Credit Valley Conservation

Terrestrial Monitoring Program Report

2003 - 2009

Monitoring Wetland Integrity within the Credit River Watershed - Status and Trends

CREDIT VALLEY CONSERVATION

PREFACE

This program report is a compilation of a series of chapters, each focused on one of the monitored variables within the Terrestrial Monitoring Wetland Program. Each chapter is a stand alone document; however, as the status of wetland integrity within the Credit River Watershed is dependant upon all of the variables, the chapters have been assimilated here, unabridged. These reports were originally prepared as part of the Integrated Watershed Monitoring Program (IWMP) 10 Year Review, within which summaries of these reports can be found. For additional information regarding the content of this report or the Terrestrial Monitoring Program in general please contact any of the authors of the following chapters.

The chapters contained within this report are as follows:

1. Wetland Water Quality and Hydrology
2. Wetland Anurans
3. Wetland Vegetation
4. Credit River Watershed Landscape Analysis

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Credit Valley Conservation

Terrestrial Monitoring Program Report

2006 - 2008

Monitoring Wetland Integrity within the Credit River Watershed
Chapter 1: Wetland Hydrology and Water Quality

CREDIT VALLEY CONSERVATION

Monitoring Wetland Integrity within the Credit River Watershed

Chapter 1: Wetland Hydrology and Water Quality 2006-2008

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ABSTRACT

Wetland Hydrology and Water Quality in the Credit River Watershed Summary of Monitoring Results 2006-2008
Water Quality: Currently, it is not feasible to compare water quality between sites or draw conclusions about quality of water at each site. The preliminary summary assessed water quality results in terms of the Ontario Provincial Water Quality Objectives (PWQO's) and Ontario Drinking Water Standards (ODWS's) for each site to provide some basis for presenting water quality information, although these guidelines were not developed specifically for wetlands. In groundwater samples, only uranium concentrations at Speersville exceeded the ODWS'S. Therefore, the groundwater at all other sites was in compliance with drinking water standards. The PWQO's were exceeded at all three sites where surface water was sampled. One parameter only was exceed at both Belfountain (total phosphorus) and Speersville (zinc), while aluminum, iron and total phosphorus guidelines were all exceeded at Warwick.
Hydrology: Four of the seven hydroperiod classes were represented by monitored wetlands, with Saturated wetlands being the most common. Wetlands fed through surface water inputs appeared more dominant that groundwater fed wetlands. Monitored wetlands tended to be wettest in the spring and early summer, which is consistent with snow melt and high rainfall during these times. Wetlands were driest in late summer and fall due to higher temperatures and lower rainfall. As more data become available with additional years of monitoring, hydrological classifications of the monitored wetlands will be refined and significant trends may be elucidated.
Recommendations: To measure water quality, the use of a Water Quality Index (WQI) developed specifically for inland wetland surface waters would allow both temporal and spatial comparison of wetland water quality within the watershed. Budget and time constraints need to be examined closely in order to determine if a WQI is a feasible indicator to add to wetland monitoring. Before piezometer installation occurs it is recommended that a detailed methodology for the hydrology component be developed. This will ensure that resources are used in the most optimal fashion to acquire high quality data and that hydrological monitoring will be performed consistently into the future.

Wetland hydrology and water quality monitoring was conducted as part of the terrestrial component of an overall Integrated Watershed Monitoring Program established at Credit Valley Conservation. Eighteen permanent wetland monitoring sites were established in 2002 throughout the Credit River Watershed for the initial purpose of monitoring changes in vegetation. In 2006, piezometers were installed to monitor surface and groundwater levels and water quality at a select group of monitored wetlands to develop an understanding of site hydrology.

Currently, it is not feasible to compare water quality between sites or even draw conclusions about the state of the quality of water at each individual site. The preliminary summary of the water quality results are presented in terms of the Ontario Provincial Water Quality Objectives (PWQO's) and Ontario Drinking Water Standards (ODWS's) for each individual site to provide some basis for presenting water quality information, even though these guidelines are not developed specifically for wetlands. In groundwater samples, only uranium concentrations at Speersville exceeded the ODWS'S. Therefore, the groundwater at all other sites was in compliance with drinking water standards. PWQO's were exceeded at all three sites where surface water was sampled. One parameter only was exceed at both Belfountain (total phosphorus) and Speersville (zinc), while aluminum, iron and total phosphorus guidelines were all exceeded at Warwick.

Four of the seven hydroperiod classes (Mitsch and Gosselink 2007) were represented in the watershed, with Saturated wetlands being the most common. All four of the groundwater flow patterns (Mitsch and Gosselink 2007) were represented in monitored wetlands; however, wetlands fed through surface water inputs appear to be more common than groundwater fed wetlands. Monitored sites tended to be wettest in the spring and early summer, which is consistent with snow melt and high rainfall during these times. Wetlands were driest in late summer and fall due to higher temperatures and lower rainfall. As more data become available with additional years of monitoring, hydrological classifications of the monitored wetlands will be refined and significant trends regarding changes in hydrology will be better elucidated.

In order to examine temporal and spatial trends in wetland surface water quality, the use of a Water Quality Index (WQI) developed specifically for this application would be ideal. There are a variety of options for developing a water quality monitoring program around this WQI; however, it must fist be confirmed that this coastal wetland WQI is applicable in inland wetlands as well. In addition, budget and time constraints need to be examined closely in order to determine if the addition of the WQI is feasible for the program. Plans have already been made to expand the hydrology portion of the wetland monitoring program through the planned installation of 3 piezometers at each wetland site. The addition of these piezometers will assist in confirming the preliminary hydrological classifications in this report as well as complementing data already collected on flora and fauna indicators at the permanent wetland plots. This will allow the program to link observed trends to understand the relationships between water quality, wetland hydrology, plant and animal communities more holistically.

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1.0 INTRODUCTION

1.1 BACKGROUND

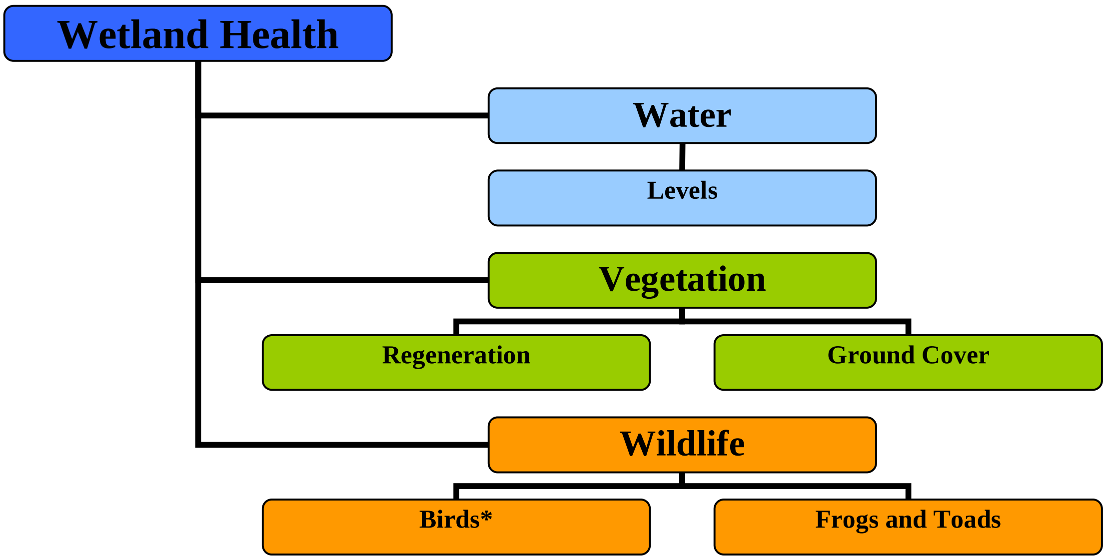
Wetlands provide numerous, irreplaceable hydrological and ecological functions, including stabilization of water supplies, flood abatement, water purification, erosion control, recharge of groundwater aquifers and carbon sequestration (Zedler and Kercher 2005; Mitsch and Gosselink 2007). Despite these benefits, wetlands have historically been viewed as unsightly waste areas and were often drained and filled to accommodate agriculture (Grand River Conservation Authority 2003). Approximately 70% of original wetland area within southern Ontario is estimated to have been drained for agricultural purposes, with the proportion increasing to over 80% near major urban centres (Natural Resources Canada 2009).

The types of alterations which lead to wetland degradation can be grouped into four categories, including: a) geomorphic and hydrologic, b) nutrient and contaminant, c) harvest, extinction and invasion, and d) climate change (Brinson and Malvarez 2002; Zedler and Kercher 2005). Current stressors specific to wetlands within the Credit River Watershed include habitat removal, nutrient enrichment, organic loading, contaminants (e.g. road salts), sedimentation, turbidity, thermal changes, dehydration, inundation, exotic species, habitat fragmentation, climate change and unsustainable use (e.g. species harvesting) (Dougan and Associates 2009). The relative impact of each of these processes on wetland health within the watershed is still unknown. Through monitoring wetlands as part of the Terrestrial Monitoring Program, it is hoped that a deeper insight into the impact of each of these processes will be gained. This will help to guide future adaptive management and restoration efforts within the Credit River Watershed.

1.2 THE IMPORTANCE OF LONG-TERM MONITORING

There is an increasing demand for better accounting of the state and health of the environment to determine whether conditions are improving or deteriorating (Niemi and McDonald 2004). An ecological monitoring program increases understanding of the trends, processes, structure and composition of a given ecosystem. A superior monitoring program should address both biotic and abiotic components across multiple scales. Monitoring is useful in providing managers with information that can be used for long-term planning because temporal trends can be employed to infer future conditions. From an ecological perspective, a monitoring program can also provide insight into cause and effect relationships between environmental stressors and ecosystem responses (Reeves et al. 2004).

Within the Credit River Watershed, 18 wetland monitoring stations have been established in which soil and vegetation parameters are measured to describe wetland conditions in the watershed and to monitor trends over a 25 year period (Fig. 1). Information gathered over time will enable the detection of changes in ecosystem structure and function and will be used to guide watershed-level management decisions.



* monitoring will begin in 2009

Figure 1. Wetland health parameters in the Credit River Watershed.

1.3 WHY MONITOR WETLAND HYDROLOGY AND WATER QUALITY?

Hydrology is possibly the most important factor for the establishment and maintenance of specific wetland types and processes (Mitsch and Gosselink 2007). It is an important component of the cycle that determines the conditions and biota within a wetland (Fig. 2). Hydrologic conditions affect many abiotic factors (e.g. soil anaerobiosis and nutrient availability) which are, in turn, responsible for the development of the biota within the wetland. This, consequently, can actively alter the wetland hydrology and physiochemical features (Mitsch and Gosselink 2007). Improved understanding of wetland function can be gained through the study of hydroperiod, water budget and water turnover time. In wetlands, hydrology can affect species composition and richness, primary productivity, organic accumulation and nutrient cycling (Mitsch and Gosselink 2007). Monitoring wetland hydrology also allows for detection of the impacts of climate change. As the climate becomes warmer, it is expected that wetland communities will transition to drier habitat types: marsh, fen and bog communities changing to swamp communities and swamps to non-wetland communities (Dougan and Associates 2009). By monitoring yearly wetland hydroperiods, detection of long-term changes to wetland hydrology should be possible.

Although wetlands are known to have the ability to cleanse contaminated water flowing into them, this contamination may negatively affect wetland biota. Wetlands may be damaged through high levels of sediment, nutrients, metals and toxic compounds (Albert and Minc 2004; Mitsch and Gosselink 2007). Sedimentation may result in high turbidity, which can lead to inadequate conditions for many aquatic macrophytes and algae to photosynthesize and survive (Albert and Minc 2004). In addition, seed germination of both emergent and submergent aquatic plants can be lost due to the deposition of thick sediments. This deposition can lead to a severe decline in plant diversity in the submergent zone. Aggressive colonizers such as Dock-leaf Smartweed

(*Polygonum lapathafolium*), Nodding Beggar-ticks (*Bidens cernua*), Spotted Touch-me-not (*Impatiens capensis*), Rice Cutgrass (*Leersia orizoides*), Bog Yellow-cress (*Rorippa palustris*), Purple Loosestrife (*Lythrum salicaria*), Common Reed (*Phragmites australis*) and Reed Canary Grass (*Phalaris arundinacea*) may take the place of other less aggressive emergent species (Albert and Minc 2004). Nutrient loading may be caused by sewage effluent or fertilizers from agricultural activities. Water carrying high levels of nutrients can result in eutrophication and associated increases in productivity, followed by increased decay rates and higher community respiration rates (Mitsch and Gosselink 2007). Nutrient enrichment can also lead to high production of blue-green algae and several species of floating-leaved plants including Duckweed (*Lemna spp.* and *Spirodela spp.*) and Watermeal (*Wolfia spp.*). Dense growth of these plants can limit available light for submergent aquatic flora (Albert and Minc 2004). Therefore, monitoring wetland water quality can identify areas where urbanization and agricultural practices are impairing water quality and if changes in water quality are occurring over time.

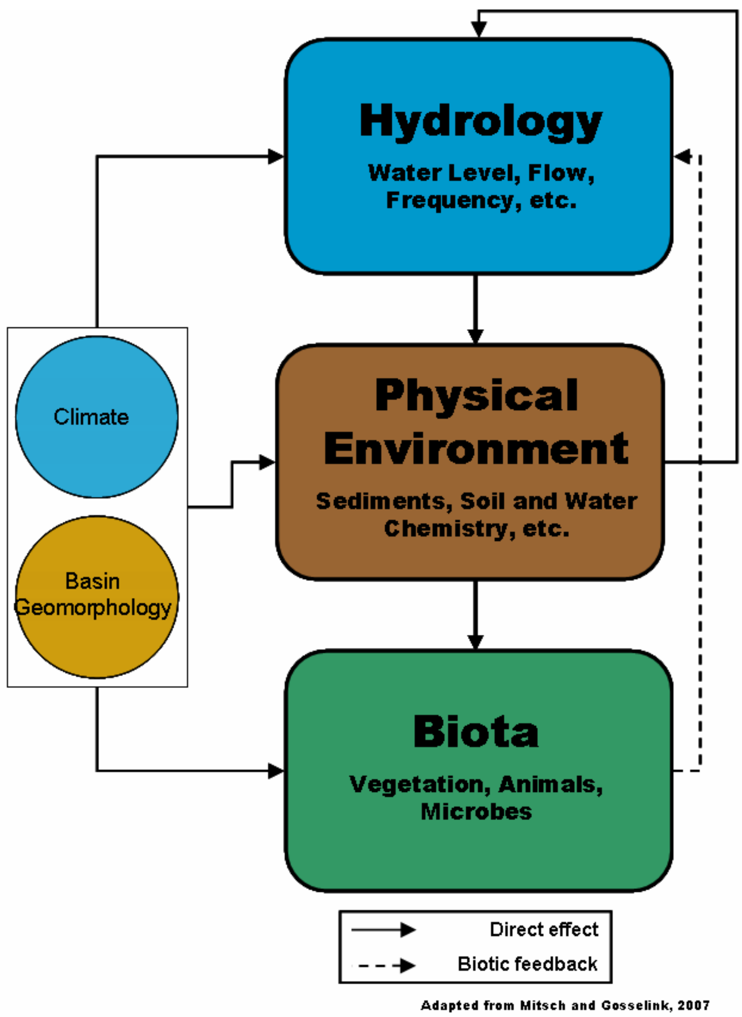


Figure 2. Conceptual diagram illustrating the effects of hydrology on wetland function and the biotic feedbacks that affect wetland hydrology (Adapted from Mitsch and Gosselink 2007).

1.4 WETLAND HYDROLOGY AND WATER QUALITY IN A CHANGING LANDSCAPE

The Credit River Watershed encompasses 950 square kilometres of land in southern Ontario, Canada (Credit Valley Conservation 2003). The Credit River flows southeast for nearly 100 km from its headwaters in Orangeville to its drainage point at Lake Ontario. It has been estimated that 33% of wetlands in the watershed have been lost (Dougan and Associates 2009). Most of this loss has occurred in areas where urban development has been most prominent. The watershed is comprised of 23% natural communities, 11% successional communities, 37% agricultural land use, and 29% urban area (Credit Valley Conservation 2007a).

Though encompassing many unique landscape formations, the Credit River Watershed can be divided into three main physiographic zones based on topography, physiographic regions and subwatershed boundaries (Fig. 3) (Credit Valley Conservation 2007b). The Lower physiographic zone is highly urbanized, containing over 85% of the population of the watershed. The topography of this area is relatively flat with a gentle slope towards Lake Ontario and it has been significantly altered by human development. The Lower watershed is comprised of lower permeability sediments, such as silt and clay till, lying on top of Georgian Bay and Queenston shales (Credit Valley Conservation 2007b). The soils in the Lower zone are comprised of clay loams associated with the Peel Plain and the South Slope physiographic regions (Credit Valley Conservation 2007b). These soils have low permeability in relation to the rest of the watershed; however, localized pockets of sand and gravel exist. When compared against government standards (Environment Canada 2004), the amount of both wetland and forest cover in the Lower watershed is classified as poor.

The Middle zone contains the Niagara Escarpment, a landform comprised of steep slopes, rocky outcrops, and thin soil (Credit Valley Conservation 2005); much of this zone is protected under the Greenbelt Plan (OMMAH 2005). The steep topography of this area leads to relatively high runoff volumes and velocities; however, the high forest cover slows runoff and increases infiltration. Soils in this zone are variable as a result of the changing physiography between the Upper and Lower zones (Credit Valley Conservation 2007b). The east portion of this zone is underlain by Queenston Shale, whereas the west portion is underlain by dolomite of the Amabel/Lockport Formation. Surface geology is dominated by silt to clay tills and silty sand to sandy silt, with pockets of ice-contract stratified drift and bedrock or bedrock drift (Credit Valley Conservation 2007b). Due to topographic factors, lower levels of urbanization and protective legislation, the Middle zone contains the greatest proportion of natural cover in the watershed.

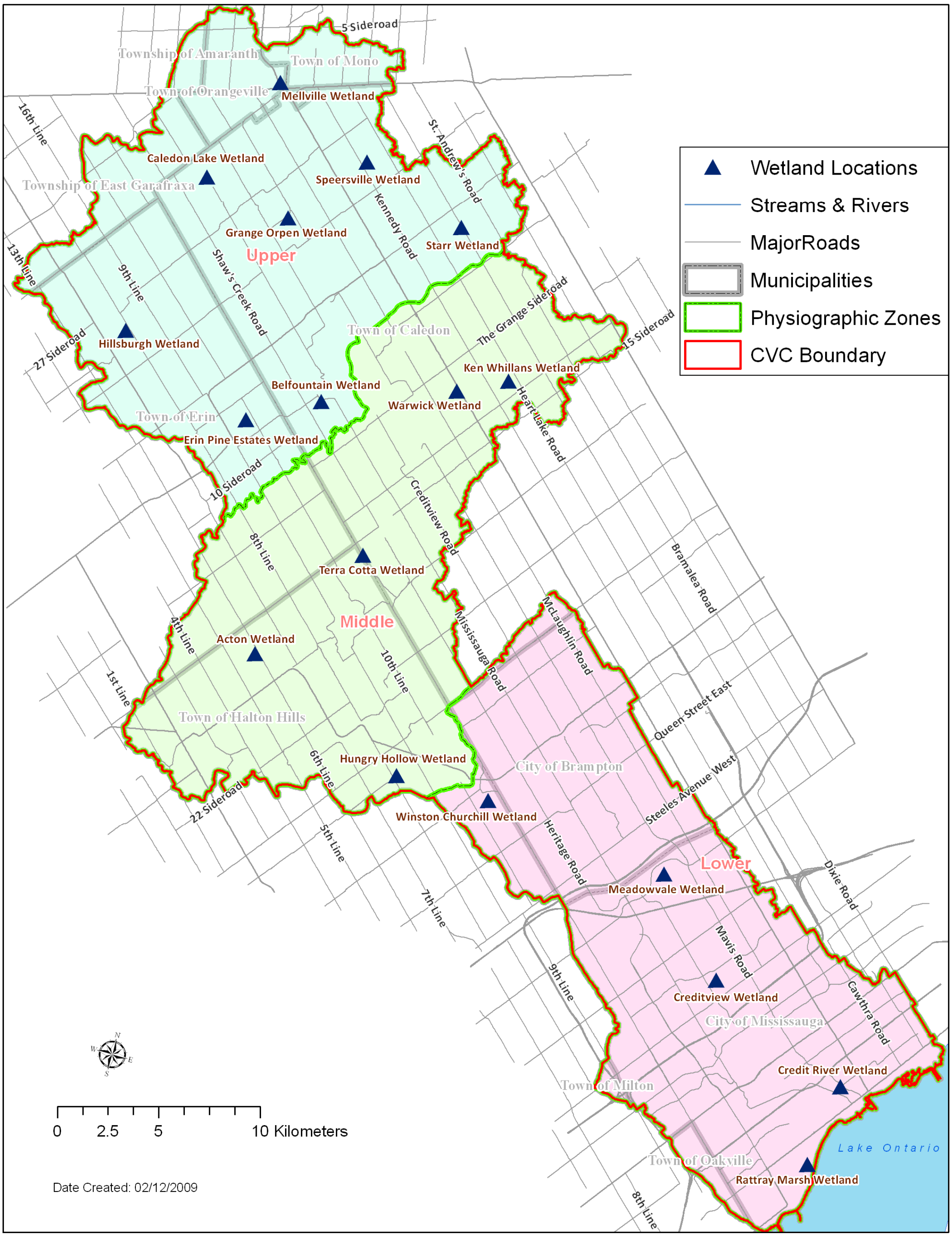


Figure 3. Wetland monitoring site locations in the Credit River Watershed.

The Upper physiographic zone lies above the escarpment and is characterized by till plains, moraines and glacial spillways (Credit Valley Conservation 2007b). This area is dominated by sandy loam soils that are associated with the Guelph drumlin field and the Hillsburgh Sandhills (Credit Valley Conservation 2007b). The high groundwater recharge rates in the Upper watershed are supported by the relatively high permeability rates of these soils. The Amabel/Lockport Formation is the dominant bedrock type in this area with pockets of the Guelph and Manitoulin dolostones (Credit Valley Conservation 2007b). Surficial geology is dominated by ice-contact stratified drift, silty sand to sandy silt and glaciofluvial gravel. Agricultural land use dominates portions of the Upper watershed, though it also contains several large wetland complexes and numerous headwater streams (Credit Valley Conservation 2007b).

Scientific research in Ontario has demonstrated that in order for good water quality to be maintained, the proportion of undeveloped land in a watershed should remain above 50% (Chow-Fraser 2006). Watersheds with predominantly agricultural land use tend to have the lowest water quality, followed closely by those watersheds dominated by urban uses (Chow-Fraser 2006). Overall, changes in landscape metrics have the ability to alter ecosystem processes, nutrient availability and water availability. All of these changes can alter wetland water quality in the watershed. Therefore, observed trends will be considered in the context of the changing landscape of the Credit River Watershed.

1.5 OBJECTIVE: MONITORING QUESTIONS

WHAT IS THE CURRENT STATE OF WETLAND WATER QUALITY AND HYRDOLOGY IN THE WATERSHED?

HOW CAN WETLAND WATER QUALITY AND HYRDOLOGY MONITORING IN THE WATERSHED BE IMPROVED?

More specifically, 1) in 2006, what was the level of groundwater and surface water quality at wetland monitoring sites?; 2) what were the apparent trends in hydrology at wetland monitoring sites between 2006 and 2008?; 3) what improvements can be made to the water quality and hydrology portion of the wetland monitoring program (Table 1)?

Table 1. Wetland water quality and hydrology monitoring framework for the Credit River Watershed between 2006 and 2008.

Monitoring Question	Monitoring Variable	Unit of Measurement	Analysis Method
Did any tested water quality parameters exceed PQWO and ODWS guidelines?	Inorganics and Metals	various	Comparison with PWQO and ODWS Guidelines
What were the apparent trends in hydrology at wetland monitoring sites?	Water Depth	meters	Descriptive

2.0 METHODS

2.1 WETLAND HEALTH MONITORING

Eighteen permanent wetland monitoring sites were established in 2002 throughout the Credit River Watershed. These sites were monitored for changes in vegetation from 2005 to 2010. The wetland sites represent 6 swamp and 12 marsh communities (Appendix A). The monitored wetlands represent 11 Provincially Significant Wetlands (PSW's), 11 Environmentally Sensitive Areas (ESA's) and six Life Science Areas of Natural and Scientific Interest (ANSI). Wetland monitoring sites consisted of 12 paired sub-plots spaced ten metres apart, established along a 50 m linear transect, which ran along the wetland's hydrological gradient (from dry to wet).

In 2006, 12 Solinst Drive-Point shielded piezometers were manually installed to the depths specified in Table 2 at 11 of the 18 wetland monitoring sites (a nest of two piezometers was installed at Erin Pines Estate Wetland). In 2008, a thirteenth piezometer was added to Rattray Marsh. These piezometers were installed to monitor surface and groundwater levels at each wetland to develop an understanding of site hydrology. Sites were selected to accommodate various types of wetlands (i.e. groundwater fed versus surface water fed). Piezometers were installed at sites where there was typically standing water for 6 months of the year. At sites where standing water is typically present for shorter periods, a two piezometer nest was installed in order to obtain more accurate information about the site.

A piezometer is a small diameter well or pipe that is only partially screened and is designed to measure groundwater conditions at a single point within an aquifer (Fig. 4) (Commonwealth of Australia 2006; Mitsch and Gosselink 2007). Individual piezometers may be installed at a site in order to measure groundwater levels or a nest of piezometers may be installed, where each piezometer is installed to a different depth. Nested piezometers provide information regarding the vertical hydraulic gradient at any of the depths to which the piezometers are installed (Commonwealth of Australia 2006).

The piezometers used by CVC have a pointed shield at the end of the piezometer to allow direct installation into the ground without having to pre-drill a hole (Fig. 4). This shield must be removed from the end of the piezometer after installation to reveal the screening. In order to do this, after the piezometer has been installed to the desired depth, it must be pulled slightly back out of the hole to release the shield. This shield cannot be recovered if the piezometer is removed and needs to be reinstalled. Therefore, it is important that no rocks or impenetrable objects are present in the subsurface layer that would prevent the piezometer from reaching its desired depth. Therefore, the piezometer installation process is as follows (Fig. 5):

Table 2. Water quality analyses and water level measurements completed at wetland monitoring sites throughout the Credit River Watershed.

Site #	Site Name	Physiographic Zone	Wetland Type	Approximate Distance from Transect to Piezometer	Piezometer Depth (mbgs)	Ground and Surface Water Levels	Inorganics and Metals		Temperature, Dissolved Oxygen, Conductivity, pH	
							Groundwater	Surface Water	Groundwater	Surface Water
W-01	Rattray Marsh Wetland	Lower	Marsh	< 40m	2.75	X				
W-02	Credit River Wetland	Lower	Marsh	-- [a]	--					
W-03	Creditview Wetland	Lower	Marsh	--	--					
W-04	Meadowvale Wetland	Lower	Marsh	>50m	0.98	X			X	X
W-12	Winston Churchill Wetland	Lower	Marsh	--	--					
W-06	Hungry Hollow Wetland	Middle	Swamp	--	--					
W-07	Acton Wetland	Middle	Swamp	<15m	1.62	X	X		X	
W-08	Terra Cotta Wetland	Middle	Swamp	--	--					
W-09	Ken Whillans Wetland	Middle	Swamp	<15m	1.94	X	X		X	
W-10	Warwick Wetland	Middle	Marsh	<5m	1.85	X	X	X	X	X
W-11	Erin Pine Estates Wetland [b]	Upper	Marsh	<35m	1.01 Shallow 2.35 Deep	X	X		X	
W-13	Hillsburgh Wetland	Upper	Marsh	<15m	2.34	X	X		X	
W-15	Grange Orpen Wetland	Upper	Swamp	<5m	2.41	X	X		X	
W-16	Starr Wetland	Upper	Marsh	<30m	2.33	X	X		X	X
W-17	Speersville Wetland	Upper	Marsh	<5m	1.99	X	X	X	X	X
W-18	Caledon Lake Wetland	Upper	Swamp	--	--					
W-19	Melville Wetland	Upper	Marsh	<5m	2.2	X	X		X	
W-20	Belfountain Wetland	Upper	Marsh	<5m	1.79	X	X	X	X	X

[a] Indicates that a piezometer is not currently installed at the wetland monitoring site.
[b] Two piezometers installed at this site.

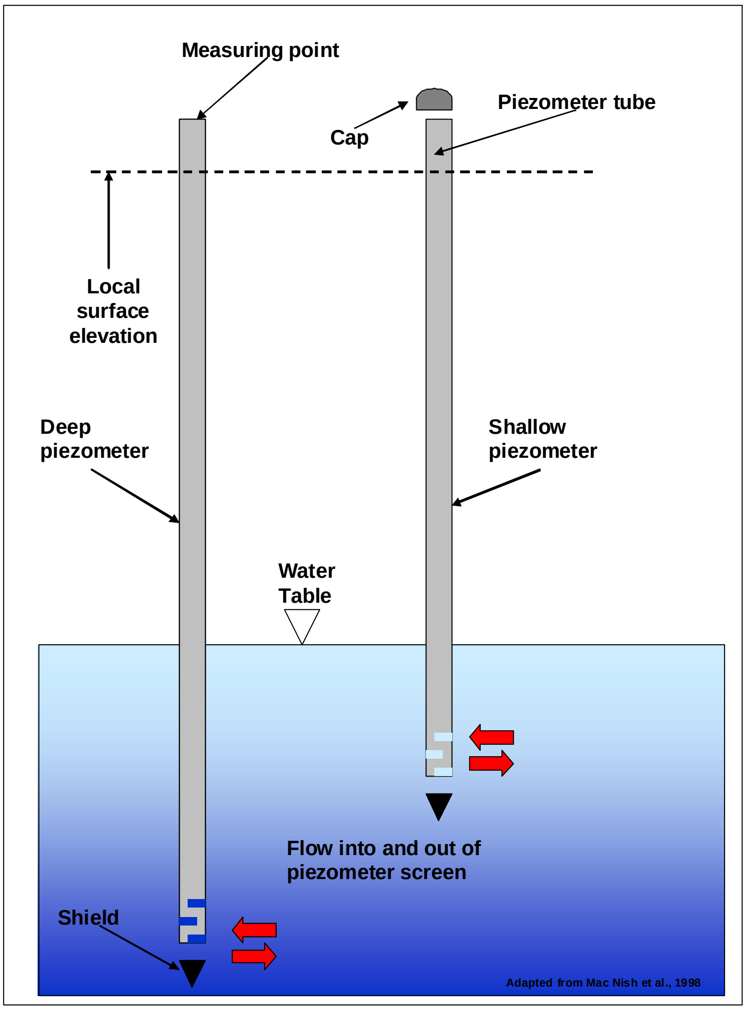


Figure 4. Example schematic of a piezometer nest (Adapted from Mac Nish et al. 1998).

1. A steel rod (A) may be hammered into the substrate using a manual slide hammer (B) previous to installing the piezometer to ensure that no impenetrable objects are present in the subsurface layer.
2. The steel rod is removed once it is determined that it is safe to install the piezometer.
3. The piezometer (C) is then hammered into the resulting hole using the manual slide hammer to the desired depth.
4. The piezometer is then pulled slightly back out of the hole to release the shield (D) located at the bottom of the piezometer.

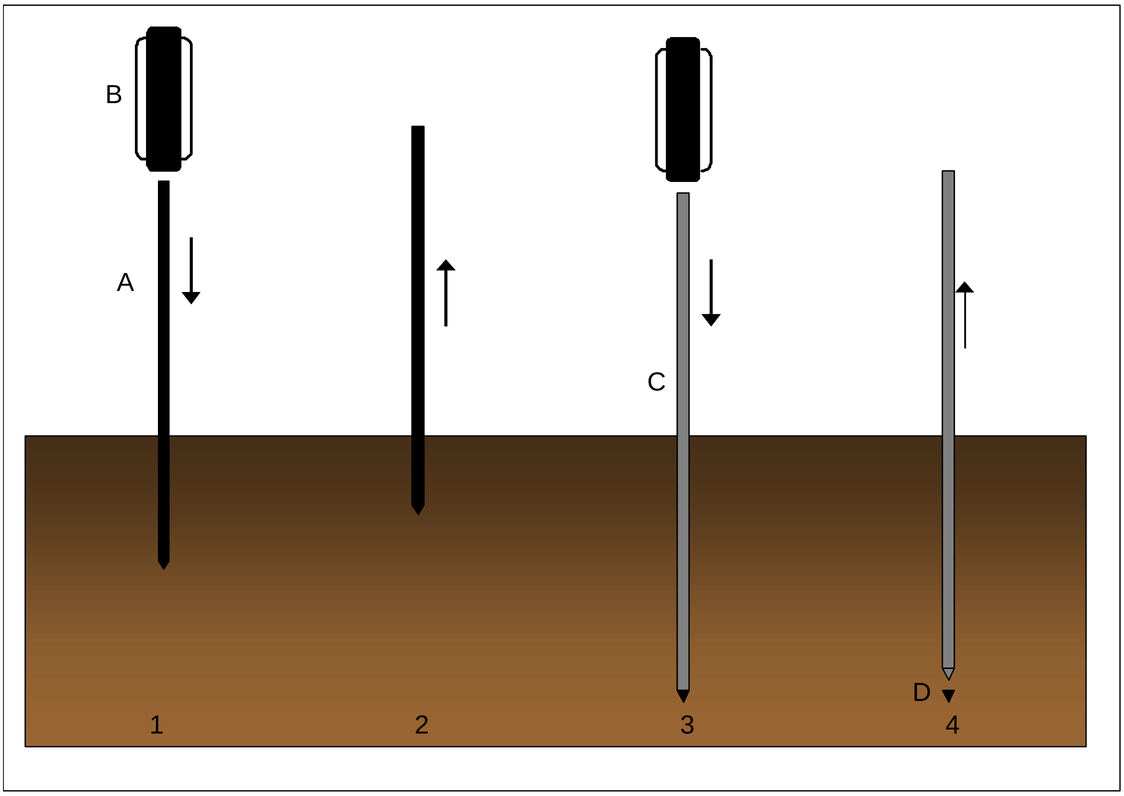


Figure 5. Piezometer installation process.

Groundwater depth is measured using a water level meter, which is a narrow measuring tape with a probe at the end. The tape is lowered into the piezometer and when the probe comes into contact with water, it activates a buzzer and light. The measurement is then taken from the specified measuring point at the top of the piezometer. The same water level meter can be used to measure surface water depth. This is done by measuring from the measuring point to the surface water and to the ground surface. Piezometers can also be used to sample groundwater quality using the water lift method as follows (CH2M Hill 2009):

1) A one-way check valve is installed in a length of Teflon or plastic tubing long enough to reach the bottom of the piezometer and facilitate easy dispensing of water into sample bottles.
2) The tubing is inserted into the piezometer.
3) The piezometer is purged by quickly raising and lowering the tubing in the piezometer to lift water past the check valve.
4) The piezometer should be purged once prior to taking a sample to ensure that the water is representative of *in situ* groundwater.
5) After the water in the piezometer has recharged, the piezometer is purged to acquire a groundwater sample. Piezometer recharge time will vary by site.
6) Sample water should be dispensed into labelled sample bottles and placed in a cooler until relinquished for analysis.

Water depth of both ground and surface water was measured at all piezometers approximately every 2 weeks from May until freeze (approximately October to December) from 2006 to 2008. In June 2006, at the time of piezometer instalment, water quality was sampled for the analysis of a suite of parameters, including inorganics and metals. Groundwater was sampled via the method described above. Surface water was sampled by obtaining a grab sample from standing water at the location of each piezometer, where water depth allowed. At Starr Wetland, a sample was also taken from the spring nearby the piezometer. Water samples were then sent to Maxxam Analytics Inc. for analysis. Field measurements including temperature, dissolved oxygen (DO), conductivity and pH were also measured for both ground and surface water. This was completed by placing the appropriate HydroLab probe into each water sample and recording stabilized values. Field measurements were taken between 8:00am and 3:00pm. Surface water samples were taken to confirm that surface water was not directly leaking into the groundwater in the piezometer, as groundwater is expected to be more pristine than wetland surface water. Unfortunately, water quality sampling was not completed at all sites. Table 2 outlines analyses and measurements taken at each site.

3.0 PRELIMINARY RESULTS

3.1 WATER QUALITY

Currently, it is not feasible to compare water quality between sites or even draw conclusions about the state of quality of water at each individual site. This is because water quality was analyzed only once in three years and not all parameters were measured at all sites. Comparing water quality between sites is easiest when employing a water quality index (WQI). Although a WQI has been developed specifically for use in coastal wetlands (Chow-Fraser 2006), the parameters tested in 2006 are not consistent with those used in the index, nor is there confirmation that this WQI is reliable in inland wetland systems. In addition, the current samples provide only a brief “snap-shot” of water quality at each site. This may not be reflective of the long-term pattern of water quality at the sites that may affect biogeochemical cycles and wetland biota (U.S. EPA 2008). More frequent sampling is required in order to develop reasonable estimates of water quality to account for changes in water quality during and between seasons. In order to overcome these limitations, suggestions for program improvement are provided in Section 5.

The preliminary summary of the water quality results will instead be discussed in terms of the Ontario Provincial Water Quality Objectives (PWQO’s) and Ontario Drinking Water Standards (ODWS’s) for each individual site to provide some basis for presenting water quality information. Neither set of guidelines were designed for wetland systems; The PWQO’s are surface water guidelines developed to protect aquatic life, while the ODWS’s were developed to provide safe drinking water to the public. Although groundwater and surface water are interconnected, the properties of these two water sources can be very different. In general, surface water has lower mineral content but higher toxin contaminate concentrations (bacteria, pesticides, industrial waste) than groundwater. Surface water is also generally faster moving, with higher dissolved oxygen concentrations and greater risk of point source pollution. Groundwater is slower moving with lower oxygen contents. Because of the percolation of water through the soil column and its long stay in underground aquifers, groundwater tends to have less contamination and higher mineral contents. Due to these differences, the PWQO’s may be more applicable to the measured surface water samples, while the ODWS’s may be more applicable to the groundwater samples. Therefore, although comparisons are provided for reference for both set of guidelines for surface and groundwater the ODWS’s will be used to determine the level of water quality for groundwater samples and the PWQO’s will be used to determine the level of water quality for surface water samples.

Laboratory analysis was completed for a suite of over 45 water quality parameters including inorganics and metals; five additional parameters were measured in the field (Appendix B). Table 3 lists the parameters analyzed for which PWQO and/or ODWS guidelines are associated. The guidelines listed in Table 3 indicate maximum allowable concentrations for all parameters except pH, which indicates a range within which pH should fall. Although PWQO guidelines are available for pH and DO, these guidelines were not used for comparisons in this report because they are not applicable in wetland ecosystems. Many of the parameters in Table 3 were not detected at any of the monitoring sites and will not be discussed further.

Table 3. Provincial Water Quality Objectives (PWQO's) and Ontario Drinking Water Standards (ODWS's) for parameters tested in wetland ground and surface water in the Credit River Watershed.

Parameter	Unit	PWQO [b]	ODWS [b]
Metals			
Aluminum (Al)	µg/L	75	N/A
Antimony (Sb)	µg/L	20	6
Arsenic (As)	µg/L	5	25
Barium (Ba)	µg/L	N/A	1000
Beryllium (Be) [a]	µg/L	11 or 1100 [e]	N/A
Boron (B)	µg/L	200	5000
Cadmium (Cd) [a]	µg/L	0.2	5
Chromium (Cr) [a]	µg/L	1	50
Cobalt (Co)	µg/L	0.9	N/A
Copper (Cu)	µg/L	5	N/A
Iron (Fe)	µg/L	300	N/A
Lead (Pb)	µg/L	5	10
Molybdenum (Mo)	µg/L	40	N/A
Nickel (Ni)	µg/L	25	N/A
Selenium (Se) [a]	µg/L	100	10
Silver (Ag)	µg/L	0.1	N/A
Thallium (Tl) [a]	µg/L	0.3	N/A
Tungsten (W) [a]	µg/L	30	N/A
Uranium (U)	µg/L	5	20
Vanadium (V)	µg/L	6	N/A
Zinc (Zn)	µg/L	20	N/A
Zirconium (Zr) [a]	µg/L	4	N/A
Inorganics			
Total Phosphorus (TP)	µg/L	20 - 30 [d]	N/A
Nitrite (N)	mg/L	N/A	1
Nitrate (N)	mg/L	N/A	10
Nitrite + Nitrate	mg/L	N/A	10
Other			
pH [c]	N/A	6.5-8.5	N/A
Dissolved Oxygen (DO) [c]	mg/L	Temperature dependant	N/A

[a] Parameters not detected at any monitored wetland.
[b] Indicates the maximum allowable concentrations for all parameters except pH, which indicates a range within which pH should fall.
[c] Although PWQO guidelines are available, these guidelines were not used for comparisons because they are not applicable in wetland ecosystems.
[d] 20mg/L is the guideline for lakes, whereas 30mg/L is the guideline for rivers
[e] 11µg/L when Hardness as CaCO3 (mg/L) <75; 1100µg/L when Hardness as CaCO3 (mg/L) >75

All metals were analysed using dissolved concentrations for groundwater and total concentrations for surface water. Phosphorus is present in aquatic systems in three forms: inorganic phosphorus, particulate organic phosphorus, and dissolved (soluble) organic phosphorus (Canadian Council of Ministers of the Environment 2004). Phosphate, the most dominant type of inorganic phosphorus, is required by aquatic biota for nutrition. However, due to the relative difficultly in estimating phosphate, total phosphorus (TP) (the sum of all three forms) is typically used for measuring phosphorus in surface waters (Canadian Council of Ministers of the Environment 2004).

3.1.1 Groundwater

3.1.1.1 Laboratory Analyses: Aluminum, antimony and zinc were each detected at only two wetlands: aluminum and antimony at Acton and Erin Pine Estates and zinc at Belfountain and Speersville. Observed levels of these parameters were well below the PWQO and/or ODWS guidelines. Barium, boron, molybdenum, nickel and nitrogen compounds (nitrite, nitrate and nitrite + nitrate) were detected at the majority of the locations, but concentrations always fell below prescribed guidelines (Appendix B).

At Speersville, the uranium concentrations reached 35 μg/L, exceeding both the PWQO guideline and the 25 μg/L ODWS guideline (Fig. 6). The 5 μg/L PWQO guideline for uranium was also exceeded at Acton and Erin Pine Estates Wetlands. It is unclear why concentrations were so high at this site. Although uranium is a radionuclide, studies show that chemical toxicity is of higher concern than radiological toxicity (Sheppard et al. 2005). Uranium toxicity can lead to malfunctioning of several mammalian organs, most notably the kidneys. Invertebrates and aquatic plants appear to be the most sensitive to uranium toxicity, and may be negatively affected by the uranium concentrations found at the three aforementioned sites; however, these levels are well below toxicity levels for fish (Sheppard et al. 2005).

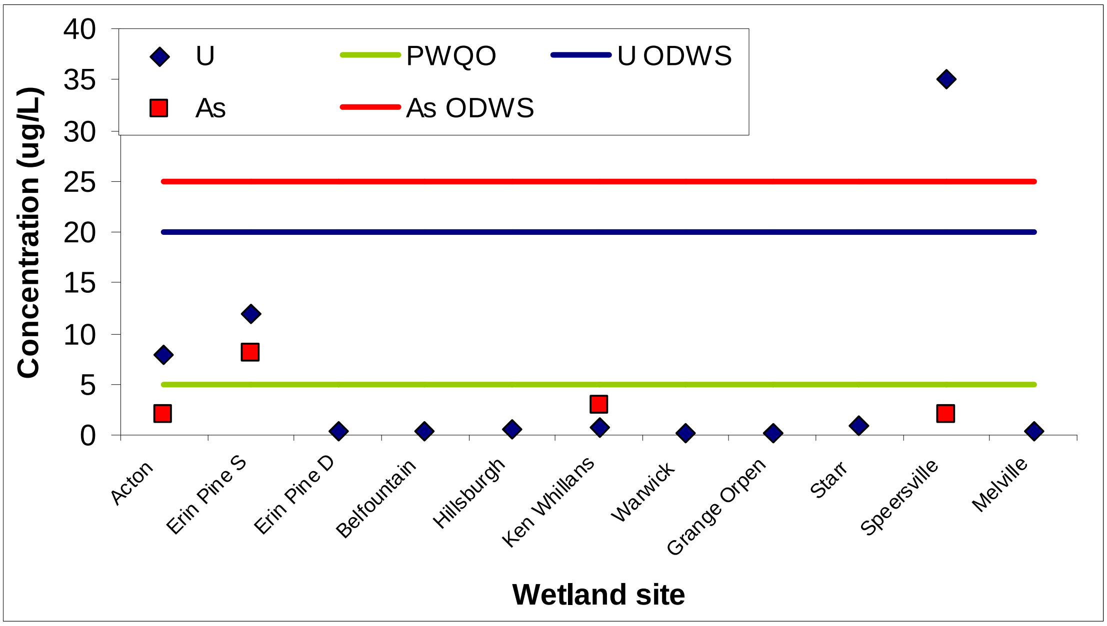


Figure 6. Arsenic (As) and Uranium (U) groundwater concentrations (points) at monitored wetlands compared to PWQO and ODWS guidelines (solid lines). Erin Pine S, Erin Pine Estates shallow piezometer; Erin Pine D, Erin Pine Estates deep piezometer. Parameter was not detected at a given site if no point is present on the graph.

The 25 µg/L ODWS guideline for arsenic was not exceeded at any site (Fig. 6). However, the 5 µg/L PWQO guideline was exceeded at Erin Pine Estates Wetland. Arsenic is an ecotoxic metal that enters water through the natural weathering of rocks and soils and through smelting and refining industries. Though arsenic does not appear to be biomagnified within the food chain, some species of aquatic plants appear to be the most sensitive aquatic biota to arsenic toxicity (Canadian Council of Ministers of the Environment 2001).

No ODWS guidelines exist for copper, vanadium, silver or cobalt. However, the PWQO guideline for each of these metals was exceeded at one or more sites. The 5 µg/L PWQO guideline for copper was exceeded at Ken Whillans Wetland and the 6 µg/L PWQO guideline for vanadium was exceeded at Starr and Erin Pine Estates Wetlands (Fig. 7). Microorganisms such as algae and plankton are the most sensitive to copper, while macrophytes are the least (Nor 1987). Anthropogenic sources of elevated copper levels include mining, industrial discharges, sewage sludge disposal and fertilizer and pesticide applications (Nor 1987). Similar to other metals, a large portion of the vanadium in water originates from erosion of rock and soil. However, concentrations can be elevated through acid rain and contaminated fall-out from the combustion of coal and crude oil (Beusen and Neven 1987). Vanadium has been found to be toxic to both aquatic invertebrates and several species of fish and appears to result in mortality rather than reproduction inhibition (Beusen and Neven 1987).

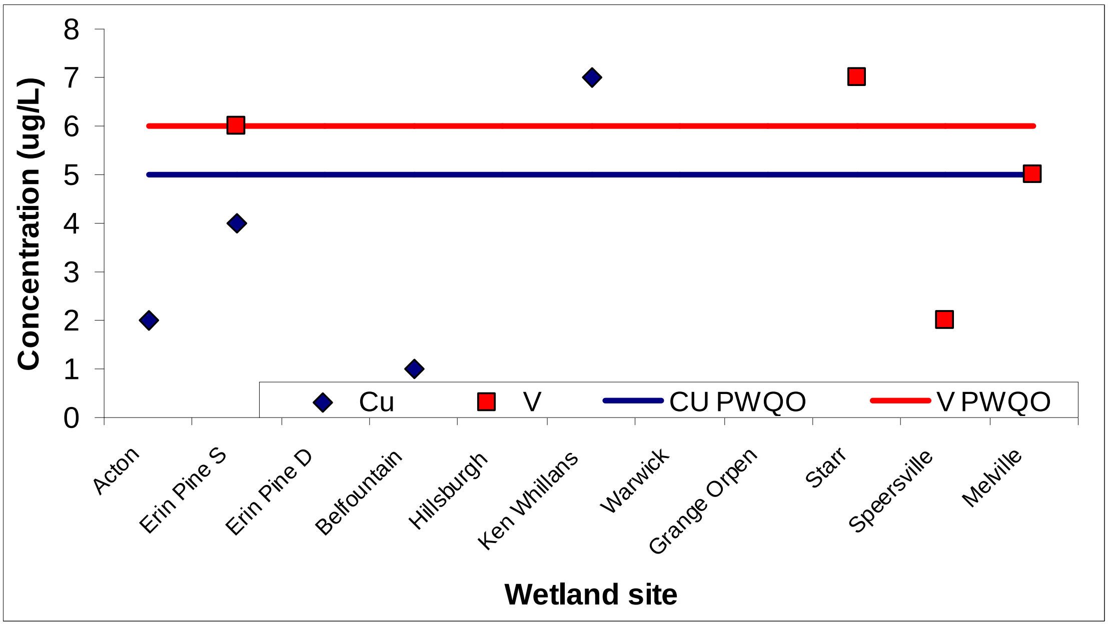


Figure 7. Copper (Cu) and Vanadium (V) groundwater concentrations (points) at monitored wetlands compared to PWQO guidelines (solid lines). Erin Pine S, Erin Pine Estates shallow piezometer; Erin Pine D, Erin Pine Estates deep piezometer. Parameter was not detected at a given site it no point is present on the graph.

The cobalt PWQO guideline was exceeded at Acton Wetland, while the PWQO guideline for silver was exceed at Starr, Acton and Speersville Wetlands (Fig. 8). Negative toxic effects of cobalt on aquatic organisms may include mortality, bioaccumulation and changes to biota physiology, biochemistry, growth, development, populations and behaviour. Cobalt has been found to be highly toxic to fish, amphibians and zooplankton (Kegley et al. 2009). The concentration of silver in water is usually caused by natural sources; however, mining and photographic processing are the main anthropogenic factors which may increase these concentrations. In fish and crustaceans, the toxic mechanism of silver involves an interference with ionic uptake at the gill surface; therefore, acute silver toxicity is higher in smaller organisms than larger ones because gill area is inversely related to body mass (Bianchini et al. 2002).

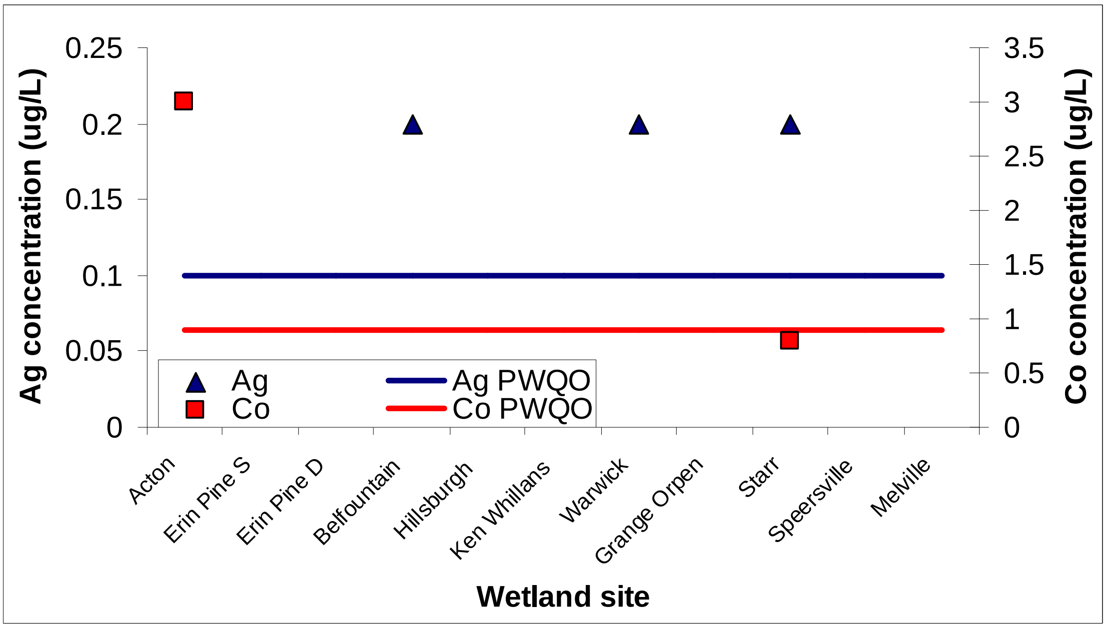


Figure 8. Silver (Ag) and Cobalt (Co) groundwater concentrations (points) at monitored wetlands compared to the PWQO guidelines (solid lines). Erin Pine S, Erin Pine Estates shallow piezometer; Erin Pine D, Erin Pine Estates deep piezometer. Parameter was not detected at a given site it no point is present on the graph.

There is also no ODWS guideline for total phosphorus (TP), but these concentrations met or exceeded both the 20 µg/L and 30 µg/L PWQO guidelines at all sites (Fig. 9). Phosphorus is a key macronutrient required by biota for biological metabolism. It tends to be the least abundant macronutrient, and therefore, is most often the limiting factor for biological productivity (Canadian Council of Ministers of the Environment 2004). Water with low P concentrations tends to support relatively diverse aquatic life. However, high P concentrations can lead to adverse effects on aquatic ecosystems, such as decreased biodiversity, anoxic conditions, increased turbidity, increased plant and animal biomass, a decline in ecologically sensitive species and an increase in nutrient tolerant species (Canadian Council of Ministers of the Environment 2004).

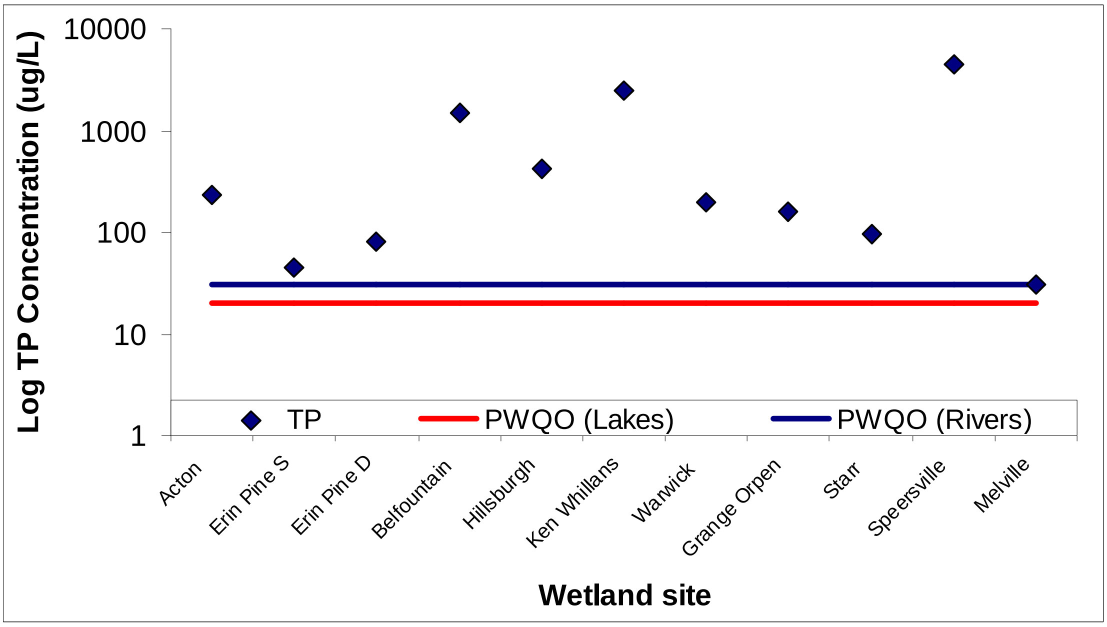


Figure 9. Total Phosphorus (TP) groundwater concentrations (points) at monitored wetlands compared to the PWQO guidelines (solid lines). Erin Pine S, Erin Pine Estates shallow piezometer; Erin Pine D, Erin Pine Estates deep piezometer. Parameter was not detected at a given site if no point is present on the graph.

3.1.1.2 Field Analyses: Provincial Water Quality Objective guidelines for flowing water for pH range from 6.5-8.5. The deviation of pH from this recommended range can be lethal for fish and invertebrates, such as caddis flies. In addition, the toxicity of other water components (e.g. trace metals and carbon dioxide) may increase with changes in pH (Ministry of the Environment 1979). These guidelines, however, are not applicable in wetland systems. No standards for pH exist in the ODWS. Mean pH in the Upper and Middle zones was 7.37and 7.75, respectively (Fig. 10). At Meadowvale Wetland, the only site sampled in the Lower zone, pH was 7.04. The lowest pH was detected at Belfountain (6.70), while the highest pH was detected at Speersville (8.68).

Similarly, no standards for DO exist in the ODWS. In flowing waters with temperatures ranging from 10-25^{o}C, PWQO's suggest that the minimum DO requirements for aquatic life ranges from 4 to 8 mg/L depending on temperature and biota type (warm vs. cold water). The absence of DO leads to anaerobic decomposition and the production of noxious gases. Low levels of DO cause fish to expend high amounts of energy to continue breathing. This reduces energy available for other essential activities, may increase lethal effect of other toxins and can eventually lead to death (Ministry of the Environment 1979). However, these standards do not apply to groundwater due to the inherent difference in the properties of groundwater. Mean DO concentrations in the Upper and Middle zones were 6.49 mg/L and 7.15 mg/L, respectively (Fig. 11). At Meadowvale Wetland, the only site sampled in the Lower zone, the DO concentration was 5.52 mg/L.

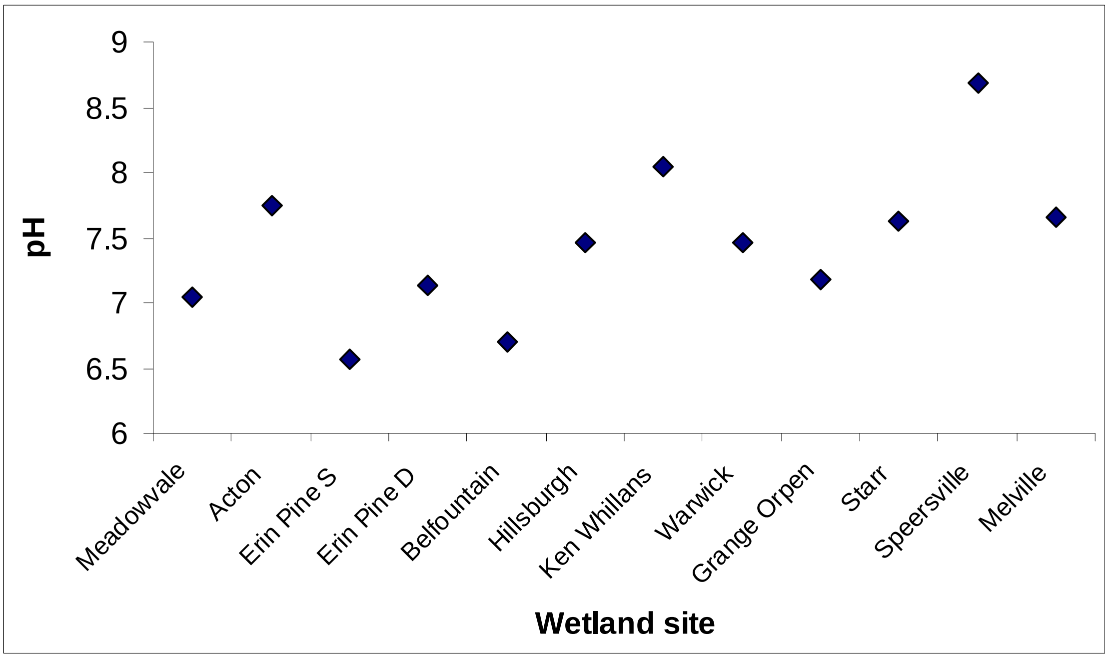


Figure 10. Groundwater pH at monitored wetlands. Erin Pine S, Erin Pine Estates shallow piezometer; Erin Pine D, Erin Pine Estates deep piezometer.

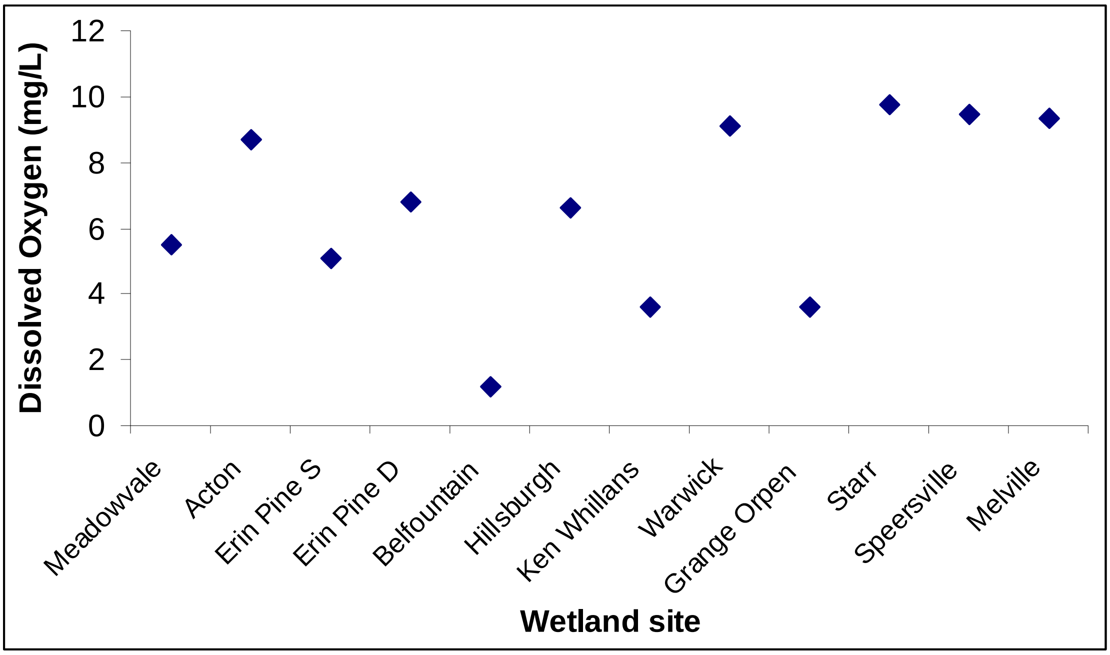


Figure 11. Groundwater dissolved oxygen (DO) concentrations at monitored wetlands. Erin Pine S, Erin Pine Estates shallow piezometer; Erin Pine D, Erin Pine Estates deep piezometer.

Neither the ODWS or the PWQO guidelines set specific recommended ranges for water temperature. Instead, the ODWS set an aesthetic objective of 15°C to enhance drinking water taste, ordor and colour. In the PWQO guidelines it is stated that "the natural thermal regime of any body of water shall not be altered so as to impair the quality of the natural environment. In particular, the diversity, distribution and abundance of plant and animal life shall not be significantly changed" (Ministry of Environment and Energy 1995). Mean temperature in the Upper and Middle zones was 14.5 ^{o}C and16.8^{o}C, respectively (Fig. 12). At Meadowvale Wetland, the only site sampled in the Lower zone, temperature was 24.5^{o}C. The dramatically higher temperature in the Lower watershed is likely the result of sampling at only one site.

Electrical conductivity refers to the ability of water to pass an electrical current (U.S. EPA 1997). This is affected by the concentrations of inorganic dissolved solids such as chloride, nitrate, sulphate, and phosphate anions and sodium, magnesium, calcium, iron, and aluminum cations in the water. Conductivity increases with increasing water temperature and is therefore reported as conductivity at 25^{o}C (U.S. EPA 1997). The ODWS's and PWQO's do not set guidelines for electrical conductivity. Harter (2003) indicates that, in the U.S., water with conductivity above 0.83 microSiemens per centimetre cubed (mS/cm) is not recommended for drinking water and water with conductivity above 2.5 - 4 mS/cm is problematic for irrigation. This is because at high levels of conductivity water becomes saline, which is undesirable for drinking and stressful for plants with low to medium salt tolerance. These limits are indicated in Fig. 10 to provide a reference point for the conductivity measured at wetland sites. Conductivity generally ranged from 0-1 mS/cm (Fig. 13). The very high conductivity reading at Meadowvale, 12.2 mS/cm, may have been caused by measurement error in the field or due to pollution from road salting, as the piezometer is located in close proximity to residential roads. Unfortunately, laboratory analyses were not completed at Meadowvale. This would have provided information on the chloride concentration of the water which could be used to confirm if road salt pollution was an issue at this site.

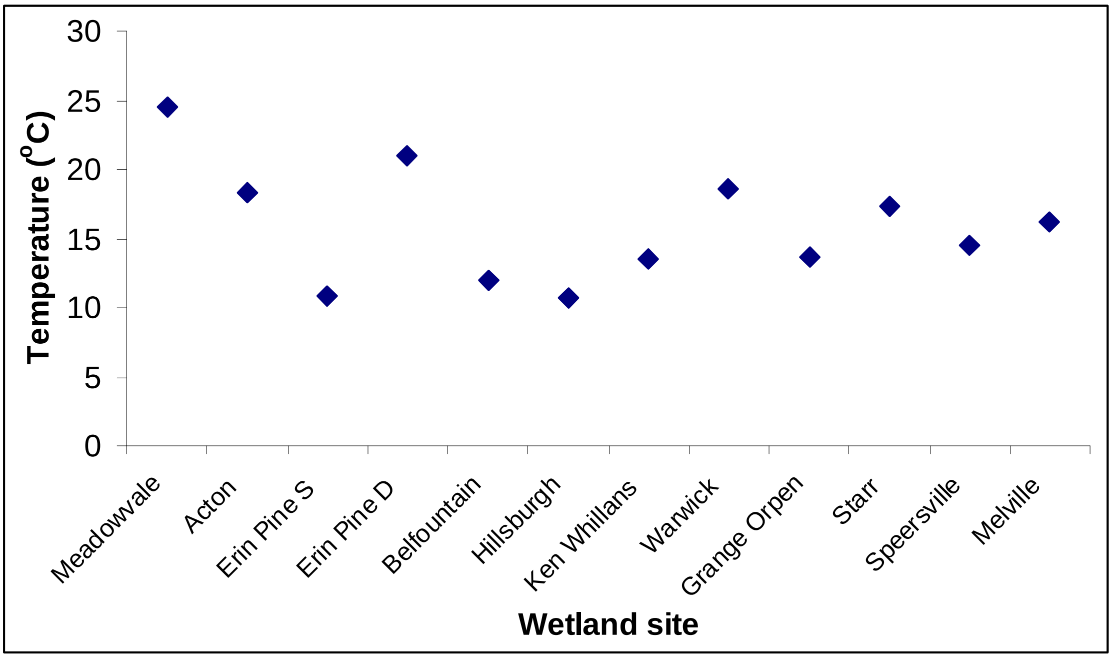


Figure 12. Groundwater temperature at monitored wetlands. Erin Pine S, Erin Pine Estates shallow piezometer; Erin Pine D, Erin Pine Estates deep piezometer.

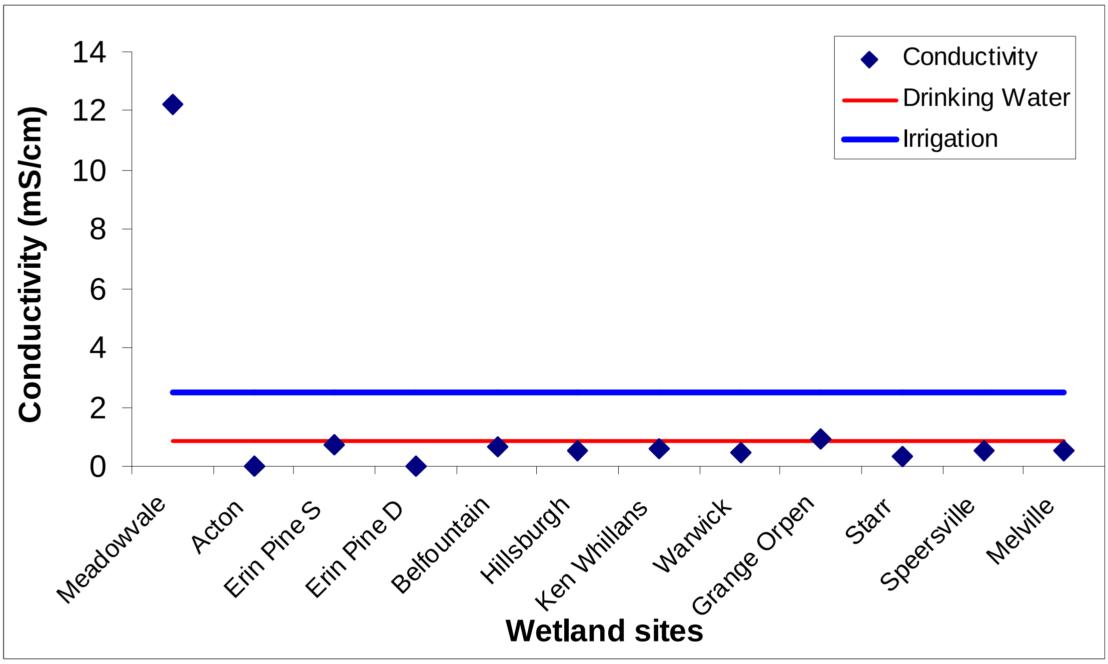


Figure 13. Groundwater conductivity at monitored wetlands. Erin Pine S, Erin Pine Estates shallow piezometer; Erin Pine D, Erin Pine Estates deep piezometer.

3.1.2 Surface Water

3.1.2.1 Laboratory Analyses: Laboratory analyses were completed for surface water at Belfountain, Warwick and Speersville Wetlands. The majority of the parameters in Table 3 were not detected (Appendix B). None of the ODWS guidelines were exceeded in sampled surface water. The PWQO guidelines were equalled or exceeded for four parameters – zinc, iron, aluminum and total phosphorus. The 75 μg/L guideline for aluminum was met at Warwick. Although aluminum is the third most abundant crustal element, it serves little purpose in the functioning of biotic systems (Gensemer and Playle 1999). In aquatic systems, aluminum toxicity increases as water becomes more acidic. Aluminum is toxic to all forms of aquatic life, from macrophytes to invertebrates. Therefore, it can be responsible for the demise of aquatic communities, especially in acidic conditions (Gensemer and Playle 1999). Also at Warwick, iron was present in concentrations over three times the 300 μg/L guideline. Similar to other metals, iron concentrations in water are elevated from natural levels through anthropogenic factors (e.g. mining and processing iron and metal fabricating). Iron is an essential element to all organisms and is used in a variety of biological processes (Ministry of the Environment 1979). Iron is moderately toxic to plants and only minimally toxic to mammals. Iron can lead to a decrease in water pH and an increase in turbidity through the formation of flocs (small, loose masses of fine particles suspended in a solution). This can smother bottom fauna, reduce primary productivity and affect species which rely on visual cues (Ministry of the Environment 1979). Zinc exceeded the 20 μg/L limit at Speersville by 18 μg/L. Zinc is also an essential nutrient for plants and animals in that it is necessary for formation of DNA and RNA, growth and reproduction. Fish appear to be the most sensitive aquatic organisms to zinc, high levels of which can lead to reduced egg production and mortality (Ministry of the Environment 1979). Total phosphorus exceeded both the 20 and 30 μg/L guidelines at Belfountain and Warwick Wetlands.

3.1.2.2 Field Analyses: In surface water samples, pH ranged from 6.6 to 8.2 (Fig. 14). Surface water DO concentrations ranged from 1.98 to 10.69 mg/L (Fig. 15), while temperature ranged from 10.8oC to 23.2oC (Fig. 16).Conductivity was typically below 1 mS/cm (Fig. 17). At Meadowvale conductivity reached nearly 3.5 mS/cm, although this may have been elevated for similar reasons as mentioned for groundwater. The PWQO guidelines are also not applicable in wetland surface waters as they may be inappropriately high for wetland systems, which tend to have naturally low DO concentrations (U.S. EPA 2009; Minnesota Pollution Control Agency 2009). In order to deal with these naturally anoxic conditions, wetland biota, ranging from unicellular organisms to plants and animals, have adapted to deal with the low levels of oxygen (Mitsch and Gosselink 2007). These adaptations include biochemical, physiological, structural and behavioural changes. For example, fauna may modify respiratory pigments to improve oxygen carrying capacity in order to control gas exchange in anoxic conditions (Mitsch and Gosselink 2007). It is important to note that DO and temperature were sampled as *insitu* spot measurements. Dissolved oxygen is highly affected by temperature and season. In addition, in areas with aquatic macrophytes oxygen levels are also affected by photosynthesis with higher DO levels during the day and lower levels at night during respiration. Macrophytes can also affect DO levels in a diel cycle, with higher oxygen levels during the growing season and low levels later in the season during vegetation dieback and decomposition (Canadian Council of Ministers of the Environment 1999). Temperature is also highly variable over the duration of a day and throughout seasons. Although, all measurements were taken within 4 days of each other, time of day varied and could significantly affect the measured values of both parameters.

It is important to remember that currently, no guidelines exist for wetland water quality. The PWQO guidelines were developed in order to ensure water quality levels that protect all forms of aquatic life during all aquatic life cycles for surface waters (e.g. lakes and rivers) (Ministry of Environment and Energy 1995). The ODWS guidelines were developed to protect public health through the provision of safe drinking water (Ontario Ministry of the Environment, 2003). Therefore, neither set of guidelines are explicitly applicable to a wetland ecosystem, however, the ODWS's will be used for groundwater samples and the PWQO's will be used for surface water samples to provide an initial approximation of site water quality (Table 4).

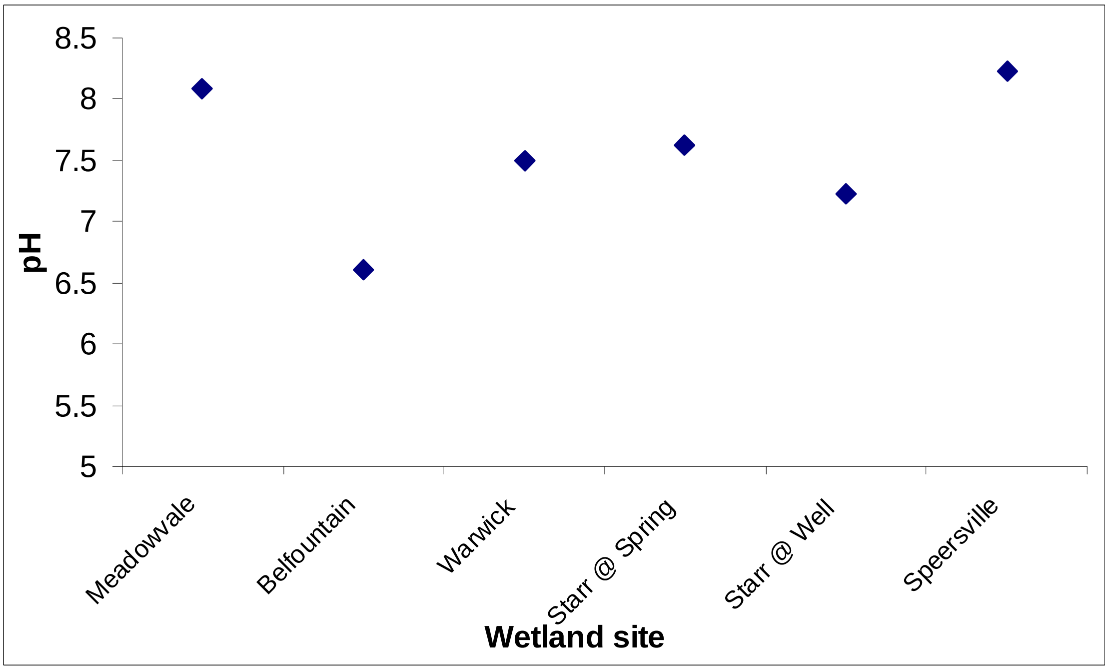


Figure 14. Surface water pH at monitored wetlands. Starr @ Spring, Starr Wetland sampled at flowing spring; Starr @ Well, Starr Wetland sampled at piezometer.

