Terrestrial Long Term Monitoring

Spatial and Temporal Trends 2008-2014

August, 2015

Toronto and Region Conservation for The Living City

Report prepared by: Lyndsay Cartwright, Data Analyst

Reviewed by: Sue Hayes, Project Manager, Terrestrial Field Inventories
Scott Jarvie, Associate Director, Environmental Monitoring and Data Management Section

This report may be referenced as:

Toronto and Region Conservation Authority (TRCA). 2015. Terrestrial Long Term Monitoring: Spatial and Temporal Trends 2008-2014.

Table of Contents

page

List of Tables

List of Figures

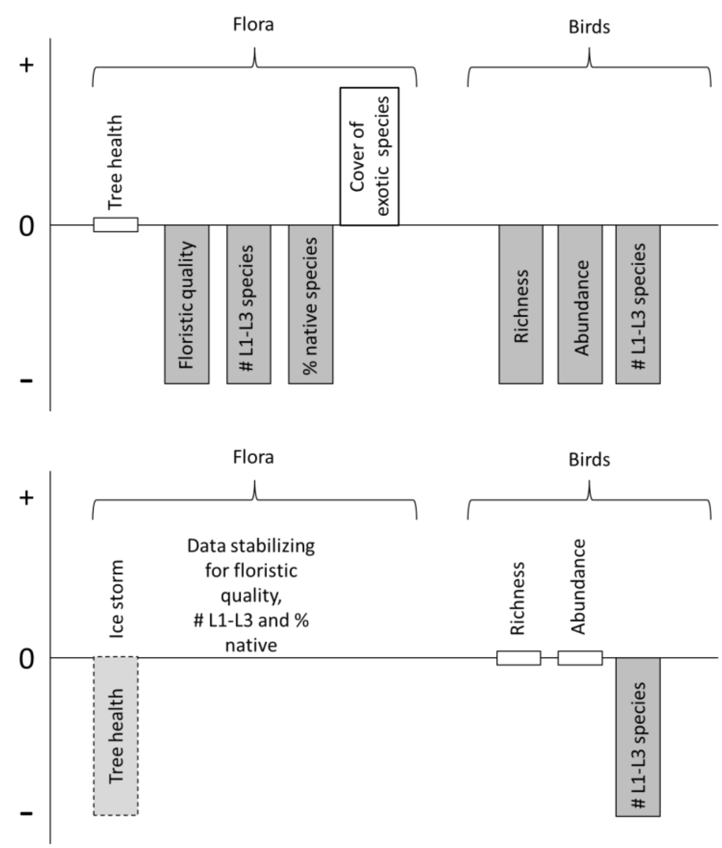
Executive Summary

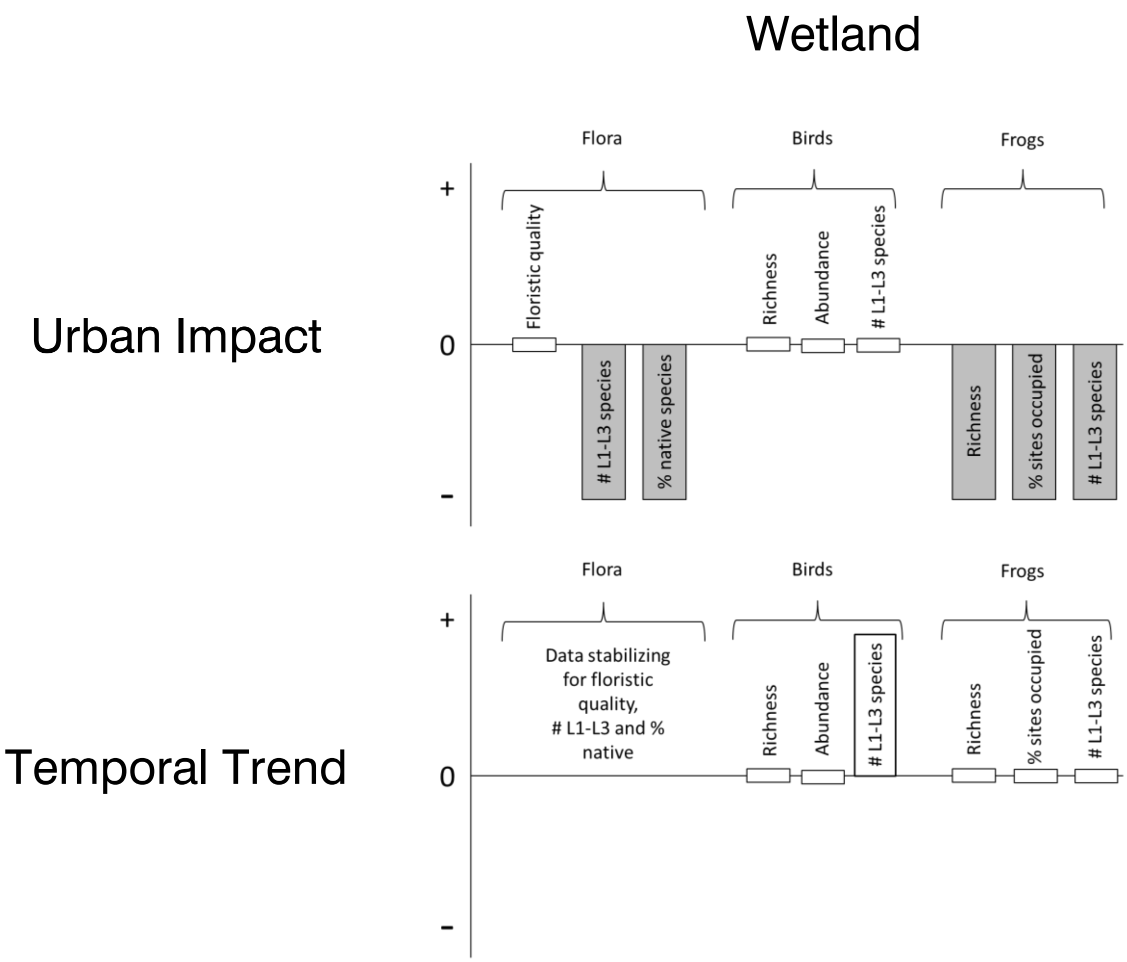
The purpose of TRCA's Terrestrial Long-Term Monitoring Program is to detect spatial and temporal trends in vegetation, breeding bird and amphibian communities within the TRCA jurisdiction. The program has been collecting data in forests, wetlands and meadows since 2008 and this report summarizes the results from 2008 to 2014. The goal of detecting spatial trends is to determine the impact of urbanization on flora and fauna communities by comparing natural areas in the rural and urban land use zones. Temporal trends were analyzed between 2008 and 2014 to determine if flora and fauna communities are increasing or decreasing over time. Below are several visual schematics to represent the findings of this report for forests, wetlands and meadows. The impact of urbanization and temporal trends are shown for each high-level indicator. If there was a negative impact of urbanization, a grey bar is displayed towards the minus sign. If there was a positive impact, a white bar is displayed towards the plus sign. No effect is indicated with a thin white bar along the zero line. For temporal trends, a grey bar toward the negative sign means a decrease over time and a white bar toward the plus sign means an increase over time. Species ranked L1-L3 are specialist, sensitive and possibly rare species in the region, richness is the number of species, and abundance is the number of individuals.

Forest

Urban Impact

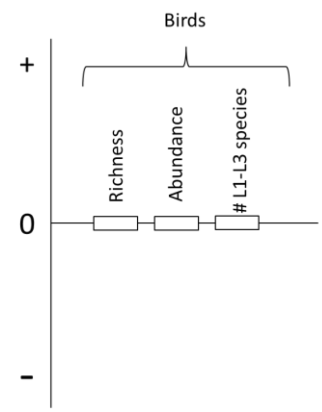
Temporal Trend



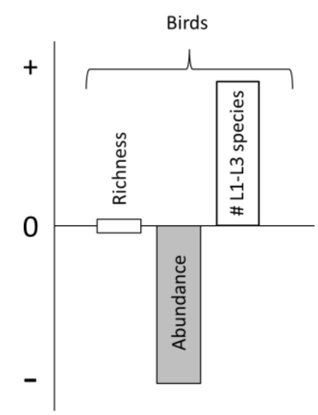

Wetland
Urban Impact
Temporal Trend
Flora
Birds
Frogs
+
0
-
Floristic quality
L1-L3 species
% native species
Richness
Abundance
L1-L3 species
Richness
% sites occupied
# L1-L3 species
Data stabilizing for floristic quality, # L1-L3 and % native

Meadow

Urban Impact



Temporal Trend



This report shows that while many of the high-level indicators are stable over time, almost all indicators are showing impairment due to urbanization. If the jurisdiction foresees the conversion of more rural land to urban land uses, we will expect to see drastic declines in both flora and fauna communities in the years to come. Monitoring should continue in both the urban and rural land use zones and should also continue to track the temporal impacts of development on natural areas that are currently changing from rural to urban land uses. In addition, more research needs to be done on development methods and land use planning that maintain habitat for sensitive species while still meeting the demands of a growing population to ensure the health and persistence of natural areas in TRCA's watershed.

1.0 Introduction

The Toronto and Region Conservation Authority (TRCA) has developed and implemented a long-term Regional Watershed Monitoring Program that is designed to assess the health of the Region's watersheds and natural heritage features. In 2008, this program was augmented with the addition of a number of terrestrial long-term fixed plots. The long-term monitoring plots represent an addition to our other projects: the systematic natural heritage inventory and assessment information that maps comprehensive vegetation community, flora and fauna species data across the landscape, which began in the late 1990s (TRCA 2007); and the terrestrial volunteer monitoring program (started in 2002) that focuses on occurrences of a limited number of indicator species.

Toronto and Region Conservation Authority staff established forest vegetation and forest bird fixed plots across the jurisdiction beginning in 2008. In 2009, additional regional plots were set-up to monitor wetland vegetation, wetland birds, frogs, Plethodontid salamanders and meadow birds (Figure 1). Plots were placed in forest, wetland and meadow habitat types using the TRCA's Long Term Monitoring Program (LTMP) protocols (TRCA 2011a-g). In contrast to the systematic natural heritage inventory, which provides a one-time picture of the flora and fauna, the purpose of the LTMP is to detect regional spatial and temporal trends in the vegetation, breeding bird, amphibian, and Plethodontid salamander communities. Through the use of standardized scientific data collection protocols, the response of the terrestrial system to various landscape changes such as increased natural cover through reforestation efforts or to increased use of the natural area due to recent nearby urbanization can be quantitatively documented. The assessment of changes in these natural systems can then be used to better guide management actions on site with the aim of improving overall biodiversity.

There are now seven years of data that have been collected in the terrestrial portion of the Regional Watershed Monitoring Program. The purpose of this report is to 1) summarize spatial trends by comparing selected biodiversity indicators between urban and rural land use zones, 2) summarize temporal trends in selected biodiversity indicators between 2008 and 2014, and 3) provide an update on the overall health of TRCA's wetland, forest and meadow communities based on these analyses.

