ, with SWMFs being less diverse. I based this prediction on the inferior water quality I expect would occur in SWMFs, which are designed to receive and treat contaminated stormwater runoff (Bishop et al., 2000).

3.2 Methods

3.2.1 Field methods

The field work took place in Brampton, Ontario and its immediate environs within the Greater Toronto Area (Fig. 3-1; details in chapter 2.2.1). In brief, I selected 21 SWMFs ranging in size from 1,001–3,685 m^2 (SD 876.9; See Appendix 6-2 for the range of landcover surrounding sampled SWMFs). These sites spanned a gradient of impervious cover (10.9 to 55.2%) and an independent gradient in open water cover (0 to 7.6%) within a 300 m radius buffer (Detailed in chapter 2.2.1). An additional 21 natural wetland sites were chosen from the Toronto and Region Conservation Authority (TRCA) Long-term Monitoring Program, which is designed to measure temporal changes in ecosystems within the region. Sites were paired to SWMFs covering an urban to rural gradient.

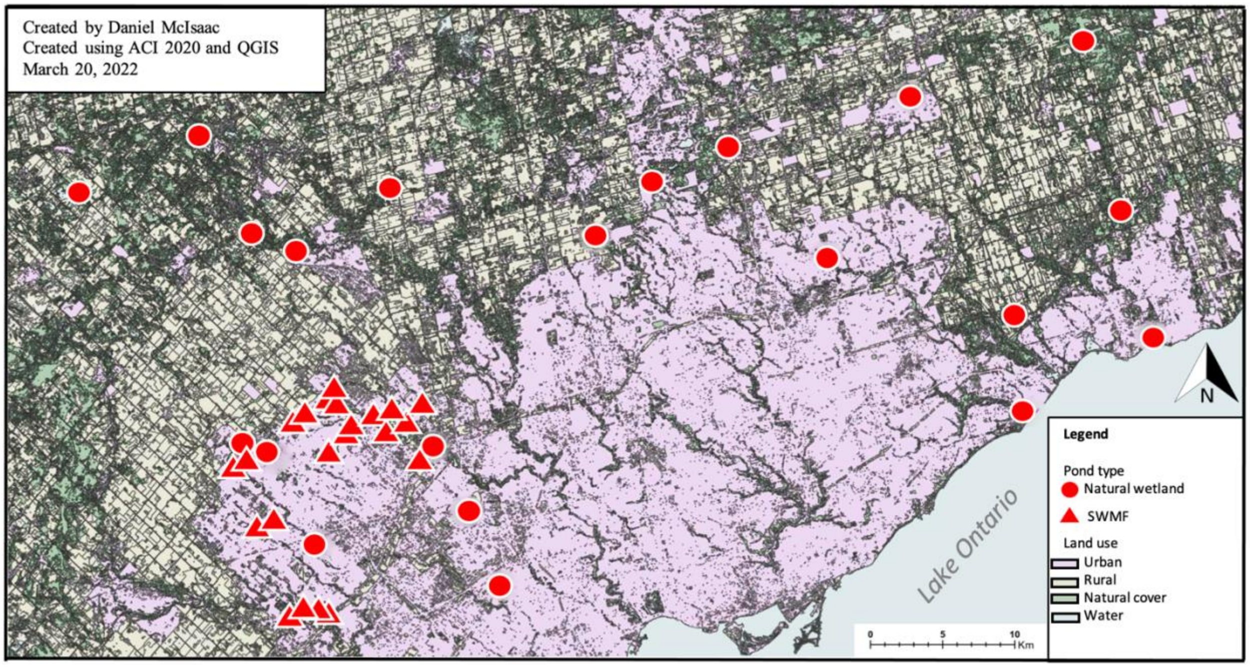


Figure 3- 1. A map of the 42 ponds surveyed during the 2021 field season. Sites are symbolized based on pond type (total n = 42; SWMF (triangle) n = 21, rural natural wetlands (circle) n=11, and urban natural wetlands (square) n=11). Land use is colour coded based on urban (pink), rural (beige), natural cover (green) and water (blue) from the 2020 Annual Crop Inventory database.

Anuran sampling at SWMFs and natural wetlands followed the Great Lakes Marsh Monitoring Program protocol, which is designed to cover the peak breeding period of all 13 anuran species found in southern Ontario (MMP Amphibians, 2009). Three auditory surveys were conducted between April and June of 2021 at each site. The first survey was completed in April to monitor the early breeding species: spring peepers (*Pseudacris crucifer*), western chorus frogs (*Pseudacris triseriara*) and wood frogs (*Lithobates sylvaticus*). The second survey was conducted in May to detect spring peepers, northern leopard frogs (*Lithobates pipiens*), American toads (*Bufo [Anaxyrus] americanus*), pickerel frog (*Lithobates palustrisis*) and Fowlers toad (*Anaxyrus fowleri*). The final survey was done in June to survey the late breeding anurans: tetraploid gray treefrog (*Hyla versicolor*), Cope's gray treefrog (*Hyla chrysoscelis*), mink frog (*Lithobates septentrionalis*), American bullfrog and green frogs (*Lithobates clamitans*). All surveys started during the half an hour after sunset and ended before midnight, though this was extended to after midnight during the June surveys. Though I carried out the surveys of SWMFs personally, the natural wetlands were surveyed by expert staff from the TRCA. The TRCA also categorized the natural wetlands into urban and rural subcategories on the basis of their landscape context, though not through a quantitative assessment of cover, as in the manner that I characterized the SWMFs.

Before anuran surveys took place, wind speed, cloud cover, temperature, precipitation, and background noise were recorded (MMP Amphibians, 2009). Air temperatures were at optimal levels for peak calling activity during each survey (greater than 5°C for the first round, 10°C for the second and 17°C for the third period), with little to no wind or rainfall. At each SMWF, a station was set up roughly 1 m from the shore where a three-minute-long auditory survey was completed. Every species heard was recorded along with a species abundance index (See Appendix 4 for abundance index description). For species abundance, I recorded the highest abundance index of each species at every site during the three auditory surveys. To determine the species richness, I summed the total number of species recorded at each site from each sampling period. Species that were recorded multiple times over each sampling period were only counted once. Note that anuran species in the study area have been assigned a Local rank (L-rank) by the TRCA based on habitat dependence and area sensitivity (Detailed in TRCA, 2017). L-ranks ranging from 1-3 are considered species of regional conservation concern, while 4-5 are of least concern.

Fish absence or presence was established using minnow traps, as detailed in 2.2.2. These traps were supplemented with visual detections of fish spawning or feeding. The extent of emergent vegetation (broken down by vegetation type), water quality and landscape characterization were determined at each site, as described in section 2.2.1 and 2.2.2

3.2.2 Statistical analysis

3.2.2.1 Local and landscape predictors of anuran richness in SWMFs

Prior to analysis, one site (site 16, see Appendix 3) was removed based on suspicion that it was overly influenced by the nearby conservation area and was not representative of SWMFs generally. Substantiating this, it has a statistically outlying NH_3 concentration and contained two rare anuran species: western chorus frog and northern leopard frog. These anurans were likely from the conservation area.

Due to the discrete nature of the anuran richness, I used generalized linear models with a Poisson distribution. I used step-wise Akaike's Information Criteria, corrected for small sample size (AIC_c; Burnham and Anderson, 1998), to undertake model competitions to identify the combination of 1) local, 2) landscape, and 3) local + landscape variables that best predicted the species richness of anurans in SWMFs (n = 21). This approach to model competition is described in detail in section 2.2.4

3.2.2.2 Community composition of anurans in SWMFs

I used non-metric multidimensional scaling (NMDS) ordination to visualize trends in community composition of anurans in SWMFs and to relate community composition to variation in local- and landscape-level variables. One SWMF had no anurans detected in it, and so was dropped from the analysis, such that n = 19. I calculated a Bray-Curtis dissimilarity matrix using the chorus abundance index from each site using the 'metaMDS' function in *vegan* package in R version 4.1.3 (Oksanen et al. 2020; Ricotta and Podani, 2017). I used scree plots to visualize stress against dimensionality to decide on the optimal number of dimensions for each NMDS, considering stress values below 0.1 fair and below 0.05 to indicate a good fit. To visualize the relationships between variation in community composition and local- and landscape-level variables, I used the 'envfit' function in vegan (Oksanen et al., 2020) to plot variables that were reasonably correlated ($r^2 > 0.10$) with at least one NMDS axis for the ordination of local variables, and to plot all variables for the landscape ordination. To conduct the ordination, I used R 4.1.3 (R Core Team 2021) and the 'ggplot' function in *ggplot2* (Wickham, 2021).

To test whether the community composition of anurans differed between SWMFs with fish detected vs. SWMFs with no fish detected, I used a permutated analysis of variance (PerMANOVA; Anderson, 2017) based on the same Bray-Curtis dissimilarity matrix derived from chorus abundance index values for each anuran detected. I implemented this analysis using R version 4.1.3 (R Core Team 2021) and the 'adonis' function in the *vegan* package (Oksanen et al., 2020).

3.2.2.3 Comparing anurans in SWMFs and natural wetlands

To visualize differences in community composition between SWMFs and natural wetlands, I again used NMDS on anuran abundance index scores from each site. Ordinations were performed using a Bray-Curtis dissimilarity matrix as in section 3.2.2. To determine if there was a significant difference between the community composition of anurans in natural sites compared to SWMFs, I again used a PerMANOVA on the same Bray-Curtis dissimilarity matrix, also as described in section 3.2.2.

Lastly, to test for differences in anuran species richness among the SWMFs and natural wetlands, I performed a permutational ANOVA using the 'aovp' function in the *LmPerm* package (Wheeler and Torchiano, 2016) in R 4.1.3 (R Core Team 2021). The non-parametric test was used because data did not meet the assumptions of normality.

3.3 Results

A total of 6 anuran species were found in SWMFs during the 2021 field season. The most common and abundant anuran was the green frog, which was detected at 95% of the sites, while all other species were relatively rare, detected at <25% of the SWMFs. Breaking the natural wetlands down into urban and rural sub-categories, urban natural wetlands had seven species with green frogs most abundant (detected at 70% of urban natural sites). Rural natural wetlands had 5 species, though detections of each species were more common than in either SWMFs or urban wetlands. Wood frogs and spring peepers were the most common and abundant species in rural natural wetlands with 100% and 95% detection rates, respectively. The occurrence, L-rank, life history and general tolerance to disturbance of each species is detailed in Table 3-1.

Table 3- 1. The detection, life history, ecology and disturbance tolerance of the eight species of anurans found during the 2021 sampling period.

Species	Local rank	SWMF Detection	Size and habitat preference	Life history and ecology	Disturbance tolerance	References
American bullfrog	2	• 20% of sites • Abundance: • Min – 0 • Max – 1 • Median - 1	• Large bodied (10-20 cm) • Highly aquatic • Require large permanent waterbodies • Abundant vegetation	• Breed: May – September • Eggs hatch within one week • Juveniles overwinter as aquatic tadpoles • Metamorphosis in 1-3 years • Dispersal rate unknown, potentially < 1km • Hibernate in deep waterbodies • Lifespan 5-10 years	• Disturbance tolerant • Tolerant of a variety of contaminants including high Cl^- levels, when in permanent ponds • Invasive species outside home range (western North America) • Moderately susceptible to road mortality	(Boone et al., 2008; Harding and Holman, 1992; Matlaga et al., 2014; MMP Amphibians, 2009; Peterson et al., 2013)

Species	Local rank	SWMF Detection	Size and habitat preference	Life history and ecology	Disturbance tolerance	References
American toad	4	• 15% of sites • Abundance: • Min – 0 • Max – 1 • Median – 1	• Medium bodied (5-11 cm) • Open woodlands, meadows, and shallow ponds • Dense terrestrial vegetation	• Breed: April – June • Eggs hatch in 3-12 days • Metamorphosize in 2-3 months • Dispersal ~1 km during breeding • Hibernate in terrestrial habitat • Lifespan 1-10 years	• Disturbance tolerant • Generalists, insensitive to landscape changes • Predated by wood frogs	(Harding and Holman, 1992; MMP Amphibians, 2009)
Tetraploid gray treefrog	2	• 15% of sites • Abundance: • Min – 0 • Max – 2 • Median – 1	• Small bodied (3-5 cm) • Terrestrial woody areas near temporary or permanent waterbodies • Juveniles prefer vegetation in ponds	• Breed: May – June • Eggs hatch in 3-7 days • Metamorphosize In 1-2 months • Limited information, > 1 km from breeding site • Hibernate in terrestrial habitat • Unknown	• Moderate disturbance tolerant • Moderately tolerant to contaminants • Few studies differentiate between diploid and tetraploid species complex	(Conte and Roble, 1903; Harding and Holman, 1992)
Green frog	4	• 95% of sites • Abundance: • Min – 0 • Max – 3 • Median – 1	• Large bodied (7-12 cm) • Wide variety of habitat from small ponds to lakes • Prefer vegetated shores for breeding	• Breed: May – September • Eggs hatch in 3-6 days • Juveniles may overwinter as aquatic tadpoles • Metamorphosis in 3-22 months • Large dispersal rate (> 1 km) • Hibernate at the bottom of waterbodies • Lifespan 5-10 years	• Disturbance tolerant • Able to survive in almost any lentic waterbody • Moderately susceptible to road mortality • Moderately susceptible to Cl^-	(Collins and Russell, 2009; Conan et al., 2022; Harding and Holman, 1992)

Species	Local rank	SWMF Detection	Size and habitat preference	Life history and ecology	Disturbance tolerance	References
Northern leopard frog	3	• 10% of sites • Abundance: • Min – 0 • Max – 1 • Median – 1	• Medium bodied (6-11 cm) • Permanent and semi-permanent pools, ponds, marshes, and lakes • Moist upland meadows	• Breed: April – July • Eggs hatch in less than 9 days • Metamorphosis within 60-90 days • Large dispersal rates (1-3 km, up to 10 km in some populations) • Hibernate in well oxygenated waterbodies • Life span 4-5 years	• Moderate disturbance tolerant • Susceptible to fragmentation and habitat loss • Susceptible to road mortality • Requires high connectivity for dispersal	(COSEWIC, 2009; Harding and Holman, 1992)
Spring peeper	2	• 0% of sites	• Small bodied (2-2.5 cm) • Marshy woods and lowlands close to pools • Prefer fishless ponds	• Breed April – June • Eggs hatch within 6-12 days • Metamorphosis within 45-90 days • Dispersal rates unknown, likely ~ 1 km • Hibernate in terrestrial habitat • Life span 3-4 years	• Disturbance intolerant • Require high water quality • Susceptible to Cl^- (>125 mg/L) • Susceptible to road mortality and fragmentation	(Collins and Russell, 2009b; Harding and Holman, 1992; MMP Amphibians, 2009)
Western chorus frog	2	• 5% of sites • Abundance: • Min – 0 • Max – 1 • Median - 1	• Small bodied (2-4 cm) • Grasslands and forests close to breeding sites • Shallow, temporary permanent waterbodies for breeding • Predator free habitats	• Breed: March – May • Eggs hatch within two weeks • Metamorphosis within 30-90 days • Small dispersal rate (<500 m) • Hibernate in terrestrial habitat • Life span 1-3 years	• Disturbance intolerant • Highly susceptible to habitat loss due to high site fidelity, primarily urbanization and agriculture	(COSEWIC, 2008; MMP Amphibians, 2009; Ethier et al., 2021)
Wood frog	2	• 0% of sites	• Medium bodied (3-8 cm) • Wet meadows, forests, and shallow pools • Temporary and permanent,	• Breed: April – May • Eggs hatch within 10-90 days • Metamorphosis within 2 months • Potentially high, though few studies	• Disturbance intolerant • Require high water quality • Susceptible to Cl^- (>175 mg/L) • Susceptible to road mortality and fragmentation	(Collins and Russell, 2009b; MMP Amphibians, 2009; Muths et al., 2005)

Species	Local rank	SWMF Detection	Size and habitat preference	Life history and ecology	Disturbance tolerance	References
			fish free ponds for breeding	• Hibernate in terrestrial habitats • Life span 4-5 years		

3.3.1 Local and landscape predictors of anuran richness in SWMFs

The average species richness of chorusing anurans per site was 1.6 SD ± 0.87. The starting models for backwards step-AICc included the following local-level variables were area of robust emergent vegetation, Cl^-, TSS and fish presence; or/and the following landscape-level variables: canopy cover, water cover and road area within 300 m (Table 3-2). See section 2.3 for a detailed description of why these variables were included. NH_3 was not included in this model since no levels were above the toxic envelope to anurans in the SWMFs (see Byram and Nickerson, 2012). The best subset of variables to predict anuran richness for both the local as well as the local and landscape models only included the extent of robust emergent vegetation at the pond which was positively correlated (Table 3-2; $\Delta AIC_c = 0.00$, $W_{AICC} = 0.77$, $r^2 = 0.50$) while the best subset for the landscape model included only the area of roads within the 300 m radius buffer around each SWMF which negatively correlated with richness (Table 3-2; $\Delta AIC_c = 0.00$, $W_{AICC} = 0.77$, $r^2 = 0.08$). The local model had the lowest AIC_c, which exceeded 2 ΔAIC_c (Table 3-3).