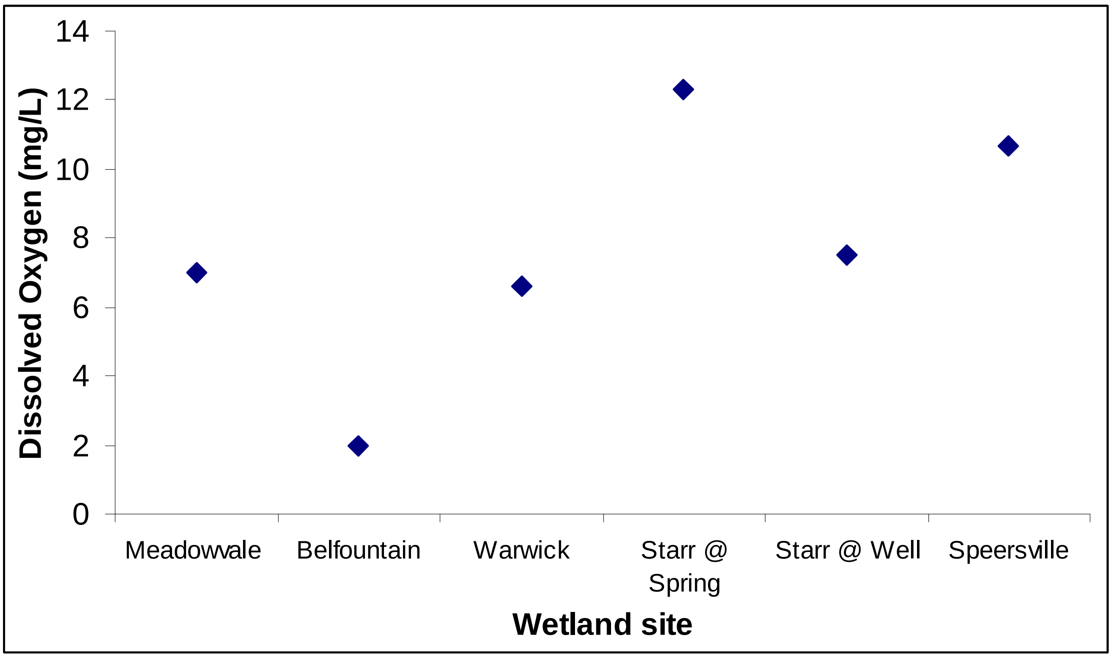


Figure 15. Surface water Dissolved Oxygen (DO)concentrations at monitored wetlands. Starr @ Spring, Starr Wetland sampled at flowing spring; Starr @ Well, Starr Wetland sampled at piezometer.

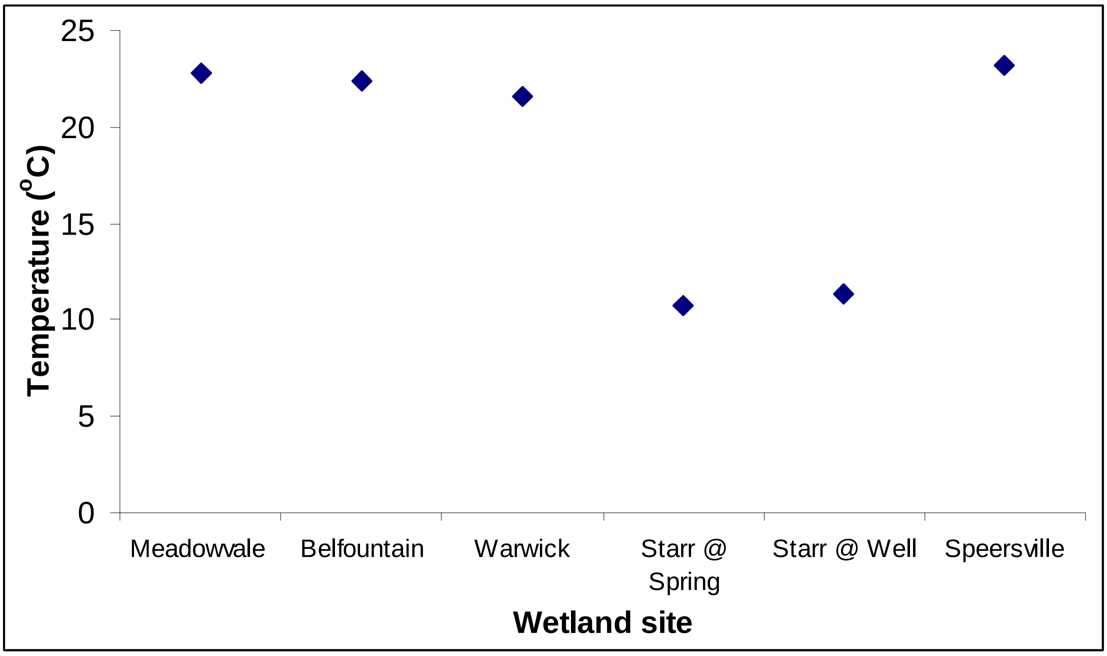


Figure 16. Surface water temperature at monitored wetlands. Starr @ Spring, Starr Wetland sampled at flowing spring; Starr @ Well, Starr Wetland sampled at piezometer.

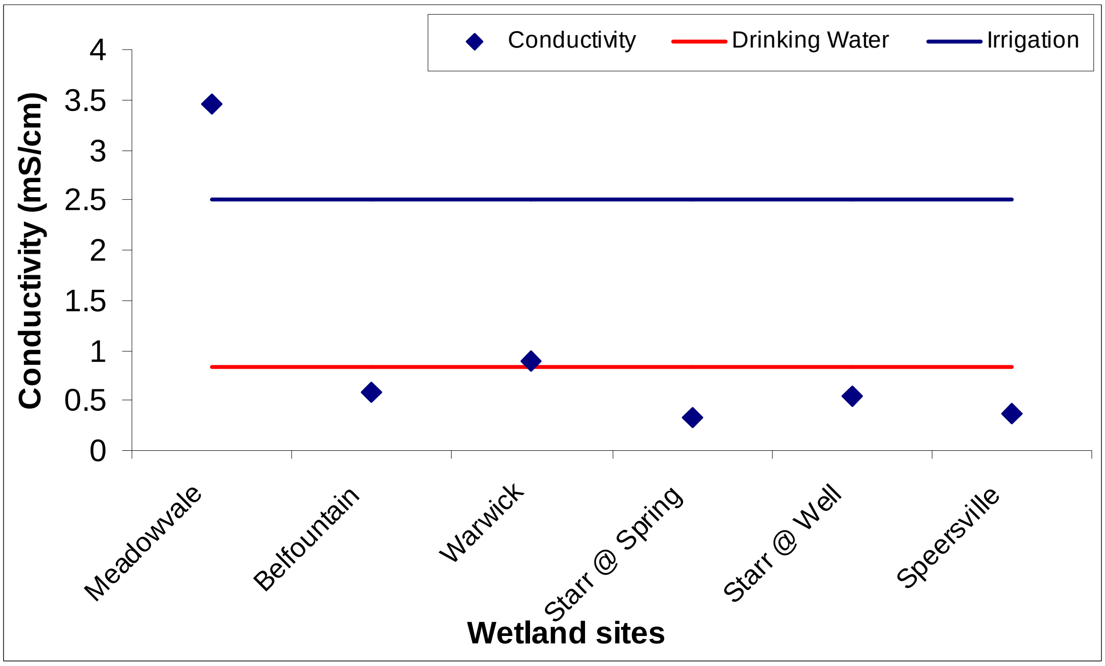


Figure 17. Surface water conductivity at monitored wetlands. Starr @ Spring, Starr Wetland sampled at flowing spring; Starr @ Well, Starr Wetland sampled at piezometer.

Table 4. Exceedance of PWQO and ODWS Guidelines for inorganics and metals in ground and surface water at wetland sites in the Credit River Watershed.

Site #	Site Name	Physiographic Region	Guidelines Not Met					
			Groundwater		Surface Water		Total	
			PWQO	ODWS [a]	PWQO	ODWS	PWQO	ODWS
W-04	Meadowvale Wetland	Lower	TP				1	0
W-13	Hillsburgh Wetland	Upper	TP				1	0
W-19	Melville Wetland	Upper	TP				1	0
W-15	Grange Orpen Wetland	Upper	TP				1	0
W-20	Belfountain Wetland	Upper	TP		TP		2	0
W-09	Ken Whillans Wetland	Middle	Cu, TP				2	0
W-16	Starr Wetland	Upper	V, Ag, TP				3	0
W-07	Acton Wetland	Middle	U, Co, Ag, TP				4	0
W-10	Warwick Wetland	Middle	TP		Al, Fe, TP		4	0
W-11	Erin Pine Estates Wetland	Upper	U, As, V, TP				4	0
W-17	Speersville Wetland	Upper	U, Ag, TP	U	Zn		4	1

[a] Blue shaded cells indicate the guideline upon which an initial approximation of site water quality should be based.

3.2 HYDROLOGY

3.2.1 Wetland Hydrologic Classification

Wetlands can be classified in many different ways including by vegetation, geographical location and hydrology. Hydroperiod describes the seasonal water level patterns of a wetland. This can be used as a broad classification tool, grouping wetlands with similar flooding regimes. Non-tidal wetlands may be placed into one of the seven hydroperiod categories listed in Table 5. Hydroperiod is important because it directly modifies the physiochemical environment of a wetland, altering aspects such as oxygen availability, soil and water chemistry, nutrients, toxicity and sedimentation. These alterations then directly impact wetland biota (Fig. 2) through changes in species composition, richness and ecosystem productivity (Mitsch and Gosselink 2007). For example, significant changes to water levels in a wetland may affect the ability of submergent or emergent vegetation to survive. If water levels drop significantly, it may be impossible for submergent aquatic vegetation to persist.

Table 5. Hydroperiod categories for freshwater wetlands (Mitsch and Gosselink 2007).

Hydroperiod Category	Description
Permanently Flooded	- flooded throughout the year in all years
Intermittently Exposed	- flooded throughout the year except in years of extreme drought
Semipermanently Flooded	- flooded during the growing season in most years
Seasonally Flooded	- flooded for extended periods during the growing season, but usually no surface water by the end of the growing season
Saturated	- substrate is saturated for extended periods during the growing season, but standing water is rarely present
Temporarily Flooded	- flooded for brief periods during the growing season, but water table is otherwise well below surface
Intermittently Flooded	- surface is usually exposed with surface water present for variable periods without detectable seasonal pattern

Wetlands can also be classified according to their groundwater patterns (Mitsch and Gosselink 2007). In some wetlands, groundwater plays an important role in regulating water levels. In others, surface water is the major contributor to the wetland. Wetland groundwater interactions can occur in two ways: discharge and recharge. Groundwater discharge occurs when the groundwater feeds the wetland; this usually takes place when the wetland is hydrologically lower than the groundwater table. Recharge occurs when water from wetlands is released into the groundwater (Mitsch and Gosselink 2007). Freshwater wetlands may be classified as one of the four wetland types described in Table 6.

Table 6. Groundwater flow pattern classification for freshwater wetlands (Mitsch and Gosselink 2007).

Groundwater Flow Pattern Classification	Characteristics
Surface Water Depression Wetland	- Dominated by surface runoff and precipitation - Little groundwater outflow due to low-permeability soils - Limited groundwater recharge possible - Water table usually below wetland
Surface Water Slope Wetland	- Generally found adjacent to a lake or stream - Fed predominantly by overbank flooding from adjacent water body - Some inflow from runoff and precipitation - Some groundwater recharge possible, but this usually discharges shortly back to water body - Water table usually below wetland
Groundwater Depression Wetland	- Wetland in depression low enough to intercept the groundwater table - Water-level fluctuations relatively less dramatic than surface fed wetlands due to the relative stability of groundwater levels
Groundwater Slope Wetland	- Located on slopes where groundwater discharges to surface through seeps or springs - Groundwater discharge to wetland can be continuous or seasonal depending on geohydrology and evapotranspiration rates

3.2.2 Site-specific Wetland Hydrological Classifications

According to the three years of data acquired through measurement of surface and groundwater levels, the studied wetlands were tentatively classified by hydroperiod and groundwater flow pattern (Table 7). However, these classifications are used with caution because of limited data and piezometer placement. Only three years of water level data are currently available. Within that time frame, there were both relatively hot-dry and cool years, which may skew results from the typical hydrological patterns of the sites (Fig. 18). In addition, piezometers were not consistently placed within the wetland hydrological gradient (e.g. at the wettest position in the wetland) or along the sampling transect. Consequently, it becomes very difficult to accurately assign a hydroperiod classification and compare between wetlands. As more data become available, these classifications will be updated. Annual piezometer water levels at each monitored wetland are displayed graphically in Appendix C and summarized in Appendix D, while Appendix E contains maps and photos of each site.

Table 7. Hydroperiod classifications of wetland monitoring sites in the Credit River Watershed.

Site #	Site Name	Hydroperiod Classification	Groundwater Flow Pattern
W-01	Rattray Marsh Wetland	Seasonally Flooded	Surface Water Slope
W-04	Meadowvale Wetland	Permanently Flooded	Surface Water Slope
W-07	Acton Wetland	Seasonally Flooded	Groundwater Depression
W-09	Ken Whillans Wetland	Temporarily Flooded	Surface Water Depression
W-10	Warwick Wetland	Permanently Flooded	Surface Water Slope
W-11	Erin Pine Estates Wetland	Saturated	Surface Water Slope
W-13	Hillsburgh Wetland	Saturated	Surface Water Slope
W-15	Grange Orpen Wetland	Saturated	Surface or Groundwater Slope
W-16	Starr Wetland	Saturated	Groundwater Slope
W-17	Speersville Wetland	Seasonally Flooded	Surface Depression
W-19	Melville Wetland	Saturated	Surface Water Slope
W-20	Belfountain Wetland	Seasonally Flooded	Surface Water Depression

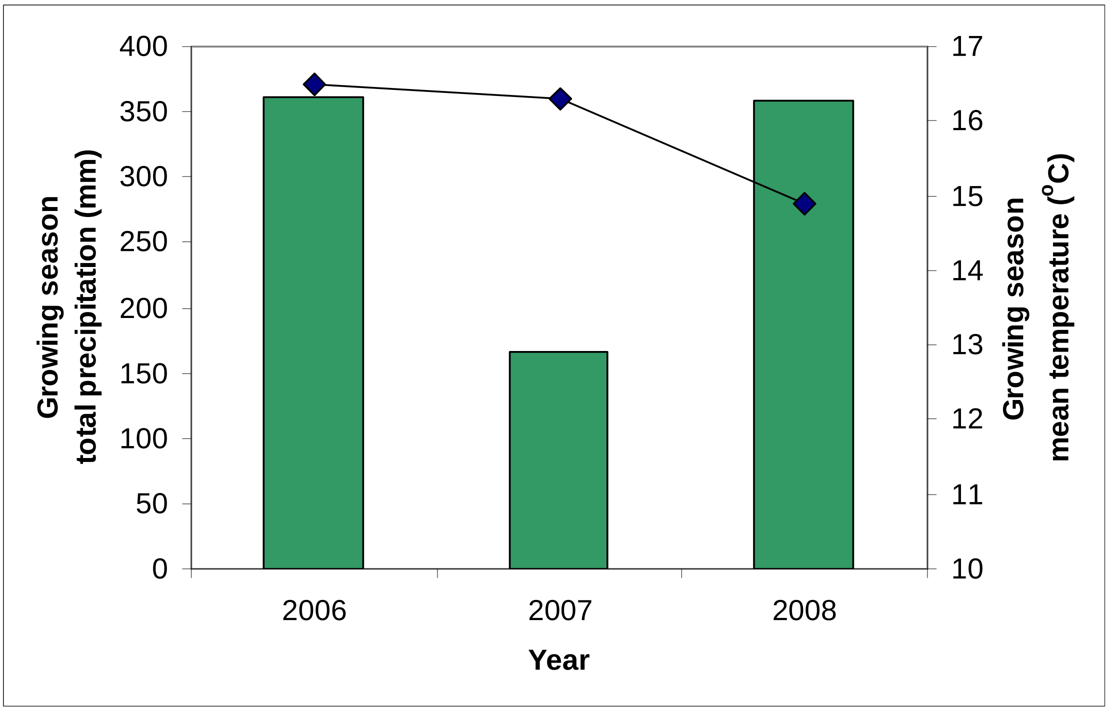


Figure 18. Total monthly precipitation (bars) and mean monthly temperature (points) from 2006 to 2008 in Georgetown, ON during the growing season (April – September). Mean temperature and precipitation over the growing season were 16.5°C and 360.7 mm, 16.3°C and 166.5 mm, 14.9°C and 358.0 mm in 2006, 2007 and 2008, respectively.

3.2.2.1 <u>Lower Watershed:</u> The Rattray Marsh Wetland is classified as a Surface Water Slope Wetland, as it is located on the shore of Lake Ontario. Although the hydrograph indicates groundwater discharge throughout the 2008 season, this may not be characteristic of the flow pattern of the marsh. Groundwater discharge in 2008 may have been due to high groundwater levels observed throughout the watershed during that year (Appendix C). However, it is not uncommon for some groundwater discharge to occur in

Surface Water Slope Wetlands (Table 7). Therefore, additional years of data are required to confirm the groundwater flow pattern classification at this site. Although a large portion of Rattray Marsh Wetland is permanently flooded, the monitoring site is located on the edge of the wetland were standing water is typically absent in late summer. It has been classified as Seasonally Flooded until more data are available to confirm these visual observations.

Meadowvale Wetland is also deemed a Surface Water Slope Wetland because it is located in the floodplain of the Credit River. Certain parts of this wetland have standing water throughout the year; therefore, it is classified as a Permanently Flooded wetland.

3.2.2.2 Middle Watershed: Acton Wetland is a swamp that is classified as Seasonally Flooded. There is an ephemeral stream running through the site during the early growing season, but by late summer and fall, standing water is usually absent. It appears from the hydrograph that Acton Wetland is fed by groundwater (Appendix C) and is therefore classified as a Groundwater Depression Wetland.

Ken Whillans Wetland is also a swamp located in the Middle watershed. This swamp has standing water for short periods during the growing season and is classified as Temporarily Flooded. In contrast to Acton Wetland, this swamp appears to be fed by surface water inflow, such as overland flow and precipitation, and is deemed a Surface Water Depression Wetland.

Warwick Wetland is flooded throughout the year, even in dry years such as 2007. Accordingly, it is classified as a Permanently Flooded wetland. The piezometer at this site was not functioning properly, possibly because a tight seal was not formed around the piezometer due to the high level of organics at the site. Therefore, accurate groundwater levels were not obtained at this wetland and, currently, it is not possible to determine the groundwater flow patterns at this site. However, this wetland is adjacent to a large lake and, as such, is tentatively classified as a Surface Water Slope Wetland. Reinstallation of the piezometer at Warwick Wetland is planned, and future data will be used to confirm this initial groundwater flow pattern classification.

3.2.2.3 Upper Watershed: Erin Pine Estates, Hillsburgh, Grange Orpen and Melville Wetlands all have similar hydroperiods and groundwater flow patterns. Standing water is rarely present at these four sites, but the substrate is usually saturated (Appendix C). Accordingly, all four wetlands were classified as Saturated. Standing water in 2006 and 2007 at Melville was due to a beaver dam that was altering water levels from their normal patterns. The dam was removed, however 2008 was an exceptionally wet year and a representation of the wetland's natural condition may not be observed until future years. Future data will assist in confirming this. In addition, Erin Pine Estates, Hillsburgh and Melville are all adjacent to water bodies, which influence the water levels of the wetlands. These wetlands are classified as Surface Water Slope Wetlands. Grange Orpen is also situated near a water body; however, it is not clear if this wetland is fed by ground or surface water. More information is needed to confirm if groundwater discharge plays an important role in the hydrology of this site.

At Starr Wetland, groundwater seeps are the major inflow of water, clearly indentifying it as a Groundwater Slope Wetland. Similar to the other previously mentioned wetlands in this region, the ground at Starr tends to be Saturated with little standing water (Appendix C).

It was difficult to assign classifications to Speersville Wetland because, similar to Warwick Wetland, the piezometer was not working properly. The site is situated on land that was previously an airport with engineered filled. This engineered fill may have been affecting piezometer readings. Despite the fact that the hydrograph indicates that the wetland is seasonally flooded (Appendix C), visual observations indicate that the wetland is permanently flooded. The wetland was classified as Seasonally Flooded until more reliable data are available. For these same reasons, it was not possible to determine groundwater discharge to the wetland. Visual observations indicated that Speersville Wetland may be a Surface Depression wetland. It is hoped that new data from the piezometer in subsequent years will allow more accurate classification of this site.

Finally, Belfountain is a Seasonally Flooded wetland. It is classified as a Surface Depression wetland because groundwater does not appear to play an important role of the hydrology at this site.

3.2.2.4 Watershed-wide: Four of the seven hydroperiod classes (Table 5) appear to be represented in the watershed, with Saturated wetlands being the most common. All four of the groundwater flow patterns (Table 6) are represented in monitored wetlands; however, wetlands fed through surface water inputs appear more dominant than groundwater fed wetlands. It is unclear whether surface water wetlands are more dominant in the watershed or if surface water wetlands are disproportionately represented in monitored wetlands.

From examining the hydrographs in Appendix C it appears that Belfountain, Meadowvale and Speersville Wetlands consistently had the most surface water, while Erin Pine Estates, Hillsburgh and Grange Orpen Wetlands tended to be the driest sites. Monitored wetlands tended to be wettest in the spring and early summer (typically April – June), which is consistent with snow melt and high rainfall during these times. Wetlands were driest in late summer and fall (July – August or September) due to higher temperatures and lower rainfall. It is important to note that 2007 was a relatively dry year and 2008 was a relatively cool year (Fig. 18). This was reflected in piezometer measurements. Groundwater levels tended to be lower in 2007 compared to 2006, especially in the late summer. This is likely due to reduced recharge resulting from low rainfall. Throughout the 2008 growing season, groundwater levels tended to be higher than the previous two years. Although 2008 was not wetter than normal, it was relatively cooler, which may have led to a decrease in evapotranspiration. As more data become available with additional years of monitoring, hydrological classifications of the monitored wetlands will be refined and significant trends in changes in hydrology will be elucidated.

4.0 CURRENT PROGRAM ADVANCEMENTS

Preliminary results based on analysis of three years of existing groundwater data make it clear that in order to better understand the hydrology of the Credit River watershed, and its large influence on shaping plant and animal communities across the landscape, this component of the program would have to be expanded. Proposed expansions to the program include the following:

- Establishment of eleven additional permanent plots to achieve a more representative sample of wetland communities (marsh versus swamp) in the watershed.
- Elevation mapping of all monitored wetlands (required in order to determine optimal placement of hydrological monitoring equipment).
- Characterization of monitored wetland communities according to Ecological Land Classification principles.
- Installation of three piezometers at all wetland monitoring plots to monitor groundwater levels and movement throughout the site.
- A staff gauge will be installed at all wetland sites to measure surface water depth and assist in the determination of hydroperiod.
- Data loggers will accompany installation of piezometers and staff gauges to increase sampling frequency and accuracy without increasing disturbance to the sensitive wetland communities.
- Commitment to long-term monitoring and analysis of wetland hydrology to ensure that extreme weather years are not used to establish baseline levels.

The above proposed changes to monitoring protocols focus on increasing understanding of wetland hydrology. Additional information collected from wetland hydrology monitoring will complement data already collected on flora and fauna indicators at the permanent wetland plots. This will allow the program to link observed trends to understand the relationships between water quality, wetland hydrology, plant and animal communities more holistically.

The proposed Wetland Hydrology component of the Terrestrial Monitoring Program has been broken up into 4 phases to ensure the workload and cost is reasonable over time. The phases are as follows:

<u>Phase 1</u> (August 2010-December 2010)
1. Selection of new wetland plots to ensure adequate representation of wetland types across the watershed, and adequate statistical power (11 additional sites)
2. Development and refinement of wetland hydrology methodology with water department
3. Completion of ELC at all established and proposed wetland plots
4. Purchase of hydrology monitoring equipment for 9 wetland plots

Phase 2 (January 2011-December 2011)

1. Completion of elevation surveys at all established and proposed wetland sites (to be completed in Phase 1 if possible)
2. Selection of 9 permanent wetland plots for piezometer and staff gauge installation
3. Installation of piezometers and staff gauges and accompanying data loggers at 9 wetland sites based on elevation survey results

Phase 3 – optional expansion of program to all wetland plots (August 2011-August 2012)

1. Selection of 10 additional permanent wetland plots for piezometers and staff gauge installation
2. Installation of piezometers and staff gauges at 10 wetland sites based on elevation survey results

Phase 4 - optional expansion of program to all wetland plots (August 2012-August 2013)

1. Installation of piezometers and staff gauges at 11 remaining permanent wetland monitoring sites based in elevation survey results
2. Wetland water quality surveys will be conducted at all permanent plots (TBD)

5.0 PROGRAM RECOMMENDATIONS

5.1 WATER QUALITY

5.1.1 Water Quality Index

If wetland water quality is to be monitored in the Credit River Watershed, it is essential that a suite of parameters be selected for monitoring that will not only provide a comprehensive assessment of water quality, but also be economically and operationally feasible and allow for the comparison of water quality between sites and over time. Chow-Fraser (2006) developed a WQI for coastal wetland surface waters as an environmental indicator of wetland health in response to degradation of costal marshes in the Great Lakes due to land use changes.

This WQI was originally developed using 12 water quality parameters (Table 8). These parameters were chosen for ease of measurement and their potential for indicating unique information (Chow-Fraser 2006). Results of analyses of these parameters are entered into a predictive equation which provides the WQI score. This 12-parameter model (Eq. 1, Table 9) describes the total variation in WQI scores and can be replaced instead with a number of other equations which use some combination of four to seven of the parameters in Table 8. Equations 2-6 (Table 9) have uniformly high r^2-values and should generate comparable WQI scores. Equations 7 -9 have lower r^2-values, but are included because they may be more feasible for some monitoring programs because they include only four or five very commonly measured parameters. Water Quality Index scores calculated with one of the equations in Table 9 are interpreted according the WQI score descriptions (Table 10). Water Quality Index scores range from -3 to +3 with positive scores indicating good water quality and negative scores indicating degraded water (Chow-Fraser 2006).

This WQI has been used to track the improved health of Cootes Paradise Marsh over the course of a marsh-wide carp exclusion program as part of the Hamilton Harbour Remediation Action Plan (Chow-Fraser 2006). Improvements in the WQI score over the course of this program were consistent with improvements in the health of zooplankton, plant and fish communities observed over the same time. The WQI was also effective at ranking wetlands according to water quality across the entire Great Lakes Basin, producing results that were comparable to published biotic indicators (Chow-Fraser 2006). This WQI was also significantly related to a wetland fish index and a wetland zooplankton index, both of which were also developed to determine the quality of coastal wetlands (Lougheed and Chow-Fraser 2002; Seilheimer and Chow-Fraser 2004). This indicates that the WQI provides a good approximation of wetland health in relation to other estimators.

Table 8. Summary of water quality variables included in the original Water Quality Index (WQI) (Chow-Fraser 2006).

Variable	Acronym	Unit	Analysis Method
Turbidity	TURB	NTU	*in situ*
Temperature	TEMP	^{o}C	*in situ*
pH	pH	N/A	*in situ*
Conductivity	COND	µS/cm	*in situ*
Chlorophyll-a	CHL	µg/L	Laboratory
Total suspended solids	TSS	mg/L	Laboratory
Total inorganic suspended solids [a]	TISS	mg/L	Laboratory
Total phosphorus	TP	µg/L	Laboratory
Soluble reactive phosphorus	SRP	µg/L	Laboratory
Total ammonium nitrogen	TAN	µg/L	Laboratory
Total nitrate nitrogen	TNN	µg/L	Laboratory
Total nitrogen [b]	TN	µg/L	Laboratory

[a] This is a calculated parameter. Volatile suspended solids test is needed for TISS calculation.
[b] This is a calculated parameter. Total Kjehldahl nitrogen (TKN) test needed for TN calculation.

Table 9. Summary of regression equations to predict WQI scores (Chow-Fraser 2006).

Eq. #	Variables in model	Associated r^2 value	Predictive Equation
1	TURB, TSS, TISS, TP, SRP, TAN, TNN, TN, COND, TEMP, pH, CHL	1.00	+10.0239684 - 0.3154965 * log TURB -0.3656606 * logTSS -0.3554498 * log ISS -0.3760789 * log TP -0.1876029 * log SRP -0.0732574 * log TAN -0.2016657 * log TNN -0.2276255 * log TN -0.5711395 * log COND -1.1659027 * log TEMP -4.3562126 * log pH -0.2287166 * log CHL
2	TURB, TSS, TP, COND, TN	0.965	+5.2427978 -0.298509 * log TURB -0.865436 * log TSS -0.626229 * log TP -0.818190 * log COND -0.330760 * log TN

Eq. #	Variables in model	Associated r^2 value	Predictive Equation
3	TURB, COND, TEMP, PH, TP, TN, CHL	0.964	+10.753047 -0.946098* log TURB -0.837294 * log COND -1.319621 * log TEMP -4.604864 * log pH -0.387189 * log TP -0.353713 * log TN -0.337888 * log CHL
4	TP, TN, SRP, TNN, TAN, TSS, CHL	0.963	+3.8311461 -0.629834 * log TP -0.271059 * log TN -0.083724 * log SRP -0.211261 * log TNN -0.119190 * log TAN -0.995406 * log TSS -0.243290 * log CHL
5	TURB, COND, TEMP, PH, SRP, TNN, TAN	0.947	+11.88597 -1.147966 * log TURB -1.048255 * log COND -2.308968 * log TEMP -4.653771 * log pH -0.278112 * log SRP -0.324002 * log TNN -0.116383 * log TAN
6	TURB, COND, TEMP, PH, TP, TN	0.947	+11.590154 -1.073765 * log TURB -0.916011 * log COND -1.684796 * log TEMP -4.677050 * log pH -0.599127 * log TP -0.306512 * log TN
7	TURB, COND, TEMP, PH	0.898	+9.2663224 -1.367148 * log TURB -1.577380 * log COND -1.628048 * log TEMP -2.371337 * log pH
8	TP, TN, COND, CHL	0.867	+5.2333056 -0.832012 * log TP -0.313032 * log TN -0.982628 * log COND -0.583014 * log CHL
9	TP, TAN, TNN, TN, CHL	0.853	+3.5161294 -0.985870 * log TP -0.195332 * log TAN -0.261192 * log TNN -0.171508 * log TN -0.599259 * log CHL

Table 10. Water Quality Index score descriptions (Chow-Fraser 2006).

WQI Score	Category
+3 to +2	Excellent
+2 to +1	Very Good
+1 to 0	Good
0 to -1	Moderately Degraded
-1 to -2	Very Degraded
-2 to -3	Highly Degraded

5.1.2 Recommendations

When adding a new component to a monitoring program it is important to ensure that the component is consistent with the goals of the monitoring program and that it functions as an effective monitoring indicator. The main goals of the Terrestrial Monitoring Program are as follows:

1. Identify status and trends in the health of terrestrial communities at the watershed scale and link to watershed health.
2. Identify spatial patterns in terrestrial community health parameters.
3. Provide meaningful data on which watershed management decisions can be based.

Water quality, using the WQI, may be a beneficial wetland monitoring indicator to add to the wetland monitoring program. The WQI will allow both temporal and spatial trends to be identified that may be useful in developing watershed management strategies. Water Quality Index component metrics are easy to measure and understand and do not require a high level of technical or biological expertise to monitor. The WQI provides key information about the structure, function and/or composition of the ecosystem, is quantifiable in a manner that is suitable for statistical analysis and is known to have relationships with other terrestrial variables. In addition, as shown by Chow-Fraser (2006) the WQI is sensitive to change and anthropogenic stresses. The use of this WQI would allow CVC to not only track changes in water quality at monitored wetland sites over time, but also compare water quality among sites. This would allow for comparisons between physiographic zones within the watershed and identify sites where water quality is of concern. This information would be complementary to the current data on wetland vegetation already being collected to asses the health of wetlands in the watershed. Unfortunately, this WQI was designed using coastal wetland ecosystems. It must first be determined if this is system is scientifically valid for inland, non-costal wetlands, before it can be used in this monitoring program.

With the consideration that the majority of wetlands monitored by the program are surface water fed and that the WQI was developed for wetland surface water, it seems reasonable to suggest that water quality sampling be completed for surface water only. This would be more costs effective for the program than completing both ground and surface water sampling.

Using the 12-parameter model for determining WQI would theoretically be ideal, as it provides the most accurate WQI estimations. However, this may not be practical due to the number of parameters being analyzed which would not only increase costs, but also the time to process samples. For the 12-parameter model, it would cost approximately $188.60 per sample in lab fees, not including the costs of the four *in situ* parameters that need to be analyzed (Table 11). Therefore, using one of the alternative models may be more practical. There are different benefits and drawbacks of using of each of the remaining equations. For example, Equation 7 uses only four parameters, all of which can be obtained by field measurements using portable equipment. This would eliminate sample processing and laboratory fees. Costs would include only those required to purchase (or borrow) the field equipment and required reagents/supplies. Purchasing equipment to take the four required *in situ* measurements would range in cost. The purchase of a Quanta T system, which would allow the measurement of all four *in situ* parameters using one device, is estimated at approximately $6000 (Appendix F); however, other options at lower costs ($1000-$2000) are also available (Appendix F). In contrast, Equations 4 and 9 both require no in situ measurements, with only laboratory samples required. Laboratory fees would range from $147.65 to $173.10 per sample for these equations. Although laboratory costs are high for these equations, no additional equipment would be needed to be purchased or borrowed for *in situ* analysis. The remaining equations use a combination of laboratory and *in situ* parameters. Budget and time constraints would need to be examined closely in order to determine which equation would be most suitable for the program's needs.

Table 11. Estimated laboratory costs for each WQI equation. Prices do not include cost of measuring *in situ* parameters. [a]

Equation	Number of Laboratory Parameters	Number of *in situ* parameters	Cost per sample	Annual sampling program	Biannual sampling program	Bimonthly sampling program
1	10	4	$188.60	$3017.60	$6,035.20	$36,211.20
2	3	2	$39.20	$627.20	$1,254.40	$7526.40
3	3	4	$130.95	$2095.20	$4,190.40	$25,142.40
4	7	0	$173.10	$2769.60	$5,539.20	$33,235.20
5	3	4	$33.90	$542.40	$1,084.80	$6,508.80
6	2	4	$30.95	$495.20	$990.40	$5,942.20
7	0	4	$0.00	$0.00	$0.00	$0.00
8	3	1	$130.95	$2095.20	$4,190.40	$25,142.40
9	5	0	$157.65	$2522.40	$5,044.80	$30,268.80

[a] Costs estimated from individual analyses cost as per Appendix F.

Although purchasing *in situ* equipment has a high up-front cost, the initial investment may be cheaper than annual laboratory costs over the life of the program. However, the expected lifespan and replacement costs of the equipment must be considered. Also, the possibility of sharing equipment with other CVC departments may decrease costs if required equipment is available. It is imperative that the most appropriate equation is chosen and that the same equation is consistently applied spatially and temporally.

Although a WQI has been developed for monitoring wetland water quality, no protocols have been specifically developed for wetland water quality sampling. Bimonthly sampling has been recommended to obtain relatively accurate readings (Jen Dougherty, pers. comm.).This may not be feasible for wetland monitoring due to time and budgetary constraints. A more feasible option may be annual or bi-annual sampling carried out at the time of wetland vegetation monitoring visits. Water quality is difficult to quantify due to the ever changing state of the water. Annual sampling would provide merely a "snapshot" in time. In addition, standard procedures relating to the timing of sampling (e.g. during storm events, etc.) will need to be developed. Estimated laboratory costs for annual, biannual and bimonthly sampling programs are presented in Table 11. Additional information regarding options for water quality monitoring costs is available in Appendix F.

In addition to water quality monitoring within the wetland, future projects may also involve monitoring the hydrological and chemical ecosystem services that wetlands offer. Wetlands provide both water quality and quantity services by capturing overland flow, reducing flow velocity and providing the opportunity for water to infiltrate and evapotranspire. As a result, pollutants transported by the overland water are deposited or filtered in the wetlands. This provides sustainable flows and improved water quality down stream of the wetlands. In order to monitor these ecosystem services the following two objectives would need to be met:

1. Monitor the water quality and quantity services provided by the wetlands
2. Monitor how quality of water received by the wetlands impact their ecosystem

To meet both the objectives it would be important to monitor water quality and quantity upstream of the wetlands, in the wetlands, and down stream of the wetlands.

Although valuable information is sure to be gained from such a project, this is a very large undertaking and would require the collaborative effort of several CVC groups, including Terrestrial Monitoring, Water and Restoration. A partnership among these groups would be essential to the success of such an undertaking as the program would demand highly of funds and man hours.

5.2 HYDROLOGY

Studying the hydrology of all monitored wetlands in the watershed is beneficial because it assists in correctly classifying wetlands according to their hydrology, leading to a better understanding of each individual wetland system. The piezometers will also provide data to assist in the determination of temporal changes in wetland hydrology, as concerns exist that wetlands are becoming dryer with climate change and increasing urbanization. In addition, it is hoped that this information will provide a better understand the hydrology of the Credit River watershed, and its large influence on shaping plant and animal communities across the landscape.

It is believed that the current proposed expansions to the hydrology component of the monitoring program will generate the information needed to answer the many questions that exist surrounding the hydrology of wetlands within the Credit River

Watershed. However, before piezometer installation occurs in 2011 it is recommended that a detailed methodology for the hydrology component be developed. This would include the rationale for the program, methodology for piezometer placement and installation as well as detailed monitoring methods. This will ensure that resources are used in the most optimal fashion to acquire high quality data and that hydrological monitoring will be performed consistently into the future.

6.0 CONCLUSIONS

Under the current monitoring configuration and the available data based thereon, few conclusive results can be drawn regarding the water quality or hydrology of wetlands in the watershed. Speersville and Warwick Wetlands appear to have the worst water quality of sampled wetlands. However, this was determined using the ODWS and PWQO guidelines which are not designed specifically for wetland water quality. If wetland water quality is a desired component for addition to the wetland monitoring program, it is suggested that a WQI developed for wetland surface water be employed in order to determine spatial and temporal changes in wetland water quality. However, this may be a costly addition to the program.

Wetland hydrology data provided enough information to begin preliminary hydrological classifications of each wetland. However, additional years of data under the new proposed wetland hydrology monitoring program will be required for definitive classifications. These data will also assist in discerning temporal changes in wetland moisture regime. Monitoring of wetland hydrology and water quality may provide information that is useful for determining temporal and spatial trends in wetland health in the Credit River Watershed. However, it is important that detailed goals and objectives be developed for hydrology monitoring (and water quality monitoring if adopted by the program) in order to use program resources in the most efficient manner, producing the highest-quality and most pertinent results.

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APPENDIX A: WETLAND MONITORING SITES

Table 1. Wetland Health Monitoring Sites established throughout the Credit River Watershed.

Site #	Site Name	Township	County	Wetland type	Physiographic zone	PSW, ESA and/or ANSI designation [a]	Habitat patch size (ha)
W-01	Rattray Marsh Wetland	Mississauga	Regional Municipality of Peel	Marsh	Lower	PSW, ESA, ANSI	92.27
W-02	Credit River Wetland	Mississauga	Regional Municipality of Peel	Marsh	Lower	ESA, ANSI	0.26
W-03	Creditview Wetland	Mississauga	Regional Municipality of Peel	Marsh	Lower	PSW, ANSI	5.38
W-04	Meadowvale Wetland	Mississauga	Regional Municipality of Peel	Marsh	Lower	ESA	42.29
W-12	Winston Churchill Wetland	Halton Hills	Regional Municipality of Peel	Marsh	Lower		114.06
W-06	Hungry Hollow Wetland	Halton Hills	Regional Municipality of Peel	Swamp	Middle	PSW, ESA	127.86
W-07	Acton Wetland	Halton Hills	Regional Municipality of Peel	Swamp	Middle	PSW, ESA	753.26
W-08	Terra Cotta Wetland	Caledon	Regional Municipality of Peel	Swamp	Middle	ESA, ANSI	165.92
W-09	Ken Whillans Wetland	Caledon	Regional Municipality of Peel	Swamp	Middle		191.18
W-10	Warwick Wetland	Caledon	Regional Municipality of Peel	Marsh	Middle	PSW, ESA, ANSI	93.12
W-11	Erin Pine Estates Wetland	Erin	Wellington County	Marsh	Upper	PSW	24.66
W-13	Hillsburgh Wetland	Erin	Wellington County	Marsh	Upper	PSW, ESA	14.37
W-15	Grange Orpen Wetland	Caledon	Regional Municipality of Peel	Swamp	Upper	PSW, ESA	137.59
W-16	Starr Wetland	Caledon	Regional Municipality of Peel	Marsh	Upper		291.34
W-17	Speersville Wetland	Caledon	Regional Municipality of Peel	Marsh	Upper	PSW, ESA	218.85
W-18	Caledon Lake Wetland	Caledon	Regional Municipality of Peel	Swamp	Upper	PSW, ESA, ANSI	554.82
W-19	Melville Wetland	Orangeville	County of Dufferin	Marsh	Upper	PSW	5.53
W-20	Belfountain Wetland	Caledon	Regional Municipality of Peel	Marsh	Upper		30.86

[a] PSW, Provincially Significant Wetland; ESA, Environmentally Sensitive Area; ANSI, Area of Natural and Scientific Interest.

APPENDIX B: WETLAND WATER QUALITY TESTING

Table 1. Water quality parameters tested at selected wetlands in the Credit River Watershed in June 2006.

Parameter	Unit	Parameter	Unit
Metals [a]		**Inorganics**	
Aluminum (Al)	ug/L	Dissolved Inorganic Carbon	mg/L
Antimony (Sb)	ug/L	Total Ammonia-N	mg/L
Arsenic (As)	ug/L	Total Carbonaceous BOD	mg/L
Barium (Ba)	ug/L	Total Chemical Oxygen Demand (COD)	mg/L
Beryllium (Be)	ug/L	Conductivity	umho/cm
Boron (B)	ug/L	Total Dissolved Solids	mg/L
Cadmium (Cd)	ug/L	Fluoride (F-)	mg/L
Chromium (Cr)	ug/L	Hardness (CaCO3)	mg/L
Cobalt (Co)	ug/L	Total Kjeldahl Nitrogen (TKN)	mg/L
Copper (Cu)	ug/L	Dissolved Organic Carbon	mg/L
Iron (Fe)	ug/L	Orthophosphate (P)	mg/L
Lead (Pb)	ug/L	pH	pH
Molybdenum (Mo)	ug/L	Total Phosphorus	mg/L
Nickel (Ni)	ug/L	Total Suspended Solids	mg/L
Selenium (Se)	ug/L	Sulphide	mg/L
Silver (Ag)	ug/L	Alkalinity (Total as CaCO3)	mg/L
Thallium (Tl)	ug/L	Nitrite (N)	mg/L
Tungsten (W)	ug/L	Chloride (Cl)	mg/L
Uranium (U)	ug/L	Nitrate (N)	mg/L
Vanadium (V)	ug/L	Nitrate + Nitrite	mg/L
Zinc (Zn)	ug/L	Sulphate (SO4)	mg/L
Zirconium (Zr)	ug/L	**RCAP Calculations** [b]	
Field Measurements		Bicarb. Alkalinity	mg/L
pH	pH	Carb. Alkalinity	mg/L
Conductivity	µS/cm	Hydrox. Alkalinity	mg/L
Dissolved Oxygen	mg/L	**Nutrients**	
Dissolved Oxygen	%	Dissolved Phosphorus (P)	ug/L
Temperature	^{o}C		

[a] Dissolved metal concentrations were analyzed for groundwater samples, while total metal concentrations were analyzed for surface water samples.
[b] Calculated as CaCO3

APPENDIX B: WETLAND WATER QUALITY TESTING (Cont'd)

Table 2. Wetland groundwater quality results for tested parameters with PWQO and/or ODWS guidelines [a].

Parameter	Units	ODWS Guideline	PWQO Guideline	W-07	W-09	W-10	W-11 S [b]	W-11 D [b]	W-13	W-15	W-16	W-17	W-19	W-20
Total Phosphorus	ug/L	N/A	20 – 30 [c]	**230**	**2500**	**200**	**44**	**81**	**420**	**160**	**94**	**4400**	**30**	**1500**
Dissolved Aluminum	ug/L	N/A	75	6	ND	ND	9	ND	ND	ND	ND	ND	ND	ND
Dissolved Antimony	ug/L	6	20	2	ND	ND	1	ND	ND	ND	ND	ND	ND	ND
Dissolved Arsenic	ug/L	25	5	2	3	ND	**8**	ND	ND	ND	ND	2	ND	ND
Dissolved Barium	ug/L	1000	1000	66	150	63	280	120	31	150	58	32	150	200
Dissolved Beryllium	ug/L	N/A	11	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Dissolved Boron	ug/L	5000	200	ND	27	24	15	17	ND	20	16	ND	48	10
Dissolved Cadmium	ug/L	5	0.2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Dissolved Chromium	ug/L	50	1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Dissolved Cobalt	ug/L	N/A	0.9	**3**	ND	ND	ND	ND	ND	ND	0.8	ND	ND	ND
Dissolved Copper	ug/L	N/A	5	2	**7**	ND	4	ND	ND	ND	ND	ND	ND	1
Dissolved Iron	ug/L	N/A	300	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Dissolved Lead	ug/L	10	5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Dissolved Molybdenum	ug/L	N/A	40	30	ND	2	8	ND	2	1	2	1	2	1
Dissolved Nickel	ug/L	N/A	25	8	ND	3	2	ND	ND	1	9	2	2	2
Dissolved Selenium	ug/L	10	100	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Dissolved Silver	ug/L	N/A	0.1	ND	ND	**0.2**	ND	ND	ND	ND	**0.2**	0.1	ND	**0.2**
Dissolved Thallium	ug/L	N/A	0.3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Dissolved Tungsten	ug/L	N/A	30	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Dissolved Uranium	ug/L	20	5	**7.8**	0.8	0.1	**12**	0.3	0.6	0.2	0.9	**35**	0.3	0.4
Dissolved Vanadium	ug/L	N/A	6	ND	ND	ND	**6**	ND	ND	ND	**7**	2	5	ND
Dissolved Zinc	ug/L	N/A	20	ND	ND	ND	ND	ND	ND	ND	ND	12	ND	12
Dissolved Zirconium	ug/L	N/A	4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Nitrite	mg/L	1	N/A	ND	ND	ND	0.01	ND	ND	0.01	ND	ND	ND	0.01
Nitrate	mg/L	10	N/A	ND	ND	ND	72	56	ND	99	21	ND	ND	13
Nitrite + Nitrate	mg/L	10	N/A	ND	ND	ND	ND	2.7	ND	ND	4.6	ND	ND	0.2
pH	N/A	N/A	6.5-8.5	7.74	8.05	7.46	6.56	7.13	7.46	7.18	7.63	8.68	7.65	6.7
Dissolved Oxygen	mg/L	N/A	Temp. dependant	8.69	3.63	9.12	5.09	6.78	6.65	3.62	9.73	9.48	9.34	1.21

[a] ND, Not detected; Grey text indicates guidelines which are not applicable to wetlands; Values exceeding applicable guidelines are highlighted in bold.
[b] S, shallow well; D, deep well.
[c] 20mg/L is the guidelines for lakes, whereas 30mg/L is the guideline for rivers

APPENDIX B: WETLAND WATER QUALITY TESTING (Cont'd)

Table 3. Wetland surface water quality results for tested parameters with PWQO and/or ODWS guidelines [a].

Parameter	Units	ODWS Guideline	PWQO Guideline	W-04	W-10	W-16 S [b]	W-16 W [b]	W-17	W-20
Total Phosphorus	ug/L	N/A	20 – 30 [c]		**100**			16	**2300**
Total Aluminum	ug/L	N/A	75		**75**			18	22
Total Antimony	ug/L	6	20		ND			ND	ND
Total Arsenic	ug/L	25	5		2			1	ND
Total Barium	ug/L	1000	1000		110			26	69
Total Beryllium	ug/L	N/A	11		ND			ND	ND
Total Boron	ug/L	5000	200		16			14	10
Total Cadmium	ug/L	5	0.2		ND			ND	ND
Total Chromium	ug/L	50	1		ND			ND	ND
Total Cobalt	ug/L	N/A	0.9		ND			ND	ND
Total Copper	ug/L	N/A	5		ND			ND	1
Total Iron	ug/L	N/A	300		**970**			260	64
Total Lead	ug/L	10	5		0.5			ND	ND
Total Molybdenum	ug/L	N/A	40		ND			ND	ND
Total Nickel	ug/L	N/A	25		ND			ND	ND
Total Selenium	ug/L	10	100		ND			ND	ND
Total Silver	ug/L	N/A	0.1		ND			ND	ND
Total Thallium	ug/L	N/A	0.3		ND			ND	ND
Total Tungsten	ug/L	N/A	30		ND			ND	ND
Total Uranium	ug/L	20	5		0.2			0.9	ND
Total Vanadium	ug/L	N/A	6		ND			ND	ND
Total Zinc	ug/L	N/A	20		6			**38**	12
Total Zirconium	ug/L	N/A	4		ND			ND	ND
Nitrite	mg/L	1	N/A		ND			ND	ND
Nitrate	mg/L	10	N/A		ND			ND	ND
Nitrite + Nitrate	mg/L	10	N/A		ND			ND	ND
pH	N/A	N/A	6.5-8.5	8.08	7.5	7.62	7.22	8.23	6.6
Dissolved Oxygen	mg/L	N/A	Temp. dependant	6.9983.1	6.59	12.3	7.52	10.69	1.98

[a] ND, Not detected; Grey text indicates guidelines which are not applicable to wetlands; Values exceeding applicable guidelines are highlighted in bold; shaded cells indicate that analysis was not completed for that parameter.
[b] S, at spring; W, at piezometer.
[c] 20mg/L is the guidelines for lakes, whereas 30mg/L is the guideline for rivers

APPENDIX C: HYDROGRAPHS AT SELECTED WETLAND MONITORING SITES IN THE CREDIT RIVER WATERSHED

For hydrographs in this appendix, depth indicates the depth of water in relation to the ground surface. Therefore, positive numbers represent the depth of water above the ground and negative numbers represent the distance from the ground surface to water below. Surface water levels are determined from the depth of water on the ground surface directly outside of the piezometers. Groundwater levels are determined by the depth of water within the piezometers and may be either above or below the ground surface. Groundwater being higher than the surface water may be an indication of groundwater discharge into the wetland; however more in-depth hydrological studies are required to confirm these estimations. Two sampling dates occurred in May 2006 at each site, once in mid-May and once in late May. A dramatic difference between the measurements was observed only at Meadowvale Wetland. This difference could be attributed to either measurement error in the field or the effect of the Credit River on the water table in the spring, as this is a riparian wetland to the Credit River.

APPENDIX C: HYDROGRAPHS AT SELECTED WETLAND MONITORING SITES IN THE CREDIT RIVER WATERSHED (Cont'd)

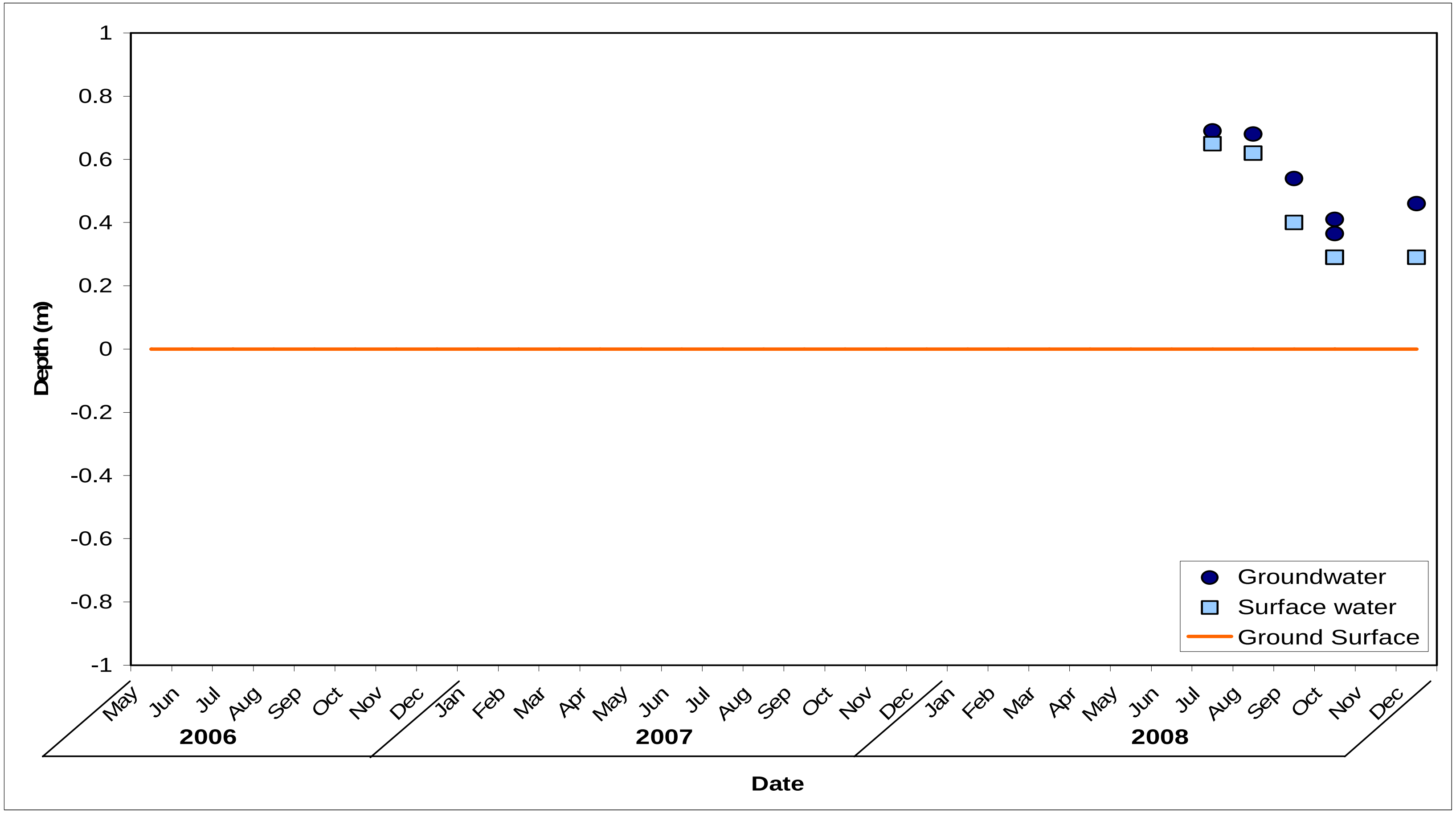

Figure 1. Hydrograph for Rattray Marsh Wetland (W-01) from Summer 2008 to Fall 2008. Depth (m) indicates the depth of water in relation to the ground surface. Therefore, positive numbers represent the depth of water above the ground and negative numbers represent the distance from the ground surface to water below.

APPENDIX C: HYDROGRAPHS AT SELECTED WETLAND MONITORING SITES IN THE CREDIT RIVER WATERSHED (Cont'd)

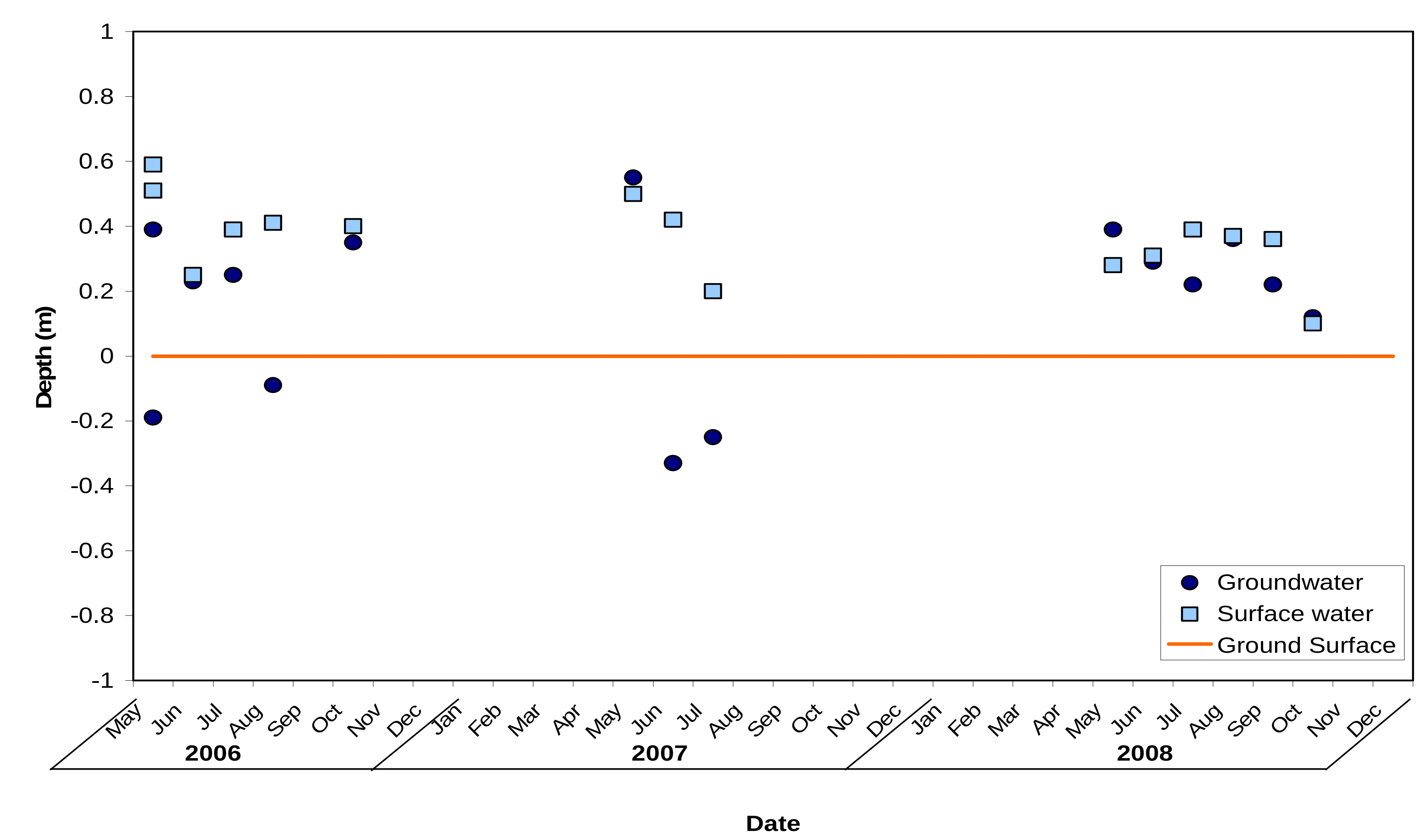

Figure 2. Hydrograph for Meadowvale Wetland (W-04) from Spring 2006 to Fall 2008. Depth (m) indicates the depth of water in relation to the ground surface. Therefore, positive numbers represent the depth of water above the ground and negative numbers represent the distance from the ground surface to water below.

APPENDIX C: HYDROGRAPHS AT SELECTED WETLAND MONITORING SITES IN THE CREDIT RIVER WATERSHED (Cont'd)

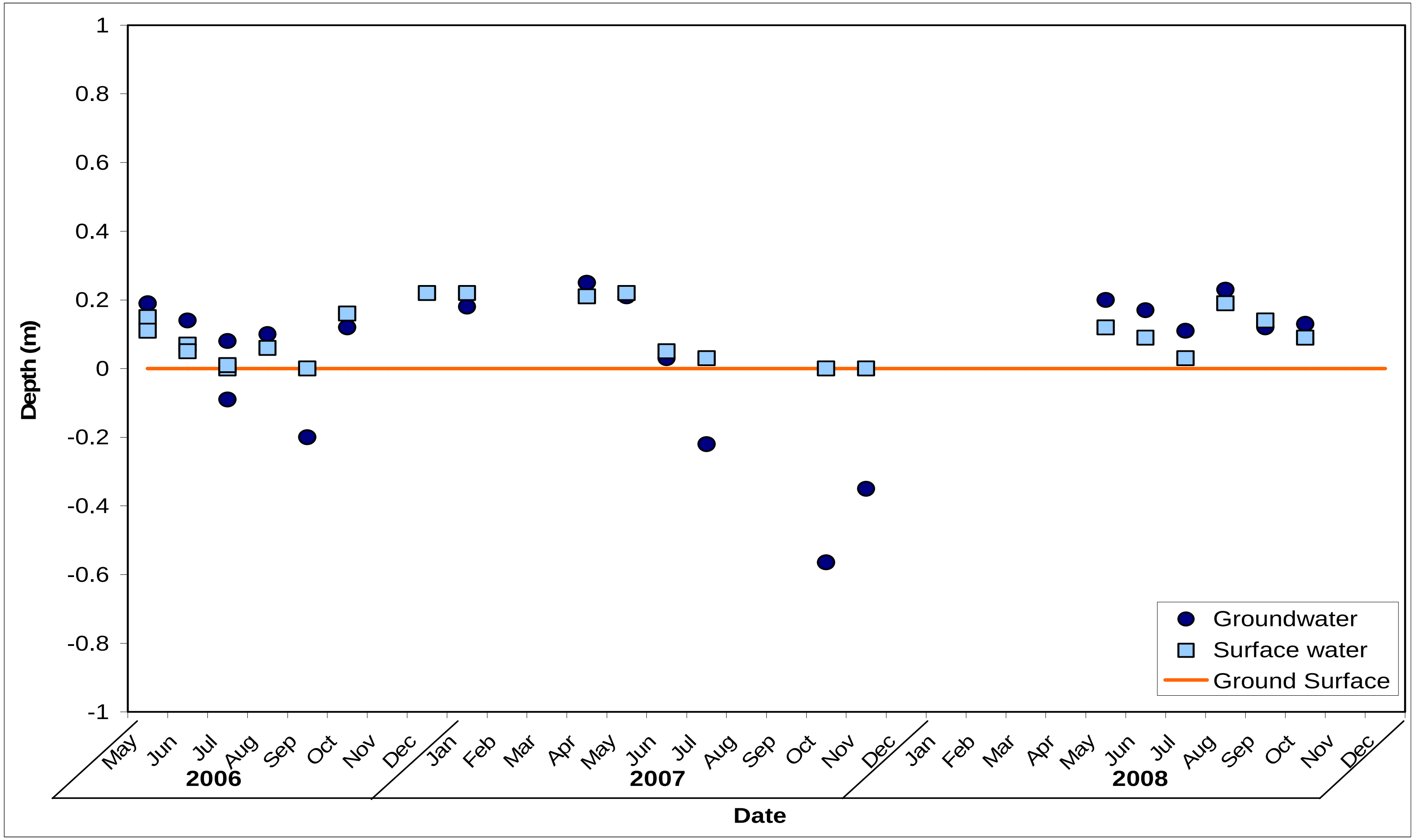

Figure 3. Hydrograph for Acton Wetland (W-07) from Spring 2006 to Fall 2008. Depth (m) indicates the depth of water in relation to the ground surface. Therefore, positive numbers represent the depth of water above the ground and negative numbers represent the distance from the ground surface to water below.

APPENDIX C: HYDROGRAPHS AT SELECTED WETLAND MONITORING SITES IN THE CREDIT RIVER WATERSHED (Cont'd)

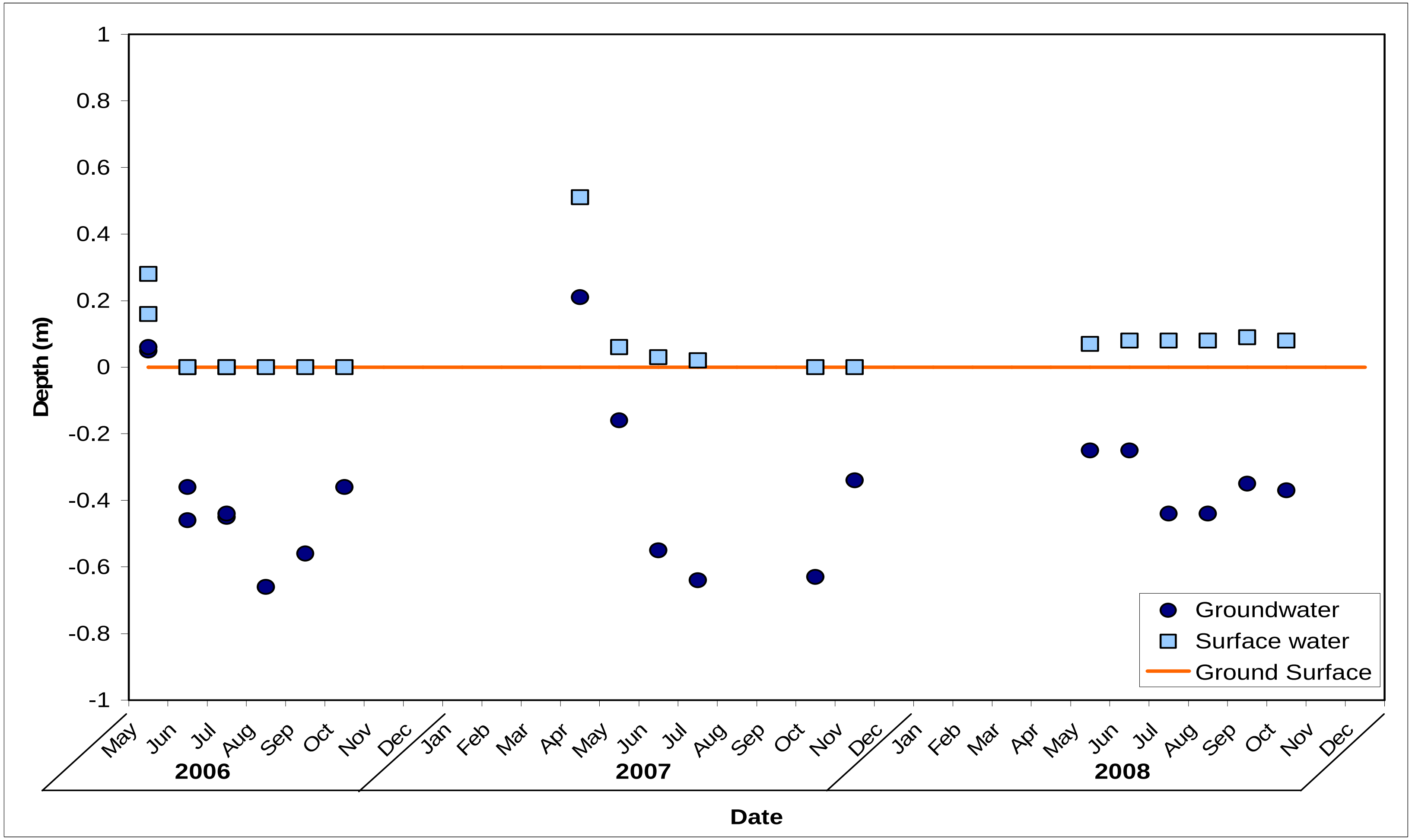

Figure 4. Hydrograph for Ken Whillans Wetland (W-09) from Spring 2006 to Fall 2008. Depth (m) indicates the depth of water in relation to the ground surface. Therefore, positive numbers represent the depth of water above the ground and negative numbers represent the distance from the ground surface to water below.

APPENDIX C: HYDROGRAPHS AT SELECTED WETLAND MONITORING SITES IN THE CREDIT RIVER WATERSHED (Cont'd)

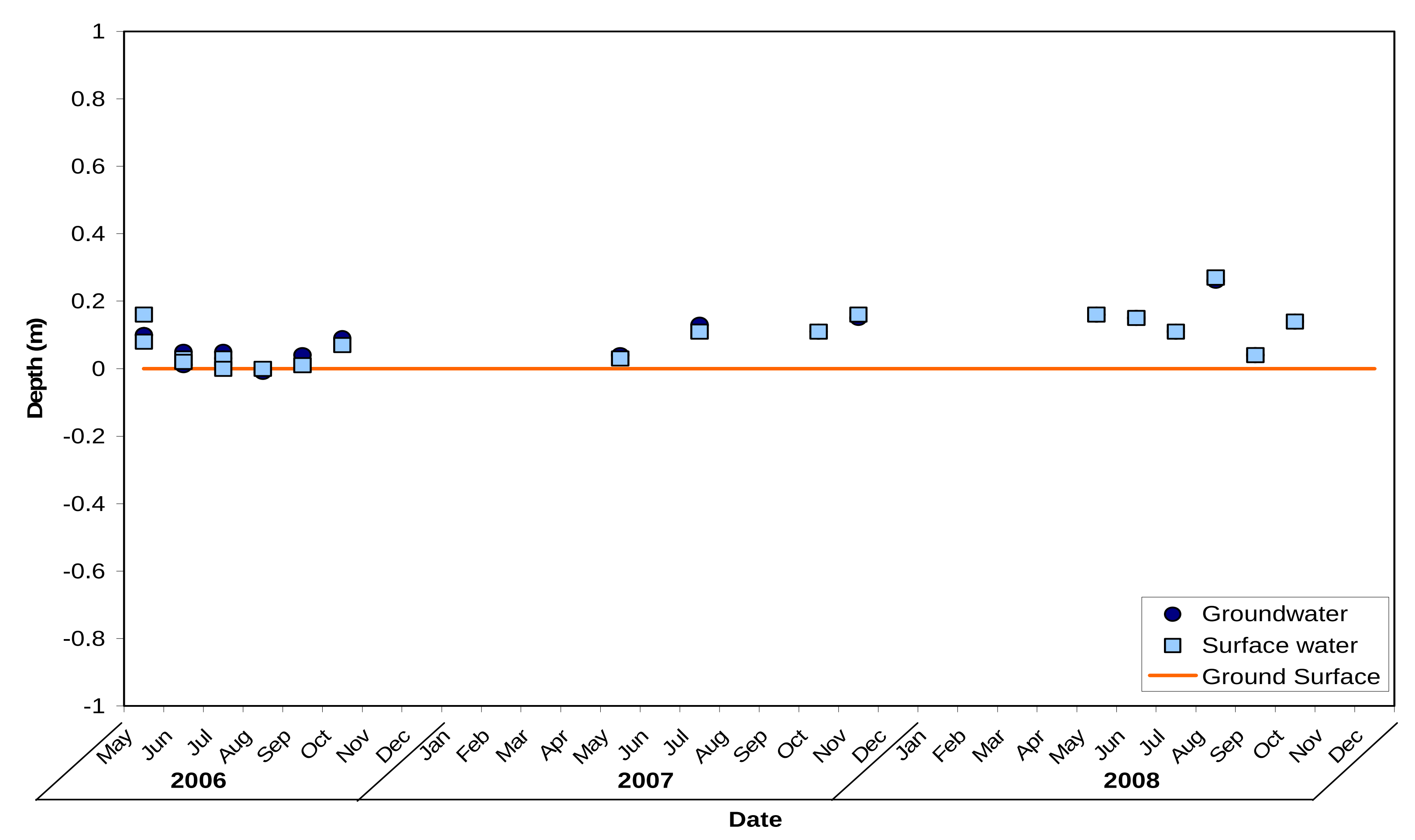

Figure 5. Hydrograph for Warwick Wetland (W-10) from Spring 2006 to Fall 2008. Depth (m) indicates the depth of water in relation to the ground surface. Therefore, positive numbers represent the depth of water above the ground and negative numbers represent the distance from the ground surface to water below.

APPENDIX C: HYDROGRAPHS AT SELECTED WETLAND MONITORING SITES IN THE CREDIT RIVER WATERSHED (Cont'd)

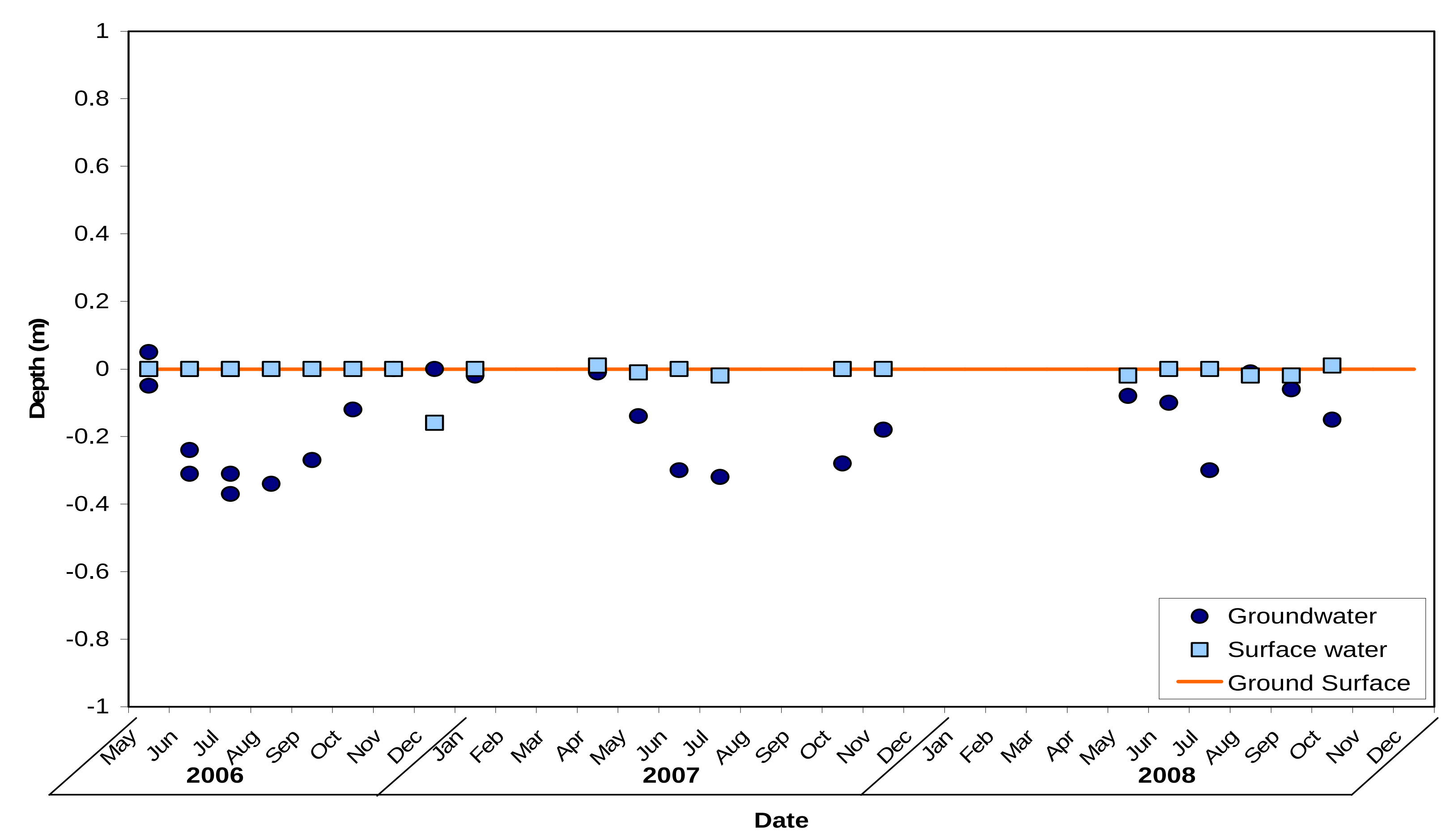

Figure 6. Hydrograph for Erin Pine Estates Wetland (Deep Piezometer) (W-11) from Spring 2006 to Fall 2008. Depth (m) indicates the depth of water in relation to the ground surface. Therefore, positive numbers represent the depth of water above the ground and negative numbers represent the distance from the ground surface to water below.