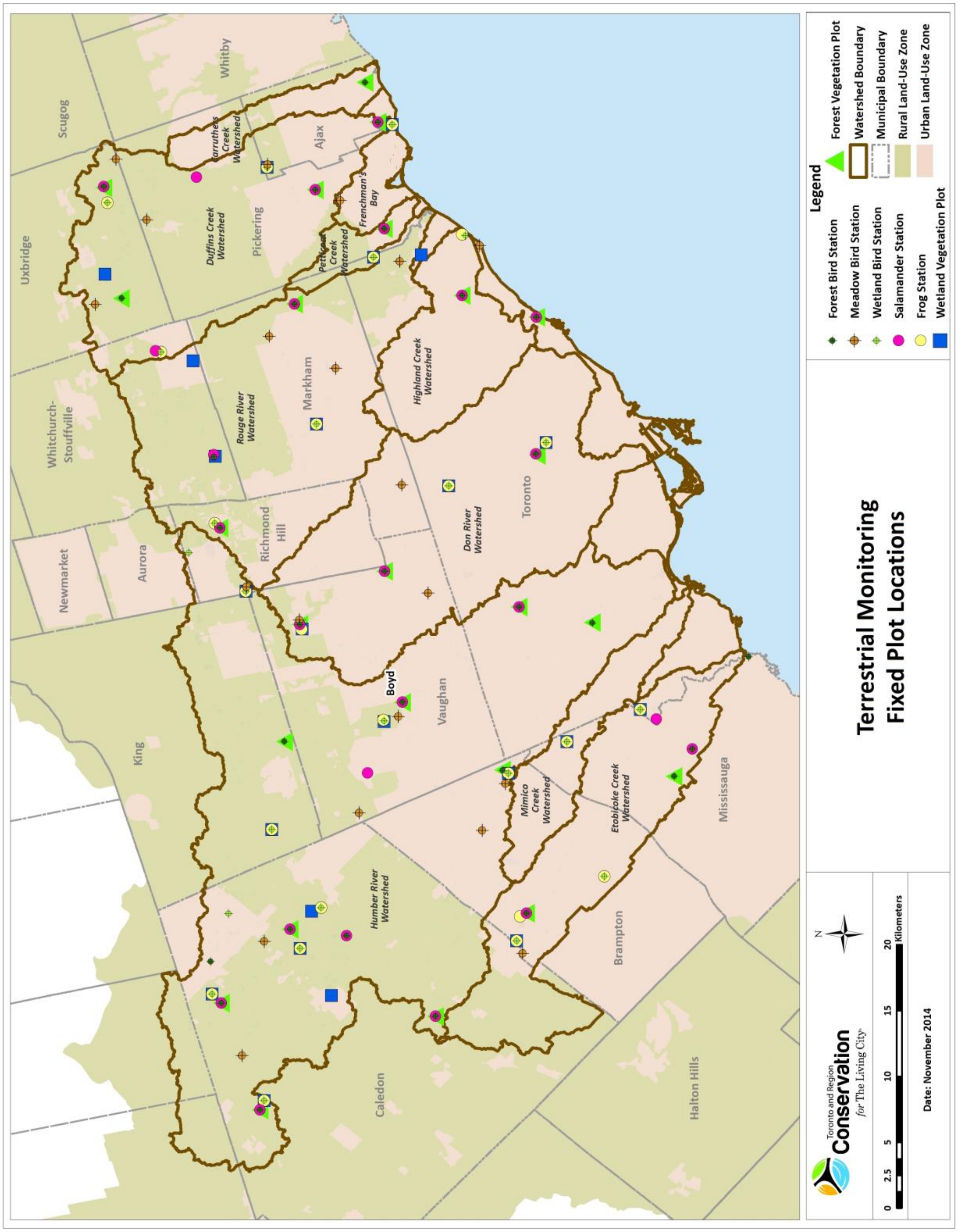


Figure 1. Terrestrial monitoring plots in the TRCA jurisdiction, 2014

2.0 Methodology

The monitoring methodology employed by TRCA is very closely based on that which is used by Environment Canada in its Ecological Monitoring and Assessment Network (EMAN) and the Credit Valley Conservation Authority (CVC) (EMAN 2004a, EMAN 2004b, CVC 2010a). For the full monitoring methodology used by TRCA for its forest, wetland, and meadow stations refer to TRCA (2011a-g).

For the purposes of this report, the TRCA jurisdiction has been divided into rural and urban zones, based on 2013 existing land use patterns observed using aerial ortho-photos. Sites were classed as urban or rural in an initial step where the number of hectares designated as greenbelt area was determined. If this amount was greater than 50% of the total area within a 2km radius buffer of the plot/station then the station was classified as rural. If this amount was less than 50% of the total area within a 2km radius buffer then the station was classified as urban. If two stations in the same site had opposite land use designations, an average of the total hectares of greenbelt area was used. If this average was >50%, the site was designated as rural and if this average was <50% the site was designated as urban. This was a rare occurrence and only applied to one site for meadow bird surveys (ORMCP,MB-8). Since there were so few "urbanizing" sites they could not be included as their own category for statistical analysis and therefore land use analysis will be limited to urban and rural designations. A summary of long-term monitoring plots found in each land use zone if shown in Table 1.

Table 1. Summary of the number of long-term monitoring plots located in the rural and urban land use zones

Plot type	Rural Land Use Zone	Urban Land Use Zone
Forest vegetation	11	13
Forest birds	14	15
Plethodontid salamanders	11	12
Wetland vegetation	11	9
Wetland birds	10	12
Frogs	11	10
Meadow birds	10	10

2.1 Selection of Site Quality Indicators

Long-term monitoring plots were established to identify the health and condition of key biological communities (i.e. vegetation, bird, frog, plethodontid) associated with forest, wetland and meadow habitat features and to track changes in their condition over time. Ecosystem health can be measured with various indicators, including tree health, flora and fauna species richness, the representation of native versus exotic species, and the presence and abundance of sensitive species (those of conservation concern). Objectives based on such indicators, specific to each habitat type, are outlined below.

Forest monitoring plots were designed to:

- Determine the health of forests in the TRCA jurisdiction
- Determine regeneration rate and species composition of understorey saplings and shrubs
- Determine if the population and abundance of flora species, including those of conservation concern, are changing over time
- Determine the floristic quality of the site
- Determine the rate of spread of selected invasive species
- Determine if non-native invasive species are replacing native species
- Facilitate identification of any regional trends in the status of forest-associated bird species, and in particular to identify any changes in the proportions of variously ranked suites of species present at forest sites in both rural and urban zones.
- Track region-wide changes in the status of Plethodontids

Wetland monitoring plots were designed to:

- Determine the health of wetlands in the TRCA jurisdiction
- Determine if the population and abundance of flora and fauna species, including those of conservation concern, are changing over time
- Determine the floristic quality of the site
- Determine the rate of spread of selected invasive species
- Determine if non-native invasive species are replacing native species

Meadow monitoring plots were designed to:

- Assess overall trends in meadow bird species richness and abundance in the TRCA region

Indicators were selected in accordance with these monitoring objectives prior to plot set-up. Table 2 provides an overview of the indicators chosen to interpret site quality.

Table 2. List of monitoring high-level indicators chosen for the long-term monitoring program

Habitat type	Monitoring indicator(s)	Description
Forest	Tree health	Proportion healthy trees
	Mean floristic quality index (FQI)	Proportion of habitat sensitive species
	Flora species richness	Number of plant species
	Flora species abundance	Proportion of different L-ranked species
	Bird species richness	Presence of forest guild species
		Proportion of different L-ranked species
	Plethodontid abundance	Count of red-backed salamanders
Wetland	Mean floristic quality index (FQI)	Proportion of habitat sensitive species
	Flora species richness	Number of plant species
	Flora species abundance	Proportions of different L-ranked species
	Bird species richness	Presence of wetland guild species
		Proportions of different L-ranked species
	Amphibian species richness	Presence of frog and toad species
Meadow	Bird species richness	Presence of meadow guild species
		Proportions of different L-ranked species

The assessment of tree health provides a wealth of information on the condition and resilience of forest communities. Variables such as tree mortality and crown vigour are measures of tree health that are standard monitoring variables used throughout the world. While there is a long history of assessing tree health, the measurement and interpretation of species richness and biodiversity are a more recent development and some clarification is provided here.

Species richness (i.e. the number of different species) and the relative dominance of native or exotic species are important indicators of ecosystem health. A closer look at the native flora and fauna present at any given site reveals that they vary in their degrees of tolerance to disturbance. Some are indicators of high-quality remnant habitat, thus of successful preservation or restoration efforts. They are of greater regional conservation concern. Others occur in a wide range of disturbed habitats. Various methods of assessment can be used to interpret any observed changes in composition of plants or animals. Toronto and Region Conservation Authority has developed a local ranking system for flora and fauna species; this ranking system was designed to reflect the ability of each species to thrive in the changing landscape of the Toronto region. The ranks range from the extremely sensitive species (L1) to the largely urban tolerant species (L5), with an additional L-rank for exotic (non-native) species (L+). Ranks are reviewed annually and subject to updates (TRCA 2010). Species with ranks of L1 to L3 are considered to be of concern throughout the TRCA jurisdiction, while those ranked L4 are of intermediate sensitivity and are of conservation concern within urban and suburban landscapes only.

An additional ranking system for plants, the coefficient of conservatism (CC) was used for calculating Floristic Quality Index (FQI) of the plots. The CC is assigned to native plants and is a measure of a plant's fidelity to high-quality remnant habitats (with 10 being the most sensitive score and 0 the lowest). This system is used for various regions across North America (Masters 1997). It therefore provides us with a continent-wide standard for assessing site biodiversity and quality. The CC values used by the TRCA are those assigned for southern Ontario plants by Oldham *et al.* (1995).

Breeding bird diversity is tracked by referring to habitat preferences; these preferences are listed in the Appendix (Table A.1) and were produced primarily through staff understanding of the various species' nesting requirements.

2.2 Forest Monitoring Methodology

2.2.1 Forest Vegetation Plots

Forest plots were set up according to standards developed by Environment Canada's Ecological Monitoring and Assessment Network (EMAN 2004a, EMAN 2004b, Roberts-Pichette and Gillespie 1999), with slight modifications. This protocol is almost identical to that used by the Credit Valley Conservation in its forest vegetation plot monitoring, although there are differences in sapling assessment (CVC 2010).

Detailed information on plot set-up can be found in TRCA (2011a). In summary, each forest vegetation plot consists of one 20 x 20 m square plot (i.e. 400 m^2) for monitoring tree health; and five 2 x 2 m subplots (i.e. 4 m^2) for monitoring woody regeneration (tree saplings, shrubs and woody vines). Four of the subplots are placed 1 m outside the perimeter of the 20 x 20 m tree health plot, and the fifth is located in its centre. Ground vegetation is measured in a 1 x 1 m subsection (1 m^2) of each subplot at its southwest quarter. Two visits are conducted per year: in the spring and in early-to-mid summer.

2.2.2 Forest Bird Stations

Forest birds were monitored using the Ontario Forest Bird Monitoring Program (FBMP) protocol designed by the Canadian Wildlife Service (TRCA 2011b). The forest bird stations are monitored twice per year at times considered optimum for recording forest breeding bird species. The first count is conducted between May 24th and June 17th; the second count is conducted no sooner than 10 days after the first visit and between the dates June 15th and July 10th. Many species that are recorded before the first week of June may still be passing through the area as migrants, therefore registering a second observation in late June or July supports the indication of a territorial and likely breeding individual. All counts are completed between 05:00 a.m. and 10:00 a.m. The second visit is completed at the same time of day as the first visit and an attempt is made to maintain the same timing schedule of visits in subsequent years.

Counts are conducted in weather conditions that optimize the detection of songbird species.

Ideally there should be very little to no wind, and precipitation should be at most a light rain. The FBMP requires the biologist to plot every individual bird observed and heard within a 100 m circle centred on the point station over a 10 minute period. In addition, any birds identified at distances beyond the 100 m circle are mapped at their approximate position. For the purposes of analysis it was decided to consider only those individuals and species located within the 100 m count circle.

2.2.3 Plethodontid Salamander Plots

Plethodontid salamanders are represented in the TRCA region by just one species, the eastern red-backed salamander (*Plethodon cinereus*). Local populations of Plethodontids are monitored by establishing grids of 40 artificial cover boards at forest stations across the jurisdiction (TRCA 2011c). Monitoring occurs over a 5 week period shortly after spring thaw (late April and early May) when frost is no longer a threat.