Table 3- 2. AIC_c table for the top models in the Step-wise AIC for local and landscape, local and landscape models of frog species richness within two AIC from the top model. Direction of estimate is labeled as **(+)** for a positive correlation and **(-)** for a negative correlation. Local and local and landscape models produced the same top models.

Model type	Model	K	AIC	AICc	ΔAICc	W_{AICC}	$W_{cum.}$	LL	r^2
Local / Local and landscape	**(+)** RobustVeg	4	53.50	41.53	0.00	0.77	0.77	-17.01	0.50
	(+) RobustVeg + **(-)** Cl	5	55.38	33.89	2.36	0.23	1.00	-16.61	0.52
Landscape	**(-)** Road	3	56.56	53.77	0.00	0.77	0.77	-23.13	0.08
	(-) Road + **(+)** Canopy	4	58.31	56.23	2.46	0.23	1.00	-22.78	0.12

Table 3- 3. AIC_c model selection table of only local, only landscape, and local and landscape variables following backwards step-wise selection using AIC_c. The local and landscape model was not included due to it being the same as the local only model (see Table 3-2).

Model	K	AICc	ΔAICc	W_{AICC}	$W_{cum.}$	LL	r^2
Local **(+)** RobustVeg	3	41.53	0.00	1.00	1.00	-14.64	0.61
Landscape **(-)** Road	3	53.77	12.24	0.00	1.00	-23.13	0.08

3.3.2 Community composition of anurans in SWMFs

The optimal NMDS ordination solution for the anuran community composition among 20 SWMFs had two axes, with a final stress of 0.045 after 20 iterations (Figure 3-2; Procrustes: rmse = 0.000059, max residual = 0.00014). The local variables (Figure 3-2B) reasonably correlated with the ordination included area of robust emergent vegetation, Cl^-, TSS, pond area, and surface water temperature (Table 3-3). The landscape variables (Figure 3-2C) included road area, canopy cover, impervious cover, and open water area, all within 300 m radius buffers of each wetland, since no r^2 criterion was assigned (Table 3-3). The community composition of anurans seems to diverge between SWMFs with and without fish detected, but not in a statistically significant manner (perMANOVA: p = 0.27).

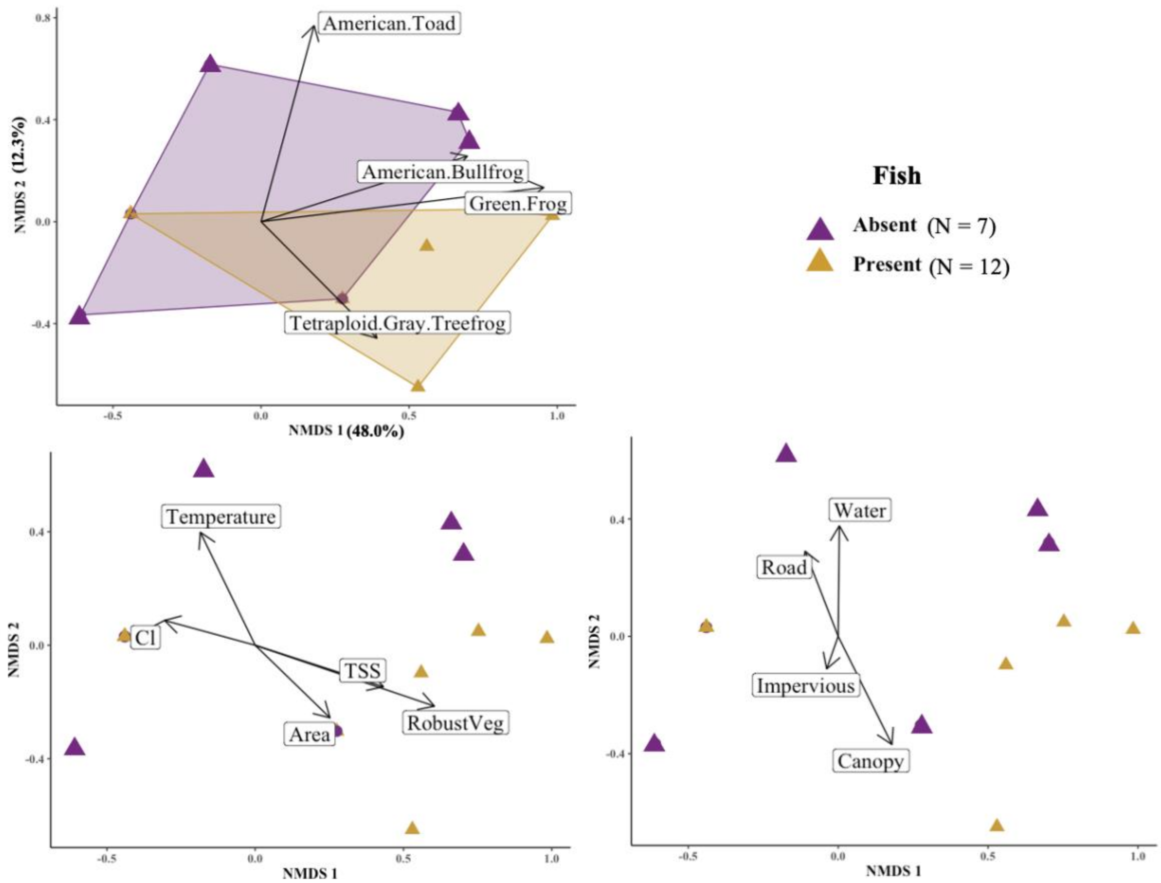


Figure 3- 2. Non-metric multidimensional scaling ordinations of anuran community composition in SWMFs, with SWMFs colour coded based on whether fish were detected with minnow traps (yellow) or not (purple). Panel A visualizes the correlation between the chorus-scores of anuran species with convex hulls highlighting the difference between ponds with and without fish detected in them. B depicts local environmental variables correlated ($r^2 > 0.1$) with at least one axis, and C depicts all landscape variables within 300 m of the SWMF perimeter as vectors. Points with the same species (e.g., one green frog) overlap in ordination space at -0.5, 0.0 (n = 9).

Table 3- 4. The NMDS scores, r^2 and p-values of all species and covariate vectors in relation to axes 1 and 2 of Figure 3-2. Environmental variables reasonably correlated ($r^2 > 0.2$) with the axes are bolded.

	Vector	Vector type	MDS1	MDS2	r^2	P-value
Species vectors	**American.Toad**	Species	0.22652	0.95740	**0.6238**	0.002
	American. Bullfrog	Species	0.93822	0.34603	**0.5511**	0.004
	Tetraploid.Gray.Treefrog	Species	0.64981	-0.76009	**0.3619**	0.040
	Green.Frog	Species	0.99027	0.13914	**0.9297**	0.001
Local-level vectors	Area	Local	0.70212	-0.71206	0.1290	0.325
	RobustVeg	Local	0.94231	-0.33473	**0.4114**	0.012
	Cl	Local	-0.96144	0.27500	0.1010	0.831
	Temperature	Local	-0.42148	0.90684	0.1926	0.183
	TSS	Local	0.94725	-0.32049	**0.2085**	0.154
Landscape-level vectors	Canopy cover	Landscape	0.43752	-0.89921	0.1682	0.225
	Impervious cover	Landscape	-0.32677	0.94511	0.0138	0.891
	Road density	Landscape	-0.35525	0.93477	0.0965	0.448
	Water area	Landscape	0.01025	0.99995	0.1421	0.280

3.3.3 Comparing anurans in SWMFs and natural wetlands

The SWMF and natural wetland ordination had an optimal NMDS solution of three dimensions with a stress of 0.054 after 40 iterations (Figure 3-3; Procrustes: rmse = 0.000080, max residual = 0.00038). SWMFs cluster together more tightly, whereas natural wetlands are more broadly spread across ordination space (Figure 3-3), yielding a statistically significant difference in community composition (perMANOVA: $p < 0.001$). A visual evaluation of the ordination revealed that natural wetlands are highly variable and SWMFs are more homogenous with less beta diversity (Figure 3-3). Spring peepers, tetraploid gray treefrogs, and wood frogs chorusing index scores were correlated negatively with axis 1 and were associated with natural wetlands, while northern leopard frogs and western chorus frogs had chorusing index scores negatively correlated with axes 2 and 3, though they were still more abundant in natural sites than SWMFs. Green frogs and American bullfrogs were the only two species that were associated with SWMFs. Their chorusing index scores were positively correlated with axis one, two and three (See Appendix 5 for NMDS scores).

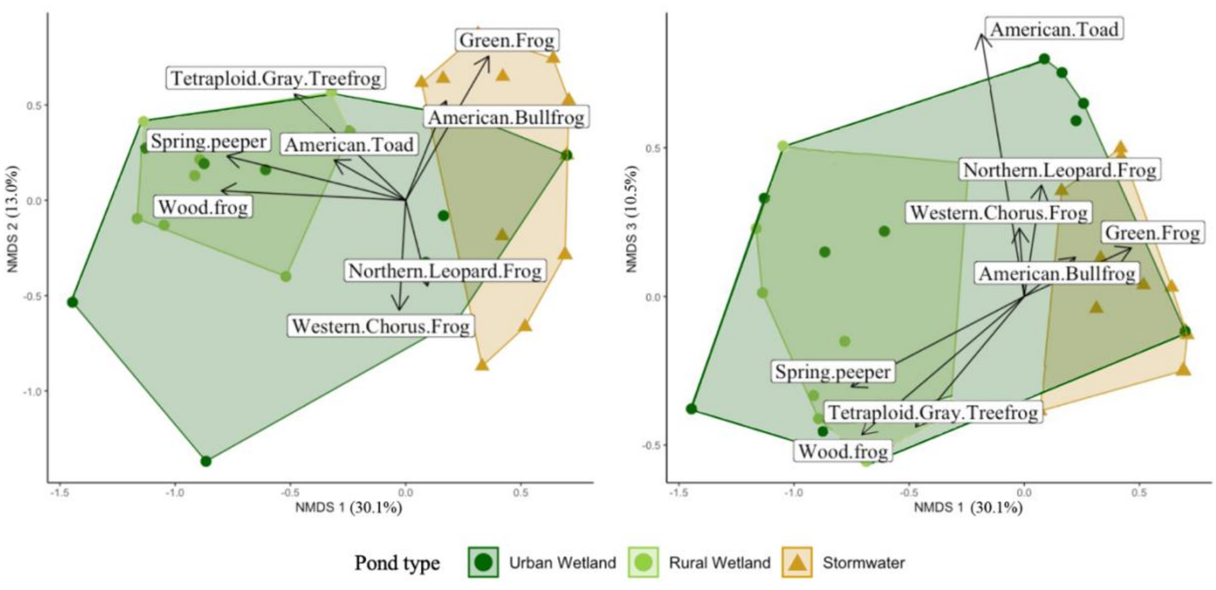


Figure 3- 3. Non-metric multidimensional scaling ordinating with three dimensions (**(A)** axis 1 and 2 **(B)** axis 1 and 3) of anuran abundance in 40 ponds. Pond type is visualized by shapes (SWMF = triangle; Natural wetland = circle) and colours (SWMF = yellow; Rural wetland = light green; urban wetland = dark green) and species were overlayed as vectors, providing they had an $r^2 > 0.1$ on at least one axis. NMDS scores were recorded in Appendix 6-5.

Lastly, I observed a significant difference in anuran species richness between natural wetlands and SWMFs (permutational ANOVA $F_{[1\text{-}38]} = 30.12$, p = 0.00001). The species richness was more variable amongst the natural wetlands compared to SWMFs (Fig. 3-4). Interestingly, it does seem as if anuran richness tended to be higher in natural wetlands classified as rural than in natural wetlands classified as urban, though I did not test for this due to the smaller sample size of 10 wetlands in each landcover type (Fig 3-4).

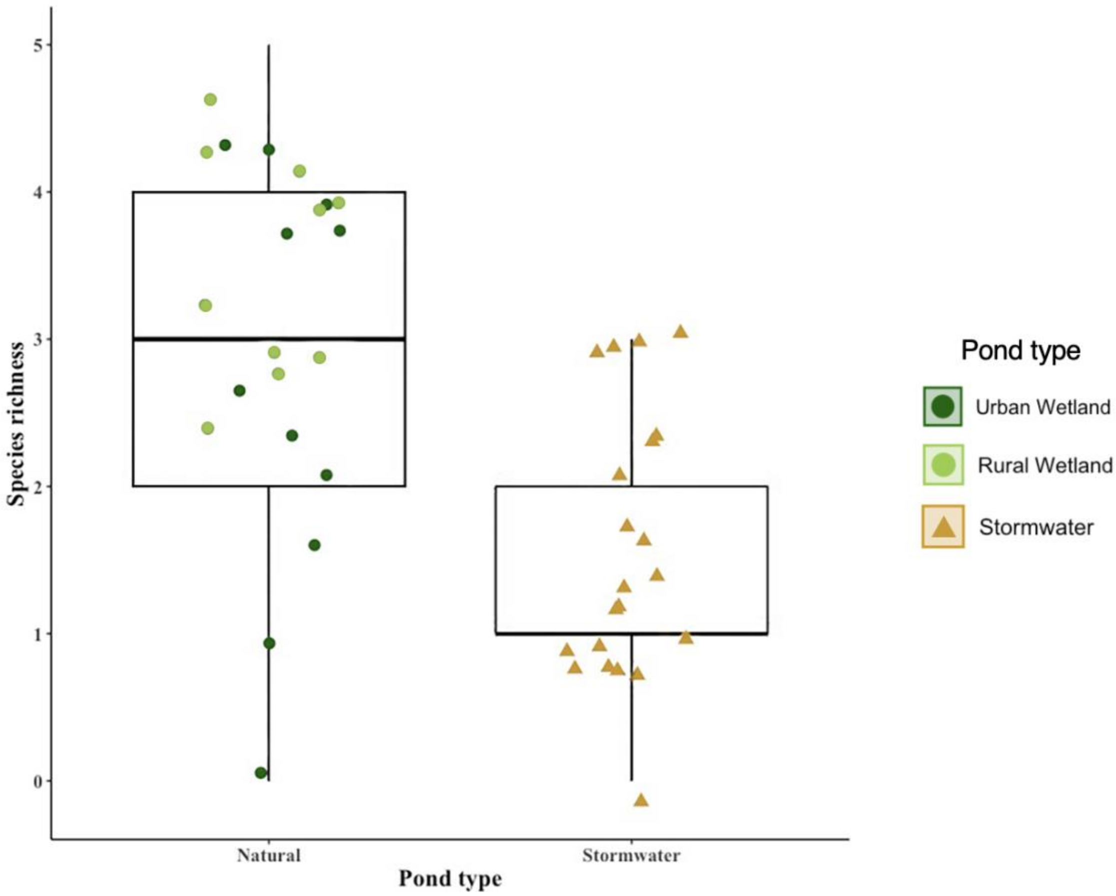


Figure 3- 4. Boxplots visualizing average species richness of anurans in each of three different pond types. Pond type is visualized by shapes (SWMF = triangle; natural wetland = circle) and colours (SWMF = yellow; rural wetland = light green; urban wetland = dark green)

3.4 Discussion

I evaluated relationships between anuran richness and local and landscape variables in stormwater management facilities (SWMFs) and in describing trends in anuran community composition among SWMFs in the City of Brampton, Ontario. Local variables were better predictors of anuran species richness and were more strongly correlated with variation in anuran community composition than were land covers in the surrounding 300 m radius buffers. These results are surprising considering previous literature which suggests both local and landscape variables are important determinants of anuran occurrence in urban wetlands (Almeida-Gomes et al., 2016; Hamer et al., 2008; Hamer and Parris, 2011). Of these local variables, the area of robust emergent vegetation and to a degree the concentration of Cl^- were the best predictors, whereas fish presence or absence did not appear as important. This is surprising also, as many authors reported that fish had a negative influence on anurans (Ficetola and de Bernardi, 2004; Hamer et al., 2008; Hamer and Parris, 2011; McCarthy and Lathrop, 2011). This is likely due to fish and anurans being sensitive to the same contaminants and pond design, or due to native fish increasing nutrient availability for anurans (Vanni et al., 2006).