APPENDIX C: HYDROGRAPHS AT SELECTED WETLAND MONITORING SITES IN THE CREDIT RIVER WATERSHED (Cont'd)

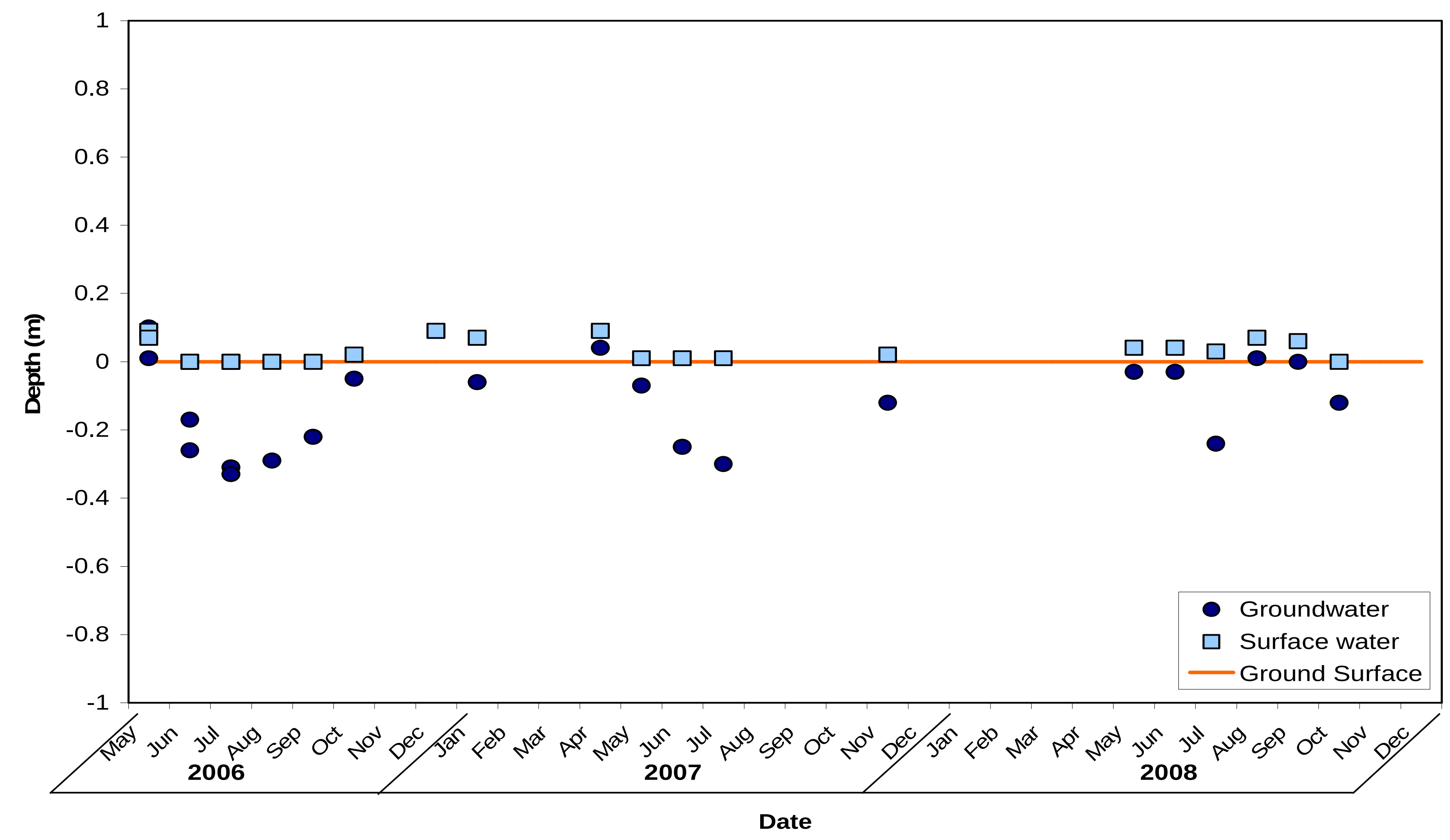

Figure 7. Hydrograph for Erin Pine Estates Wetland (Shallow Piezometer) (W-11) from Spring 2006 to Fall 2008. Depth (m) indicates the depth of water in relation to the ground surface. Therefore, positive numbers represent the depth of water above the ground and negative numbers represent the distance from the ground surface to water below.

APPENDIX C: HYDROGRAPHS AT SELECTED WETLAND MONITORING SITES IN THE CREDIT RIVER WATERSHED (Cont'd)

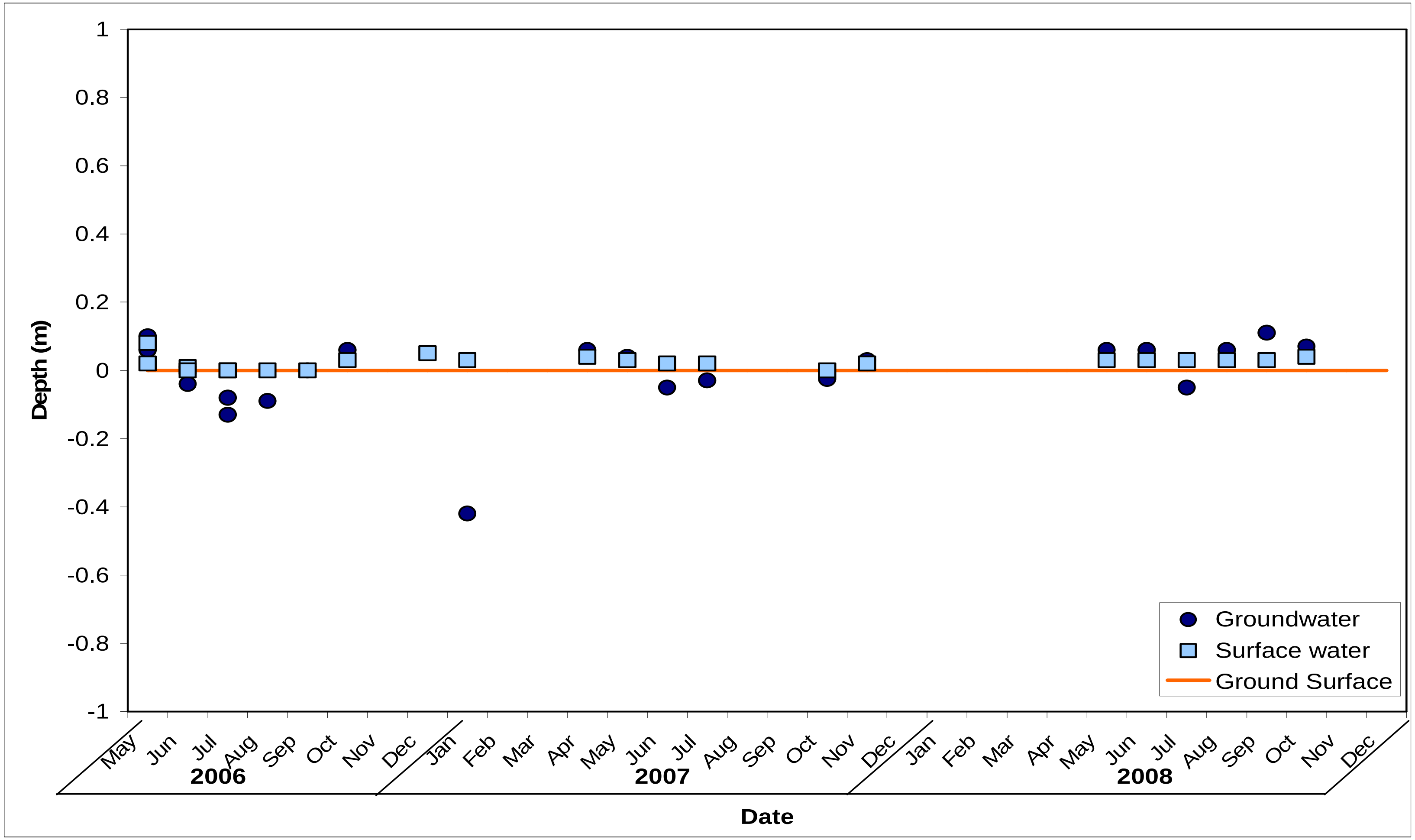

Figure 8. Hydrograph for Hillsburgh Wetland (W-13) from Spring 2006 to Fall 2008. Depth (m) indicates the depth of water in relation to the ground surface. Therefore, positive numbers represent the depth of water above the ground and negative numbers represent the distance from the ground surface to water below.

APPENDIX C: HYDROGRAPHS AT SELECTED WETLAND MONITORING SITES IN THE CREDIT RIVER WATERSHED (Cont'd)

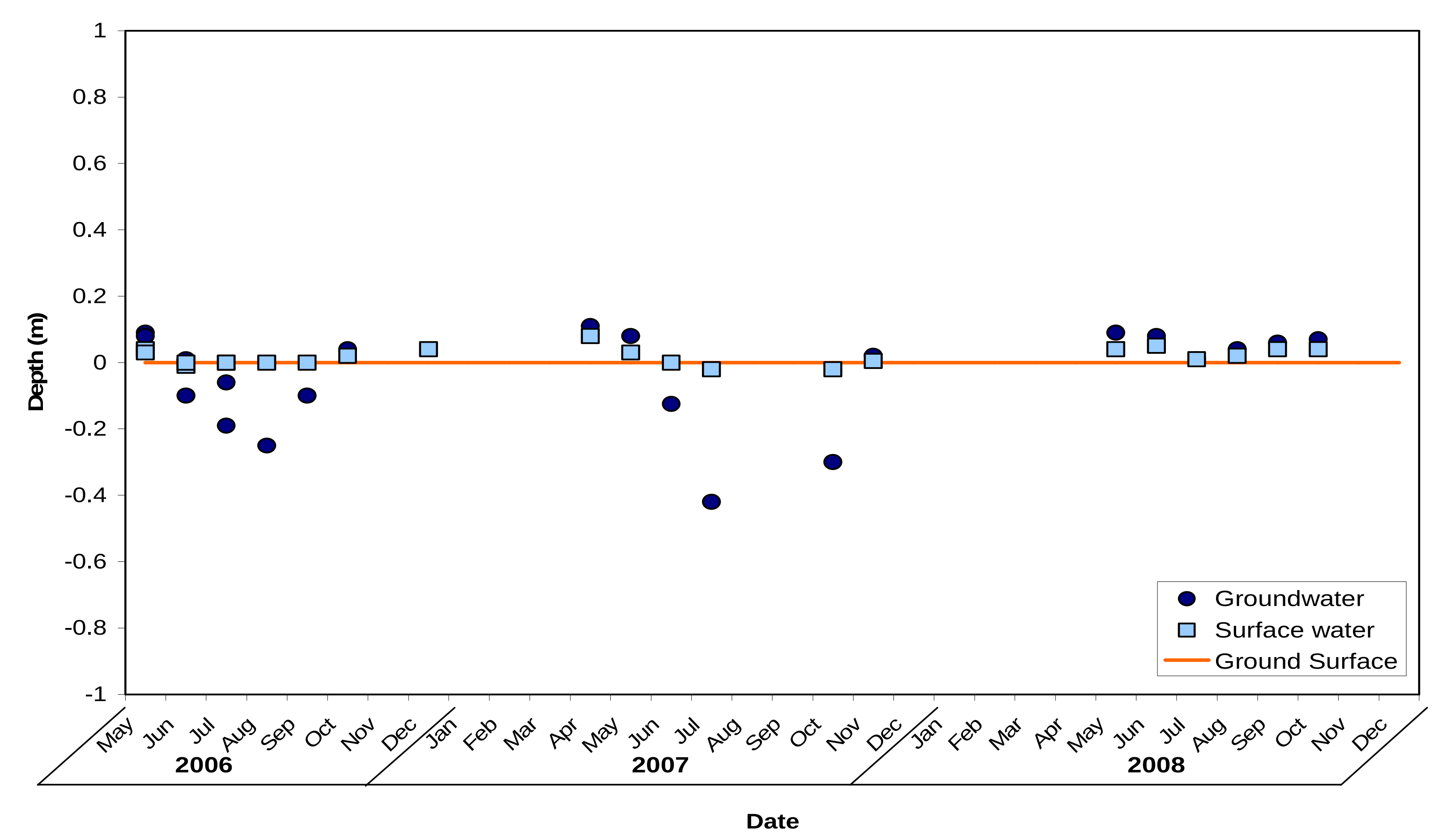

Figure 9. Hydrograph for Grange Orpen Wetland (W-15) from Spring 2006 to Fall 2008. Depth (m) indicates the depth of water in relation to the ground surface. Therefore, positive numbers represent the depth of water above the ground and negative numbers represent the distance from the ground surface to water below.

APPENDIX C: HYDROGRAPHS AT SELECTED WETLAND MONITORING SITES IN THE CREDIT RIVER WATERSHED (Cont'd)

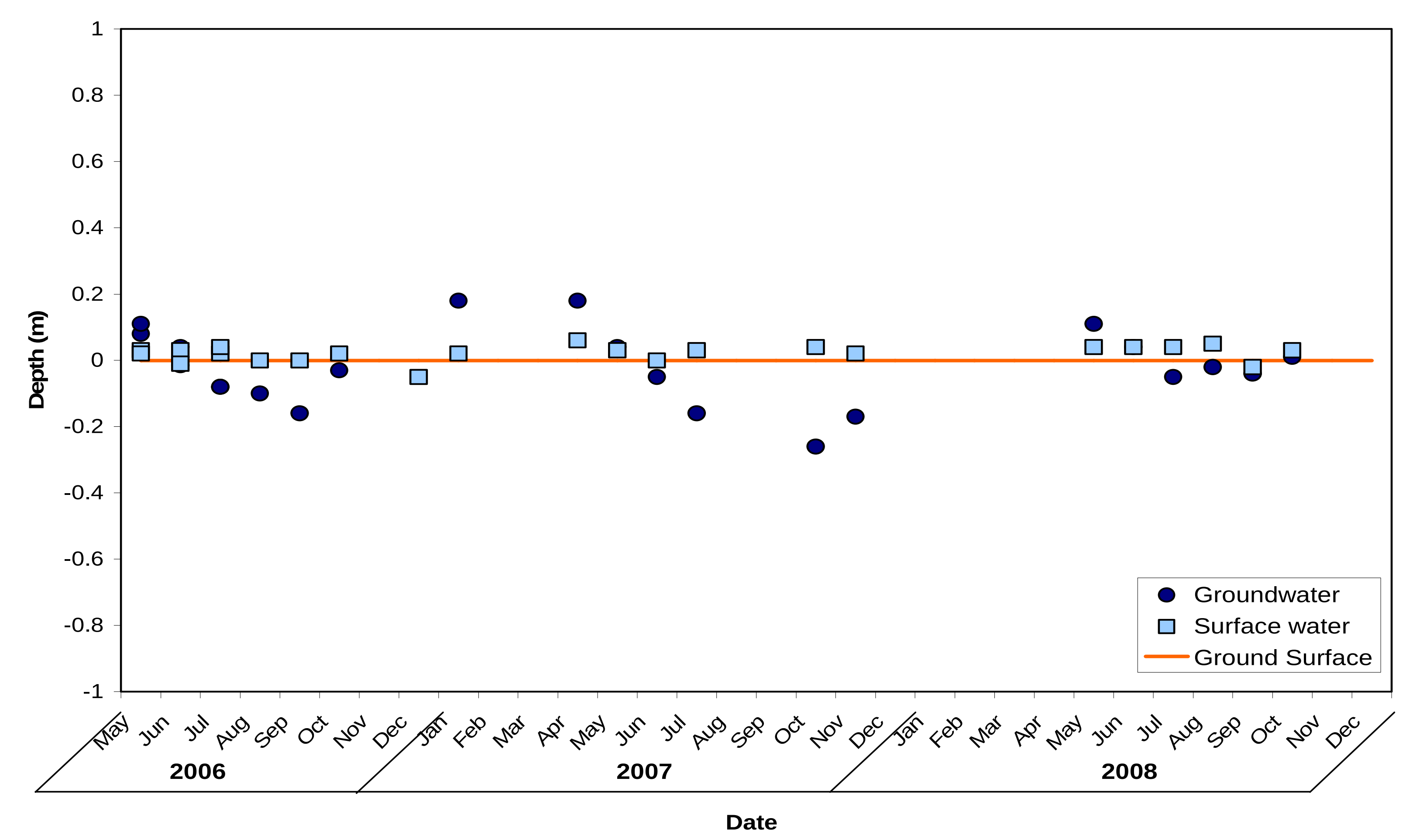

Figure 10. Hydrograph for Starr Wetland (W-16) from Spring 2006 to Fall 2008. Depth (m) indicates the depth of water in relation to the ground surface. Therefore, positive numbers represent the depth of water above the ground and negative numbers represent the distance from the ground surface to water below.

APPENDIX C: HYDROGRAPHS AT SELECTED WETLAND MONITORING SITES IN THE CREDIT RIVER WATERSHED (Cont'd)

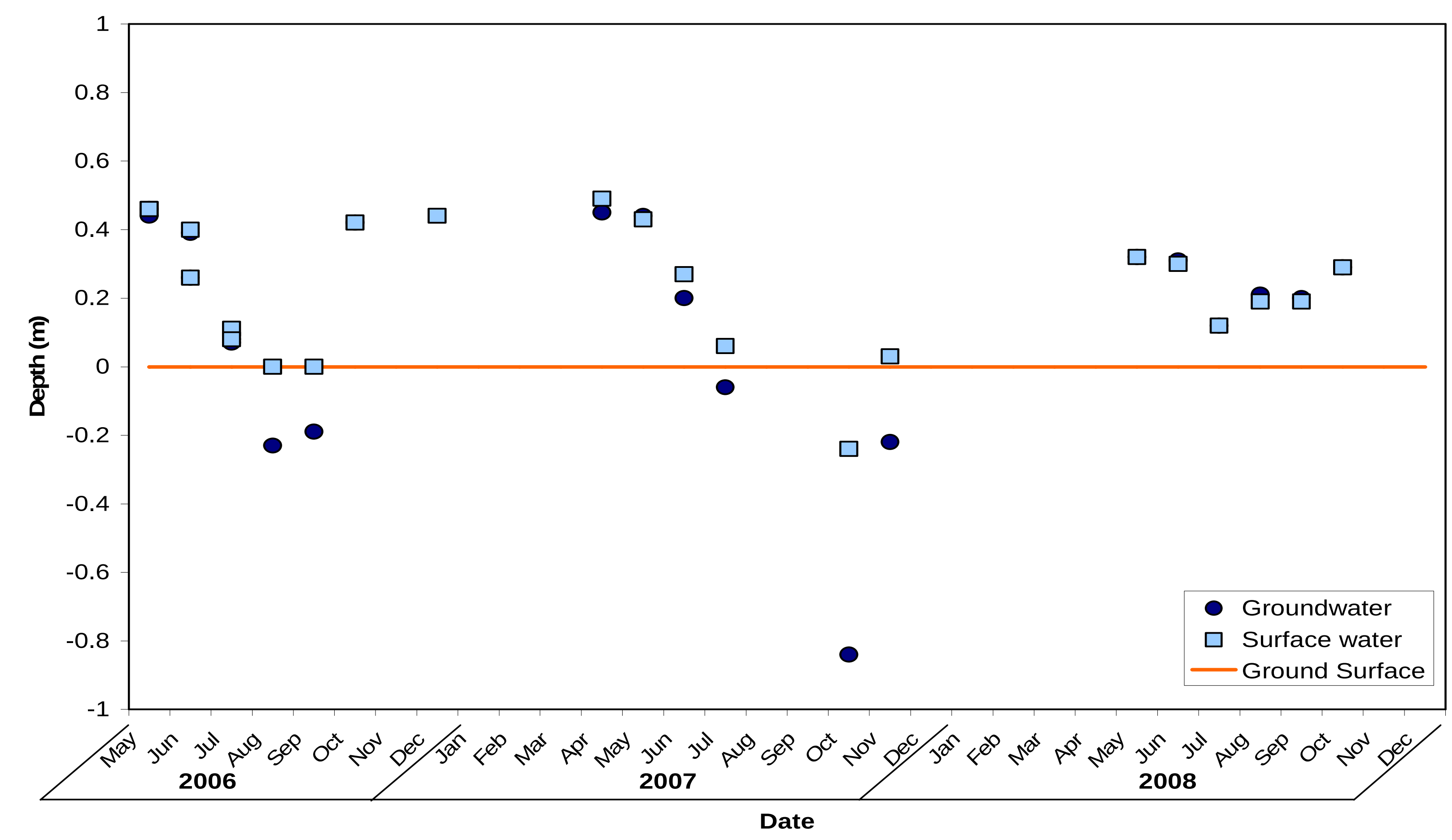

Figure 11. Hydrograph for Speersville Wetland W-17) from Spring 2006 to Fall 2008. Depth (m) indicates the depth of water in relation to the ground surface. Therefore, positive numbers represent the depth of water above the ground and negative numbers represent the distance from the ground surface to water below.

APPENDIX C: HYDROGRAPHS AT SELECTED WETLAND MONITORING SITES IN THE CREDIT RIVER WATERSHED (Cont'd)

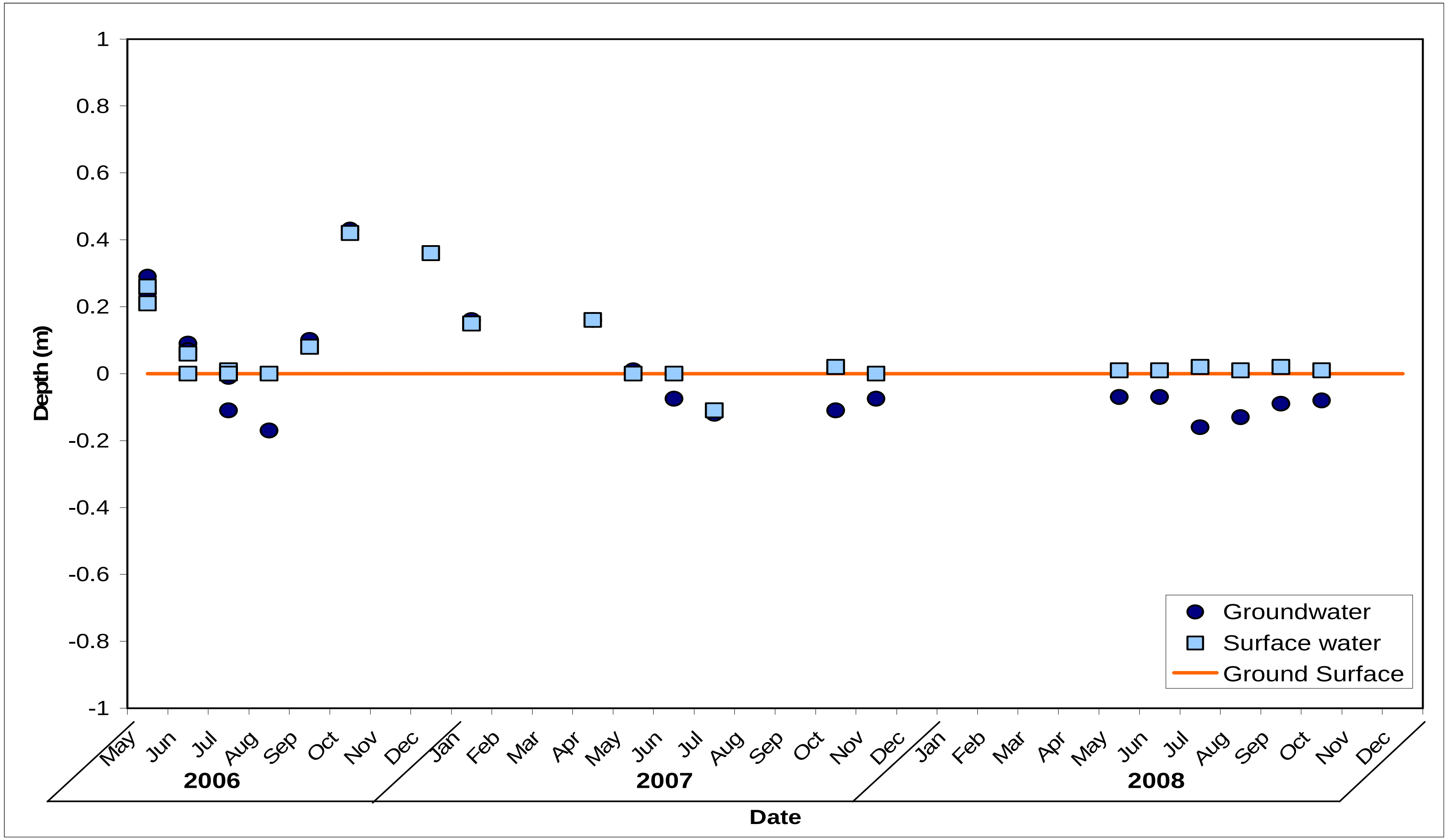

Figure 12. Hydrograph for Melville Wetland (W-19) from Spring 2006 to Fall 2008. Depth (m) indicates the depth of water in relation to the ground surface. Therefore, positive numbers represent the depth of water above the ground and negative numbers represent the distance from the ground surface to water below.

APPENDIX C: HYDROGRAPHS AT SELECTED WETLAND MONITORING SITES IN THE CREDIT RIVER WATERSHED (Cont'd)

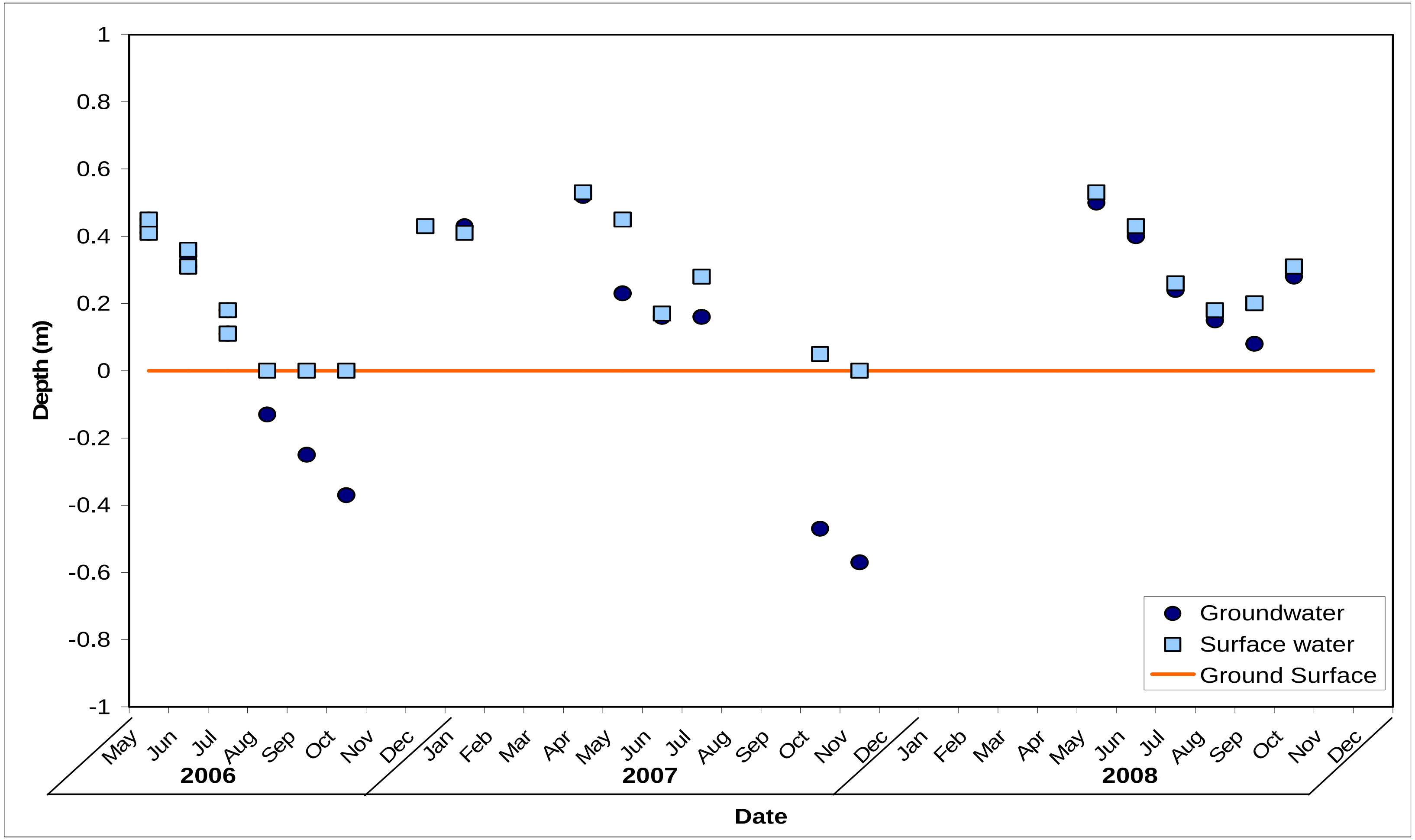

Figure 13. Hydrograph for Belfountain Wetland (W-20) from Spring 2006 to Fall 2008. Depth (m) indicates the depth of water in relation to the ground surface. Therefore, positive numbers represent the depth of water above the ground and negative numbers represent the distance from the ground surface to water below.

APPENDIX D: WETLAND SITE HYDROLOGY SUMMARIES

Table 1. Summary of local hydrology at selected wetland monitoring sites in the Credit River Watershed based on piezometer water levels.

Wetland Site	2006	2007	2008	Combined
Meadowvale Wetland	•During dryer periods, groundwater drops below ground surface •No periods of groundwater discharge to wetland •Surface water always present (0.25-0.59m above ground surface)	•During dryer periods, groundwater drops below ground surface •Groundwater discharge into wetland in May •Surface water always present (0.2-0.5m above ground surface)	•Groundwater does not drop below ground surface – 2008 not as dry as 2007 •Groundwater discharge entire season •Surface water always present (0.3-0.65m above ground surface)	•Limited groundwater discharge to wetland •Typical groundwater drops in dry periods •Generally wet
Rattray Marsh Wetland			•Groundwater discharge into wetland in May and October •Surface water always present (0.1-0.39m above ground surface) Ground and surface water levels decrease during dryer periods	•Moderate amounts groundwater discharge to wetland •Typical groundwater drops in dry periods •Generally wet
Acton Wetland	•During dryer periods (July & September), groundwater drops below ground surface •Groundwater discharge to wetland from May – June, and in late summer •Surface water absent at dry time with average of 0.1m above ground surface	•During dryer periods (July – Nov), groundwater drops below ground surface (up to 0.57m below) •Groundwater discharge into wetland in Apr •Surface water always present (0.03-0.22m above ground surface)	•Groundwater does not drop below ground surface – 2008 not as dry as 2007 •Groundwater discharge into wetland over the majority of the season •Surface water always present (0.03-0.19m above ground surface)	•Moderate amounts groundwater discharge to wetland •Typical groundwater drops in dry periods •Generally moist
Warwick Wetland				•Surface water almost always present, sometimes dry during dry season, seems to fluctuate over season •Organics likely not providing a compact enough seal, resulting in surface water monitoring only, well may be more suitable
Ken Whillans Wetland	•Groundwater drops below ground surface for the majority of the season (except May) (mean 0.47m) •No groundwater discharge to wetland •Surface water present in May, absent for remainder of season	•Groundwater drops below ground surface for the majority of the season (except Apr) (mean 0.46m) •No groundwater discharge to wetland •Surface water at 0.5m in May but drops to mean of 0.04m into May and June, absent later in season	•Groundwater drops below ground surface for entire season (mean 0.35m) •No groundwater discharge to wetland •Surface water always present (mean 0.08m)	•No groundwater discharge to wetland •Groundwater below surface for majority of season •Generally dry •Groundwater not a large contributor to wetland hydrology
Erin Pine Estates Wetland	•Groundwater drops below ground surface for the majority of the season •Groundwater discharge to wetland only in May •Surface water generally absent, present in shallow early and late in season	•Groundwater drops below ground surface for the majority of the season •No groundwater discharge to wetland •Surface water generally absent or minimal after April	•Groundwater drops below ground surface for entire season •No groundwater discharge to wetland •Surface water absent from deep well, minimal at shallow well	•No groundwater discharge to wetland •Groundwater below surface for majority of season

Wetland Site	2006	2007	2008	Combined
Hillsburgh Wetland	•Groundwater drops below ground surface during dry periods •Groundwater discharge to wetland early and late in season •Surface water absent during summer season	•Groundwater drops below ground surface during dry periods •Minimal groundwater discharge to wetland early and late in season •Surface water minimal (mean 0.03m) and stable	•Groundwater drops below ground surface only in July •Potential groundwater discharge to wetland for majority of season •Surface water minimal (mean 0.03m) and stable	•Limited potential groundwater discharge to wetland, depending on till •Groundwater below surface only in dry times •Minimal surface water and stable •Creek may moderate surface water levels
Grange Orpen Wetland	•Groundwater drops below ground surface from June-Sept •Groundwater discharge to wetland in spring and fall •Surface water dry from June - Sept	•Groundwater drops below ground surface from June-Oct •Groundwater discharge to wetland in spring and fall •Surface water dry from June - Oct	•Groundwater does not drops below ground surface for entire season •Groundwater discharge to wetland for majority of season •Surface water always present (mean 0.03m)	•Potential groundwater discharge to wetland •Groundwater below surface during dry events •Generally moist
Starr Wetland	•During dryer periods (June – Oct.), groundwater drops below ground surface •Groundwater discharge to wetland from May – June •Surface water absent in Aug, with average of 0.03m above ground surface	•During dryer periods (June – Nov), groundwater drops below ground surface •Groundwater discharge into wetland Jan- May •Surface water always present (mean 0.03m above ground surface)	•Groundwater drops below surface but by no more than 0.05m from July-Sept. •Groundwater discharge into wetland only in May •Surface normally present (mean 0.04m)	•Moderate amount of groundwater discharge to wetland •Typical groundwater drops in dry periods •Generally moist •Annual spring flow at site
Speersville Wetland	•Groundwater drops below ground surface during dry periods •No groundwater discharge to wetland •Surface water absent Aug – Sept, decreases from spring to summer and then increases in fall	•Groundwater drops below ground surface during dry periods •Essentially no groundwater discharge to wetland •Surface water absent Aug – Oct, decreases from spring to summer and then increases in fall	•Groundwater never drops below ground surface •Essentially no groundwater discharge to wetland •Surface water always present (mean 0.24m)	•Piezometer may be leaking, may just be monitoring surface was, as groundwater follows similar trends, may also be result of engineered fill which is why groundwater falls below surface during dry periods •Essentially no groundwater discharge to wetland •Groundwater below surface during dry periods •Usually standing water
Melville Wetland	•Groundwater drops below ground surface in summer •Minimal groundwater discharge in spring and fall •Surface water present for entire season, but fluctuates greatly due to beaver dam	•Groundwater drops below ground surface in summer and fall •Essentially no groundwater discharge to wetland •Surface water variable, higher in spring and fall, also affected by dam?	•Groundwater drops below ground surface for entire season (mean 0.10m) •No groundwater discharge to wetland •Surface water always present (mean 0.01m)	•2006 and 2007 surface data affected by beaver dam •Very limited groundwater discharge to wetland •Groundwater below surface for large portions of season •Generally dry after removal of dam
Belfountain Wetland	•Groundwater drops below ground surface in dry season (Aug – Oct) •No groundwater discharge to wetland •Surface water present until Aug, absent for remainder of season until Dec	•Groundwater drops below ground surface in fall •Groundwater discharge to wetland only in Jan •Surface water present until Nov. (mean 0.32m)	•Groundwater does not drop below ground surface •No groundwater discharge to wetland •Surface water always present (mean 0.29m)	•Groundwater has very little input in wetland, surface water more important, outflow location from wetland •Surface water depth decreases throughout the season •Groundwater below surface during dry spells

APPENDIX E: WETLAND MONITORING SITE MAPS AND PHOTOS

Figure 1. Rattray Marsh Wetland (W-01) site map and photograph.

APPENDIX E: WETLAND MONITORING SITE MAPS AND PHOTOS (Con't)

Figure 2. Meadowvale Wetland (W-04) site map and photograph.

APPENDIX E: WETLAND MONITORING SITE MAPS AND PHOTOS (Con't)

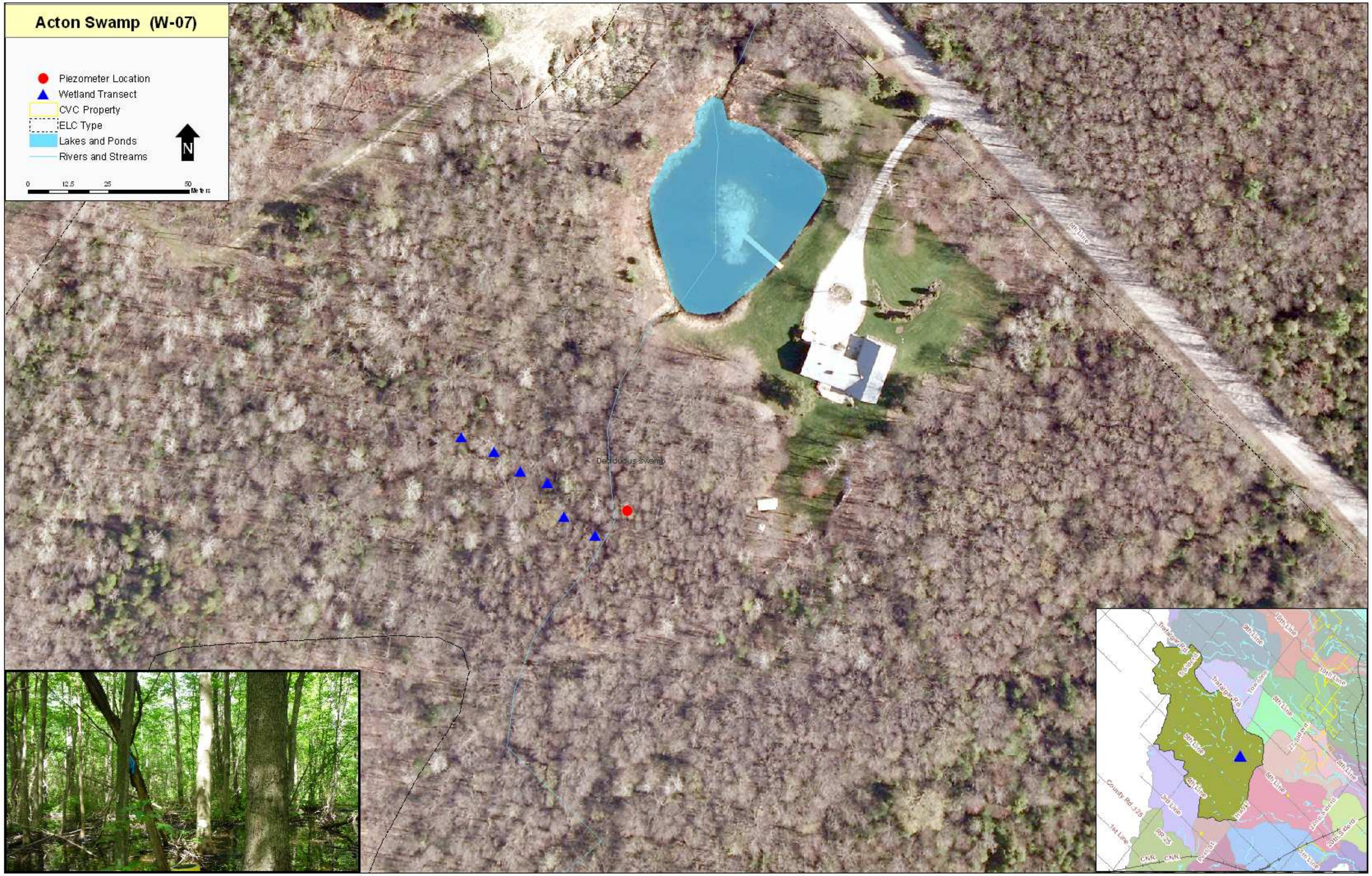

Figure 3. Action Wetland (W-07) site map and photograph.

APPENDIX E: WETLAND MONITORING SITE MAPS AND PHOTOS (Con't)

Figure 4. Ken Whillans Wetland (W-09) site map and photograph.

APPENDIX E: WETLAND MONITORING SITE MAPS AND PHOTOS (Con't)

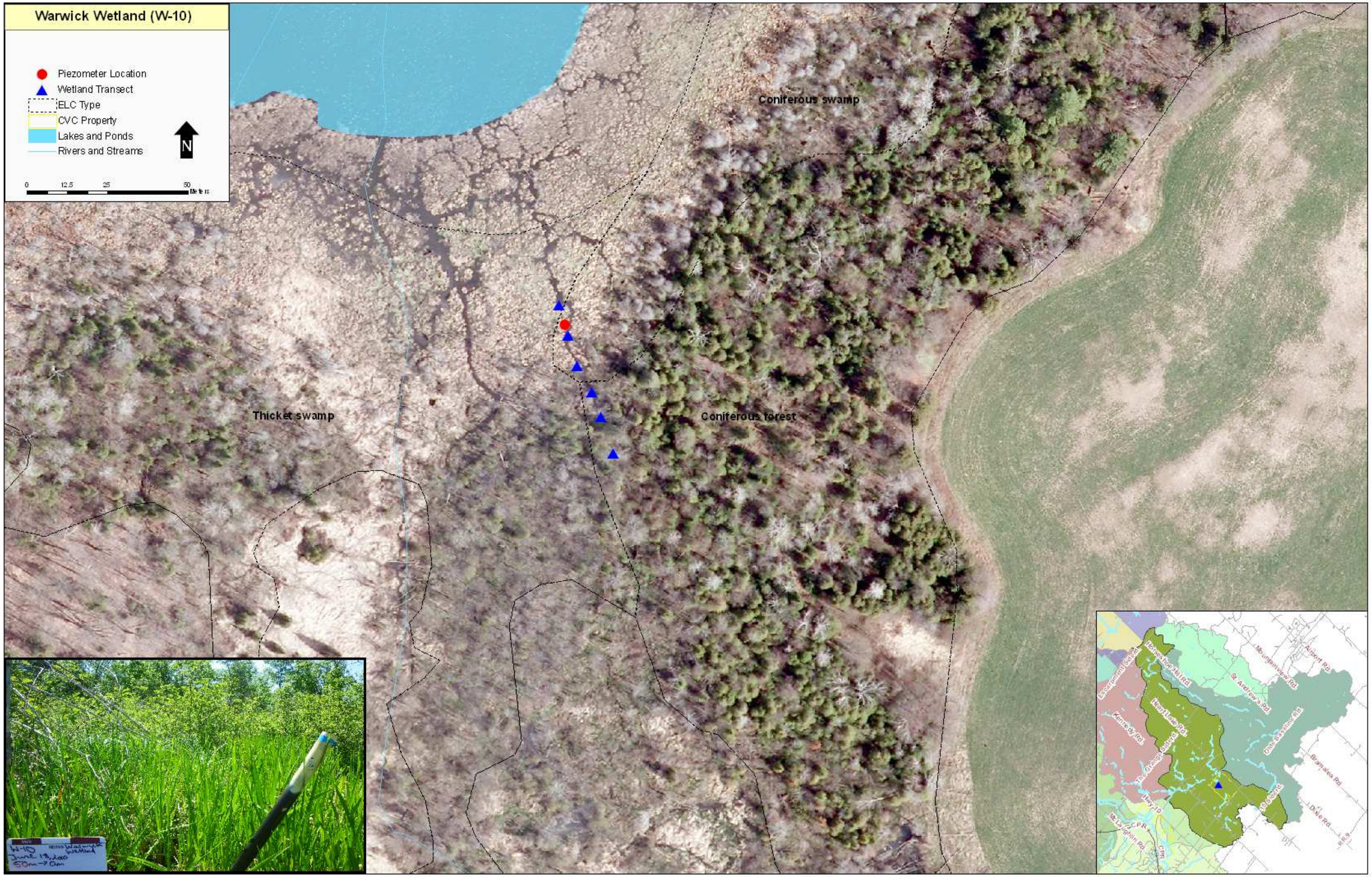

Figure 5. Warwick Wetland (W-10) site map and photograph.

APPENDIX E: WETLAND MONITORING SITE MAPS AND PHOTOS (Con't)

Figure 6. Erin Pine Estates Wetland (W-11) site map and photograph.

APPENDIX E: WETLAND MONITORING SITE MAPS AND PHOTOS (Con't)

Figure 7. Hillsburgh Wetland (W-13) site map and photograph.

APPENDIX E: WETLAND MONITORING SITE MAPS AND PHOTOS (Con't)

Figure 8. Grange Orpen Wetland (W-15) site map and photograph.

APPENDIX E: WETLAND MONITORING SITE MAPS AND PHOTOS (Con't)

Figure 9. Starr Wetland (W-15) site map and photograph.

APPENDIX E: WETLAND MONITORING SITE MAPS AND PHOTOS (Con't)

Figure 10. Speersville Wetland (W-17) site map and photograph.

APPENDIX E: WETLAND MONITORING SITE MAPS AND PHOTOS (Con't)

Figure 11. Melville Wetland (W-19) site map and photograph.

APPENDIX E: WETLAND MONITORING SITE MAPS AND PHOTOS (Con't)

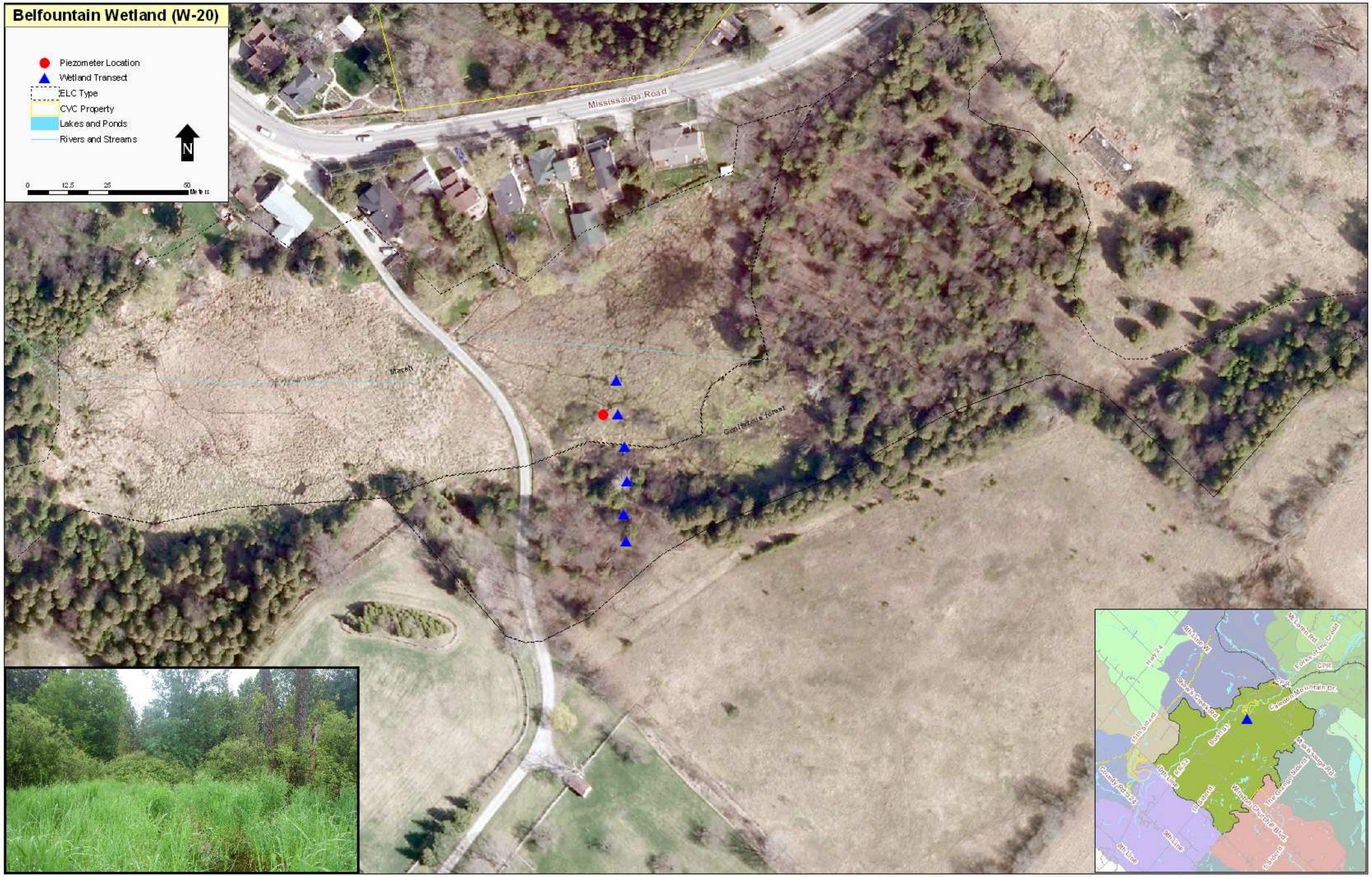

Figure 12. Belfountain Wetland (W-20) site map and photograph.

APPENDIX F: ADDITIONAL LABORATORY AND EQUIPMENT COSTS

LABORATORY COSTS

Table 1. Laboratory costs for each parameter in the WQI (excludes *in situ* parameters) as quoted from Maxxam Analytics. [a]

Parameter	Cost per Sample	Notes
Chlorophyll-a	$100	
Total suspended solids	$8.25	
Total inorganic suspended solids	$15.50	Cost for Volatile suspended solids test, needed for TISS calculation (TISS=TSS-VSS)
Total phosphorus	$15.50	
Soluble reactive phosphorus	$7.20	
Total ammonium nitrogen	$12.30	
Total nitrate nitrogen	$14.40	
Total nitrogen	$15.45	Cost for TKN test, needed for TN calculation

[a] Displayed costs are on a per sample basis. Lower costs may be available for mass sample submissions.

IN SITU EQUIPMENT COSTS

Quanta T System

The Quanta T system is a complete water quality monitoring system capable of monitoring multiple water quality parameters simultaneously. The system is comprised of three components: the Quanta transmitter, the Quanta display, and a connecting cable. The system can be custom configured to monitor your desired parameters (it will measure all four *in situ* parameters required for the WQI). More detailed information regarding this equipment can be found at the following website: http://www.campbellsci.ca/Catalogue/QUANTAT.html. The purchase cost before taxes for this system is estimated at under $6000.00 (Table 2). Please note that this estimate was generated from the 2009 Campbell Scientific Hydrolab Products price list. Additional supplies, such as reagents or carrying case, may need to be purchased. A quote from Campbell Scientific should be requested prior to purchase.

Table 2. Estimated purchase costs of a Quanta T transmitter unit as estimated from Campbell Scientific Price List.

Product #	Description	Cost
Quanta T	Basic Quanta Transmitter [a]	$1730.00
Quanta D	Quanta Display	$1110.00
H8047	Specific Conductance Sensor	$520.00
H8041	pH Sensor	$690.00
H8060	4Beam Turbidity Sensor	$1850.00
Total Estimate		$5630.00

[a] Includes temperature sensor, SDI-12 Format and 5 m cable.

APPENDIX F: ADDITIONAL LABORATORY AND EQUIPMENT COSTS (Cont'd)

Other Options

Portable testers for individual parameters can also be purchased (Table 3). There is a broad range of portable testers available for each parameter. The Pocket Pal testers are at the low end of the cost scale, while other equipment can cost over $1000.00. Range and accuracy of the Pocket Pal testers are comparable to that of the Quanta (Table 4); however, the Pocket Pal testers are not as convenient as the Quanta because they do not provide the ease of measuring all parameters at once and they do not store data within the device. More information on portable testers can be found at the following website: http://www.fondriest.com/waterquality.htm.

Table 3. Estimated purchase costs of individual portable testers.

Product	Cost [a]
Hach Pocket Pal Conductivity Tester	$64.50
Hach Pocket Pal pH Tester	$56.67
Hach Pocket Pal Temperature Tester	$53.06
Hach 2100P Portable Turbidity Meter	$815.00
Total Estimate	$989.23

[a] Prices in US dollars and do not include the cost of reagents.

Table 4. Comparison of range and accuracy between the Quanta system and portable meters.

Parameter	Range	Accuracy
Temperature		
Quanta T	-5 °C to 50 °C	+/-0.20°C
Pocket Pal	-15 °C to 170 °C	+/-1.0 °C
Conductivity		
Quanta T	0 to 100 mS/cm	+/-1% of reading +/- 1 count
Pocket Pal	10 to 1990 uS/cm	+/-2% of reading
pH		
Quanta T	0 to 14 units	+/-0.2 units
Pocket Pal	0 to 14 units	+/-0.1 units
Turbidity		
Quanta T	0 - 1000 NTU	± 5% of reading ±1 NTU
Portable Turbidity Meter	0 - 1000 NTU	± 2% of reading or ± 1 least significant digit from 0-500 NTU ± 3% of reading between 500-1000 NTU

Credit Valley Conservation

Terrestrial Monitoring Program Report

2003 - 2008

Monitoring Wetland Integrity within the Credit River Watershed
Chapter 2: Wetland Anurans

CVC
CREDIT VALLEY
CONSERVATION

Monitoring Wetland Integrity within the Credit River Watershed

Chapter 2: Wetland Anurans 2003-2008

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ABSTRACT

Anurans of the Credit River Watershed: Summary of Monitoring Results 2003-2008
Temporal Trends: No significant temporal trends in total anuran richness and total occupancy were detected. A significant increase in the proportion of sites with full chorus was detected across the whole of the watershed and in the Middle physiographic zone.
Spatial Trends: Species richness was significantly higher in the Middle physiographic zone when compared to the Lower zone for five of the six survey years. Non-statistical observations revealed consistently lower anuran measures in the Lower zone when compared to the Middle and Upper zones. There were no significant differences in anuran parameters between the Middle and Upper physiographic zones.
Individual Species Trends: Green Frog occupancy increased in the Lower watershed while Leopard Frog occupancy increased in the Upper watershed. Green Frog and Spring Peeper were the most commonly identified species during the surveys. Spring Peeper was consistently absent from sites within the Lower physiographic zone. American Bullfrog was absent from the Lower and Upper zones, while Western Chorus Frog was absent from the Middle zone.
Relationships with Landscape and Vegetation: Measures associated with habitat loss and fragmentation (habitat patch size, percent urban land cover within 2 kilometres, non-native vegetation) appeared to have the greatest correlation with anuran richness, occupancy, and abundance.
Recommendations: Continue to track trends and relationships closely over subsequent survey years in order to determine if intervention is necessary to maintain stable anuran populations in the watershed. Land use and wetland vegetation data should be kept up-to-date. Call-detection methodology should be reviewed due to current limitations in the estimation of abundance counts.

Monitoring Wetland Integrity within the Credit River Watershed.
Chapter 2: Wetland Anurans 2003-2008

Anuran monitoring was conducted as part of the terrestrial component of an integrated watershed monitoring program established at Credit Valley Conservation. Anuran species were surveyed at 26 wetland sites throughout the Credit River Watershed from 2003 to 2008. Surveys were based on call detection and followed protocols of the Marsh Monitoring Program. Collected data were used to examine temporal and spatial trends in anuran species richness, site occupancy, and abundance, as well as the relationship between anuran parameters and both landscape and wetland vegetation measures. Eight frog species and one toad species were detected over the six year study period. Green Frog (*Rana clamitans*) and Spring Peeper (*Pseudacris crucifera*) were the most commonly detected species. At this point in monitoring, the nature of collected data favoured the use of non-parametric equivalents to chosen parametric statistical tests. No significant trends were detected in total species richness or total site occupancy, though Green Frog occupancy was found to be increasing in the Lower watershed and Leopard Frog occupancy was found to be increasing in the Upper watershed. In addition, there were significant increases in the number of sites with full chorus calling at both the watershed level and within the Middle physiographic zone. From a spatial perspective, anuran measures were consistently diminished in the lower physiographic zone when compared to those in the Middle and Upper zones. This pattern is likely a result of extensive urbanization and development present in the Lower watershed. Urban effects were also evident in comparisons with landscape and vegetation measures in that measures associated directly or indirectly with habitat loss and fragmentation (habitat patch size, percent urban land cover within 2 kilometres, non-native vegetation) had the greatest correlation with anuran richness, occupancy, and abundance. Further years of monitoring data and updated landscape information are required to determine if anuran populations in the Credit River watershed are stable.

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1.0 INTRODUCTION

1.1 BACKGROUND

Ontario has undergone dramatic change since the arrival of European settlers. A landscape that was historically dominated by forest cover and wetlands has experienced rapid removal of wildlife habitat and fragmentation of remaining natural areas by agriculture, residential development and roads (Larson et al. 1999). The amount of natural cover in a landscape often dictates what forest or wetland dependent wildlife species can be supported. Although this is especially evident with the disappearance of large mammals such as the gray wolf (*Canis lupus*) and black bear (*Ursus americanus*) from the highly developed landscapes of southern Ontario, this is also true for smaller organisms such as amphibians, birds, and plants. Habitat loss, and therefore reductions in overall natural cover can negatively affect a species in many ways; a) some species are observed to disappear from the landscape, b) some species are observed to decline in abundance, while c) others fail to reproduce (Noss and Cooperrider 1994). These effects are not mutually exclusive, and may feed back on one another to exacerbate negative impacts.

Fragmentation generally results in the reduction of total habitat available, the isolation of remaining patches, decreases in patch size and often an increase in total patch number (Noss and Cooperrider 1994; Fahrig 2002). Plant and wildlife species that are adapted to living in well-connected landscapes now face the challenge of acquiring all resources crucial to survival and reproduction from isolated habitat patches (Beier and Noss 1998). This is especially difficult for species that are poor dispersers or require large home ranges to fulfill their foraging and breeding needs (Andren and Nurnburger 1994). Isolation from other patches can result in reduced ability to secure required resources, reduced gene flow and ultimately extirpation of a species (Fleury and Brown 1997). Habitat loss and fragmentation can therefore have serious implications for populations, biodiversity and ecosystem functioning (Hess and Fischer 2001; Noss and Cooperrider 1994). In the case of amphibians, the richness and site occurrence of individual species were found to be influenced by environmental variables at both local and landscape scales (Hecnar and M'Closkey 1998; Buskirk 2005).

1.2 THE IMPORTANCE OF LONG TERM MONITORING

The baseline classification of natural areas and populations is important for understanding spatial patterns of structure, composition, processes, and functions. However, the mapping and classification of populations usually produce only a "snapshot" of community status and do not capture the dynamics of the system (Gebhardt et al. 2005). A monitoring program must be implemented if one hopes to obtain an understanding of trends in dynamics and processes. Monitoring, or the investigation of the status of something over time (Clarke et al. 2004), is useful in providing managers with information on which long-term plans can be based because trends over time can be used to infer future conditions. A superior monitoring program should address both biotic and abiotic environments, as well as function across multiple scales. From an ecological perspective, a monitoring program can also provide insight into cause and effect relationships between environmental stressors and ecosystem responses (Reeves et al. 2004). However, this is not always the end result of monitoring. Though statistical associations can be established between indicators, Sutter (2001) suggests that monitoring does not necessarily address causation.

1.3 WHY MONITOR ANURANS?

Despite continued debate on the subject (Blaustein 1994), most researchers are convinced that significant amphibian population declines have occurred both locally and globally in recent times (Alford and Richards 1999; Blaustein et al. 1994; Skerratt et al. 2007). As a result, approximately one-third of the world's 6000 amphibian species have now been classified as threatened (Stuart et al. 2004). Biological and behavioural characteristics of these species may be contributing to the perceived decline. Amphibians, for instance, tend to have relatively narrow habitat tolerances that can amplify the effects of habitat loss (Findlay and Houlahan 1997). In addition, a biphasic life cycle requires access to both aquatic environments and suitable terrestrial habitat for post-metamorphic stages (Todd et al. 2009). The effects of habitat loss are also exacerbated by the limited dispersal abilities of most amphibian species (Gibbs 1998; Bowne and Bowers 2004). Beyond these innate biological explanations, several potential external causes of this global decline have been proposed by researchers. Chief among these causes are habitat loss, disease, toxins in the environment, and large-scale climate change (Alford and Richards 1999).

Most threats to amphibian populations are either directly or indirectly related to habitat loss and fragmentation (Cushman 2006). Not only does habitat loss directly remove breeding and overwintering sites (Hecnar and M'Closkey 1996), but the resulting disturbance and loss of habitat connectivity in the surrounding landscape matrix can have a detrimental effect on isolated populations. Connectivity within a landscape is particularly important for anuran survival because populations have been shown to experience frequent local extinction and turnover (Hecnar and M'Closkey 1996). Positive relationships have been found between amphibian population and the area of forest in the surrounding landscape (Knutson et al. 1999) and between species richness and increased forest cover (Gibbs 1998), while negative relationships have been found between amphibian populations and both urban land (Knutson et al. 1999) and landscape barriers such as roads (Fahrig et al. 1995). This interference in the dispersal of individuals is of concern because studies have revealed considerable reductions in dispersal success and juvenile survival in fragmented landscapes (Cushman 2006; Becker et al. 2007).

Anthropogenic toxins in the environment can also have detrimental effects on amphibians. Beyond any immediate lethal consequences, environmental toxins can affect anuran populations by increasing the susceptibility of both young and mature individuals to disease (through immuno-suppression), reducing reproductive potential, retarding the growth and development, and causing physical deformities that could decrease the ability to avoid predation (Carey and Bryant 1995; Gilbertson et al. 2003). Biological pathogens can have similar health-related impacts. Viruses and fungal infections can cause “waves” of disease to move rapidly through amphibian populations (Laurance et al. 1996). They can also linger within a population, remaining present in healthy individuals and becoming active only when sub-lethal stress suppresses immune capabilities (Bishop 1992). The most critical microbiological threat currently facing anuran populations is Chytridiomycosis, a skin infection caused by the fungal pathogen *Batrachochytrium dendrobatidis* (Longcore et al. 1999). This pathogen has been identified as the most significant disease affecting biodiversity in vertebrates (Skerratt et al. 2007) due to its impact on amphibians worldwide. Though Chytridiomycosis is not of critical concern in Ontario as of 2009, the associated fungus has been found to be widespread in anurans of the Northeastern United States as recently as 2007 (Longcore et al. 2007).

There is evidence in the literature that climate change is influencing anuran behavior and phenology. Broad patterns seem to suggest that some temperate-zone anuran populations are now breeding earlier in the year (Blaustein et al. 2001). In a historical study near Ithaca NY, four of six monitored frog species were found to be calling 10 to 13 days earlier in 1999 than they were in 1900 (Gibbs and Breisch 2001). Data suggested that this temporal shift was due to daily temperature increases during 5 of the 8 months key to gametogenesis. A similar climate-influenced pattern of earlier calling in amphibians was detected by Beebee (1995) in Great Britain. In addition to temperature, it has been proposed that large-scale decreases in rainfall or snow could also affect anuran survivorship and reproductive behavior by causing ponds to fill later in the season and persist for a shorter period. This would provide less time for metamorphosis and potentially lead to increased competition and predation as individuals become concentrated at fewer breeding sites (Donnelly and Crump 1998). In modeling experiments, the ability of anurans to maintain global population levels in the face of climate alteration was found to be highly dependant on long-range dispersal abilities of the affected species (Araujo et al. 2006). Warming in cooler northern ranges would create new colonization opportunities for species with a limited habitat range but unlimited dispersal abilities.

1.4 THE LANDSCAPE OF THE CREDIT RIVER WATERSHED

The Credit Valley Conservation Authority is a municipal-level agency mandated with environmental and hydrological monitoring and regulation within the borders of the Credit River watershed. This watershed encompasses approximately 980 square kilometres of land in southern Ontario, Canada (Credit Valley Conservation 2003). The River flows southeast for nearly 100 kilometres from its headwaters in Orangeville to its drainage point at Lake Ontario. There are 21 subwatersheds within the main watershed boundaries, as well as zones which drain directly into Lake Ontario. These subwatersheds contain almost 1500 kilometres of streams and creeks that empty into the Credit River (Credit Valley Conservation 2003). As of 2007 the watershed was composed of 23% natural communities, 10% successional communities, 37% agricultural land use, and 29% urban area (Credit Valley Conservation 2007c). Approximately 6% of the watershed (about 32% of the natural cover) is classified as wetlands, while 12% of the watershed (63% of natural cover) is forested (Credit Valley Conservation 2007c).

Though encompassing many unique landscape formations, the Credit River watershed can be divided into three main physiographic zones based on topography, morphology, geologic origins, and the boundaries of individual subwatersheds. The Lower physiographic zone is highly urbanized, containing over 85% of the human population of the watershed. The soils in the Lower watershed are primarily clay, resulting in poor surface water infiltration (Credit Valley Conservation 2007b). The topography of this area is relatively flat and has, in places, been significantly altered by human development. The amount of both wetland and forest cover in the Lower watershed has been classified as poor when applying proposed government guidelines (Environment Canada, 2004). The Middle zone contains the Niagara Escarpment, a region of steep slopes, rocky outcrops, and thin soil (Credit Valley Conservation 2005). These topographic factors, along with limited urbanization and the implementation of protective legislation, have resulted in the Middle watershed containing the greatest proportion of natural cover in the watershed. The Upper physiographic zone lies above the escarpment and is characterized by hilly moraines, glacial spillways, and permeable loamy soils (Credit Valley Conservation 2005; Credit Valley Conservation 2007b). Agricultural land use dominates portions of the Upper watershed, though it also consists of several large wetland complexes and a protected moraine (Oak Ridge) that is shared with the Middle zone.

2.0 OBJECTIVE: MONITORING QUESTION

The purpose of the anuran component of the terrestrial monitoring program at Credit Valley Conservation is a) to detect long-term trends in a selected set of indicators in order to identify trends in anuran population health at the watershed scale, b) to infer the impact of these trends on overall watershed and aquatic health, and c) to provide meaningful data on which watershed management decisions can be based. The central question being asked is as follows:

ARE POPULATION TRENDS OF ANURAN SPECIES IN THE CREDIT RIVER WATERSHED STABLE?

More specifically, 1) are species richness and anuran abundance changing through time; 2) have anuran species richness and abundance differed spatially in the watershed; 3) has anuran site occupancy changed through time; and 4) has anuran community similarity changed through time. The framework for anuran monitoring at this stage in the program is reviewed in Table 1. The analysis techniques listed in the table are those chosen to fit the quality and scope of data currently available. This analytical framework may change as the monitoring program progresses and data quality improves.

Table 1. Anuran Monitoring Framework.

Monitoring Question	Monitoring Variable(s)	Unit of Measurement	Analysis Method
Part 1: Anuran Species Richness			
Is anuran species richness changing over time in the Credit River watershed?	Total number of species detected in the watershed	Count	N/A
	Number of species detected per monitoring site	Count	- Temporal Analysis: Friedman Test - Power Analysis: Linear Regression
Are there spatial differences in anuran species richness in the Credit River watershed?	Total number of species detected in each physiographic zone	Count	- Chi-Square test for independence
	Mean number of species detected per site in each physiographic zone.	Count	- Spatial Analysis: Kruskal-Wallis
Is Anuran richness correlated with landscape parameters in the Credit River watershed?	Mean number of species detected per site compared to three landscape metrics	Count	- Correlation: Spearman Rank
Is anuran richness correlated with vegetation parameters in the Credit River watershed?	Mean number of species detected per site compared to several wetland vegetation parameters	Count	- Correlation: Spearman Rank
Part 2: Anuran Site Occupancy			
Is the proportion of sites occupied by anurans changing over time in the Credit River watershed?	Proportion of total sites occupied	Proportion (%)	-Trend Analysis: Cochrane-Armitage - Power Analysis: Cochrane-Armitage
Are there spatial differences in anuran site occupancy in the Credit River watershed?	Number of sites occupied per physiographic zone	Count	- Chi-Square test for independence
Is there a relationship between anuran site occupancy and landscape metrics in the Credit River watershed?	Site occupancy by anurans tested against three landscape metrics	Presence/Absence	- Trend Analysis: Logistic Regression - Power Analysis: Logistic Regression
Is there a relationship between anuran site occupancy and vegetation parameters in the Credit River watershed?	Site occupancy by anuran species tested against several wetland vegetation parameters	Presence/Absence	- Trend Analysis: Logistic Regression - Power Analysis: Logistic Regression

Monitoring Question	Monitoring Variable(s)	Unit of Measurement	Analysis Method
Part 3: Anuran Site Occupancy			
Is anuran abundance changing over time in the Credit River watershed?	Proportion of sites at which full chorus was detected	Proportion (%)	- Trend Analysis: Cochrane-Armitage - Power Analysis: Cochrane-Armitage
Are there spatial differences in anuran abundance in the Credit River watershed?	Mean full chorus detections per site per physiographic zone	Count	Spatial Analysis: Kruskal-Wallis
Is there a relationship between anuran abundance and landscape metrics in the Credit river watershed?	Mean full chorus detections per site tested against three landscape metrics	Count	- Trend Analysis: Logistic Regression - Power Analysis: Logistic Regression
Is there a relationship between anuran abundance and vegetation parameters in the Credit River watershed?	Mean full chorus detections per site tested against several wetland vegetation parameters	Count	- Trend Analysis: Logistic Regression - Power Analysis: Logistic Regression
Part 4: Anuran Community Similarity			
Is anuran community similarity changing over time in the Credit River watershed?	Jaccard's Similarity Index	Index	- Temporal Analysis: N/A - Power Analysis: Linear Regression

3.0 METHODS

3.1 ANURAN SURVEYS

Data were collected for six years (2003 to 2008) at 26 wetland monitoring sites distributed throughout the Credit River watershed (Figure 1). The 26 anuran monitoring sites are listed in Appendix 1. Eight of these sites were located in the Lower physiographic zone, eight were located in the Middle physiographic zone, and ten were located in the Upper zone. Data collection followed protocols of the Marsh Monitoring Program (Konze and McLaren 1997) and involved three yearly visits to each site spaced a minimum of 15 days apart. Surveys were conducted within the last 15 days of April, of May, and of June in order to capture both early and late breeding species. Monitoring activities during each 15 day period were split into four specific groups of stations, often referred to as travel routes. Each "route" was surveyed on a single night and involved a specific sequence of sites. These surveys began no less than half an hour after sundown and were completed by midnight. Monitoring was restricted to nights with winds measuring less than "3" on the Beaufort scale (less than 12 km/hr) and to nights consistent with a set temperature established for each survey period.

The monitoring station at each wetland site consisted of an area in front of the observer that was semi-circular in shape and roughly 100 m in radius (Figure 2). Protocols specified a three minute survey period at each station during which an estimate of anuran numbers could be established through auditory detection of calls. Calls estimated to be originating beyond the 100m radius were still counted, but noted in the data as being "distant". The estimate of anuran abundance was based on a 4-level ordinal calling code recommended by the marsh monitoring program. A calling code of "0" noted that no frog or toad was heard. A code of "1" (C1) was assigned to a species when calls were not overlapping and, as a result, a specific count of individuals could be achieved. A code of "2" (C2) was an estimate count of a species because some calls were overlapping and not all individuals could be distinguished. A code of "3" (C3) indicated a full chorus, in which calls were continuous and overlapping. It was still possible to distinguish between anuran species when full chorus was present, but it was not possible to estimate the number of individuals. Additional collected information, recorded on-site with a portable weather station (Kestral 3000), included air temperature, relative humidity, maximum wind speed, average wind speed, and wind direction. Water temperature was measured using a digital thermometer (Fisher Scientific).

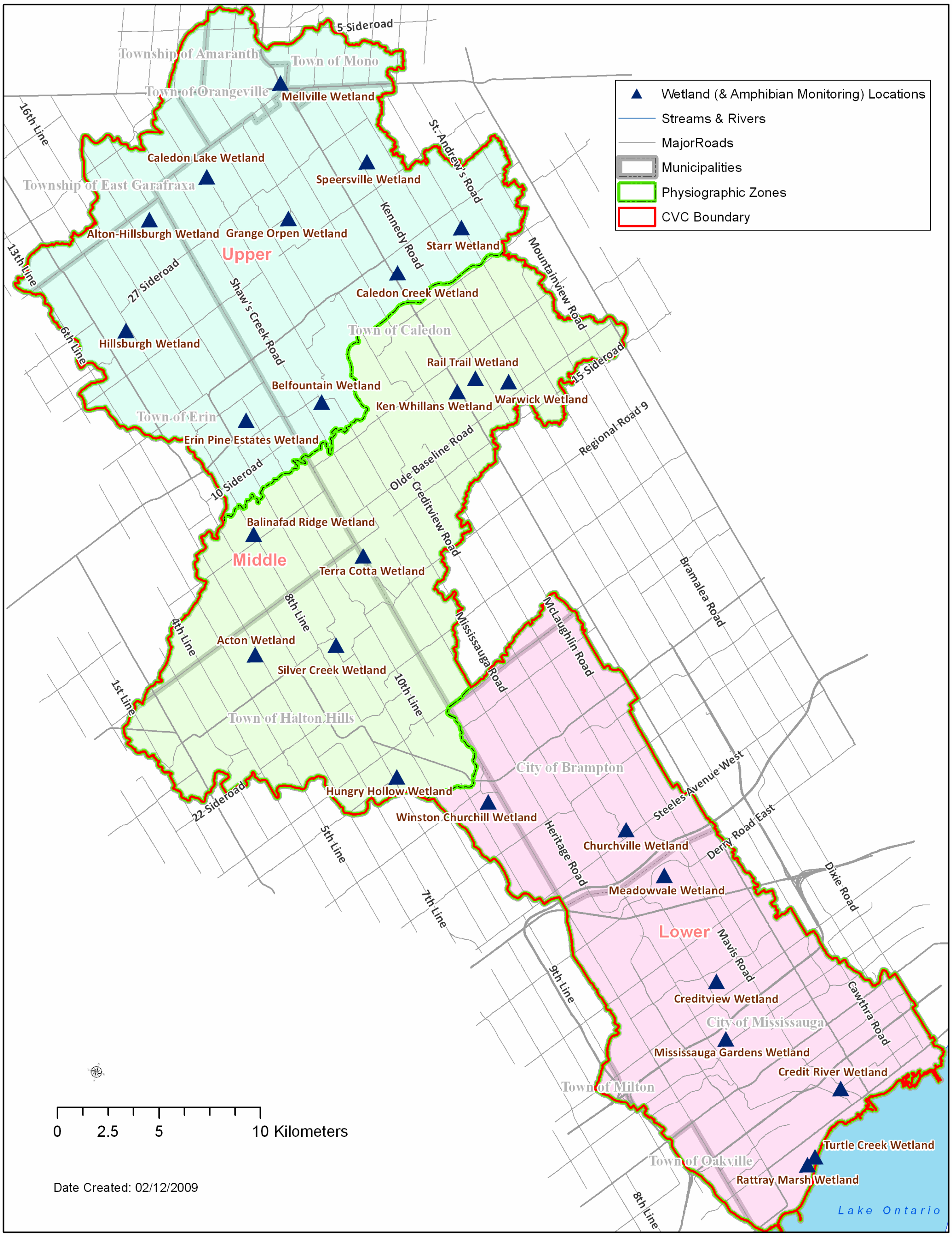

Figure 1. A map of the Credit River Watershed and the jurisdiction of the Credit Valley Conservation authority showing the location of 26 anuran monitoring stations. Physiographic zones have been separated using colour.

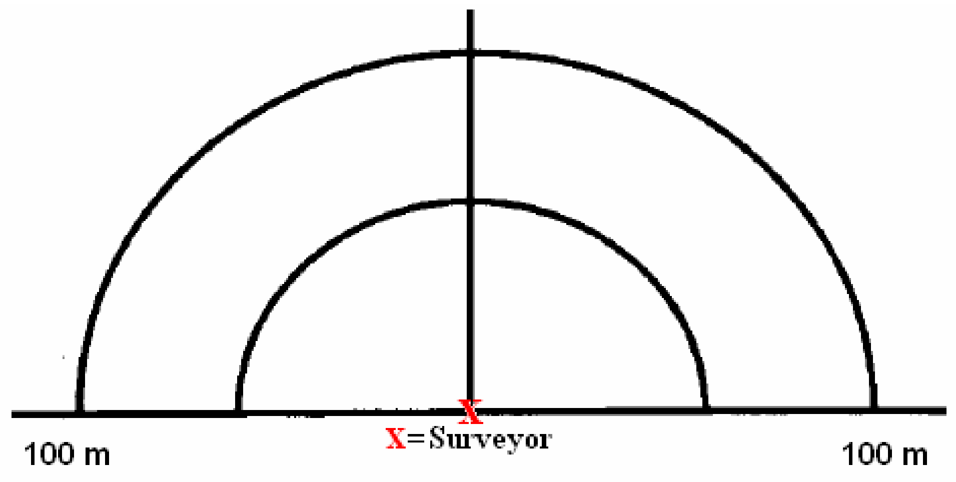

Figure 2. An anuran monitoring station as defined by the Marsh Monitoring Program. The curved lines show the semi-circular boundaries of the station.

3.2 LANDSCAPE PARAMETERS

The three landscape parameters referenced within this report are habitat patch size, the distance from monitoring plots to the nearest road, and the percent urban land use within 2 kilometres of the plot. They are based on orthophotography collected between 2005 and 2007 over the Credit River watershed. Aerial photographs were interpreted and analysed using ArcGIS, a software package capable of generating information on patterns, structure, and change within a landscape (Credit Valley Conservation Authority 2007a). The measure of "habitat patch size" refers to the total size of the contiguous area of natural habitat within which an anuran monitoring station was nested. The Ecological Land Classification (ELC) community series included in the definition of natural habitat are listed in Appendix 2. Habitat patch size was selected as a landscape parameter because several studies have found negative relationships between anuran population trends and both habitat loss and landscape fragmentation (Cushman 2006). The "distance to the nearest road" measure was chosen because the presence of roads can often limit the movement of species (Gibbs 1998), lead to increased anuran mortality rates (Fahrig et al. 1995; Hels and Buchwald 2001; Mazerolle 2004), or result in altered reproductive behaviour (Eigenbrod et al. 2009; Parris et al. 2009). The percent urban land use within 2 kilometres was measured from the outside of a 30 metre buffer surrounding the approximate centroid of each wetland site. The ELC land use types included in the definition of urban land use are listed in Appendix 2. This matrix measurement was chosen because studies have found negative associations between urban expansion and both anuran abundance (Knutson et al. 1999) and the presence of suitable anuran habitat (Baldwin and DeMaynadier 2009). In addition, Findlay and Houlahan (1997) found that taxa richness increased with an associated increase in the amount of forest cover within 2 kilometres of wetland sites in southeastern Ontario.