2.3 Wetland Monitoring Methodology

2.3.1 Vegetation Transects

Wetland vegetation is monitored along a 50 m transect, capturing a gradient of conditions (terrestrial to aquatic) that occur in most wetlands (TRCA 2011d). Where possible, the transect starts immediately outside the wetland in an adjacent terrestrial system, while the remainder of the transect lies within the wetland proper. Posts (lengths of white polyvinyl chloride or "PVC" pipe) are placed at 10 m intervals along the transect, and vegetation monitoring subplots occur 5 m on either side of each post. Thus, there are paired subplots at the 0, 10, 20, 30, 40 and 50 m points along the transect: 12 in total. Subplots for woody regeneration (tree saplings, shrubs and woody vines) are 2 x 2 m (4 m^2), while the rear outer quarter (1 x 1 m subplot) of each 4 m^2 subplot is used for ground vegetation). Detailed information on wetland transect layout can be found in TRCA (2011d).

All wetland vegetation data are collected concurrently, in mid-to-late summer (late July to mid-September). This corresponds with full vegetation expansion before autumnal die-back and with relatively low water levels. The timing also harmonizes with the schedule for the forest plots, which are sampled earlier in the season.

2.3.2 Wetland Bird Stations

Monitoring stations were set-up following the Marsh Monitoring Program (MMP) protocol that was established by Bird Studies Canada (TRCA 2011e). Observations and counts are undertaken in a 100 m-radius semi-circle from the station marker since in general, stations are located at the edge of the wetland. Multiple stations within the same site were separated by 250 m in order to avoid double-counting the same individual. The wetland stations are monitored twice per year at times considered optimum for recording wetland bird breeding species. The first count is conducted between May 20th and July 5th; the second count is conducted no sooner than 10 days after the first visit.

Counts are conducted in weather conditions that optimize the opportunity for the biologist to hear and observe wetland bird species. Ideally, there should be no wind (very light wind is acceptable), and precipitation should be light rain at the very most. The surveys are conducted in the morning hours a half hour before sunrise and end by 10:00 a.m. during appropriate weather conditions for bird activity. The field protocol for monitoring wetland birds requires counts to be made of individuals located only within the 100 m-radius semi-circle.

2.3.3 Frog Stations

Stations were set-up and monitored following the MMP in the same manner as wetland birds (TRCA 2011f). The frog stations are 100 m semi-circles with orientation noted and maintained on each visit; these frog stations need to be at least 500 m apart. Temperature guidelines change with each visit. For the first visit in the spring, night temperatures should be above 5°C, at least 10°C for the second visit and at least 17°C for the third and final visit. Surveys begin one half hour after sunset and end before midnight. Frogs were recorded as present and the observer estimated the number of individuals present along with the call code (1=no overlap of calls and an exact measurement of abundance of frogs calling can be determined, 2=calls begin to overlap and an estimate of abundance of frogs can be determined, 3=full chorus and the number of individuals cannot be counted).

2.4 Meadow Monitoring Methodology

2.4.1 Meadow Bird Stations

In the absence of any bird monitoring protocols designed specifically for meadow habitat it was decided to simply use the FBMP protocol and to adjust the suite of target species during analysis (TRCA 2011g). Each station is sampled twice per year with the first visit occurring between May 15th and May 30th, and the second visit between May 30th and June 15th, with at least 10 days between visits. Counts are conducted between 05:00 a.m. and 10:00 a.m., and at approximately the same time of day on subsequent visits from year to year. The field protocol for monitoring meadow birds is adapted from the forest bird protocol which requires counts to be made of individuals located both within and beyond the 100 m count circle. For the analysis of results, as with the forest and wetland results, it was decided to consider only those individuals and species located within the 100 m count circle.

2.5 Data Analysis

2.5.1 Temporal Trends

Temporal trends were analyzed using Mann-Kendall tests in an established Microsoft Excel™ spreadsheet provided by the Ministry of Natural Resources and Forestry (MNRF). The Mann-Kendall test is a non-parametric test for identifying monotonic trends in time series data. This test was chosen over traditional regression analyses because the data did not meet the assumption of independent samples required for regression analyses. When analyzing time-series data, data collected at the same site from one year to the next are not independent. This made the Mann-Kendall test the best option. The Mann-Kendall test uses the S statistic to determine an associated p-value. If the value of S is zero, there is no trend in the data. If a data value from a later time period is higher than a data value from an earlier time period, S is incremented by one. On the other hand, if the data value from a later time period is lower than a data value sampled earlier, S is decremented by one. The net result of all such increments and decrements yields the final value of S (TRCA 2011h). For example, a very high positive value of S is an indicator of an increasing trend, and a very low negative value indicates a decreasing trend (TRCA 2011h). A p-value of less than 0.05 denotes a significant trend (increasing or decreasing) and a p-value of greater than 0.05 indicates that there is no increase or decrease over time and that the variable of interest is stable. Tables showing statistical results will contain a red arrow indicating the direction of the temporal trend if $p<0.05$ and the word "caution" will be included if $0.05<p<0.10$ (approaching significance). All temporal trend graphs show means ± 1 standard error unless otherwise indicated.

2.5.2 Spatial Trends

All analyses were conducted using SAS JMP statistical software (SAS Institute Inc. 2008). Differences between urban and rural land use zones were analyzed using independent t-tests. An independent t-test is a parametric test that compares the mean value between two groups (e.g. urban and rural land use zones). This test is reported using the test statistic, t, and an associated p-value where a p-value of less than 0.05 indicates a difference between groups. A p-value of greater than 0.05 indicates that there is no difference between groups. Before performing t-tests, all data were checked for normality and homoscedasticity because these are two assumptions of using parametric statistics. If these assumptions were not met, data transformations were attempted to improve normality or heteroscedasticity. If data transformations were not effective, a Wilcoxon test was conducted (Z-statistic). This is the non-parametric version of an independent t-test and is the appropriate test to proceed with if the data do not meet assumptions. A Fisher's exact test was used to examine differences in the percent of sites occupied by each frog species between urban and rural zones. The Fisher's exact test is a modification of a chi-square test and is used when one of your cells has an expected frequency of less than five. Tables showing statistical results will contain a red arrow indicating the direction of the spatial trend if $p<0.05$ and the word "caution" will be included if $0.05<p<0.10$ (approaching significance). All spatial trend graphs show means ± 1 standard error unless otherwise indicated.

The terrestrial long-term monitoring program varies in the number of sites monitored for each taxa. In addition to variability among taxa, new sites have been added over time for various reasons. When examining temporal trends it is important that the same set of sites are measured each year and that there are no added or removed sites to ensure a valid comparison. A detailed description of which sites were included in each analysis of temporal and spatial trends for each taxa/indicator can be found in Appendix A (Tables A.3-A.13). Temporal trends were analyzed using data to maximize the number of years included and spatial trends were analyzed using data to maximize the number of sites included in the analysis.

2.5.3 Power Analysis

An *a priori* power analysis for various temporal and spatial analyses for each taxa was completed at the outset of the program (Zorn 2008). Power analyses are important to conduct to determine the appropriate sample sizes needed to detect temporal and spatial trends when setting up a new monitoring program. Power is important to consider in statistical analysis because it measures your ability to detect a real difference or a trend in the data.

3.0 Results

3.1 Power Analysis

The power recommendations of Zorn (2008) for TRCA's Terrestrial Long-term Monitoring Program were met for the majority of high-level indicators. A summary of the recommend sample sizes and whether TRCA met this requirement is shown in Tables 3, 4 and 5.

Zorn (2008) used specific statistical tests to calculate power *a priori*. Once the data were collected, several of the assumptions required for the use of many of the tests proposed by Zorn (2008) were violated (e.g. normality, homoscedasticity and independent samples). Due to these violations, the nonparametric version of the test was used (e.g. Mann-Kendall in place of linear regression and the Wilcoxon test in place of a t-test). Nonparametric statistics have a greater probability of committing a type II error (not detecting a difference when one is in fact present). Often the difference in power is not great, and can be compensated with a small increase in sample size for the nonparametric test (Zar 1999). For many of the statistical tests in this report, sample size was above that recommended by Zorn (2008).

The only analyses that did not meet the sample sizes recommended by Zorn (2008) for the original parametric tests were temporal trends in forest, wetland and meadow birds when analyzed solely in the urban zone or solely in the rural zone. Power was recalculated for these variables based on current sample sizes using the same analysis as Zorn (2008). Power for forest birds ranged from 72-82%, wetland birds from 62-82% and meadow birds from 62-90%. Several of these values are below the acceptable 80% power. This means that there is a lower ability to detect declines when declines are occurring.

Table 3. Suggested sample sizes and whether or not these met the assumptions of Zorn (2008) for forest indicators.

Monitoring measure	Monitoring objective	Suggested annual sample size and sampling unit to achieve 90% power (Zorn 2008)	Power assumptions met?
Tree mortality	Temporal trend	40 trees	Yes
	Spatial trend	30 trees per strata	Yes
Crown vigour	Temporal trend	30 trees	Yes
	Spatial trend	30 trees per strata	Yes
Mean FQI	Temporal trend	11 plots	n/a
	Spatial trend	7 plots per strata	Yes
Forest bird species richness	Temporal trend	14 sites	Yes/caution
	Spatial trend	8 sites per strata	Yes
Forest bird abundance	Temporal trend	14 to 20 sites	Yes/caution
	Spatial trend	8 to 11 sites per strata	Yes
Plethodontid abundance	Temporal trend	14 plots	n/a
	Spatial trend	8 plots per strata	n/a
Invasive/exotic plants	Temporal trend	8 plots	Yes
	Spatial trend	8 plots per strata	Yes

Table 4. Suggested sample sizes and whether or not these met the assumptions of Zorn (2008) for wetland indicators.

Monitoring measure	Monitoring objective	Suggested annual sample size and sampling unit (Zorn 2008)	Power assumptions met?
Frog chorus call	Temporal trend	14 stations	Yes
	Spatial trend	6 stations per strata	Yes
Wetland bird species richness	Temporal trend	10 sites	Yes/caution
	Spatial trend	7 sites per strata	Yes
Wetland bird abundance	Temporal trend	14 to 20 sites	Yes/caution
	Spatial trend	8 to 11 sites per strata	Yes
Mean FQI	Temporal trend	12 sites	n/a
	Spatial trend	8 sites per strata	Yes
Invasive/exotic plants	Temporal trend	22 sites	n/a
	Spatial trend	8 sites per strata	n/a

Table 5. Suggested sample sizes and whether or not these met the assumptions of Zorn (2008) for meadow indicators.

Monitoring measure	Monitoring objective	Suggested annual sample size and sampling unit (Zorn 2008)	Power assumptions met?
Meadow bird species richness	Temporal trend	7 sites	Yes/caution
	Spatial trend	6 sites per strata	Yes
Meadow bird abundance	Temporal trend	14 to 20 sites	Caution
	Spatial trend	8 to 11 sites per strata	Yes

3.2 Forest Monitoring

3.2.1 Forest Vegetation

Forest vegetation showed significant increasing trends for both species richness and the floristic quality index (FQI) between 2008 and 2014. These increases are likely due to biologists finding rare species in more recent years that may have been missed in earlier years. This is a common situation when starting monitoring programs and in scientific studies and it is expected that these values will plateau at a point when all species have been identified (Grandin 2011). In addition to finding rare species in more recent years, surveys were not timed and therefore different levels of effort were likely used in different years or even among plots within the same year. In 2015, a pilot study was conducted to see how adding a time limit on surveys will affect the results. Temporal trends for FQI, number of L1-L3 species and the percent native species will not be analyzed in this report based on this limitation. A baseline for tracking temporal trends in these variables will be established once the vegetation identified has stabilized.

Spatial Trends

Regionally between 2008 and 2014 forest plots contained a total of 289 species including 63 exotic species (22%) and 226 native species (78%). Urbanization had a significant impact on all three forest high-level indicators: FQI, number of L1-L3 species and percent native species (Figure 2, Table 6). The mean FQI was significantly lower in urban forests (22.1) compared to rural forests (28.2). Rural forests had on average three L1-L3 species while urban forests on average had only one L1-L3 species. Urban forests consisted of flora communities with a lower percentage of native species (74% native species) compared to rural sites (88% native species). This suggests that urban forests contain a higher percentage of exotic plant species. In general, the impact of urbanization on these forest vegetation high-level indicators is similar.