Landscape drivers, namely area of road and canopy, did not improve the model for species richness when combined with local variables, and likely only influences variables such as Cl^- and robust emergent vegetation which are better predictors. The unexpectedly weak relationship between anuran richness and landscape-level factors like road density, canopy cover, or the cover of impervious or open water surfaces could be attributed to the high degree of urbanization in my study system compared to the much less urbanized control sites detailed by Hamer et al. (2008). My study also utilized a slightly smaller buffer size (300 m) than the average of 500 m discussed by Clevenot et al. (2018). I chose this to reflect strong effects of local cover within urban areas (Semlitsch and Bodie, 2003), and find it unlikely that a larger buffer would increase the predictive capacity of landscape-level variables in predicting anuran richness. However, I acknowledge that it may have been an insufficient size to capture dispersal limitation in anurans, as some species are able to disperse up to 10 km (Hamer et al., 2008).

The gradient of community composition and species richness, where rural wetlands are the most pristine with the highest amount of L-rank 2 species and species richness, SWMFs are the lowest quality with a few L-rank 2 species and lowest species richness and urban wetlands are highly variable, suggesting a highly variable gradient of environmental characteristics, although this wasn't measured. These findings suggest that certain high quality SWMFs can have higher anuran diversity and more L-rank 2 species than some lower quality urban wetlands highlighting the consideration of legally considering these ponds wildlife habitat. This agrees with previous literature suggesting amphibians and other taxa such as macroinvertebrates can have comparable diversity in SWMFs compared to urban wetlands, and as in Brand and Snodgrass, (2010) and Hassall and Anderson, (2015), the variability is likely determined by environmental drivers.

3.4.1 Do local and/or landscape variables effect anurans in SWMFs?

Most of the six species found within SWMFs are large-bodied anurans with a high tolerance for poor water quality and other stressors associated with urbanization (Table 3-1). From this set of species, four are currently ranked as of regional conservation concern (L-Rank 2: American bullfrog, tetraploid gray treefrog, western chorus frog, and L-Rank 3: northern leopard frog) while two are listed as least concern (L-Rank 4: green frog and American toad) (TRCA, 2017). Although four of these species are of regional concern in the GTA, none are currently listed as threatened on the IUCN red list (IUCN, 2021). These finding suggest that

SWMFs are capable of holding high priority species that are of regional conservation concern, however their low abundances and relatively limited richness reveal that SWMFs are not providing excellent habitat quality to the same diversity of anurans as natural wetlands.

The extent of robust emergent vegetation growing at a SWMF was the strongest correlate of anuran species richness and with the abundance of every species apart from the American toad. This finding aligns with previous studies which have highlighted the importance of naturalizing SWMFs with aquatic vegetation for anurans (e.g., McCarthy and Lathrop, 2011). Aquatic vegetation, such as cattails, are vital for the survival and reproduction of anurans as it provides a food source, habitat refuge from predators, and oviposition sites (Harding and Holman, 1992; McCarthy and Lathrop, 2011). However, I did not differentiate among plant species in mapping the extent of vegetation cover, and robust emergent species can include both native and non-native plants. Anecdotally, the vast majority of robust emergent vegetation was from the genus *Typha*, which might include the native species *T. latifolia* (broad-leaf cattail) and the invasive species *T. angustifolia* (narrow-leaf cattail) as well as their hybrid (T. x glauca). Other common robust emergent plants in my study included European *Phragmites australis.* Invasive populations of *T. angustifolia* L. and *P. australis* may affect anurans differently than native robust cattail *T. latfolia* L. Although invasive plant species should be avoided when naturalizing or enhancing SWMFs, the need to manage invasive plants to maximize habitat value in SWMFs is unclear. Future studies should evaluate which plant species are most important to anurans in SWMFs and whether the evolutionary history and nativity status of robust emergent vegetation matters to anuran diversity or not.

Community composition of green frogs, American bullfrogs and tetraploid gray treefrogs was also negatively correlated with Cl^-. High levels of Cl^- from road salts are widely known to impair reproduction in anurans (Brand and Snodgrass, 2010; Brown et al., 2012; Collins and Russell, 2009; Gallagher et al., 2014), and can negatively impact plant density (Simmons, 2012). Concentrations of Cl^- in SWMFs averaged at 1,100 mg L^- SD ± 757 with several ponds exceeding 3,000 mg L^-. In mesocosm experiments, Brown et al. (2012), found median Cl^- concentrations for egg mortality in green frogs at 2479 mg L^- and copes gray tree frog, which is in a species complex with tetraploid gray tree frogs and therefore may have similar tolerances, at 1855 mg L^-. Similarly, Matlaga et al. (2014), found American bullfrog tadpole tolerances to reach up to 3926 mg L^-. The high tolerance of these species, which still needs to be quantified for tetraploid gray tree frogs, may suggest that Cl^- in SWMFs directly affects anurans above their specific threshold, and indirectly affects them at similar or lower levels through the reduction of emergent vegetation.

Green frogs, American bullfrogs and to a lesser degree, tetraploid gray treefrogs were the most abundant species detected at SWMFs (Figure 3-2). Green frogs and bullfrogs were somewhat surprisingly positively associated with total suspended solids of the water, though this might be because larger SWMFs tended also to have higher suspended solids.

Also surprisingly, these three anuran species were generally more abundant in SWMFs that contained fish, which contradicts expectations from the published literature (e.g., Ficetola and de Bernardi, 2004; Hamer et al., 2008; Hamer and Parris, 2011; McCarthy and Lathrop, 2011). This discrepancy is likely attributable to my counting all fish, even if they are not known to prey on anurans. Indeed, all the fish I detected except creek chubs, located only at one of the SWMFs, were not considered predatory on adult anurans (Table 2-1). The rest of the fish were all opportunistic benthivores, so if predation was occurring it would be during the egg stage of anurans, which may not be as strong of an effect on anuran population persistence (Davenport et al., 2013). Still, it is interesting that these three species were generally found in higher abundances where fish were present, particularly the tetraploid gray treefrog which is known to avoid ponds with fish (Binckley and Resetarits, 2008; Resetarits and Wilbur, 1989; Vonesh et al., 2009). Instead, I interpret these results to reveal that fish and anurans using SWMFs are constrained by similar environmental variables. Green frogs and American bullfrogs both overwinter as fully aquatic juveniles, therefore their life histories more closely resemble those of fish (Harding and Holman, 1992). I hypothesize that one important factor constraining the abundance and distribution of fish, green frogs and American bullfrogs could be dissolved oxygen levels in SWMFs under winter ice (Huang et al., 2021). Future research on fish and anuran populations in SWMFs should examine the availability of dissolved oxygen in greater detail, particularly during winter. However, presence of fish in some sites, with high Cl^-, relate to lower compositions of anurans. These results could be affected by sites with high Cl^- are indicative of the presence of extreme generalist fish such as fathead minnows or goldfish which would require sub optimal conditions for survival (Holm et al., 2009). It is likely that sites with more rare species of fish, which require more optimal conditions, are correlated with sites with higher anuran composition, due to their sensitivity to contaminants.

To a lesser extent, green frogs, bullfrogs, and gray treefrogs were also more abundant in SWMFs surrounded by higher canopy cover and gray treefrogs in particular, were more abundant in SWMFs surrounded by lower road density. These results support previous findings that suggest that, in areas where local conditions support anuran populations, connectivity for dispersal in an important landscape variable (Clevenot et al., 2018a; Hutto and Barrett, 2021).

American toads are characterised in the literature as ubiquitous anurans with broad niche requirements (Harding and Holman, 1992; TRCA, 2015), yet they were surprisingly rare in my study, being detected in only 25% of the SWMFs. They tended to occur in sites with less robust emergent vegetation and higher water temperatures, but given their anticipated ubiquity and high tolerance range I expected they would be more common. The cause of their apparent rarity may be their lower detectability, though I did not model occupancy and detection probabilities in my study. American toads call intermittently, depending on weather conditions and traffic noise, and a three minute survey time in urban areas with nearby roads may not have been adequate (P. Prior, personal communication, March 1, 2022). Another possible reason is that American toads can exhibit scramble mating behaviour, where males actively search for females instead of attracting via calling (Vargas-Salinas et al., 2014). I hypothesize that due to the elevated ambient noise typical of urban areas, American toads may adopt this scramble mating behavior over calling more frequently. However, several studies have identified that traffic noise does not affect the call behaviour of American toads (Cunnington and Fahrig, 2010, 2012; Vargas-Salinas et al., 2014). Future research investigating anurans with American toads in the region should employ a variety of sampling methods, including egg detections, sweep netting, fish traps (deployed in the spring), and transect walk surveys to directly observe American toads (discussed in a review by Hamer et al., 2008).

3.4.2 Should SWMFs be protected as wildlife habitat?

The anuran communities in stormwater management facilities were a nested subset of the anurans occurring in natural wetlands in the same region, comprising both lower alpha (richness) and beta diversity than natural wetlands. As such, SWMFs should not be considered adequate habitat compensation for the destruction of natural wetlands in urban areas, as they cannot replace the full biodiversity value of natural wetlands. However, they are clearly providing ecological habitat to a similar degree as natural urban wetlands, which are managed as legal habitat, so do they provide sufficient value to warrant managing them as part of the natural heritage portfolio of a city?

The anuran communities in SWMFs in Brampton, Ontario were primarily characterized by green frogs and American bullfrogs, which are both relatively tolerant, generalist anuran species. Conversely, the natural wetlands contained two species which I never detected in SWMFs: the spring peeper and the wood frog. Both of these are categorized as anurans of regional conservation concern (L-rank: 2) and they were more common in the natural wetlands classified as rural, along with gray treefrogs (Figure 3-4). These findings suggest that landcover composition, which is how rural and urban sites were delineated, is likely an important factor effecting anurans, and I would see a stronger effect on SWMF composition if they spanned the same gradient from urban to rural.

Spring peepers and wood frogs are both highly sensitive to even moderate levels of Cl^-, and are both unlikely to persist in water with a concentration over 175 mg/L (Collins and Russell, 2009). On average, no SWMF sites fell within acceptable ranges for spring peepers and wood frogs. I also found that Cl^- was highest during the spring months, after the snowmelt and dropped come the summer. Given that both spring peepers and wood frogs begin breeding in March, they will potentially be subject to the highest concentration of this contaminant (MMP Amphibians, 2009).

However, it bears noting that American bullfrogs – also an L-rank 2 of regional conservation concern – were exclusively detected in SWMFs. This reveals that the SWMFs are contributing to regional diversity, even if they do not support the full complement of species typical in natural wetlands. The occurrence of bullfrogs in SWMFs presents two contrasting perspectives when considering species protection. One of the potential reasons for the exclusive occurrence of bullfrogs in SWMFs is that these species require deep permanent waterbodies with abundant vegetation for both breeding and hibernating through the winter (Harding and Holman, 1992). If some of the more naturalized, higher quality SWMFs are able to offer this habitat to bullfrogs, then they may prefer these sites over natural wetlands which are generally shallower and subject to seasonal drawdown. Alternatively, bullfrogs from the pet trade may be being released into SWMFs, due to their close proximity to suburban households. This raises questions regarding the origin of the bullfrogs I observed as the introduction of any organism to an ecosystem by humans can have drastic effects, even if the species is native to the region. In western North America, where American bullfrogs are not native, they are problematic and considered highly invasive (Ficetola et al., 2010). Yet locally, they are native and even ranked L-Rank 2 to reflect their local status as of conservation concern. Regardless of their origin and the role of humans in their introduction to SWMFs, clearly SWMFs are offering a habitat to anurans in an urban area. The anuran communities in SWMFs more closely resembled the anuran communities in natural wetlands categorized as urban, rather than those categorized as rural by the Toronto Region Conservation Authority. Although SWMFs differ from rural natural wetlands, their overlap with urban wetlands suggests they can be of importance to wildlife and perhaps should be managed more actively by Conservation Authorities to protect and promote regional biodiversity. Yet, I conclude that they must be considered distinct from natural habitats and certainly not be considered compensation for any natural wetland loss. I only looked at the presence of anurans in SWMFs for a single field season. Yet the SWMFs have many characteristics typical of sink habitats, where fragmentation, and harsh living conditions can cause extinctions when there is no influx from higher quality habitat (Clevenot et al., 2018; Furrer and Pasinelli, 2016). Important questions to address before such a claim could be made include whether the populations, I detect are viable, by monitoring the sites over an extended period, how the fitness of anurans in SWMFs compares to those in urban natural wetlands (i.e., quantify if they are successfully reproducing), and whether SWMFs are source habitat, sink habitat, or even ecological traps. A greater understanding of the dispersal dynamics and metacommunity dynamics among SWMFs and natural wetlands in urban areas is needed.

3.4.3 Conclusion and areas of future work

My research constitutes the most intensive dataset on anurans within southern Ontario SWMFs to date, and to my knowledge is the first time anuran communities in SWMFs have been compared to anuran communities in surrounding natural wetlands in this area. I found that SWMF anuran community composition and species richness were highly correlated to local variables, specifically richness increased in SWMFs with greater extent of robust emergent vegetation and decreased in SWMFs with higher concentrations of Cl^-. These findings suggest that to increase the suitability of SWMFs to anurans, attention should be given to naturalizing the SWMFs by increasing vegetation cover, particularly of native plant species. Stormwater management facilities were capable of supporting anuran richness and composition similar to that of naturally occurring urban wetlands, and they merit further research to determine if they should be managed as legal natural habitats for wildlife and biodiversity in urban areas.

SWMFs had the lowest species richness and least abundant L-rank 2 species, compared to natural wetlands, although there was overlap with natural wetlands classified as ‘urban’ compared to those classified as ‘rural’, suggesting the factors responsible for the distinction between anuran communities in SWMFs and natural wetlands are likely stressors and isolation imposed by the urban landscape context, rather than any morphological or hydrologic features specific to SWMFs that distinguish them from natural wetlands more generally. I advise additional research be done to quantify the mechanisms that drive the differences in anuran community that I observed, potentially using larger buffer sizes and nested buffers to capture more of the landscape context that might fragment or otherwise isolate SWMFs and natural wetlands in urban environments. Discovering these mechanisms will not only be important to the future of SWMFs but also to naturally occurring wetlands in a highly urbanized area.

A practical question that their possible designation as legal wildlife habitat raises is how they should be managed in the future as SWMFs. For example, SWMFs are typically dredged to extend their lifespan as they fill with sediment and organic matter over time. The SWMFs in my study have yet to be dredged since their creation over a decade ago and several are nearing the age when dredging may become necessary to maintain their storm water management function. This presents an interesting opportunity to quantify how dredging will affect anuran communities. Dredging will pose a major risk to amphibians and can be a factor to SWMFs becoming sink habitats, but the ability of anurans and other wildlife to recolonize SWMFs after dredging is severely lacking in the literature (Clevenot et al., 2018a). Continuing research and monitoring of anuran communities in the ponds will strengthen our understanding of SWMFs, and will start to determine how or even whether they should be included in local or regional planning to enhance urban biodiversity.

4.0 Synthesis and conclusion

Stormwater management facilities (SWMF) are key components of urban landscapes, as they provide critical flood prevention and runoff control (Dhalla, 2012). Although protection from flooding is the main purpose of SWMFs, their high abundance and indirect replacement of wetlands (Birch et al., 2022) in urban areas means they are often inhabited by aquatic wildlife and provide ecological habitat (Oertli and Parris, 2019). Despite the use of SWMFs by aquatic organisms, they are often not considered legal habitat and are therefore not included in regional biodiversity monitoring due to their classification as infrastructure rather than natural heritage features. However, since SWMFs may provide some of the only aquatic refuge habitat in highly urbanized areas, the habitat suitability of these ponds requires ecological assessment. It is possible that they should be classified as novel ecosystems (i.e., human-made) (Hobbs et al., 2006).