3.3 VEGETATION PARAMETERS

Measures of vegetation used in this report are based on data collected through the wetland component of the Terrestrial Monitoring program at Credit Valley Conservation. Data were collected from ground vegetation and regeneration subplots arranged along a 50 metre hydrological gradient at selected wetland monitoring plots. Most anuran monitoring stations had been intentionally paired with a wetland vegetation monitoring plot so that direct comparisons could be made between fauna and vegetation. Vegetation parameters were separated into three general groups (Table 2). Parameters in the ground vegetation group included all herbaceous vegetation regardless of height, as well as all woody vegetation less than 16 cm in height. Parameters in the Regeneration group contained woody vegetation greater than 16 cm in height and less than 10 cm DBH, excluding woody vines. The parameter in the combined vegetation group included all ground vegetation as well as all regenerating woody vegetation greater than 16 cm in height.

Table 2. Wetland vegetation parameters used in comparisons with anuran monitoring data.

Data Group	Parameters
Wetland Ground Vegetation	- Total species per site - Total native species per site - Total non-native species per site - Proportion of non-native species per site
Wetland Woody Regeneration	- Total species per site - Total native species per site - Total non-native species per site - Proportion of non-native species per site
Wetland Combined Vegetation	- Mean coefficient of conservatism score per site

It has been shown that vegetation density and structure can influence amphibian survival (Purrenhage and Boone 2009). Ground vegetation was compared to anuran parameters because the associated plants in this category are the dominant living component of most wetland communities and are likely to have an influence on ground dwelling fauna. Regenerating woody vegetation was compared to anuran parameters because climbing frog species may show a preference for wetlands with a greater variety of vertical structure. The mean Coefficient of Conservatism (mCC), the only combined vegetation parameter, was chosen to examine the influence of vegetation quality on anurans. mCC is a measure based on species-specific scores listed in the Floristic Quality Assessment System for Southern Ontario (Oldham et al. 1995). These Coefficient of Conservatism scores, ranging from zero to ten and assigned only to native species, are estimates of sensitivity and habitat fidelity. The higher the mean Coefficient of Conservatism of a given site, the greater the habitat sensitivity and specificity of the constituent native plants. The ubiquitous Canada Goldenrod (*Solidago canadensis*), for instance, has been assigned a score of 1, while the rare alvar species Dwarf Lake Iris (*Iris lacustris*) has been assigned a score of 10.

3.4 DATA PREPARATION

All frog observations not confirmed to species (including those labelled as "unknown") were removed from the data set before analysis. Observations of Pickerel Frog (*Rana palustris*) were included in species richness and abundance analyses but excluded from all other statistical testing because the associated sample size (only three observations over six years) was too small.

A species presence at a given monitoring station on a given year was based on any call detection of that species, regardless of calling code or the number of visits (out of three) at which it was recorded. The maximum calling code recorded for each species during the three yearly site visits was used when analysing abundance data.

The mean species richness over six years of monitoring was calculated for each site so that correlations could be made between species richness and landscape parameters. This was necessary because landscape data were not available in a year-by-year form but rather as a single "snapshot" representation of land parameters during the general period of monitoring. For logistic regression tests between site occupancy (presence/absence) over the six-year survey period and landscape parameters, a site was considered occupied if the species or group in question was detected in at least two of the six survey years. This was done to avoid the inclusion of temporally isolated occurrences.

Comparisons between anuran data and wetland vegetation parameters could only be made for the years 2005 through 2008 because vegetation community information prior to these years is considered unreliable due to staff turnover and sampling inconsistencies. In addition, comparisons with vegetation data could only be made using 18 of the 26 anuran monitoring stations because this is the number of sites at which both anurans and vegetation were surveyed for the years 2005 through 2008. For logistic regression analyses between site occupancy (species presence/absence) and vegetation parameters, a site was considered occupied if the anuran species was detected in at least two of the four survey years. For similar analyses using full chorus calls, positive detection was defined as full chorus occurring at a site during at least two of four survey years. Vegetation parameters at each site were averaged over the four years in order to be comparable to these anuran occupancy and abundance data.

The Jaccard's Similarity Index was calculated for total species richness between each set of two consecutive survey years (five calculations in total) for both the whole of the watershed and within physiographic zones. It was also calculated between the first and last survey years (2003 and 2008) for both the whole of the watershed and within physiographic zones. Finally, the index was calculated between each two-zone combination (Lower with Middle, etc.) for each separate survey year. The Similarity Index is a calculated value used to compare the species similarity between two samples. A high index value is indicative of low species turnover between tested groups. The Jaccard's Index equation is as follows:

$$Sj = \frac{a}{a + b + c}$$

Where S_j = Jaccard's Similarity coefficient
a = The number of species in group 1 and group 2 (joint occurrences)
b = The number of species in group 1 but not in group 2
c = The number of species in group 2 but not in group 1

3.5 STATISTICAL ANALYSIS

Linear regression and repeated measures ANOVA, the parametric statistical tests chosen in the original study framework to analyse anuran richness and abundance data, could not be performed at the current stage of monitoring. The small number of species detected throughout the survey, combined with a data set replete with zero or near-zero values, made it difficult to meet parametric assumptions. Data were tested for normality and homoscedacticity with Shapiro-Wilks and Levene's test for homogeneity of variance, respectively. The variance of anuran parameters was often consistent between years, but normal distributions could not be obtained even after data transformations. In addition, linear trend analyses were not performed on any variable consisting of less than ten data points because outliers in such a case would have too great an influence on test results. Parametric and non-parametric tests used to analyse anuran data at the current stage of monitoring are described below. All analysis was completed using STATISTICA version 7 (Statsoft Inc.) unless otherwise noted, with results being considered significant at $p < 0.05$.

3.5.1 Trend Analysis

Trend analysis was preformed on anuran occupancy data to examine possible sequential patterns across the six study years. Cochrane-Armitage trend analysis was employed for these tests because the anuran occurrence data was in the form of proportions (such as the percent of total sites occupied per year). The Cochrane-Armitage test is a non-parametric analysis of trend designed to examine proportion instead of count data, and is the closest proportion-based equivalent of parametric linear regression. Tests were conducted using *XL*STAT, a third party add-on to Microsoft Excel.

3.5.2 Temporal Analysis

Temporal analysis was performed in order to examine patterns in anuran richness data among the six survey years. Per-site data were analysed using the Friedman test (Zar 1974), a non-parametric equivalent of a repeated measures ANOVA which tests the null hypothesis that observations in different groups share the same median values. The Friedman test is preferable over other non-parametric options when dealing with trend data because it takes into account that values between groups (years in this case) are dependant. This is not a true trend analysis, however, because it is not testing for a sequential change in the dependant variable. It is instead testing whether there are significant differences in the median values of at least two of the groups. In lieu of post-hoc analysis, which is not available for the Freidman test, a trend must be inferred after visually examining the size and directions of any differences between individual groups.

3.5.3 Spatial Analysis

Spatial analysis was performed to examine differences in anuran data among the three physiographic zones of the Credit River watershed. The Chi-square test for independence was used to look for interaction between year and zone effects. This non-parametric test for independence is based on a probability distribution of observed and expected values between groups (year) and conditions (zone). Chi-square comparisons were completed using free software provided by Preacher (2001). If independence was shown between year and zone, physiographic zones were then compared using a Kruskal-Wallis test. This analysis of variance by ranks (Zar 1974) is a non-parametric version of a One-Way ANOVA in which treatment levels of a predictor variable (zones in this case) are independent of one another. Zone effects had to be analysed on a year by year basis because Kruskal-Wallis cannot account for year effect when all survey years are used in the same analysis. An associated post hoc test was then run to determine which zone relationships were significant.

3.5.4 Comparisons with Landscape and Vegetation Parameters

Relationships between continuous anuran variables (such as "mean species per site") and continuous landscape or vegetation parameters were examined using Spearman rank correlations. The binary nature of anuran occupancy and full chorus detection data, however, made comparisons using rank-based correlation inappropriate. Instead, binary occupancy and abundance data was tested against landscape and vegetation parameters using simple logistic regression with one continuous predictor. Though still considered a parametric option, logistic regression is a non-linear estimation method in which the binary response variable is assumed to follow a binomial distribution (Quinn and Keough 2002). The assumptions of normality and homoscedacticity do not apply as with linear regression.

3.5.5 Statistical Process Control

Statistical Process Control (SPC) is an application which uses ecological time series to develop monitoring thresholds. Though similar to regression, SPC is not based on hypothesis testing. Instead, this application seeks to identify instances when a time series exhibits non-random behaviour. A series demonstrating this non-random behaviour is considered "out of control" and, as such, unsuitable for developing a monitoring baseline. If the time series exhibits natural random variability around a reference point (usually the mean), the series is considered "in control". Data that are in control can be treated as a baseline from which monitoring thresholds can be generated. A minimum of five years of "in control" data are required to set monitoring thresholds.

SPC control charts are divided into six zones based on the mean value of a time series and its standard deviation adjusted to sample size (Figure 3). The Upper and Lower critical limits, defined as +/- 3 standard deviations from the mean, encompass the “in control” range of the time series. Control charts for stable systems will not contain any points outside the +/- 3 SD “in control” range. In such a case, the series represents appropriate reference conditions from which the +/- 3 SD critical limit thresholds can be adopted. Series with points outside the “in control” range may represent an ecosystem under stress or moving slowly toward some alternate state. A data set in this type of flux would not provide appropriate reference conditions on which thresholds could be based.

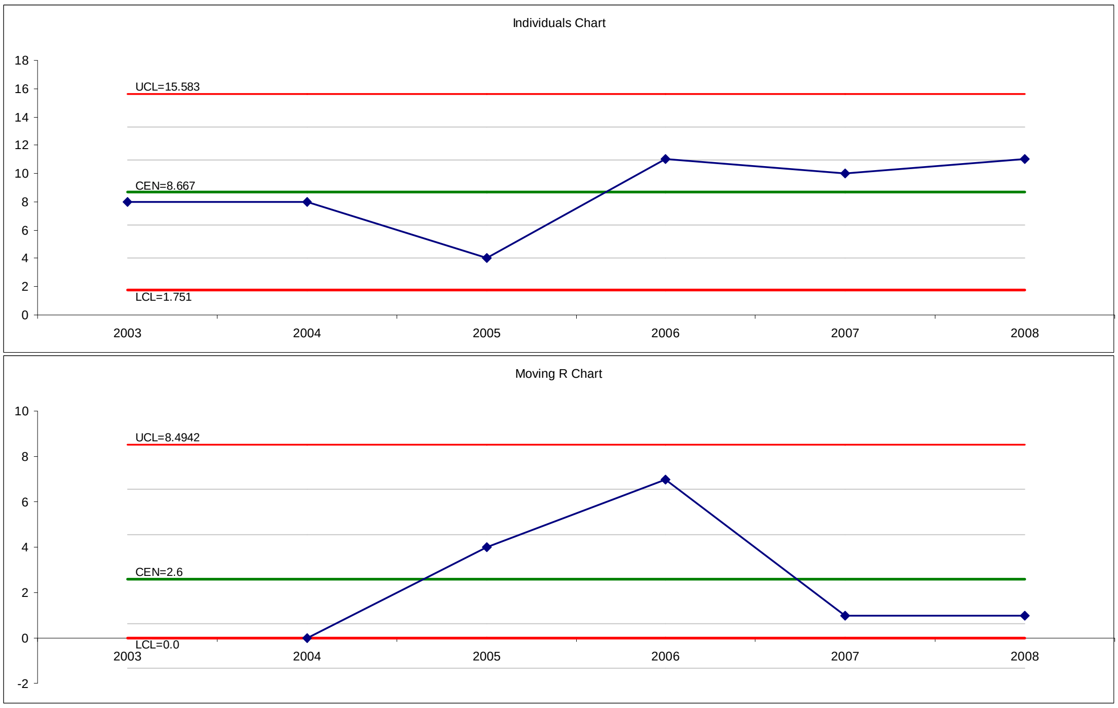

Figure 3. Statistical Process Control charts for Leopard Frog site occupancy. Both the individual moving average chart (showing the parameter values for each year) and the Moving Range chart (showing the amount of change in the parameter between years) are “in control” from 2003 to 2008. The green line indicates the time series mean and the red lines show the upper and lower critical limits (+/- 3standard deviations). Since the data set is in control, the mean from 2003-2008 can be used as a baseline for future monitoring.

Values that exceed the critical limits are associated with significant increasing or decreasing trends and should always be investigated thoroughly. Often in monitoring, however, early detection of changing trends or significant decline in parameters is desired in order to recognise problems before the effects become irreversible. To address these early detection concerns, the upper and lower warning limits on control charts are represented by +/- 2 standard deviations from the mean. Trends of series that exceed the warning limits but not the critical limits should be tracked closely for further changes. A time series with values falling within the warning limits is exhibiting natural variability and is considered to be stable and not of concern based on SPC alone. All statistical control charts for anuran data were generated using SPC XL software package, a third-party add-on application to Microsoft Excel.

3.5.6 Power Analysis

Power analysis uses baseline monitoring data – such as the "in control" time series detected using SPC – to determine if additional sampling is required to detect significant trends or desired effect sizes. In ecological monitoring, additional sampling can be interpreted as an increase in the number of monitoring sites at which data is collected. Effect size refers to the detectable change in the time series over a given interval, and is often related to a specific monitoring threshold. Power analysis is most frequently used to calculate the sample size required to detect a significant change, or to calculate a specific effect size given the variance of the time series being considered. It can also be used to determine the maximum effect size detectable in a time series given the scope of sampling. All power analysis presented in this report were conducted using PASS (NCSS Inc.). The form of Power Analysis applied to a given parameter was dependant on the specific statistical test being used to examine the associated data. Tests were run using a set power level (1-beta) of 90% and a significance level (alpha) of 20%, the analysis design recommended by an external statistician (P. Zorn, personal communication). Effect sizes were designed to be detected over a five year monitoring period.

Power analysis of the anuran data was conducted for the linear trend analyses stipulated in the original monitoring framework and for the Cochrane-Armitage and logistic regression analyses actually applied in this report. Though linear regression was not applied to monitoring data at this time, linear trend analysis was still examined for power because monitoring thresholds (1 SD, 2 SD, and 3 SD) were calculated for the planned linear analyses using SPC. Power analysis specific to both linear regression and Cochrane-Armitage allowed for the calculation of required site numbers for detecting SPC thresholds as well as the calculation of minimum effect sizes. Power analysis specific to logistic regression requires continuous covariates (the predictor variables) to be normally distributed, so landscape and vegetation parameters (the covariates) were - when required - normalized by transformation before power analysis proceeded. It should be noted that the actual logistic regressions reported in the results section were performed with raw (pre-normalized) covariates. Though not necessarily identical in results, the normalized and non-normalized results should be comparable because covariate transformations did not change the significance status of relationships with binary anuran data. PASS presents the effect size for logistic regressions as an odds ratio, a descriptive relationship with limited interpretive value. As a result, power analysis was only used to calculate the minimum number of monitoring sites required to detect a significant logistic relationship between the binary anuran parameter and chosen landscape or vegetation covariate.

4.0 RESULTS AND DISCUSSION

4.1 DESCRIPTIVE RESULTS

4.1.1 Identified Species

Nine anuran species were detected at monitoring stations during the study period (Table 3). The associated species of conservation concern (SCC) tier rankings listed in Table 3 are based on rarity and status of concern within the boundaries of the Credit River watershed (Credit Valley Conservation 2009), while the S ranking of species is a provincial signifier assigned by the Ministry of Natural Resources. Tier 3 species American Toad (*Bufo americanus*), Gray Tree Frog (*Hyla versicolor*), Green Frog (*Lithobates clamitans*), Leopard Frog (*Rana pipiens*), and Spring Peeper (*Pseudacris crucifera*) are Species of Urban Interest that are thought to be relatively secure in more natural environments. Tier 2 species Bullfrog (*Rana catesbeiana*), Pickerel Frog (*Rana palustris*), and Wood Frog (*Lithobates sylvaticus*) are known to be locally rare or are exhibiting signs of decline. Western Chorus Frog is ranked as Tier 1 because it is formally considered a species of conservation concern. This denotes that it has been designated as threatened or endangered in Canada, or has been designated rare at a provincial level. The only anuran previously detected in the watershed but not appearing in the survey records was Mink Frog (*Rana septentrionalis*), a Tier 2 species listed as "secure" provincially.

Table 3. Anuran species detected at anuran monitoring stations in the Credit River watershed from 2003 to 2008.

Common Name	Latin Name	S Ranking (Provincial)	SCC Tier (Local)
American Bullfrog	*Rana catesbeiana*	Uncommon	2
American Toad	*Bufo americanus*	Secure	3
Gray Tree Frog	*Hyla versicolor*	Secure	3
Green Frog	*Lithobates clamitans*	Secure	3
Northern Leopard Frog	*Rana pipiens*	Secure	3
Pickerel Frog	*Rana palustris*	Uncommon	2
Spring Peeper	*Pseudacris crucifera*	Secure	3
Western Chorus Frog	*Pseudacris triseriata*	Uncommon	1
Wood Frog	*Lithobates sylvatica*	Secure	2

Gray Tree Frog, Spring Peeper, and Western Chorus Frog are all classified as tree frogs, or Hylidae. They can be distinguished by the presence of discs on the tips of their toes, a morphological adaptation that assists in vertical movement (Harding 1997). Despite this climbing ability, tree frogs such as Western Chorus Frog may never leave near-ground terrestrial habitats for arboreal ones (MacCulloch 2002). American Bullfrog, Green Frog, Northern Leopard Frog, Pickerel Frog, and Wood Frog are known as true frog species, or Ranidae. These frogs are usually associated with aquatic habitats, though some species (such as Wood Frog) may favour wooded terrestrial habitats during non-breeding periods (MacCulloch 2002). As a member of the Bufonidae family, American Toad is distinguishable from the eight recorded frog species by its short legs, stocky body shape, and thickened skin that protects against water loss (Harding 1997).

4.1.2 Anuran Site Occupancy

It is not surprising that Green Frog and Spring Peeper were detected at the greatest number of monitoring sites throughout the watershed (Figure 4), as these are considered two of the most abundant frog species in the Great Lakes region (Harding 1997). American Bullfrog, Western Chorus Frog, and Wood Frog – the three recorded species assigned an SCC Tier ranking of 2 (locally rare) - occupied fewer total sites than those species listed as Tier 3. Pickerel Frog was only recorded on multiple years at a single site (Silver Creek). It should be noted that "detection", in this case, has been defined as presence at a site in at least two of the six survey years. Based on a non-statistical observation of trends in occupancy patterns of individual species (Appendix 3), Green Frog and Spring Peeper were also the only anurans to exhibit relatively stable levels of site occupancy over the whole of the watershed. Year-to-year occupancy numbers for the other seven species were notably more unstable. American Toad had the most erratic between-year occupancy of any detected species, both at the whole of the watershed and within physiographic zones. This may be a product of low site fidelity in the reproductive behaviour of this species because Petranka and Holbrook (2006) found that American Toads rarely use the same breeding sites from year to year. It should also be noted that Mink Frog, though not detected in the surveys, was frequently heard by other monitoring crews in the upper portions of the watershed.

The above conclusions assume that the detection probability for all species at all sites was 1. In reality, detectability may vary between species and will usually be less than absolute. MacKenzie et al. (2002) recommend a likelihood-based method for estimating anuran occupancy rates when detection probability is less than 1. As an example, they determined that estimated toad occupancy using the likelihood model was 44% higher than the proportion of sites at which the species was actually identified. Evidence of competition between species for site occupancy was not examined at this point in monitoring, though patterns of intra-species competition may not be significant when other site-level characteristics are considered. In a Swiss study, Buskirk (2005) found that occurrence of amphibian species was positively correlated with the densities of other species and concluded that competition was a less important predictor than variation in site quality.

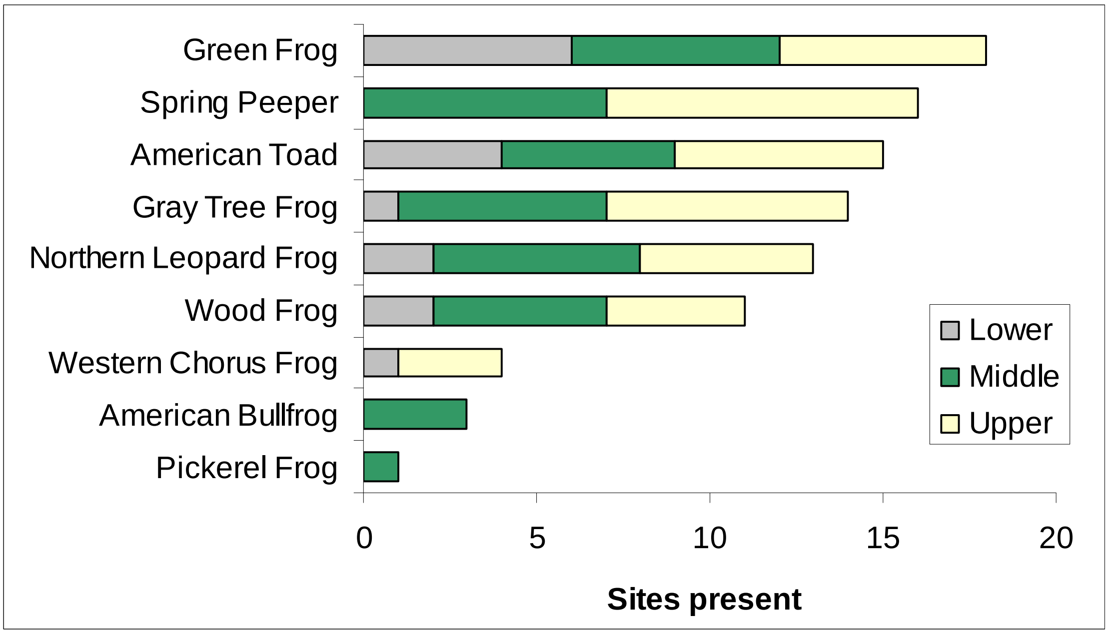

Figure 4. The total number of monitoring sites each anuran species occupied over the course of the study period, separated by physiographic zone. Occupancy has been defined as presence at a site in at least two of the six survey years.

4.1.3 Anuran Abundance

The call-based method of estimating anuran abundance places strict limitations on the type of population data that can be reported and analysed. Foremost, the semi-ordinal nature of the call numbering system makes the call values non-proportional. This prohibits any combination of call values for analytical purposes, including the calculation of such parameters as “Mean calling level per species”. It is relatively easy to obtain an accurate count when individuals are spread out evenly across the sampling area but difficult if the same numbers of individuals are clustered. In addition, call counts provide no information on age structure or caller health (Van Wieren and Zorn 2005). Clearly, surveys were dominated by the detection of call code one, or individual calls (Figure 5). This level was nonetheless excluded from analysis because its detection, though providing the desired individual counts, tells only part of the population story. The other part is hidden within the records of call codes two and three. Accurate individual counts are obscured by the overlapping calling associated with these two higher levels, so the three call codes are not relatable in terms of establishing complete anuran abundance numbers. However, full chorus detection can be used to identify wetland sites with large meta-populations of certain species or to identify those species which occur at the highest numbers in the watershed. These points form our justification for including only call code three in all abundance analyses.

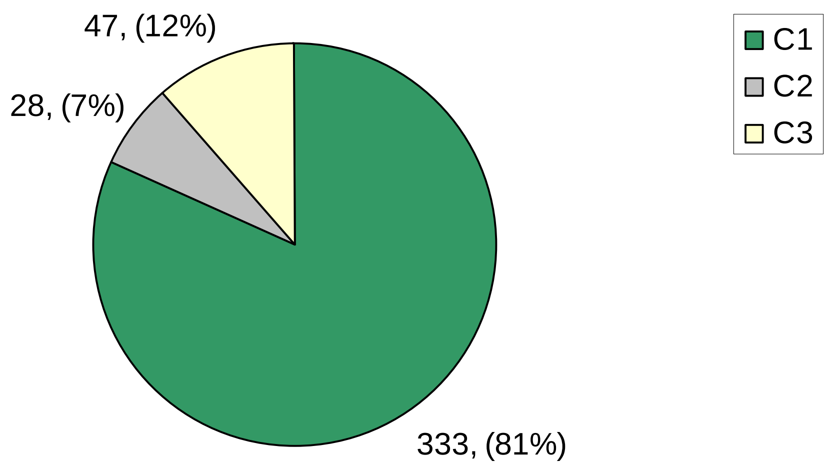

Figure 5. The total number and proportion (%) of calls detected in each of three calling levels, based on the maximum level detected per site per year.

Relative rankings of anuran species based on call detections (Figure 6) are similar to the rankings by site occupancy (Figure 4) save that Spring Peeper outranks Green Frog in terms of total detections. American Toad had a relatively low total detections count given the species' high occupancy ranking, suggesting that individuals are well dispersed throughout the watershed but not locally abundant. The breakdown of total full chorus detections (n = 47) by anuran species (Figure 7) shows that Spring Peeper alone was responsible for 68% of all full chorus detections while tree frog species together were responsible for over 90% of full chorus detections.

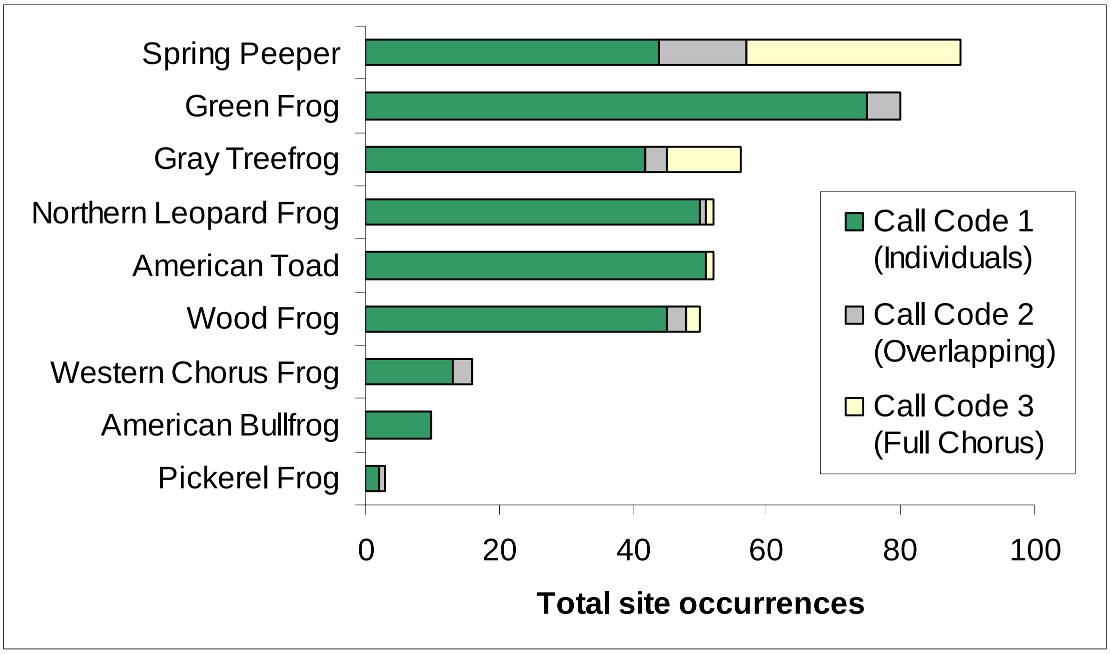

Figure 6. The distributions of C1, C2, and C3 call detections as separated by anuran species. Data is presented for all detections from 2003 to 2008 (408 in total), based on the maximum level detected per site per year.

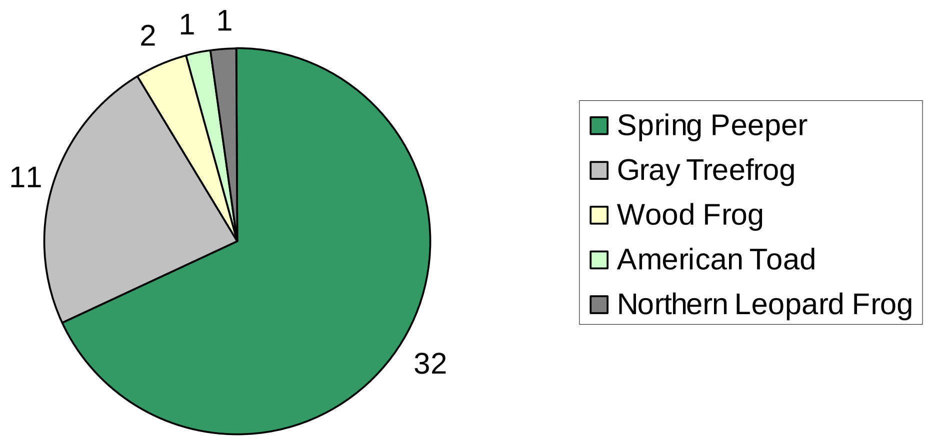

Figure 7. The distributions of C1, C2, and C3 call detections as separated by anuran species. Data is presented for all detections from 2003 to 2008 (408 in total), based on the maximum level detected per site per year.

4.1.4 Anuran Health at Individual Monitoring Sites

The watershed health perspectives adopted by the CVC integrated monitoring program require that results presented in Parts 1 through 4, along with the statistical process control and power analysis sections, focus primarily on temporal and spatial trends at the level of either the entire watershed or within and between the three physiographic zones. However, it seems pertinent to highlight some individual stations (Table 4) that may in the future function as reference sites. The qualitative "Healthy" and "Unhealthy" rankings are based on high or low values of mean per-site species richness, mean per-site full chorus detection, and the number of survey years in which anurans occupied the sites. Appendix 4 summarizes anuran parameters for each individual monitoring station in the study.

Table 4. The monitoring sites with the healthiest and unhealthiest anuran communities. [b]

Zone	Monitoring Site	Mean Species Richness [a]	Years Occupied	Mean Full Chorus Detections [a]
		Healthiest Anuran Communities		
Lower	Winston Churchill Wetland	4	6	0
Middle	Balinafad Ridge	5	6	1
Both Upper and Whole of the Watershed	Speersville Wetland	5.17	6	1.33
		Unhealthiest Anuran Communities		
Both Lower and Whole of the Watershed	Mississauga Gardens	0.33	2	0
Middle	Hungry Hollow Wetland	0.83	4	0
Upper	Hillsburgh Wetland	0.83	3	0
	Grange Orpen Wetland	0.67	3	0

[a] Per year, over six years
[b] Rankings are qualitative but based on high or low values of mean per-site species richness, mean per-site full chorus detection, and the number of survey years in which anuran occupied the sites.

The healthiest monitoring site in the whole of the watershed was Speersville wetland, a marsh located in the Upper physiographic zone. An isolated location, in combination with the permanent presence of water, may be influencing the ranking of this site. The healthiest site in the Lower physiographic zone (Winston Churchill Wetland) is also relatively isolated, contains permanent water, and is located in a portion of that zone with the least amount of urban influence. Conversely, the unhealthiest site in the Lower zone (Mississauga Gardens) is located close to a highway and in the Middle of a major urban centre. Hungry Hollow, the unhealthiest site in the Middle physiographic zone, is in a moist regenerating lowland that has little permanent water present and is at times dominated by red ants. These ants produce a stinging bite and have the potential to ward off other fauna species (Todd et al. 2008). Balinifad ridge may be the healthiest site in the Middle watershed due to the presence of ample permanent water and a variety of vegetation, while Grange Orpen wetland may be one of the unhealthiest sites in the Upper watershed because conditions there were often quite dry. Exogenous or site-level factors influencing the ranking of Hillsburgh Wetland remain unclear.

Identifying populations of species of concern in the Lower watershed is important from a conservation perspective due to both the scarcity of natural area in that zone and the threats to sensitive species presented by continued urbanization. Western Chorus

Frog, a locally-ranked species of concern (Credit Valley Conservation 2009), was detected each survey year at Creditview Wetland in the urbanized Lower zone. This is unexpected given that Creditview is a small patch of natural habitat isolated within a densely populated urban community. The site is likely supporting a remnant population left over from habitation that occurred before development. The persistence of Chorus Frog at this site could be due to restoration efforts and an absence of direct disturbance, as the wetland is fenced-off and public access is restricted. Lower levels of direct disturbance, along with proximity to the Middle watershed, may also be responsible for the fact that Winston Churchill Wetland was the only site in the Lower physiographic zone at which both Gray Treefrog and Spring Peeper were detected over the six survey years.

4.2 TEMPORAL ANALYSIS

4.2.1 Anuran Species Richness

Monitoring Question: Is anuran species richness changing over time in the Credit River Watershed?

- Trend analysis could not be applied to total and mean richness at this time.
- There was a significant difference in mean species richness between at least two of the survey years.

Trend analysis could not be performed on total species counts at either the watershed level or within zones due to a deficit of data points (only six over the survey period). There were no observable trends in the richness data over the six year study period (Figure 8). The slight increase in total species observed in 2004, 2005, and 2008 was due to the detection of Pickerel Frog during those three years. Aside from this anomaly, however, each of the other eight anuran species found in the watershed were detected during every year of the study. These results are consistent with a study of anuran richness in southwestern Ontario ponds by Hecnar and M'Closkey (1998), which found no significant change in regional species richness over time. Figure 8 also shows the total number of anuran species detected in each physiographic zone by year. The lower richness in the Upper zone in 2005 is due to the absence of American Toad and Northern Leopard Frog that year.

There appeared to be no strong trend in mean species richness over the study period either watershed-wide or within physiographic zones (Figure 9). The Friedman test did show, however, that there was a significant difference in mean per-site species richness (watershed-wide) between at least two of the six survey years ($X^2 = 13.514$, $p = 0.019$). This difference was likely between the lower means in 2004/2005 and the higher mean in 2008. Friedman tests on individual physiographic zones also found that there was a significant difference in per-site species richness in the Upper watershed between at least two of the six survey years ($X^2 = 15.608$, $p = 0.008$), likely a result of the decreased richness in 2005.

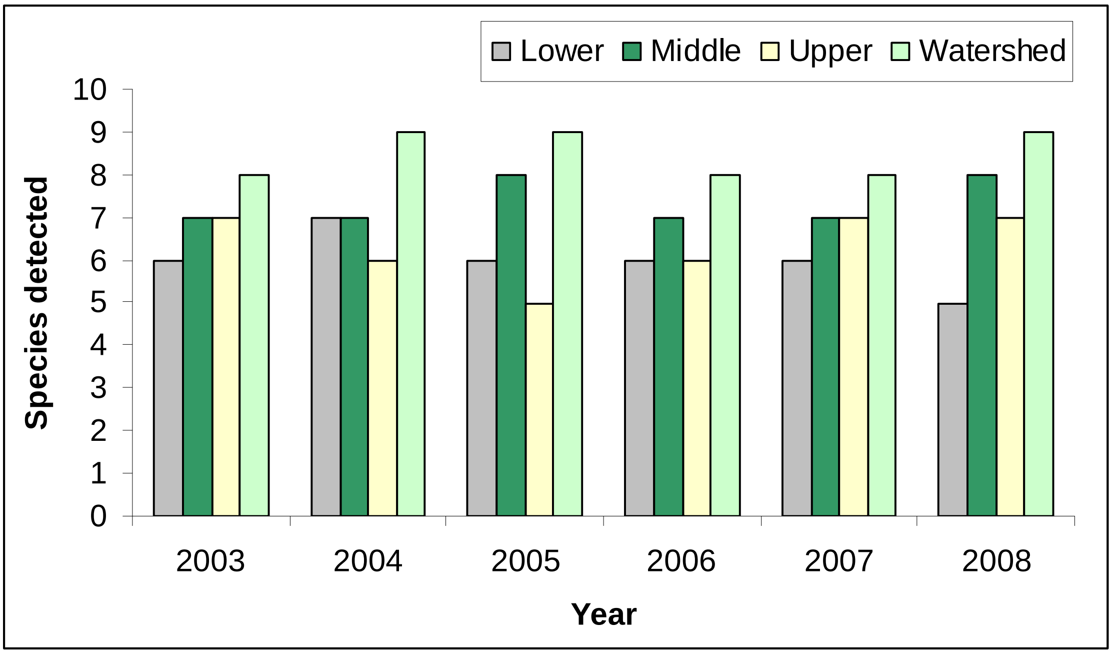

Figure 8. The total number of anuran species detected in the Credit River watershed as a whole and separated by physiographic zone for the years 2003 through 2008.

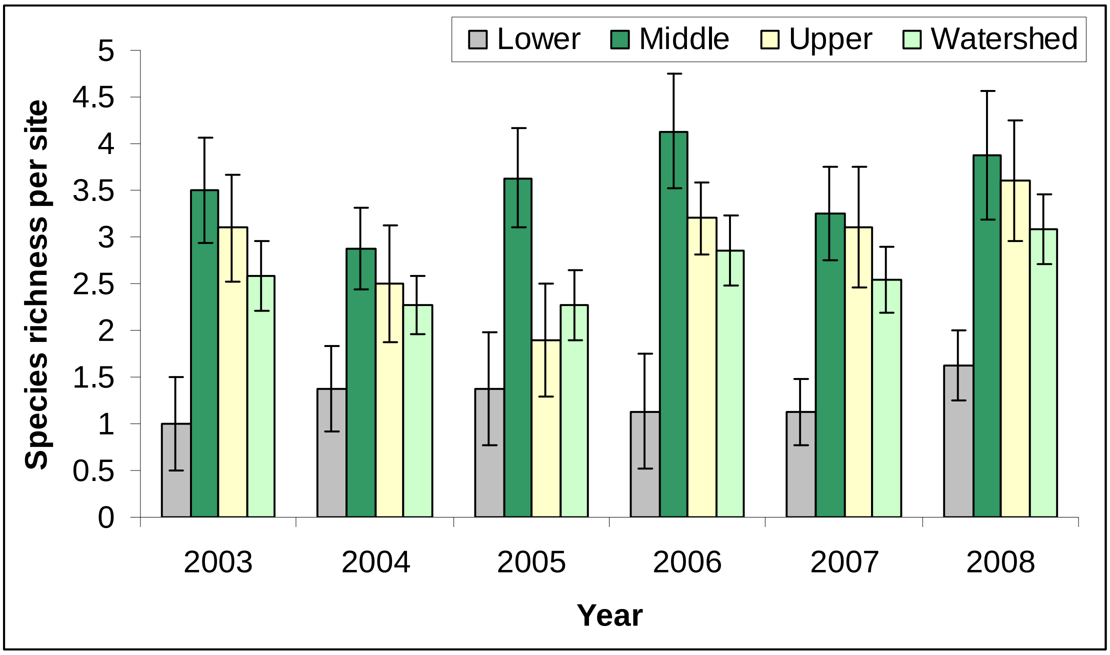

Figure 9. The mean per-site richness of anuran species detected in the Credit River watershed as a whole and separated by physiographic zone for the years 2003 through 2008, +/- standard error.

4.2.2 Anuran Site Occupancy

***Monitoring Question: Is the total number of sites occupied by anurans changing over time in the Credit River watershed*?**

- There was no significant trend in total site occupancy (all species combined) over the monitoring period.
- Proportional site occupancy of Green Frog increased over time in the Lower physiographic zone while proportional occupancy of Leopard Frog increased over time in the Upper zone.
- Green Frog occupied the greatest number of monitoring sites over the course of the study.
- Spring Peeper occupied the greatest proportion of monitoring sites in the Middle and Upper physiographic zones over the course of the monitoring period.

Proportion of monitoring sites occupied by at least one anuran species, by year, was chosen for this analysis due to discrepancies between the numbers of sites included in each zone. Cochrane-Armitage trend analysis found no significant trend in the proportion of sites occupied at either the watershed level (z = 1.132, p = 0.258, Figure 9) or within physiographic zones (Lower: z = 1.255, p = 0.209; Middle: z = 0.423, p = 0.672; Upper: z = 0.667, p = 0.505, Figure 10). This is in agreement with results of the Ontario Backyard Frog Survey, which found no consistent trends in site occupancy between 1995 and 2001 (De Solla et al. 2006). However, site occupancy was noticeably lower in the Upper physiographic zone in 2005. As with anuran richness in that zone in 2005, this low proportion could have been a result of the absence of both American Toad and Northern Leopard Frog.

There were only two significant trends in the proportional site occupancy of individual species at either the whole of the watershed or within individual physiographic zones. The proportion of sites occupied by Green Frog increased over time in the Lower watershed and the proportion of sites occupied by Leopard Frog increased over time in the Upper watershed (z = 2.051, p = 0.040 and z = 2.151, p = 0.031, respectively, Figure 11). The proportional site occupancy of all species both watershed-wide and within zones is presented as figures in Appendix 3. It is unclear whether the two detected trends are harbingers of long-term population changes or simply products of the relatively short study duration. The significant results for Leopard Frog, for instance, were probably a result of that species' absence from the Upper zone in 2005. Additional years of data may be required to determine if the patterns are unidirectional, cyclic, or merely part of random fluctuations.

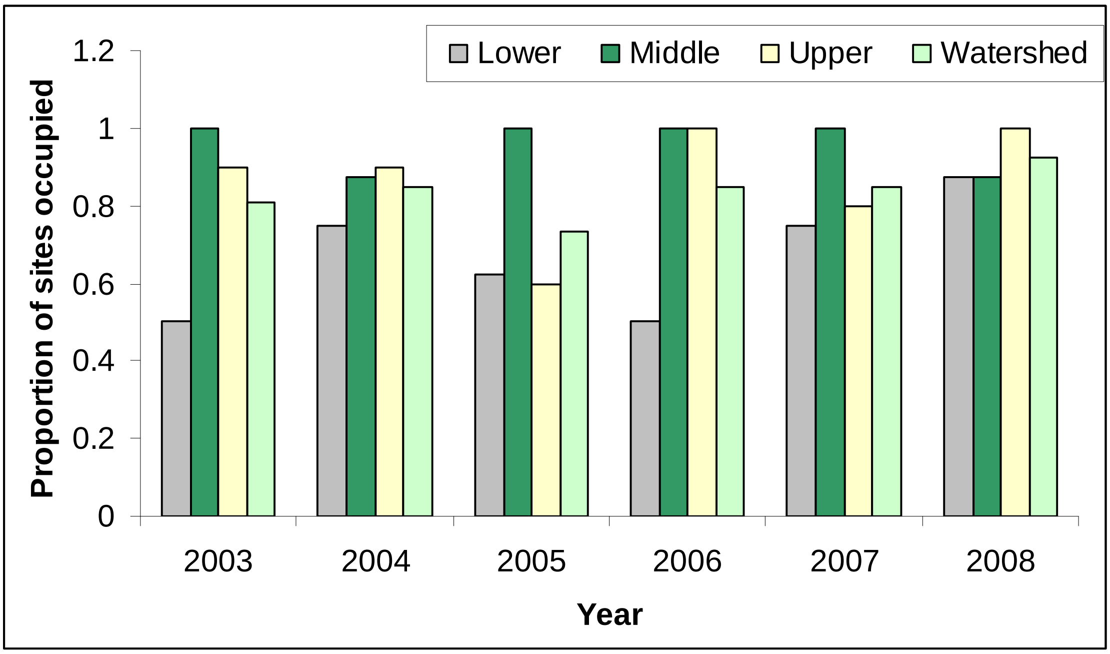

Figure 10. The proportion of monitoring sites occupied by at least one anuran species by year, presented for the whole of the Credit River watershed and by individual physiographic zones. No significant trends in total occupancy were identified.

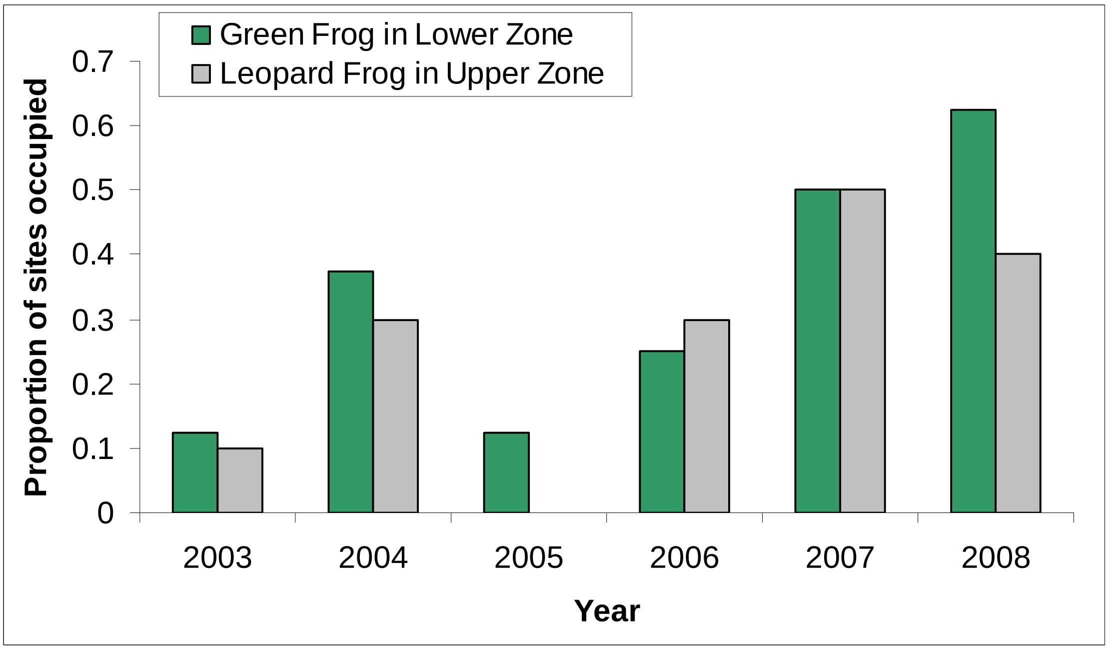

Figure 11. The proportion of monitoring sites occupied by Green Frog in the Lower physiographic zone and the proportion of sites occupied by Leopard Frog in the Upper zone from 2003 to 2008. Both data sets show a significantly increasing trend over time.

4.2.3 Anuran Abundance

Monitoring Question: Is anuran abundance changing over time in the Credit River watershed?

- There was a significant increase in the proportion of sites at which full chorus was detected in both the whole of the watershed and in the Middle physiographic zone.

Cochrane-Armitage analysis revealed a significant increase in the proportion of sites with full chorus at both the watershed level ($Z = 2.48$, $p = 0.013$) and within the Middle physiographic zone ($Z = 2.16$, $p = 0.031$). Full chorus was never detected in the Lower physiographic zone and the increasing trend apparent in the Upper zone was not significant ($Z = 1.65$, $p = 0.099$) (Figure 12). It is difficult to explain these significant trends, partially because similar patterns were not visible in richness and occupancy data. For example, we would perhaps expect to see an increase in Spring Peeper site occupancy associated with an increase in the proportion of sites with full chorus because Spring Peeper was the species most likely to be detected calling at full chorus (Figure 11). It is still possible that the proportional increase in full chorus is due to rapid growth of localized populations. It is possible, however, that the increase is instead due to physical factors such as humidity and springtime temperatures or some unknown sampling or non-sampling error.

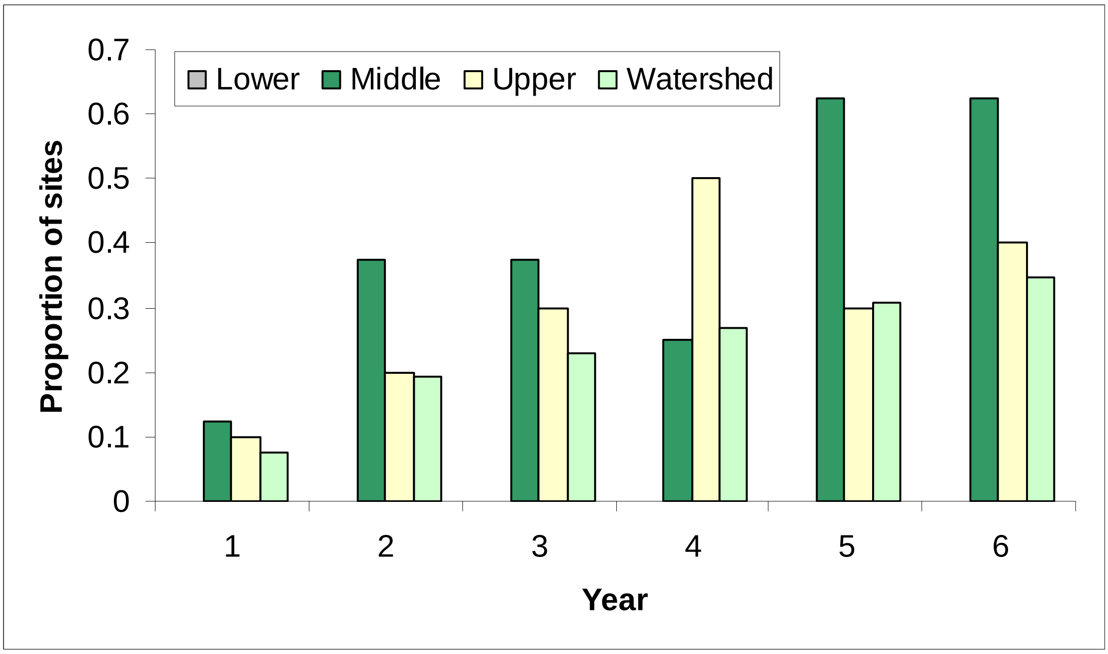

Figure 12. The proportion of monitoring sites at which full chorus (C3) was detected, separated by year and physiographic zone. Significant increases were found across the whole of the watershed and in the Middle zone.

4.2.4 Anuran Community Similarity

Monitoring Question: Is anuran community similarity changing between consecutive years in the Credit River watershed?

- Trend analysis could not be applied to community similarity calculations at this time
- Similarity was lowest in the Lower watershed when considering change between the first and last survey years.

Trend analysis could not be performed on Jaccard's Similarity Index Scores at either the watershed level or within zones due to a deficit of data points (only five over the survey period). There was no obvious trend in community similarity over the six year study period (Figure 13), with the low index score for the Upper physiographic zone in 2006 likely being a product of the absence of American Toad and Northern Leopard Frog in 2005. Values in the first five bar-groupings are based on the anuran species turnover between consecutive years. The index values can be viewed as a measure of species turnover between survey years. Higher index values indicate a greater species similarity between two groups, with a value of 1.0 indicating complete similarity. When comparing similarity index calculations between each combination of two zones for each year of the study, there were no observable trends in any of the zone relationships over the six survey years (Figure 14).

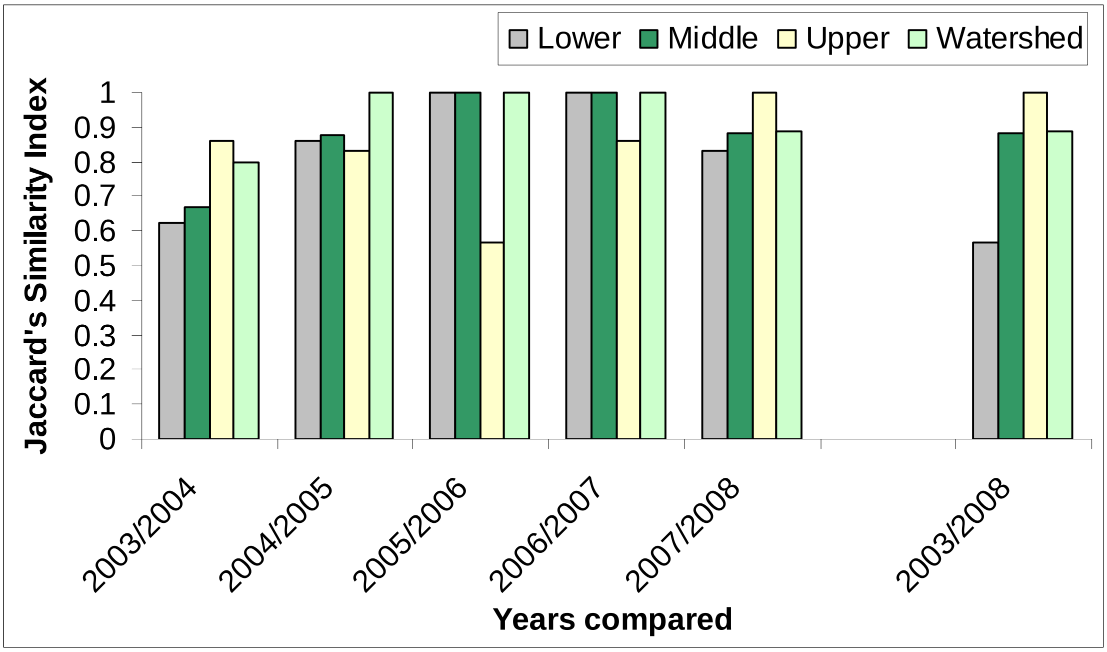

Figure 13. Jaccard's Similarity Index values between consecutive years for the whole of the watershed and within each individual physiographic zone. The set of bars on the right side of the figure compare first and last survey years.

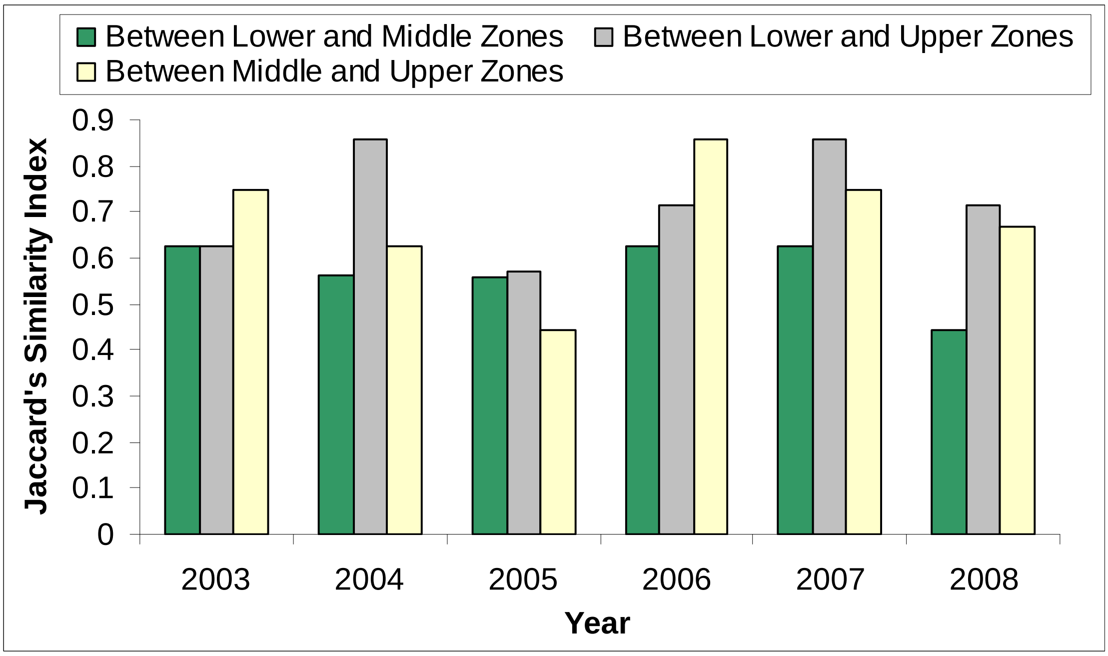

Figure 14. Jaccard's Similarity Index values between each combination of two physiographic zones for each survey year.

Overall species composition did not change in the Upper zone when the first and last survey years were compared (Figure 13, similarity index of 1.0), while compositional change was more pronounced in the Lower watershed over the same period (similarity index of 0.57). American Bullfrog and Gray Treefrog were present in the Lower zone in 2003 but not in 2008, whereas Wood Frog was present in 2008 but not in 2003. It is unclear what factors, if any, were responsible for this change in species over the study period. In general, large changes in similarity index values must be evaluated closely before any conclusions can be made. This is because there are so few choices in the available "pool" of potential anuran species. Outside the introduction to the watershed of a previously undetected invader or rare native such as Fowler's Toad, some combination of the same nine or ten species will be detected each survey year.

4.3 SPATIAL ANALYSIS

4.3.1 Anuran Species Richness

Monitoring Question: Are there spatial differences in anuran species richness in the Credit River watershed?

- Mean per-site species richness was significantly less in the Lower physiographic zone when compared to the Middle physiographic zone for all years except 2004.

The Chi-Square test for independence did not find a significant association between Year and Zone ($X^2 = 0.503$, $p > 0.999$). Non-parametric Kruskal-Wallis tests showed significant per-site species richness differences between physiographic zones in all study years except 2004 (Table 5). Post-hoc testing revealed that these results were primarily due to significant differences between values in the Lower and Middle physiographic zones (Figure 9), though the near significant differences ($p < 0.1$) that existed between the Lower and Upper zones in 2003, 2006, and 2007 could be considered biologically significant. These patterns in richness are likely being driven by differences in land use, as the greatest contrasts in both land composition and the associated measures of landscape disturbance within the watershed exist between the predominantly urban areas of the Lower zone and the protected escarpment and moraine areas of the Middle zone. It should be noted that there were no significant relationships detected post-hoc between the Middle and Upper zones. Urbanizing areas do exist in the Upper watershed and in pockets of the Middle watershed, though they are not present to the same extent or intensity as in the Lower zone.

Table 5. Test statistics, p values, and the associated significant ($p<0.05$) or near-significant post-hoc results for Kruskal-Wallis tests of per-site species richness performed between physiographic zones for each survey year.

Year	Test Statistic (H)	p value	Zone differences determining results (post-hoc) [a]
2003	8.367	0.015	**Lower-Middle (p = 0.028)** **Lower-Upper (p = 0.052)**
2004	4.086	0.130	None
2005	6.088	0.048	**Lower-Middle (p = 0.069)**
2006	9.426	0.009	**Lower-Middle (p = 0.010)** Lower-Upper (p = 0.099)
2007	7.260	0.027	**Lower-Middle (p = 0.053)** Lower-Upper (p = 0.072)
2008	6.622	0.037	**Lower-Middle (p = 0.056)**

[a] Significant and near-significant post-hoc results are presented in bold.

4.3.2 Anuran Site Occupancy

Monitoring Question: Are there spatial differences in anuran site occupancy in the Credit River watershed?

- There are insufficient data to make any conclusions regarding spatial relationships in overall site occupancy.
- Spring Peeper was consistently absent from sites in the Lower physiographic zone.

The binary nature of anuran occupancy data precludes statistical spatial analysis. However, patterns in total sites occupied (Figure 10) seem to suggest that fewer sites were occupied by anurans in the Lower physiographic zone when compared to the Middle and Upper physiographic zones. In 2003 and 2006, for instance, anurans were detected in only 50% of sites in the Lower physiographic zone while all sites in the Middle zone were occupied. This may have been due to lower than normal occupancy rates of the common species Green Frog in the lower zone during those two years

(Appendix 3), or the absence of Wood Frog from that zone in 2003. The pattern seemed to have diminished by 2008, a year in which proportional site occupancy was similar between the three physiographic zones and Green Frog occupancy was at its highest level in the Lower zone. The Chi-Square test for independence using "percent of sites occupied" data found no significant association between Year and Zone ($X^2 = 16.339$, $p = 0.090$).

Spring Peeper was consistently absent from the Lower physiographic zone despite being the most common species in the Middle and Upper zones. This spatial discrepancy may be due to the fact that Spring Peeper appears especially sensitive to habitat patch size (Table **7**). The dominance of urbanization in the Lower zone has resulted in decreased patch sizes and connectivity of natural areas and fewer significant wooded wetlands. Alternatively, the lack of detection in the Lower zone may be a product of inequalities in the sampling design. Spring Peeper requires wooded habitats for at least part of its life cycle (MacCulloch 2002), yet a majority of monitoring stations in the Lower watershed are located at open marsh sites. Though American Bullfrog and Western Chorus Frog were also consistently missing from individual physiographic zones (the Lower/Upper and Middle, respectively), their low occurrence numbers make it difficult to speculate on the root causes of observed spatial distributions.

4.3.3 Anuran Abundance

Monitoring Question: Are there spatial differences in anuran abundance in the Credit River watershed?

- Data are insufficient to make any spatial comparisons of full chorus detection between the three physiographic zones.
- Full chorus has not been detected in the Lower watershed

The Kruskal-Wallis test could not be performed on per-site C3 calling data between physiographic zones for each year because full chorus was never detected in the Lower zone from 2003 to 2008 (Kruskal-Wallis requires a minimum of three groups for comparisons). Results of Mann-Whitney U tests for each separate survey year found no significant differences in full chorus detection between the Middle and Upper physiographic zones (*p* values were all greater than 0.6). It is difficult to conclude whether the absence of full chorus detection in the Lower watershed is due to exogenous factors affecting all species or simply the lack of Spring Peeper at sites in the Lower physiographic zone.

4.4 RELATIONSHIPS WITH LANDSCAPE PARAMETERS

4.4.1 Anuran Species Richness

Monitoring Question: Is anuran species richness correlated with landscape parameters in the Credit River watershed?

- Mean per-site anuran species richness was positively correlated with habitat patch size and negatively correlated with the percent urban land cover within 2 kilometres of the monitoring site.

The Spearman Rank tests using mean species richness data revealed a moderately strong positive correlation between mean per-site anuran species richness and habitat patch size ($\rho = 0.523$, $p = 0.006$) and a strong negative correlation between per-site richness and percent urban land use within 2 kilometres ($\rho = -0.745$, $p < 0.001$). No significant correlation was detected between mean species richness and the distance from monitoring sites to the nearest road ($\rho = 0.029$, $p = 0.885$. The significant results here are consistent with Parris (2006), who found an increase in species richness with pond size within urban areas of Melbourne, Australia. In general, a larger size will be more habitat-diverse and, hence, increase the likelihood that a patch will encompass the specific habitat requirements of a greater number of species. Inversely, a small habitat patch may be homogenous in structure and encompass the specific habitat requirements of only a few species. The negative correlation with percent urban cover is not surprising, as bulk habitat loss and reduction of patch connectivity are associated with increased urban development. There may be no significant correlation with distance to the nearest road because the nearest road may not bisect an anuran corridor. Further study of such landscape configuration issues is required.

4.4.2 Anuran Site Occupancy

Monitoring Question: Is there a relationship between anuran site occupancy and landscape parameters in the Credit River watershed?

- There was a significant positive relationship between habitat patch size and site occupancy of both Gray Tree Frog and Spring Peeper.
- There were no significant relationships between site occupancy and the distance to the nearest road for any anuran species.

There was a significant positive relationship between habitat patch size and site occupancy of both Gray Tree Frog ($X^2 = 11.756$, $p = 0.006$) and Spring Peeper ($X^2 = 14.433$, $p < 0.001$), and a nearly significant positive relationship between patch size and Wood Frog ($X^2 = 3.694$, $p = 0.055$) (Table 6). These three species favour wooded habitats during non-breeding periods and require temporary or permanent ponds in woodlands for breeding purposes (MacCulloch 2002). An increase in patch size would likely result in an associated increase in suitable wooded and permanent water habitat, a relationship supported by the fact that Gray Tree Frog and Spring Peeper were detected at every station located within a habitat patch greater than 170 hectares (8 stations) and 115 hectares (12 stations) respectively. A nearly significant positive relationship with patch size was also found for Bullfrog ($X^2 = 3.58$, $p = 0.058$), but there were too few observations of this species to have any amount of confidence in apparent trends.

Though roads have been shown to have a significant influence on anuran population dynamics (Mazerolle 2004; Fahrig et al 2005; Eigenbrod et al. 2009), there were no significant relationships found between site occupancy of individual species and the distance of monitoring sites from the nearest road (Table 7). This may be partially due to the fact that "distance to the nearest road" is a weak surrogate for road disturbance in the surrounding landscape, missing the complexity of landscape configuration and failing to address the often restricted nature of frog dispersal routes. It ignores the influence of multiple roads surrounding a monitoring site, and it fails to consider differences in road size, type, and frequency of usage. As might be expected, the road parameter is not significantly correlated with habitat patch size (P = 0.106, $p = 0.607$). In addition, roads represent a mortality risk (Fahrig et al. 1995; Hels and Buchwald 2001; Mazerolle 2004) so they may have a greater influence on anuran abundance than on occupancy (results section 3). The density of roads surrounding monitoring sites - a measure appearing more frequently in the literature - would likely be a more useful surrogate for landscape disturbance.

Table 6. Summarised results for logistic regressions between habitat patch size and site occupancy of eight anuran species.

Anuran Species	Test Statistic (X^2)	*p* value	Significant Relationship
American Toad	0.223	0.637	None
Bullfrog	3.58	0.058	None
Gray Tree Frog	11.756	**0.006**	**Positive**
Green Frog	1.007	0.316	None
Leopard Frog	0.035	0.852	None
Spring Peeper	14.433	**< 0.001**	**Positive**
Western Chorus Frog	0.243	0.622	None
Wood Frog	3.694	0.055	None

Table 7. Summarised results for logistic regressions between distance to the nearest road and site occupancy of eight anuran species.

Anuran Species	Test Statistic (X^2)	*p* value	Significant Relationship
American Toad	3.409	0.065	None
Bullfrog	0.041	0.839	None
Gray Tree Frog	2.473	0.116	None
Green Frog	1.692	0.193	None
Leopard Frog	0.272	0.602	None
Spring Peeper	0.381	0.537	None
Western Chorus Frog	1.168	0.280	None
Wood Frog	1.403	0.236	None

There were significant negative relationships between percent urban cover within 2 kilometres and site occupancy of Gray Tree Frog, Leopard Frog, Spring Peeper, and Wood Frog (Table 8). In general, it has been shown that landscape characteristics are important in determining the spatial pattern of frog occurrence at ponds (Mazerolle et al. 2005). This is particularly true in urbanizing areas, where development has been shown to negatively influence the amount of suitable anuran habitat in the landscape (Baldwin and DeMaynadier 2009). As is the case with decreasing patch size or the presence of roads, widespread urbanization can lead to isolated metapopulations. This is turn can increase the risk of local extinctions because highly mobile juveniles are unable to disperse between local population centres (Cushman 2006, Becker et al. 2007).

Table 8. Summarised results for logistic regressions between the percent urban cover within 2 kilometres of the monitoring site and site occupancy of eight anuran species.

Anuran Species	Test Statistic (X^2)	*p* value	Significant Relationship
American Toad	0.566	0.452	None
Bullfrog	3.294	0.069	None
Gray Tree Frog	15.759	**<0.001**	**Negative**
Green Frog	0.350	0.554	None
Leopard Frog	9.297	**0.002**	**Negative**
Spring Peeper	14.975	**<0.001**	**Negative**
Western Chorus Frog	0.150	0.700	None
Wood Frog	6.640	**0.010**	**Negative**

4.4.3 Anuran Abundance

Monitoring Question: Is there a relationship between anuran abundance and landscape parameters in the Credit River Watershed?

- There were significant positive relationship between full chorus detection and both habitat patch size and the distance to the nearest road.
- There was a significant negative relationship between full chorus detection and the percent urban cover within 2 kilometres.

Logistic comparisons between the detection of full chorus and the three chosen landscape parameters produced several significant results. There was a significant positive relationship between detection of full chorus and both habitat patch size (X^2 = 12.790, p < 0.001) and the distance to the nearest road (X^2 = 3.856, p = 0.049), while a significant negative relationship existed between full chorus and the percent urban cover within 2 kilometres (X^2 = 15.970, p < 0.001). The strong relationships with both habitat patch size and the percent urban cover in the landscape could be the result of an associated increase in the amount of suitable wetland and forest habitat supporting larger anuran populations. The relationship with patch size could also be driven by the fact that site occupancy of Spring Peeper, the species most commonly detected at full chorus, had a strong positive relationship with habitat patch size (Table 6). The positive relationship between full chorus detection and distance to the nearest road could be a result of anuran population sensitivities in that the habitat fragmentation and mortality risk suggested by the roads parameter could be leading to smaller populations and, hence, decreased probability of detecting full chorus. Noise from adjacent roadways could also be making it more difficult to detect anuran calls. These detection problems were noted at the Hutttonville and Ken Whillans sites, while low flying jets occasionally delayed the start of surveys at the Meadowvale site. Bee and Swanson (2007), for instance, found that realistic levels of traffic noise limited the range of anuran acoustic signals in Minnesota.

4.5 RELATIONSHIPS WITH VEGETATION PARAMETERS

4.5.1 Anuran Species Richness

Monitoring Question: Is anuran species richness correlated with wetland vegetation parameters in the Credit River watershed?

- The only significant finding was a negative correlation between anuran species richness and non-native species richness in the wetland ground vegetation layer.

Each selected vegetation parameter was converted to mean per-site values across all years so they could be compared to mean per-site anuran richness values. The only significant relationship was a negative correlation between anuran species richness and non-native species richness in the wetland ground vegetation layer (ρ = -0.542, p = 0.020, Table 9). Though there is the temptation to look for a direct relationship between non-native vegetation on anuran richness, this and any other significant correlations must be considered cautiously and no causal relationships should be inferred. The non-native vegetation richness parameter may not be directly influencing anuran richness but simply be acting as an indirect surrogate of landscape disturbance and urbanization, elements that are themselves expressed more directly by landscape measures. Non-native vegetation is likely to be more prevalent at stations located in habitat patches with greater edge length and closer proximity to anthropogenic dispersal routes.

Table 9. Spearman Rank correlations between anuran richness and wetland vegetation parameters, ranked by the absolute size of the correlation.

Vegetation Parameters	Correlation Coefficient (ρ)	*p* Value
Mean non-native richness (ground vegetation)	**-0.542**	**0.020**
Mean proportion non-native species (ground vegetation)	-0.429	0.076
Mean native richness (regeneration)	0.426	0.078
Mean coefficient of conservatism (combined vegetation)	0.412	0.090
Mean total species richness (regeneration)	0.373	0.127
Mean non-native richness (regeneration)	-0.284	0.253
Mean proportion non-native species (regeneration)	-0.257	0.302
Mean total species richness (ground vegetation)	-0.115	0.648
Mean native richness (ground vegetation)	0.065	0.797

4.5.2 Anuran Site Occupancy

Monitoring Question: Is there a relationship between anuran site occupancy and vegetation parameters in the Credit River watershed?

- There were several significant negative relationships between occupancy of individual species and measures of non-native vegetation.
-There were significant positive relationships between occupancy of individual species and both total and native regeneration richness.

Several significant logistic relationships were found between site occupancy of individual anuran species and the chosen wetland vegetation parameters (Table 10). As with the rank correlations between anuran richness and the same vegetation parameters, the significant negative results here were associated with non-native plant richness covariates. Again, these measures of non-native vegetation may be acting as surrogates for habitat loss and landscape fragmentation. Many of the significant positive relationships between vegetation and occupancy involved tree frogs (Gray Treefrog, Spring Peeper) and richness measures of regenerating woody plant species. The increased structural diversity at the monitoring stations that is implicit in an increase in regeneration diversity may be encouraging tree frog species to remain in the immediate vicinity of the surveyed wetlands even after breeding has been completed. Open marsh sites with little woody plant diversity would not provide the stratified habitat required by these species during non-breeding periods. The positive relationship between Spring Peeper occupancy and mean coefficient of conservatism is tougher to explain, as a) the literature provides no evidence that the quality of native plant species has any effect on anuran populations and b) mCC could be expressing any number of environmental variables. Vegetation with high habitat fidelity may be more prevalent within the interior communities associated with increasing habitat patch size, which is itself a landscape parameter significantly related to Spring Peeper occupancy.

Table 10. The significant and near significant results of logistic regression tests between site occupancy of individual anuran species and the chosen wetland vegetation parameters.

Species	Vegetation Parameter	Test Statistic (X^2)	*p* Value	Significant Trend
Gray Tree Frog	Ground vegetation non-native richness	3.649	0.056	Negative
	Ground vegetation proportion non-natives	3.747	0.053	Negative
	Regeneration total species richness	3.928	0.048	Positive
	Regeneration native richness	4.299	0.038	Positive
Leopard Frog	Ground vegetation non-native richness	4.225	0.040	Negative
Spring Peeper	Ground vegetation non-native richness	8.434	0.004	Negative
	Ground vegetation proportion natives	11.576	0.001	Positive
	Regeneration non-native richness	3.880	0.049	Negative
	Regeneration proportion non-natives	4.764	0.029	Negative
	Combined vegetation Mean Coefficient of Conservatism	8.815	0.003	Positive
Wood Frog	Regeneration total species richness	6.674	0.010	Positive
	Regeneration native richness	6.173	0.013	Positive

4.5.3 Anuran Abundance

Monitoring Question: Is there a relationship between anuran abundance and wetland vegetation parameters in the Credit River watershed?

- There were significant negative relationships between full chorus detection and measures of non-native vegetation.
- There were significant positive relationships between full chorus detection and both total and native regeneration richness.

There were several significant logistic relationships found between full chorus detection and selected wetland vegetation parameters (Table 11). As with the relationships between vegetation and richness or occupancy, the significant negative results are mostly associated with measures of non-native vegetation. This could again be related to the kind of landscape alteration that promotes the growth and transport of invasive plant species. The negative trends in abundance could also be due to properties of the vegetation itself. For example, it has been shown that larval amphibian development can be slowed by secondary compounds leached from invasive plants such as Purple Loosestrife (*Lythrum salicaria*) (Maerz et al. 2005). The significant positive results were associated with regeneration richness (both native and total) and mean Coefficient of Conservatism. Please see the occupancy results section (page 33) for a discussion of the potential connections between anuran parameters and both regeneration richness and mean coefficient of conservatism values.

Table 11. Significant results of logistic regressions between full chorus detection and selected wetland vegetation parameters, listed by confidence level.

Vegetation Parameter	Test Statistic (X^2)	*p* value	Significant Relationship
Mean coefficient of conservatism	7.313	0.007	Positive
Native richness (regeneration)	6.550	0.010	Positive
Proportion non-natives (ground vegetation)	6.481	0.011	Negative
Total species richness (regeneration)	5.671	0.017	Positive
Non-native richness (ground vegetation)	5.426	0.020	Negative
Proportion non-natives (regeneration)	4.743	0.029	Negative
Non-native richness (regeneration)	4.490	0.034	Negative

4.6 STATISTICAL PROCESS CONTROL

The upper and lower critical and warning limits calculated for each anuran trend parameter with at least five years of available data (Table 12) will be adopted as monitoring thresholds to which future anuran monitoring data can be compared. Future values that exceed the upper or lower warning limits (2 standard deviations) will signify emergent increasing or declining population trends, while values that exceed the upper and lower critical limits (3 standard deviations) will signify a significant deviation from natural variability. Such deviations would warrant immediate management action before trends became irreversible. No monitoring thresholds can be applied to those parameters deemed to be out of control or those displaying no variance over the monitoring period.

In general, the spread in values between the upper and lower critical limits of individual parameters seem larger in the Upper physiographic zone when compared to similar parameters in the Lower zone. For instance, the spread of “mean species per site” values between upper and lower limits in the Lower zone was 1.2 species units while the same spread in the Upper zone was 3.2 units. This pattern is likely due to increased variability in the yearly values of parameters associated with the Upper watershed. Among measures of site occupancy by individual species, American Toad had the largest spread in values between the upper and lower critical limits while Green Frog and Western Chorus Frog had the smallest spreads. It should be noted that critical and warning thresholds presented in the table do not exceed the constraints of realistic values, such as the maximum proportion of monitoring sites occupied (100%) or the maximum Jaccard’s score (1.0), even if calculated values exceeded these constraints. Values that exceed these constraints cannot be realistically detected and, therefore, cannot be adopted as thresholds.

Table 12. Monitoring thresholds for all anuran trend parameters extracted from Statistical Process Control individual (moving average) IMR charts for data collected from 2003 through 2008.