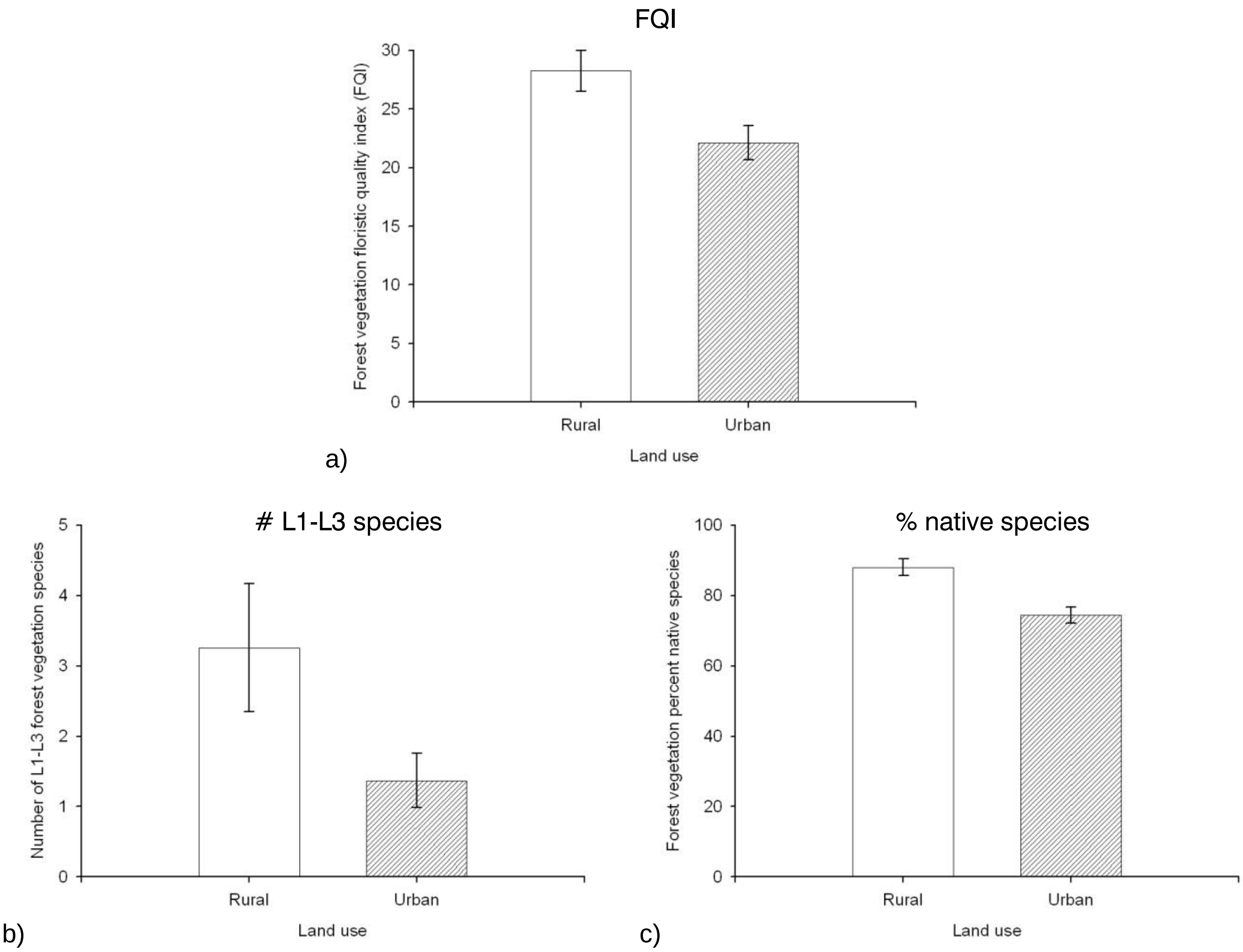


Figure 2. Spatial trends in forest vegetation high-level indicators a) Floristic Quality Index (FQI), b) number of L1-L3 forest vegetation species and c) percent native forest vegetation species.

Table 6. Statistical results for spatial trends in forest vegetation high-level indicators.

Metric	t-value	P-value	Effect of urbanization
Floristic Quality Index (FQI)	2.74	0.01	↓
Number of L1-L3 species	2.02	<0.001	↓
Percent native species	4.12	<0.001	↓

Tree Composition

As of 2014, a total of 554 live trees were being monitored in regional tree health plots. There were a total of 30 tree species including five exotic species: common buckthorn (*Rhamnus cathartica*), Manitoba maple (*Acer negundo*), apple (*Malus pumila*), black locust (*Robinia pseudoacacia*) and pear (*Pyrus communis*) (Figure 3). Tree health plots were dominated by native species (83%) with sugar maple (*Acer saccharum* ssp. *saccharum*) having the highest relative abundance of 34%. Since the 2012 baseline report, several species changed position based on relative abundance ranking although the differences primarily involved changes of one position (TRCA 2012a). For example, eastern hemlock (*Tsuga canadensis*) increased from 4th to 3rd and bur oak (*Quercus macrocarpa*) decreased from 3rd to 4th. The larger changes occurred for species ranked 25th or greater in abundance. For example, trembling aspen (*Populus tremuloides*) dropped from 26th in 2008 to 30th in 2014.

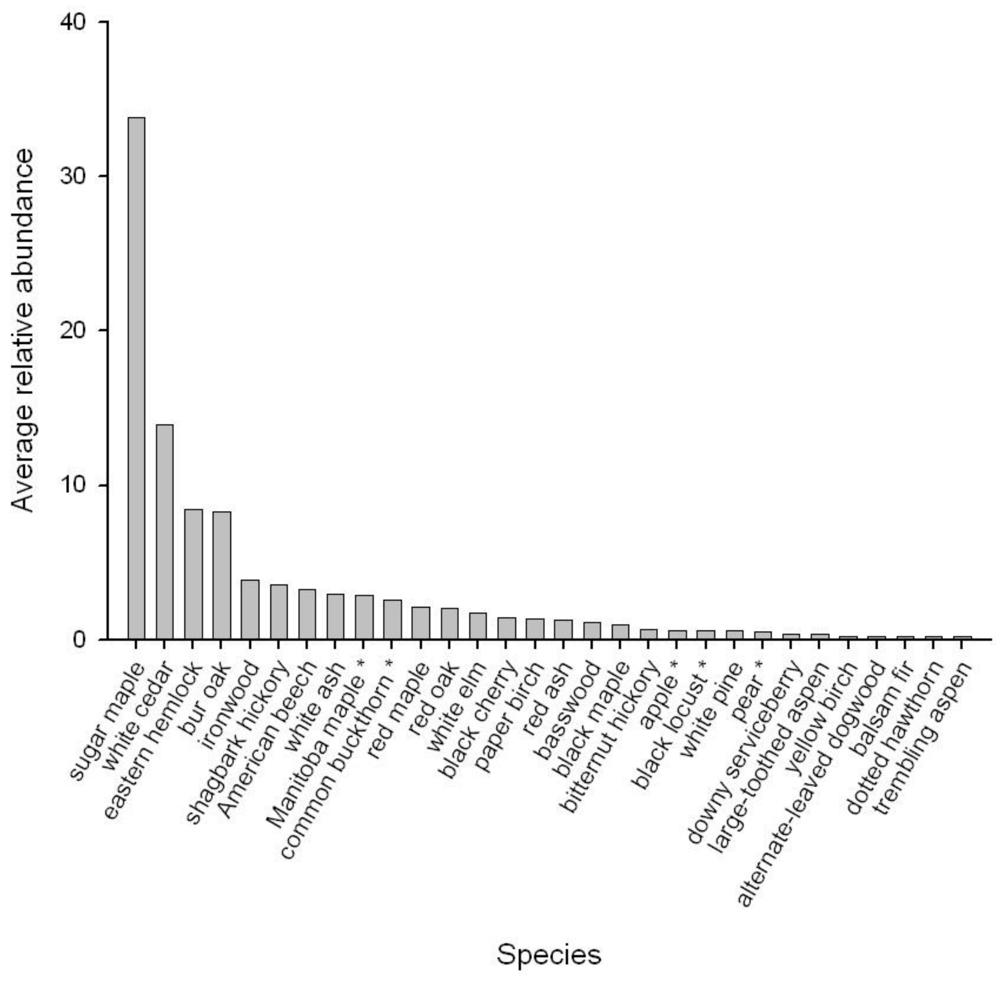
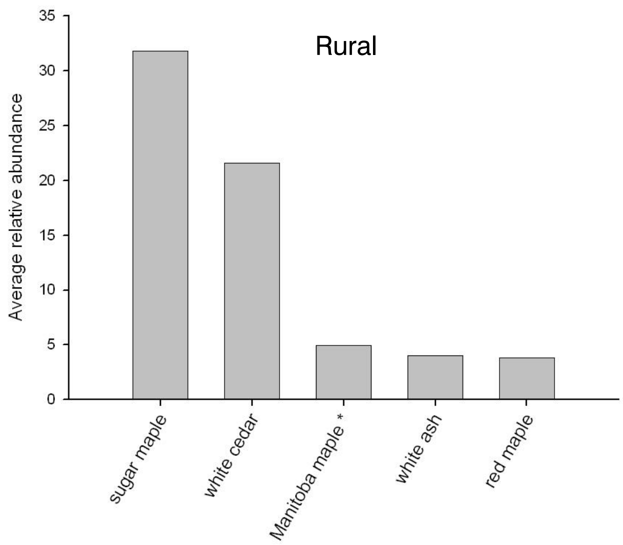
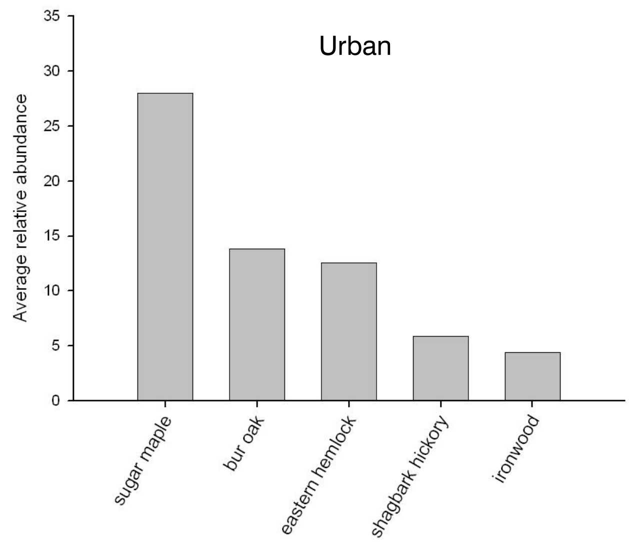


Figure 3. Average relative abundance of tree species in regional tree health plots (2008-2014). Exotic species are indicated with an asterisk (*).

There was some variation between urban and rural zones in the relative abundance of the top five tree species (Figure 4). Both rural and urban forests were dominated by sugar maple but relative abundance was slightly higher at rural sites (32%) compared to urban sites (28%). Sugar maple was the only tree species found in both zones within the top five most abundant tree species, and all other species were unique between zones. The species composition of the top five species has remained consistent in the urban zone since the baseline report; however, in the rural zone Manitoba maple has moved from 4th to 3rd, white ash (*Fraxinus americana*) moved from 5th to 4th and red maple (*Acer rubrum*) dropped from 3rd to 5th.



a)



b)

Figure 4. Average relative abundance of the five most common tree species (2008-2014) in the a) rural and b) urban land use zones. Exotic species are indicated with an asterisk (*).