The habitat suitability of SWMFs is often debated (see reviews in Clevenot et al., 2018 and Oertli and Parris, 2019). Studies have investigated the habitat value of SWMFs using bioindicators, but report conflicting results. For example, studies in Canada using macroinvertebrates as bioindicators found that SWMFs are capable of supporting biodiversity comparable to unmanaged natural wetlands (Hassall and Anderson, 2015b; Perron and Pick, 2020), while studies in France and Australia using anurans (true frogs and toads) as bioindicators report conflicting results; some studies suggest that SWMFs are toxic ecological traps (France: Conan et al., 2022; Australia: Sievers et al., 2018, 2019), while others suggest they are suitable habitats, but dependent on surrounding environmental variables (Australia: Hamer et al., 2012; Hamer and Parris, 2011; USA: McCarthy and Lathrop, 2011). Studies using fish as bioindicators are rarer (Oertli and Parris, 2019), and to my knowledge, only two studies have determined that the presence of invasive species (Taiwan: Huang et al., 2021) and poor water quality (Canada: Bishop et al., 2000) lead to lower native fish diversity. My research advances our knowledge about the habitat suitability of SWMFs for both anurans and fish in the southern Ontario region of Canada.

4.1 Summary of thesis

In this thesis, I have demonstrated the use of SWMFs by fish and anurans in southern Ontario, Canada (Figure 4-1). Fish richness was correlated with local variables (e.g., extent of shrub vegetation and chloride) in SWMFs, however I highlight the likely importance of dissolved oxygen and dispersal for future studies. Anuran composition and species richness was strongly related to the increases in area of robust emergent vegetation and the composition of serval species was related to reductions in concentration of chloride ions. Stormwater management facilities with high area of emergent vegetation were able to have comparable composition and higher species richness when compared to urban wetlands.

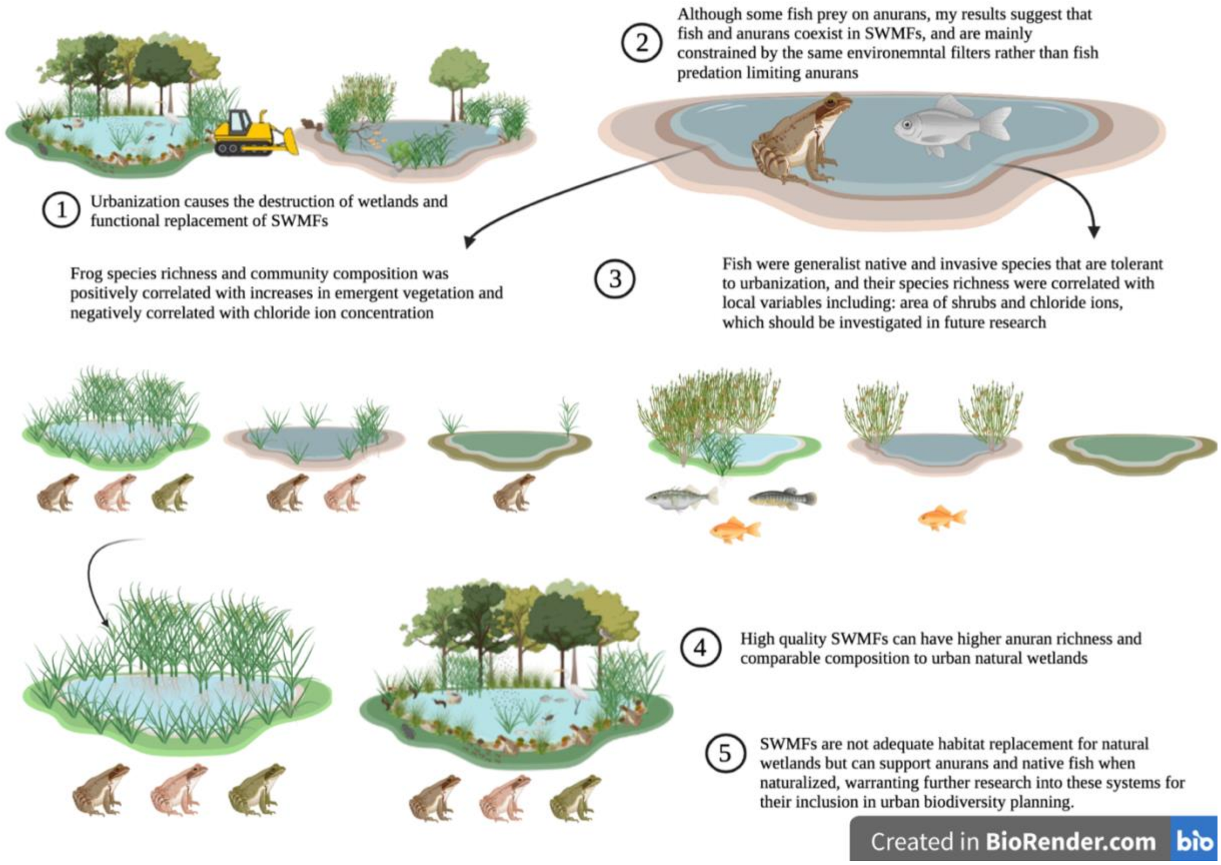


Figure 4- 1. Conceptual figure outlining the key findings of SWMF occupancy of fish and anurans, depicting the drivers of species richness and composition as well as contrasting anurans in SWMFs and natural wetlands. Image was created using BioRender.

4.2 Research findings

The purpose of my research was to 1) evaluate the effects of local and landscape-level environmental drivers of urban SWMFs on the community dynamics of fish and anurans, and 2) contrast the community dynamics of anurans in these novel ecosystems to natural wetlands in the surrounding area. Quantifying these results helps elucidate if SWMFs should be included in regional planning for biodiversity conservation. In chapter one I provided a literature review of SWMFS, as well as the current literature surrounding the use of SWMF and urban areas by anurans and fish. I also discussed the ecology of both fish and anurans and how they interact together in aquatic systems.

In chapter two, I explored the prevalence and diversity of fish in SWMFs and quantified the local and landscape variables associated with patterns in species richness and catch per unit effort (CPUE) of fish. I found that fish richness was low, and that community composition was primarily composed of habitat generalist species that are tolerant to urbanization. I concluded that several water quality and vegetation parameters should be investigated in future studies to better analyze their effects on fish diversity in urban aquatic habitats. I also suggest that there are likely additional constraints, such as winter minimum dissolved oxygen concentrations or species dispersal limitations that influence fish distributions and abundance in SWMFs.

In chapter three, my objectives were twofold: first, I quantified local and landscape environmental drivers to determine if water quality and vegetation type or landcover (impervious, canopy, road and water cover) could predict anuran species richness in SWMFs or patterns in anuran community composition. Second, I compared community composition and species richness in SWMFs to surrounding natural wetlands to determine if SWMFs support similar anuran communities to natural wetlands and to inform future biodiversity planning by the Toronto Region Conservation Authority (TRCA).

I found that anurans were more sensitive to local variables, where species richness and composition of several species were positively correlated with robust emergent vegetation (e.g., cattails) and several species were negatively correlated with chloride ion concentration. I concluded that naturalizing SWMFs by increasing the presence of robust emergent vegetation and efforts to improve water quality, particularly by reducing chloride ion concentrations, might increase the diversity of anurans using SWMFs. Increasing the extent of robust emergent vegetation might also improve water quality (Ross et al., 2018), such that these factors may interact in their influence on anurans.

Natural wetlands supported distinct anuran community composition compared to SWMFs, including more anuran species of regional concern and higher species richness than SWMFs. Natural wetlands had higher variability both in composition (beta diversity) and species richness, which was explained by the ponds being in an urban or rural landscape. I concluded that SWMFs can have comparable anuran community composition and higher species richness than some urban natural wetlands if they are naturalized, which suggests that there is the potential for higher quality SWMFs to be implemented into regional conservation planning.

4.3 Anuran and fish community dynamics

Although only one species of fish recorded in this study is known as a predatory species, which may feed on metamorphosizing and smaller anurans, predation from fish could be influencing the community dynamics of SWMFs. For example, even the smaller, generalist-feeding fish found in this study can be voracious predators of anuran eggs and newly hatched tadpoles (Davenport et al., 2013). This could leave anurans vulnerable to predation prior to reaching an adequate size, particularly in ponds that are dominated by high abundances of fathead minnows and goldfish, as in this study. However, it is interesting that even when fish CPUE and diversity were relatively higher, anuran diversity and abundance did not necessarily decline, and most species were generally associated with the presence of fish (Figure 3-2). Only western chorus frogs and northern leopard frogs were completely absent from ponds with fish, which is not surprising considering chorus frogs tend to breed in vernal pools where fish absenter rare (COSEWIC, 2008). Wood frogs were not detected in SWMFs, however their presence in the natural wetlands represents an interesting avenue for future research. Are wood frogs excluded by the presence of fish? Research from Albertan lakes found that wood frog populations have been experiencing declines due to increases in stickleback and fathead minnows (Eaton et al., 2005). Although my study was predominantly focused on the effects of anthropogenic activity on the occurrence and richness of anurans and fish, future research should more thoroughly investigate the effects of fish predation on anurans in SWMFs (e.g., Davenport et al., 2013), as there are likely trophic interactions to consider.

Although there is likely an effect of fish predation on juvenile anurans in SWMFs, my results indicate that some species of anurans are found in higher abundance at sites with fish. As detailed in section 3.4.1, green frogs, tetraploid gray treefrogs and American bullfrogs were all associated with sites where fish were present (Figure 3-2). I surmise that this association is more likely the result of shared environmental constraints (e.g., chloride levels, emergent vegetation or dissolved oxygen levels under winter ice) rather than any biological interaction between fish and these anuran species. Bullfrogs and green frogs may be particularly prone to poor water quality over winter since they overwinter as juveniles for up to three years (Harding and Holman, 1992). Consequently, both fish and tadpoles of green frogs and bullfrogs may experience stress from high biological or chemical oxygen demand under winter ice (Datry et al., 2004; Eaton et al., 2005). More surprising, tetraploid gray treefrog, which are well documented to be sensitive to fish predation (Binckley and Resetarits, 2008; Resetarits and Wilbur, 1989; Vonesh et al., 2009), were only present in SWMF ponds with fish. I speculate that this association reflect shared habitat preferences between gray treefrogs and fish (e.g., for SWMFs in landscapes with more canopy cover or with a greater cover of robust emergent vegetation) rather than any mutualism. However, it is still plausible that increases in rare species of fish can contribute to optimal nutrient cycling that may be beneficial for higher abundances of certain frog species (Vanni et al., 2006). My results indicate that future work should investigate the effects of fish on anurans to not only better understand predation, but to also gain a better understanding of what is truly affecting biodiversity within these systems.

4.4 Areas of future work

My research contributes to the urban ecology literature pertaining to the occurrence and diversity of anurans and fish in southern Ontario SWMFs, however future research is still required to gain a further understanding of these systems as habitat. Although I did not detect any species of concern, SWMFs are still being inhabited by several generalist fish species. However, my study only used minnow traps, limiting the detections to small-bodied species. Future research should expand on this study by increasing the variety of sampling gear (e.g., fyke nets, dip nets, electrofishing or environmental DNA) in order to get more reliable estimates of fish diversity, and to increase the probability of catching large and rare species (Perez et al., 2017). These methods can be employed in conjunction with a larger sample size in order to increase the detection rate at SWMFs to gain a better understanding of their response to environmental conditions. Since few fish were captured in my study, it was difficult to quantify the environmental variables that affected CPUE and species richness. Future research should examine the effects of emergent vegetation (mainly shrub), chloride ions, ammonia and surrounding water and canopy connectivity when investigating fish in SWMFs, as my study found these variables to show correlations with the fish that were detected. Finally, although I did not measure these variables, there is likely a strong effect of dissolved oxygen and dispersal for the occupancy of many fish in SWMFs, which needs to be quantified in any future study regarding these systems.

I found that anuran community composition and richness was strongly positively correlated with robust emergent vegetation in SWMFs, however, vegetation at the sites was only broadly categorized following MMP Amphibians (2009) and OWES (2013). Due to this, I did not differentiate between robust vegetation species from *Typha* and *Phragmites*. Future research should investigate what species of vegetation in SWMFs promotes that highest species richness and composition of anurans in order to give proper management advice for the planting of native plant species.

Clearly SWMFs are ecological habitats for several anuran and fish species, and my study is the precursor step to the inclusion of these systems as legal habitat (i.e., protected and monitored by a public agency). These novel ecosystems can support similar or increased biodiversity compared to natural urban wetlands, so if these natural heritage features are considered legal habitat, then should SWMFs be as well? My study alone is not adequate to change the legality of SWMF habitat classification and future studies need to address three critical issues before any legal implementation can occur, to avoid potential harm to biodiversity in the long runs. The first issue is dredging, where ponds are drained, and all of the sediment is removed in order to regulate the water quality of the pond (Clevenot et al., 2018). There is a large gap in the current literature on the effects of dredging on anurans, though the potential for deadly impacts has been noted (Clevenot et al., 2018; McCarthy and Lathrop, 2011b). Future studies should investigate the effect of dredging to determine if anurans can recolonize SWMFs after the removal of contaminants. If biodiversity is unable to recolonize SWMFs after dredging, then any ecological work towards species conservation would be undone and perhaps allocation of funding could be better spent elsewhere. The next issue is the effects that contaminants may have through biomagnification, where organisms accumulate and increase the concentration of contaminates in their system through the ingestion of prey (Coelho et al., 2018; Søberg et al., 2016). Anurans and fish were both able to occupy sites with relatively high contaminant levels (e.g., bioavailable nitrogen, phosphorus, and chloride ions), however the effect of biomagnification on the predators that would ingest these organisms could be drastic. Future research should investigate the role of biomagnification in food web dynamics of SWMFs to gain a better understanding of contaminants and biological interactions. Improving the diversity and abundance of wildlife using SWMFs may have unintended outcomes if biota mobilize contaminants or create ecological traps, in which case attracted more wildlife to these ponds might be the wrong approach (Drygiannaki et al., 2020; Sievers, Parris, et al., 2018). Finally, future research needs to closely examine the trade-offs between biodiversity services and ecosystem services of SWMFs. For example, SWMFs are generally designed with a steep slope in order to hold higher capacities of stormwater during heavy rainfall events, however a gentle slope will facilitate the growth of plants and likely lead higher diversity (Rooney et al., 2015). Although naturalizing SWMFs may be a promising route for the improvement of urban biodiversity, it should not come at the expense of the ecosystem function of these ponds. However, future research should explore the use of SWMFs in southern Ontario by other taxa (e.g., birds, invertebrates, microorganisms, reptiles, etc.) to determine the breadth of their habitat value and ensure that future management can sustain both biodiversity and humans.

4.5 Conclusion and implications

My results have important implications for the design and management of SWMFs. However, as of now, I do not believe that SWMFs should be managed as legal habitat for current planning and monitoring of biodiversity. My study is a precursor step to the inclusion of SWMFs as legal habitat, and I have found that they should not be ruled out, simply for the reason they are human made habitats (As in Hobbs et al., 2006). Nevertheless, the issues of dredging, the fate of the contaminants and the trade-offs between biodiversity and ecosystem services need to be addressed before any implementation can occur. I recommend that the TRCA and City of Brampton dredges the sites that I have surveyed in my study, and that it should be carried out in the fall after anurans have completed breeding. Not only will this provide an opportunity to examine the effects dredging has on SWMFs, but it is also a process that the general public often demanded during sampling since they had not been dredged for over 10 years, or in most cases ever. I also recommend that the implementation of gentle slopes should be required when designing SWMFs in order to facilitate more plant growth, provided it does not impair the ecosystem services of flood mitigation. If these issues are quantified and regional management and monitoring of SWMFs becomes more prolific, I recommend the development of novel techniques to monitor these sites. These are novel ecosystems, which presents novel assemblages and challenges (Hobbs et al., 2014), and as such these systems may not function the same as natural heritage systems. For example, in my study I employed monitoring techniques (e.g., minnow traps and auditory surveys) that align with standardized protocols in marsh systems, though certainly some SWMFs are more representative of lake systems and others of no natural system. I also compare SWMFs to reference natural wetlands, though this may be inappropriate since these are different systems, and it may be more suitable to use reference SWMFs in natural areas if applicable to the region in the study. Either way this is likely highly dependent on the objective of the study. If studies aim to address the biodiversity services that SWMFs provide to urban areas, as in the first chapter of my study, it may be more suitable to use a reference approach with SWMFs in natural landcover. However, when addressing if SWMFs should be considered legal habitat, it may be more appropriate to use reference natural heritage sites that are already managed and protected for their biodiversity services. In any case novel approaches to conservation for SWMFs will be critical to strengthen our understating of how these systems support biodiversity in urban areas.