Parameter	Thresholds [a]			
	Upper Critical Threshold	**Upper Warning Threshold**	**Lower Warning Threshold**	**Lower Critical Threshold**
Species Richness				
Total species	9.6	9.21	7.79	7.4
Total species (Lower zone)	7.2	6.9	5.5	5.1
Total species (Middle zone)	Not "in control"			
Total species (Upper zone)	8.5	7.8	4.9	4.2
Species per site	3.5	3.2	2.0	1.7
Species per site (Lower zone)	1.9	1.7	0.9	0.7
Species per site (Middle zone)	5.3	4.7	2.4	1.7
Species per site (Upper zone)	4.5	4.0	1.8	1.3
Proportional Site Occupancy (%)				
Sites Occupied	>100	95.6	71.1	64.9
Sites Occupied (Lower)	>100	97.7	35.6	20.1
Sites Occupied (Middle)	>100	>100	82.5	75.9
Sites Occupied (Upper)	>100	>100	47.7	28.2
American Bullfrog	Not "in control"			
American Bullfrog (Lower)	Not "in control"			
American Bullfrog (Middle)	32.1	27.6	9.9	5.5
American Bullfrog (Upper)	No Records			
American Toad	>100	87.9	<0	<0
American Toad (Lower)	75.7	60.2	<0	<0
American Toad (Middle)	>100	>100	<0	<0
American Toad (Upper)	>100	>100	<0	<0
Gray Tree Frog	70.7	59.1	12.7	1.1
Gray Tree Frog (Lower)	Not "in control"			
Gray Tree Frog (Middle)	>100	86.0	<0	<0
Gray Tree Frog (Upper)	>100	94.2	9.1	<0
Green Frog	60.9	57.5	43.8	40.4
Green Frog (Lower)	86.5	68.8	<0	<0
Green Frog (Middle)	88.7	82.1	55.5	48.8
Green Frog (Upper)	92.6	78.4	21.6	7.4
Leopard Frog	59.9	51.1	15.6	6.7
Leopard Frog (Lower)	Not "in control"			
Leopard Frog (Middle)	>100	94.1	14.3	<0
Leopard Frog (Upper)	85.2	65.7	<0	<0
Spring Peeper	73.4	69.3	46.4	40.7
Spring Peeper (Lower)	15.4	10.9	<0	<0
Spring Peeper (Middle)	>100	96.6	70.0	63.4
Spring Peeper (Upper)	>100	>100	55.2	42.8
Chorus Frog	20.5	17.1	3.4	0
Chorus Frog (Lower)	No Variance			
Chorus Frog (Middle)	No Records			
Chorus Frog (Upper)	43.3	34.4	<0	<0
Wood Frog	54.6	47.1	17.0	9.5
Wood Frog (Lower)	34.5	27.9	1.3	<0
Wood Frog (Middle)	83.3	72.2	27.8	16.8
Wood Frog (Upper)	68.9	56.5	6.8	<0

Parameter	Thresholds [a]			
	Upper Critical Threshold	Upper Warning Threshold	Lower Warning Threshold	Lower Critical Threshold
Abundance				
Total C3 call records	13.7	11.7	3.9	2.0
Total C3 call records (Lower)	No Records			
Total C3 call records (Middle)	6.7	5.6	1.4	0.4
Total C3 call records (Upper)	9.1	7.7	1.9	0.5
Community Similarity				
Jaccard's index	1.00	1.00	0.78	0.71
Jaccard's index (Lower zone)	1.00	1.00	0.62	0.50
Jaccard's index (Middle zone)	1.00	1.00	0.66	0.56
Jaccard's index (Upper zone)	1.00	1.00	0.50	0.35

[a] Limits are presented in the specific units used to measure each given parameter.

4.7 POWER ANALYSIS

The Calculated numbers of monitoring sites required to detect linear trends that exceed each threshold identified through Statistical Process Control (1 SD, 2 SD or 3 SD) were found to be identical between all anuran trend parameters (Table 13). This odd pattern is a product of the Power Analysis procedure specific to linear regression, which itself uses standard deviation as a key variable in calculations. Though results show that a one standard deviation change in values cannot be detected in any parameter by the current program (approximately 32 sites are required), values that remain within this "green" zone are not considered deviant from a management perspective. Of greater importance is the fact that there are currently enough monitoring plots in the program to detect warning thresholds (7.2 plots needed) and critical thresholds (less than 3 plots needed) across the whole of the watershed and within the Lower, Middle, and Upper physiographic zones.

The smallest minimum detectable effect size based on the current complement of monitoring sites (26 in the watershed, 8 in the Lower and Middle zones, 10 in the Upper zone) was 6.6% for the total species trend in the whole of the watershed (Table 13). This means that the monitoring program has sufficient power to detect a 6.6% change from the baseline mean (about 1 species) in the mean number of anuran species in the watershed over a five year period. High variability in some parameters, along with the small number of sites in each physiographic zone, resulted in proportionally higher minimum detectable effect sizes. For example, it is only possible to detect a minimum change of 102.7% from the baseline mean (or about 5 detections) in the number of full chorus records in the Upper physiographic zone.

The current program of 26 monitoring sites only has the power to detect the two standard deviation warning threshold for Toad occupancy across the whole of the watershed, while the three standard deviation critical threshold can only be detected for Toad occupancy across the whole of the watershed and within the Upper physiographic zone (Table 14). The minimum detectable effect sizes were greater than three standard deviations for every other proportion parameter. For instance, a minimum change of 7.9 standard deviations can be detected in Spring Peeper occupancy at the watershed level given the current complement of monitoring sites (Table 14). Two conclusions can be drawn from these results: either a) there are not enough anuran monitoring sites in the program to detect the desired changes in proportional occupancy, or b) the 1, 2, and 3 standard deviation thresholds used for linear regression are not appropriate when dealing with proportional data.

The calculated number of monitoring sites required to determine a significant logistic relationship between the binary anuran variables and the landscape or wetland vegetation covariates differed greatly between the many variable and covariate combinations (Table 15). All test results with landscape covariates have been presented. However, only statistically significant relationships ($p < 0.05$) with vegetation covariates were tested for power due to the sheer number of possible parameter combinations. The exceptions are that no relationships with the percent urban cover within 2 kilometres, mean coefficient of conservatism values, non-native regeneration richness, and proportion non-native regenerating species are presented because the distribution of these parameters could not be normalised. The calculated N values (required number of sites) were within the current complement of sites (26) for each statistically significant logistic relationship, save for full chorus detection compared to the distance to the nearest road ($p = 0.0496$, n =75). For example, the significant relationship between Leopard Frog occupancy and non-native ground vegetation richness had a significance level of 0.004 and could legitimately be detected with a sample size of 19. Required sample sizes were often quite high for non-significant logistic relationships. For instance, over 30 000 monitoring sites would be required to detect a significant relationship between Spring Peeper occupancy and the distance to the nearest road. Further examination of these results is necessary because the number of sites required to detect a relationship may change as more survey years are added to the data set or as updated landscape information becomes available.

Table 13. Results of linear regression power analysis for anuran parameters that will be eventually be examined for temporal trends.

Anuran Parameter [b]	**Sites Required to Detect SPC Thresholds**			**Minimum Detectable Effect Size**	
	1 SD	**2SD (Warning)**	**3SD (Critical)**	**% Change**	**Units Change** [a]
Anuran Species Richness					
Mean species per site (Watershed)	31.8	7.2	2.6	7.0	0.35
Mean species per site (Lower zone)	31.8	7.2	2.6	35.2	0.4
Mean species per site (Middle zone)	31.8	7.2	2.6	24	0.85
Mean species per site (Upper zone)	31.8	7.2	2.6	36.2	1.05
Total species (Watershed)	31.8	7.2	2.6	7.0	1 species
Total species (Lower zone)	31.8	7.2	2.6	20	2 species
Total species (Middle zone)	31.8	7.2	2.6	13.6	1 species
Total species (Upper zone)	31.8	7.2	2.6	22.1	2 species
Anuran Abundance					
Total full chorus detections (Watershed)	31.8	7.2	2.6	54.3	5 detections
Total full chorus detections (Middle zone)	31.8	7.2	2.6	81.4	3 detections
Total full chorus detections (Upper zone)	31.8	7.2	2.6	102.7	5 detections
Anuran Community Similarity					
Jaccard's Similarity Index (Watershed)	31.8	7.2	2.6	10.9	0.1
Jaccard's Similarity Index (Lower zone)	31.8	7.2	2.6	29.0	0.25
Jaccard's Similarity Index (Middle zone)	31.8	7.2	2.6	29.1	0.25
Jaccard's Similarity Index (Upper zone)	31.8	7.2	2.6	30.3	0.25

[a] Rounded up to the nearest whole number for parameters that cannot be m
[b] Listed are the numbers of monitoring sites required to detect SPC thresholds, as well as the minimum detectable effect size given the current study design

Table 14. Power analysis results for trends in proportional site occupancy. [a]

Occupancy Category	Spatial Level	Sites required to detect SPC thresholds			Minimum detectable effect size (SD)
		1 SD	2 SD	3SD	
Total Occupancy	Watershed	154	76	50	5.66
	Lower zone	62	31	20	7.43
	Middle zone	150	73	49	17.4
	Upper zone	63	31	20	6.03
Bullfrog	Watershed	503	247	163	18
	Lower zone	191	94	62	22
	Middle zone	141	69	46	16.4
	Upper zone	-	-	-	-
American Toad	Watershed	46	**23**	**15**	1.74
	Lower zone	74	36	24	8.7
	Middle zone	37	18	12	4.49
	Upper zone	27	13	**9**	2.62
Gray Tree Frog	Watershed	93	46	30	3.48
	Lower zone	191	94	62	22
	Middle zone	50	25	16	5.97
	Upper zone	48	24	16	4.65
Green Frog	Watershed	171	84	56	6.28
	Lower zone	46	23	13	5.5
	Middle zone	141	69	46	16.4
	Upper zone	67	33	22	6.38
Leopard Frog	Watershed	93	46	30	3.48
	Lower zone	94	46	30	11
	Middle zone	50	25	16	5.97
	Upper zone	51	25	16	4.85
Spring Peeper	Watershed	217	106	70	7.9
	Lower zone	191	94	62	22
	Middle zone	150	73	49	17.4
	Upper zone	87	43	28	8.24
Chorus Frog	Watershed	247	121	80	8.96
	Lower zone	-	-	-	-
	Middle zone	-	-	-	-
	Upper zone	92	45	30	8.74
Wood Frog	Watershed	135	66	44	4.97
	Lower zone	102	50	33	11.9
	Middle zone	85	42	28	10.04
	Upper zone	56	27	18	5.34
Full Chorus Calling	Watershed	100	49	32	3.72
	Lower zone	-	-	-	-
	Middle zone	47	23	15	5.6
	Upper zone	67	33	22	6.38

[a] Presented are the calculated number of monitoring sites required to detect the SPC thresholds and the minimum detectable effect size expressed in standard deviations.

Table 15. The calculated number of monitoring sites (N) required to detect a significant logistic relationship between the binary anuran variables and landscape or wetland vegetation covariates.

Binary Dependant Variable	Continuous Covariate	Logistic Regression *p* Value	Sites Required (N)
Bullfrog Occupancy	Patch Size (Sqrt)	0.058	67
	Distance to Road (Sqrt)	0.839	14769
Toad Occupancy	Patch Size (Sqrt)	0.637	2666
	Distance to Road (Sqrt)	0.065	95
Gray Tree Frog Occupancy	Patch Size (Sqrt)	0.006	6
	Distance to Road (Sqrt)	0.116	215
	Ground Vegetation Non-Native Richness	0.056	22
	Ground Vegetation Proportion Non-Natives	0.053	21
	Regeneration Total Species Richness	0.048	21
	Regeneration Native Richness	0.038	20
Green Frog Occupancy	Patch Size (Sqrt)	0.316	206
	Distance to Road (Sqrt)	0.193	248
Leopard Frog Occupancy	Patch Size (Sqrt)	0.852	287
	Distance to Road (Sqrt)	0.602	3388
	Ground Vegetation Non-Native Richness	0.04	19
Spring Peeper Occupancy	Patch Size (Sqrt)	< 0.001	5
	Distance to Road (Sqrt)	0.537	31425
	Ground Vegetation Non-Native Richness	0.004	7
	Ground Vegetation Proportion Non-Natives	0.001	3
Chorus Frog Occupancy	Patch Size (Sqrt)	0.622	3642
	Distance to Road (Sqrt)	0.280	187
Wood Frog Occupancy	Patch Size (Sqrt)	0.055	45
	Distance to Road (Sqrt)	0.236	415
	Regeneration Total Species Richness	0.010	9
	Regeneration Native Richness	0.013	12
Full Chorus Detection	Patch Size (Sqrt)	< 0.001	6
	Distance to Road (Sqrt)	0.0496	75
	Ground Vegetation Proportion Non-Natives	0.011	15
	Ground Vegetation Non-Native Richness	0.020	9
	Regeneration Total Species Richness	0.017	13
	Regeneration Native Richness	0.010	7

5.0 CONCLUSIONS

No critical signs of decline were apparent in anuran populations over the study period. Parameters appeared stable (or at least oscillated randomly) over the six years of monitoring. At times, the detection of temporal and spatial trends was hindered both by a deficit of sampling points and by an inability to apply more robust statistical techniques to the collected data, namely linear trend analysis and Repeated Measures ANOVA. These problems may be alleviated as additional years of survey data are incorporated into analyses or as patterns in anuran richness, occupancy, and abundance change through time. If, however, the analytical problems persist, the study framework or monitoring protocol may have to be altered to better address monitoring questions. For example, the usefulness of the ordinal call classification system in accurately estimating anuran abundance numbers may have to be reviewed. Unfortunately, there appears to be few alternatives available that fit within the resource constraints of the current monitoring program.

Four significant temporal trends were found to exist between 2003 and 2008. Proportional site occupancy of Green Frog increased in the Lower physiographic zone while proportional occupancy of Leopard Frog increased in the Upper zone. An unexplainable significant increase in the proportion of sites with full chorus calling was also observed, both watershed-wide and within the Middle physiographic zone. Species richness and total site occupancy did experience yearly drops and surges in both the whole of the watershed and in individual physiographic zones, but there was no overall movement of values towards improvement or decline. As such, few predictions can be made concerning future conditions of anuran populations in the watershed. Additional years of monitoring may be required to recognise any temporal patterns outside unstable population dynamics (De Solla et al. 2006) or the cycle of extinction and re-colonization often associated with metapopulations (Marsh and Trenham 2000; Smith and Green 2005) that have been reported in the literature.

Spatial trends in anuran parameters were more pronounced over the study period. Though the only statistically significant spatial pattern was detected in species richness between the Lower and Middle zones for five of six survey years, anuran parameter values were consistently diminished in the Lower physiographic zone when compared to those in the Upper and Middle zones. Fewer monitoring sites were occupied by any species in the Lower zone, and no full chorus calls were detected there over the six year study. The absence of full chorus in the Lower zone could be an early indicator of declining populations and is likely capturing the influence of pervasive urban development and associated habitat fragmentation in the lower portion of the watershed, as well as a sampling inequality that is biased towards marsh sites in urban areas. Despite dissimilarities in topography and land use, spatial distinctions between the Middle and Upper zones were rarely substantial.

Green Frog and Spring Peeper were the most commonly detected anuran species in the watershed. Based on a non-statistical observation of trends, these two species were also the only ones to exhibit relatively stable levels of site occupancy over the whole of the watershed (Appendix 3). Year-to-year occupancy numbers for the other seven species were notably more unstable, with American Toad occupancy being the most erratic. The regional distribution of some species appeared to favour or exclude certain landscape divisions. Spring Peeper was mostly absent from the Lower physiographic zone, whereas American Bullfrog was not recorded in either the Lower or Upper zones. Western Chorus Frog was absent from the Middle zone while exhibiting seemingly stable occupancy at a few sites in the Lower and Upper zones. Unfortunately, a relative shortage of detections excludes us from making any conclusions regarding population trends in this species of concern.

Direct or indirect measures of habitat loss and fragmentation (such as habitat patch size, the percent urban land cover within 2 kilometres of monitoring sites, and the richness of non-native vegetation) had the strongest correlation with anuran richness, occupancy, and abundance. These relationships between anuran parameters and potential environmental predictors should be subject to more comprehensive multiple-variable modeling as analysis continues, and be tracked intently as updated landscape and vegetation community data becomes available. In reference to individual monitoring sites, two locations appeared to represent the best and worst of anuran communities in the watershed. Speersville Wetland, a marsh in the Upper physiographic zone, was rated the healthiest anuran community because it had the highest mean species richness, the second highest mean full chorus detection, and was occupied by anurans in each of the six survey years. The Mississauga Gardens site, a marsh located within a major urban centre in the Lower physiographic zone, was rated the least healthy anuran community because it had the lowest mean species richness, no detections of full chorus calling, and was occupied by anurans for only two of the six survey years.

Additional years of data may be required to confirm current trends, so it is unnecessary to pursue a change in the number of monitoring sites at this time. It is recommended, however, that trends and spatial relationships in anuran richness, occupancy, and abundance continue to be tracked closely over subsequent survey years in order to determine if intervention is necessary to maintain stable populations in the watershed. Land-use and composition information, as well as wetland vegetation data, should be kept up-to-date so that analysis can be based on the most accurate data available. In addition, call detection methodology should be reviewed due to current limitations in the estimation of abundance counts. Through this integration and continued focus on program improvement, a more complete and holistic understanding of anuran population health in the Credit River watershed can be gained.

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APPENDIX A: ANURAN MONITORING SITES IN THE CREDIT RIVER WATERSHED 2003 TO 2008.

Site Name	Site Number	Upper Tier Municipality	Lower Tier Municipality	Physiographic Zone	Habitat Patch Size (Ha)
Rattray Marsh	W-01	Peel Region	Mississauga	Lower	92.3
Credit River Marsh	W-02	Peel Region	Mississauga	Lower	0.3
Creditview Wetland	W-03	Peel Region	Mississauga	Lower	5.4
Meadowvale ESA Wetland	W-04	Peel Region	Mississauga	Lower	42.3
Winston Churchill Wetland	W-12	Halton Hills	Halton Hills	Lower	24.7
Turtle Creek	WNV-01	Peel Region	Mississauga	Lower	92.3
Mississauga Gardens	WNV-02	Peel Region	Mississauga	Lower	52.4
Churchville	WNV-03	Peel Region	Brampton	Lower	16.8
Hungry Hollow Wetland	W-06	Halton Hills	Halton Hills	Middle	114.1
Acton Swamp	W-07	Halton Hills	Halton Hills	Middle	127.9
Terra Cotta Wetland	W-08	Peel Region	Town of Caledon	Middle	753.3
Ken Whillans RMA	W-09	Peel Region	Town of Caledon	Middle	165.9
Warwick Wetland	W-10	Peel Region	Town of Caledon	Middle	191.2
Rail Trail	A-01	Peel Region	Town of Caledon	Middle	158.9
Balinafad Ridge	A-02	Wellington	Town of Erin	Middle	223.6
Silver Creek	A-03	Halton Hills	Halton Hills	Middle	370.1
Erin Pine Estates Wetland	W-11	Wellington	Town of Erin	Upper	93.1
Hillsburgh Wetland	W-13	Wellington	Town of Erin	Upper	14.4
Grange Orpen Wetland	W-15	Peel Region	Town of Caledon	Upper	137.6
Starr Wetland	W-16	Peel Region	Town of Caledon	Upper	291.3
Speersville Wetland	W-17	Peel Region	Town of Caledon	Upper	218.9
Caledon Lake Bog	W0-18	Peel Region	Town of Caledon	Upper	554.8
Mellville Marsh	W-19	Amaranth	Orangeville	Upper	5.5
Belfountain Wetland	W-20	Peel Region	Town of Caledon	Upper	31.2
Caledon Creek	WNV-4	Peel Region	Town of Caledon	Upper	78.3
Alton-Hillsburgh	A-04	Wellington	Town of Erin	Upper	321.3

APPENDIX B: ECOLOGICAL LAND CLASSIFICATION (ELC) COMMUNITY SERIES AND LAND USE TYPES INCORPORATED INTO THE DEFINITIONS OF NATURAL, AGRICULTURAL, AND URBAN LAND COVER.

ELC Community Series	ELC Code	Land use Cover (aggregated)
Coniferous forest	FOC	Natural
Coniferous plantation	CUP3	
Coniferous swamp	SWC	
Cultural meadow	CUM	
Cultural savannah	CUS	
Cultural thicket	CUT	
Cultural woodland	CUW	
Deciduous forest	FOD	
Deciduous plantation	CUP1	
Deciduous swamp	SWD	
Marsh	MA	
Mixed forest	FOM	
Mixed plantation	CUP2	
Mixed swamp	SWM	
Open Aquatic	OAO	
Open Beach/Bar	BBO	
Open Bluff	BLO	
Shrub bluff	BLS	
Shrub bog	BOS	
Thicket swamp	SWT	
Treed beach/Bar	BBT	
Treed bog	BOT	

Land Use Type	ELC Code	Land use Cover (aggregated)
Commercial/Industrial Open Space	MOC	Agriculture
Educational/Institutional Open Space	MOI	
Inactive Aggregate	AI	
Intensive Agriculture	AGI	
Manicured Open Space	MOS	
Non-intensive Agriculture	AGN	
Other Open Space	MOO	
Private Open Space	MOP	
Recreational Open Space	MOR	
Wet Meadow	WET	
Active Aggregate	AA	Urban
Airport	TPA	
Collector	TPC	
Commercial/Industrial	CIC	
Construction [a]	CON	
Educational/Institutional	CII	
High Density Residential	URH	
High Rise Residential	URR	
Highway	TPH	
Landfill	LF	
Low Density Residential	URL	
Medium Density Residential	URM	
Railway	TPX	
Regional Road	TPR	
Residential Estate	URE	
Rural Development	RD	
Urban	URB	
Mixed Residential	URX	

[a] New addition Nov 2009

APPENDIX C: ANURAN PROPORTIONAL OCCUPANCY

Table 1. Proportion of monitoring sites occupied by American Bullfrog.

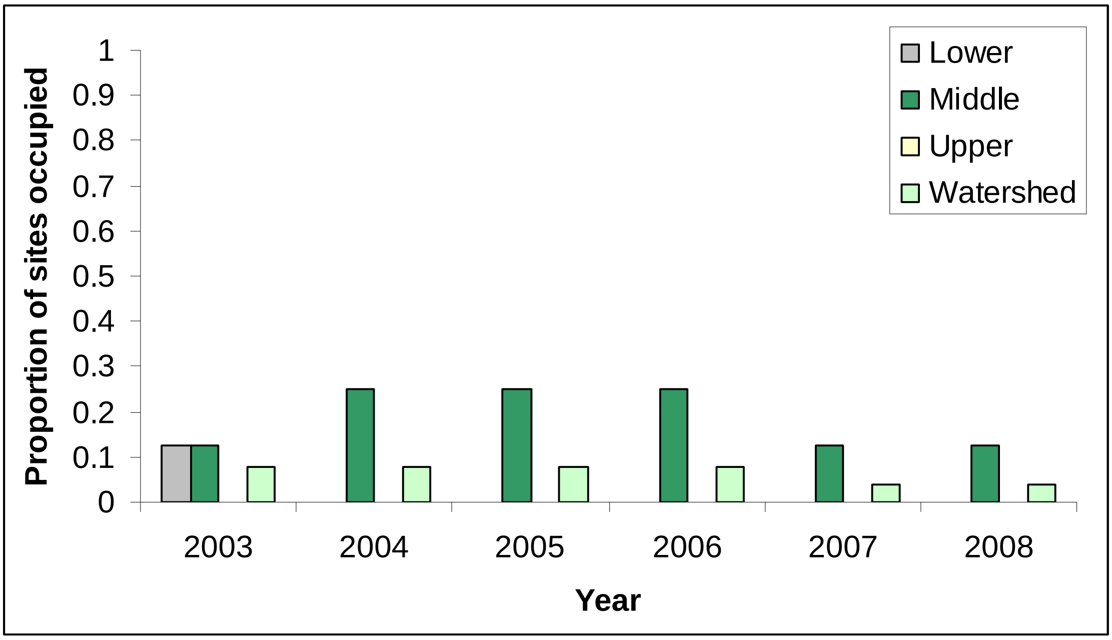
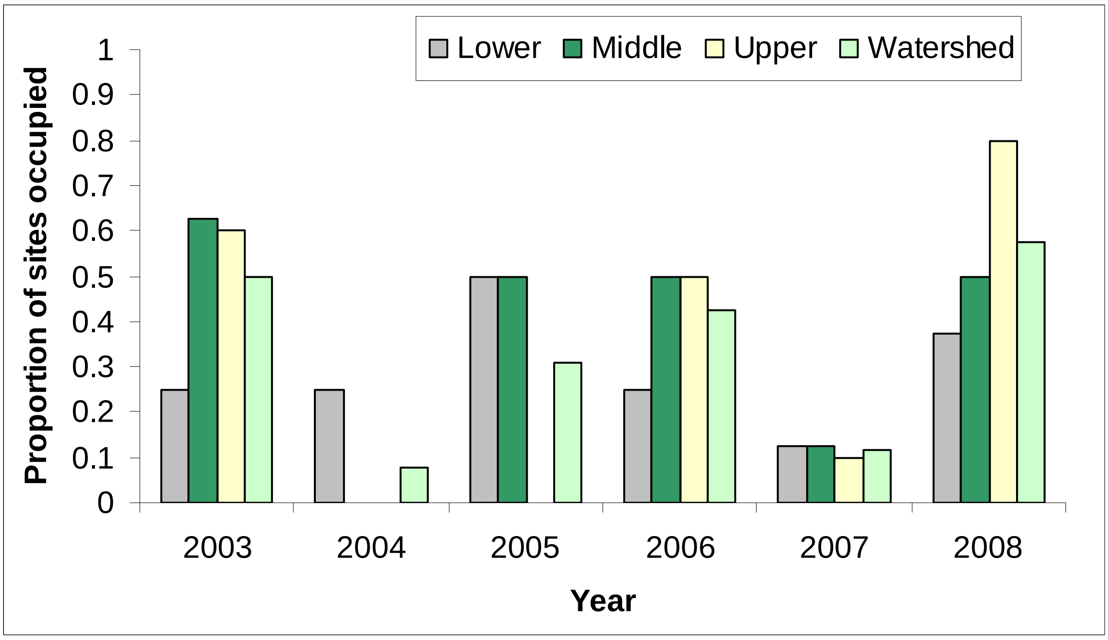


Table 2. Proportion of monitoring sites occupied by American Toad.



APPENDIX C: ANURAN PROPORTIONAL OCCUPANCY (Cont'd)

Table 3. Proportion of monitoring sites occupied by Grey Tree Frog.

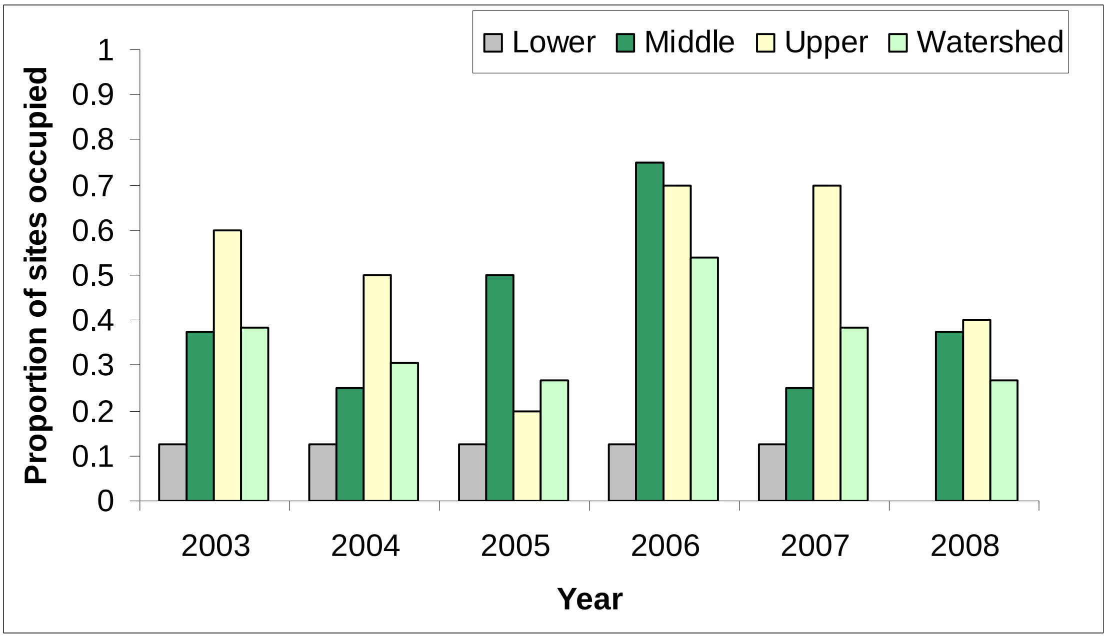
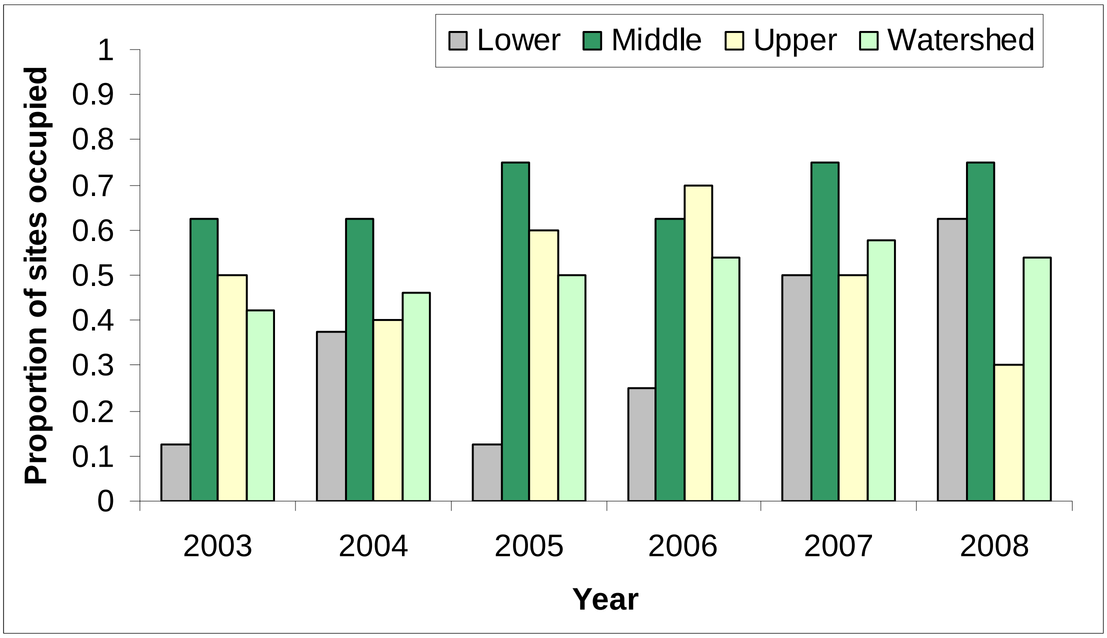


Table 4. Proportion of monitoring sites occupied by Green Frop.



APPENDIX C: ANURAN PROPORTIONAL OCCUPANCY (Cont'd)

Table 5. Proportion of monitoring sites occupied by Northern Leopard Frog.

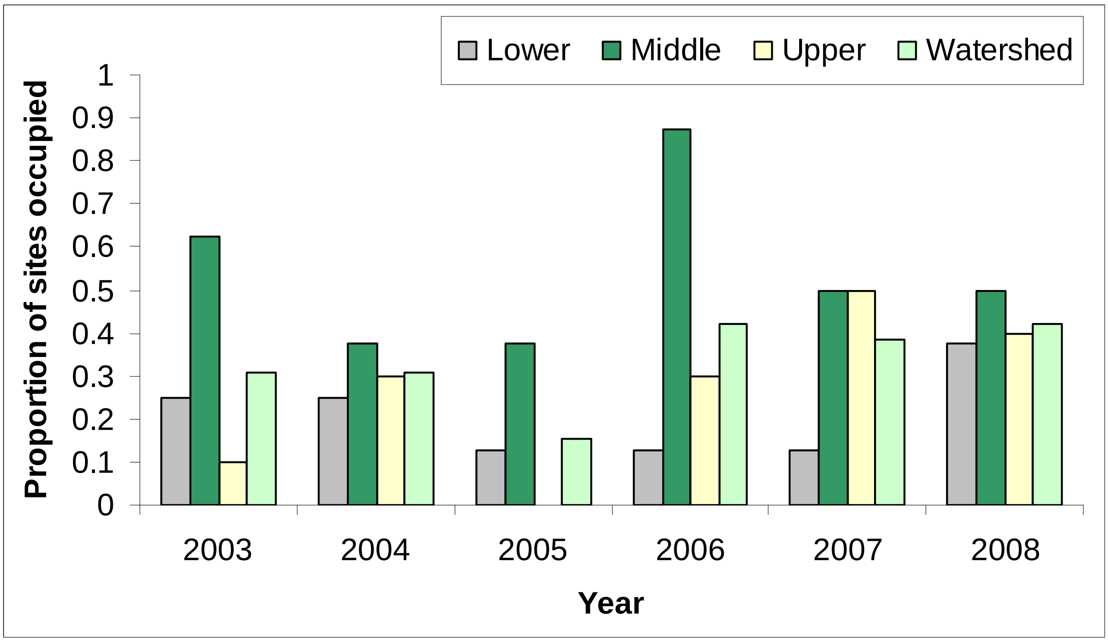
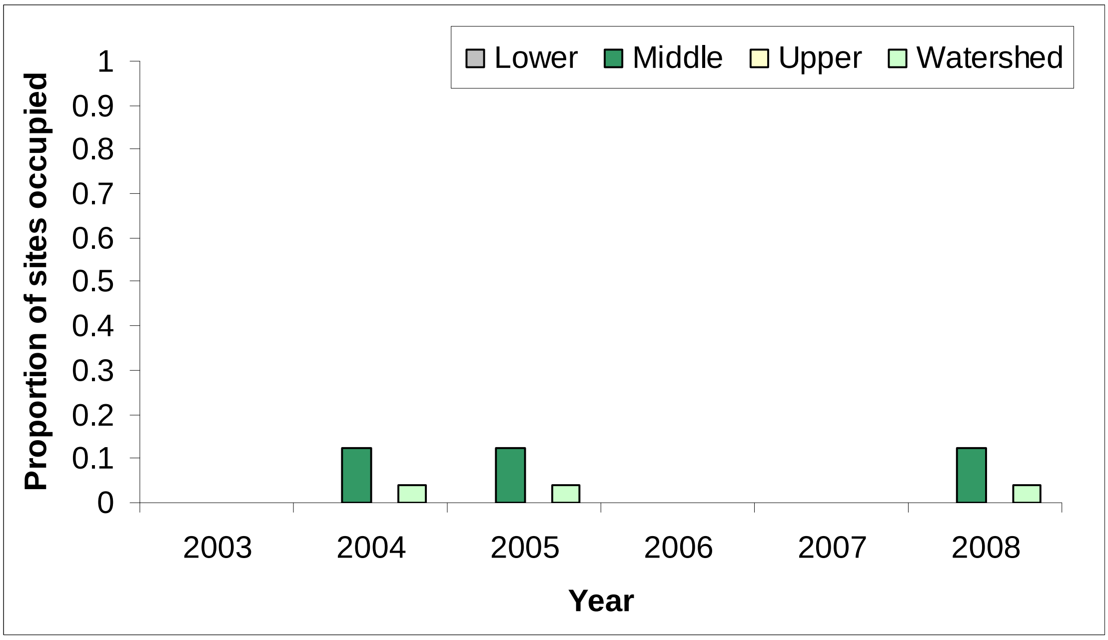


Table 6. Proportion of monitoring sites occupied by Pickerel Frog.



APPENDIX C: ANURAN PROPORTIONAL OCCUPANCY (Cont'd)

Table 7. Proportion of monitoring sites occupied by Spring Peeper.

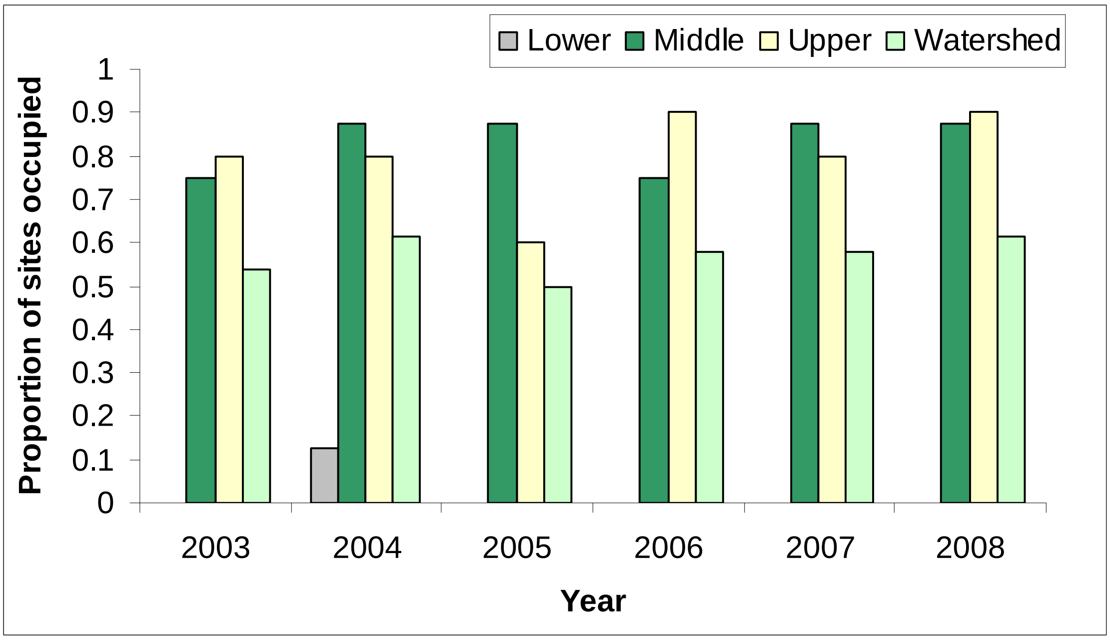
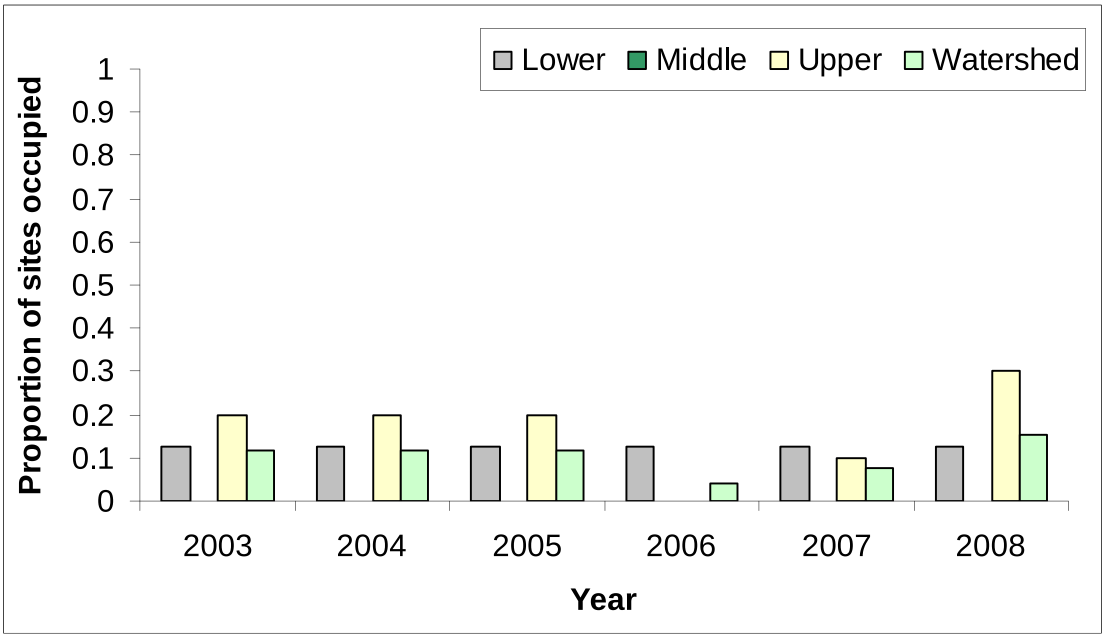
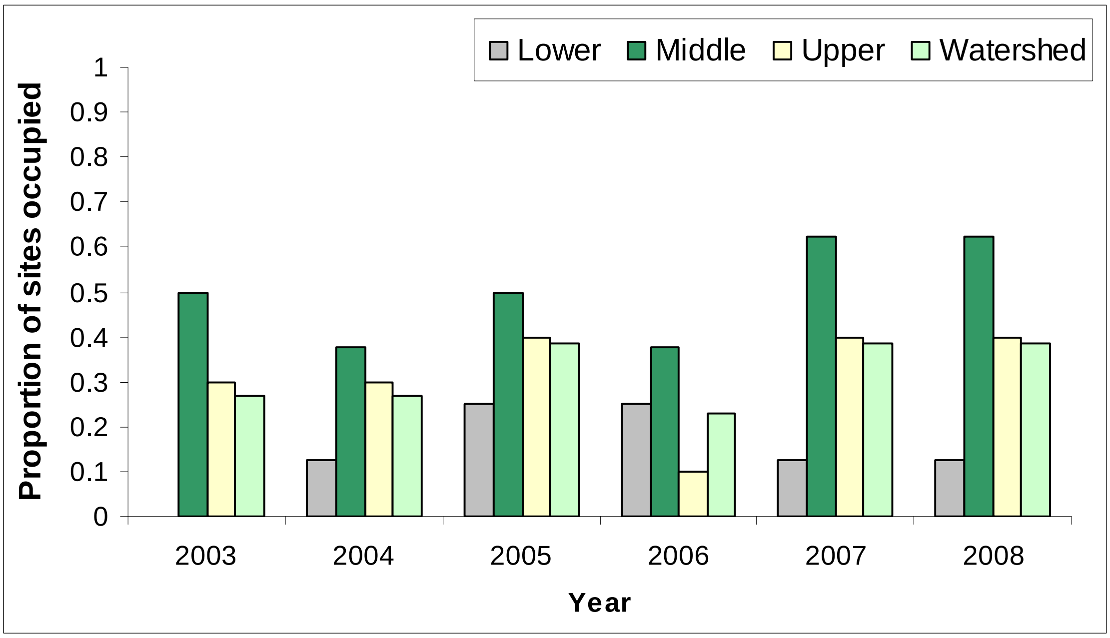


Table 8. Proportion of monitoring sites occupied by Western Chorus Frog.



APPENDIX C: ANURAN PROPORTIONAL OCCUPANCY (Cont'd)

Table 9. Proportion of monitoring sites occupied by Wood Frog.



APPENDIX D: PARAMETER VALUES ASSOCIATED WITH EACH INDIVIDUAL ANURAN MONITORING STATION IN THE CREDIT RIVER WATERSHED.

Site Name	Site Number	Physiographic Zone	Mean Species Richness [a, b]	Years Occupied [a, b]	Mean Full Chorus Detections [a,b]
Rattray Marsh	W-01	Lower	1.17	5	0
Credit River Marsh	W-02	Lower	0.5	4	0
Creditview Wetland	W-03	Lower	1.83	6	0
Meadowvale ESA Wetland	W-04	Lower	1	3	0
Hungry Hollow Wetland	W-06	Middle	0.83	4	0
Acton Swamp	W-07	Middle	3.67	6	0.5
Terra Cotta Wetland	W-08	Middle	4	6	0.83
Ken Whillans RMA	W-09	Middle	2.5	6	0
Warwick Wetland	W-10	Middle	4.17	6	0.33
Erin Pine Estates Wetland	W-11	Upper	1.83	5	0
Winston Churchill Wetland	W-12	Lower	4	6	0
Hillsburgh Wetland	W-13	Upper	0.83	3	0
Grange Orpen Wetland	W-15	Upper	0.67	3	0
Starr Wetland	W-16	Upper	2	5	0.17
Speersville Wetland	W-17	Upper	5.17	6	1.33
Caledon Lake Bog	W-18	Upper	4.17	6	0.33
Mellville Marsh	W-19	Upper	3.17	6	0
Belfountain Wetland	W-20	Upper	4.67	6	1
Rail Trail	A-01	Middle	4.67	6	0
Balinafad Ridge	A-02	Middle	5	6	1
Silver Creek	A-03	Middle	3.5	6	0.83
Alton-Hillsburgh	A-04	Upper	4.33	6	1.5
Turtle Creek	WNV-01	Lower	0.5	3	0
Mississauga Gardens	WNV-02	Lower	0.33	2	0
Churchville	WNV-03	Lower	0.83	4	0
Caledon Creek	WNV-04	Upper	2.17	6	0

[a] Per year, over six survey year
[b] The highest values within each parameter are highlighted in green, while the lowest values are highlighted in red.

Credit Valley Conservation

Terrestrial Monitoring Program Report

2005 - 2009

Monitoring Wetland Integrity within the Credit River Watershed
Chapter 3: Wetland Vegetation

CVC
CREDIT VALLEY CONSERVATION

Monitoring Wetland Integrity within the Credit River Watershed

Chapter 3: Wetland Vegetation 2005-2009

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ABSTRACT

Wetland Vegetation in the Credit River Watershed Summary of Monitoring Results 2005-2009
Temporal Trends: Species richness, species diversity and weedy species richness all increased between 2005 and 2009. The proportion of native species and species evenness both appeared to decrease. Many of the remaining parameters fluctuated between years; however, no significant trends were observed and these parameters appear to be relatively stable over the five year monitoring period.
Spatial Trends: Species richness and diversity were not different among the three physiographic zones. The Lower zone contained a lower proportion of native species than the Middle and/or Upper zones and weediness was higher in the Lower zone compared to the Middle and/or Upper zones. There was no difference among the zones for any of the floristic quality parameters or for rare species.
Relationships with Landscape Metrics: A number of vegetation parameters indicative of wetland health were positively correlated with habitat patch size, percent natural cover and matrix quality and negatively correlated with percent urban cover. The number of individuals with a weediness score of -3 and weedy species richness both displayed opposite relationships with landscape metrics. Species richness was not correlated to habitat patch size. All three selected species showed some combination of significant positive relationships with percent urban cover or significant negative relationships with percent natural cover and/or habitat patch size. None of the studied vegetation parameters or selected species displayed a significant relationship with the distance to the nearest road landscape metric.
Recommendations: Trends of changing parameters must be watched closely to determine if intervention will be needed to maintain healthy wetland vegetation communities. Of particular concern is the increase in the number of potentially problematic weedy species, the apparent decrease of the proportion of native species over the five year monitoring period and the low proportion of native species in the Lower watershed.

Wetland vegetation monitoring was conducted as part of the terrestrial component of an overall Integrated Watershed Monitoring Program established at Credit Valley Conservation. Vegetation was surveyed at 18 wetland sites throughout the Credit River Watershed from 2005 to 2009. Monitoring was conducted according to protocols developed by Credit Valley Conservation adapted from Environment Canada's Environmental Monitoring and Assessment Network.

Three hundred and seventeen wetland vegetation species were detected over the four year monitoring period, representing 26% of the vascular plant species known to occur in the watershed. Of these species, 78% were native to Ontario, 55 of which were considered locally rare and 32 of which were considered regionally rare (Kaiser 2001). In addition, 12 of the detected species were considered to be potentially problematic weedy species and are a potential threat to natural areas in the Credit River Watershed.

Species richness, diversity and weedy species richness all increased between 2005 and 2009. However, these increases may be attributed to improved species identification, site knowledge and changes to the monitoring protocol. The proportion of native species appeared to decrease, while the number of problematic weedy species (weediness score = -3) (Oldham et al. 1995) increased between 2005 and 2008. Mean Conservation Coefficient (mCC) and species evenness displayed differences among years, but appeared to be fluctuating rather than increasing or decreasing over time. The remaining parameters remained stable over the monitoring period.

The three physiographic zones in the Credit River Watershed exhibited significant differences for some vegetation parameters. Differences were observed among the Lower watershed and the other physiographic zones. The Middle and Upper watershed were not significantly different for any of the vegetation parameters examined. The Lower zone contained a lower proportion of native species, whereas the Upper and/or Middles zones contained fewer weedy species and problematic weedy species (Weediness Score = -3).

Several of the vegetation parameters including the proportion of native species, native ground vegetation diversity and FQI were significantly positively correlated with some or all of the following landscape metrics: habitat patch size, percent natural cover and matrix quality. Many of the same parameters were also negatively correlated to percent urban cover. The number of individuals with a weediness score of -3 and weedy species richness were significantly negatively correlated with habitat patch size, percent natural cover and matrix quality. Garlic Mustard, Purple Loosestrife and Common Buckthorn all showed some combination of significant positive relationships with percent urban cover or significant negative relationships with percent natural cover and/or habitat patch size. None of the studied vegetation parameters or selected species displayed a significant relationship with the distance to the nearest road.

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1.0 INTRODUCTION

1.1 BACKGROUND

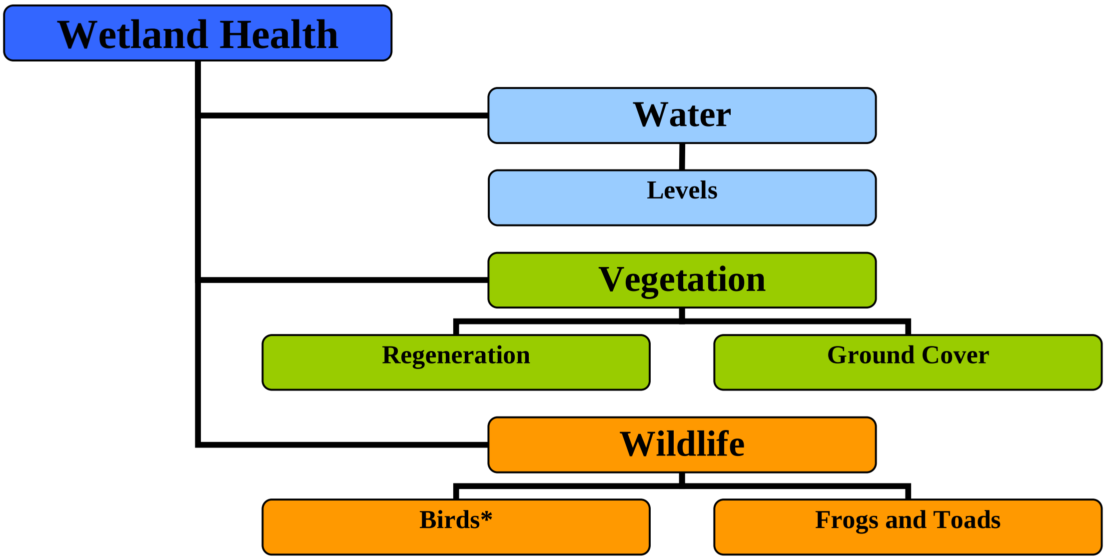
Wetlands provide numerous, irreplaceable hydrological and ecological functions, including stabilization of water supplies, flood abatement, water purification, erosion control, recharging groundwater aquifers and carbon sequestration (Zedler and Kercher 2005; Mitsch and Gosselink 2007). Despite these benefits, wetlands have historically been viewed as unsightly waste areas, and were often drained and filled to accommodate agriculture (Grand River Conservation Authority 2003). Within southern Ontario, approximately 70% of original wetland area is estimated to have been drained for agricultural purposes, increasing to over 80% near major urban centres (Natural Resources Canada 2009).

The types of alterations which lead to wetland degradation can be grouped into four categories, including: a) geomorphic and hydrologic, b) nutrient and contaminant, c) harvest, extinction and invasion, and d) climate change (Brinson and Malvarez 2002; Zedler and Kercher 2005). Current stressors specific to wetlands within the Credit River watershed include habitat removal, nutrient enrichment, organic loading, contaminants (e.g. road salts), sedimentation, turbidity, thermal changes, dehydration, inundation, exotic species, habitat fragmentation, climate change and unsustainable use (e.g. species harvesting) (Dougan and Associates 2009). Within the watershed, the relative impact of each of these processes on wetland health is still unknown. Through monitoring wetlands as part of the Terrestrial Monitoring Program, it is hoped that a deeper insight into the impact of these processes will be gained, to guide future adaptive management and restoration efforts within the Credit River Watershed.

1.2 The Importance of Long-Term Monitoring

There is an increasing demand for better accounting of the condition and health of the environment to determine whether conditions are improving or deteriorating (Niemi and McDonald 2004). An ecological monitoring program increases understanding of the trends and processes of a given ecosystem. A superior monitoring program should address both biotic and abiotic components, as well as function across multiple scales. Monitoring is useful in providing managers with information which can be used for long-term planning, because trends over time can be used to infer future conditions. From an ecological perspective, a monitoring program can also provide insight into cause and effect relationships between environmental stressors and ecosystem responses (Reeves et al. 2004).

In 2002, the Terrestrial Monitoring Program was initiated at Credit Valley Conservation (CVC) to examine the integrity of biotic and abiotic indicators in forest, wetland and riparian ecosystems throughout the watershed. The main goals of the Terrestrial Monitoring Program are to 1) measure indicators of the structure, composition and function of terrestrial ecosystems to assess the ecosystem integrity of the Credit River Watershed, 2) identify status and trends in the integrity of terrestrial communities at the watershed scale and linkages to overall watershed integrity, 3) identify spatial patterns in terrestrial community integrity, and 4) provide meaningful data on which watershed management decisions can be based. Within the watershed several wetland monitoring stations have been established in which soil, vegetation and wildlife parameters are measured to describe wetland conditions in the watershed and to monitor trends over a 25 year period (Fig. 1).



* monitoring began in 2009

Figure 1. Wetland health parameters in the Credit River Watershed.

1.3 Why Monitor Wetland Vegetation?

Vegetation monitoring provides several useful indicators of ecosystem health, including estimates of species richness, species diversity, non-native species richness, community biomass, and community productivity (Geomatics International Inc. 1999). Monitoring herbaceous vegetation provides an early warning system for environmental degradation, including climate change, environmental pollution and land-use change (Roberts-Pichette and Gillespie 1999). Furthermore, monitoring seedling and sapling regeneration provides a measure of vertical structure and allows for understanding of vegetation succession within each community.

Within wetlands, vegetation is known to be a useful indicator of biotic integrity (Simon et al. 2001; Albert and Minc 2004). Plant communities respond to water quality, hydrologic modifications, chemical pollution, and nutrient enrichment (Lopez and Fennessy 2002; Albert and Minc 2004). Submergent species richness is affected by high sediment levels, nutrient enrichment and turbidity, while emergent species also respond to culturally enriched inputs (reviewed in Van Wieren and Zorn 2005). Furthermore, non-native species are characteristic of wetland degradation (Zedler and Kercher 2004). Albert and Minc (2004) suggested that coverage of non-native species is a good measure of wetland condition, as wetlands within heavily urbanized environments tend to be dominated by non-native species.

Monitoring wetland vegetation also allows for early detection of the impacts of climate change. As the climate becomes warmer, it is expected that wetland communities will transition to drier habitat types: marsh, fen and bog communities changing to swamp communities and swamps to non-wetland communities. Wetland vegetation is expected to be highly sensitive to climate change; however, very little is known about the adaptive capacity of these habitats in the face of climate change impacts (reviewed in Dougan and Associates 2009).

1.4 Wetland Vegetation in a Changing Landscape

The Credit River Watershed encompasses 950 square kilometres of land in Southern Ontario, Canada (Credit Valley Conservation 2003). The Credit River flows southeast for nearly 100 km from its headwaters in Orangeville to its drainage point at Lake Ontario. It has been estimated that 33% of wetlands in the watershed have been lost (Dougan and Associates 2009). Most loss has occurred in areas where urban development has been most prominent. The watershed is comprised of 23% natural communities, 11% successional communities, 37% agricultural land use, and 29% urban area (Credit Valley Conservation 2007a).

Though encompassing many unique landscape formations, the Credit River Watershed can be divided into three main physiographic zones based on topography, physiographic regions and subwatershed boundaries (Fig. 2) (Credit Valley Conservation 2007b). The Lower physiographic zone is highly urbanized, containing over 85% of the population of the watershed. The topography of this area is relatively flat with a gentle slope towards Lake Ontario and has been significantly altered by human development. The Lower watershed is comprised of lower permeability sediments, such as silt and clay till, laying on top of Georgian Bay and Queenston shales (Credit Valley Conservation 2007b). The soils in the Lower zone are comprised of clay loams associated with the Peel Plain and the South Slope (Credit Valley Conservation 2007b). These soils have low permeability in relation to the rest of the watershed; however, localized pockets of sand and gravel exist. When compared against government standards (Environment Canada 2004), the amount of both wetland and forest cover in the Lower watershed is classified as poor.

The Middle zone contains the Niagara Escarpment, a landform comprised of steep slopes, rocky outcrops, and thin soil (Credit Valley Conservation 2005); much of this zone is protected under the Greenbelt Plan (OMMAH 2005). The steep topography of this area leads to relatively high runoff volumes and velocities; however, the high forest cover slows runoff and increases infiltration. Soils in this zone are variable as a result of the changing physiography between the Upper and Lower zones (Credit Valley Conservation 2007b). The east portion of this zone is underlain by Queenston Shale, whereas the west portion is underlain by dolomite of the Amabel/Lockport Formation. Surface geology is dominated by silt to clay tills and silty sand to sandy silt, with pockets of ice-contract stratified drift and bedrock or bedrock drift (Credit Valley Conservation 2007b). Due to topographic factors, lower levels of urbanization and protective legislation, the Middle zone contains the greatest proportion of natural cover in the watershed.

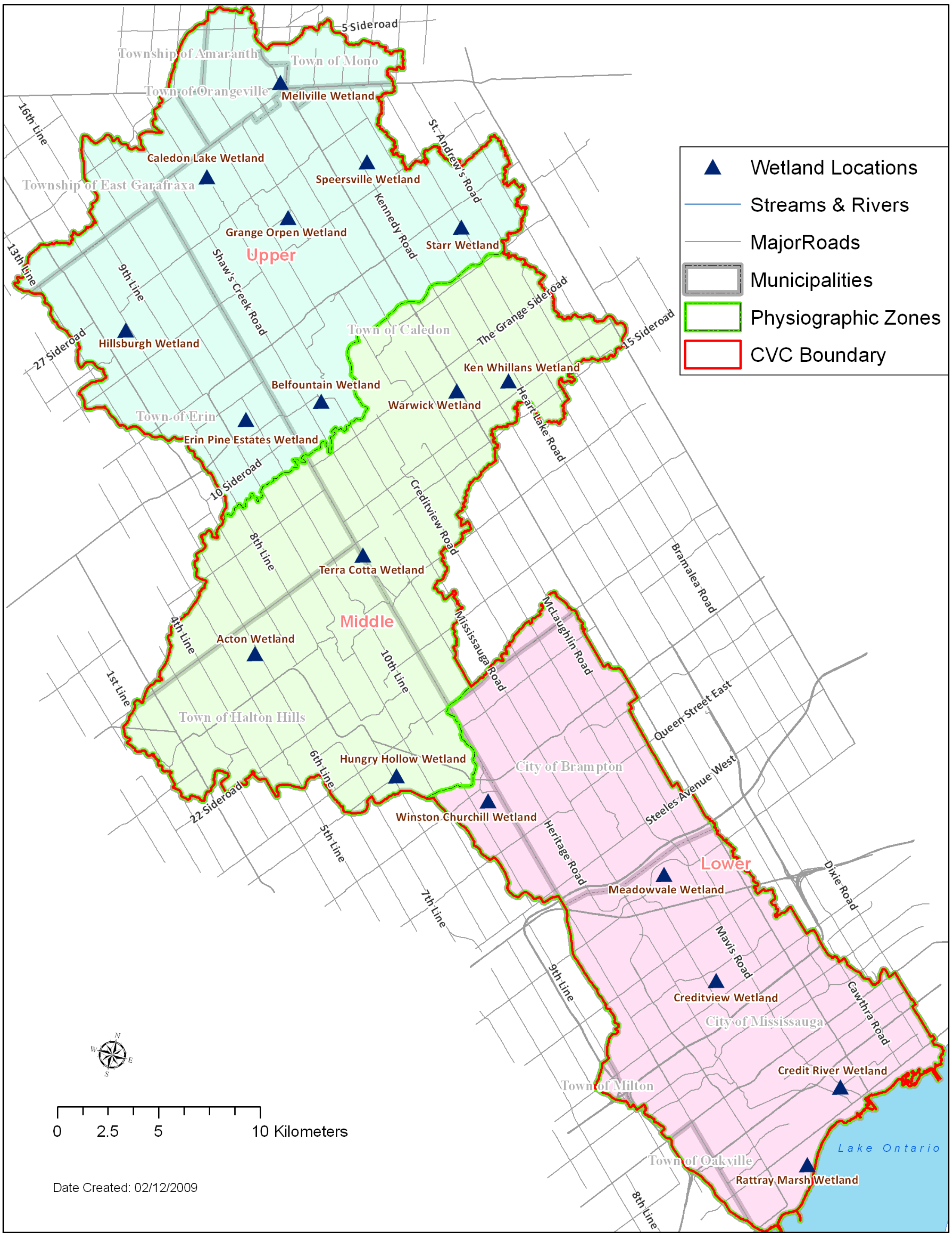


Figure 2. The Credit River Watershed wetland monitoring site locations.

The Upper physiographic zone lies above the escarpment and is characterized by till plains, moraines and glacial spillways (Credit Valley Conservation 2007b). This area is dominated by sandy loam soils that are associated with the Guelph drumlin field and the Hillsburgh Sandhills (Credit Valley Conservation 2007b). The high groundwater recharge rates in the Upper watershed are supported by the relatively high permeability rates of these soils. The Amabel/Lockport Formation is the dominant bedrock type in this area with pockets of the Guelph and Manitoulin dolostones (Credit Valley Conservation 2007b). Surficial geology is dominated by ice-contact stratified drift, silty sand to sandy silt and glaciofluvial gravel. Agricultural land use dominates portions of the Upper watershed, though it also contains several large wetland complexes and numerous headwater streams (Credit Valley Conservation 2007b).

Research in Ontario has demonstrated that the proportion of natural land cover surrounding wetlands increases the diversity of mammals, herptiles, birds and plants (Findlay and Houlahan 1997). In fact, a reduction in wetland area of 50% is expected to result in a loss of 10-16% of wildlife in any taxa group (Findlay and Houlahan 1997). In addition, the development of roads has been shown to alter both groundwater and stream flow (Forman and Alexander 1998), which is in turn expected to have direct impacts on wetland vegetation. Road construction has also been suggested to assist in the spread of non-native species such as Purple Loosestrife (*Lythrum salicaria*) (Forman and Alexander 1998).

Overall, the extent and configuration of natural land cover in the landscape have the ability to alter ecosystem processes, nutrient availability, water availability, and dispersal. These changes can then alter wetland vegetation communities in the watershed. Therefore, observed trends in wetland vegetation will be considered in the context of the changing landscape of the Credit River Watershed.

1.5 Objective: Monitoring Question

Several measures of wetland vegetation are currently being monitored to examine the ecosystem integrity of wetland communities within the Credit River Watershed (Credit Valley Conservation 2010a). This report examines the integrity of the ground vegetation and regeneration layers using a variety of vegetation parameters, including: species richness, evenness and diversity, and measures of floristic quality (mCC, FQI, wetness and weediness). In addition, vegetation parameters were compared to several landscape measures because reliable indicators of ecosystem integrity are expected to be sensitive to anthropogenic disturbance (Noss 1999). Finally, several individual species were examined to determine if their presence was changing over time and to determine if they are sensitive to anthropogenic disturbance. The overall monitoring question being examined in this report is:

ARE INDICATORS OF HEALTHY WETLAND VEGETATION IN THE CREDIT RIVER WATERSHED STABLE?

More specifically, 1) are species richness and diversity differing spatially or temporally through the watershed; 2) does vegetation community similarity differ spatially or temporally through the watershed; 3) are indicators of floristic quality differing spatially or temporally throughout the watershed; 4) do any plant groups differ spatially or temporally throughout the watershed; 5) are there relationships among wetland vegetation and landscape metrics in the watershed; and 6) are there relationships between selected species and landscape metrics in the watershed (Table 1).

Trend analyses were used to examine parameters of interest for increasing or decreasing trends over time, whereas temporal analyses were used to examine if there were differences in parameters among years. Spatial analyses were used to examine whether there was a difference among the three physiographic zones in the Credit River Watershed. Plant groups analyzed include the number of species with high conservatism scores (CC 8-10), the number of locally and regionally rare species, and the number of potentially problematic weedy species (Weediness Score = -3) (see Methods section for details).

Table 1. Wetland vegetation monitoring framework for the Credit River Watershed.

Monitoring Question	Monitoring Variable [a]	Unit of Measurement	Vegetation Layer [b]	Analysis Method
Trend Analyses				
Is species richness changing over time in the Credit River Watershed?	Number of species	Count	Combined vegetation and Ground vegetation	- Linear Regression - Power Analysis
Is total species diversity changing over time in the Credit River Watershed?	Shannon-Weiner Function	Index	Ground vegetation	- Linear Regression - Power Analysis
Is native species diversity changing over time in the Credit River Watershed?	Shannon-Weiner Function	Index	Ground vegetation	- Linear Regression - Power Analysis
Is the mean Coefficient of Conservatism changing over time in the Credit River Watershed?	Mean Coefficient of Conservatism value	Index	Combined vegetation	- Linear Regression - Power Analysis
Is the Floristic Quality Index changing over time in the Credit River Watershed?	FQI	Index	Combined vegetation	- Linear Regression - Power Analysis
Is weedy species richness changing over time in the Credit River Watershed?	Number of species	Count	Combined vegetation	- Linear Regression - Power Analysis
Temporal Analyses				
Did species richness differ among years in the Credit River Watershed?	Number of species	Count	Combined vegetation Ground vegetation, Regeneration	- Repeated Measures ANOVA - Friedman test
Did the proportion of native species differ among years in the Credit River Watershed?	Proportion native species	Proportion	Ground vegetation, Combined vegetation, Regeneration	- Friedman test
Did species diversity differ among years in the Credit River Watershed?	Shannon-Weiner Function	Index	Total ground vegetation, Native ground vegetation, Regeneration	- Repeated Measures ANOVA - Friedman test
Did species evenness differ among years in the Credit River Watershed?	Simpson's Dominance Index	Index	Total ground vegetation, Native ground vegetation, Regeneration	- Friedman test
Did the mean Coefficient of Conservatism differ among years in the Credit River Watershed?	Mean Coefficient of Conservatism value	Index	Combined vegetation	- Repeated Measures ANOVA

Monitoring Question	Monitoring Variable [a]	Unit of Measurement	Vegetation Layer [b]	Analysis Method
Temporal Analyses (Continued)				
Did the number of species with a Conservatism score of 8-10 differ among years in the Credit River Watershed?	Number of species with a Conservatism score of 8-10	Count	Combined vegetation	- Friedman test
Did the mean Floristic Quality Index differ among years in the Credit River Watershed?	FQI	Index	Combined vegetation	- Repeated Measures ANOVA
Did weedy species richness differ among years in the Credit River Watershed?	Number of species	Count	Combined vegetation	- Repeated Measures ANOVA
Did the mean number of species with a weediness score of -3 differ among years in the Credit River Watershed?	Mean number of species with a weediness score of -3	Count	Combined vegetation	- Friedman test
Did the number of locally and regionally rare species differ among years in the Credit River Watershed?	Number of locally and regionally rare species	Count	Combined vegetation	- Friedman test
Spatial Analyses				
Did species richness differ among physiographic zones in the Credit River Watershed?	Number of species	Count	Combined vegetation, Ground vegetation, Regeneration	- Repeated Measures ANOVA - One-way ANOVA - Kruskal-Wallis test - Power Analysis
Did the proportion of native species differ among physiographic zones in the Credit River Watershed?	Proportion native species	Proportion	Ground vegetation, Combined vegetation, Regeneration	- Kruskal-Wallis test
Did species diversity differ among physiographic zones in the Credit River Watershed?	Shannon-Weiner Function	Index	Total ground vegetation, Native ground vegetation, Regeneration	- Repeated Measures ANOVA - Kruskal-Wallis test - Power Analysis
Did species evenness differ among physiographic zones in the Credit River Watershed?	Simpson’s Dominance Index	Index	Total ground vegetation, Native ground vegetation, Regeneration	- Kruskal-Wallis test
Did the mean Coefficient of Conservatism differ among physiographic zones in the Credit River Watershed?	Mean Coefficient of Conservatism value	Index	Combined vegetation	- Repeated Measures ANOVA - Power Analysis

Monitoring Question	Monitoring Variable [a]	Unit of Measurement	Vegetation Layer [b]	Analysis Method
Spatial Analyses (Continued)				
Did the number of species with a Conservatism score of 8-10 differ among physiographic zones in the Credit River Watershed?	Number of species with a Conservatism score of 8-10	Count	Combined vegetation	- Kruskal-Wallis test
Did the Floristic Quality Index differ among physiographic zones in the Credit River Watershed?	FQI	Index	Combined vegetation	- Repeated Measures ANOVA - One-way ANOVA - Power Analysis
Did the number of locally and regionally rare species differ among physiographic zones in the Credit River Watershed?	Number of locally and regionally rare species	Count	Combined vegetation	- Kruskal-Wallis test
Did weedy species richness differ among physiographic zones in the Credit River Watershed?	Number of species	Count	Combined vegetation	- Repeated Measures ANOVA - Power Analysis
Did the mean number of species with a weediness score of -3 differ among physiographic zones in the Credit River Watershed?	Mean number of species with a weediness score of -3	Count	Combined vegetation	- Kruskal-Wallis test
Species Level Analyses				
Was there a relationship between selected species and landscape metrics in the Credit River Watershed?	Selected species compared to habitat patch size, distance to the nearest road and % urban and natural cover	Presence / Absence	Ground vegetation	- Logistic Regression
Did site occupancy of selected species change over time in the Credit River Watershed?	Proportion of sites occupied by selected species	Proportion	Ground vegetation	- Cochrane-Armitage - Power Analysis
Landscape Analysis				
Were wetland vegetation parameters correlated with landscape metrics in the Credit River Watershed?	All above vegetation parameters compared to habitat patch size, distance to the nearest road, % urban, natural and agriculture cover and matrix quality	Variable	Combined vegetation, Ground vegetation, Regeneration	- Spearman Rank Correlation

[a] All monitoring variables were summarized for each individual site in each year across the monitoring period.
[b] Ground vegetation includes all herbaceous plants, woody vines, and shrubs and trees <16cm. Regeneration includes trees and shrubs $\geq$ 16cm and < 4cm in diameter at breast height (DBH). Combined vegetation includes data from both the ground vegetation and regeneration layers.

2.0 METHODS

2.1 WETLAND HEALTH MONITORING

Eighteen permanent wetland monitoring sites were established in 2002 throughout the Credit River Watershed. These sites were monitored for changes in vegetation from 2005-2009. The wetland sites represent six swamp and 12 marsh communities (Appendix A). The monitored wetlands represent 11 Provincially Significant Wetlands (PSW's), 11 Environmentally Sensitive Areas (ESA's) and six Life Science Areas of Natural and Scientific Interest (ANSI). Wetland Health in the Credit River Watershed was monitored following modified plot-based methodologies developed by Environment Canada's Environmental Monitoring and Assessment Network (EMAN) and CVC. All vegetation parameters were measured annually or bi-annually at the monitoring sites. Wetland monitoring sites consisted of 12 paired sub-plots spaced ten metres apart, established along a 50 m linear transect, which ran along the wetland's hydrological gradient (from dry to wet) (Fig. 3). At each subplot a 1x1m ground vegetation subplot was nested within a 2x2 m regeneration subplot, which shared the same centre point.

2.1.1 Ground Vegetation

Ground vegetation is acutely sensitive to finer-scale disturbances in microclimate (e.g. soil moisture and temperature), so it is an integral early indicator of changes in wetland health (Geomatics International Inc. 1999; Simon et al. 2001; Albert and Minc 2004). At each 1x1m monitoring subplot, species identification was recorded for all herbaceous vegetation (grasses, sedges, ferns etc.) and trees and shrubs with a height <16cm. Each species was also assigned to one of the following cover classes: <1%, 1-5%, 6-15%, 15-30%, 31-50%, 51-75% and 76-100%. In 2008, methodology was revised for assessing wetland vegetation abundance from "percent cover" classes to the actual percent cover a species occupies in a subplot. This was done to enable better analysis of data as the cover classes were found to be too coarse to yield meaningful statistical results. For analysis, 2008 and 2009 percent cover data were reassigned to the appropriate cover class for consistency in data presentation across all years. Ground vegetation was monitored bi-annually, in spring and late summer, to ensure accurate identification of species and to capture plants blooming at different times throughout the season. Methodology followed standardized EMAN protocol (Roberts-Pichette and Gillespie 1999).

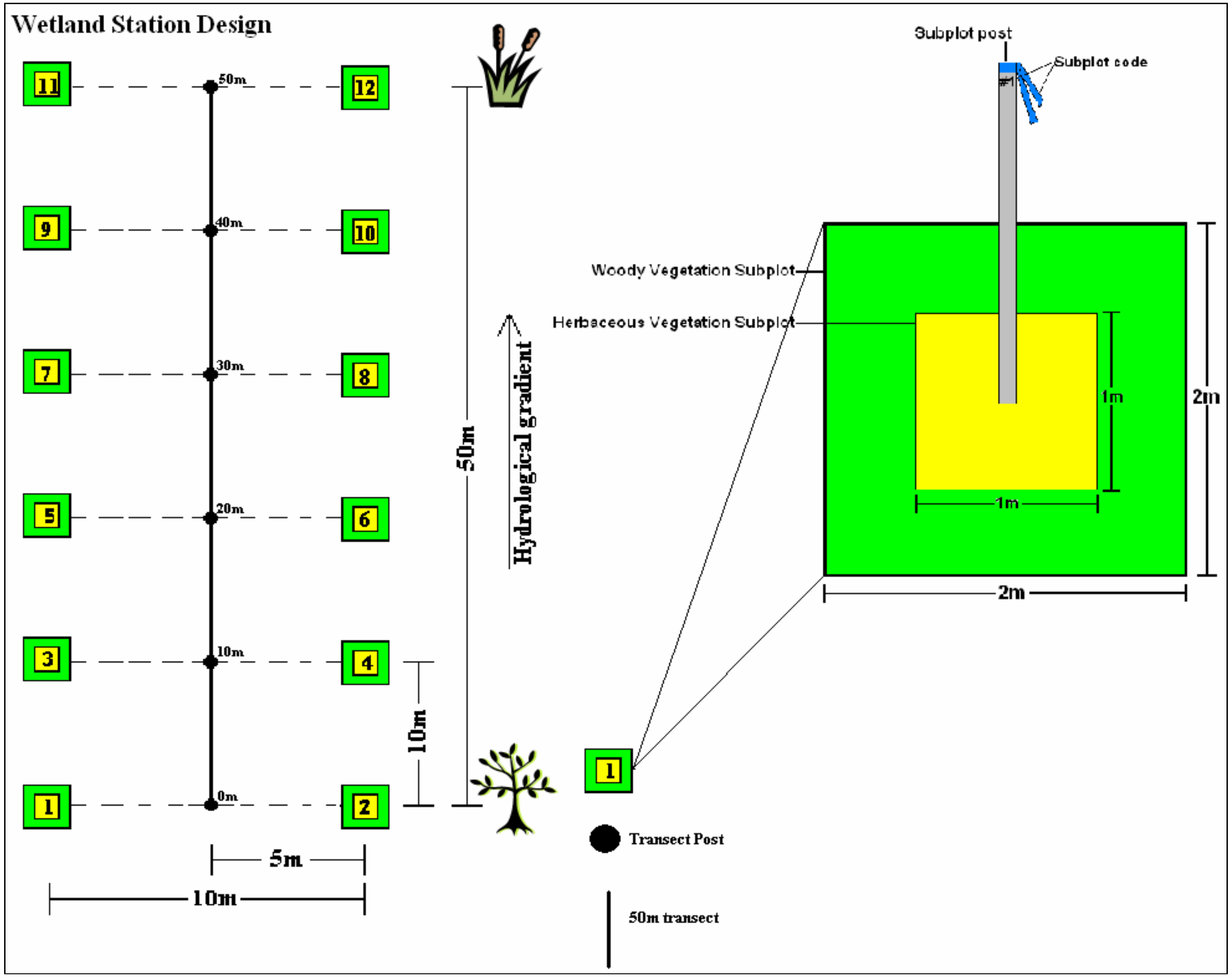


Figure 3. Wetland monitoring site layout.

2.1.2 Regeneration

Regeneration monitoring provides insight into what trees will comprise the future canopy layer of wetlands of the Credit River Watershed. Regeneration was monitored annually in the late summer. Following modified EMAN protocols, tree and shrub stems ≥16cm in height and <4cm Diameter at Breast Height (DBH) originating within each subplot were identified to species and assigned to one of the seven cover classes noted above (Sajan 2000).