Forest Sapling and Shrub Composition

Total Quantity of Woody Regeneration

Density of woody regeneration (tree saplings, shrubs and woody vines) varied enormously across the subplots (Figure 5). Between 2008 and 2014, the densest plot was Portage Trail in the Lower Humber subwatershed and the least dense plot was Reesor Road – Hwy 7 in the Little Rouge subwatershed. There was no significant change over time in the density of woody stems in the regeneration layer (Figure 5, Table 7). Choke cherry (*Prunus virginiana* var. *virginiana*), one of the most important native species in the regeneration layer, was found to be approaching a significant increase in relative abundance although the increase would be small given the small increases seen between years (Figure 5c). Trends were not analyzed for choke cherry in the rural zone alone because it was only found at two sites and a temporal analysis of these data would not be a representative sample. There was no significant difference between rural and urban sites in the overall density of woody stems, the relative percent cover of choke cherry or the relative abundance of choke cherry (Figure 6, Table 8) .

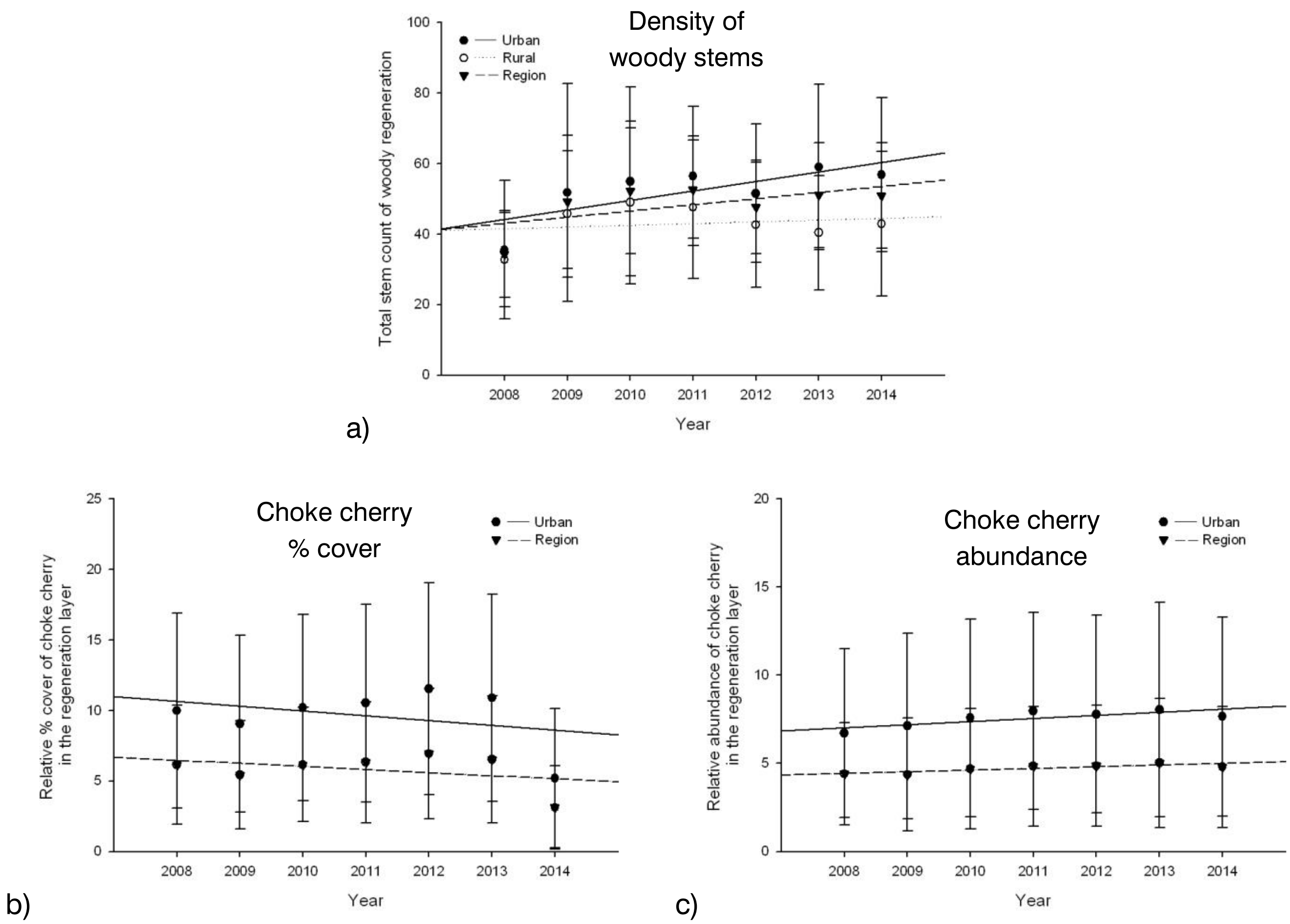
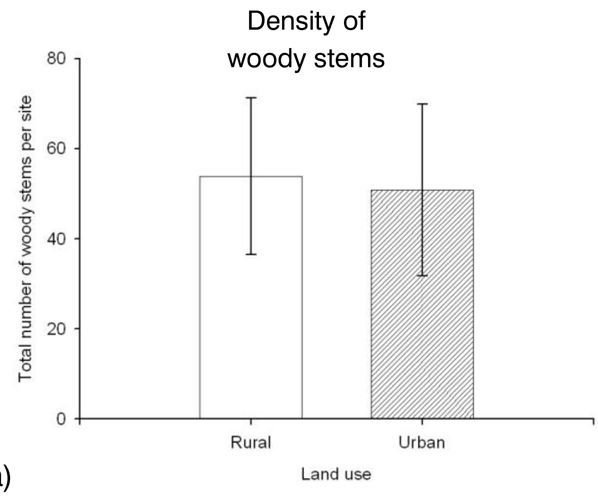


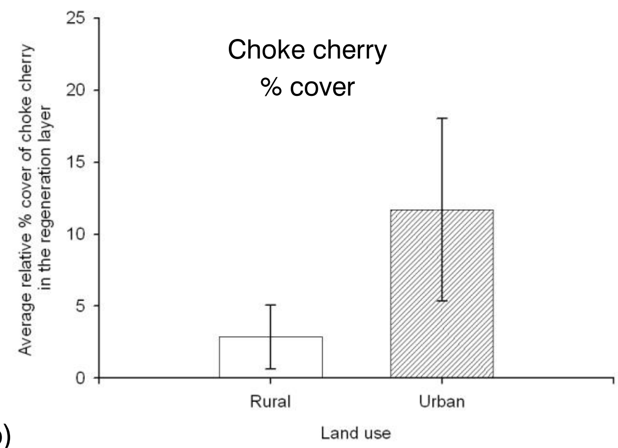
Figure 5. Temporal trends in a) density of woody regeneration b) choke cherry relative % cover and c) choke cherry relative abundance between 2008 and 2014.

Table 7. Statistical results for temporal trends in woody regeneration and choke cherry (2008-2014).

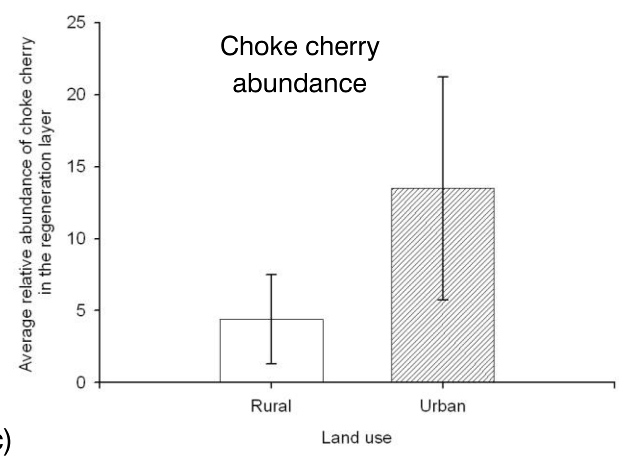
Metric	Land use	S-value	P-value	Increase, decrease or stable?
Total number of stems of woody regeneration	Region	0.601	0.548	Stable
	Urban	1.802	0.072	↑ Caution
	Rural	0.000	1.000	Stable
Choke cherry (relative % cover in the regeneration layer)	Region	0.601	0.548	Stable
	Urban	0.601	0.548	Stable
	Only 1 rural site with choke cherry cover greater than 1% to determine trends			
Choke cherry (relative abundance)	Region	1.802	0.072	↑ Caution
	Urban	1.802	0.072	↑ Caution
	Only 2 rural sites with choke cherry present to determine trends			



a)



b)



c)

Figure 6. Regeneration in urban and rural sites between 2010 and 2014 measured as a) density of woody vegetation, b) choke cherry average relative % cover and c) choke cherry average relative abundance.

Table 8. Statistical results for spatial trends in woody regeneration and choke cherry.

Metric	Test statistic	P-value	Effect of urbanization
Total number of stems of woody regeneration	Z = 0.116	0.908	Non-significant
Choke cherry (relative % cover in the regeneration layer)	Z = 0.304	0.76	Non-significant
Choke cherry (relative abundance)	Z = 0.04	0.97	Non-significant

Woody Regeneration Species Composition

A total of 55 woody species were found in the regeneration layer across the region with an average of 7 species per site (Table A.2 in Appendix). A total of 38 species were found in the regeneration layer in the rural zone and 39 species in the urban zone (Figure 7). Portage Trail had the highest species richness of woody regeneration across the region with on average 14 species found at the site between 2010 and 2014. Reesor Rd & Hwy 7 had no regeneration.

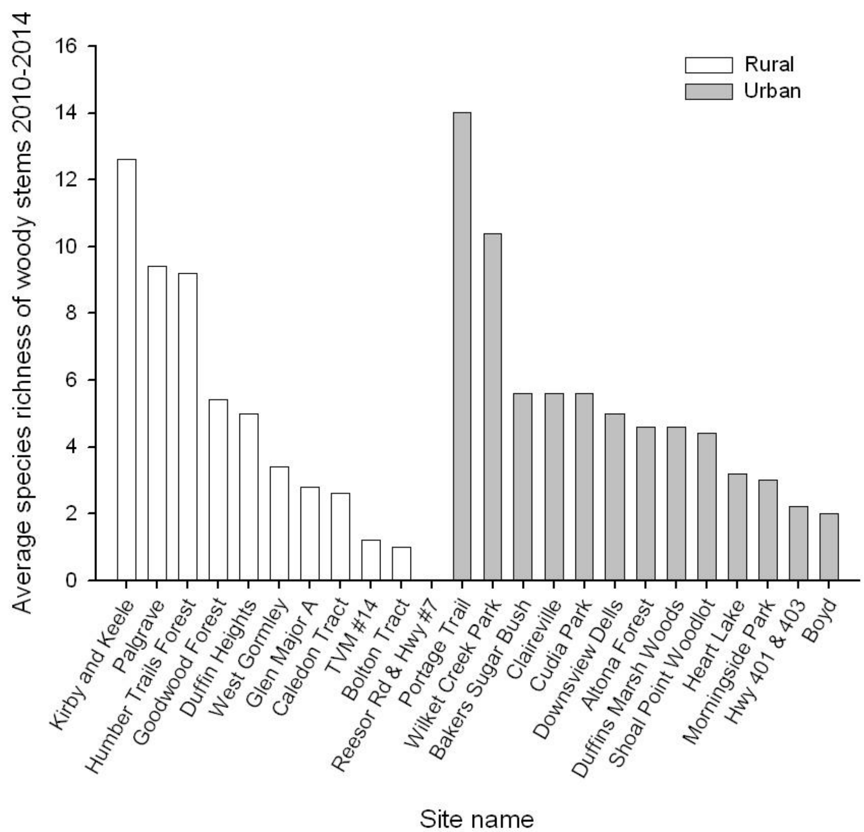


Figure 7. Species richness of woody vegetation in the regeneration layer by site measured as the average number of species per site between 2010 and 2014.