Although I do not believe SWMFs should be considered legal habitat at this point, certain processes can still be applied to maximize their biodiversity and ecosystem services both to existing and new ponds. I recommend that the TRCA works with the City of Brampton to begin naturalizing more SWMFs by increasing the extent of robust emergent vegetation at both new and existing sites. Robust emergent vegetation is not just important for anurans but can also play an important role to improve water quality in SWMFs, e.g., the presence of cattail (*Typha*) was associated with reductions in chloride in constructed wetlands in serval studies (Guesdon et al., 2016; Jesus et al., 2014; Schück and Greger, 2022). Planting more emergent vegetation in existing ponds should be done in conjunction with dredging SWMFs in the fall season, when the water will be low, and the seeds can take and naturally germinate the following spring (Ross et al., 2018). This will also allow the vegetation to grow before high concentrations of contaminants are present in the system, which in turn can lead to more effective phytoremediation (Guesdon et al., 2016). Indeed, local characteristics show a stronger effect on the biodiversity in my study compared to landscape variables, and future SWMF design can implement strategies to improve local conditions even when in unfavourable landscape conditions (e.g., high impervious cover). Future SWMFs can be designed with a more gradual slope to mimic a natural wetland, which can help facilitate plant growth and allow emerging organisms to leave the water more readily (Ross et al., 2018). When practical, this can be combined with a larger pond area, which did not limit either anurans or fish in my study, to allow for more water holding capabilities of the SWMF, in order to not impair the flood mitigation services they provide. Finally planning for areas of connectivity either to other SWMFs or corridors to higher quality natural heritage sites will allow high quality SWMFs to act as refuge in urban areas as biota disperse through urban areas.

It is clear that the release of pets has led to high abundances of goldfish and potentially American bullfrogs in the SWMFs surveyed in my study. I recommend that the TRCA and City of Brampton begin initiatives for the education in the release of exotic pets into urban systems if they are not already in place. This could include signs posted around SWMFs that describe the dangers of releasing goldfish into these ponds or through social media platforms. I recommend the disconnectivity techniques to keep fish out stay in place, and to consider allocating funding towards the monitoring of goldfish in the neighbouring streams connected to the SWMFs. I also recommend further investigation of American bullfrogs within the jurisdiction to determine if they are naturally occurring in the area or if they too are released pets.

These steps need to be done in conjunction with an overall restoration of the idea of SWMFs. Even if these ponds are not considered legal habitat, they are clearly ecological habitat to many charismatic and several rare species such as tetraploid gray treefrogs and brook sticklebacks. If these ponds are held in low regard to their biodiversity values, then certainly they will be continued to be treated as such, acting as landfill sites for garbage or cesspools for invasive species. Identifying with SWMFs as habitat will be critical for the future stewardship of these systems reconnecting people and the landscape. I recommend that future sampling of biological organisms, using novel techniques, in SWMFs be carried out through community science programs led by municipal bodies (e.g., TRCA Citizen Science Volunteer Program). Community science programs have been effective in many ecological studies and has been increasingly seen as reliable and valuable (McKinley et al., 2017). This monitoring strategy will not only be beneficial for a compressive understanding of SWMF ecology but also provide strategies for the public to become more connected with these novel ecosystems.

The transition to a more sustainable SWMF design, that improves ecosystem and biodiversity services is an essential step to the future of conservation in urban areas. My findings are the precursor step to improving the sustainability of SWMFs, while prompting new questions as to if these highly abundant urban blue spaces should be considered protected habitat.

References

Abdi, R., and Endreny, T. (2019). A river temperature model to assist managers in identifying thermal pollution causes and solutions. *Water (Switzerland)*, *11*(5). https://doi.org/10.3390/w11051060

Alexander, K., Hettiarachchi, S., Ou, Y., and Sharma, A. (2019). Can integrated green spaces and storage facilities absorb the increased risk of flooding due to climate change in developed urban environments? *Journal of Hydrology*, *579*, 124201. https://doi.org/10.1016/J.JHYDROL.2019.124201

Alikhani, S., Nummi, P., and Ojala, A. (2021). Urban wetlands: A review on ecological and cultural values. In *Water (Switzerland)* (Vol. 13, Issue 22). MDPI. https://doi.org/10.3390/w13223301

Almeida-Gomes, M., Rocha, C. F. D., and Vieira, M. v. (2016). Local and Landscape Factors Driving the Structure of Tropical Anuran Communities: Do Ephemeral Ponds have a Nested Pattern? *Biotropica*, *48*(3), 365–372. https://doi.org/10.1111/BTP.12285

Anderson, M. J. (2017). Permutational Multivariate Analysis of Variance (PERMANOVA). *Wiley StatsRef: Statistics Reference Online*, 1–15. https://doi.org/10.1002/9781118445112.STAT07841

Ankley, G. T., and Villeneuve, D. L. (2006). The fathead minnow in aquatic toxicology: Past, present and The fathead minnow in aquatic toxicology: Past, present and future. https://doi.org/10.1016/j.aquatox.2006.01.018

Arnold, C. L., and Gibbons, C. J. (1996). Impervious Surface Coverage: The Emergence of a Key Environmental Indicator. *Journal of the American Planning Association*, *62*(2), 243–258. https://doi.org/10.1080/01944369608975688

Bateman, J. A. (2014). *Effects of stormwater ponds on calling amphibian communities in Monroe County, NY*.

Becker, G. (1983). *Fishes of Wisconsin*. University of Wisconsin press.

Beninde, J., Veith, M., and Hochkirch, A. (2015). Biodiversity in cities needs space: A meta-analysis of factors determining intra-urban biodiversity variation. In *Ecology Letters* (Vol. 18, Issue 6, pp. 581–592). Blackwell Publishing Ltd. https://doi.org/10.1111/ele.12427

Binckley, C. A., and Resetarits, W. J. (2008). Oviposition behavior partitions aquatic landscapes along predation and nutrient gradients. *Behavioral Ecology*, *19*(3), 552–557. https://doi.org/10.1093/BEHECO/ARM164

Birch, W. S., Drescher, M., Pittman, J., and Rooney, R. C. (2022). Trends and predictors of wetland conversion in urbanizing environments. *Journal of Environmental Management, 310*, 114723. https://doi.org/10.1016/j.jenvman.2022.114723

Bishop, C. A., Struger, J., Barton, D. R., Shirose, L. J., Dunn, L., Lang, A. L., and Shepherd, D. (2000). Contamination and wildlife communities in stormwater detention ponds in Guelph and the Greater Toronto Area, Ontario, 1997 and 1998. Part I - Wildlife communities. *Water Quality Research Journal of Canada*, *35*(3), 399–435. https://doi.org/10.2166/wqrj.2000.026

Bishop, C. A., Struger, J., Shirose, L. J., Dunn, L., and Campbell, G. D. (2000). Contamination and wildlife communities in stormwater detention ponds in Guelph and the Greater Toronto Area, Ontario, 1997 and 1998. Part II - Contamination and biological effects of contamination. *Water Quality Research Journal of Canada*, *35*(3), 437–474. https://doi.org/10.2166/wqrj.2000.027

Blackwell, B. F., Schafer, L. M., Helon, D. A., and Linnell, M. A. (2008). Bird use of stormwater-management ponds: Decreasing avian attractants on airports. *Landscape and Urban Planning, 86*(2), 162–170. https://doi.org/10.1016/j.landurbplan.2008.02.004

Bliss, S. M., Midwood, J. D., Stamplecoskie, K. M., and Cooke, S. J. (2015). Seasonal movements and residency of small-bodied fish in a north temperate urban watershed demonstrate connectivity between a stream and stormwater drain. *Hydrobiologia, 742*(1), 327–338. https://doi.org/10.1007/s10750-014-1996-0

Bobbink, R., Beltman, B., Verhoeven, J. T. A., and Whigham, D. F. (Eds.). (2006). *Wetlands: Functioning, Biodiversity Conservation, and Restoration*. *191*. https://doi.org/10.1007/978-3-540-33189-6

Boeuf, G., and Payan, P. (2001). How should salinity influence fish growth? *Comparative Biochemistry and Pysiology*, *130*(4), 411–423.

Boone, M. D., and Bridges, C. M. (2003). *Effects of pesticides on amphibian populations*. Smithsonian Institution.

Boone, M. D., Semlitsch, R. D., and Mosby, C. (2008). Suitability of Golf Course Ponds for Amphibian Metamorphosis When Bullfrogs Are Removed. *Conservation Biology, 22*(1), 172–179. https://doi.org/10.1111/J.1523-1739.2007.00817.X

Brand, A. B., and Snodgrass, J. W. (2010). Value of artificial habitats for amphibian reproduction in altered landscapes: Contributed paper. *Conservation Biology, 24*(1), 295–301. https://doi.org/10.1111/j.1523-1739.2009.01301.x

Brix, K. v., Keithly, J., Santore, R. C., DeForest, D. K., and Tobiason, S. (2010). Ecological risk assessment of zinc from stormwater runoff to an aquatic ecosystem. *Science of the Total Environment, 408*(8), 1824–1832. https://doi.org/10.1016/j.scitotenv.2009.12.004

Brown, M. G., Dobbs, E. K., Snodgrass, J. W., and Ownby, D. R. (2012). Ameliorative effects of sodium chloride on acute copper toxicity among Cope's gray treefrog (*Hyla chrysoscelis*) and green frog (*Rana clamitans*) embryos. *Environmental Toxicology and Chemistry*, *31*(4), 836–842. https://doi.org/10.1002/etc.1751

Burnham, K. P., and Anderson, D. R. (1998). *Model selection and inference*. Springer.

Byram, J.K. and Nickerson, M.A. (2012). Effects of nitrogen and ammonia and MS-222 on *Xenopus laevis* development, growth, and foraging behaviour. *Bulletin of Florida Museum of Natural History*. *51*(4), 217-226.

Campbell, D. (2021). *Our growing goldfish problem*.

Campbell, K. R., and Johns, S. (1994). environmental contamination and toxicology concentrations of heavy metals associated with urban runoff in fish living in stormwater treatment ponds. In *Arch. Environ. Contam. Toxicol* (Vol. 27).

CCME. (2010). *Canadian water quality guidlines for the protection of aquatic life*.

Chivers, D. P., Brown, G. E., and Smith, R. J. F. (1996). The evolution of chemical alarm signals: Attracting predators benefits alarm signal senders. *American Naturalist*, *148*(4), 649–659. https://doi.org/10.1086/285945

Clevenot, L., Carré, C., and Pech, P. (2018a). A Review of the factors that determine whether stormwater ponds are ecological traps and/or high-quality breeding sites for amphibians. In *Frontiers in Ecology and Evolution* (Vol. 6, Issue APR). Frontiers Media S. A. https://doi.org/10.3389/fevo.2018.00040

Coelho, C., Foret, C., Bazin, C., Leduc, L., Hammada, M., Inácio, M., and Bedell, J. P. (2018). Bioavailability and bioaccumulation of heavy metals of several soils and sediments (from industrialized urban areas) for Eisenia fetida. *Science of the Total Environment*, *635*, 1317–1330. https://doi.org/10.1016/j.scitotenv.2018.04.213

Collins, S. J., and Russell, R. W. (2009a). Toxicity of road salt to Nova Scotia amphibians. *Environmental Pollution*, *157*(1), 320–324. https://doi.org/10.1016/J.ENVPOL.2008.06.032

Conan, A., Jumeau, J., Dehaut, N., Enstipp, M., Georges, J. Y., and Handrich, Y. (2022). Can road stormwater ponds be successfully exploited by the European green frog (*Pelophylax* sp.)*? Urban Ecosystems*, *25*(1), 35–47. https://doi.org/10.1007/S11252-021-01129-Z/FIGURES/3

Connor, T. P. O., Muthukrishnan, S., Barshatzky, K., Connor, T. P. O., Muthukrishnan, S., Barshatzky, K., and Wallace, W. (2012). Trace metal accumulation in sediments and benthic macroinvertebrates before and after maintenance of a constructed wetland trace metal accumulation in sediments and benthic macroinvertebrates before and after maintenance of a constructed wetland. *84*(4). https://doi.org/10.2175/106143012X13347678384486

Conte, L., and Roble, S. M. (1903). Dispersal movements and plant associations of juvenile gray treefrogs, *Hyla versicolor* Le Conte. *Source: Transactions of the Kansas Academy of Science*, *82*(4), 235–245.

Copp, G. H., Warrington, S., and Wesley, K. J. (2008). Management of an ornamental pond as a conservation site for a threatened native fish species, crucian carp Carassius carassius. *Hydrobiologia*, *597*(1), 149–155. https://doi.org/10.1007/S10750-007-9220-0/FIGURES/2

COSEWIC - Assessment and update status report on the Western Chorus frog Pseudacris triseriata in Canada. (2008).

COSEWIC Assessment and update status report on the Northern Leopard frog Lithobates pipiens . (2009).

Cronk, J. K., and Fennessy, M. S. (2009). Wetland plants. *Encyclopedia of Inland Waters*, 590–598. https://doi.org/10.1016/B978-012370626-3.00060-0

Cunnington, G. M., and Fahrig, L. (2010). Plasticity in the vocalizations of anurans in response to traffic noise. *Acta Oecologica*, *36*(5), 463–470. https://doi.org/10.1016/j.actao.2010.06.002

Cunnington, G. M., and Fahrig, L. (2013). Mate attraction by male anurans in the presence of traffic noise. *Animal conservation, 16*(3), 275-285. https://doi.org/10.1111/j.1469-1795.2012.00598.x

Datry, T., Malard, F., and Gibert, J. (2004). Dynamics of solutes and dissolved oxygen in shallow urban groundwater below a stormwater infiltration basin. *Science of the Total Environment*, *329*(1–3), 215–229. https://doi.org/10.1016/j.scitotenv.2004.02.022

Davenport, J. M., Seiwert, P. A., Fishback, L. A., and Cash, W. B. (2013). The effects of two fish predators on Wood frog (*Lithobates sylvaticus*) tadpoles in a subarctic wetland: Hudson Bay Lowlands, Canada. *Canadian Journal of Zoology, 91*(12), 866–872. https://doi.org/10.1139/CJZ-2013-0091

Dearborn, D. C., and Kark, S. (2010). Motivations for conserving urban biodiversity. *Conservation Biology*, *24*(2), 432–440. https://doi.org/10.1111/j.1523-1739.2009.01328.x

Dhalla, S. (2012). *Stormwater management criteria*.

Drygiannaki, I., Bejar, M., Reible, D. D., Dawson, J. A., Rao, B., Hayman, N. T., Rosen, G. H., and Colvin, M. A. (2020). Assessing biota accumulation due to contamination of sediments by storm water heavy metals. *Environmental Toxicology and Chemistry*, *39*(12), 2475–2484. https://doi.org/10.1002/etc.4862

Eaton, B. R., Tonn, W. M., Paszkowski, C. A., Danylchuk, A. J., and Boss, S. M. (2005). Indirect effects of fish winterkills on amphibian populations in boreal lakes. *Canadian Journal of Zoology*, *83*(12), 1532–1539.

Edwards, D. P., Gilroy, J. J., Thomas, G. H., Uribe, C. A. M., and Haugaasen, T. (2015). Land-sparing agriculture best protects avian phylogenetic diversity. *Current Biology, 25*(18), 2384–2391. https://doi.org/10.1016/j.cub.2015.07.063

Ehrenfeld, J. G. (2000). *Evaluating wetlands within an urban context*. *4*, 69–85.