2.2 DATA PREPARATION

Although non-vascular plants also provide valuable information about ecosystem health, due to resource constraints and the specialized knowledge required for their identification, only vascular plants were included in analyses. Therefore, any observations of mosses, liverworts and algae were removed from the dataset. All vascular plants identified to species were included in the dataset. Plants identified to genus were included in a grouping if no other plant of that genus was identified to species within the group in question, unless otherwise noted below. These genus-only records were included to provide an accurate representation of the number of species within a site, even if species-level identification was not possible.

Woody species excluded from the regeneration survey included woody vines such as Poison Ivy (*Toxicodendron radicans ssp. negundo*) and woody species that do not have the potential to contribute to the dominant canopy layer in the future, such as Bunchberry (*Cornus canadensis*). A complete list of all woody species not included in the regeneration analyses is presented in Table 2. These species were instead included in those ground vegetation analyses which did not rely on percent cover data.

Table 2. List of woody species removed from the regeneration analyses

Common Name	Scientific Name
Bog Rosemary	*Andromeda polifolia var. glaucophylla*
Bunchberry	*Cornus canadensis*
Climbing Nightshade	*Solanum dulcamara*
Dwarf Red Raspberry	*Rubus pubescens*
Large Cranberry	*Vaccinium macrocarpon*
Poison Ivy	*Toxicodendron radicans ssp. negundo*
Riverbank Grape	*Vitis riparia*
Small Cranberry	*Vaccinium oxycoccos*
Thicket Creeper	*Parthenocissus inserta*
Twinflower	*Linnaea borealis ssp. longiflora*
Virgin's-Bower	*Clematis virginiana*

2.2.1 Vegetation Parameters

Data summaries were conducted for several vegetation parameters listed below. Data was summarized for ground vegetation and regeneration layers separately, as well as for combined vegetation, which incorporates data from both the ground and regeneration layers.

2.2.1.1 Most Common Species: Lists of the most common species in the watershed were developed using only plants identified to species level. The lists were determined by summing the total number of monitoring sites in which each species was observed per year, regardless of the abundance of the species at each site.

2.2.1.2 Species Richness: Richness was calculated by counting the number of unique vegetation species identified watershed-wide and at each site per year. Richness calculations were performed for ground vegetation, regeneration and combined vegetation. In addition, the proportion of native species (ground vegetation, regeneration and combined vegetation) was determined watershed-wide and at each site per year. Proportion of native species was calculated using plants identified to species-level, as:

$$\mathbf{P = \frac{N}{T}}$$

Where P = Proportion of native species
N = Number of native species
T = Total number of species (native + non-native).

2.2.1.3 Species Diversity: Vegetation species diversity in the Credit River Watershed was calculated using the Shannon-Wiener Function:

$$H' = -\sum_{i=1}^{S} p_i \ln p_i$$

Where H' = Index of species diversity
s = Number of species
p_i = Proportion of total sample belonging to ith species determined by percent cover

Diversity values at each monitoring site were calculated separately for each subplot and then averaged to obtain the diversity score for each site. Traditionally, p_i is calculated using the number of individuals of each species. Unfortunately, these data were not available. Instead, cover class was used to determine the proportion of the subplot occupied by each species. To do this, cover classes were first converted to whole numbers by using the mid-point of each cover class (e.g. the 5-10% cover class was converted to 7.5%). The total cover for the subplot was then calculated by summing the cover classes for all detected species. Next, p_i was calculated by dividing the cover class of $species_i$ by the total cover of the subplot. Diversity for each subplot was then calculated using the formula described above. It is important to note that because p_i was calculated using percent cover, opposed to abundance, H' values in this study cannot be compared to other diversity indices for vegetation within the literature.

In a community with only one species $H' = 0$. As H' increases, communities increase in diversity (Krebs 1999). This index was calculated at each site per year. Total diversity was calculated for both ground vegetation and regeneration. However, diversity was not calculated for regeneration in 2009 because of changes in the regeneration sampling protocol which rendered the 2009 incomparable to previous years data. Native diversity was only calculated for ground vegetation, using plants identified to species-level, because of low non-native richness in the regeneration data.

2.2.1.4 Evenness: Species evenness, *E*, was calculated based on Simpson's dominance index (Simpson 1949):

$$E = (1/\sum_{i=1}^{S} p_i^2)/S$$

where p_i is the proportional cover of species *i* and *S* is species richness. Cover class was determined using the same process as for diversity. Evenness is a measure of how evenly the cover within a plot is distributed between species. Evenness values range from 0 to 1. An evenness value of 1 indicates that plant cover within a plot is evenly shared among the species present (e.g. only one species is present, two species both account for 50% of the cover, etc.). A low evenness value indicates that one or a few plants account for the majority of the cover, whereas other plants cover very little of the plot. Regeneration could not be calculated in 2009 for regeneration due to changes in sampling protocol.

2.2.1.5 Floristic Quality Assessment: Plants can reflect the quality or condition of an ecosystem. This relationship between vegetation and site quality was assessed using the Ontario Ministry of Natural Resources Floristic Quality Assessment System for Southern Ontario (Oldham et al. 1995). This system assigned each vegetation species known to occur in southern Ontario several values, including a wetness index, coefficient of conservatism and weediness score. Each index is based on species habitat requirements and these scores were used to calculate several vegetation parameters described below.

Wetness Index

Mean wetness index was calculated for each site using the species specific wetness values listed in the Floristic Quality Assessment System for Southern Ontario (Oldham et al. 1995). Wetness values were assigned to species based on their requirement for wet habitats on a scale of -5 (obligate wetland) to 5 (obligate upland). The mean wetness index was calculated by averaging the wetness scores for each species detected at a given site, per year.

Coefficient of Conservatism

Coefficient of conservatism (CC) scores have been assigned to native vegetation species in the Floristic Quality Assessment System for Southern Ontario (Oldham et al. 1995) and are based on a species tolerance to disturbance and habitat fidelity. These scores were used to calculate three vegetation parameters, including the mean Coefficient of Conservatism (mCC), Floristic Quality Index (FQI) and the number of species with CC values of 8-10 (CC 8-10). Coefficient of Conservatism scores for each species range from 0 (low conservatism) to 10 (high conservatism) and a conservatism value of 8, 9 or 10 indicates that a species is a habitat specialist (OMNR 2008). Mean Coefficient of Conservatism was calculated by averaging the CC scores for each native species detected at a given site. In addition to mCC, the total number of species with a CC value of 8-10 was also calculated for each site. Finally, the Floristic Quality Index was calculated as follows:

$$\mathbf{FQI = mCC} \times \sqrt{\mathbf{N}}$$

where mCC is the mean Coefficient of Conservatism and N is native species richness.

Mean Coefficient of Conservatism and FQI are both measures of the floristic integrity of a given site. Mean Coefficient of Conservatism is solely based on the habitat requirements of detected species, while FQI incorporates species richness into its calculation. Both parameters are useful for examining vegetation integrity, as a site with a high FQI may have a low mCC value and vice versa. For example, a relatively degraded site may have a high FQI value if a large number of disturbance-tolerant native species present (Taft et al. 1997). Given that mCC and FQI have the potential to rank sites differently, it is important to consider both when examining the overall quality of a given location. Mean Coefficient of Conservatism, FQI and CC 8-10 were calculated for each individual site, per year.

Weediness

Weediness was evaluated using the weediness scores provided for non-native species in the Floristic Quality Assessment System for Southern Ontario (Oldham et al. 1995). Weediness scores have been assigned to all non-native species and range from -1 (low impact of the species on natural areas) to -3 (high impact of the species on natural areas). The weedy species richness was calculated for each site by totalling the number of species with a weediness score of either -1, -2 or -3 at a given site. Also, in order to quantify the impact of non-native species that may have serious effects on natural areas in the watershed, the total number of species with a weediness score of -3 (weediness score -3) was calculated. These weediness parameters were calculated for each individual site, per year.

2.2.1.6 Locally and Regionally Rare Species: In order to understand the composition of rare species within the watershed, the number of both locally and regionally rare species was determined using plants identified to species-level watershed-wide and at each site per year. Local and regional rareness designations were based on

Kaiser (2001). The criteria used for determining if a plant was locally or regionally rare were as follows:

- Locally rare - a species that occurred fewer than 10 times in the Region of Peel
- Regionally rare - a species that occurred fewer than 40 times in the Greater Toronto Area (Kaiser 2001).

2.2.1.7 Community Similarity: Vegetation community change was calculated with plants identified to species-level using Jaccard's Similarity Index (Krebs 1999):

$$S_j = \frac{a}{a + b + c}$$

Where S_j = Jaccard's similarity coefficient
a = number of species in year 1 and year 2 (joint occurrences)
b = number of species in year 2 but not in year 1
c = number of species in year 1 but not year 2

Jaccard's Similarity Index is a calculated value used to compare the species similarity between two samples. The index was calculated watershed-wide for the combined vegetation data for 2005 vs. 2006, 2006 vs. 2007, 2007 vs. 2008, 2008 vs. 2009 and 2005 vs. 2009. In addition, Jaccard's Similarity Index was also determined across all years among the physiographic zones, including Lower zone vs. Middle zone, Lower zone vs. Upper zone, and Middle zone vs. Upper zone.

2.2.2 Statistical Analyses

2.2.2.1 Trend and Temporal Analysis: Population trends between 2005 and 2009 were analyzed using linear regression in STATISTICA 7.0 (Statsoft Inc.), with results considered significant when $p<0.05$. Prior to analyses, data were tested for normality and homogeneity of variance using the Shapiro-Wilk test and Levene's test, respectively. If required, data were transformed in order to improve normality and homogeneity (Appendix B). Trend analysis was completed for the vegetation parameters listed in Table 3. Regeneration parameters could not be analysed using linear regression due to a spatial correlation among sites in the regeneration datasets. Trends could not be determined in the proportion of native species and evenness parameters because of the ordinal nature of the data, which necessitated the use of non-parametric tests (McDonald 2009). Weediness score -3, CC 8-10, locally rare and regionally rare parameters were analyzed with non-parametric methods because of the low number of species detected within these groups at each site.

In addition, data were also examined to determine if differences occurred among years. Differences among years were determined with either the parametric repeated measures ANOVA test or the non-parametric Friedman test, depending on the parameter. Although a one-way ANOVA would have been sufficient to test for differences among years, the repeated measures ANOVA was used because this test was already being utilized to determine spatial differences in the data and provided equivalent temporal comparisons to a one-way ANOVA. The Friedman test, a non-parametric equivalent of the repeated measures ANOVA, was used to identify differences among sampling periods (2005-2009) for non-parametric data. In lieu of post-hoc analysis, which is not available for the Freidman test, differences among groups must be inferred through visual examination of the data.

Table 3. Summary of temporal and spatial analyses used to examine vegetation parameters within the Credit River Watershed.

Vegetation Parameters	Parametric Trend Analysis	Parametric Temporal and Spatial Analyses	Non-Parametric Temporal and Spatial Analyses
Ground Vegetation			
Species Richness	X	X	
Species Diversity	X	X	
Species Evenness			X
Proportion of Native Species Richness			X
Native Species Diversity	X	X	
Native Species Evenness			X
Regeneration			
Species Richness			X
Species Diversity			X
Species Evenness			X
Proportion of Native Species Richness			X
Combined Vegetation			
Species Richness	X	X	
Proportion of Native Species Richness			X
Mean Wetness Index [a]	X		
mCC [b]	X	X	
CC 8-10 [b]			X
FQI [b]	X	X	
Weedy Species Richness	X	X	
Weediness Count -3			X
Locally Rare Species			X
Regionally Rare Species			X

[a] Spatial analysis not conducted because start and end points of wetness gradient vary by site.
[b] mCC, Mean Coefficient of Conservatism; CC 8-10; number of species with a Coefficient of Conservatism between 8 and 10; FQI, Floristic Quality Index.

A selected group of wetland vegetation species were also examined for trends using the Cochrane-Armitage test in XLStat (Addinsoft) (Agresti 2002). This test is often used to examine linear trends in binomial proportions of response across increasing levels of dosage in medical studies. In this study, the test has been used to examine if species occurrence was increasing or decreasing over time. Vegetation species chosen for this analysis were selected due to interest in a species distribution and the suitability of its distribution for the required statistical tests. Species were considered for selection if they were of some interest to the program either because they were non-native, uncommon in the watershed or a common native plant. In addition, the species' distribution needed to be appropriate as the statistical analyses would not be meaningful if the species were present at very few or almost all of the sites. Based on the preceding criteria, ground vegetation species examined included: Garlic Mustard (*Alliaria petiolata*), Common Buckthorn (*Rhamnus cathartica*) and Purple Loosestrife.

2.2.2.2 Spatial Analysis: To examine spatial patterns of wetland vegetation parameters in the Credit River Watershed, monitoring sites were grouped into Lower, Middle and Upper zones, with five, five and eight sites established in each zone, respectively (Fig. 2). It is hypothesized that differences in land cover and landscape configuration among the three zones are likely to influence wetland health parameters, including species richness, diversity and floristic quality.

Data were analyzed with STATISTICA 7.0 (Statsoft Inc.), with results considered significant when $p<0.05$ (Zar 1999). Data were tested for normality and homoscedasticity using the Shapiro-Wilk test and Levene's test for homogeneity of variance, respectively. If required, data were transformed in order to improve normality and homogeneity (Appendix B). Where data could not be normalized, the equivalent non-parametric test was used to complete analyses. Repeated measures ANOVA (Analysis of Variance) was used to examine differences in vegetation parameters among the three physiographic zones (Table 3). Tukey's HSD test for unequal sample sizes was used for post-hoc comparisons in order to compensate for the larger number of sites in the Upper zone compared to the Middle and Lower zones (Zar 1999). When a significant interaction is detected between zone and year in the repeated measures ANOVA it is not possible to distinguish the main effects of the parameters (Underwood 1997). In this situation, one-way ANOVAs were completed for each individual year in order to determine differences among the physiographic zones. The Kruskal-Wallis test was conducted on annual data sets to identify differences among the three physiographic zones for non-normalized data. This is a non-parametric version of a one-way ANOVA in which treatment levels of an independent variable (e.g. physiographic zone) are independent of one another. Zone effects had to be analysed on a year by year basis because Kruskal-Wallis cannot account for year effect when all survey years are used in the same analysis. An associated post-hoc test was subsequently run to determine which zone relationships were significant.

2.2.2.3 Correlation between Vegetation Parameters and Landscape Metrics: To increase understanding of the relationships between wetland vegetation populations and the landscape which surrounds them, Spearman Rank correlations were used to determine if monotonic associations exist between vegetation parameters and landscape metrics. Monotonic associations are instances where one variable increases or decreases with the other variable, but not necessarily in a linear manner. Correlations were run pairing the mean of each vegetation parameter across the monitoring period (Table 3) to habitat patch size, distance to nearest road, matrix quality, and finally percent agriculture, natural and urban area within a 2 km radius. Landscape parameters were based on orthophotography collected between 2005 and 2007 over the Credit River watershed. Aerial photographs were interpreted and analysed using ArcGIS, a software package capable of generating information on patterns, structure, and change within a landscape (Credit Valley Conservation Authority 2007a).

Habitat patch is defined as a continuous patch of natural land cover in a given area, in which the monitored wetland area is included, and is comprised of forests, wetlands and successional communities. To calculate percent natural, urban and agricultural area, a 30 m buffer was developed around the centre of the monitoring site, and the percentage of land use for each category was calculated within a 2 km radius from the edge of the buffer. The 30 m buffer was used to develop a centre point which would encompass each wetland site. Finally, matrix quality is an index developed to account for the amount of varying types of land uses in an area, with higher matrix quality values indicating more natural land cover in the area. Matrix influence is calculated by summing percent natural area multiplied by positive one, with percent agricultural area multiplied by zero, and percent urban area multiplied by negative one (TRCA 2007).

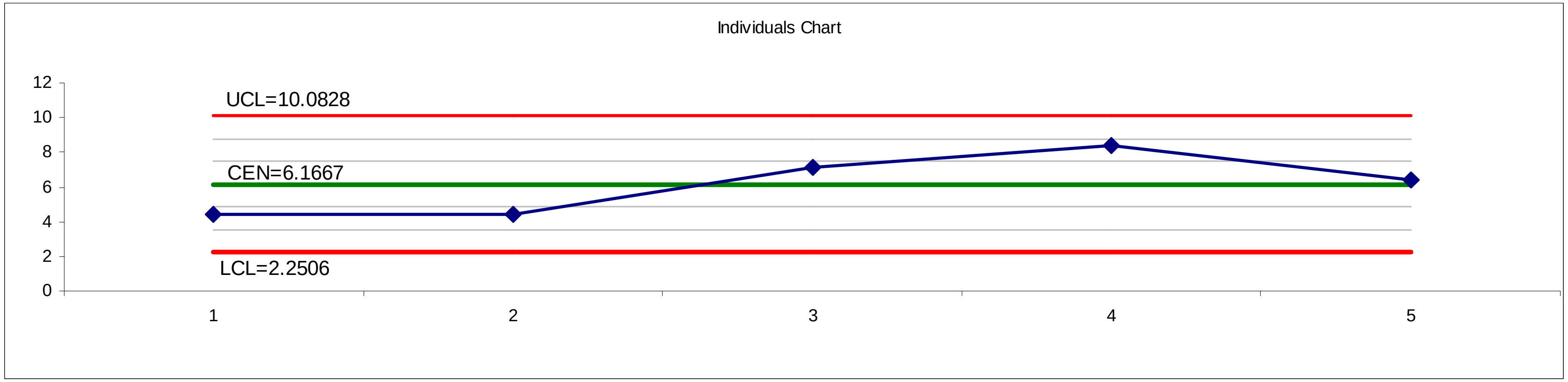
Spearman Rank correlations were completed using STATISTICA 7.0 (Statsoft Inc.), with results considered significant when $p<0.05$. Habitat patch size and distance to the nearest road were included as metrics in the analysis as research has shown that they both tend to negatively affect vegetation communities (Findlay and Houlahan1997; Hooftman and Diemer 2002). A radius of 2 km was chosen for the scope of this urban composition parameter because Findlay and Houlahan (1997) found that landscape effects on wetland taxa richness were strongest when considering composition at this distance. The mean of each vegetation parameter over four years of monitoring was calculated for each site so that correlations could be made between vegetation parameters and landscape metrics. This was necessary because landscape data were not available in a year-by-year form but rather as a single “snapshot” representation of land parameters during the general period of monitoring.

In addition, relationships between the most predominant non-native species with a weediness score of -3 and landscape metrics were also analyzed with logistic regression using STATISTICA 7.0 (Statsoft Inc.). The presence of Purple Loosestrife, Common Buckthorn and Garlic Mustard at each site was tested against habitat patch size, distance to the nearest road, percent natural cover and percent urban cover using simple logistic regression with one continuous predictor. The chosen species were selected according to the same criteria used in the Cochrane-Armitage analyses. Though still considered a parametric option, logistic regression is a non-linear estimation method in which the binary response variable is assumed to follow a binomial distribution (Quinn and Keough 2002). The assumptions of normality and homoscedacticity do not apply as with linear regression. For logistic regression tests between site occupancy (presence/absence) over the five-year survey period and landscape parameters, a site was considered occupied if the species in question was detected in at least two of the four survey years.

2.2.2.4 Statistical Process Control: Statistical Process Control (SPC) is an application which uses time series data to develop thresholds (Maurer et al. 1999). Though similar to regression, SPC is not based on hypothesis testing. Instead, this application seeks to identify instances when a time series exhibits non-random behaviour. A series demonstrating this non-random behaviour is considered "out of control" and, as such, unsuitable for developing a monitoring baseline. If the time series exhibits natural random variability around a reference point (usually the mean), the series is considered "in control". Data that are in control can be treated as a baseline from which monitoring thresholds can be generated. A recommended minimum of five years of "in control" data have been used to set monitoring thresholds (Paul Zorn, pers. comm.).

SPC control charts are divided into six zones based on the mean value of a time series and its standard deviation adjusted to sample size (Fig. 4). The upper and lower critical limits, defined as +/- 3 standard deviations (SD) from the mean, encompass the "in control" range of the time series. Control charts for stable systems will not contain any points outside the +/- 3 SD "in control" range. In such a case, the series represents appropriate reference conditions from which the +/- 3 SD critical limit thresholds can be adopted. Series with points outside the "in control" range may represent an ecosystem under stress or moving slowly toward some alternate state. A data set in this type of flux would not provide appropriate reference conditions on which thresholds could be based.

For this study Individual Moving Average and Moving Range Charts were examined to determine if each vegetation parameter was in control (Paul Zorn, pers. comm.). Values that exceed the critical limits and are associated with significant increasing or decreasing trends should always be investigated thoroughly. Often in monitoring, early detection of changing trends or a significant decline in parameters is desired in order to recognise problems before the effects become irreversible. To address these early detection concerns, the upper and lower warning limits on control charts are represented by +/- 2 SD from the mean. Trends of series that exceed the warning limits but not the critical limits should be tracked closely for further changes. A time series with values falling within the warning limits is exhibiting natural variability and is considered to be stable and not of concern based on SPC alone. All statistical control charts for vegetation data were generated using SPC XL (SigmaZone), a third-party add-on application to Microsoft Excel.



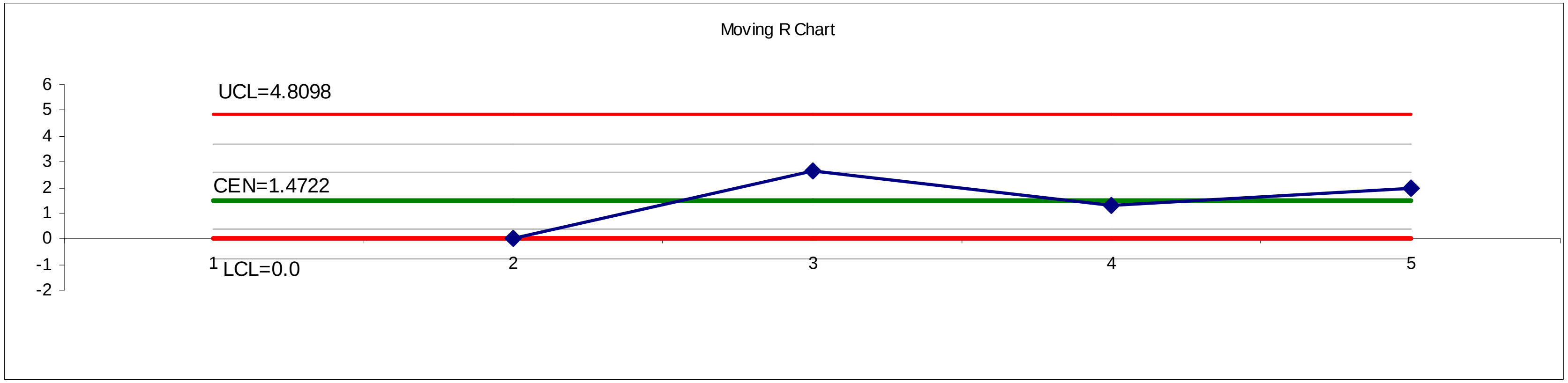


Figure 4. Statistical Process Control charts for weedy species richness. Both the individual moving average chart (showing the parameter values for each year) and the Moving Range chart (showing the amount of change in the parameter between years) are “in control” from 2005 to 2009. On the Individuals Chart, the green line indicates the time series mean and the red lines show the upper and lower critical limits (+/- 3SD). Since the data set is in control, the mean from 2005-2009 can be used as a baseline for future monitoring.

2.2.2.5 Power Analysis: Power analyses were conducted in PASS (NCSS Inc.) to determine the number of monitoring sites required in order to detect significant levels of change in various vegetation parameters. Power analysis also assessed the ability of the current monitoring program to detect statistical differences in wetland vegetation parameters among physiographic zones and among years, given the number of established monitoring sites. Power analysis uses baseline monitoring data to determine if additional monitoring sites are required to detect significant trends. Effect size refers to the detectable change in the time series over a given interval, and is often related to a specific monitoring threshold. Based on SPC, which also uses baseline monitoring data to generate monitoring thresholds, changes in a parameter of ± 1 SD represent natural variability in stable systems and are not considered to be of concern. Changes in a parameter of ± 2 SD are considered outside of normal variability expected in stable communities and represent early warning signs. Changes in a parameter of ± 3 SD represent instability and situations in which causal factors must be investigated. The effect sizes considered in the power analyses have therefore been set at 1, 2 and 3 SD to represent ecologically relevant statistical differences in wetland vegetation communities. Currently, five years of in control data will be used to determine the five year effect size.

In addition to the calculation of required *n*, the minimum effect size that can currently be measured under the monitoring program was determined for each analysis. These results were used to determine the quality of current monitoring data. Data quality was assessed according to Table 4, developed by Dobbie et al. (2006) and CVC. Although these rules are subjective, they are not arbitrary and are based on emerging common practice among conservation agencies, personal experience, and logistical considerations (Dobbie et al. 2006). For linear regression, the upper effect size limit (data are red) is set to 40% because changes above this point are too coarse to provide early warning of decline in a parameter. A change of nearly 50% or greater would be needed before changes were detected by the monitoring program (Dobbie et al. 2006). The lower effect size limit (data are green) is set to 20% because a smaller effect size is too fine and would likely result in an unaffordable monitoring program. An effect size of less than 20% may be reasonable for some parameters, but this is likely not the case for all monitored parameters. And although smaller effect sizes may allow for the detection of significant differences more often, without knowledge regarding natural variability in the parameter it may be difficult to determine if these changes are meaningful (Dobbie et al. 2006). It is important to note that the data quality rules developed by Dobbie et al. (2006) were set according to power analysis of a paired *t*-test. This test does not reflect changes in trends but changes in status and usually requires a smaller sample size to achieve the same power and confidence as a regression test (Dobbie et al. 2006). Therefore, data quality assessment is appropriate, if not more stringent than necessary, for analyses completed in this report.

For the Cochrane-Armitage test, percent change effect sizes could not be calculated; therefore, SD was instead used to determine data quality (Table 4). The upper effect size limit (data are red) was set to 3 SD because that is the critical warning limit in SPC. Therefore, data in this quality range are not capable of detecting the most important ecological changes. The lower effect size limit (data are green) is set to <2 SD because that is the early warning limit in statistical process control. Data in this quality range can therefore detect both early and critical warming limits and should be able to provide managers with necessary information for making informed management decisions. It should be noted that data quality will naturally be better for measures that display more variance over the study period. This is because the critical and warning thresholds are themselves based on the standard deviation of baseline data. For example, it is easier (e.g. requires a smaller sample size) to detect a 2 SD change in mortality when the underlying variability of the mortality data is high. If variability was low, 2 SD would represent a very small change in proportion. Smaller changes are innately more difficult to detect than larger changes over the same time period.

Currently, no type of effect size could be calculated for logistic regression. For that reason, two data quality groupings were developed for this analysis. Data were considered green if enough site-years were available to detect a significant logistic relationship between parameters. Data were considered red if not enough site-years were available to detect a significant logistic relationship between parameters. No yellow data category was used.

Confidence and power levels are also subjectively, but not arbitrarily, selected. Although research standards typically set confidence to 95% and power to 80%, this may not be appropriate for ecological monitoring (Dobbie et al. 2006). Confidence refers to the probability of not committing a Type I statistical error. In ecological monitoring, this may be considered a “false alarm”. Power refers to the probability of not making a Type II error, which may be considered a “missed signal”. In ecological monitoring, missing a signal has more severe management implications than raising a false alarm. For that reason, power is usually set higher than confidence in this type of monitoring (Dobbie et al. 2006).

Table 4. Rule set for assessing data quality in monitoring measures (adapted from Dobbie et al. 2006).

Data Quality	**Effect size**			**Confidence**	**Power**
	Linear Regression	**Cochrane-Armitage**	**Logistic Regression**		
Green	<20% change	<2 SD	Currently have enough site to detect a significant relationship	80%	90%
Yellow	>20% <40% change	2-3 SD	N/A	80%	90%
Red	>40% change	>3 SD	Currently do not have enough site to detect a significant relationship	80%	90%

3.0 RESULTS AND DISCUSSION

3.1 DESCRIPTIVE ANALYSIS

3.1.1 Most Common Species

Monitoring Question: What vegetation species were most commonly encountered in the Credit River Watershed?

- The most common species in the watershed were: Red Osier Dogwood, Climbing Nightshade, Marsh Bedstraw, Spotted Touch-me-not, Northern Bugleweed, Brown-seed Dandelion, Fowl Manna-grass, Riverbank Grape, Canada Blue Joint and Wild Mint.

The ten most common combined wetland vegetation species across all years are presented in Table 5 in descending order. The most common wetland species span a range of wetness values from facultative upland to obligate wetland species (+3 to -5). They also occupy a range of Conservation Coefficients, between 0 and 5, indicating that they tolerate major to mild disturbance. Of the top ten species, eight are native to Ontario, whereas Climbing Nightshade (*Solanum dulcamara*) and Brown-seed Dandelion (*Taraxacum officinale*) are non-native.

Two of the most common species, Canada Blue Joint (*Calamagrostis canadensis*) and Marsh Bedstraw (*Galium palustre*), are Species of Urban Interest as designated by Credit Valley Conservation (CVC). This designation is specific to CVC, which has ranked species inhabiting the watershed according to their sensitivity to environmental change and disturbance. Species in the watershed may be ranked as Species of Conservation Concern (SoCC; Tier 1), Species of Interest (SoI; Tier 2), Species of Urban Interest (SoI; Tier 3), Secure (Tier 4) or Non-native (Tier 5). Species of Conservation Concern are designated at risk either federally and/or provincially, or are provincially rare according to OMNR’s Natural Heritage Information Centre. According to a draft version of the ranking, a total of 87 species in the Credit River Watershed have been designated SoCC to date. Species of Interest are either uncommon in the watershed and/or have exhibited a significant population decline in the past 25 years. Finally, Species of Urban Interest are exhibiting some form of decline within urban areas due to some sensitivity, but are thought to be relatively secure in more natural environments.

The five most common non-native wetland vegetation species across all years are presented in Table 6 in descending order. In 2009, CVC ranked priority invasive vegetation species into five categories. Species in Category 1, or Transformer species, have the ability to exclude all other species, dominate sites indefinitely and provide a threat to natural areas. Category 2 consists of Highly Invasive Species that tend to dominate certain niches or that are problematic, but do not spread rapidly from major concentrations. These species can persist in dense populations for long periods. Category 3 consists of Moderately Invasive species that can become locally dominant under specific conditions. Category 4 species are considered Minimally Invasive and do not pose an immediate threat to natural areas, but compete with native species. Finally, Category 5 species are Potentially Invasive and are known to reproduce aggressively on occasion, but have not yet posed a serious threat to natural areas in Ontario (Credit Valley Conservation 2009).

Of the most common non-native species recorded in the surveys, Purple Loosestrife, Common Buckthorn, Manitoba Maple (*Acer negundo*) and Garlic Mustard are all considered Transformer species. Additionally, all but Manitoba Maple have a weediness score of -3 according to the Floristic Quality Assessment System for Southern Ontario (Oldham et al. 1995). There is conflicting evidence as to whether or not Purple Loosestrife has the ability to decrease native species richness in wetlands (Hager and McCoy 1998; Farnsworth and Ellis 2001). However, it has been demonstrated that Purple Loosestrife alters nutrient cycling and decomposition rates, posing a major threat to wetland ecosystems (Blossey et al. 2001). Common Buckthorn has many physiological traits which confer advantages over native species. It is shade tolerant, has rapid growth rates and higher photosynthetic rates when compared with many native species (Knight et al. 2007). It also has a large range of habitat tolerances, grows well in drought and moisture rich environments, has high fecundity and high germination rates and has bird dispersal of fruit which allows the shrub to spread rapidly and successfully (Knight et al. 2007). Garlic Mustard is a biennial forb that is shade tolerant and capable of thriving in the understory. Community disturbance is also not required for Garlic Mustard to become established at a site (Rodgers et al. 2008). In addition, Garlic Mustard produces a variety of secondary compounds that make it unpalatable to many herbivores, affect seed germination and growth of surrounding native plants, and alter the activity of soil biota (Rodgers et al. 2008). Manitoba Maple is an aggressively opportunistic tree species which thrives in alluvial soils. It is most commonly found in riparian areas, floodplains and anthropogenic waste areas and tends to shade out herbaceous plants, especially in wetlands (Wisconsin Department of Natural Resources 2004).

Climbing Nightshade is considered a Moderately Invasive species by CVC, as it is known to invade forests and wetlands. Brown-seed Dandelion has not been ranked by CVC, but is considered a -2 species by the Floristic Quality Assessment System. True Forget-me-not (*Myosotis scorpioides*), a garden escapee, is considered a Minimally Invasive species by CVC because it does not pose a threat to natural areas and mainly dominates shaded seepage areas.

In general, the most common ground vegetation species were the same as the most common combined vegetation species, with the exception that in ground vegetation Red Osier Dogwood was the tenth most common species in conjunction with Reed Canary Grass (*Phalaris arundinacea*) (Table 7). The five most common non-native wetland ground vegetation species are also similar to those for combined vegetation, with the exception that Common Buckthorn and Manitoba Maple were absent.

The five most common wetland regenerating species across all years are presented in Table 8 in descending order. Common Buckthorn is the only non-native species appearing on the list. White Elm (*Ulmus americana*) was the only tree species present on the list, while the remaining four species were shrubs. Wetness values ranged from -3 to +3 and CC values ranged from 0 to 3.

Table 5. The ten most common combined vegetation species across all years in the Credit River Watershed.

Common Name	Scientific Name	Native	Wetness Index	SCC Tier	CC Value	Mean # of Sites per Year
Red Osier Dogwood	*Cornus stolonifera*	Native	-3	4	2	16
Climbing Nightshade	*Solanum dulcamara*	Non- Native	0	5	N/A	13.6
Marsh Bedstraw	*Galium palustre*	Native	-5	3	5	12.6
Spotted Touch-me-not	*Impatiens capensis*	Native	-3	4	4	12
Northern Bugleweed	*Lycopus unifloris*	Native	-5	4	5	11.2
Brown-seed Dandelion	*Taraxacum officinale*	Non- Native	3	5	N/A	10.8
Fowl Manna-grass	*Glyceria striata*	Native	-5	4	3	10.6
Riverbank Grape	*Vitis riparia*	Native	-2	4	0	10.6
Canada Blue Joint	*Calamagrostis canadensis*	Native	-5	3	4	9.8
Wild Mint	*Mentha arvensis*	Native	-3	4	3	9.8

Table 6. The five most common non-native combined vegetation species across all years in the Credit River Watershed.

Common Name	Scientific Name	CVC Invasive Priority Category	Wetness Index	Weediness Score	Mean # of Sites per Year
Climbing Nightshade	*Solanum dulcamara*	3	0	-2	13.6
Brown-seed Dandelion	*Taraxacum officinale*	N/A	3	-2	10.8
Common Buckthorn	*Rhamnus cathartica*	1	0	-3	8.2
Purple Loosestrife	*Lythrum salicaria*	1	-5	-3	6.8
Manitoba Maple	*Acer negundo*	1	-2	N/A	5.4
Garlic Mustard	*Alliaria petiolata*	1	0	-3	5.4
True Forget-me-not	*Myosotis scorpioides*	4	-5	-1	5.4

Table 7. The ten most common ground vegetation species across all years in the Credit River Watershed.

Common Name	Scientific Name	Native	Wetness Index	SCC Tier	CC Value	Mean # of Sites per Year
Climbing Nightshade	*Solanum dulcamara*	Non- Native	0	5	N/A	13.6
Marsh Bedstraw	*Galium palustre*	Native	-5	3	5	12.6
Spotted Touch-me-not	*Impatiens capensis*	Native	-3	4	4	12
Northern Bugleweed	*Lycopus unifloris*	Native	-5	4	5	11.2
Brown-seed Dandelion	*Taraxacum officinale*	Non- Native	3	5	N/A	10.8
Fowl Manna-grass	*Glyceria striata*	Native	-5	4	3	10.6
Riverbank Grape	*Vitis riparia*	Native	-2	4	0	10.6
Canada Blue Joint	*Calamagrostis canadensis*	Native	-5	3	4	9.8
Wild Mint	*Mentha arvensis*	Native	-3	4	3	9.8
Reed Canary Grass	*Phalaris arundinacea*	Native	-4	4	0	9.4
Red Osier Dogwood	*Cornus stolonifera*	Native	-3	4	2	9.4

Table 8. The five most common regenerating species across all years in the Credit River Watershed.

Common Name	Scientific Name	Native	Wetness Index	SCC Tier	CC Value	Mean # of Sites per Year
Red Osier Dogwood	*Cornus stolonifera*	Native	-3	4	2	15.8
Choke Cherry	*Prunus virginiana ssp. virginiana*	Native	1	4	2	8
Common Buckthorn	*Rhamnus cathartica*	Non-Native	3	5	N/A	7.2
White Elm	*Ulmus americana*	Native	-2	4	3	6.4
Common Red Raspberry	*Rubus idaeus*	Native	-2	N/A	0	6

3.1.2 Species Richness and Proportion of Natives

Monitoring Questions: How many vegetation species were found in the Credit River Watershed? What proportion of wetland vegetation species was native to the Credit River Watershed?

- Three hundred and seventeen species were detected.
- Seventy-eight percent of these species were native to Ontario.

Over the five year monitoring period 317 woody and herbaceous vegetation species were identified to species level, representing 26% of the 1205 vascular vegetation species known to occur in the Credit River Watershed (Credit Valley Conservation 2002). Seventy-eight percent of the species detected were native to Ontario. Of the species identified, 16% were shrubs, 8% were trees and the remaining 76% were herbaceous plant types (ferns, grasses, forbs, etc.) (Fig. 5). Combined vegetation species richness was highest at Warwick and Caledon Lake Wetlands. Warwick is located within the Middle watershed, and Caledon Lake is located within the Upper Watershed, both on CVC property. The sites with the lowest richness were Meadowvale Wetland and Ken Whillans Wetland, both of which are also on CVC property in the Lower and Middle watershed, respectively.

The proportion of native combined wetland vegetation species, watershed-wide, ranged from 79.8% to 84.6% per year. Between 2005 and 2009, the number of native species increased from 186 to 220 species (18.3%), while the number of non-native species increased from 41 to 49 (19.5%). It therefore appears that the rate of increase for non-native species is similar to the rate of increase of native species. Sites with the greatest proportion of native species were Caledon Lake Wetland and Acton Wetland, within the Upper and Middle watersheds, respectively. Sites with the lowest proportion of native species were Rattray Marsh and Meadowvale Wetlands, both found within the Lower watershed. The proportion of native species detected in wetlands was higher than the proportion of native species found in the watershed overall, as it is estimated that approximately 69% of the flora in the Credit River Watershed is native (Credit Valley Conservation 2009).

From 2005 to 2009, 305 plants identified to species were observed in the ground vegetation layer. Approximately 78% of these species were native to Ontario, while the remaining 21% were non-native. The number of species observed each year ranged from 222 to 298, with natives accounting for 78%-85% of these species (Fig. 6).

Over the five year monitoring period 66 regeneration species were identified to species level, approximately 90% of which were native to Ontario. The proportion of native regeneration species watershed-wide ranged from 82.2% to 87.1% across the monitoring period. Only seven non-native regeneration species were detected: Manitoba Maple, Norway Maple (*Acer platanoides*), Tartarian Honeysuckle (*Lonicera tatarica*), Common Buckthorn, Garden Red Currant (*Ribes rubrum*), Purpleosier Willow (*Salix purpurea*) and European Highbush Cranberry (*Viburnum opulus*).

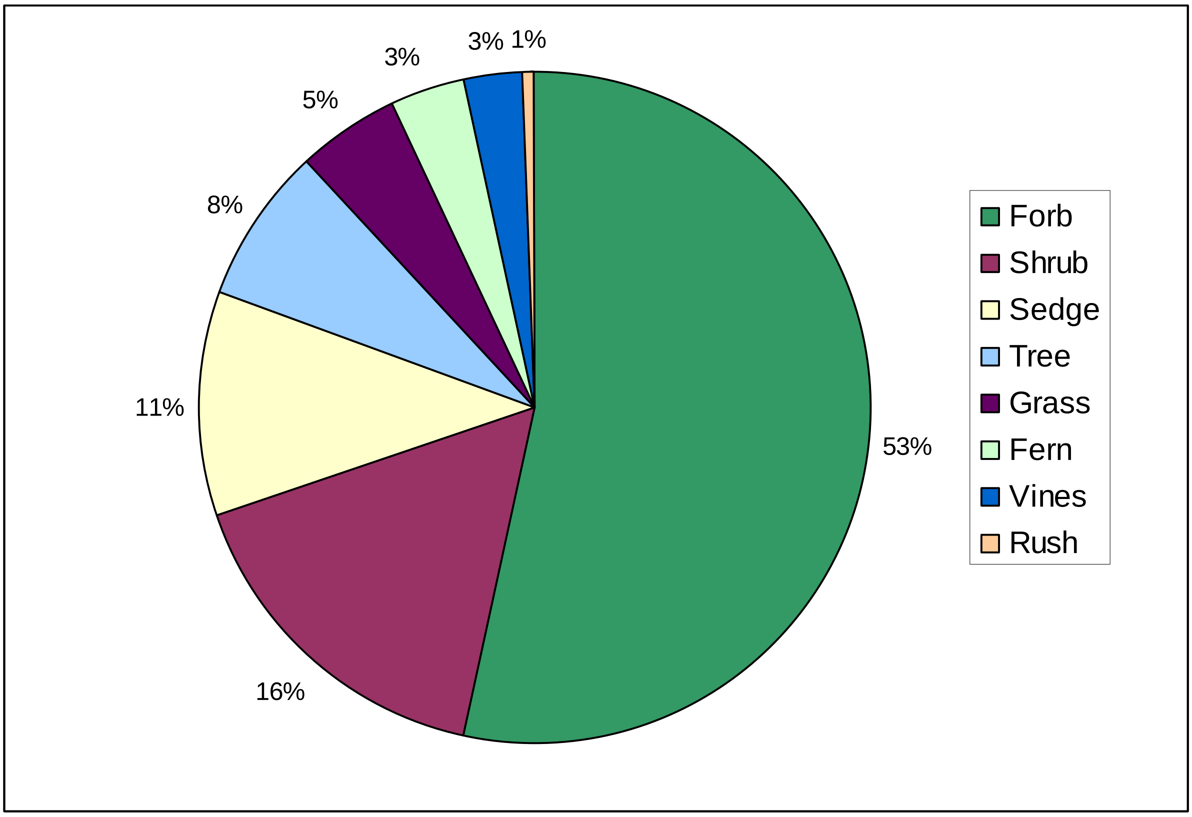


Figure 5. Species richness by plant type detected between 2005 and 2009 in the Credit River Watershed.

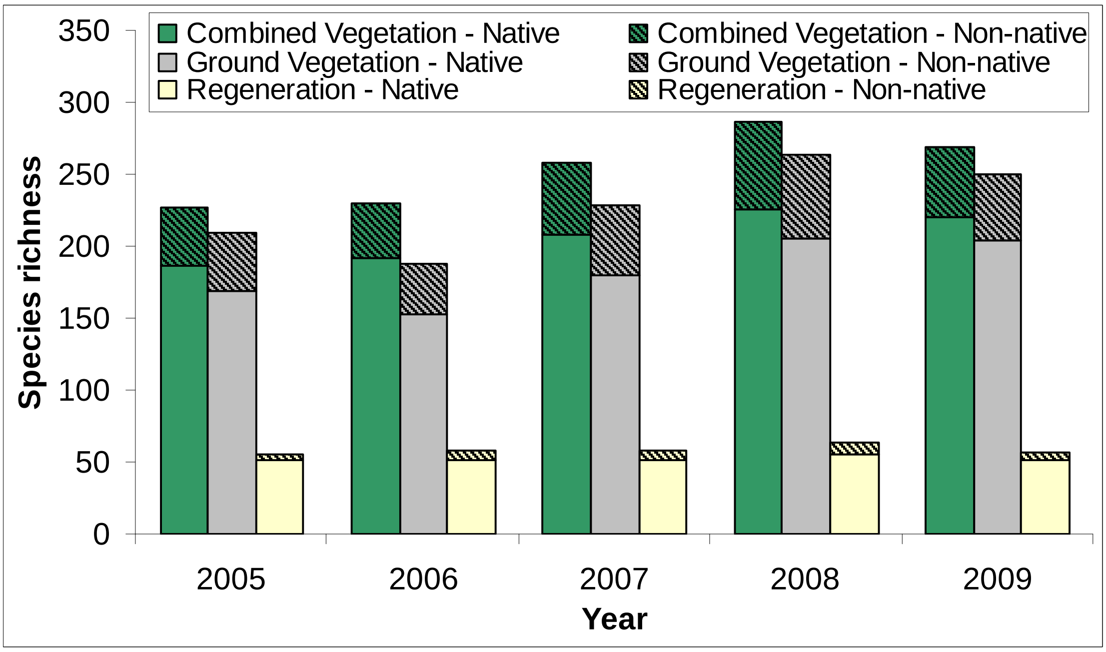


Figure 6. Total species richness throughout the Credit River Watershed over the monitoring period for combined vegetation, ground vegetation and regeneration.

3.1.3 Species Diversity

Monitoring Question: What was the species diversity for ground vegetation and regeneration within the Credit River Watershed?

- Mean total ground vegetation diversity over the monitoring period ranged from 1.32 to 1.83.
- Mean native ground vegetation diversity over the monitoring period ranged from 0.97 to 1.52.
- Mean regeneration diversity over the monitoring period ranged from 0.35 to 0.45.

Because diversity calculations were dependant on cover class estimations and subplot sizes differed between ground vegetation and regeneration, diversity was calculated separately for ground vegetation and regenerating species. However, diversity was not calculated for regeneration in 2009 because of changes in the regeneration sampling protocol which rendered the 2009 incomparable to previous years data.

Over the five year monitoring period, diversity averaged 1.53 and 1.26 per site over the entire watershed for total and native ground vegetation, respectively (Fig. 7). The sites which had the greatest total and native ground vegetation diversity were Grange Orpen Wetland (Total, H' = 2.36; Native, H' = 2.22) and Caledon Lake Wetland (Total, H' = 2.20; Native, H' = 2.11), both located within the Upper Watershed. The sites with the lowest total ground vegetation diversity were Meadowvale Wetland (Total, H' = 0.85; Native, H' = 0.48) and Creditview Wetland (Total, H' = 1.05; Native, H' = 0.61), both located within the Lower watershed.

The mean diversity score per site for regenerating species ranged from 0.35 to 0.45 across the monitoring period (Fig. 7). The sites which had the greatest regeneration diversity were Caledon Lake Wetland (H' = 1.06) and Grange Orpen Wetland (H' = 0.77), both located within the Upper Watershed. The sites with the lowest regeneration diversity were Rattray Marsh (H' = 0.05) and Meadowvale Wetlands (H' = 0.08), both located within the Lower watershed.

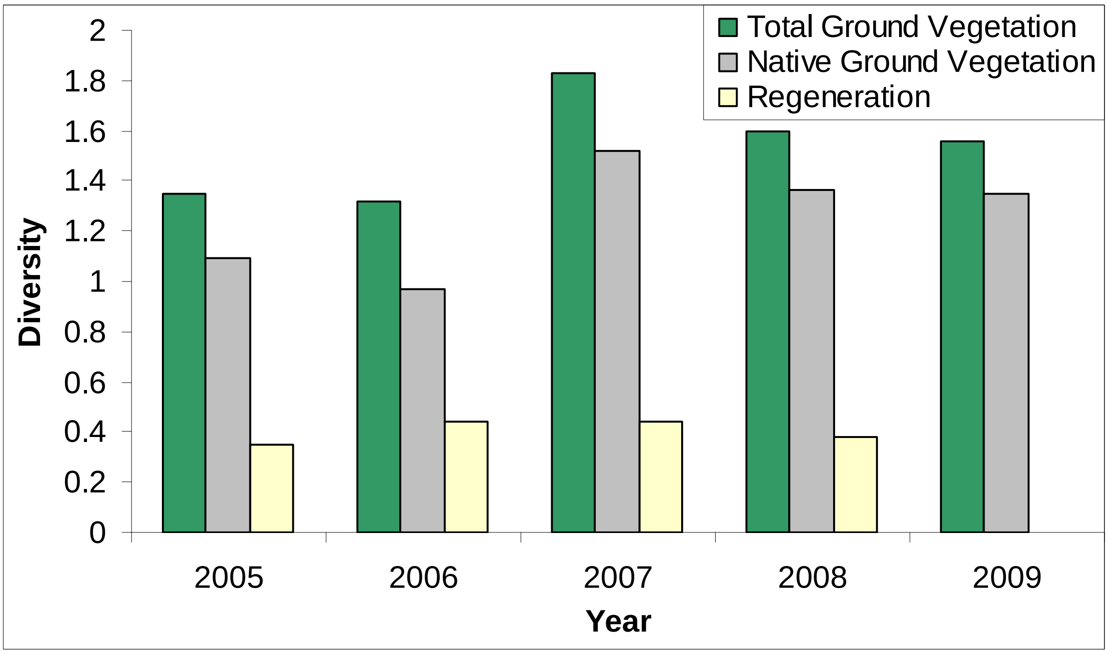


Figure 7. Mean diversity per site throughout the Credit River Watershed over the monitoring period for total and native ground vegetation and regeneration.

3.1.4 Species Evenness

Monitoring Question: What was the species evenness for ground vegetation and regeneration within the Credit River Watershed?

- Mean total ground vegetation evenness over the monitoring period ranged from 0.41 to 0.59.
- Mean native ground vegetation evenness over the monitoring period ranged from 0.48 to 0.66.
- Mean regeneration evenness over the monitoring period ranged from 0.75 to 0.83.

Similar to diversity, evenness calculations were dependant on cover class estimations; therefore evenness was calculated separately for ground vegetation and regenerating species and was not calculated for regeneration in 2009. Over the five year monitoring period, evenness averaged 0.50 and 0.57 per site over the entire watershed for total and native ground vegetation, respectively (Fig. 8). The sites which had the greatest total ground vegetation evenness were Acton Wetland (E = 0.66) and Caledon Lake Wetland (E= 0.60). The sites which had the greatest native ground vegetation evenness were Meadowvale Wetland (E = 0.73) and Terra Cotta Wetland (E= 0.69). The sites with the lowest total and native ground vegetation evenness were Erin Pine Estates Wetland (Total, E= 0.38; Native, E= 0.38) and Warwick Wetland (Total, E= 0.40; Native, E= 0.43). The mean evenness score per site for regeneration species ranged from 0.75 to 0.83 across the monitoring period (Fig. 8). The sites which had the greatest regeneration diversity were Meadowvale Wetland (E = 1.00) and Hungry Hollow Wetland (E = 0.92).

The sites with the lowest regeneration diversity were Caledon Lake (E = 0.45) and Belfountain Wetlands (E= 0.67).

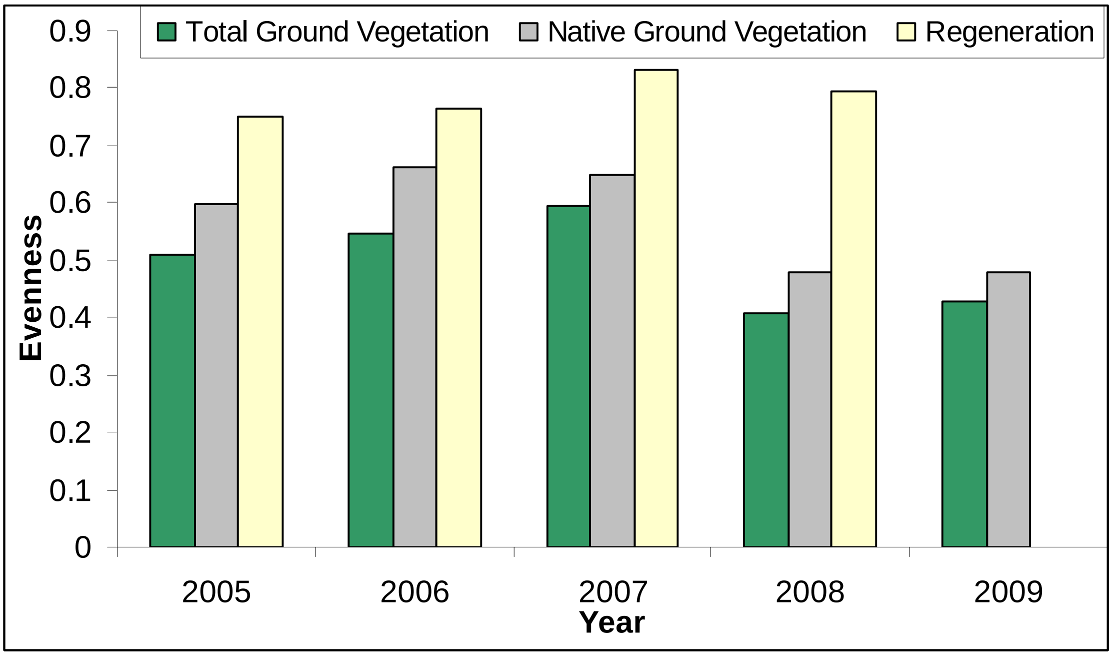


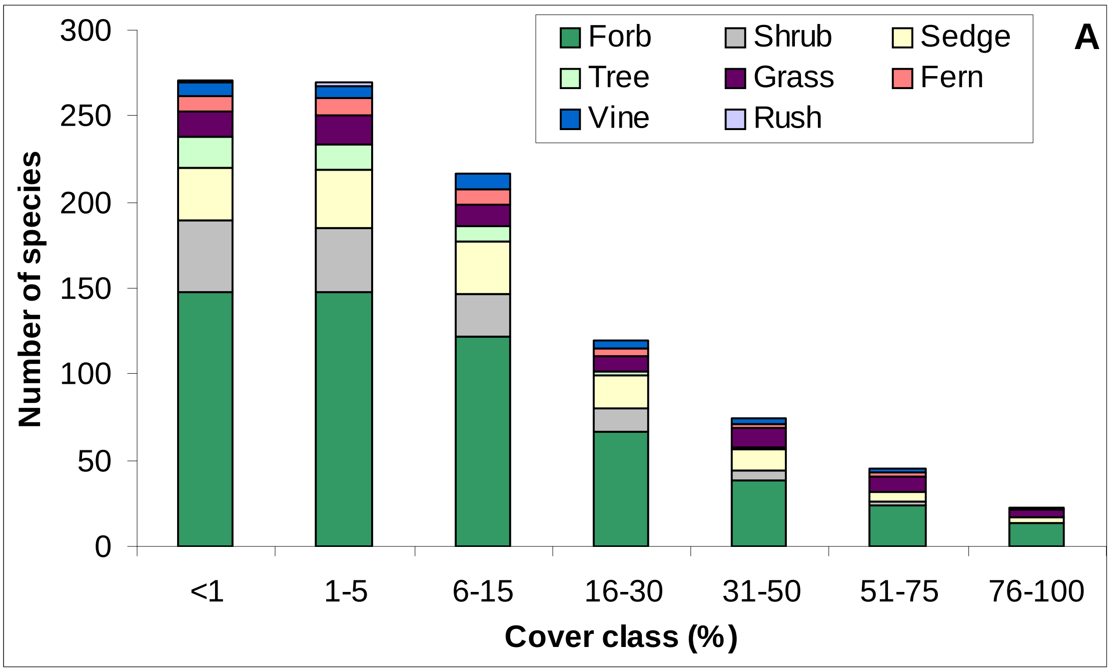
Figure 8. Mean evenness per site throughout the Credit River Watershed over the monitoring period for total and native ground vegetation and regeneration.

3.1.5 Cover Classes

Monitoring Questions: How many species were detected in each cover class? What plant types dominated the different cover classes?

- For ground vegetation, the number of species in each cover class ranged from 271 in the <1% class to 22 in the 76-100% class.
- For regeneration, the number of species in each cover class ranged from 73 in the 1-5% class to 13 in the 76-100% class.
- Ground vegetation was dominated by forbs, while regeneration was dominated by shrubs.

The number of species detected in each cover class ranged from 22 to 271 for ground vegetation and 13 to 76 for regeneration. Species richness tended to decrease from lower classes to higher classes (Fig. 9A & B). In other words, wetland ground vegetation species are more likely to occupy a low proportion of cover within a given area. Forbs were the most dominant plant type in all cover classes in the ground vegetation, making up approximately 55% of the species (Fig. 9A). Rushes were the least represented plant type in all cover classes. For regeneration, shrubs were more dominant than trees in all cover classes (Fig. 9B). This is reflective of monitoring methodology as monitoring plots were not set up in treed locations. Therefore, this does not depict future regeneration trends.



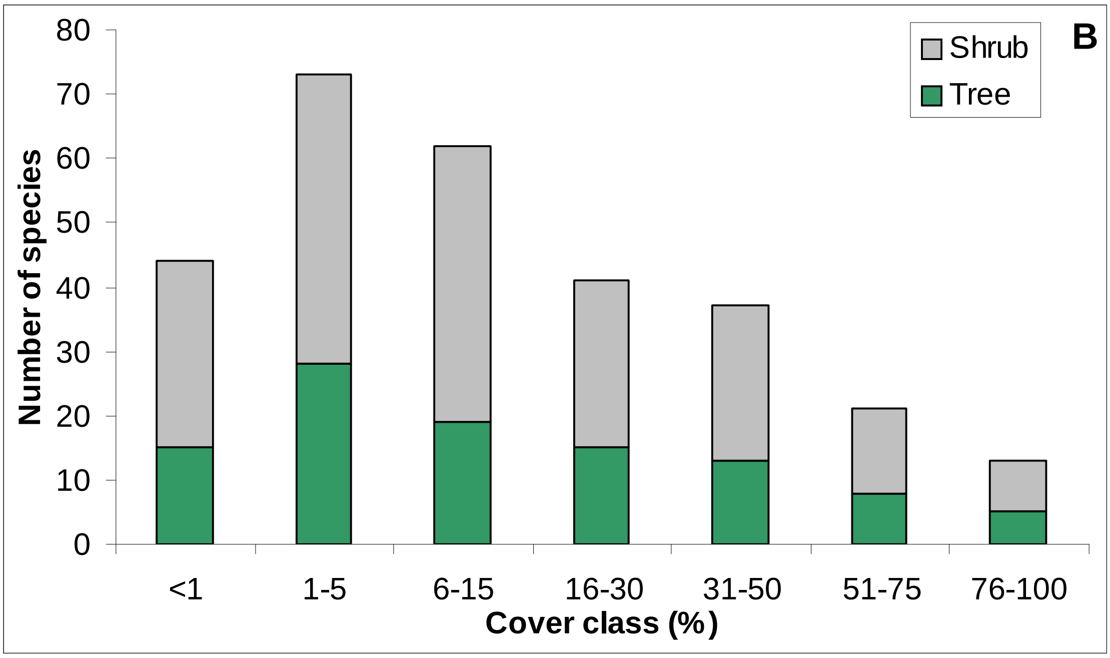


Figure 9. Number of species in each cover class by plant type, for ground vegetation (A) and regeneration (B).

3.1.6 Coefficient of Conservatism

Monitoring Questions: What were the most common coefficient of conservatism (CC) values in the watershed? How many species with a high conservatism score (CC 8-10) were present in the watershed? What was the mean floristic quality index (FQI) value in the watershed?

- The majority of detected species had CC values of between 4 and 6.
- Fifteen species with high CC values were detected.
- mCC scores ranged from 3.91 to 4.08.
- Mean FQI scores ranged from 22.0 to 25.5.

Native wetland plants detected in the Credit River Watershed varied in CC value across the entire range of values, from 0 to 10 (Fig. 10). Species with a score of 0 to 3 are indicative of wide-ranging taxa that are usually tolerant of disturbed sites. Taxa which tolerate mild disturbance have CC rankings of 4 to 6, and those which are found in communities in an advanced successional stage have rankings of 7 to 8. Finally, taxa which tend to be found in pristine communities are assigned a ranking of 9 to 10 (Oldham et al. 1995).

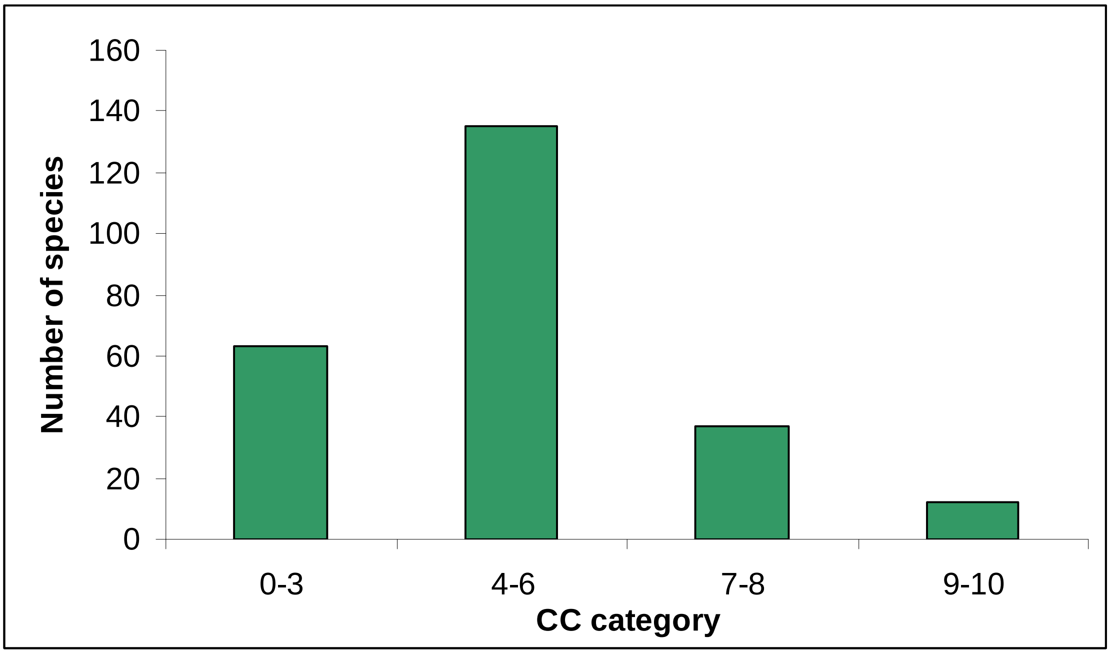


Figure 10. Number of species within each Coefficient of Conservatism (CC) category for plants detected during the monitoring period in the Credit River Watershed.

The CC values for taxa detected across the watershed were used to determine the mean Coefficient of Conservatism for each site (mCC), as well as the Floristic Quality Index of each site (FQI). The mean mCC for sites in each year ranged from 3.91 to 4.08, while the mean FQI for sites in each year ranged from 22.0 to 25.5. Since these two parameters both rely on CC values for their calculation, they were highly correlated with each other among wetland plots (ρ=0.637, p=0.005). This is consistent with other studies using mCC and FQI, which have shown that these two parameters are highly correlated in natural areas (Bourdaghs et al. 2006; Bowers and Boutin 2008).

Caledon Lake Wetland had the highest mCC (5.76) and FQI (46.7) scores of all sites, which is not surprising as it contains a number of specialized vegetation species which are characteristic of a bog/fen habitat. The sites that contained the lowest mCC scores were Credit River (3.48) and Creditview Wetlands (3.18), while Creditview (15.12) and Meadowvale Wetlands (14.83) had the lowest FQI scores. In general, mCC and FQI ranked sites in a similar fashion in terms of their floristic quality.

Fifteen habitat specialists (CC 8-10) (OMNR 2008), were detected throughout the monitoring period (Appendix C). Each of these species was detected at only one to three sites. Thirteen of the detected habitat specialists were found at Caledon Lake Wetland.

3.1.7 Rare Species

Monitoring Question: How many locally and regionally rare species were found watershed-wide?

- A total of 56 locally and/or regionally rare species were detected.

A total of 55 locally rare species were observed during the monitoring period. Of these species, 31 were also classified as regionally rare. In addition, one regionally rare species, Late Lowbush Blueberry (*Vaccinium angustifolium*), was observed in the watershed. Twenty-four locally/regionally rare species were observed at Caledon Lake Wetland, the most at any monitored wetland in the watershed. Eighteen of these species were not observed at any other monitoring site, including three different species of both *Vaccinium* and *Salix* and 11 known bog/fen indicator species. Warwick Wetland and Erin Pine Estates Wetland had the next highest number of rare species at 11 and eight species, respectively. Meadowvale and Hillsburgh Wetlands had the fewest rare species, with only one present at each site. Each monitored wetland in the watershed contained an average of 5.6 locally and/or regionally rare species. Only one Species of Conservation Concern was detected at monitoring sites: Field Thistle (*Cirsium discolor*); however, it was only observed at Belfountain Wetland in 2005. Approximately 44% of species are considered of interest, while 50% of species are considered secure or non-native (Table 9).

Table 9. Ranking of wetland vegetation species detected in the Credit River Watershed according to CVC's Species of Conservation Concern.

Species of Conservation Concern Ranking	# of Species
Species of Conservation Concern	1
Species of Interest	75
Species of Urban Interest	64
Secure	96
Non-native	66

3.1.8 Wetness Index

Monitoring Question: What was the mean wetness index per site across the watershed?

- Mean wetness index ranged from -1.87 to -2.03.

The mean wetness score averaged across all sites ranged from -1.87 to -2.03 per year. Vegetation detected at monitoring sites in the Credit River Watershed ranged in wetness values from -5 (Obligate Wetland) through +5 (Obligate Upland). The majority (58%) of vegetation species detected ranged from Facultative Wetland to Facultative Upland Species (-4 to +4) (Fig. 11), indicating that they can persist in a range of wet to dry habitats. Thirty-one percent of species detected were Obligate Wetland Species, indicating that they occur almost exclusively in wetlands under natural conditions. The remaining 11% of species detected were Obligate Upland species, indicating that they almost never occur in wetlands under natural conditions. The inclusion of so many Obligate Upland species in the monitoring data is likely due to the fact that the wetland monitoring sites were established along a wetness gradient, beginning in dry areas along the edge of the wetland. The invasion of non-native species may also be a factor, as 59% of the detected obligate upland species were non-native.

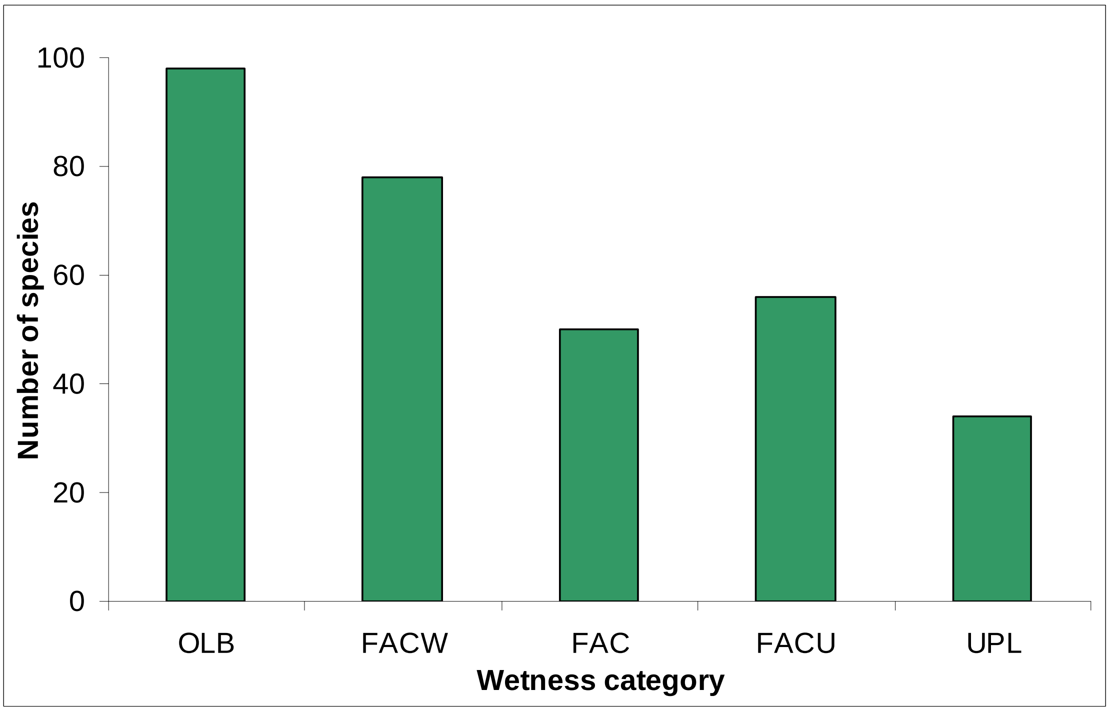


Figure 11. Number of species within each wetness category detected during the monitoring period in the Credit River Watershed. OBL, Obligate Wetland (-5); FACW, Facultative Wetland (-2 to -4); FAC, Facultative (-1 to +1); FACU, Facultative Upland (+2 to +4); UPL, Obligate Upland (+5).

3.1.9 Weediness

Monitoring Questions: What was weedy species richness across the watershed? How many species with a weediness score of -3 were present in the watershed?

- Weedy species richness ranged from 4.44 to 8.39.
- Twelve species had a weediness score of -3.

Sixty-nine non-native plants were detected during the monitoring period (Appendix D), all but five of which have been assigned a Weediness ranking in Ontario by Oldham et al. (1995). Weedy species richness across all sites for each year ranged from 4.44 to 8.39. The sites with the highest weedy species richness across the monitoring period were Credit River and Creditview Wetlands, both located in the Lower watershed. The sites which contained the lowest weedy species richness were Caledon Lake and Speersville Wetlands, both within the Upper watershed.

Forty-one percent of species had a weediness score of -1, suggesting that they have little or no impact on natural areas (Fig. 12). Thirty-five percent of the species detected had a weediness score of -2, indicating that they have the potential to be locally problematic. The remaining 17% of taxa with a score of -3 were considered problematic, or have the potential to become seriously problematic weeds. Nine of the 12 taxa with a weediness score of -3 were detected at three or fewer sites (Table 10).

Detecting problematic non-native species before they are widespread allows for rapid and effective management of invasive species (CFIA 2004). Management efforts may be difficult for species such as Purple Loosestrife, Common Buckthorn and Garlic Mustard as they already appear to be widespread and are found in at least seven sites across the watershed. Species with a more limited distribution, such as Tartarian Honeysuckle, may be easier to control because it has been demonstrated that control is most successful and economical when species are in small populations (Moody and Mack 1988; Welling and Becker 1993; Blossey et al. 2001).

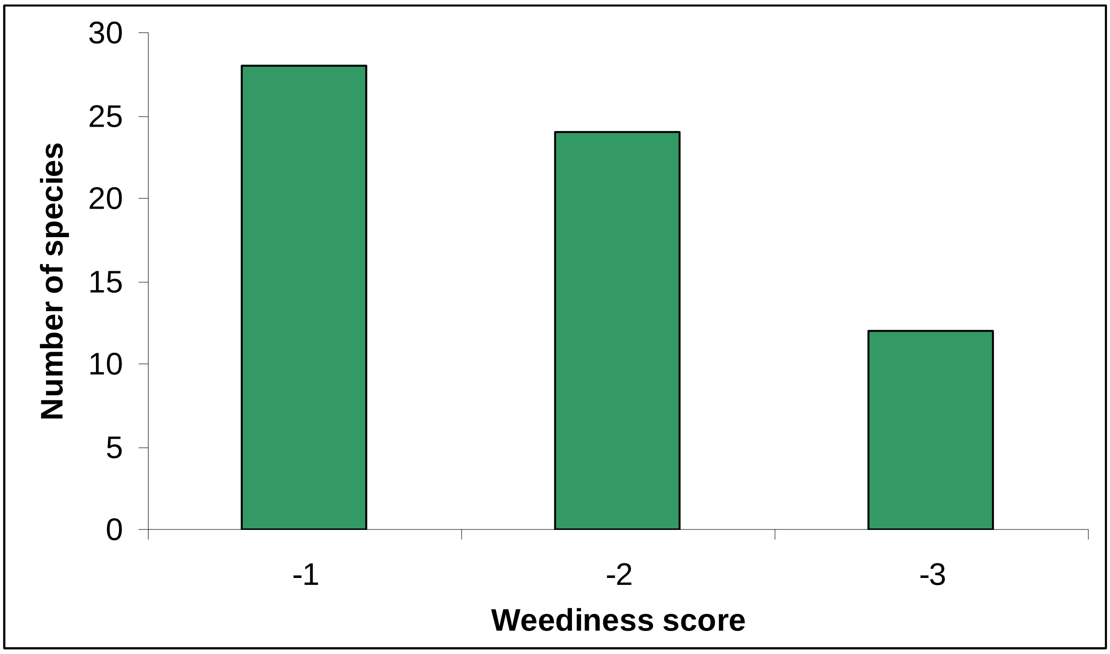


Figure 12. Number of species detected in each weediness category during the monitoring period in the Credit River Watershed.

Table 10. Species detected in the Credit River Watershed between 2005 and 2009 with a weediness score of -3.

Common Name	Latin Name	# Of Sites	Wetness Value
Norway Maple	*Acer platanoides*	1	5
Garlic Mustard	*Alliaria petiolata*	7	0
Greater Celadine	*Chelidonium majus*	1	5
Quackgrass	*Elymus repens*	1	3
Dame’s Rocket	*Hesperis matronalis*	3	5
Common St. John's-wort	*Hypericum perforatum*	3	5
Tartarian Honeysuckle	*Lonicera tatarica*	2	3
Creeping Jennie	*Lysimachia nummularia*	3	-4
Purple Loosestrife	*Lythrum salicaria*	10	-5
White Sweet Clover	*Melilotus albus*	1	3
Curly Pondweed	*Potomogeton crispus*	2	-5
Common Buckthorn	*Rhamnus cathartica*	12	3

3.1.10 Community Composition

Monitoring Questions: Was species composition different among physiographic zones? Did species composition change over the monitoring period?

- Community similarity averaged about 35% between physiographic zones.
- Species turnover at monitoring sites averaged about 17% between years.

Species composition in the Lower zone was 34.1% similar to the Middle zone and 33.6% similar to the Upper zone. Similarity between the Middle and Upper zones was slightly higher at 36.6%. Therefore, although all three zones tended to have relatively dissimilar community composition, the Middle and Upper are more alike than is either to the Lower zone. This is expected due to higher urban cover in the Lower zone. In addition, the Lower zone did not contain any swamps, unlike the Middle and Upper zones. Community similarity values may be more reflective of actual differences with a more balanced sampling design.

Caledon Lake Wetland was the only monitoring site which contained a large number of bog/fen indicator species. The vegetation at Caledon Lake is very different than the other sites, as seen by the large number of rare species present and the large quantity of species which were not observed at other sites. Therefore, Caledon Lake Wetland may not be representative of the majority of wetlands found within the Upper zone. To determine the effect that this site had on community composition, Jaccard's index was recalculated with Caledon Lake removed. This resulted in increased similarity between the Upper and other two zones by 56%. However, removing Caledon Lake from the analyses did not have a significant effect on results; therefore, all results are presented with Caledon Lake included in the analyses.

Similarity in combined community composition between consecutive years ranged from 80.5% to 86.6% (Table 11). Over the entire monitoring period community similarity was 70.5%. Although this indicates that community dissimilarity averaged around 17% between years, the amount of dissimilarity was not increasing between years. Dissimilarity between years can probably largely be attributed to changes in sampling intensity over the monitoring program. However, varying precipitation patterns (Fig. 13) and natural community turnover rate may be responsible for some changes in community composition.

Table 11. Percent community similarity and dissimilarity for combined vegetation watershed wide between years.