The regeneration community in the rural land use zone was dominated by sugar maple when measured based on the number of stems and percent cover (Figures 8 and 9). In the urban zone sugar maple was also dominant based on percent cover; however, choke cherry was more dominant based on the number of stems. Common buckthorn, an invasive plant species, was present in both the urban and rural land use zones and only varied slightly in percent cover and abundance.

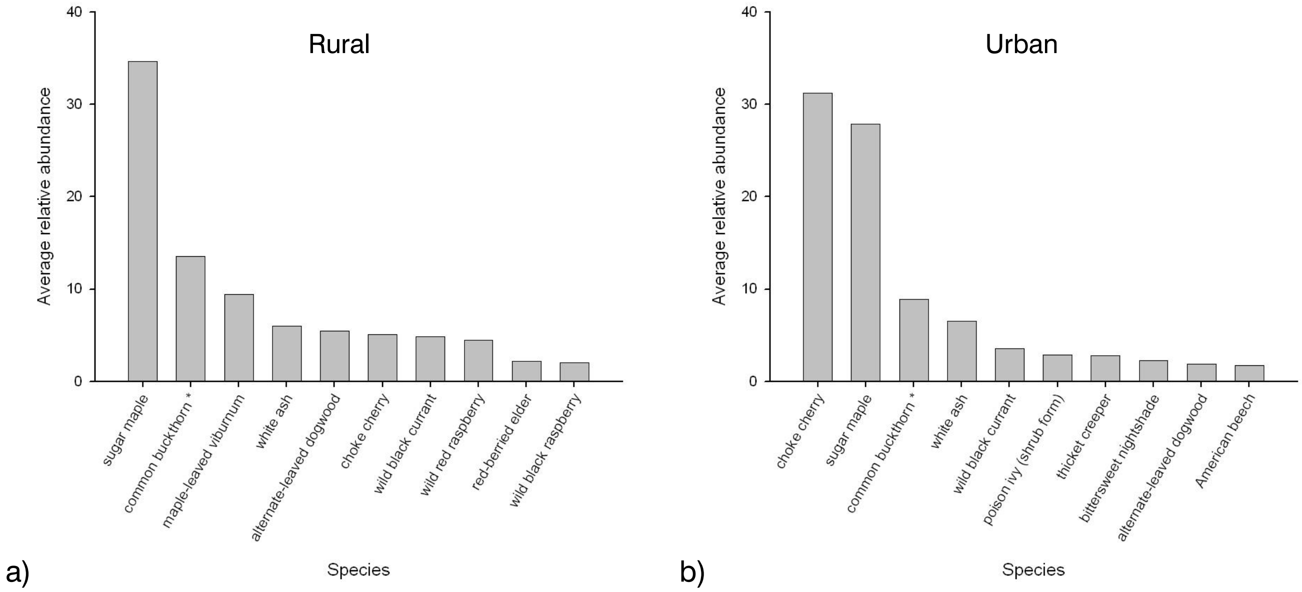


Figure 8. Average relative abundance of the 10 most common species of woody regeneration (2008-2014) in the a) rural and b) urban land use zones. Exotic species are indicated with an asterisk (*).

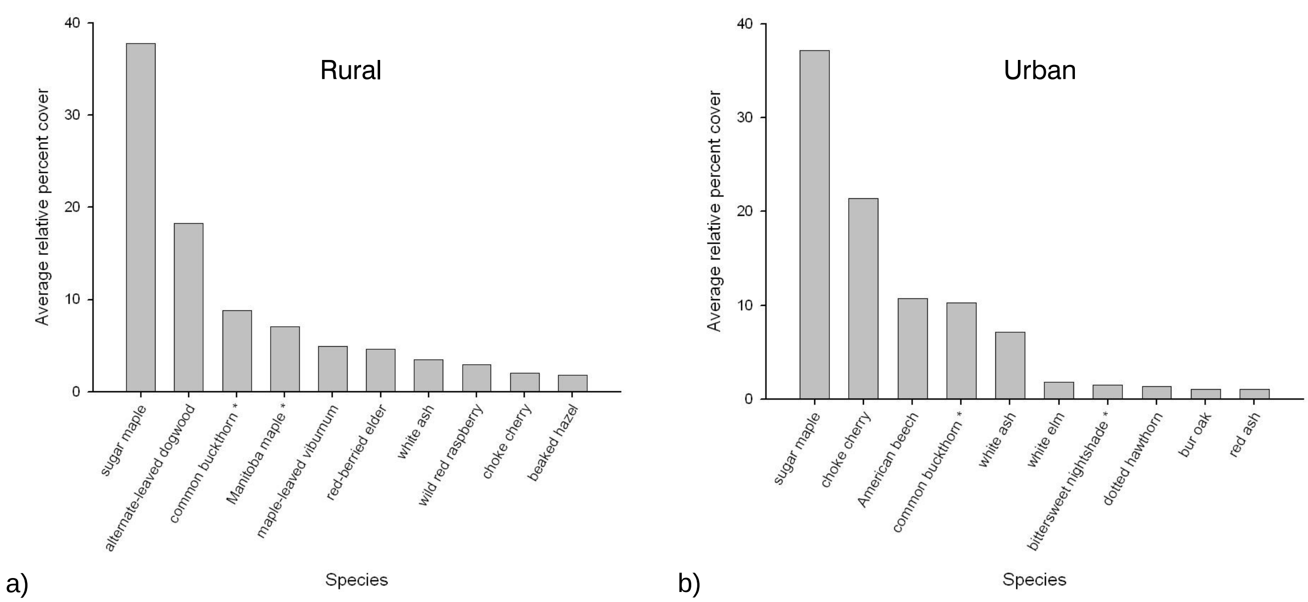


Figure 9. Average relative percent cover of the 10 most common woody regeneration species (2008-2014) in the a) rural and b) urban land use zones. Exotic species are indicated with an asterisk (*).

The relative abundance and cover of native species was considerably higher than exotic species in both the rural and urban land use zones (Table 9). This demonstrates that the woody regeneration layer in our forests is still dominated by native species.

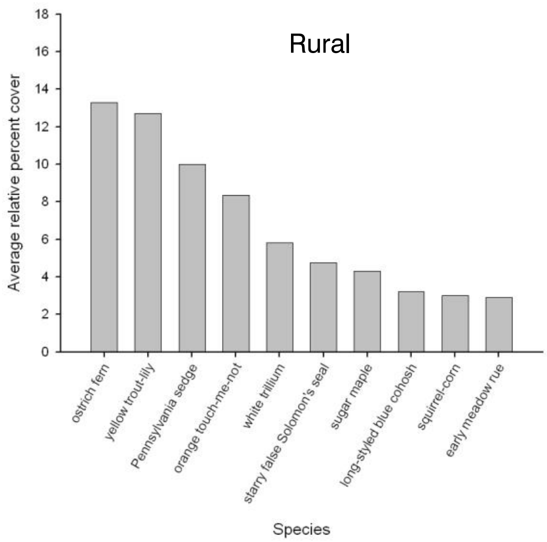
Table 9. Relative abundance and cover of native and exotic woody regeneration species by land use zone (2008-2014).

	Rural		Urban	
Measure	**Relative abundance (%)**	**Relative cover (%)**	**Relative abundance (%)**	**Relative cover (%)**
Native	83.1	84.7	87.4	87.6
Exotic	16.9	15.3	12.6	12.4

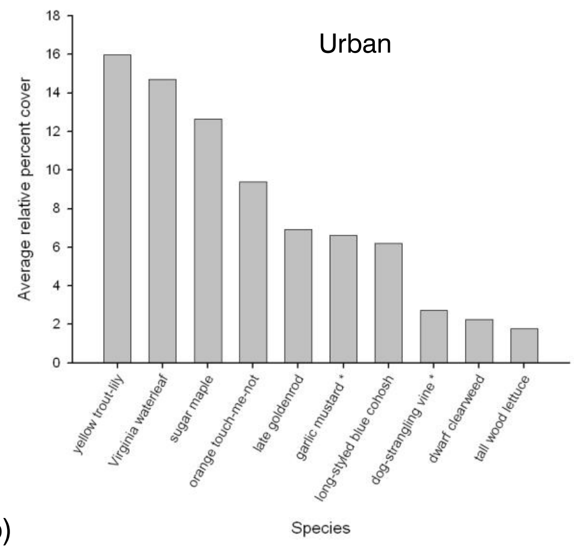
Forest Ground Vegetation Composition

Ground vegetation analysis included only data from 2009-2014 because many plots were just being set-up in the summer of 2008 and thus the spring ground vegetation visits were missing for many sites. Across the jurisdiction, 177 species were identified in ground vegetation plots excluding species that could only be identified to genus. Of the 177 species, 38 were exotic (21.5%) and 139 were native (78.5%).

Yellow trout lily (*Erythronium americanum* ssp. *Americanum*; 14%), orange touch-me-not (*Impatiens capensis*; 9%), sugar maple (8%) and ostrich fern (*Matteuccia struthiopteris* var. *pensylvanica*; 7%) had the top 5 highest values of relative cover in the jurisdiction. Relative cover of each species varied by zone with rural sites dominated by ostrich fern, yellow trout lily and Pennsylvania sedge (*Carex pensylvanica*) and urban sites dominated by yellow trout lily, Virginia waterleaf (*Hydrophyllum virginianum*) and sugar maple (Figure 10). Rural sites contained squirrel-corn (*Dicentra canadensis*), the only species of concern (rank of L3) to fall in the top 10 in both land use zones. Relative cover of garlic mustard (*Alliaria petiolata*) and dog-strangling vine (*Cynanchum rossicum*) ranked in the top 10 species in the urban sites (6th and 8th, respectively) and ranked 25th and 30th in the rural zone. Garlic mustard was found at 7 of 13 urban sites and only 1 of 11 rural sites. Sites with particularly high cover of garlic mustard were Cudia Park (FV-14) at 25%, Downsview Dells (FV-4) at 13.1% and Morningside Park (FV-15) at 11.3% relative cover. Maximum relative cover of exotic species was higher in urban sites (12.7%) compared to rural sites (7.8%; Table 10).



a)



b)

Figure 10. Average relative percent cover of the 10 most common species in the ground vegetation layer (2009-2014) in the a) rural and b) urban land use zones. Exotic species are indicated with an asterisk (*).

Table 10. Relative maximum % cover of native and exotic species in the rural and urban zones.

	Rural	Urban
Measure	**Relative maximum cover (%)**	**Relative maximum cover (%)**
Native	92.2	87.3
Exotic	7.8	12.7

Spread of Invasive Species

The spread of invasive species was measured by examining changes in the relative cover in the ground vegetation layer for all exotic species combined and garlic mustard independently. A list of species included in the analysis of all exotic species combined can be found in Table A.14 in the Appendix. The spread of common buckthorn was examined in the regeneration layer.

Temporal Trends

The relative cover of all exotic species combined in the ground vegetation layer was relatively stable between 2009 and 2014 (Figure 11, Table 11). Garlic mustard when analyzed separately showed increasing trends that were very close to being significant ($p = 0.06$) at both the regional scale and in the urban land use zone. Particularly high years for garlic mustard were 2013 and

2014. There was no significant increase or decrease in garlic mustard in the rural zone alone. The relative cover and abundance of common buckthorn in forests was stable and showed no signs of increase regionally or in the urban zone. There was a noticeable increase in relative cover between 2013 and 2014 although this was not consistent with an increase in relative abundance. There was no significant increase or decrease in common buckthorn in the rural zone alone. Other virulent invasive flora species such as dog-strangling vine or honeysuckles ((*Lonicera x bella, Lonicera tatarica, Lonicera morrowi*) were not analyzed because of their low level of occurrence in the long-term monitoring plots.