Ethier, J. P., Fayard, A., Soroye, P., Choi, D., Mazerolle, M. J., and Trudeau, V. L. (2021). Life history traits and reproductive ecology of North American chorus frogs of the genus Pseudacris (*Hylidae*). *Frontiers in Zoology*, *18*(1), NA-NA. https://doi.org/10.1186/S12983-021-00425-W

Fahrig, L. (2003a). Effects of habitat fragmentation on biodiversity. In *Annual Review of Ecology, Evolution, and Systematics* (Vol. 34, pp. 487–515). Annual Reviews Inc. https://doi.org/10.1146/annurev.ecolsys.34.011802.132419

Ficetola, G. F., and de Bernardi, F. (2004). Amphibians in a human-dominated landscape: the community structure is related to habitat features and isolation. *Biological Conservation*, *119*(2), 219–230. https://doi.org/10.1016/J.BIOCON.2003.11.004

Ficetola, G. F., Maiorano, L., Falcucci, A., Dendoncker, N., Boitani, L., Padoa-Schioppa, E., Miaud, C., and Thuiller, W. (2010). Knowing the past to predict the future: land-use change and the distribution of invasive bullfrogs. *Global Change Biology*, *16*(2), 528–537. https://doi.org/10.1111/J.1365-2486.2009.01957.X

Ficken, K. L. G., and Byrne, P. G. (2013). Heavy metal pollution negatively correlates with anuran species richness and distribution in south-eastern Australia. *Austral Ecology*, *38*(5), 523–533. https://doi.org/10.1111/J.1442-9993.2012.02443.X

Fischer, J. and Quist, M.C. (2014). Gear and seasonal bias associated with abundance and size structure estimates for lentic and freshwater fishes. *Journal of Fish and Wildlife Management*, *5*(2), 394-412.

Furrer, R. D., and Pasinelli, G. (2016). Empirical evidence for source–sink populations: a review on occurrence, assessments and implications. *Biological Reviews*, *91*(3), 782–795. https://doi.org/10.1111/BRV.12195

Gallagher, M. T., Snodgrass, J. W., Brand, A. B., Casey, R. E., Lev, S. M., and van Meter, R. J. (2014). The role of pollutant accumulation in determining the use of stormwater ponds by amphibians. *Wetlands Ecology and Management*, *22*(5), 551–564. https://doi.org/10.1007/s11273-014-9351-9

Gillis, P. L., Parrott, J. L., and Helm, P. (2022). Environmental fate and effects of road run-off. In *Archives of Environmental Contamination and Toxicology* (Vol. 82, Issue 2, pp. 159–161). Springer. https://doi.org/10.1007/s00244-021-00906-3

Grimm, N. B., Faeth, S. H., Golubiewski, N. E., Redman, C. L., Wu, J., Bai, X., and Briggs, J. M. (2008). Global change and the ecology of cities. In *Science* (Vol. 319, Issue 5864, pp. 756–760). Science. https://doi.org/10.1126/science.1150195

Guesdon, G., de Santiago-Martín, A., and Galvez-Cloutier, R. (2016). Phytodesalinization potential of *Typha angustifolia*, *Juncus maritimus*, and *Eleocharis palustris* for removal of de-icing salts from runoff water. *Environmental Science and Pollution Research*, *23*(19), 19634–19644. https://doi.org/10.1007/s11356-016-7176-1

Guo, J. C. Y. (2008). Volume-based imperviousness for storm water designs. *Journal of Irrigation and Drainage Engineering*, *134*(2), 193–196. https://doi.org/10.1061/(ASCE)0733-9437(2008)134:2(193)

Hamer, A. J., Mcdonnell, M. J., Au, A. J., Hamer, M. J., and Mcdonnell). (2008). Amphibian ecology and conservation in the urbanising world: A review. https://doi.org/10.1016/j.biocon.2008.07.020

Hamer, A. J., and Parris, K. M. (2011). Local and landscape determinants of amphibian communities in urban ponds. *Ecological Applications*, *21*(2), 378–390. https://doi.org/10.1890/10-0390.1

Hamer, A. J., Smith, P. J., and McDonnell, M. J. (2012). The importance of habitat design and aquatic connectivity in amphibian use of urban stormwater retention ponds. *Urban Ecosystems*, *15*(2), 451–471. https://doi.org/10.1007/s11252-011-0212-5

Harding, J. H., and Holman, J. A. (1992). *Michigan anurans, toads and salamanders* (1st ed.). Michigan State University.

Hassall, C. (2014). The ecology and biodiversity of urban ponds. *Wiley Interdisciplinary Reviews: Water*, *1*(2), 187–206. https://doi.org/10.1002/wat2.1014

Hassall, C., and Anderson, S. (2015a). Stormwater ponds can contain comparable biodiversity to unmanaged wetlands in urban areas. *Hydrobiologia*, *745*(1), 137–149. https://doi.org/10.1007/s10750-014-2100-5

Heller, C. (2020). *The public perception of urban stormwater ponds as environmental amenities*.

Hintz, W. D., Mattes, B. M., Schuler, M. S., Jones, D. K., Stoler, A. B., Lind, L., and Relyea, R. A. (2017). Salinization triggers a trophic cascade in experimental freshwater communities with varying food-chain length. In *Ecological Applications* (Vol. 27, Issue 3).

Hirsch, P. E., N'Guyen, A., Muller, R., Adrian-Kalchhauser, I., and Burkhardt-Holm, P. (2018). Colonizing Islands of water on dry land—on the passive dispersal of fish eggs by birds. *Fish and Fisheries*, *19*(3), 502–510. https://doi.org/10.1111/FAF.12270

Hobbs, R. J., Arico, S., Aronson, J., Baron, J. S., Bridgewater, P., Cramer, V. A., Epstein, P. R., Ewel, J. J., Klink, C. A., Lugo, A. E., Norton, D., Ojima, D., Richardson, D. M., Sanderson, E. W., Valladares, F., Vilà, M., Zamora, R., and Zobel, M. (2006). Novel ecosystems: theoretical and management aspects of the new ecological world order. *Global Ecology and Biogeography*, *15*(1), 1–7. https://doi.org/10.1111/J.1466-822X.2006.00212.X

Holm, E., Mandrak, N.E. and Burridge, M. (2009). *The ROM field guide to freshwater fishes of Ontario*. Royal Ontario Museum.

Holman-Dodds, J. K. (2007). Towards greener stormwater management. *Journal of Green Building*, *2*(1), 68–96. https://doi.org/10.3992/jgb.2.1.68

Holtmann, L., Brüggeshemke, J., Juchem, M., and Fartmann, T. (2019). Odonate assemblages of urban stormwater ponds: the conservation value depends on pond type. *Journal of Insect Conservation*, *23*(1), 123–132. https://doi.org/10.1007/s10841-018-00121-x

Holtmann, L., Juchem, M., Brüggeshemke, J., Möhlmeyer, A., and Fartmann, T. (2018). Stormwater ponds promote dragonfly (Odonata) species richness and density in urban areas. *Ecological Engineering*, *118*, 1–11. https://doi.org/10.1016/J.ECOLENG.2017.12.028

Holtmann, L., Kerler, K., Wolfgart, L., Schmidt, C., and Fartmann, T. (2019). Habitat heterogeneity determines plant species richness in urban stormwater ponds. *Ecological Engineering*, *138*, 434–443. https://doi.org/10.1016/J.ECOLENG.2019.07.035

Holtmann, L., Philipp, K., Becke, C., and Fartmann, T. (2017). Effects of habitat and landscape quality on amphibian assemblages of urban stormwater ponds. *Urban Ecosystems*, *20*(6), 1249–1259. https://doi.org/10.1007/s11252-017-0677-y

Holzer, K. A. (2014). Amphibian use of constructed and remnant wetlands in an urban landscape. *Urban Ecosystems*, 955–968. https://doi.org/10.1007/s11252-014-0373-0

Howard L. (1833). *The climate of London deduced from mete- orological observations* (3rd ed.). Harvey and Darton.

Huang, D.-J., Jen, C.-H., Chen, Y.-C., Shieh, B.-S., Chen, C.-C., Lee, L.-L., and Liang, S.-H. (2021). Fish communities in urban ponds of southern Taiwan: Assessment for conservation potentials. *Taiwania, 66*(3), 337–344. https://doi.org/10.6165/tai.2021.66.337

Hutto, D., and Barrett, K. (2021). Do urban open spaces provide refugia for anurans in urban environments? *PLoS ONE*, *16*(1), 1–19. https://doi.org/10.1371/journal.pone.0244932

IUCN - Summary statistics. (2021). IUCN Red List of Threatened Species. https://www.iucnredlist.org/resources/summary-statistics#Summary%20Tables

Jesus, J. M., Calheiros, C. S. C., Castro, P. M. L., and Borges, M. T. (2014). Feasibility of typha latifolia for high salinity effluent treatment in constructed wetlands for integration in resource management systems. *International Journal of Phytoremediation*, *16*(4), 334–346. https://doi.org/10.1080/15226514.2013.773284

Jordan, C., Backe, N., Wright, M. C., and Tovey, C. (2009). *Biological synopsis of pumpkinseed (Lepomis gibbosus)*.

Karamouz, M., Asce, F., Hosseinpour, A., and Nazif, S. (2010). improvement of urban drainage system performance under climate change impact: case study. *Journal of Hydrologic Engineering*, *16*(5), 395–412. https://doi.org/10.1061/(ASCE)HE.1943-5584.0000317

Kline and Brian, J. L., and Wood, M. M. (1996). Food habits and diet selectivity of the brown bullhead. *Journal of Freshwater Ecology*, *11*(2), 145–151. https://doi.org/10.1080/02705060.1996.9663473

Knapp, S., Aronson, M., Carpenter, E., Herrera-Montes, A., Jung, K., Kotze, J., A La Sorte, F., Lepczyk, C., MacGregor-Fors, I., MacIvor, J., Moretti, M., Nilon, C., Piana, M., Rega-Brodsky, C., Salisbury, A., Threlfall, A., Trisos, C., Williams, N., Hahs, A. (2021). A research agenda for urban biodiversity in the global extinction crisis, *BioScience*, *71*(3), 268–279, https://doi.org/10.1093/biosci/biaa141

Kondratyeva, A., Knapp, S., Durka, W., Kühn, I., Vallet, J., Machon, N., Martin, G., Motard, E., Grandcolas, P., and Pavoine, S. (2020). Urbanization effects on biodiversity revealed by a two-scale analysis of species functional uniqueness vs. redundancy. *Frontiers in Ecology and Evolution*, *8*, 73. https://doi.org/10.3389/FEVO.2020.00073/BIBTEX

Kozlowski, G., and Bondallaz, L. (2013). Urban aquatic ecosystems: Habitat loss and depletion of native macrophyte diversity during the 20th century in four Swiss cities. *Urban Ecosystems*, *16*(3), 543–551. https://doi.org/10.1007/S11252-012-0284-X

Lewellyn, C., Lyons, C. E., Traver, R. G., and Wadzuk, B. M. (2016). Evaluation of seasonal and large storm runoff volume capture of an infiltration green infrastructure system. *Journal of Hydrologic Engineering*, *21*(1). https://doi.org/10.1061/(ASCE)HE.1943-5584.0001257

Li, K., Chi, G., Wang, L., Xie, Y., Wang, X., and Fan, Z. (2018). Identifying the critical riparian buffer zone with the strongest linkage between landscape characteristics and surface water quality. *Ecological Indicators*, *93*, 741–752. https://doi.org/10.1016/j.ecolind.2018.05.030

Li, L., Chan, P., Wang, D., and Tan, M. (2015). Rapid urbanization effect on local climate: intercomparison of climate trends in Shenzhen and Hong Kong, 1968-2013. *Climate Research*, *63*(2), 145–155. https://doi.org/10.3354/cr01293

Lin, B. B., and Fuller, R. A. (2013). FORUM: Sharing or sparing? How should we grow the world's cities? *Journal of Applied Ecology*, *50*(5), 1161–1168. https://doi.org/10.1111/1365-2664.12118

Liu, F., Olesen, K. B., Borregaard, A. R., and Vollertsen, J. (2019). Microplastics in urban and highway stormwater retention ponds. *Science of the Total Environment*, *671*, 992–1000. https://doi.org/10.1016/j.scitotenv.2019.03.416

MacArthur, R. H., and Wilson, E. O. (1967). *The theory of island biogeography*. Princeton University Press.

Macdonald, I. (2016). *A Winnipeg constructed wetland: inciting curiosity, facilitating learning and fostering engagement*.

Massal, L. R., Snodgrass, J. W., and Casey, R. E. (2007). Nitrogen pollution of stormwater ponds: Potential for toxic effects on amphibian embryos and larvae. *Applied Herpetology*, *4*(1), 19–29. https://doi.org/10.1163/157075407779766714

Matlaga, T. H., Phillips, C. A., and Soucek, D. J. (2014). Insensitivity to road salt: an advantage for the American bullfrog? *Hydrobiologia*, *721*(1), 1–9. https://doi.org/10.1007/S10750-013-1626-2

Mazerolle, M. J. (2020). *Model Selection and Multimodel Inference Based on (Q)AIC(c)*.

McCarthy, K., and Lathrop, R. G. (2011a). Stormwater basins of the New Jersey coastal plain: Subsidies or sinks for anurans and toads? *Urban Ecosystems*, *14*(3), 395–413. https://doi.org/10.1007/s11252-011-0161-z

Mcdonald, R. I., Marcotullio, P. J., Güneralp, B., Lead, C., Mcdonald, R. I., Marcotullio, P. J., and Güneralp, B. (2013). Urbanization and global trends in biodiversity and ecosystem services. *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities: A Global Assessment*, 31–52. https://doi.org/10.1007/978-94-007-7088-1_3

McIntyre, P. B., Flecker, A. S., Vanni, M. J., Hood, J. M., Taylor, B. W., and Thomas, S. A. (2008). Fish distributions and nutrient cycling in streams: can fish create biogeochemical hotspots. *Ecology*, *89*(8), 2335–2346. https://doi.org/10.1890/07-1552.1

McKinley, DC., Miller-Rushing, A.J. and Ballard, H.L. (2017). Citizen science can improve conservation science, natural resource management and environmental protection. *Biological Conservation*, 208, 15-28.

Miranda, L. E., and Hodges, K. B. (2000). Role of aquatic vegetation coverage on hypoxia and sunfish abundance in bays of a eutrophic reservoir. *Hydrobiologia*, *427*, 51–57.

MMP Amphibians - *Marsh monitoring program participants handbook for surveying amphibians*. (2009).

Muths, E., Rittman, S., Irwin, J., Keinath, D., and Scherer, R. (2005). *wood frog (Rana sylvatica) a technical conservation assessment*.

Oertli, B., and Parris, K. M. (2019). Review: Toward management of urban ponds for freshwater biodiversity. *Ecosphere*, *10*(7), e02810. https://doi.org/10.1002/ECS2.2810

Oksanen, J., Blanchet, F., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P., O’Hara, R., Simpson, G., Solymos, P., Stevens, M., Szoecs, E., and Wagner, H. (2020). *vegan: Community Ecology Package*.

OWES - Ontario wetland evaluation system (3rd ed.). (2013). Queens Printer for Ontario.

Page, L. M., and Burr, B. M. (2011). *Peterson field guide to freshwater fishes of North America north of Mexico /*. Houghton Mifflin Harcourt. https://books.google.com/books?id=rFTqmdDqn-wCandpgis=1

Perez, C. R., Bonar, S. A., Amberg, J. J., Ladell, B., Rees, C., Stewart, W. T., Gill, C. J., Cantrell, C., and Robinson, A. T. (2017). Comparison of american fisheries society (AFS) standard fish sampling techniques and environmental DNA for characterizing fish communities in a large reservoir. *North American Journal of Fisheries Management*, *37*(5), 1010–1027. https://doi.org/10.1080/02755947.2017.1342721

Perron, M. A. C., and Pick, F. R. (2019). Stormwater ponds as habitat for Odonata in urban areas: the importance of obligate wetland plant species. *Biodiversity and Conservation*. https://doi.org/10.1007/s10531-019-01917-2

Perron, M. A. C., and Pick, F. R. (2020). Stormwater ponds as habitat for Odonata in urban areas: the importance of obligate wetland plant species. *Biodiversity and Conservation*, *29*(3), 913–931. https://doi.org/10.1007/S10531-019-01917-2/TABLES/1

Peterson, A. C., Richgels, K. L. D., Johnson, P. T. J., and McKenzie, V. J. (2013). Investigating the dispersal routes used by an invasive amphibian, Lithobates catesbeianus, in human-dominated landscapes. *Biological Invasions*, *15*(10), 2179–2191. https://doi.org/10.1007/S10530-013-0442-Y/FIGURES/4

Pimm, S. L., and Raven, P. (2000). Extinction by numbers. In *Nature* (Vol. 403, Issue 6772, pp. 843–845). Nature. https://doi.org/10.1038/35002708

Pyke, G. H. (2008). Plague minnow or mosquito fish? a review of the biology and impacts of introduced *Gambusia* Species.