Comparison	% Community Similarity	% Community Dissimilarity
2005-2006	80.63	19.37
2006-2007	82.77	17.23
2007-2008	80.46	19.54
2008-2009	86.58	13.42
2005-2009	70.45	29.55

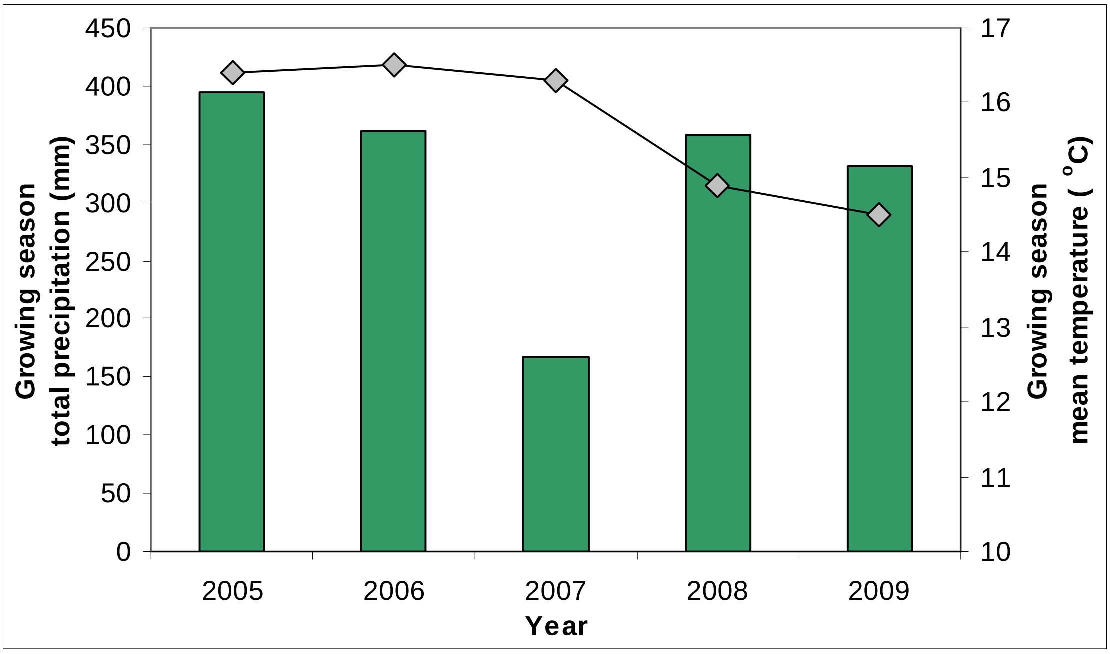


Figure 13. Total monthly precipitation (bars) and mean monthly temperature (points) from 2005 to 2009 in Georgetown, ON during the growing season (April – September). Mean temperature and total precipitation over the growing season were 16.4°C and 395.3 mm, 16.5°C and 360.7mm, 16.3°C and 166.5mm, 14.9°C and 358.0 mm and 14.5°C and 331.1 mm in 2005, 2006, 2007, 2008 and 2009, respectively.

3.1.11 Site Quality

Monitoring Questions: Which sites were the most pristine and most degraded throughout the watershed?

- Caledon Lake, Warwick, Grange Orpen and Erin Pine Estates Wetlands were the most pristine sites in the watershed.
- Creditview, Meadowvale, Winston Churchill and Credit River Wetlands were the most degraded sites in the watershed.

Wetland sites in the Credit River Watershed were ranked based on several wetland vegetation parameters, including: proportion of native species, mCC, number of -3 weedy species and number of rare species (Table 12). Floristic quality index was not included since it is correlated with mCC and would be a redundant parameter. Sites were sorted from the highest to lowest values for each parameter and given a score of between 1 (Highest Quality) and 18 (Lowest Quality). For example, Caledon Lake Wetland had the highest rare species richness, and was given a score of 1 for this parameter. Each site was ranked four times (once for each vegetation parameter) and these scores were summed to give an overall site score. The sites with the lowest score indicated the most pristine habitats. This allowed a coarse determination of which wetland monitoring sites were the most pristine and which sites were most degraded. This is a rough method of ranking the sites, as it assumes that each vegetation parameter contributes equally to the overall quality of the site. The parameters included in the ranking system were chosen because they are broad indicators of wetland health and generally were not affected by sampling quality over the monitoring period. Species richness and diversity were not included in ordering the sites, as it will be affected by wetland community type, but is included in the table for visualization purposes. It is important to note that while this ranking system provides a general overview of site health, the ranking system is subjective.

Caledon Lake, Warwick, Grange Orpen, and Erin Pine Estates were the most pristine wetland sites monitored in the watershed. As stated earlier, Caledon Lake is a very diverse site that contains many bog/fen indicator and rare species and is located in the Upper watershed in a patch of 554.82 ha uninterrupted natural habitat, the second largest habitat patch size of monitored wetlands in the watershed. The remaining three sites are situated in either the Upper or Middle watershed in relatively large habitat patches, are not heavily influenced by urbanization and have relatively highly organic substrates.

Creditview, Meadowvale, Winston Churchill and Credit River Wetlands were the most degraded wetlands monitored in the watershed. All of these sites are located in the Lower watershed and experience heavy pressure from urbanization. It is therefore not surprising that these sites had some of the highest -3 weediness species and lowest percent native species because urbanization is known to increase non-native species in natural areas (McKinney 2006). Creditview and Credit River Wetlands are located in habitat patches of less than 10 ha and are very close to roads and residential areas. Although Winston Churchill and Meadowvale wetlands are situated in relatively large habitat patches, Meadowvale is in very close proximity to a large residential development and was adversely affected by human use and other factors related to urbanization. Similarly, Winston Churchill is situated along the edge of the patch, from which it is largely isolated and connected to only through a riparian corridor. In addition, it is affected by fluctuating water levels, is in very close proximity to a commercial horticultural nursery and may have undesirable placement within the wetland, with the monitoring site starting in a very upland area and ending in open water.

Table 12. Wetland monitoring site quality within the Credit River Watershed. [abc]

Site	Site Name	Physiographic Zone	Wetland Type	% Native	mCC	Count -3	Rare Species Richness	Species Richness	Species Diversity (GV)	
18	Caledon Lake Wetland	Upper	Marsh	0.98	5.76	0.00	17.20	69.40	2.20	
10	Warwick Wetland	Middle	Marsh	0.93	4.28	1.20	5.60	75.60	1.93	**Pristine**
15	Grange Orpen Wetland	Upper	Marsh	0.94	4.34	0.20	3.20	65.00	2.36	
11	Erin Pine Estates Wetland	Upper	Marsh	0.94	4.16	1.40	5.20	68.60	2.17	
7	Acton Wetland	Middle	Marsh	0.94	4.43	0.00	1.20	33.80	1.15	
8	Terra Cotta Wetland	Middle	Swamp	0.87	4.22	0.20	2.40	54.80	1.60	
6	Hungry Hollow Wetland	Middle	Marsh	0.82	3.77	1.20	3.20	48.00	1.74	
20	Belfountain Wetland	Upper	Marsh	0.86	3.91	1.40	2.20	41.00	1.43	
16	Starr Wetland	Upper	Swamp	0.79	3.61	0.20	3.40	55.00	1.70	
19	Melville Wetland	Upper	Swamp	0.78	3.92	1.80	2.60	30.20	1.09	
17	Speersville Wetland	Upper	Swamp	0.93	3.57	0.20	1.20	33.60	1.16	
13	Hillsburgh Wetland	Upper	Marsh	0.84	3.86	1.60	1.20	59.20	1.73	
1	Rattray Marsh Wetland	Lower	Marsh	0.62	4.07	3.00	2.20	34.20	1.47	
9	Ken Whillans Wetland	Middle	Swamp	0.81	3.67	2.20	2.00	28.60	1.29	
2	Credit River Wetland	Lower	Marsh	0.66	3.48	6.00	4.20	50.00	1.41	
12	Winston Churchill Wetland	Lower	Marsh	0.82	3.60	3.20	1.80	34.60	1.21	
4	Meadowvale Wetland	Lower	Swamp	0.61	3.93	3.00	0.60	26.20	0.85	**Degraded**
3	Creditview Wetland	Lower	Marsh	0.63	3.18	3.60	3.80	43.00	1.05	

[a] It is important to note that while this ranking system provides a general overview of site health, the ranking system is subjective.
[b] Parameters represent means across all years for each site.
[c] Green, yellow and orange shading is used to indicate the quality of sites and parameters across the spectrum of most pristine to most degraded. Green shading indicates the highest quality sites/parameters, yellow indicates median ranking of sites/parameters and orange indicates the lowest rankings of sites/parameters.

3.2 TEMPORAL ANALYSIS

3.2.1 Vegetation Parameters

3.2.1.1 Species Richness

Monitoring Question: Did species richness exhibit temporal variation over the monitoring period?

- There was an increasing trend in combined and ground vegetation species richness.
- It appears that regeneration species richness may have been lower in 2009 than other years.

According to linear regression analyses, species richness showed a significant increasing trend between 2005 and 2009 in the Credit River Watershed for both combined and ground vegetation (F=7.466, p=0.008 and F=11.476, p=0.001, respectively; Table 13, Fig. 14A & B). Although p-values for these, and other significant parameters, indicated a significant trend over time, the fitted linear models exhibited low R^2 values indicating that the models explain little of the variation in the data (Table 13).

Temporal differences in species richness could not be determined for ground vegetation, as an interaction was detected in the repeated measures ANOVA; therefore, one-way ANOVAs were completed for each year (results not shown). Differences in species richness were observed among years for combined vegetation and regeneration (F=35.388, p<0.001; F=11.364, p=0.0.23; Table 14, Fig. 15). For combined vegetation, these differences were consistent with the trend analysis. Although the Friedman test indicated differences among years for regeneration, the test does not provide any post-hoc analyses to determine where these differences exist. By examining the means for each year presented in Fig. 15 it appears that regeneration species richness may have been lower in 2009 than other years.

Observed increases in species richness were likely due to improved species identification over the same period. Species identification improved over the course of monitoring due to increased knowledge of the monitoring staff as well as the development of species lists for each site from past years. Additional years of data are needed in order to improve the robustness of the trend analyses.

Table 13. Summary of trend analysis results using linear regression for vegetation parameters between 2005 and 2009.

Vegetation Parameter	*F* value	*p* value [a]	R^2	Observed Trend
Combined Vegetation				
Total Species Richness	7.466	**0.008**	0.068	Increase
Mean Wetness Index	0.001	0.970	-0.011	Stable
mCC	0.031	0.861	-0.011	Stable
Weedy Species Richness	5.104	**0.026**	0.044	Increase
FQI	3.186	0.0778	0.024	Stable
Ground Vegetation				
Total Species Richness	11.476	**0.001**	0.105	Increase
Total Species Diversity	3.923	0.051	0.032	Stable
Native Species Diversity	4.965	**0.028**	0.043	Increase

[a] Results significant when $p<0.05$; significant values highlighted in bold.

Table 14. Summary of temporal analysis results using repeated measures ANOVA or the Friedman test for vegetation parameters between 2005 and 2009.

Vegetation Parameter	Test	*F* value	*p* value [a]	Observed Differences [b]
Combined Vegetation				
Species Richness	Repeated Measures ANOVA	**35.388**	**<0.001**	2005 & 2006 < 2007 & 2009 < 2008
Proportion of Native Species	Friedman	**17.367**	**0.002**	Decreasing from 2005 to 2008
mCC	Repeated Measures ANOVA	**2.645**	**0.042**	2007 < 2009 (p=0.100)
CC 8-10	Friedman	2.554	0.635	None
FQI	Repeated Measures ANOVA	N/A	N/A	Could not be determined
Weedy Species Richness	Repeated Measures ANOVA	**17.677**	**<0.001**	2005 & 2006<2007-2009; 2009<2008
Weediness Count -3	Friedman	**23.305**	**<0.001**	Highest in 2008
Locally Rare	Friedman	**16.366**	**0.003**	Increasing from 2005 to 2008
Regionally Rare	Friedman	5.850	0.211	None
Ground Vegetation				
Species Richness	Repeated Measures ANOVA	N/A	N/A	Could not be determined
Proportion of Native Species	Friedman	**20.919**	**<0.001**	Decreasing from 2005 to 2008
Native Species Diversity	Repeated Measures ANOVA	**31.849**	**<0.001**	2005 & 2006 < 2007-2009; 2009 < 2008
Total Species Diversity	Repeated Measures ANOVA	**40.220**	**<0.001**	2005 & 2006 < 2008 & 2009 < 2007
Native Species Evenness	Friedman	**50.444**	**<0.001**	Higher in 2005-2007 than 2008-2009
Total Species Evenness	Friedman	**47.778**	**<0.001**	Higher in 2005-2007 than 2008-2009
Regeneration				
Species Richness	Friedman	**11.364**	**0.023**	Lowest in 2009
Proportion of Native Species	Friedman	5.922	0.205	None
Total Species Diversity	Friedman	**19.111**	**<0.001**	Increasing from 2005 to 2007
Total Species Evenness	Friedman	**14.012**	**0.003**	Highest in 2007

[a] Results significant when $p<0.05$; significant values highlighted in bold.
[b] Differences between years determined using Tukey's post-hoc test for repeated-measures ANOVA analyses, and by visual observation for the Friedman Test.

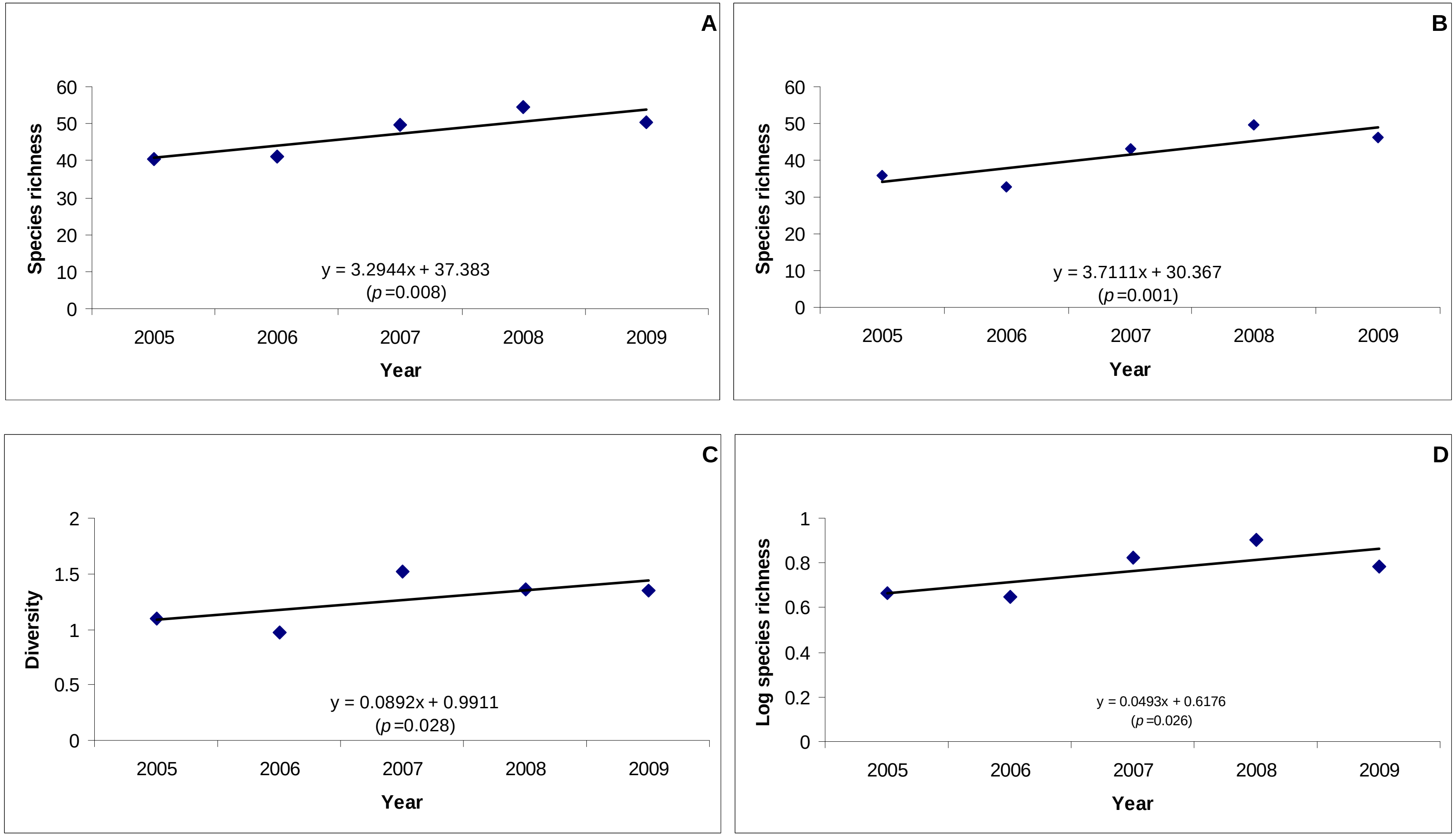


Figure 14. Linear regression for combined vegetation species richness (A), ground vegetation species richness (B), ground vegetation native species diversity (C) and weedy species richness (D) observed at 18 wetland sites in the Credit River Watershed between 2005 and 2009. Points represent the mean over the watershed for each year. Equation of the regression line and *p*-value provided. Weedy species richness was log transformed for analysis and is presented in this way.

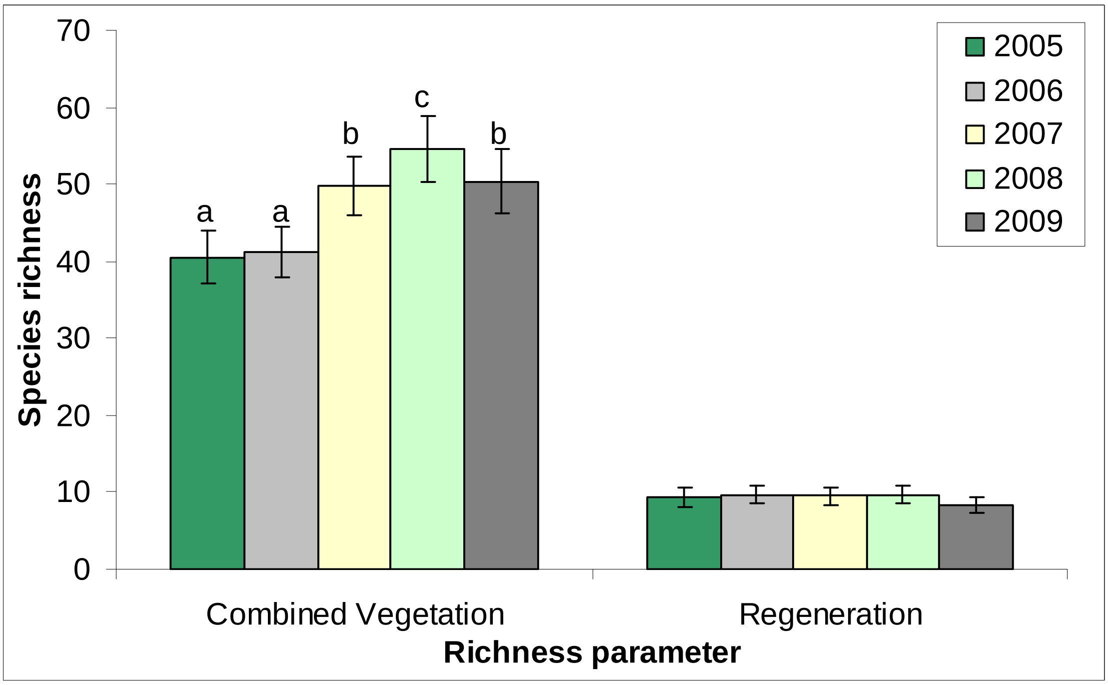


Figure 15. Mean species richness across all sites, as affected by year, for combined vegetation and regeneration. Repeated measures ANOVA was used for combined vegetation, while the Friedman test was used for regeneration. Within vegetation parameters, bars with different letters indicate a significant difference according to Tukey's HSD test for unequal sample sizes (p<0.05). Error bars indicate +/- SE.

3.2.1.2 Proportion of Native Species

Monitoring Question: Did the proportion of native species exhibit temporal variation over the monitoring period?

- The proportion of native species in both combined and ground vegetation appeared to decrease from 2005 to 2008 and increase in 2009.
- For regeneration no differences were observed between years.

Linear regression analysis was not completed on the proportion of native species parameters due to the ordinal natural of the data. The proportion of native species differed among years for combined vegetation and ground vegetation, but not regeneration (F=17.367, p<0.002; F=20.919, p<0.001; F=5.922, p=0.205, respectively; Table 14, Fig. 16). Although the Friedman test indicated differences among years for all three parameters, the test does not provide any post-hoc analyses to determine where these differences exist. By examining the means for each year presented in Fig. 16 it appears that combined and ground vegetation displayed the same pattern, with proportion of native species decreasing from 2005 to 2008 and then increasing again in 2009. The differences seen in the combined and ground vegetation may indicate an increase in non-native species within the watershed over the monitoring period. However, due to the increase in the proportion of natives in 2009, inconsistencies in species naming in the early stages of the program and improved sampling quality in the later years, it is difficult to tell if these changes are due to sampling inconsistencies, increased non-natives or a combination of both factors. This highlights the importance of additional years with improved monitoring accuracy.

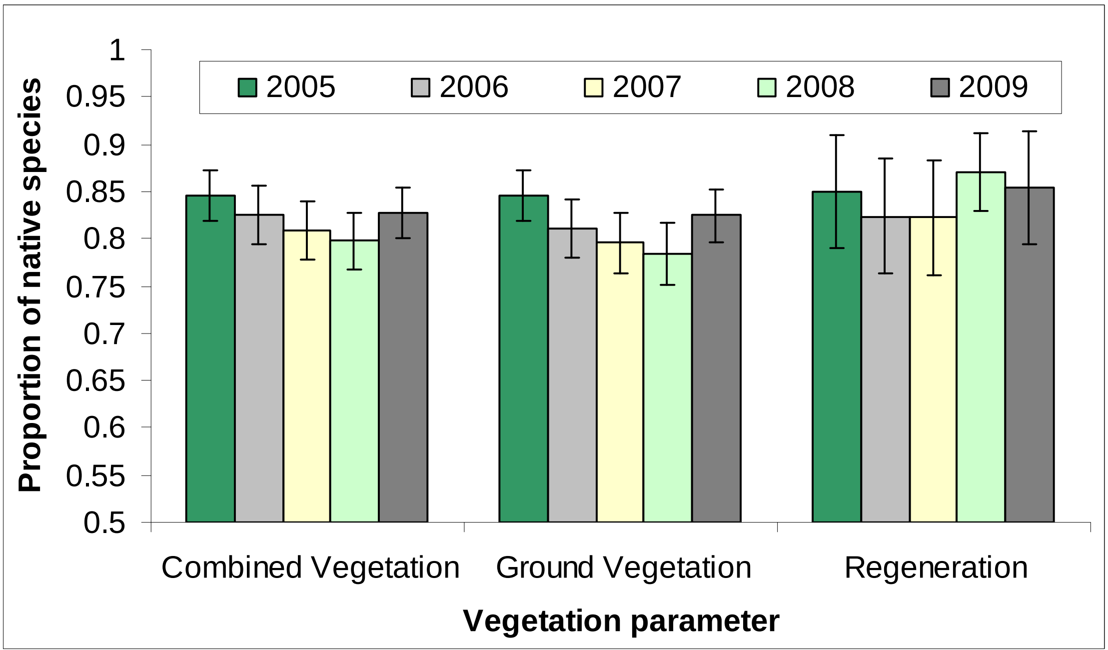


Figure 16. Mean proportion of native species across all sites, as affected by year, for combined vegetation, ground vegetation and regeneration. Friedman test was used for all parameters. Error bars indicate +/- SE.

3.2.1.3 Species Diversity

Monitoring Question: Did species diversity exhibit temporal variation over the monitoring period?

- There was a nearly significant increasing trend in total ground vegetation diversity over the monitoring period.
- Native ground vegetation species diversity significantly increased over the monitoring period.
- Regeneration diversity appeared to increase from 2005 to 2007.

According to linear regression, native ground vegetation species diversity increased over the five year monitoring period (F=4.965, p=0.028; Table 13, Fig. 14C). Total ground vegetation diversity was found to be stable over the monitoring period, although there was a nearly significant positive trend (F=3.923, p=0.051; Table 14). Temporal differences in total and native ground vegetation and regeneration diversity are presented in Fig. 17 (F=40.220, p<0.001; F=31.849, p<0.001; F=19.111, p<0.001, respectively; Table 14) and are generally consistent with the increasing trends observed in the regression analysis. Similarly to species richness, these increasing trends may be attributed to improved sampling. The decrease in 2008 and 2009 for all parameters may be attributed to the different system used to classify percent cover in these years (actual percent cover, as opposed to cover class), as percent cover was used in the diversity calculations.

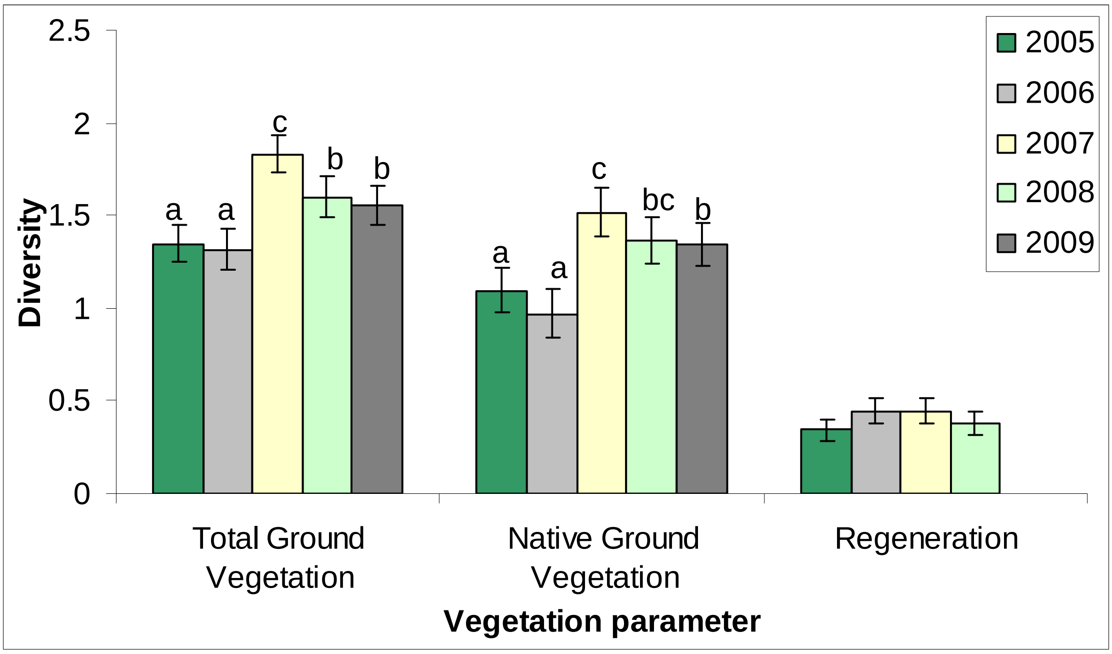


Figure 17. Mean diversity across all sites, as affected by year, for total and native ground vegetation and regeneration. Repeated measures ANOVA was used for total and native ground vegetation, while the Friedman test was used for regeneration. Within vegetation parameters, bars with different letters indicate a significant difference according to Tukey's HSD test for unequal sample sizes (p<0.05). If no letters are present, post-hoc analyses were unavailable to detect difference among years. Error bars indicate +/- SE.

3.2.1.4 Evenness

Monitoring Questions: Did species evenness exhibit temporal variation over the monitoring period?

- Total and native ground vegetation evenness appeared higher in the first three years of monitroing.
- Regeneration evenness appeared highest in 2007.

Linear regression analysis was not completed on the evenness parameters due to the ordinal natural of the data. Differences in total and native ground vegetation and regeneration evenness are presented in Fig. 18 (F=50.444, p<0.001; F=47.778, p<0.001; F=14.012, p=0.003, respectively; Table 14). Evenness for both total and native ground vegetation appeared higher in the first three years of sampling (2005 to 2007) than in 2008 and 2009. In addition, it appeared that evenness for regeneration was highest in 2007. This indicates that cover was more evenly distributed between species in the earlier sampling years. The apparent lower evenness in 2008 and 2009 may signify that either some species are becoming more dominant or that more species which occupy lower cover levels are appearing in the later years. This may be the case and could be the result of improved sampling. In conjunction, the lower evenness levels in 2008 and 2009 for all parameters may be due to the different system used to classify percent cover, similar to diversity. Overall, with additional years of reliable monitoring data, the reasons for these differences may become clearer.

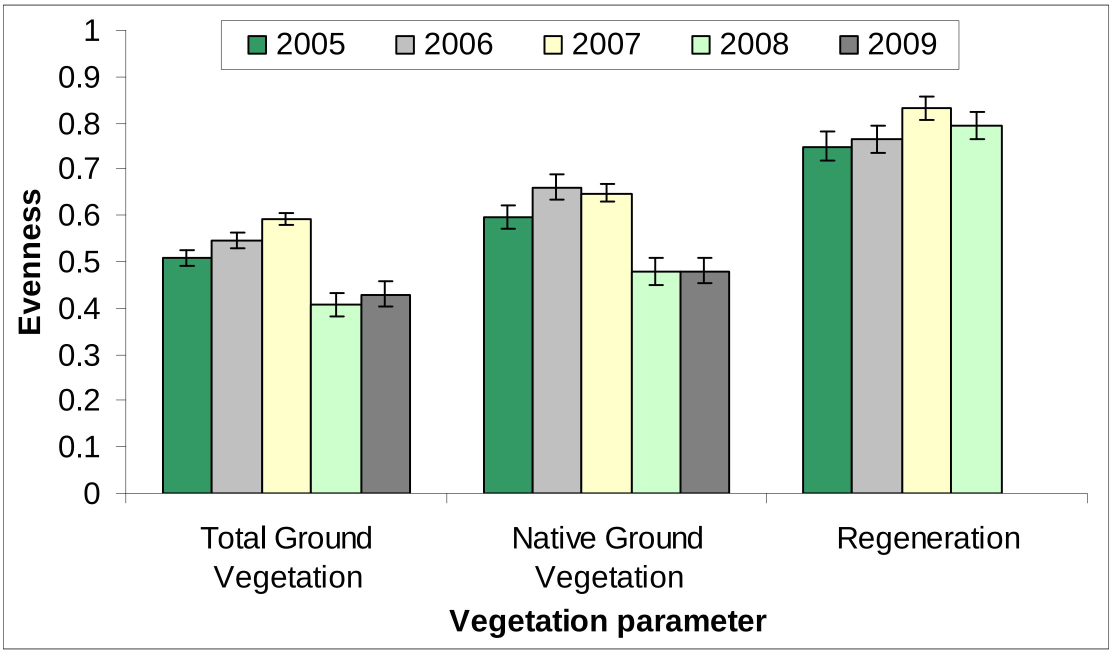


Figure 18. Mean evenness across all sites, as affected by year, for total and native ground vegetation and regeneration. The Friedman test was used for all analyses. Error bars indicate +/- SE.

3.2.1.5 Coefficients of Conservatism

Monitoring Questions: Did mCC and CC 8-10 exhibit temporal variation over the monitoring period? Did FQI exhibit temporal variation over the monitoring period?

- No trends were observed for mCC or FQI.
- There were no differences between years for CC 8-10.

Regression analyses were completed for mCC and FQI, but no significant trends were observed (Table 13); linear regression analysis could not be completed for CC8-10. Temporal analyses were also unable to detect differences in mCC, FQI and CC 8-10. Although the Repeated Measure ANOVA test indicated a significant difference between years for mCC (F=2.645, p=0.042; Table 14), post-hoc analyses were unable to detect these differences, which is consistent with regression results. Temporal differences in FQI could not be determined, as an interaction was detected in the repeated measures ANOVA; and therefore, one-way ANOVAs were completed for each year (results not shown). The number of species with a CC value between 8 and 10 did not differ among years (F=2.554, p=0.635; Table 14, Fig. 19). Therefore, it appears that the floristic quality of monitored sites within the watershed may have been relatively stable over the monitoring period.

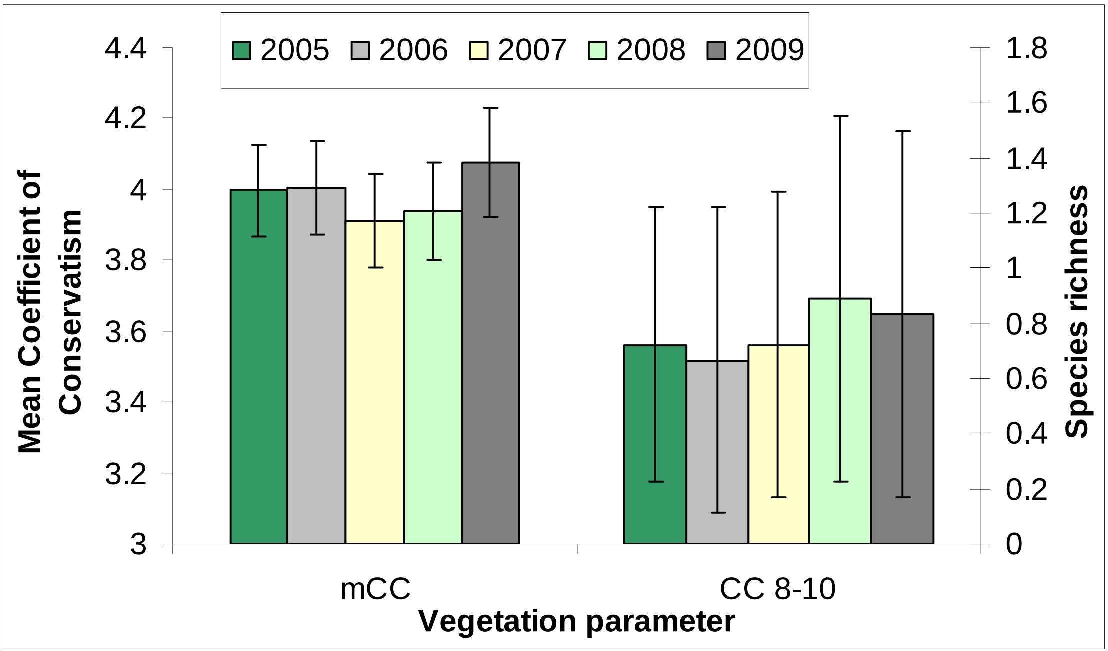


Figure 19. Mean mCC and CC 8-10 across all sites, as affected by year. Repeated measures ANOVA was used for mCC, while the Friedman test was used for CC 8-10. Error bars indicate +/- SE.

3.2.1.6 Weediness

Monitoring Question: Did the weedy species richness and weediness score -3 exhibit temporal variations over the monitoring period?   - Weedy species richness increased over the monitoring period.   - Weediness score -3 appears highest in 2008.

Linear regression analysis was not completed for weediness score -3; however, weediness score -3 differed among years (*F*=23.305, *p*<0.001; Table 14, Fig. 20). Due to the use of the Friedman test, significant differences between years could not be determined. It did appear that the number of -3 weediness species steadily increased from approximately one species per site in 2005 to approximately two species per site in 2008. This increase is consistent with the decrease in proportion of native species observed over the years in combined and ground vegetation (Fig. 16) and may indicate that very weedy species are becoming more common in the watershed. It is unlikely that the increase in weediness score -3 was due to improved sampling quality over the years because the majority of -3 species, such as Garlic Mustard, Purple Loosestrife and Common Buckthorn are relatively easy to identify.

Weedy species richness was found to increase from 2005 to 2009 (*F*=45.104, *p*=0.026; Table 13, Fig. 14D). Temporal differences between years are consistent with this (*F*=17.677, *p*<0.001; Table 14, Fig. 20). The increasing trend observed for weedy species richness may also be a result of improved sampling. However, analyses indicate that the proportion of native species and weediness count -3 were higher in the later years of the monitoring program than the early years. Therefore, the cause of this trend is unclear.

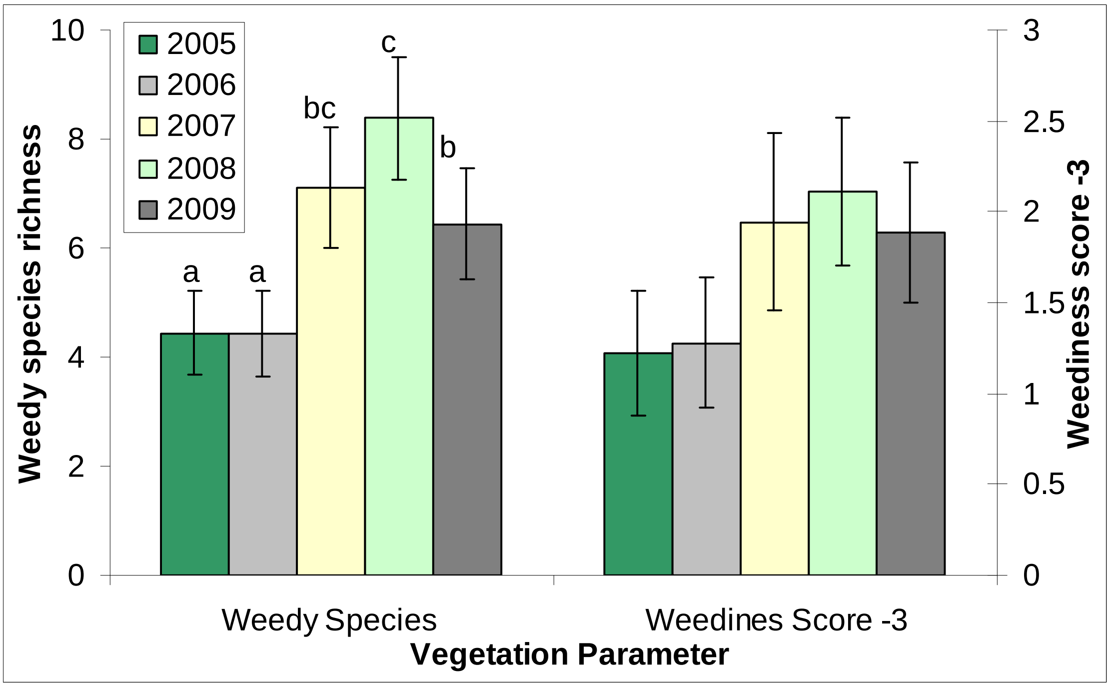


Figure 20. Mean weedy species richness and weediness score -3 across all sites as affected by year. Repeated measures ANOVA was used for weedy species richness, while the Friedman test was used for weediness score -3. Within vegetation parameters, bars with different letters indicate a significant difference according to Tukey's HSD test for unequal sample sizes (*p*<0.05). If no letters are present, post-hoc analyses were unavailable to detect difference among years. Error bars indicate +/- SE.

It is suggested that wetlands may be vulnerable to invasions, likely due to the fact that they are landscape sinks (Zedler and Kercher 2004). Landscape sinks facilitate invasions through accumulating debris, sediments, water, and nutrients that create canopy gaps or accelerate the growth of opportunistic vegetation species. Many wetland invaders form monotypes which can alter habitat structure, nutrient cycling and food webs, in addition to decreasing biodiversity (Zedler and Kercher 2004). However, over 60% of the species detected with a weediness score of -3 had a wetness value of two or higher (Table 10). This indicates that these species were facultative or obligate upland species, including Common Buckthorn which was the most widespread species with a -3 weediness score. Wetland monitoring sites were established along the hydrological gradient of each wetland, from driest to wettest conditions. For that reason, the first pair of subplots at each monitoring site was often situated on the edge of the wetland, where it was drier and very weedy. Therefore, weedy species may be becoming more dominant along the edges of wetlands in the watershed and special attention should be paid to this trend.

3.2.1.7 Rare Species

Monitoring Question: Did the number of locally rare and regionally rare species exhibit temporal variations over the monitoring period?   - Locally rare species appeared to increase from 2005 to 2008, while no change was seen in regionally rare species.

Linear regression analysis was not completed for rare species parameters. The number of locally rare species per site differed among years (F=16.366, p=0.003; Table 14, Fig. 21). However, no differences were observed among years for regionally rare species (F=5.850, p=0.211; Table 14, Fig. 21). Although a post-hoc test to differentiate years was not available for the Friedman test, it appears that locally rare species increased from 2005 to 2008. In 2008, two rare orchids were recorded at monitoring sites which were not previously observed. These include Small Yellow Lady's-slipper (*Cypripedium parviflorum var. makasin*) at Hungry Hollow Wetland and Northern Green Orchid (*Platanthera hyperborea var. hyperborean*) at Terra Cotta Wetland. The large increase in 2008 may also have been due to improved data collection and entry. Species identification improved over the monitoring period and in 2008 a consultant was commissioned to assist in species identification. This may have culminated in more rare species being identified that year.

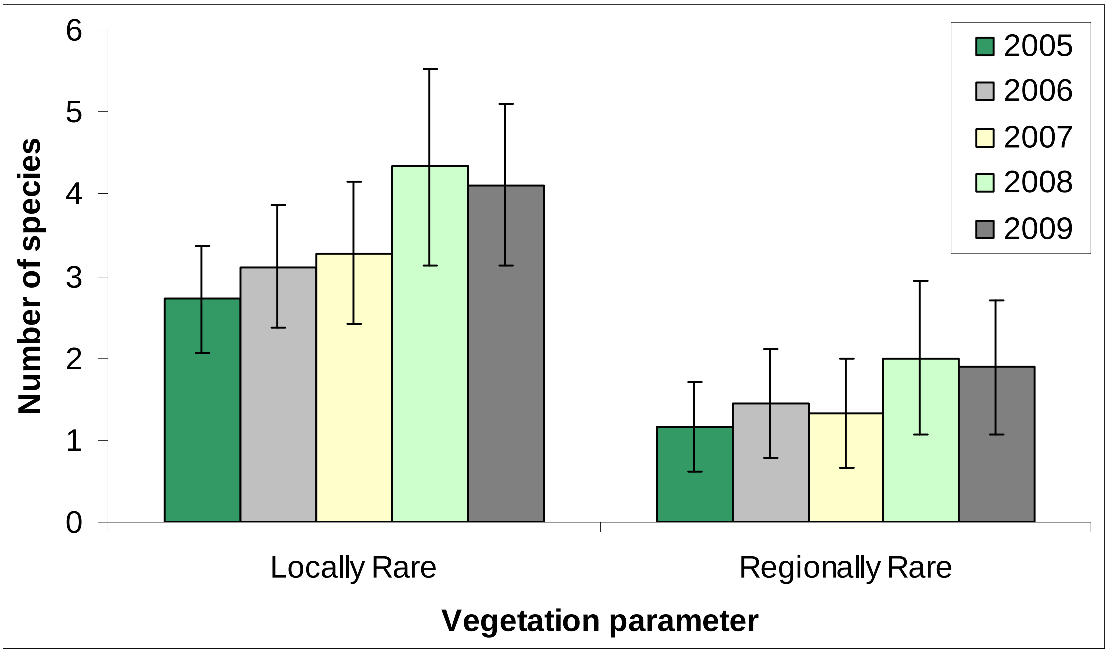


Figure 21. Mean number of locally and regionally rare species across all sites, as affected by year. Friedman's test was used for both parameters. Error bars indicate +/- SE.

3.2.1.8 Power Analysis and Statistical Process Control: For all vegetation parameters analyzed with a linear regression, power analysis determined that the following number of sites would be needed in order to detect 1, 2 and 3 SD of change: 31.8, 7.2 and 2.6 sites, respectively. Therefore, a sufficient number of sites were monitored to detect significant changes in the vegetation parameters with 90% power at an 80% confidence level for effect sizes greater than or equal to 2 SD. This is important because changes in the magnitude of 2 SD are often used as a warning threshold in statistical process control and changes in the magnitude of 3 SD are often used as the critical threshold. A change of 1 SD cannot currently be detected for any of the vegetation parameters. Detectable effect size in the form of percent change varied markedly among the parameters. Detection of less than 5% change is possible for only mCC; changes of at least 5-10% can be detected for mean wetness index and FQI; changes of over 30% can be detected for the weediness parameters; changes of between 10% and 30% can be detected for the remaining parameters (Table 15). Although minimum detectable percent change was as high as 36.5%, this is below the 40% change limit indicating poor data quality. Data quality for all parameters was either green or yellow indicating that data collected by the monitoring program thus far is of good or moderate quality.

Table 16 lists the critical and warning limits calculated for each wetland vegetation parameter. These upper and lower limits will be adopted as monitoring thresholds to which future vegetation monitoring data can be compared. Future values that exceed the upper or lower warning limits will signify emergence of possible increasing or declining population trends, while values that exceed the upper and lower critical limits will signify a significant deviation from natural variability. Such deviations would warrant immediate management action before trends became irreversible. No monitoring thresholds can be applied to those parameters deemed to be out of control or those displaying no variance over the monitoring period.

Table 15. Power analysis results for detecting effect sizes for trend analyses. Values represent the minimum effect size detectable under current monitoring conditions.

Parameter	Minimum Detectable Effect Size (% Change)	Data Quality
Combined Vegetation		
Species Richness	17	Green
Mean Wetness Index	5.1	Green
mCC	2.5	Green
CC 8-10	13	Green
FQI	9.1	Green
Weedy Species Richness	36.5	Yellow
Weediness Count -3	32.6	Yellow
Locally Rare	25.6	Yellow
Regionally Rare	28.7	Yellow
Ground Vegetation		
Species Richness	22	Yellow
Total Species Diversity	16.3	Green
Native Species Diversity	23.8	Yellow

Table 16. Monitoring thresholds for all wetland vegetation parameters extracted from Statistical Process Control individual moving average (IMR) charts for data collected from 2005 through 2009. Limits are presented in the specific units used to measure each given parameter.

Parameter	Lower Critical Threshold	Lower Warning Threshold	Upper Warning Threshold	Upper Critical Threshold
Combined Vegetation				
Species Richness	35.15	39.19	55.35	59.38
Proportion of Native Species	0.769	0.786	0.856	0.873
Mean Wetness Index	-2.19	-2.11	-1.77	-1.7
mCC‡	3.81	3.87	4.11	4.16
FQI	21.04	21.94	25.52	26.41
CC 8-10	0.545	0.619	0.915	0.988
Locally Rare	2.29	2.7	4.32	4.73
Regionally Rare	0.79	1.05	2.09	2.34
Weedy Species Richness	2.25	3.56	8.78	10.08
Weediness Count -3	0.95	1.2	2.18	2.43
Ground Vegetation				
Species Richness	25.95	31.13	51.87	57.05
Total Species Diversity	0.98	1.16	1.9	2.08
Total Species Evenness	0.303	0.368	0.626	0.69
Native Species Diversity	0.7	0.87	1.63	1.82
Native Species Evenness	0.411	0.465	0.683	0.737
Proportion of Native Species	0.744	0.767	0.857	0.881
Regeneration				
Species Richness	7.99	8.43	10.21	10.65
Species Diversity	*Not enough data*			
Species Evenness	*Not enough data*			
Proportion of Native Species	0.781	0.802	0.886	0.906

3.2.2 Species Level Analysis

Monitoring Question: Were trends observed for individual wetland vegetation species over the monitoring period?

- The number of sites at which Common Buckthorn was present in the ground vegetation increased over the monitoring period; however, this may be due to sampling quality.
- No other selected species displayed any trends.

Trends were examined for selected wetland vegetation species using the Cochrane-Armitage test. Ground vegetation species examined included: Garlic Mustard, Common Buckthorn and Purple Loosestrife (Table 17). The only species which exhibited a trend over time was Common Buckthorn, which was present at two sites in 2005 and seven sites in 2009. It is possible that the increase in Common Buckthorn was due to improved monitoring quality. Very young Common Buckthorn plants may be easily confused with young Spotted Touch-me-not (*Impatiens capensis*) plants or may have been disregarded as too young to identify in the early years of the program. No other selected species exhibited an increasing or decreasing trend over time. Therefore, it does not appear that the presence of very weedy non-native species in monitoring sites increased over the monitoring period.

Power analysis and SPC were completed for species evaluated with the Cochrane-Armitage test (Table 17 & 18). Power analysis revealed that under the current monitoring program, data quality is generally poor using the Cochrane-Armitage test to examine trends in species occurrence (Table 17). It appears that examining trends in individual species site occupancy may have limited utility within the current monitoring program, as a larger sample size is needed to detect changes of 2 SD (warning threshold). Therefore, it may be necessary for the current monitoring program to re-evaluate its current methodology for examining individual species trends, to yield results which are meaningful for ecological management.

Statistical process control indicated that data for Purple Loosestrife is not in control and therefore no baseline thresholds could be set for this species at this time. In contrast, data for Common Buckthorn and Garlic Mustard were determined to be in control, resulting in the development of thresholds that may be used for future analysis.

Table 17. Cochrane-Armitage results for selected species trends and power analysis between 2005 and 2009.

Vegetation Parameter	Cochrane-Armitage Test			Power Analysis				
	Critical Z	Observed Z	*p* value	N [a]			Minimum Effect Size (SD)	Data Quality
				1 SD	2 SD	3 SD		
Common Buckthorn	1.96	2.138	0.033	79	36	22	4.2	Red
Purple Loosestrife	1.96	1.384	0.166	107	50	31	5.6	Red
Garlic Mustard	1.96	1.139	0.255	132	63	40	6.8	Red

[a] Number of sites required to detect the specified amount of change.

Table 18. Statistical Process Control (SPC) results for selected species trends between 2005 and 2009.

Vegetation Parameter	Lower Critical Threshold	Lower Warning Threshold	Upper Warning Threshold	Upper Critical Threshold
Common Buckthorn	0.104	0.166	0.412	0.474
Purple Loosestrife	*Not in control*			
Garlic Mustard	0.152	0.201	0.399	0.448

3.3 SPATIAL ANALYSIS

3.3.1 Species Richness

Monitoring Question: Did species richness differ among physiographic zones?

- Species richness was unaffected by physiographic zone.

Repeated measures ANOVA analysis of ground vegetation richness revealed a significant interaction between year and zone. When an interaction is present, interpretation of the main effects (year and zone) are impossible, as the two factors are not independent (Underwood 1997). Therefore, when an interaction was detected, one-way ANOVA analyses were conducted separately for each year to examine differences among physiographic zones.

There was no difference detected in combined wetland vegetation species richness among the physiographic zones (F=1.544, p=0.246; Table 19). The mean number of species detected annually was between 17.6 and 25.8 in the Lower watershed, between 31 and 48.2 in the Middle watershed, and between 38.6 and 52 in the Upper watershed. Although not significant, the Lower zone consistently contained lower richness than the other two zones (Fig. 22). There were also no differences among physiographic zones when ground vegetation and regeneration were analyzed separately (F=2.222, p=0.143, F=1.044, p=0.593, respectively; Table 19).

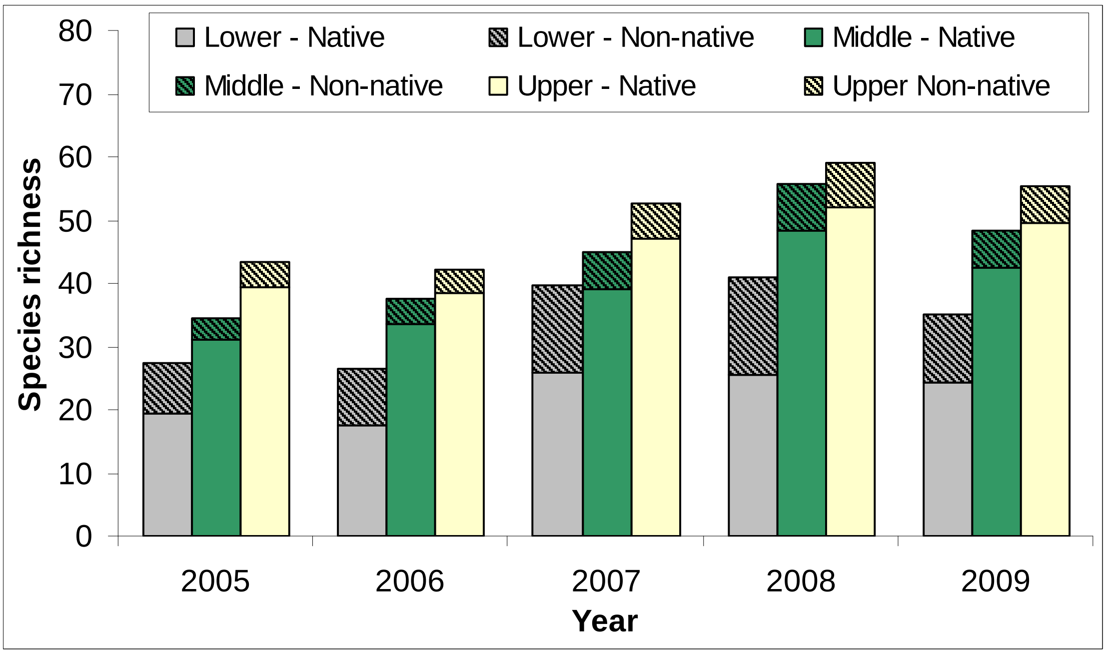


Figure 22. Mean species richness across sites within each physiographic zone in the Credit River Watershed.

Table 19. Summary of spatial analysis results using repeated measures ANOVA or the Kruskal-Wallis test for vegetation parameters between 2005 and 2009.

Vegetation Parameter	Physiographic Effect		Physiographic Differences
	F	***P*** [d]	
Species Richness			
Combined Vegetation	1.544	0.246	None
Ground Vegetation	2.222	0.143	None
Regeneration [ab]	1.044	0.593	None
Proportion of Native Species			
Combined Vegetation [a]	**6.906**	**0.032**	Lower < Upper
Ground Vegetation [a]	**7.456**	**0.024**	Lower < Middle & Upper
Regeneration [ab]	4.077	0.130	None
Diversity			
Total Ground Vegetation [c]	2.831	0.091	None
Native Ground Vegetation	3.665	0.051	None
Regeneration [ab]	4.972	0.083	None
Evenness			
Total Ground Vegetation [c]	2..233	0.328	None
Native Ground Vegetation	2.284	0.319	None
Regeneration [ab]	3.583	0.167	None
Floristic Quality Assessment			
Weedy Species Richness	**4.309**	**0.033**	Lower > Upper (*p*=0.054)
Weediness Count -3 [a]	**8.781**	**0.012**	Lower > Middle & Upper
mCC	1.689	0.218	None
CC 8-10 [a]	2.062	0.357	None
FQI	2.588	0.108	None
Locally Rare [a]	0.508	0.776	None
Regionally Rare [a]	2.509	0.285	None

[a] Kruskal Wallis test conducted for each year separately to test for effect of physiographic zone, with *H* and *p* values shown for 2009 tests only.
[b] Regeneration parameters exhibit spatial autocorrelation and were analyzed with non-parametric methods.
[c] Year by zone interaction present in repeated measures ANOVA. One way ANOVAs performed for each year separately, with F and *p*-values displayed for 2009 only.
[d] Results significant when $p<0.05$; significant values highlighted in bold.

Species richness of vegetation in wetlands has previously been shown to decrease in developed landscapes (Lougheed et al. 2008). Therefore, it was surprising that no differences were observed among physiographic zones. A possible explanation is that non-native species are often imported into urban environments. This increase in non-native species could increase the diversity and abundance of urban species to levels that are similar to surrounding natural environments (McKinney 2006).

Unequal distribution of marsh and swamp monitoring sites throughout the watershed is another possible reason that differences in species richness were not observed among physiographic zones. Richness estimates are expected to be heavily influenced by wetland type, and the current unequal representation of wetland types among physiographic zones may be obscuring biological differences among the zones.

3.3.2 Proportion of Native Species

Monitoring Question: Did the proportion of native species differ among physiographic zones?

- The Lower zone contained a lower proportion of native species than the Middle and/or Upper zone.

The proportion of native species differed among zones across the monitoring period (Table 20). For combined and ground vegetation the Lower zone generally had a significantly lower proportion of natives than the Middle and/or Upper zone (Table 20, Fig. 23A & B). For regeneration, similar differences were seen in three of five years (Table 20, Fig. 23C). The Middle and Upper zones did not contain significantly different proportions of native species (Table 20, Fig. 23C).

Table 20. Kruskal-Wallis test results for differences in the proportion of native species among physiographic zones.

Parameter	Year	Test Statistic (*H*)	*p*-value [a]	Zone Differences [b]
Combined Vegetation	2005	8.494	**0.014**	Lower-Upper (p=0.019)
	2006	9.075	**0.011**	Lower-Middle (p=0.046) Lower-Upper (p=0.014)
	2007	8.001	**0.018**	Lower-Upper (p=0.020)
	2008	9.653	**0.008**	Lower-Middle (p=0.028) Lower-Upper (p=0.013)
	2009	6.906	**0.032**	Lower-Upper (p=0.035)
Ground Vegetation	2005	7.944	**0.019**	Lower-Upper (p=0.023)
	2006	9.997	**0.007**	Lower-Middle (p=0.036) Lower-Upper (p=0.009)
	2007	8.548	**0.014**	Lower-Upper (p=0.015)
	2008	10.263	**0.006**	Lower-Middle (p=0.023) Lower-Upper (p=0.009)
	2009	7.456	**0.024**	Lower-Upper (p=0.026)
Regeneration	2005	7.228	**0.027**	Lower-Upper (p=0.046)
	2006	5.120	0.077	None
	2007	6.114	**0.047**	Lower-Upper (p=0.063)
	2008	9.184	**0.010**	Lower-Upper (p=0.013)
	2009	4.077	0.130	None

[a] Results significant when $p<0.05$; significant values highlighted in bold
[b] *p*-values in this column represent the results of post-hoc analyses between the indicated zones. Significant at $p<0.05$.

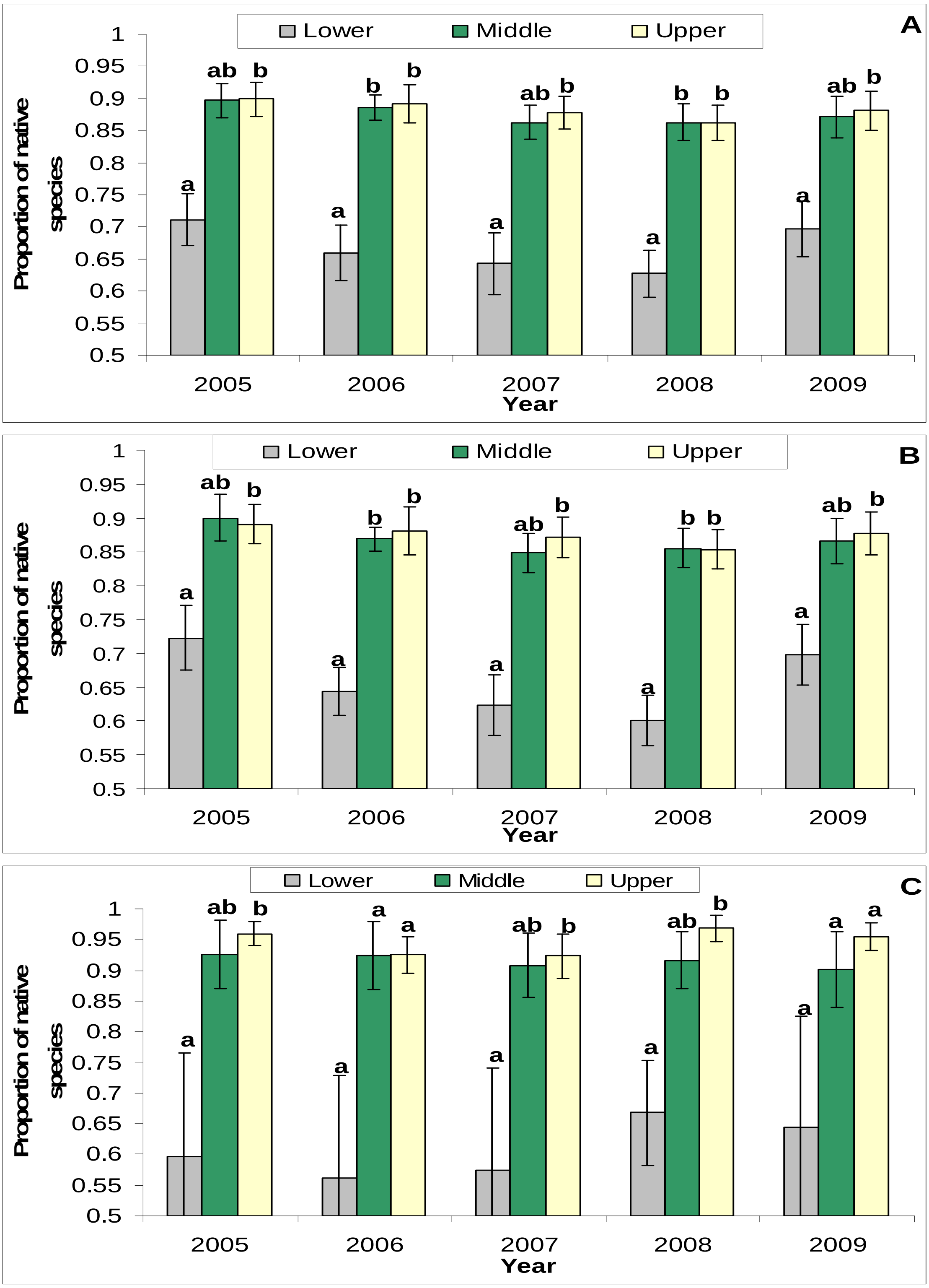


Figure 23. Kruskal-Wallis analysis of mean proportion of native species across sites within each physiographic zone for combined vegetation (A), ground vegetation (B) and regeneration (C). Within years, bars with different letters indicate a significant difference according to post-hoc analyses (p<0.05). Error bars indicate +/- SE.

It is not surprising that the Lower zone contained a lower proportion of native species. Disturbances caused by urban development are expected to extirpate native species while simultaneously promoting the establishment of non-native species that tend to persist in disturbed environments (McKinney 2006). The Lower zone is comprised of nearly four times more urban land cover than the Upper and Middle watersheds. In the Lower watershed, nearly 60% of the land has been converted to urban cover, while only 8% is occupied by natural cover (woodlands and wetlands) (Fig. 24). The lack of differences between the Upper and Middle watershed are also reflected in the land cover of those zones. Both the Upper and Middle zones have only 15% urban cover, while the percent cover of natural, agricultural and successional land uses are similar between the two zones (Fig. 24).

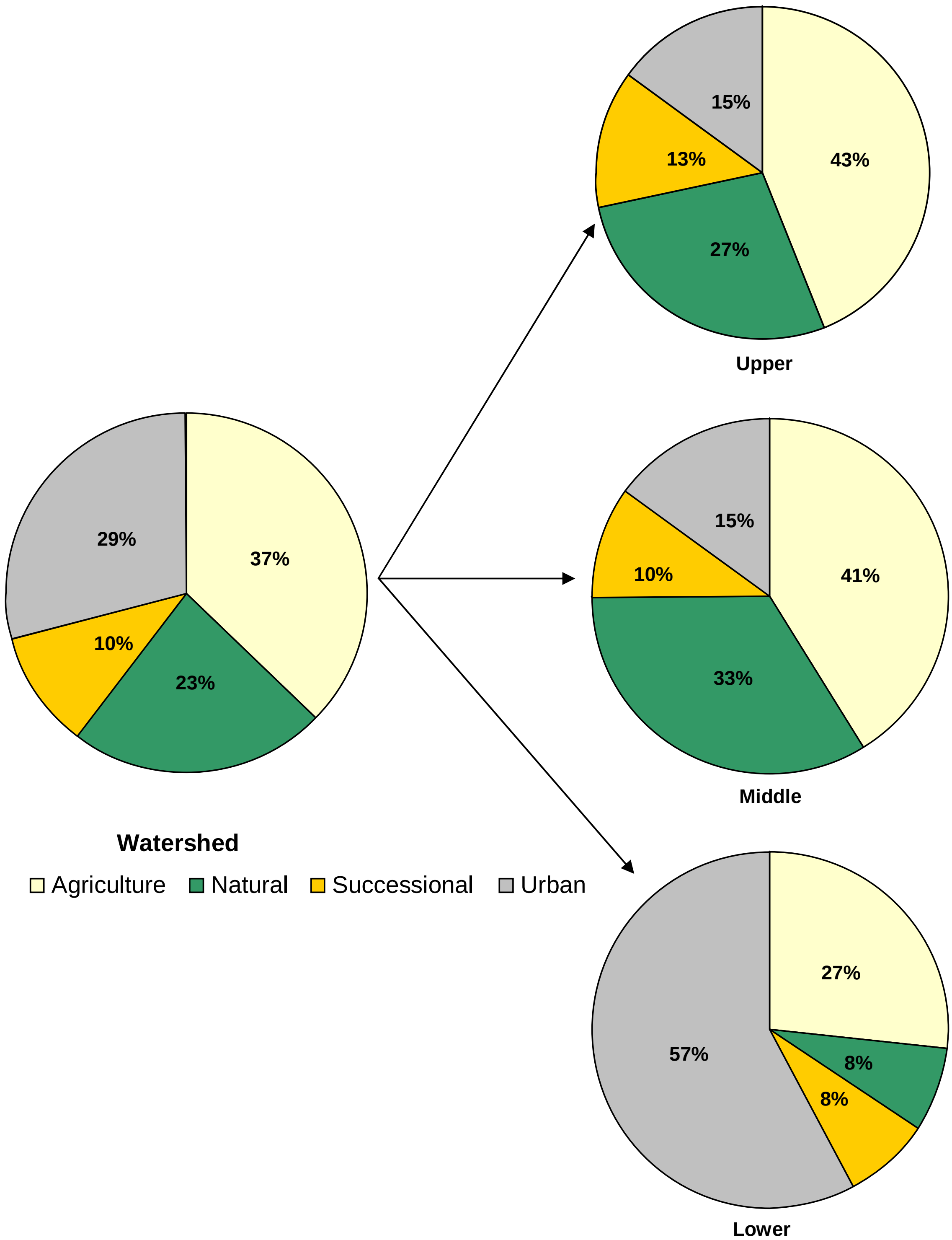


Figure 24. Land cover in the Credit River Watershed (Credit Valley Conservation 2007a).

3.3.3 Species Diversity

Monitoring Question: Did total or native species diversity differ among physiographic zones?

- No differences were observed among zones for all diversity parameters.

No significant differences were detected among zones for total and native ground vegetation (F=2.831, p=0.091; F=3.665, p=0.051; Fig. 25) according to repeated measure ANOVA analyses. Similarly, the Kruskal-Wallis test revealed no differences among zones for regeneration diversity (2005, F=4.705, p=0.095; 2006, F=4.509, p=0.105; 2007, F=4.509, p=0.105; 2008, F=4.972, p=0.083; Fig. 26). This is consistent with spatial differences observed in the species richness analyses. The diversity calculation is partially based on number of species; therefore, parallels are expected between the two types of parameters.

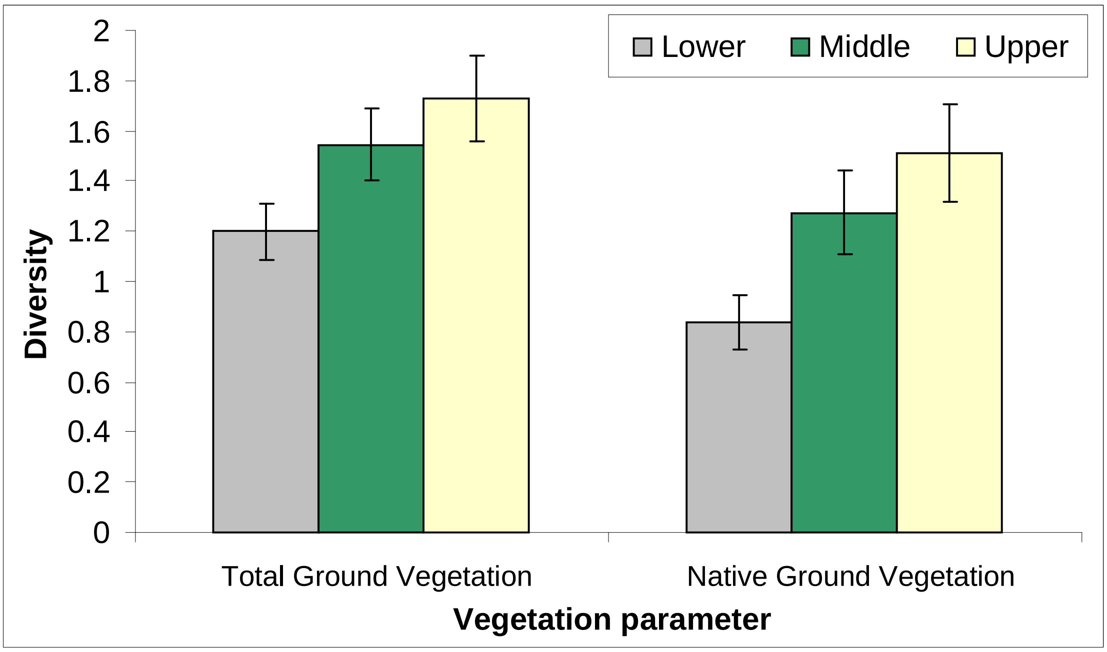


Figure 25. Mean total and native ground vegetation diversity across sites within each physiographic zone in the Credit River Watershed. Repeated measures ANOVA was used for both parameters. Within years, bars with different letters indicate a significant difference according to Tukey's HSD test for unequal sample sizes analyses (p<0.05). Error bars indicate +/- SE.

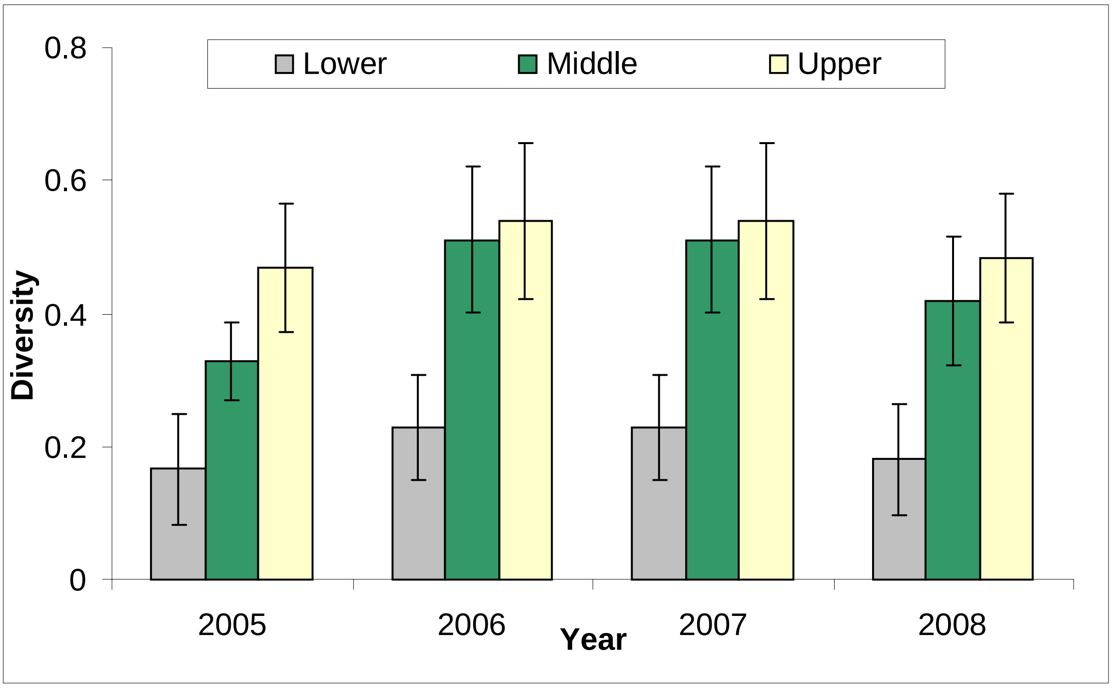


Figure 26. Kruskal-Wallis analysis of mean regeneration diversity across sites within each physiographic zone in the Credit River Watershed. Within years, bars with different letters indicate a significant difference according to post-hoc analyses (*p*<0.05). Error bars indicate +/- SE.

3.3.4 Species Evenness

Monitoring Question: Did total or native species evenness differ among physiographic zones?

- Generally there were few differences in evenness among the zones, with the lowest evenness in the Upper zone in early years.

There were generally few significant differences observed in evenness for combined and ground vegetation and regeneration (Fig. 27A & B) according to Kruskal-Wallis analyses. Exceptions include that the Upper zone had lower evenness in 2005 for both combined and ground vegetation and in 2005 and 2006 for regeneration (Fig. 27C). Although not significant, the Lower zone tended to have the highest evenness in ground vegetation and regeneration, which may indicate that in urbanized areas of the watershed there are a few dominant species.

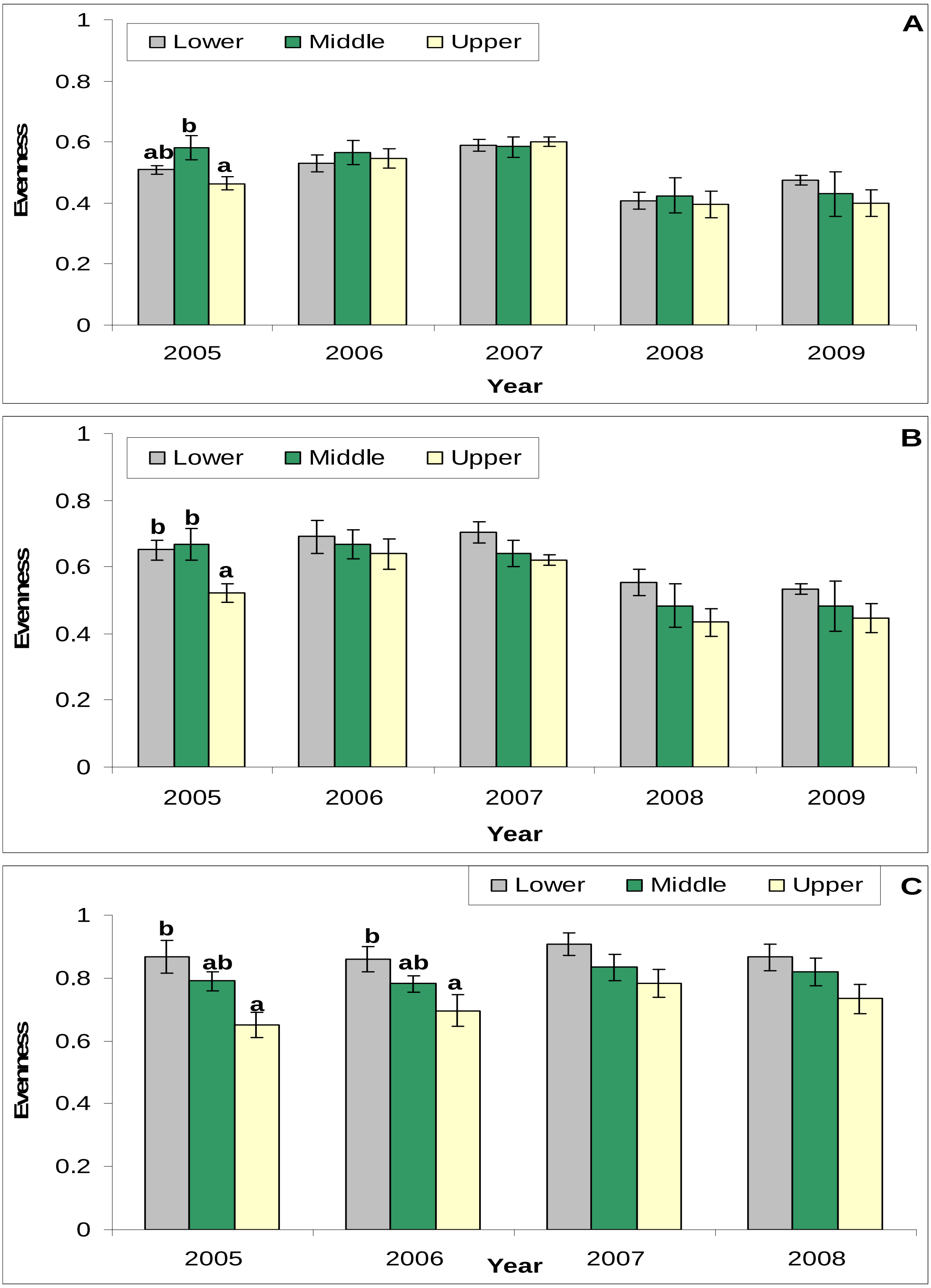


Figure 27. Kruskal-Wallis analysis of mean evenness across sites within each physiographic zone for combined vegetation (A), ground vegetation (B) and regeneration (C). Within years, bars with different letters indicate a significant difference according to post-hoc analyses (*p*<0.05). Error bars indicate +/- SE.

3.3.5 Coefficient of Conservatism

Monitoring Questions: Did the mCC differ among physiographic zones? Did the number of species with a Conservatism score of 8-10 differ among physiographic zones? Did FQI differ among physiographic zones?

- The mCC and CC 8-10 did not differ among zones.
- FQI was significantly, or nearly so, lower in the Lower zone than the Middle and/or Upper zones in 3 of 5 years.