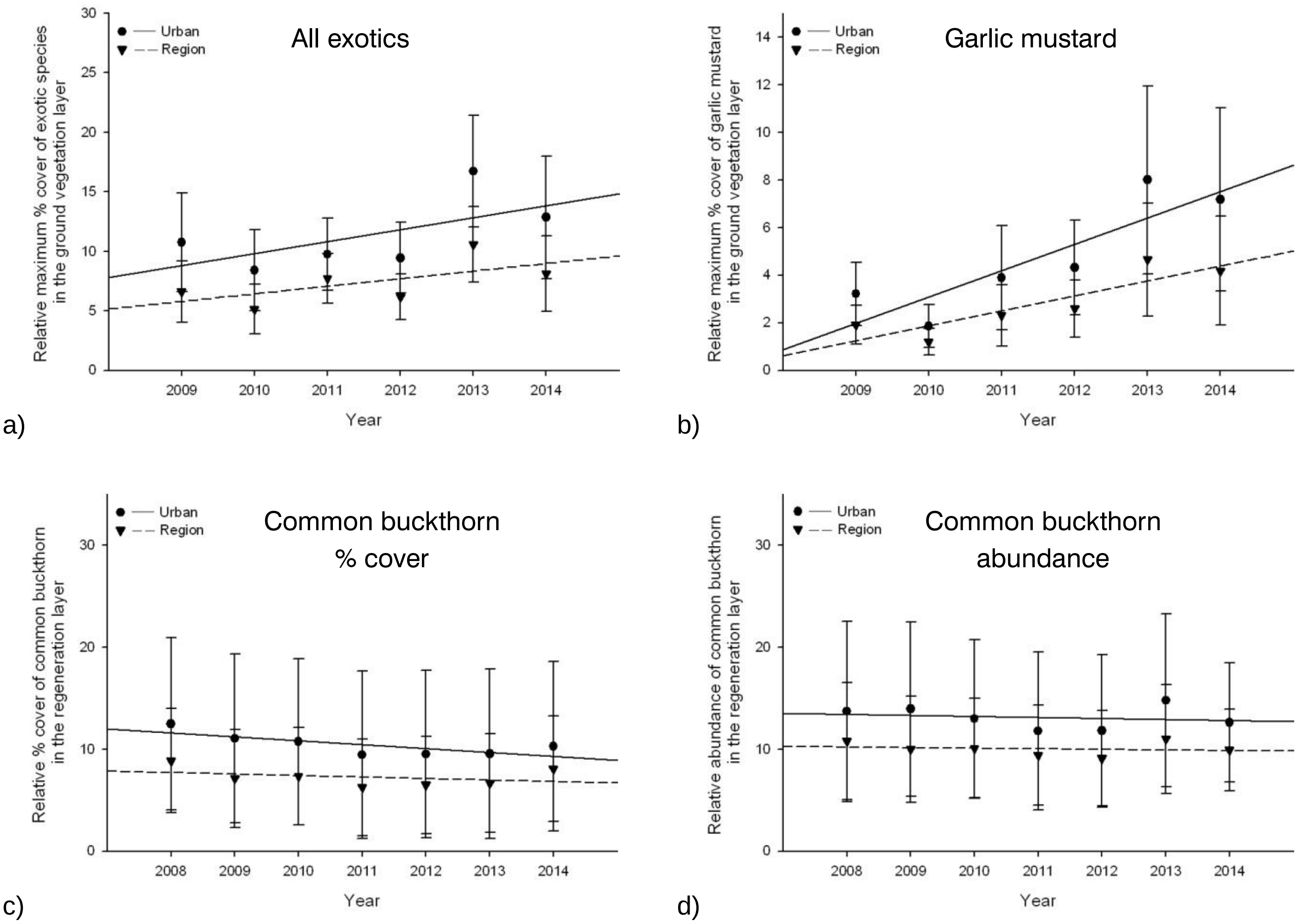


Figure 11. Temporal changes in a) relative percent cover of all exotic species b) relative percent cover of garlic mustard c) relative percent cover of common buckthorn and d) relative abundance of common buckthorn.

Table 11. Statistical results for temporal trends in the spread of invasive species in forest vegetation regional plots (2009-2014).

Metric	Land use	S-value	P-value	Increase, decrease or stable?
Relative % cover of all exotic species combined in the ground vegetation layer	Region	1.127	0.260	Stable
	Urban	0.751	0.452	Stable
	Only 2 rural sites with exotic species cover greater than 1% to determine trends			
Garlic mustard (relative % cover in the ground vegetation layer)	Region	1.879	0.060	↑ Caution
	Urban	1.879	0.060	↑ Caution
	Only 1 rural site with garlic mustard cover greater than 1% to determine trends			
Common buckthorn (relative % cover in the regeneration layer)	Region	-0.300	0.764	Stable
	Urban	-1.202	0.230	Stable
	Only 1 rural site with common buckthorn cover greater than 1% to determine trends			
Common buckthorn (relative abundance in the regeneration layer)	Region	-0.601	0.548	Stable
	Urban	-0.300	0.764	Stable
	Only 2 rural sites with common buckthorn to determine trends			

Spatial Trends

There were prominent spatial trends in the relative cover and abundance of exotic species between urban and rural land use zones. Sites in the urban zone had significantly higher relative percent cover of all exotic species combined than rural zones (Figure 12, Table 12). This suggests that the ground vegetation layer in urban forests has a higher cover of exotic species than rural forests. Garlic mustard may be heavily contributing to this trend based on the large difference in relative cover between urban and rural forests. Common buckthorn also had a higher relative cover and relative abundance in the urban zone.

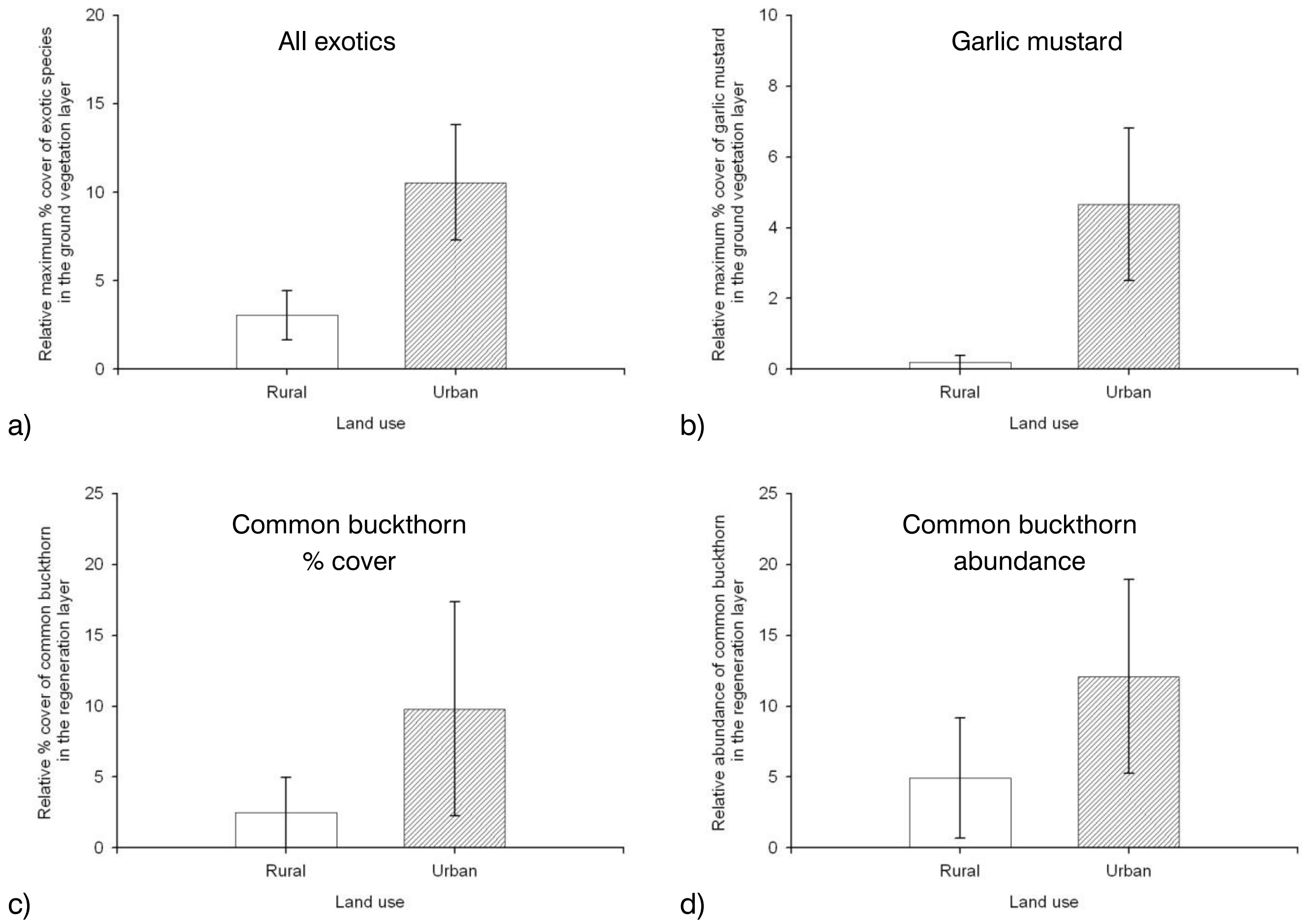


Figure 12. Spatial trends in a) relative percent cover of all exotic species b) relative percent cover of garlic mustard c) relative percent cover of common buckthorn and d) relative abundance of common buckthorn.

Table 12. Statistical results for spatial trends in the relative cover and abundance of selected exotic species.

Metric	Test statistic	P-value	Effect of urbanization
All exotic species in the ground vegetation layer	Z = 1.85	0.065	↑ Caution
Garlic mustard	Z = 2.21	0.027	↑
Common buckthorn (relative % cover)	Z = 1.73	0.083	↑ (Caution)
Common buckthorn (relative abundance)	Z = 2.20	0.028	↑

Tree Health Temporal and Spatial Trends

Crown Vigour – Crown Classes 1 and 2 (Dominant and Co-dominant)

Crown class affects crown vigour because trees with crowns that are dominant and co-dominant in the forest canopy are naturally less stressed because they receive more sunlight than crowns that are intermediate or suppressed. For this reason crown vigour was analyzed using only trees with crown classes of dominant and co-dominant (classes 1 and 2). Due to methodological inconsistencies, crown vigour category 4 (dead) could not be included in the analysis. Instead, crown vigour of only live trees was analyzed. Trees with a missing crown class or a crown class of zero were excluded from analysis.

Crown vigour of dominant and co-dominant trees consisted primarily of healthy trees (91.6% on average between 2008 and 2014 across the region; Figure 13). On average 6.7% of trees were in light to moderate decline and 1.6% were in severe decline. Regionally, between 2008 and 2014 the percent of live trees with healthy crowns is showing signs of decline and these declines were statistically significant in the rural land use zone (Table 13). This corresponds to an increase in the percent of live trees with crowns showing light/moderate or severe declines.

Urban and rural land use zones were similar in the crown vigour composition of live trees (Figure 13). There were approximately equal proportions of healthy trees and trees with light to moderate decline in urban and rural sites. There was a significant difference in the mean percent of live trees in severe decline between urban and rural land use zones (Table 14). In rural forests there was a higher percent of trees in severe decline compared to urban sites.