Http://Dx.Doi.Org.Proxy.Lib.Uwaterloo.ca/10.1146/Annurev.Ecolsys.39.110707.173451, *39*, 171–191. https://doi.org/10.1146/ANNUREV.ECOLSYS.39.110707.173451

QGIS Development Team. (2022). *QGIS Geographic Information System. Open Source Geospatial Foundation Project.*

Qiu, J., Yang, X., Cao, B., Chen, Z., and Li, Y. (2020). Effects of urbanization on regional extreme-temperature changes in China, 1960-2016. *Sustainability (Switzerland)*, *12*(16). https://doi.org/10.3390/su12166560

R Core Team. (2021). *A language and environment for statistical computing*. R foundation of statistical computing.

Ramesh R, Perry G, Farmer M, and Griffis-Kyle KL. (2017). Limited data used to make valid inference about targeting sites for conservation: a case study in urban amphibian ecology. *Herpetological Review*, *48*, 1–5.

Ravit, B., Gallagher, F., Doolittle, J., Shaw, R., Muñiz, E., Alomar, R., Hoefer, W., Berg, J., and Doss, T. (2017). Urban wetlands: restoration or designed rehabilitation? *AIMS Environmental Science*, *4*(3), 458–483. https://doi.org/10.3934/environsci.2017.3.458

Resetarits, W. J., and Wilbur, H. M. (1989). *Choice of Oviposition Site by Hyla Chrysoscelis: Role of Predators and Competitors*. *70*(1), 220–228.

Ricotta, C., and Podani, J. (2017). *On some properties of the Bray-Curtis dissimilarity and their ecological meaning*. https://doi.org/10.1016/j.ecocom.2017.07.003

Rooney, R. C., and Bayley, S. E. (2012). Community congruence of plants, invertebrates and birds in natural and constructed shallow open-water wetlands: Do we need to monitor multiple assemblages? *Ecological Indicators*, *20*, 42–50. https://doi.org/10.1016/j.ecolind.2011.11.029

Rooney, R. C., Foote, L., Krogman, N., Pattison, J. K., Wilson, M. J., and Bayley, S. E. (2015). Replacing natural wetlands with stormwater management facilities: Biophysical and perceived social values. *Water Research*, *73*, 17–28. https://doi.org/10.1016/j.watres.2014.12.035

Ross, L., Dupuis, L., Unterschultz, K., and Raizenne, J. (2018). *City of Brandon Naturalized Stormwater Pond Guidelines*. https://brandon.ca/images/pdf/developmentServices/waterTreatment/City_of_Brandon_Naturalized_Stormwater_Guidelines_Final.pdf

Sałata, A., Bak, L., and Górski, J. (2019). Ecological risk assessment of chemical contaminants in stormwater sediments. *E3S Web of Conferences*, *86*. https://doi.org/10.1051/e3sconf/20198600017

Scheffer, M., Hosper, S. H., Meijer, M. L., Moss, B., and Jeppesen, E. (1993). Alternative equilibria in shallow lakes. *Trends in Ecology and Evolution*, *8*(8), 275–279. https://doi.org/10.1016/0169-5347(93)90254-M

Scheffers, B. R., and Paszkowski, C. A. (2013). Amphibian use of urban stormwater wetlands: The role of natural habitat features. *Landscape and Urban Planning*, *113*, 139–149. https://doi.org/10.1016/j.landurbplan.2013.01.001

Schück, M., and Greger, M. (2022). Chloride removal capacity and salinity tolerance in wetland plants. *Journal of Environmental Management*, *308*. https://doi.org/10.1016/j.jenvman.2022.114553

Semlitsch, R. D., and Bodie, J. R. (2003). Biological criteria for buffer zones around wetlands and riparian habitats for amphibians and reptiles. *Conservation Biology*, *17*(5), 1219–1228. https://doi.org/10.1046/j.1523-1739.2003.02177.x

Sievers, M., Hale, R., Parris, K. M., and Swearer, S. E. (2018). Impacts of human-induced environmental change in wetlands on aquatic animals. *Biological Reviews*, *93*(1), 529–554. https://doi.org/10.1111/brv.12358

Sievers, M., Hale, R., Swearer, S. E., and Parris, K. M. (2019). Anuran occupancy of polluted wetlands in urban landscapes. *Conservation Biology*, *33*(2), 389–402. https://doi.org/10.1111/cobi.13210

Sievers, M., Parris, K. M., Swearer, S. E., and Hale, R. (2018). Stormwater wetlands can function as ecological traps for urban anurans. *Ecological Applications*, *28*(4), 1106–1115. https://doi.org/10.1002/eap.1714

Simmons, J. A. (2012). Toxicity of major cations and anions (Na+, K+, Ca2+, Cl−, and SO) to a macrophyte and an alga. *Environmental Toxicology and Chemistry*, *31*(6), 1370–1374. https://doi.org/10.1002/ETC.1819

Simon, J. A., Snodgrass, J. W., Casey, R. E., and Sparling, D. W. (2009). Spatial correlates of amphibian use of constructed wetlands in an urban landscape. *Landscape Ecology*, *24*(3), 361–373. https://doi.org/10.1007/S10980-008-9311-Y/FIGURES/2

Smiley, P. C., King, K. W., and Fausey, N. R. (2017). Eagle Hill Institute Influence of Different Habitat Factors on Creek Chub (Semotilus atromaculatus) within Channelized Agricultural Headwater Streams. *Source: Northeastern Naturalist*, *24*, 18–44. https://doi.org/10.2307/26453996

Smith, C. D. (2006). *Waterfowl management in stormwater management ponds*.

Søberg, L. C., Vollertsen, J., Blecken, G. T., Nielsen, A. H., and Viklander, M. (2016). Bioaccumulation of heavy metals in two wet retention ponds. *Urban Water Journal*, *13*(7), 697–709. https://doi.org/10.1080/1573062X.2015.1024689

Soga, M., Yamaura, Y., Koike, S., and Gaston, K. J. (2014). Land sharing vs. land sparing: does the compact city reconcile urban development and biodiversity conservation? *Journal of Applied Ecology, 51*(5), 1378–1386. https://doi.org/10.1111/1365-2664.12280

Stearman, L. W., and Lynch, D. T. (2013). Patterns of assemblage change in prairie stream fishes in relation to urban stormwater impoundments. *Hydrobiologia*, *718*(1), 221–235. https://doi.org/10.1007/s10750-013-1629-z

Stewart, D., Reist, T., Carmichael, T., Sawatzky, C., and Mochnacz, N. (2007). *Fish life history and habitat use in the Northwest Territories: brook stickleback (Culaea inconstans)*.

Stuart L. Pimm and Peter Raven. (2000). Extinction by numbers. *Nature*, *403*, 843–845.

TRCA. *In the face of urbanization, the green frog seems to be the most resilient in the greater Toronto area.* (2015).

TRCA - W*hy are there goldfish in toronto waterways?* (2016).

TRCA. *Scoring and ranking TRCA's vegetation communities, flora, and fauna species: A methodology for assessing degree of conservation concern for terrestrial communities, vascular plants and vertebrates*. (2017).

Kremen, C. (2015). Reframing the land-sparing/land sharing debate for biodiversity conservation. *The Year in Ecology and Conservation Biology*, *1335*(1), 52-76 https://doi.org/10.1111/nyas.12845

UN - United Nations: Department of social and economic affairs. (2018).

van Kleef, H., van der Velde, G., Leuven, R. S. E. W., and Esselink, H. (2008). Pumpkinseed sunfish (*Lepomis gibbosus*) invasions facilitated by introductions and nature management strongly reduce macroinvertebrate abundance in isolated water bodies. *Biological Invasions, 10*(8), 1481–1490. https://doi.org/10.1007/S10530-008-9220-7/FIGURES/4 van Meter, R. J., and Swan, C. M. (2014). Road salts as environmental constraints in urban pond food webs. *PLoS ONE*, *9*(2). https://doi.org/10.1371/journal.pone.0090168

van Seters, T. and D. J. (2019). *Data synthesis and design considerations for stormwater thermal mitigation measures*.

Vanni, M.J., Bowling, A.M., Dickman, E.M., Hale, R.S., Higgins, K.A., Horgan, M.J., Knoll, L.B., Renwick, W.H., Stein, R.A. (2006). *Ecology*. *87*(7), 1696 – 1709.

Vargas-Salinas, F., Cunnington, G. M., Amézquita, A., and Fahrig, L. (2014). Does traffic noise alter calling time in anurans and toads? A case study of anurans in Eastern Ontario, Canada. *Urban Ecosystems*, 945–953. https://doi.org/10.1007/s11252-014-0374-z

Vincent, J., Kirkwood, A. E., Vincent, J., and Kirkwood, A. E. (2014). Variability of water quality, metals and phytoplankton community structure in urban stormwater ponds along a vegetation gradient. *Urban ecosystems*, *17*(3), 839 – 835. https://doi.org/10.1007/s11252-014-0356-1

Vonesh, J. R., Kraus, J. M., Shoshana Rosenberg, J., Chase J R Vonesh jrvonesh, J. M., Rosenberg, J. S., Chase, J. M., and M Kraus, Á. J. (2009). Predator effects on aquatic community assembly: disentangling the roles of habitat selection and post-colonization processes. *Oikos*, *118*(8), 1219–1229. https://doi.org/10.1111/J.1600-0706.2009.17369.X

Walker, R. M., and Johansen, P. H. (1977). Anaerobic metabolism in goldfish (Carassius auratus). *Canadian Journal of Zoology*, *55*(8).

Wallace, J., Champagne, P., and Hall, G. (2016). Time series relationships between chlorophyll-a, dissolved oxygen, and pH in three facultative wastewater stabilization ponds. *Environmental Science: Water Research and Technology*, *2*(6), 132–14.

Walsh, C. J., Fletcher, T. D., and Burns, M. J. (2012). Urban stormwater runoff: A new class of environmental flow problem. *PLoS ONE*, *7*(9). https://doi.org/10.1371/journal.pone.0045814

Ward, K., Lauf, S., Kleinschmit, B., and Endlicher, W. (2016). Heat waves and urban heat islands in Europe: A review of relevant drivers. *Science of the Total Environment*, *569–570*, 527–539. https://doi.org/10.1016/j.scitotenv.2016.06.119

Watson, R. J., Butler, E. C., Clemenston L.A., and Berry, K. M. (2005). Flow-injection analysis with fluorescence detection for the determination of trace levels of ammonium in seawater. *Journal of Environmental Monitoring*, *7*(1), 37–42.

Weon, S.-Y., Lee, S.-I., and Koopman, B. (2004). Effect of temperature and dissolved oxygen on biological nitrification at high ammonia concentrations. In *Environmental Technology* (Vol. 25).

Wheeler, B., and Torchiano, M. (2016). *lmPerm: Permutation Tests for Linear Models*.

Wickham, H. 2021. *ggplot2: Create elegant data visualizations using the grammar of graphics.*

Wilby, R. L., Johnson, M. F., and Toone, J. A. (2015). Thermal shockwaves in an upland river. *Weather*, *70*(3), 92–100. https://doi.org/10.1002/wea.2435

Wilson, M. J., Forrest, A. S., and Bayley, S. E. (2013). Floristic quality assessment for marshes in Alberta's northern prairie and boreal regions. *Aquatic Ecosystem Health and Management*, *16*(3), 288–299. https://doi.org/10.1080/14634988.2013.825194

Wu, J. (2014). Urban ecology and sustainability: The state-of-the-science and future directions. *Landscape and Urban Planning*, *125*, 209–221. https://doi.org/10.1016/j.landurbplan.2014.01.018

Young, A., Kochenkov, V., McIntyre, J. K., Stark, J. D., and Coffin, A. B. (2018a). Urban stormwater runoff negatively impacts lateral line development in larval zebrafish and salmon embryos. *Scientific Reports*, *8*(1). https://doi.org/10.1038/s41598-018-21209-z

Young, A., Kochenkov, V., McIntyre, J. K., Stark, J. D., and Coffin, A. B. (2018b). Urban stormwater runoff negatively impacts lateral line development in larval zebrafish and salmon embryos. *Scientific Reports*, *8*(1). https://doi.org/10.1038/s41598-018-21209-z

Zhou, Q., Leng, G., Su, J., and Ren, Y. (2019). Comparison of urbanization and climate change impacts on urban flood volumes: Importance of urban planning and drainage adaptation. *Science of The Total Environment*, *658*, 24–33. https://doi.org/10.1016/J.SCITOTENV.2018.12.184

Ziter, C. D., Pedersen, E. J., Kucharik, C. J., and Turner, M. G. (2019). Scale-dependent interactions between tree canopy cover and impervious surfaces reduce daytime urban heat during summer. *Proceedings of the National Academy of Sciences of the United States of America*, *116*(15), 7575–7580. https://doi.org/10.1073/pnas.1817561116

Appendices

Appendix 1 – Fish abundances

Table 6- 1. Total fish species abundances over two sampling periods

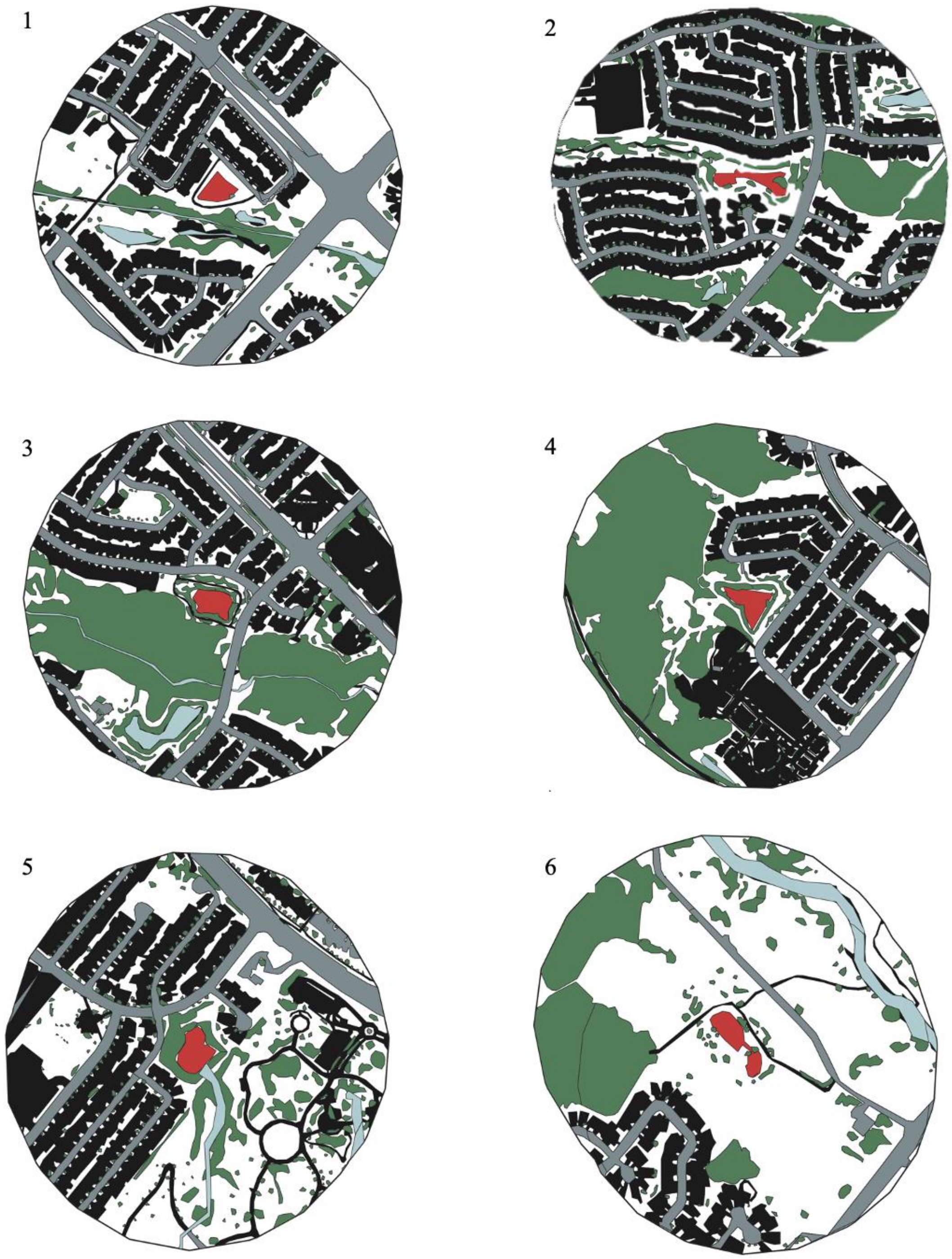
Species	August raw abundance	September raw abundance
Fathead minnow	78	51
Goldfish	3750	6624
Pumpkinseed sunfish	493	354
Brook stickleback	15	77
Brown bullhead	20	9
Creek chub	0	6