There was no difference in mCC scores among physiographic zones in the Credit River Watershed (F=1.689, p=0.218; Table 19). Despite the fact that significant differences were not observed, the Lower zone contained the lowest mean mCC and the Upper zone contained the highest mean mCC (Fig. 28). Additionally, there was no difference in the number of individuals with high conservatism scores (CC 8-10) among the physiographic zones in any year (2005, H=0.823, p=0.663; 2006, H =1.394, p=0.498; 2007, H =0.135, p=0.935; 2008, H =0.823, p=0.663; 2009, H =2.062, p=0.357; Fig. 29). Although not significant, a greater number of species with high conservatism scores were observed in the Upper watershed than the Lower and Middle watershed (Fig. 29). However, this is due to the inclusion of Caledon Lake. In three of five years, FQI scores varied significantly among physiographic zones (2005, F=3.983, p=0.041; 2006, F =4.527, p=0.029; 2007, F =3.327, p=0.064; 2008, F =5.245, p=0.019; 2009, F =2.588, p=0.108; Fig. 30). However, in all years except 2008, the post-hoc analysis found no differences among zones. Generally, the Upper and/or Middle watersheds had significantly, or nearly so, higher FQI values than the Lower watershed (Fig. 30). This indicates that FQI may be impacted by urbanization in the Lower zone.

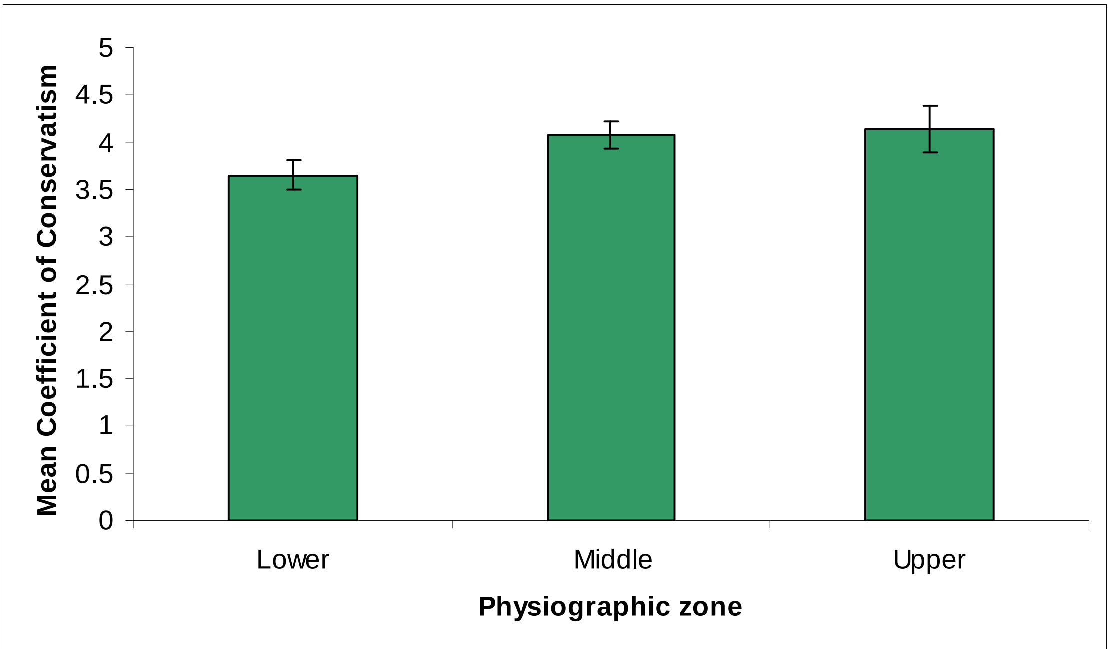


Figure 28. Mean mCC across sites within each physiographic zone throughout the Credit River Watershed. Error bars indicate +/- SE.

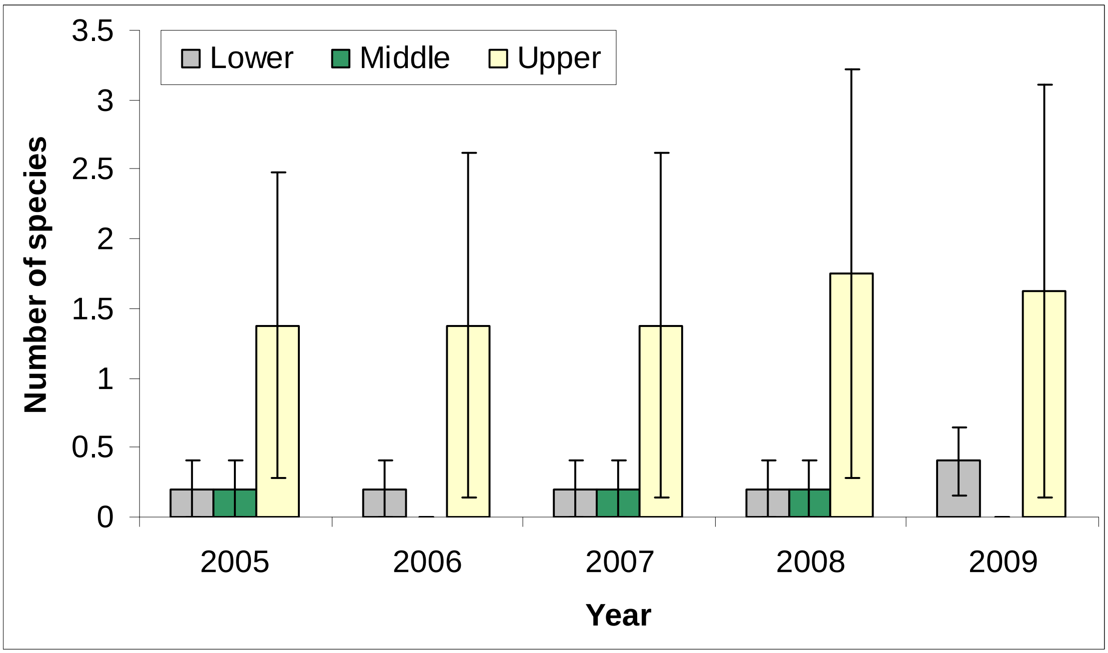


Figure 29. Mean number of species with high conservatism scores (CC 8-10) across sites within each physiographic zone within the Credit River Watershed. Error bars indicate +/- SE.

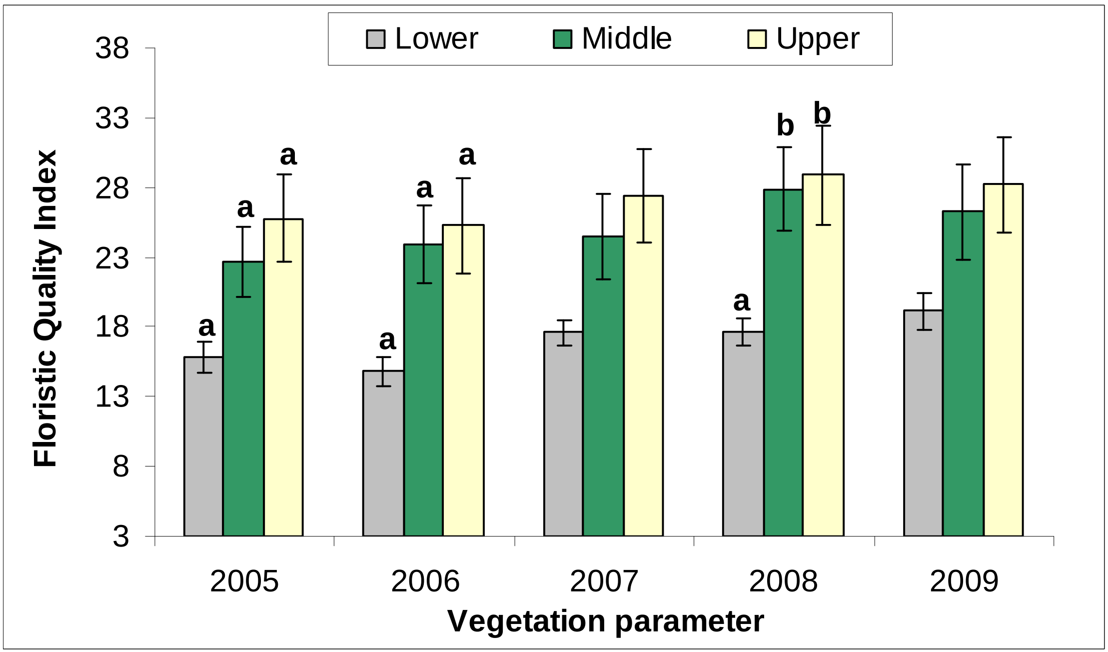


Figure 30. Mean Floristic Quality Index (FQI) across sites within each physiographic zone throughout the Credit River Watershed. Bars with different letters indicate a significant difference according to post-hoc analyses (p<0.05). If no letters are present the ANOVA analysis detected no differences. Error bars indicate +/- SE.

3.3.6 Rare Species

Monitoring Question: Did the number of locally and regionally rare species differ among physiographic zones?

- The number of locally and regionally rare species were unaffected by physiographic zone.

No differences were detected in the number of locally and regionally rare species among physiographic zones, in any year, as indicated by Kruskal-Wallis analyses (Table 19). Despite the fact that differences were not significant, the Upper watershed had a larger number of locally and regionally rare species than the other physiographic zones (Fig. 31). The Lower watershed typically had the lowest number of locally rare species, but surprisingly the Middle watershed contained the lowest number of regionally rare species. This was unanticipated. The Middle watershed is expected to contain the most pristine habitats and is under less development pressure than the other two zones due to the Niagara Escarpment and Green Belt legislation. This may be because the majority of sites monitored in the Middle watershed are swamps, while sites monitored in the Lower and Upper watersheds are composed mainly of marsh habitats. Rare species are almost always associated with species rich communities (Bedford et al. 1999) and the majority of swamps monitored in the watershed had below average species richness. The large number of sites with locally and regionally rare species and species with CC values of 8-10 in the Upper watershed is partially due to the inclusion of Caledon Lake Wetland.

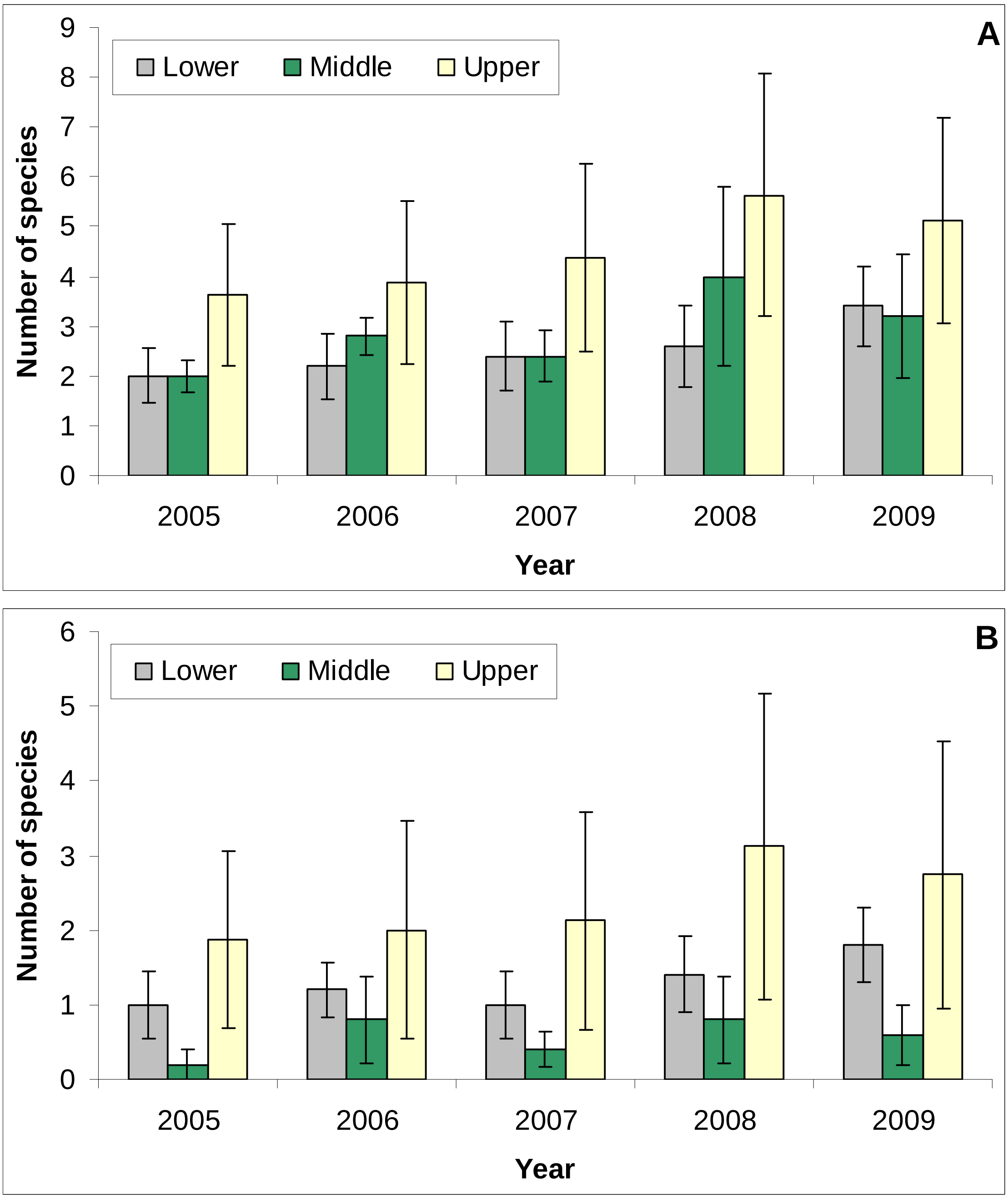


Figure 31. Mean number of locally (A) and regionally (B) rare species across sites within each physiographic zone within the Credit River Watershed. Error bars indicate +/- SE.

3.3.7 Weediness

Monitoring Question: Did weedy species richness differ among physiographic zones? Did the number of species with a weediness score of -3 differ among physiographic zones?

- There was a near significant difference between the Upper and Lower zones for Weedy species richness.
- Weediness count -3 was higher in the Lower zone than the Upper and/or Middle zones.

Spatial analysis with repeated measures ANOVA indicated a significant difference among zones for weedy species richness (F=4.309, p=0.033; Table 19, Fig. 32). However, the Tukey's HSD test only indicated nearly significant differences between the Upper and Lower zones (p=0.054). The Kruskal-Wallis test indicated that the number of individuals with a weediness score of -3 was different among physiographic zones (2005, H=9.379, p=0.009; 2006, H =11.548, p=0.003; 2007, H =10.778, p=0.005; 2008, H =10.490, p=0.005; 2009, H =8.781, p=0.012; Fig. 33). Post-hoc analyses indicated that the Lower watershed contained a significantly greater number of taxa with a weediness score of -3 than the Middle and/or Upper zones. It therefore appears that the number of weedy species and the number of very problematic species are greater in the urbanized Lower zone compared to the Upper and Middle zones, which is consistent with spatial differences in the proportion of native species in the watershed.

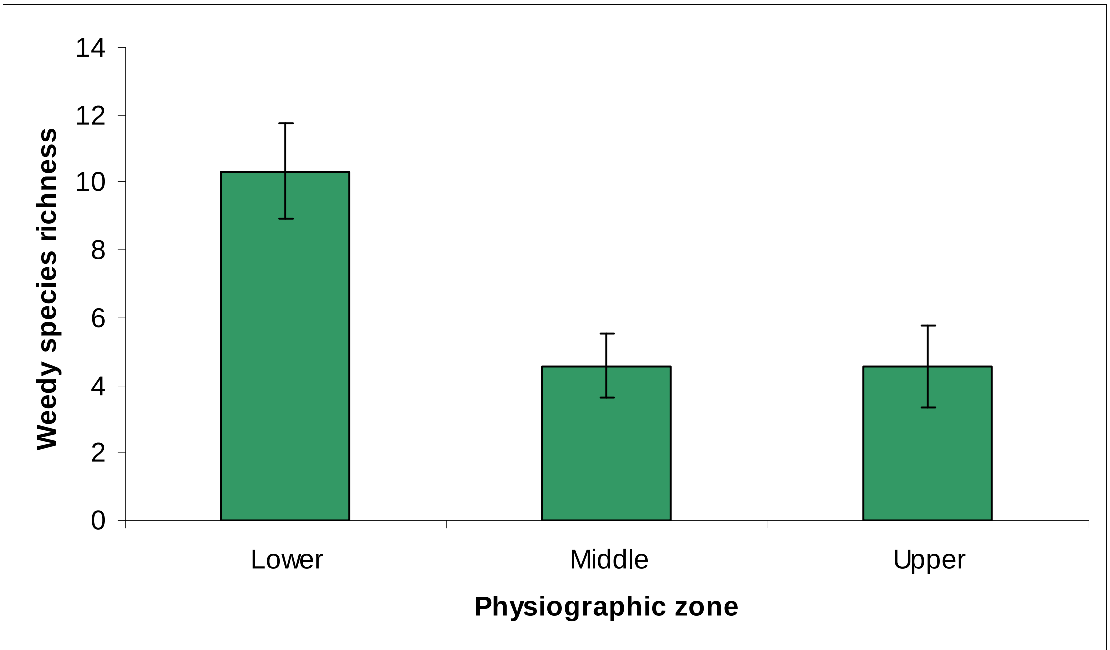


Figure 32. Mean weedy species richness across sites within each physiographic zone throughout the monitoring period in the Credit River Watershed. Error bars indicate +/- SE.

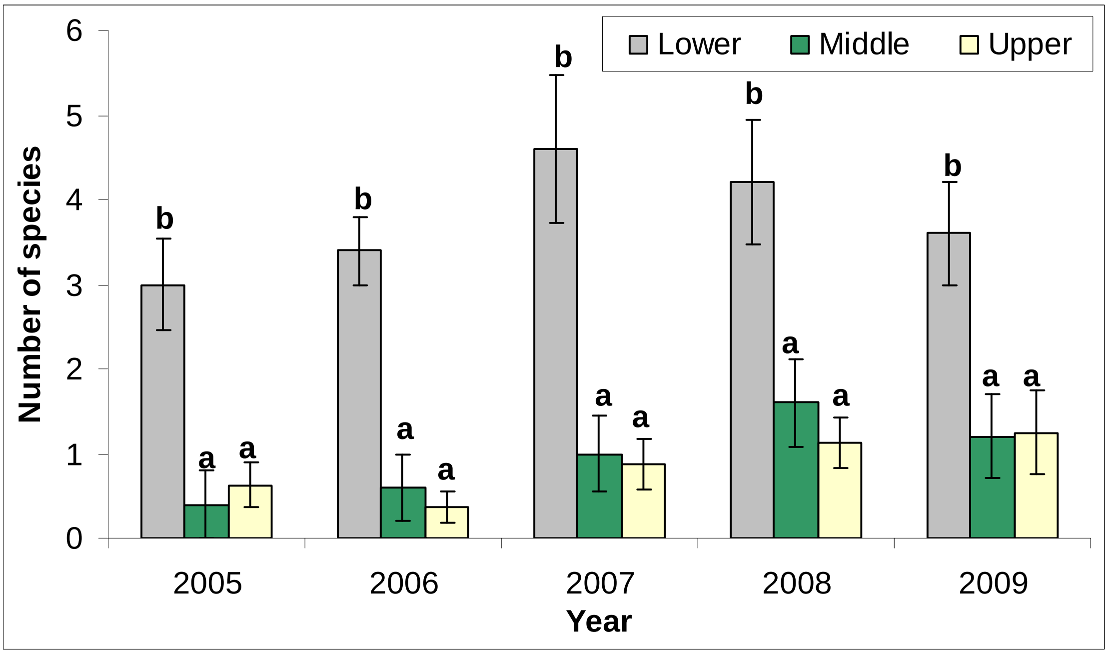


Figure 33. Mean number of species with a weediness score of -3 across sites within each physiographic zone throughout the Credit River Watershed. Within years, bars with different letters indicate a significant difference according to post-hoc analyses (p<0.05). Error bars indicate +/- SE.

3.3.8 Power Analysis

For all vegetation parameters analyzed with a Repeated Measures ANOVA analysis, power analysis determined that the following number of sites within the watershed would be needed in order to detect 1, 2 and 3 SD of change: 33, 12 and 9 sites, respectively. Additionally, the following number of sites within each zone would be needed in order to detect 1, 2 and 3 SD of change: 11, 4 and 3 sites, respectively. Therefore, a sufficient number of sites were monitored at both the watershed and physiographic zone scales to detect significant changes in the vegetation parameters with 90% power at an 80% confidence level for effect sizes greater than or equal to 2 SD.

3.4 LANDSCAPE ANALYSIS

3.4.1 Vegetation Parameters

Monitoring Question: Were wetland vegetation parameters correlated with landscape metrics in the Credit River watershed?

- Many parameters indicative of wetland health were positively correlated with habitat patch size, percent natural cover and matrix quality. The majority of the same parameters were negatively correlated with percent urban cover.
- No vegetation parameters were correlated with distance to the nearest road
- Only CC 8-10 was significantly negatively correlated to percent agricultural cover

Spearman rank correlation was used to examine relationships between wetland vegetation and landscape parameters in the Credit River Watershed. Significant correlations must be considered cautiously as they do not imply a cause-and-effect relationship between two parameters, but simply that the parameters are associated with one another (Zar 1999).

The proportion of native species (combined vegetation, ground vegetation and regeneration), native ground vegetation diversity and FQI were all positively correlated with habitat patch size (Table 21). These same parameters were also positively correlated to percent natural cover and matrix quality and negatively correlated to percent urban cover (Table 22). This is in agreement with research showing that fragmentation, leading to smaller wetland sizes, adversely affects both rare and common wetland vegetation species (Hooftman and Diemer 2002). Other research has indicated that the amount of landscape disturbance (combination of agriculture and urbanization) within a 2500 m radius of wetlands is related to a reduction in native graminoid and herbaceous perennial abundance (Galatowitsch et al. 2000). These native perennial species are found to be replaced by annuals or non-native perennials. Therefore, as habitat patch decreases, the amount of landscape disturbance increases and native populations decrease. In addition, weedy species richness and the number of individuals with a weediness score of -3 was negatively correlated to habitat patch size, percent natural cover and matrix quality and positively correlated to percent urban cover. This is consistent with current knowledge that weedy species increase in disturbed habitats (McKinney 2006). These results indicate that urbanization appears to be associated with wetland degradation, as many parameters indicative of healthy wetland vegetation decrease as natural areas become more fragmented.

It was surprising that species richness (combined vegetation and ground vegetation) was not significantly correlated with habitat patch size, percent natural cover and matrix quality. Previous studies have demonstrated a decrease in species richness in association with wetland degradation and an increase in richness with increasing wetland size and surrounding natural cover (Findlay and Houlahan 1997; Lougheed et al. 2008). The lack of association between richness and landscape parameters may be due an increase in non-native species richness in urban areas. However, a positive correlation was seen between matrix quality and regeneration richness.

Table 21. Spearman rank correlations between wetland parameters and habitat patch size and distance to the nearest road.

Vegetation Variable	Habitat Patch Size		Distance to Nearest Road	
	Spearman R	*P* value [a]	Spearman R	*P* value [a]
Combined Vegetation				
Total Species Richness	0.011	0.964	0.059	0.817
Proportion of Native Species Richness	**0.556**	**0.017**	0.350	0.155
Mean Coefficient of Conservatism	0.465	0.052	0.286	0.250
CC 8-10	0.307	0.216	0.086	0.736
Floristic Quality Index	**0.511**	**0.030**	0.216	0.399
Weedy Species Richness	**-0.558**	**0.016**	-0.447	0.063
Weediness Count -3	**-0.805**	**<0.001**	-0.168	0.504
Locally Rare Species	0.134	0.597	0.184	0.464
Regionally Rare Species	-0.072	0.776	0.343	0.164
Ground Vegetation				
Total Species Richness	0.331	0.179	0.142	0.576
Total Species Diversity	0.451	0.060	0.183	0.468
Total Species Evenness	0.082	0.748	-0.049	0.848
Proportion of Native Species Richness	**0.612**	**0.007**	0.387	0.113
Native Species Diversity	**0.472**	**0.048**	0.243	0.332
Native Species Evenness	-0.065	0.798	-0.170	0.499
Regeneration				
Total Species Richness	0.152	0.547	-0.058	0.819
Total Species Diversity	0.267	0.284	0.131	0.604
Total Species Evenness	-0.144	0.570	0.144	0.570
Proportion of Native Species Richness	**0.483**	**0.042**	0.137	0.589

[a] Results significant when $p<0.05$; significant values highlighted in bold.

Table 22. Spearman rank correlations between wetland parameters and selected landscape metrics.

Vegetation Variable	% Agriculture		% Natural		% Urban		Matrix Quality	
	Spearman R	*P* value [a]	Spearman R	*P* value [a]	Spearman R	*P* value [a]	Spearman R	*P* value [a]
Combined Vegetation								
Total Species Richness	0.256	0.305	0.089	0.699	-0.049	0.848	0.133	0.599
Proportion of Native Species	0.385	0.114	**0.622**	**0.006**	**-0.551**	**0.018**	**0.664**	**0.003**
mCC	-0.058	0.820	**0.496**	**0.036**	0.046	0.851	0.408	0.093
CC 8-10	**-0.508**	**0.031**	0.401	0.099	-0.207	0.896	0.327	0.185
Floristic Quality Index	0.263	0.291	**0.476**	**0.046**	**-0.483**	**0.043**	**0.560**	**0.016**
Weedy Species Richness	-0.297	0.232	**-0.645**	**0.004**	0.457	0.057	**-0.581**	**0.011**
Weediness Count -3	-0.263	0.291	**-0.756**	**<0.001**	**0.651**	**0.003**	**-0.758**	**<0.001**
Locally Rare Species	-0.230	0.360	0.007	0.977	0.037	0.883	0.014	0.958
Regionally Rare Species	-0.051	0.842	-0.232	0.355	0.122	0.630	-0.171	0.497
Ground Vegetation								
Total Species Richness	0.146	0.563	0.191	0.448	-0.302	0.224	0.306	0.217
Total Species Diversity	0.151	0.550	0.325	0.188	-0.293	0.239	0.395	0.105
Total Species Evenness	-0.127	0.615	0.271	0.276	-0.130	0.607	0.284	0.254
Proportion of Native Species	0.395	0.105	**0.618**	**0.006**	**-0.577**	**0.012**	**0.666**	**0.003**
Native Species Diversity	0.239	0.340	0.366	0.135	-0.386	0.114	0.459	0.055
Native Species Evenness	-0.315	0.203	0.073	0.773	0.072	0.775	0.055	0.829
Regeneration								
Total Species Richness	0.237	0.344	0.430	0.075	**-0.469**	**0.050**	**0.547**	**0.019**
Total Species Diversity	0.309	0.212	0.424	0.079	-0.317	0.120	0.445	0.064
Total Species Evenness	-0.051	0.842	**-0.606**	**0.008**	0.401	0.099	**-0.577**	**0.012**
Proportion of Native Species	0.220	0.380	**0.543**	**0.020**	-0.439	0.069	**0.529**	**0.024**

[a] Results significant when $p<0.05$; significant values highlighted in bold.

Local biodiversity may be negatively affected by roads through obstructing migration between local populations, altering wetland hydrology and siltation, increasing edge effects in habitat patches, facilitating the invasion of exotic species and increasing human access to wetlands (Findlay and Houlahan 1997). None of the studied vegetation parameters were correlated with the distance to the nearest road (Table 21). Although research has shown that the density of paved roads within 1 to 2 km of a wetland is negatively correlated with vegetative species richness (Findlay and Houlahan 1997), relationships may not have been noted in this study as the density of surrounding roads was not taken into account. The density of roads surrounding the wetland may be more indicative of the effects of roads on wetland species than the distance to a nearby road. For example, Terra Cotta Wetland is situated only 34 m from the nearest road (dirt road); however, it is located in the Middle watershed where natural land cover is more dominant than urban cover and it is located in a natural habitat patch of over 700 ha. In contrast, Meadowvale Wetland is located 155 m from the nearest road (paved city road); it is located in the Lower watershed, where urban land use is more dominant than natural cover, and is located in a natural habitat patch of only 42 ha. Although Terra Cotta is over four times closer to a road than Meadowvale, many parameters indicative of healthy wetland vegetation were consistently higher than Meadowvale. Therefore, distance to the nearest road may not be an effective indicator of the effects of roads on wetland vegetation because it does not account for the type of road and the amount of habitat fragmentation surrounding the monitoring site.

The number of species with a CC value between 8 and 10 was the only parameter found to be significantly correlated to percent agricultural cover (p=0.031, Table 22); this correlation was negative, indicating that as percent agriculture in the surrounding area increases, the number of high CC value species decreases. It is believed that agriculture may cause species loss due to edge effects from herbicide and fertilizer drift, alien plant invasions, and domestic livestock grazing (Galatowitsch et al. 2000). Therefore, these factors may be decreasing the suitability of habitat for highly conservative species.

3.4.2 Species Level Analysis

Monitoring Question: Was there a relationship between selected species and landscape metrics in the Credit River Watershed?

- Garlic Mustard and Common Buckthorn displayed significant negative logistic relationships with habitat patch size.
- None of the selected species displayed a significant relationship with the distance to the nearest road landscape metric.
- Both Purple Loosestrife and Common Buckthorn displayed a significant negative logistic relationship with percent natural cover.
- All three species displayed a significant positive logistic relationship with percent urban cover.

Logistic regression was used to test the relationship between the most weedy species in the watershed – Purple Loosestrife, Garlic Mustard and Common Buckthorn – and the following landscape metrics: habitat patch size, distance to the nearest road, percent natural cover and percent urban cover. There was no significant relationship between the presence of the three species and distance to the nearest road (Purple Loosestrife: X^2= 0.154, p=0.695; Common Buckthorn: X^2= 0.124, p=0.621; Garlic Mustard: X^2= 0.506, p=0.477). Although roads may facilitate the dispersal of invasive non-native species (Forman and Alexander 1998), the lack of relationship between selected species and distance to the nearest road was consistent with the insignificant correlations with vegetation parameters (Table 23). Examining the density of roads surrounding wetlands may provide more meaningful results.

Table 23. Logistic relationship between selected species and landscape metrics.

Species	Landscape Metric	X^2	p-value [a]
Garlic Mustard	Habitat patch size	**9.240**	**0.002**
	Distance to nearest road	0.506	0.477
	Percent natural cover	3.345	0.067
	Percent urban cover	**6.242**	**0.013**
Purple Loosestrife	Habitat patch size	3.347	0.067
	Distance to nearest road	0.154	0.695
	Percent natural cover	**9.377**	**0.002**
	Percent urban cover	**12.259**	**<0.001**
Common Buckthorn	Habitat patch size	**4.300**	**0.038**
	Distance to nearest road	0.124	0.621
	Percent natural cover	**4.233**	**0.040**
	Percent urban cover	**33.905**	**0.048**

[a] Results significant when p<0.05; significant values highlighted in bold.

Garlic mustard was significantly related to habitat patch size and percent urban cover (X^2= 9.240, p=0.002; X^2= 6.242, p=0.013, respectively; Table 23, Figure 34 & 35), However there was no significant relationship with percent natural cover (X^2= 3.345, p=0.067). It therefore appears that percent urban cover, not natural cover, is more critical to the presence of Garlic Mustard. According to the predicted probability of occurrence from the logistic regression, the probability of Garlic Mustard being found in a wetland plot is very high in small habitat patches, but decreases quickly with habitat patch size. In habitat patches of 5000 m^2 or less, the probability of Garlic Mustard occurring in wetlands is between 50-80% (Fig. 34). In habitat patches of over 25000 m^2 there is less than a 1% probability that Garlic Mustard will be found in wetlands. Similarly, there is less than 10% probability that Garlic Mustard will be present in wetlands with 5% or less urban cover in a 2 km radius, while wetlands situated in areas with greater than 70% urban cover in a 2 km radius have an 80% or greater chance of having Garlic Mustard (Fig. 35). It is important to note that habitat patch size is unequally represented by smaller areas and none of the monitored wetlands have above 80% urban cover in a 2km radius; therefore, extrapolation of occurrence probabilities should be made with caution.

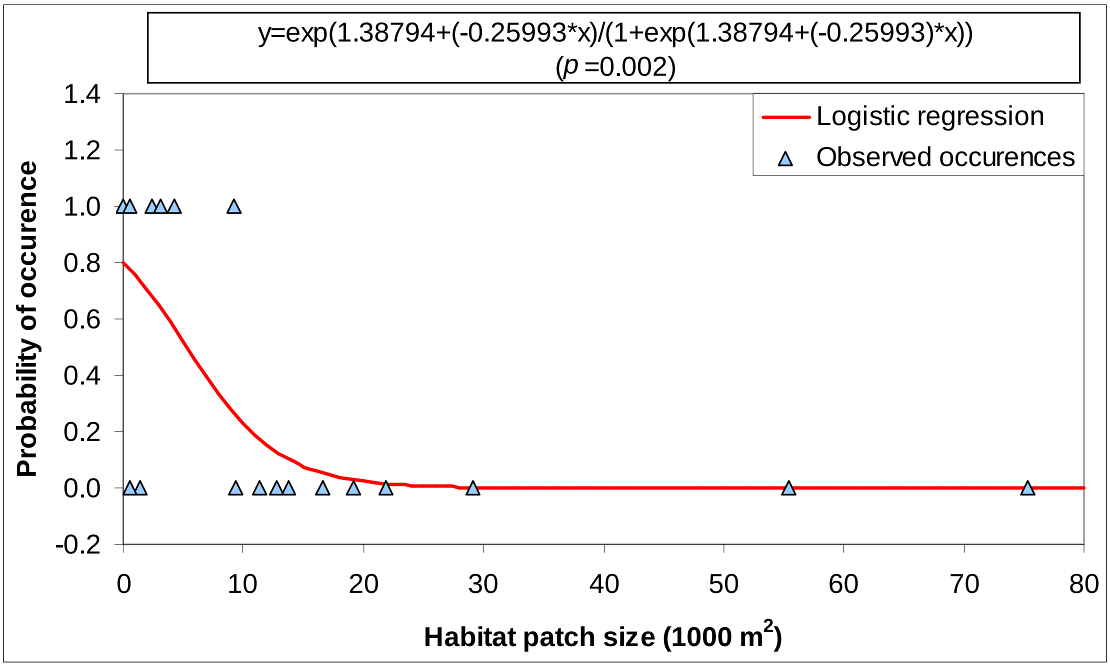


Figure 34. Logistic regression between Garlic Mustard (*Alliaria petiolata*) occurrence at a site and habitat patch size of the site. Equation of the regression line and p-value provided. Logistic regression significant when $p<0.05$.

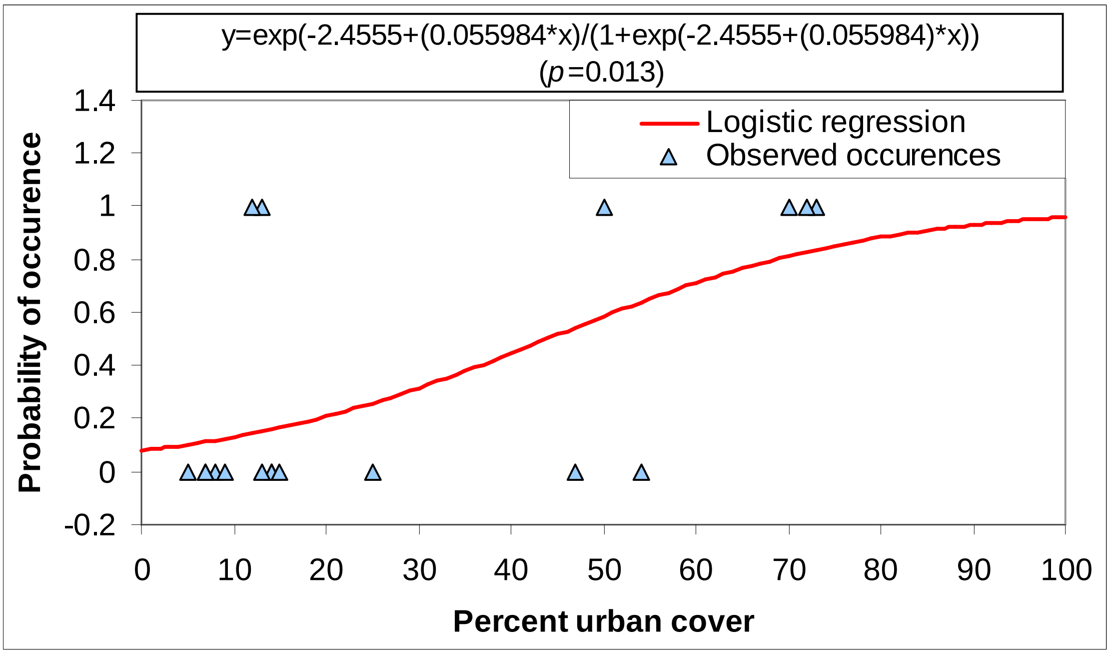


Figure 35. Logistic regression between Garlic Mustard (*Alliaria petiolata*) occurrence at a site and percent urban cover. Equation of the regression line and p-value provided. Logistic regression significant when $p<0.05$.

Although there was no significant relationship between Purple Loosestrife and habitat patch size (X^2= 3.347, p=0.067), Purple Loosestrife did exhibit significant relationships with both percent natural and urban cover (X^2= 9.377, p=0.002; X^2= 12.259, p<0.001, respectively; Table 23, Fig. 36 & 37). The occurrence of Purple Loosestrife at wetland monitoring plots decreased with increasing natural cover and increased with increasing urban cover. Monitoring sites with 50% or greater natural cover in a 2 km radius had less than a 10% probability of Purple Loosestrife being present; conversely, in areas with less than 15% urban cover the probability of occurrence was less than 20%. It appears that both urban and natural cover may be critical for the presence of Purple Loosestrife. This may not have been reflected in the relationship with habitat size because of the amount of agriculture in the area or because not enough site-years were available to detect a significant relationship (Table 23).

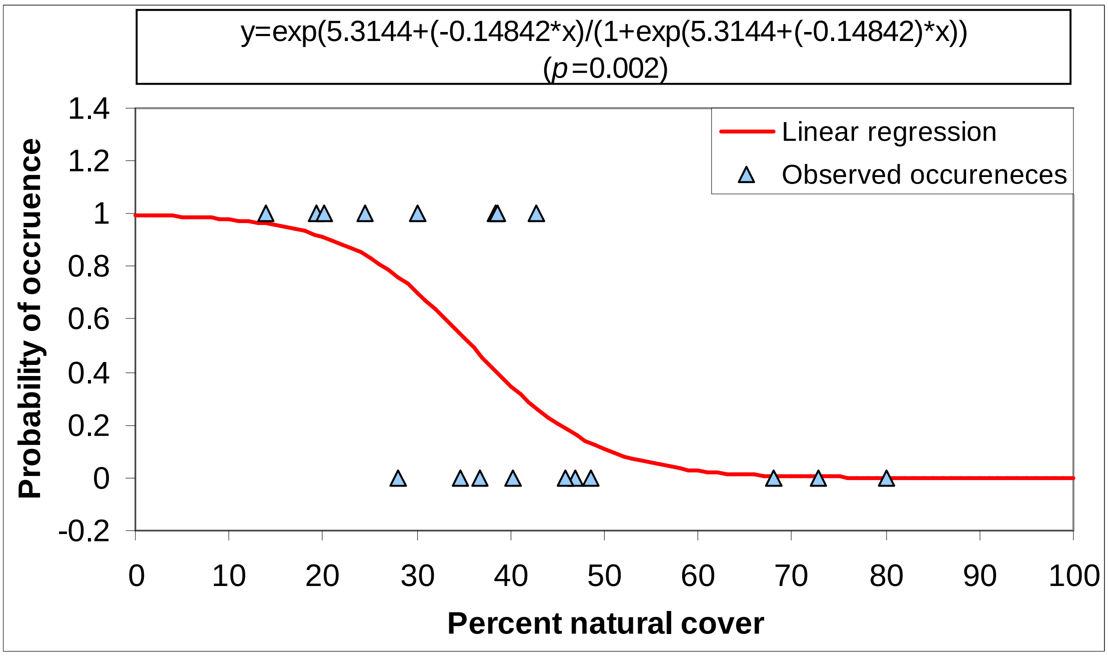


Figure 36. Logistic regression between Purple Loosestrife (*Lythrum salicaria*) occurrence at a site and percent natural cover. Equation of the regression line and p-value provided. Logistic regression significant when p<0.05.

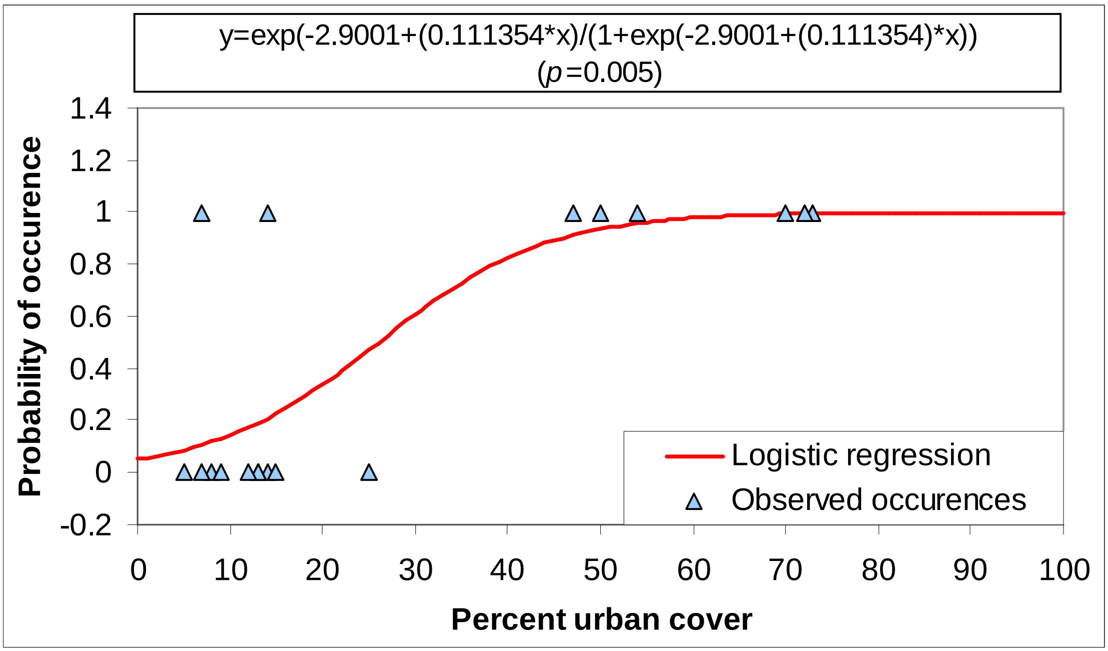


Figure 37. Logistic regression between Purple Loosestrife (*Lythrum salicaria*) occurrence at a site and percent urban cover. Equation of the regression line and p-value provided. Logistic regression significant when p<0.05.

Common Buckthorn was related to habitat patch size, percent natural cover and percent urban cover (X^2= 4.300, p=0.038; X^2= 4.233, p=0.040; X^2= 33.905, p=0.048, respectively; Fig. 38-40). In wetlands with greater than 70% natural cover in a 2 km radius, there was less than 5% probability of the occurrence of Common Buckthorn. Similarly, wetlands with greater than 15% urban cover in a 2 km radius had greater than 20% probability of the occurrence of Common Buckthorn and wetlands in a habitat patch of greater than 1000 m^2 had less than a 50% probability of the occurrence of Common Buckthorn.

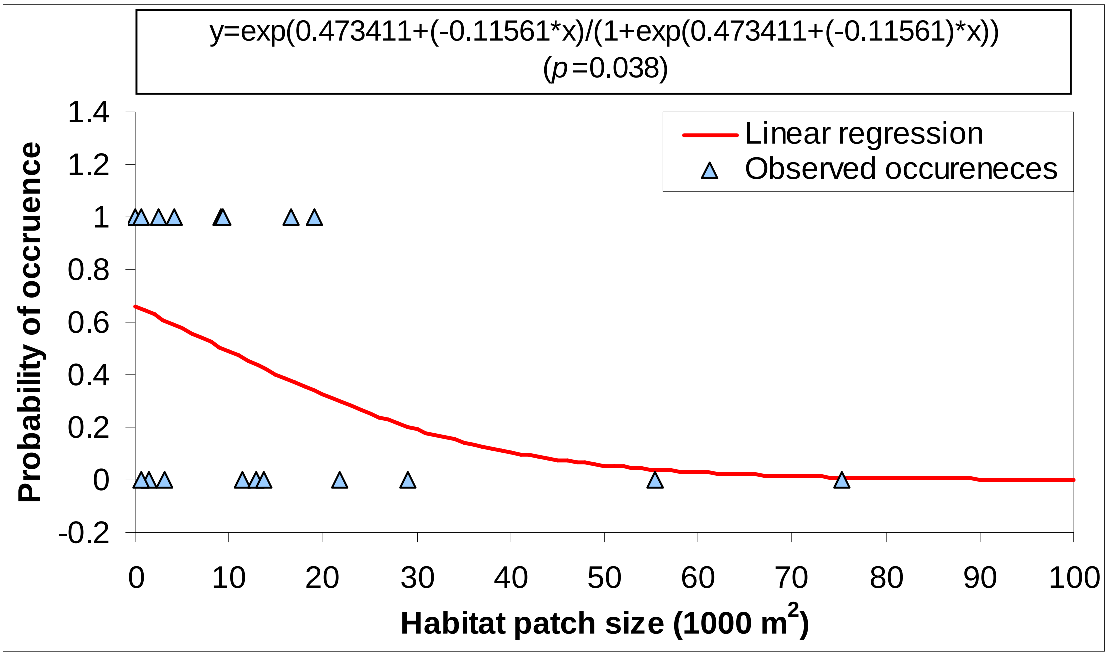


Figure 38. Logistic regression between Common Buckthorn (*Rhamnus cathartica*) occurrence at a site and habitat patch size. Equation of the regression line and p-value provided. Logistic regression significant when p<0.05.

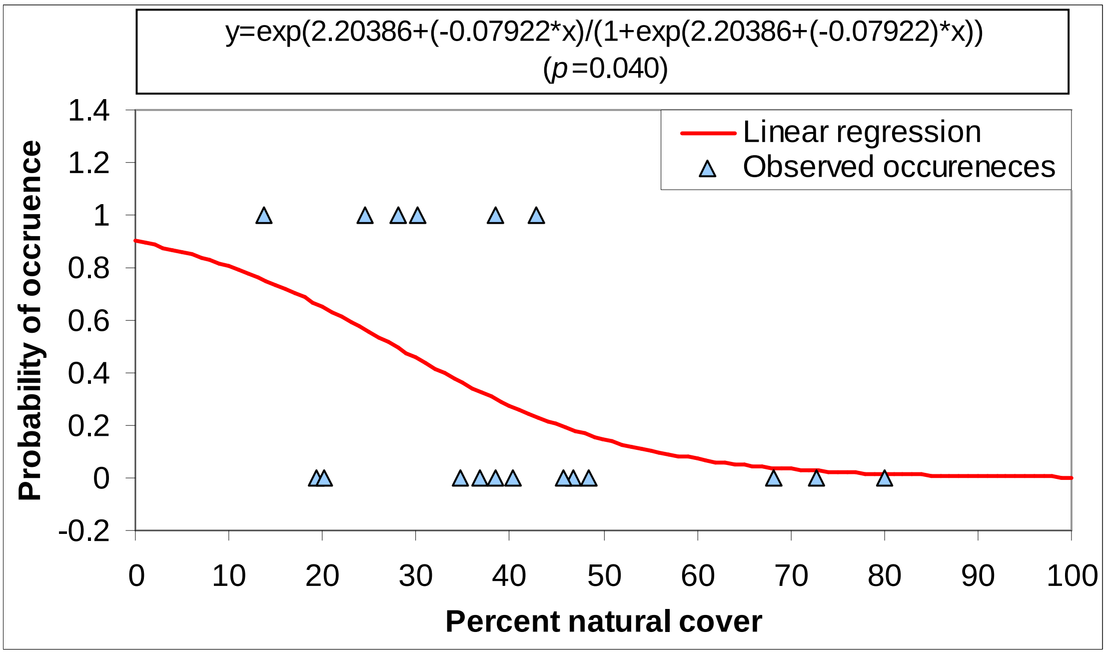


Figure 39. Logistic regression between Common Buckthorn (*Rhamnus cathartica*) occurrence at a site and percent natural cover. Equation of the regression line and p-value provided. Logistic regression significant when p<0.05.

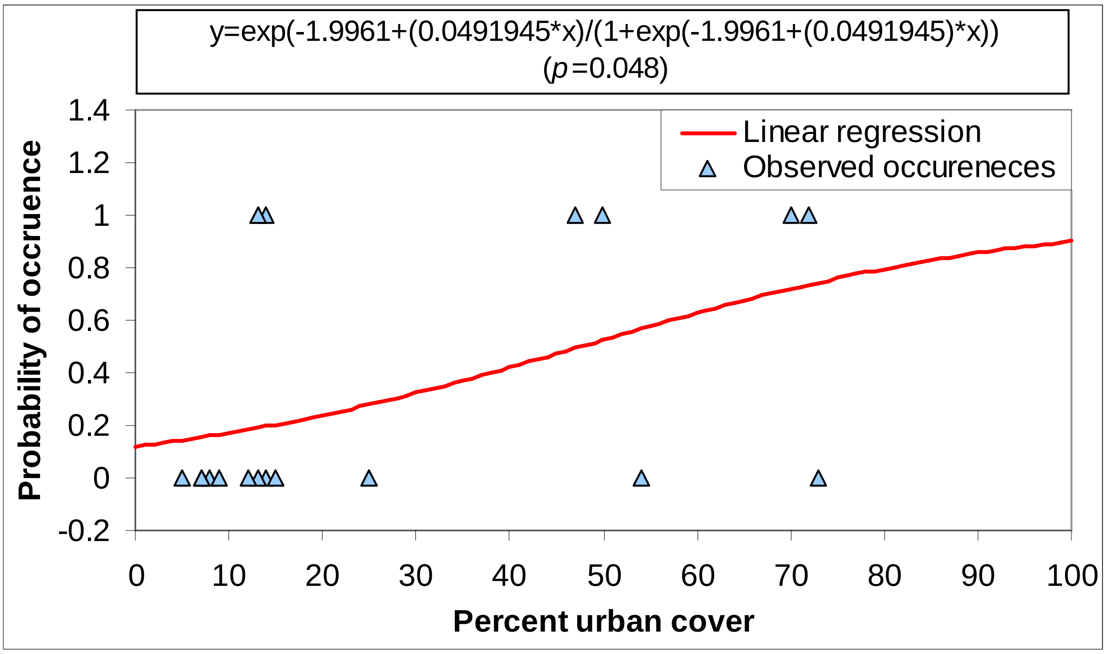


Figure 40. Logistic regression between Common Buckthorn (*Rhamnus cathartica*) occurrence at a site and percent urban cover. Equation of the regression line and p-value provided. Logistic regression significant when $p<0.05$.

3.4.3 Power Analysis

Table 24 lists the calculated number of monitoring sites required to determine a significant logistic relationship between the presence of the selected species and the landscape metrics. Analysis could not be performed for percent urban cover because of its bimodal distribution that could not be normalized. For relationships between Purple Loosestrife and Common Buckthorn and percent natural cover data quality was green. This indicates that there were sufficient site-years to detect a legitimate logistic regression between these species and percent natural cover. Although data quality for relationships between Garlic Mustard and percent natural cover was considered red, the current number of monitored wetland sites (18) is very close to N (19); and therefore, data quality may improve with additional years of data. For relationships between all three selected species and distance to the nearest road data quality was red. This indicates that there were insufficient site-years to detect a legitimate logistic regression. There were sufficient site-years to detect a legitimate regression between Garlic Mustard and habitat patch size; however this was not the case for Purple loosestrife and Common Buckthorn.

Table 24. The calculated number of monitoring sites (N) required to determine a significant logistic relationship between selected species and landscape metrics.

Species	Landscape Metric [a]	N	Data Quality
Garlic Mustard	Habitat patch size	9	Green
	Distance to nearest road	346	Red
	Percent natural cover	19	Red
	Percent urban cover	N/A	N/A
Purple Loosestrife	Habitat patch size	23	Red
	Distance to nearest road	3038	Red
	Percent natural cover	4	Green
	Percent urban cover	N/A	N/A
Common Buckthorn	Habitat patch size	21	Red
	Distance to nearest road	1670	Red
	Percent natural cover	15	Green
	Percent urban cover	N/A	N/A

[a] Landscape metrics were square root transformed for power analysis; analysis could not be performed for percent urban cover because of its bimodal distribution that could not be normalized.

4.0 CONCLUSIONS

Several wetland vegetation parameters were stable in the Credit River Watershed between 2005 and 2009. Although species richness and diversity were found to be increasing, these trends may be attributed to improved monitoring quality. As the monitoring program continues, it is expected that this bias will stabilize and it will be possible to detect biological changes in richness and diversity over time. Other wetland vegetation parameters including: mCC, FQI, weediness score -3, wetness, the number of locally and regionally rare species and CC 8-10, were either stable or fluctuated among years, but did not display an increasing or decreasing trend. Yearly fluctuations are to be expected, as weather and water levels vary over time (Wilcox et al. 2002; Touchette et al. 2008). It is interesting to note that the proportion of native species decreased and weedy species richness increased between 2005 and 2008.

There were also some notable differences among physiographic zones within the Credit River Watershed. The Lower watershed was different than the other physiographic zones for several vegetation parameters. In general, the Lower watershed contained a lower proportion of native species, higher weedy species richness and a greater number of -3 weedy species. These differences reflect the expected changes in vegetation that accompany increasing urbanization (McKinney 2006) and support the need for the current invasive species management program at CVC. The Middle and Upper watershed were not significantly different for any of the vegetation parameters examined.

Currently, wetland community types are not evenly distributed among physiographic zones. Monitoring sites consist of five marsh habitats in the Lower watershed, one marsh and four swamp habitats in the Middle watershed, and two swamps and six marsh habitats in the Upper watershed. Increasing the number of sites to include an equal distribution of marsh and swamp habitats will improve the program’s ability to detect spatial differences within the watershed.

Many of the parameters indicative of healthy wetland vegetation were significantly positively correlated with habitat patch size, percent natural cover and matrix quality. The majority of the same parameters were negatively correlated with percent urban cover. This indicated that as urbanization increases and habitat patch size and percent natural cover decreases, wetlands in the Credit River Watershed may be becoming more degraded. Parameters which are correlated with landscape metrics may be considered good monitoring variables, as they may be sensitive to anthropogenic disturbance (Noss 1999). Special attention should be paid to these vegetation parameters in the future, as increasing or decreasing trends may be an early warning sign for ecosystem degradation. All three selected species, highly invasive non-natives at wetland sites, showed some combination of significant positive relationships with percent urban cover or significant negative relationships with percent natural cover and/or habitat patch size. None of the studied vegetation parameters or species of interest displayed a significant relationship with the distance to the nearest road landscape metric. Road density may be a more appropriate metric for capturing the fragmenting effects of roads.

Overall, changes in wetland vegetation parameters between 2005 and 2009 in the Credit River Watershed appear to be linked to urbanization, and are most prominent in the Lower watershed. Deterioration of wetlands has the potential to become more severe as urbanization increases. This of concern, as a large proportion of wetlands in the watershed have already been altered or lost.

5.0 FUTURE DIRECTIONS

Several improvements and changes to the Terrestrial Monitoring Program are currently being evaluated, which will assist in future data collection and interpretation of results. Ecological Land Classification of all sites will be completed during the summer of 2010. These designations will provide insight to the effect of ecological classification on vegetation in the Credit River Watershed and assist the interpretation of temporal and spatial results. In addition, the Terrestrial Monitoring Program is examining the application of panel design to minimize the impact of monitoring on vegetation plots.

At the data analysis stage, several additional analyses will be explored that were beyond the scope of the five year review. Additional methods recommended by external reviewers (Bohdan Kowalyk, pers. comm.; Emily Gonzales, pers. comm.) will be evaluated, including generalized linear models and Bayesian analyses (Wade 2000) to examine vegetation trends over time. Several other metrics may be incorporated into the analyses as covariates, including: surveyor variation, soil moisture and pH, and canopy closure. In addition, vegetation parameters will be compared to other biotic and abiotic parameters such as soil chemistry and pH, birds and salamanders within forest communities. The possibility of monitoring wetland area will also be considered to determine if wetland area or wetland/terrestrial land boundaries are changing over the study period. A wetland hydrological monitoring program is also currently being developed, which will assist in drawing conclusions between vegetation changes and urbanization as well as other land use changes and climate change (Credit Valley Conservation 2010b).

Also, thresholds will be established for each monitoring parameter when at least five years of stable baseline data exists. Currently the Terrestrial Monitoring Program is developing a strategy for management actions to be undertaken when an indicator (or a suite of indicators) surpasses their established threshold, indicating a decline in the overall integrity of the Credit River Watershed. It should be noted that the baselines and thresholds presented in this report are preliminary in nature. The five year baseline period used to calculate indicator thresholds is based on the maximum number of years of reliable data currently available to the monitoring program. These current data represent the best available steady state information. Though indicators may naturally vary above and below these baselines over longer timeframes, we do not currently possess this information. Long term data may reveal additional patterns, such as wider than expected variability or cyclic behaviour oscillating over multiple years. As a result, established baselines and thresholds must be reviewed as additional monitoring data becomes available. To this end, a complete review of all indicator baselines and thresholds will be conducted at the 10 year point of the program.

The limitations of current baselines and thresholds should also be considered in the context of overall monitoring program design. This is because power analysis, a statistical assessment tool that can help inform decisions concerning program design, incorporates values associated with the identified baselines. This reliance on preliminary baselines, combined with the idiosyncrasies associated with individual power analysis techniques and the highly assimilative nature of most ecological indicators, preclude us from making program design decisions based solely on assessment of power. For instance, indicators with low abundance or high natural variability are more likely to provide data of lower power. Even if tests reveal a lack of power in the current program configuration, the data may still be providing valuable information. This is particularly the case when results from all monitoring questions are assessed in an integrated fashion. Hence, program design should be steered by a combination of power analysis, logistic considerations, and integrated ecological knowledge.

Finally, developing a management actions strategy will ensure that detrimental ecosystems changes observed by the Terrestrial Monitoring Program can be mitigated. Through this mitigation the Terrestrial Monitoring Program will assist CVC in achieving its corporate vision of “an environmentally healthy Credit River Watershed for present and future generations.”

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APPENDIX A: WETLAND MONITORING SITES

Table 1. Wetland Health Monitoring Sites established throughout the Credit River Watershed.

Site #	Site Name	Official Evaluated Wetland Name [b]	Township	County	Wetland Type	Physiographic zone	PSW, ESA and/or ANSI Designation [a]	Habitat patch size (ha)
W-01	Rattray Marsh Wetland	Rattray Marsh Wetland Complex	Mississauga	Regional Municipality of Peel	Marsh	Lower	PSW, ESA, ANSI	92.27
W-02	Credit River Wetland	Credit River Marshes Wetland Complex	Mississauga	Regional Municipality of Peel	Marsh	Lower	ESA, ANSI	0.26
W-03	Creditview Wetland	Creditview Wetland	Mississauga	Regional Municipality of Peel	Marsh	Lower	PSW, ANSI	5.38
W-04	Meadowvale Wetland	Churchville-Norval Wetland Complex	Mississauga	Regional Municipality of Peel	Marsh	Lower	ESA	42.29
W-12	Winston Churchill Wetland	Levi's Creek Wetland Complex	Halton Hills	Regional Municipality of Peel	Marsh	Lower		114.06
W-06	Hungry Hollow Wetland	Hungry Hollow Wetland	Halton Hills	Regional Municipality of Peel	Swamp	Middle	PSW, ESA	127.86
W-07	Acton Wetland	Acton - Silver Creek Wetland Complex	Halton Hills	Regional Municipality of Peel	Swamp	Middle	PSW, ESA	753.26
W-08	Terra Cotta Wetland	Caledon Lake Wetland Complex	Caledon	Regional Municipality of Peel	Swamp	Middle	ESA, ANSI	165.92
W-09	Ken Whillans Wetland	N/A	Caledon	Regional Municipality of Peel	Swamp	Middle		191.18
W-10	Warwick Wetland	Little Credit River Wetland Complex	Caledon	Regional Municipality of Peel	Marsh	Middle	PSW, ESA, ANSI	93.12
W-11	Erin Pine Estates Wetland	West Credit River	Erin	Wellington County	Marsh	Upper	PSW	24.66
W-13	Hillsburgh Wetland	Alton-Hillsburgh Wetland Complex	Erin	Wellington County	Marsh	Upper	PSW, ESA	14.37
W-15	Grange Orpen Wetland	Credit River at Alton Wetland Complex	Caledon	Regional Municipality of Peel	Swamp	Upper	PSW, ESA	137.59
W-16	Starr Wetland	Star Wetland Complex	Caledon	Regional Municipality of Peel	Marsh	Upper		291.34
W-17	Speersville Wetland	Speersville Wetland Complex	Caledon	Regional Municipality of Peel	Marsh	Upper	PSW, ESA	218.85
W-18	Caledon Lake Wetland	Caledon Lake Wetland Complex	Caledon	Regional Municipality of Peel	Swamp	Upper	PSW, ESA, ANSI	554.82
W-19	Melville Wetland	Orangeville Wetland Complex (MO3)	Orangeville	County of Dufferin	Marsh	Upper	PSW	5.53
W-20	Belfountain Wetland	N/A	Caledon	Regional Municipality of Peel	Marsh	Upper		30.86

[a] PSW, Provincially Significant Wetland; ESA, Environmentally Sensitive Area; ANSI, Area of Natural and Scientific Interest.
[b] Wetland plot apart of the indicated official evaluated wetland or wetland complex.

APPENDIX B: WETLAND VEGETATION PARAMETERS INCLUDED IN ANALYSIS

Table 1. Wetland vegetation parameters analyzed in the Credit River Watershed.

Vegetation Parameter	Normal Distribution	Data Transformation	Analysis Type [a]
Ground Vegetation			
Species Richness	Yes	No	Parametric
Total Species Diversity	Yes	No	Parametric
Total Species Evenness	N/A	No	Non-parametric
Native Species Diversity	Yes	No	Parametric
Native Species Evenness	N/A	No	Non-parametric
Proportion of Native Species	NA	No	Non-parametric
Regeneration [b]			
Species Richness	N/A	No	Non-parametric
Species Diversity	N/A	No	Non-parametric
Species Evenness	N/A		Non-parametric
Proportion of Native Species	N/A	No	Non-parametric
Combined Vegetation			
Species Richness	Yes	No	Parametric
Proportion of Native Species	N/A	No	Non-parametric
Mean Wetness Index	Yes	No	Parametric
mCC [c]	No	Log	Parametric
FQI [c]	Yes	Log	Parametric
CC 8-10	N/A	No	Non-parametric
Locally Rare	N/A	No	Non-parametric
Regionally Rare	N/A	No	Non-parametric
Weedy Species Richness	Yes	Log	Parametric
Weediness Count -3	N/A	No	Non-parametric

[a] Parametric analyses were Linear Regression to examine trends over time, and Repeated-Measures ANOVA to examine spatial differences between groups. Non-parametric tests were conducted for data which did not meet parametric test assumptions, and included the Friedman Test to examine temporal differences between years, and Kruskal-Wallis to examine differences over time.
[b] All regeneration data were analyzed using non-parametric methods due to spatial autocorrelation between the sites.
[c] mCC, Mean Coefficient of Conservatism; FQI, Floristic Quality Index.

APPENDIX C: RARE AND HIGHLY CONSERVATIVE SPECIES OBSERVED AT WETLAND MONITROING SITES

Table 1. Rare and highly conservative species observed at wetland monitoring sites in the Credit River Watershed.

Latin Name	Common Name	Locally Rare	Regionally Rare	CC Value	Number of Years Detected	Wetland Site
Andromeda polifolia ssp. glaucophylla	Bog Rosemary	X	X	10	5	Caledon Lake Wetland
Apios americana	American Groundnut	X	X	6	4	Credit River Wetland
Betula pumila	Dwarf Birch	X	X	9	4	Caledon Lake Wetland
Bolboschoenus fluviatilis	River Bulrush	X	X	7	4	Rattray Marsh Wetland Credit River Wetland
Campanula aparinoides	Marsh Bellflower	X	X	7	5	Erin Pine Estates Wetland
Cardamine pratensis	Cuckoo Flower	X	X	7	3	Rattray Marsh Wetland Credit River Wetland Erin Pine Estates Wetland
Carex albursina	White Bear Sedge	X		7	4	Credit River Wetland
Carex alopecoidea	Foxtail Sedge	X		6	4	Ken Whillans Wetland
Carex amphibola	Narrowleaf Sedge			9	5	Rattray Marsh Wetland Credit River Wetland
Carex aquatilis	Water Sedge	X	X	7	3	Belfountain Wetland Caledon Lake Wetland Starr Wetland Warwick Wetland
Carex atherodes	Awned Sedge	X	X	6	5	Hungry Hollow Wetland
Carex bromoides	Brome-like Sedge	X	X	7	2	Warwick Wetland
Carex comosa	Bristly Sedge	X		5	5	Creditview Wetland Hungry Hollow Wetland Warwick Wetland Hillsburgh Wetland
Carex flava	Yellow Sedge	X		5	5	Erin Pine Estates Wetland Hillsburgh Wetland
Carex leptalea	Bristly-stalk Sedge			8	5	Caledon Lake Wetland Grange Orpen Wetland Terra Cotta Wetland
Carex magellanica ssp. irrigua	Boreal Bog Sedge	X	X	10	4	Caledon Lake Wetland
Carex utriculata	Northwest Territory Sedge	X		7	5	Melville Wetland Belfountain Wetland

APPENDIX C: RARE AND HIGHLY CONSERVATIVE SPECIES OBSERVED AT WETLAND MONITORING SITES (Cont'd)

Latin Name	Common Name	Locally Rare	Regionally Rare	CC Value	Number of Years Detected	Wetland Site
Carex vesicaria	Inflated Sedge	X	X	7	5	Erin Pine Estates Wetland
Cephalanthus occidentalis	Buttonbush	X	X	7	2	Creditview Wetland
Ceratophyllum demersum	Common Hornwort	X	X	4	5	Acton Wetland Winston Churchill Wetland
Chamaedaphne calyculata	Leatherleaf	X	X	9	4	Caledon Lake Wetland
Cirsium discolor	Field Thistle			9	1	Belfountain Wetland
Cornus amomum ssp. obliqua	Silky Dogwood	X		5	4	Creditview Wetland Rattray Marsh Wetland Ken Whillans Wetland
Cuscuta gronovii	Gronovius Dodder	X		4	5	Credit River Wetland Warwick Wetland
Cypripedium parviflorum var. makasin	Small Yellow Lady's- slipper	X		7	3	Hungry Hollow Wetland
Drosera rotundifolia	Roundleaf Sundew	X	X	7	5	Caledon Lake Wetland
Elymus riparius	River Wild-rye	X	X	7	1	Hungry Hollow Wetland
Epilobium palustre	Marsh Willow-herb			10	2	Erin Pine Estates Wetland Speersville Wetland Caledon Lake Wetland
Galium aparine	Catchweed Bedstraw	X		4	2	Warwick Wetland Rattray Marsh Wetland
Impatiens pallida	Pale Touch-me-not	X		7	3	Meadowvale Wetland
Ledum groenlandicum	Labrador-tea	X	X	9	5	Caledon Lake Wetland
Lemna trisulca	Star Duckweed	X		4	3	Creditview Wetland
Linnaea borealis ssp. longiflora	Twinflower	X		7	3	Caledon Lake Wetland
Liparis loeselii	Loesel's Twayblade	X		5	2	Erin Pine Estates Wetland
Lonicera hirsuta	Hairy Honeysuckle	X	X	7	1	Caledon Lake Wetland
Lonicera oblongifolia	Swamp Fly-honeysuckle	X	X	8	1	Caledon Lake Wetland
Ludwigia palustris	Marsh Seedbox	X	X	5	4	Winston Churchill Wetland Melville Wetland

APPENDIX C: RARE AND HIGHLY CONSERVATIVE SPECIES OBSERVED AT WETLAND MONITORING SITES (Cont'd)

Latin Name	Common Name	Locally Rare	Regionally Rare	CC Value	Number of Years Detected	Wetland Site
Lysimachia thyrsiflora	Water Loosestrife	X		7	5	Hungry Hollow Wetland Warwick Wetland Erin Pine Estates Wetland Grange Orpen Wetland Starr Wetland Caledon Lake Wetland Melville Marsh Belfountain Wetland
Nemopanthus mucronatus	Mountain Holly	X	X	8	5	Caledon Lake Wetland
Orthilia secunda	One-side Wintergreen	X	X	5	4	Caledon Lake Wetland
Picea glauca	White Spruce	X		6	5	Caledon Lake Wetland Warwick Wetland Grange Orpen Wetland
Potentilla palustris	Marsh Cinquefoil	X		7	5	Caledon Lake Wetland
Rhamnus alnifolia	Alder-leaved Buckthorn	X		7	5	Credit River Wetland Acton Wetland Erin Pine Estates Wetland Terra Cotta Wetland Ken Whillans Wetland Caledon Lake Wetland Melville Wetland
Rosa palustris	Swamp Rose	X	X	7	5	Creditview Wetland Warwick Wetland
Rumex orbiculatus	Water Dock	X		6	5	Warwick Wetland Erin Pine Estates Wetland Grange Orpen Wetland Starr Wetland
Salix amygdaloides	Peach-leaved Willow	X			2	Warwick Wetland
Salix candida	Hoary Willow	X	X	10	5	Caledon Lake Wetland
Salix exigua	Sandbar Willow	X		3	5	Credit River Wetland
Salix lucida	Shining Willow	X		5	5	Caledon Lake Wetland
Salix serissima	Autumn Willow	X	X	6	2	Caledon Lake Wetland

APPENDIX C: RARE AND HIGHLY CONSERVATIVE SPECIES OBSERVED AT WETLAND MONITORING SITES (Cont'd)

Latin Name	Common Name	Locally Rare	Regionally Rare	CC Value	Number of Years Detected	Wetland Site
Sarracenia purpurea	Northern Pitcher-plant	X	X	10	5	Caledon Lake Wetland
Sparganium eurycarpum	Large Bur-reed	X		3	5	Creditview Wetland Warwick Wetland Melville Wetland
Stellaria longifolia	Longleaf Starwort	X	X	2	2	Warwick Wetland Starr Wetland Grange Orpen Wetland
Symphyotrichum pilosum var. pilosum	White Heath Aster variety	X	X	4	4	Grange Orpen Wetland Hungry Hollow Wetland Starr Wetland
Utricularia vulgaris	Greater Bladderwort	X		4	4	Speersville Wetland
Vaccinium angustifolium	Late Lowbush Cranberry		X	6	5	Caledon Lake Wetland
Vaccinium macrocarpon	Large Cranberry	X	X	10	2	Caledon Lake Wetland
Vaccinium oxycoccos	Small Cranberry	X	X	10	5	Caledon Lake Wetland
Viola renifolia	Kidney-leaf White Violet	X		7	2	Speersville Wetland Caledon Lake Wetland

APPENDIX D: YEAR OF FIRST DETECTION OF NON-NATIVE WETLAND SPECIES BY THE TERRESTRIAL MONITORING PROGRAM

Table 1. Year of first detection of non-native wetland species in the Credit River Watershed by the Terrestrial Monitoring Program

Latin Name	Common Name	Year of First Detection by Monitoring Program [a]				
		2005	2006	2007	2008	2009
Acer negundo	Manitoba Maple	X				
Acer platanoides	Norway Maple		X			
Achillea millefolium ssp. millefolium	Common Yarrow		X			
Alliaria petiolata	Garlic Mustard	X				
Arctium minus ssp. minus	Common Burdock	X				
Artemisia vulgaris	Common Wormwood			X		
Barbarea vulgaris	Yellow Rocket				X	
Capsella bursa-pastoris	Common Shepherd's Purse			X		
Cerastium fontanum	Common Mouse-ear Chickweed					X
Chelidonium majus	Greater Celadine	X				
Chenopodium album var. album	Lambsquarters			X		
Cirsium arvense	Creeping Thistle	X				
Cirsium vulgare	Bull Thistle		X			
Cynoglossum officinale	Common Hound's-tongue				X	
Dactylis glomerata	Orchard Grass	X				
Daucus carota	Queen Anne's Lace	X				
Elymus repens	Quackgrass			X		
Epilobium hirsutum	Great-hairy Willow-herb				X	
Epilobium parviflorum	Small-flower Willow-herb				X	
Epipactis helleborine	Eastern Helleborine	X				
Geranium robertianum	Herb-robert	X				
Glechoma hederacea	Ground Ivy				X	
Glyceria maxima	Reed Manna-grass		X			
Hesperis matronalis	Dame's Rocket	X				
Hieracium caespitosum	Yellow Hawkweed	X				
Hypericum perforatum	Common St. John's-wort	X				
Impatiens glandulifera	Ornamental Jewelweed	X				
Inula helenium	Elecampane Flower	X				
Leonurus cardiaca ssp. cardiaca	Common Motherwort				X	
Lithospermum officinale	European Gromwell	X				
Lonicera tatarica	Tartarian Honeysuckle	X				
Lotus corniculatus	Birds-foot Trefoil	X				
Lycopus europaeus	European Bugleweed	X				
Lysimachia nummularia	Creeping Jennie		X			
Lythrum salicaria	Purple Loosestrife	X				
Medicago lupulina	Black Medic	X				
Melilotus alba	White Sweet Clover		X			
Melitotus officinalis	Yellow Sweet Clover			X		
Mentha x piperita	Peppermint		X			
Myosotis scorpioides	True Forget-me-not	X				
Nasturtium officinale	True Watercress	X				
Nepeta cataria	Catnip		X			

APPENDIX D: YEAR OF FIRST DETECTION OF NON-NATIVE WETLAND SPECIES BY THE TERRESTRIAL MONITORING PROGRAM (Cont'd)

Common Name	Latin Name	Year of First Detection by Monitoring Program [a]				
		2005	2006	2007	2008	2009
Phleum pratense	Meadow Timothy			X		
Plantago major	Common Plantain			X		
Poa compressa	Canada Bluegrass				X	
Poa trivialis	Rough Bluegrass	X				
Polygonum hydropiper	Marshpepper Smartweed				X	
Potamogeton crispus	Curly Pondweed	X				
Ranunculus acris	Tall Buttercup	X				
Rhamnus cathartica	Common Buckthorn	X				
Ribes rubrum	Garden Red Currant				X	
Rumex crispus	Curly Dock	X				
Salix purpurea	Purpleosier Willow					
Silene vulgaris	Bladder Campion	X				
Solanum dulcamara	Climbing Nightshade	X				
Sonchus arvensis ssp. arvensis	Perennial Sowthistle	X				
Sonchus oleraceus	Common Sowthistle				X	
Sorbus aucuparia	European Mountain-ash	X				
Stellaria graminea	Little Starwort				X	
Taraxacum officinale	Brown-seed Dandelion	X				
Trifolium repens	White Clover	X				
Tussilago farfara	Colt's Foot	X				
Typha angustifolia	Narrow-leaved Cattail	X				
Typha x glauca	White Cattail	X				
Urtica dioica ssp. dioica	Stinging Nettle	X				
Veronica anagallis-aquatica	Water Speedwell			X		
Veronica officinalis	Gypsy-weed				X	
Viburnum opulus	Guelder-rose Viburnum	X				
Vicia cracca	Tufted Vetch	X				

[a] Indicates the year that non-native species were first observed in monitoring plots, not the year that non-natives were first detected within the watershed. 2005 was the year of first detection for many species as it was the first year of the monitoring program.

Credit Valley Conservation

Terrestrial Monitoring Program Report

2002 - 2009

Monitoring Wetland Integrity within the Credit River Watershed

Chapter 4: Credit River Watershed Landscape Analysis

CVC

CREDIT VALLEY CONSERVATION

CREDIT RIVER WATERSHED LANDSCAPE ANALYSIS

Landscape monitoring of the Credit River watershed is completed every five years. This analysis is complementary to the wetland community and species level monitoring, presented in the preceding chapters. Landscape monitoring is conducted at a broad scale, examining the status and trends in natural, semi-natural and non-natural communities across the watershed. Therefore, this analysis was completed as a separate exercise. However, the assessment of wetland integrity should be considered in the context of the rapidly changing landscape of the Credit River watershed. For details, please see:

Credit Valley Conservation. 2011. Landscape monitoring of terrestrial ecosystems in the Credit River watershed. Credit Valley Conservation.