Appendix 2 – Local and landscape variables

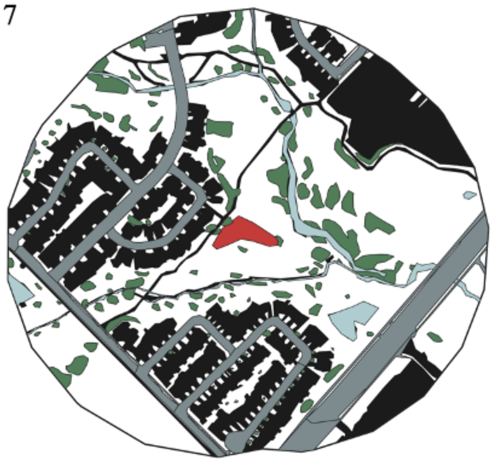
Table 6- 2. Local and landscape variables measured in this study

	Variable	Abbreviation	Mean, ± SD
Landscape (300 m)	Area of surrounding canopy cover (m^2)	Canopy	51964.42 ± 40743.16
	Area of surrounding impervious cover (m^2)	Impervious	159043.05 ± 45197.39
	Area of surrounding roads (m^2)	Road	47700.27 ± 18467.28
	Area of surrounding Water(m^2)	Water	6766.34 ± 6321.58
Local water quality	Ammonia concentration (mg L^-)	NH_3	0.20 ± 0.49
	Chloride ion concentration (mg L^-)	Cl^-	1100.06 ± 746.86
	Chlorophyll-a concentration (mg L^-)	Chla	56.05 ± 46.73
	Nitrate (mg L^-)	NO_3^-	0.04 ± 0.02
	Nitrite (mg L^-)	NO_2^-	0.02 ± 0.01
	Orthophosphate (mg L^-)	PO_4^{3-}	0.009 ± 0.01
	Phaeophytin (mg L^-)	Phaeo	46.66 ± 46.26
	Pond area	Area	2271.87 ± 851.43
	Surface water temperature (°C)	Temperature	24.02 ± 1.31
	Total suspended solids (mg)	TSS	1.20 ± 0.60
Local vegetation	Area of down trees (m^2)	DownTree	10.31 ± 18.25
	Area of Narrow leaf emergent vegetation (m^2)	NarrowLeaf	3.80 ± 14.72
	Area of Robust emergent vegetation (m^2)	RobustVeg	414.90 ± 448.03
	Area of shrubs (m^2)	Shrub	1.43 ± 2.05

Appendix 3 – Landcover surrounding SWMFs



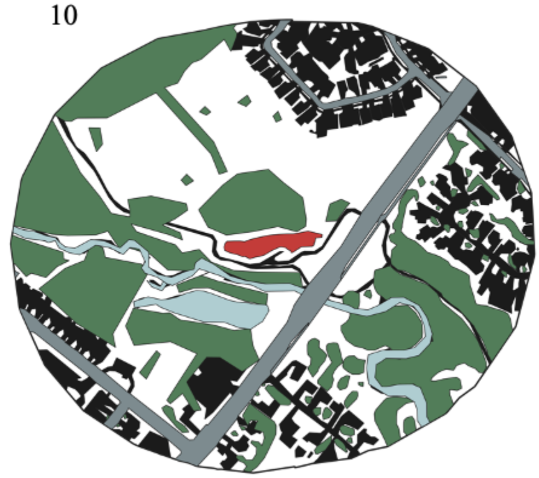
7



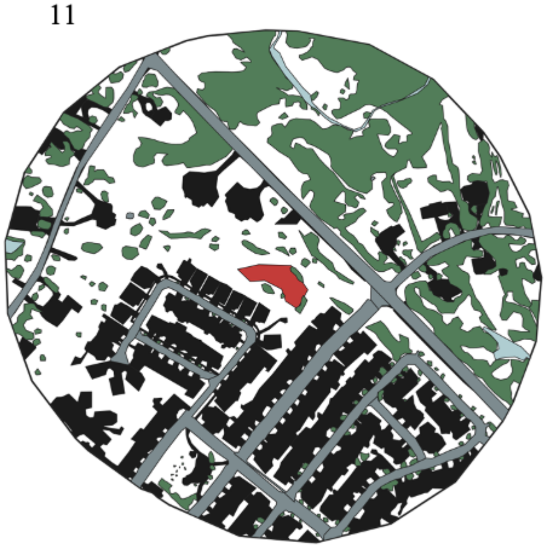
8

9

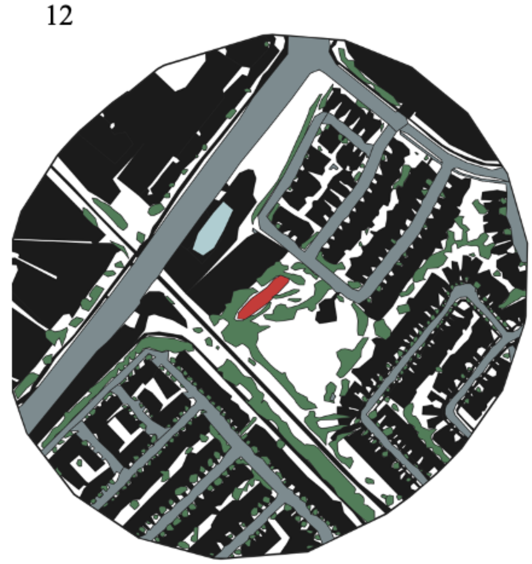
10



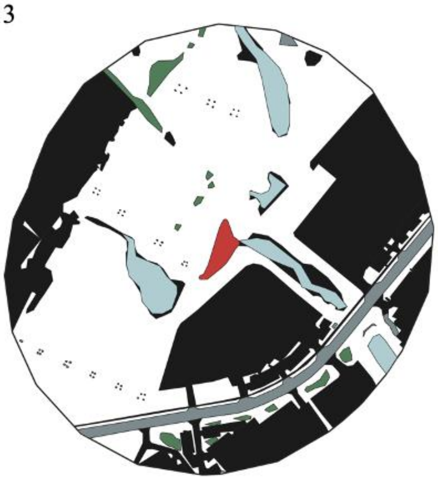
11



12

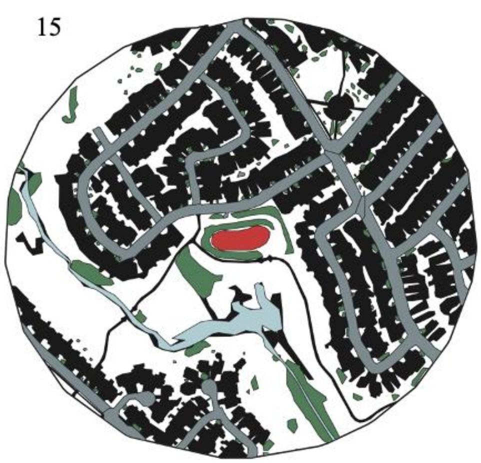


13

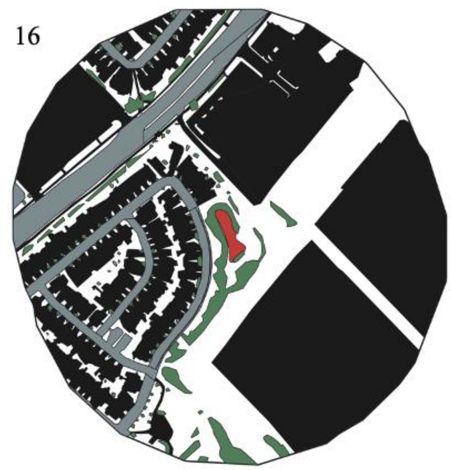


14

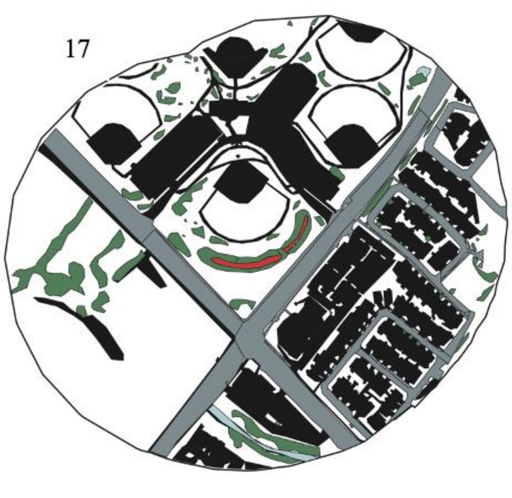
15



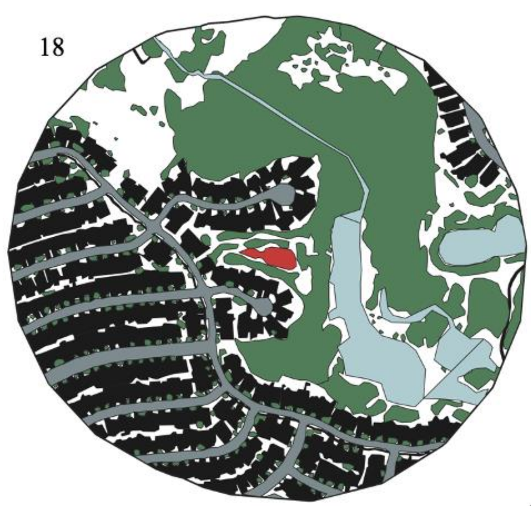
16



17



18



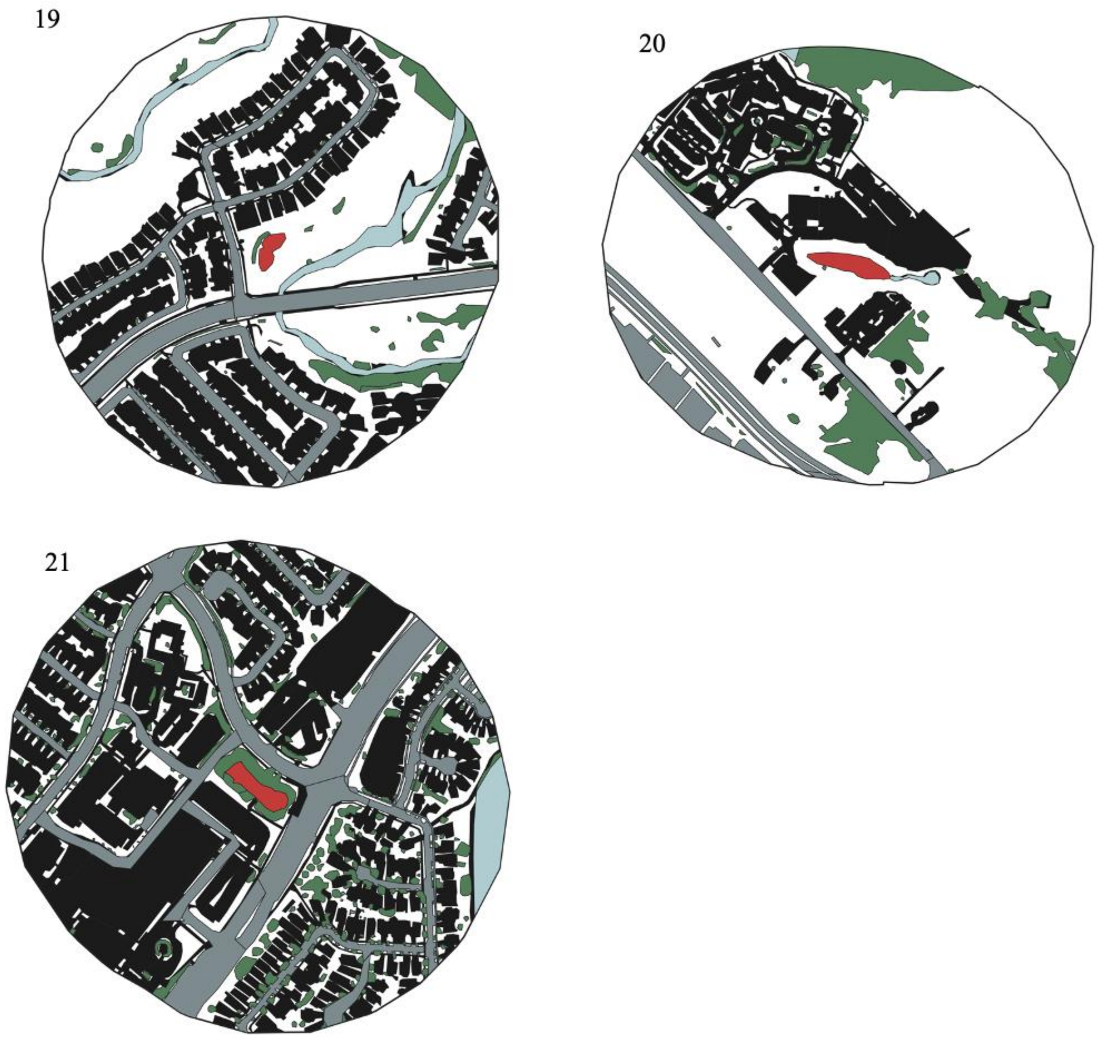


Figure 6- 1. The 21 sampled SWMFs ranging a gradient from impervious cover open water body area. Landcover are colour coded (impervious = black; road area = grey; canopy cover = green; water area = blue). Numbers correspond to Table 2-1 and Figure 2-2.

Appendix 4 - Anuran calling codes

Table 6- 3. Abundance index and description for anuran auditory surveys as in (MMP Amphibians, 2009).

Abundance index	Description
1	Calls distinguishable and easily counted, no overlap
2	Calls distinguishable and can be reliably estimated, calls overlap
3	Full chorus of overlapping calls, cannot reliably count

Appendix 5 - Community composition of anurans in SWMFS and natural wetlands

Table 6- 4. NMDS values of **(A)** axes 1 and 2 and **(B)** axes 1 and 3 corresponding to species vectors in Figure 3-3.

Vector	**A**				**B**			
	MDS1	MDS2	r^2	P-value	MDS1	MDS3	r^2	P-value
American.Toad	-0.82634	0.56318	0.1411	0.062	-0.20703	-0.97833	0.8143	0.001
Bullfrog	0.31840	0.94795	0.3029	0.003	0.85931	-0.51145	0.0664	0.332
Chorus.frog	-0.04876	-0.99881	0.3331	0.003	-0.09521	-0.99546	0.0531	0.383
Gray.Treefrog	-0.65491	0.75571	0.5442	0.001	-0.73243	0.68084	0.4189	0.001
Green.frog	0.42981	0.90292	0.7026	0.001	0.94331	-0.33191	0.2430	0.008
Leopard.frog	0.20580	-0.97859	0.2111	0.007	0.19577	-0.98065	0.1457	0.043
Spring.peeper	-0.95805	0.28661	0.6553	0.001	-0.92681	0.37552	0.6576	0.001
Wood.frog	-0.99804	0.06265	0.6423	0.001	-0.83491	0.55039	0.7158	0.001