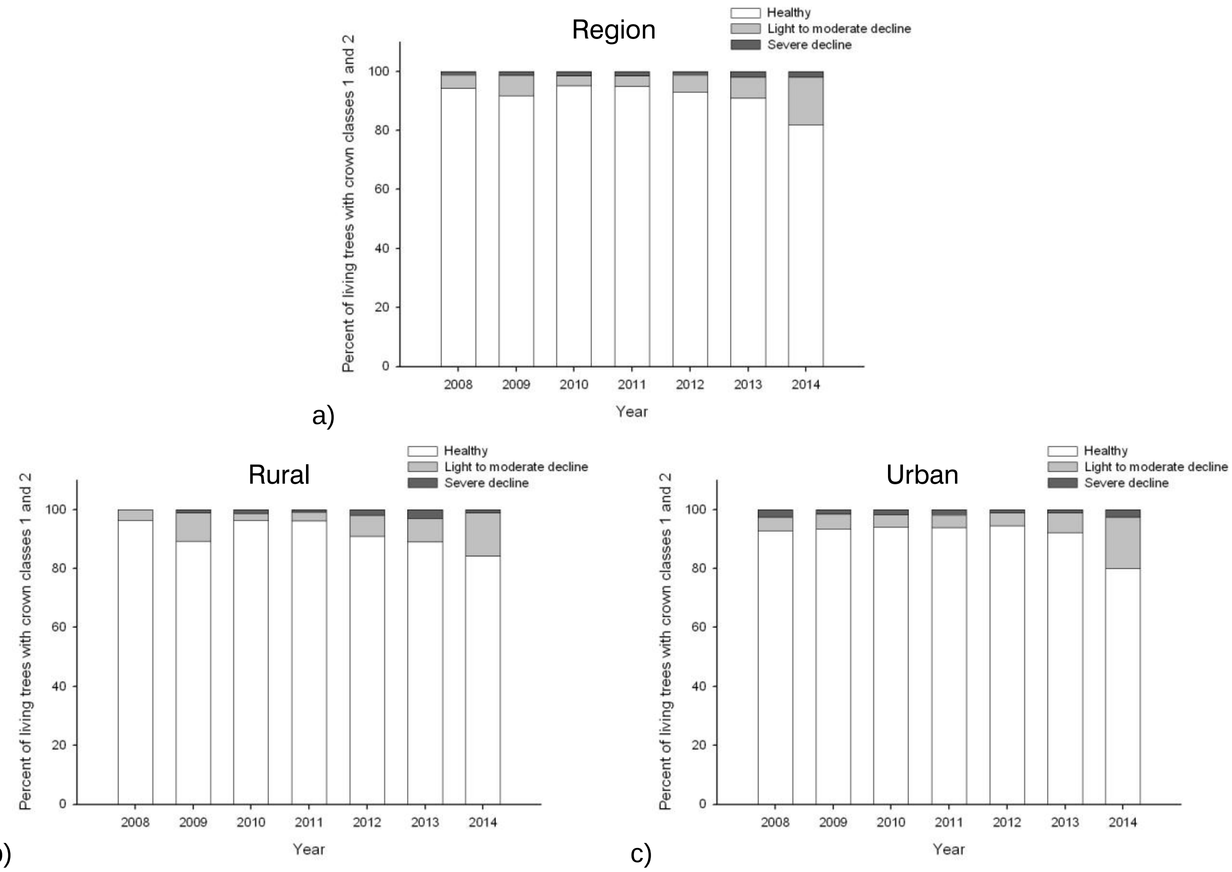


Figure 13. Temporal changes in crown vigour of living trees with crown classes 1 and 2 (dominant and co-dominant) a) region, b) rural land use zone and c) urban land use zone.

Table 13. Statistical results for temporal trends in crown vigour for crown class 1 and 2 living trees (2008-2014).

Metric	Land use	S-value	P-value	Increase, decrease or stable?
Crown vigour 1 (Healthy)	Region	-1.502	0.133	Stable
	Urban	-0.300	0.764	Stable
	Rural	-2.103	0.035	↓
Crown vigour 2 (Light to moderate decline)	Region	1.502	0.133	Stable
	Urban	0.901	0.368	Stable
	Rural	1.202	0.230	Stable
Crown vigour 3 (Severe decline)	Region	1.802	0.072	↑ Caution
	Urban	-0.150	0.881	Stable
	Rural	1.202	0.230	Stable

Table 14. Statistical results for spatial trends in crown vigour for crown class 1 and 2 living trees.

Metric	Test statistic	P-value	Mean % of trees in urban forests	Mean % of trees in rural forests
Crown vigour 1 (Healthy)	Z = 1.19	0.234	91.6	91.1
Crown vigour 2 (Light to moderate decline)	t = 0.002	0.998	6.79	6.79
Crown vigour 3 (Severe decline)	Z = 1.85	0.065	1.61 (Caution)	2.04 (Caution)

There was variation in which tree species are showing signs of decline (Figure 14). Of the five most abundant species in the region (sugar maple, white cedar (*Thuja occidentalis*), eastern hemlock, bur oak and ironwood (*Ostrya virginiana*)), all but eastern hemlock showed a decrease in health between 2013 and 2014. Black cherry (*Prunus serotina*) is included here because of the large decrease in health between 2013 and 2014 affecting five of seven black cherry trees in regional plots. In general, white ash and red ash (*Fraxinus pennsylvanica*) had more trees in the categories of light to moderate decline or severe decline than other species.

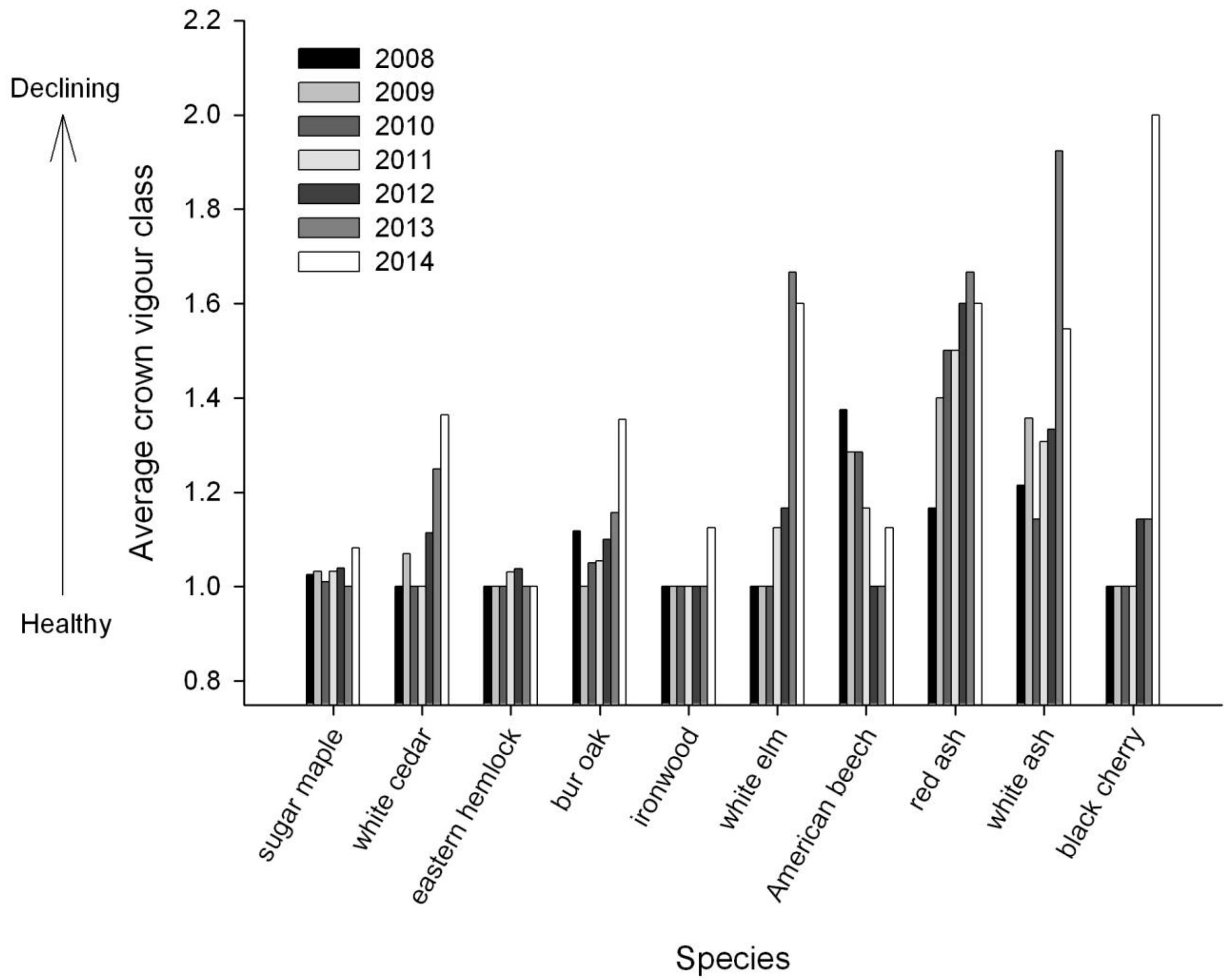


Figure 14. Temporal trends in average crown vigour (dominant and co-dominant) for selected tree species between 2008 and 2014. For each species, bars on the graph run left to right chronologically by year.

Mortality

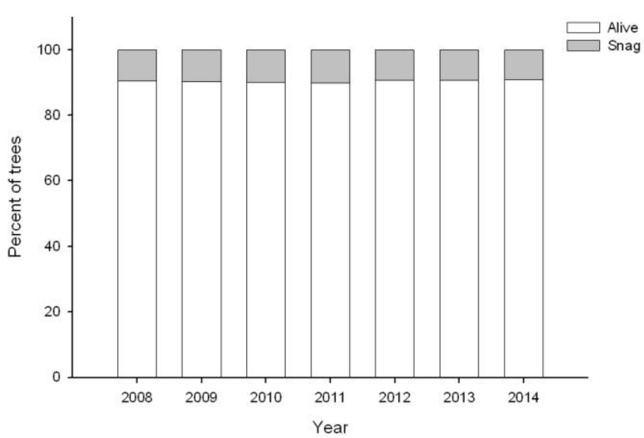
Mortality was measured by determining the number of trees that changed in status from living to dead between two consecutive years. Only trees with dominant and co-dominant crown classes and those that were surveyed annually between 2008 and 2014 were included in this analysis (n=185 trees). In general, there were low mortality rates with only one or two trees dying per year (Table 15). Most of these trees had no signs of pests or disease although the bitternut hickory (*Carya cordiformis*) did show signs of tar fungus in the year it died. The white elm (*Ulmus americana*) that died in 2014 was subject to defoliation by an unknown defoliator in 2011 and showed extreme signs of crown dieback in 2012 with Dutch elm disease suspected. Dutch elm disease was highly suspected in 2013 with >60% crown dieback with epicormic growth and missing bark. There were no signs of a significant increase or decrease in mortality between 2009 and 2014 (S = 1.13, p = 0.26). Of the six trees that died between 2009 and 2014, five were in the rural zone and one was in the urban zone.

Table 15. Annual tree mortality between 2008 and 2014 based on tree status.

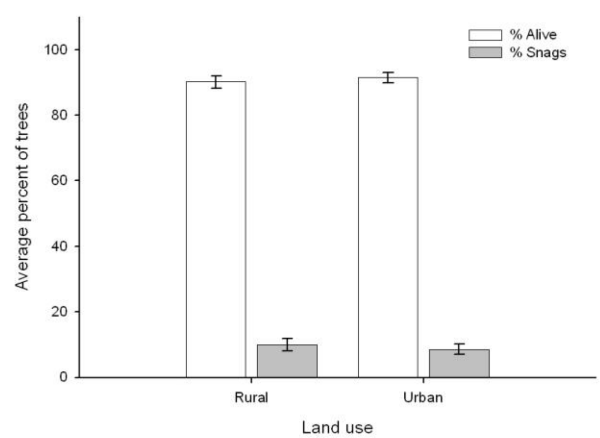
Years	Mortality (% of trees that died)	Species
2008-2009	0.54	White elm
2009-2010	0.54	White ash
2010-2011	0.55	Sugar maple
2011-2012	0.00	-
2012-2013	0.55	Bitternut hickory
2013-2014	1.10	White elm and sugar maple

Alive Trees vs. Snags

Tree conditions included as snags were dead broken (DB), dead standing (DS) and dead leaning (DL) and all live trees were included in the alive category. The percentage of trees classified as snags or alive was relatively constant between 2008 and 2014 (Figure 15, Table 16). Across the region on average 90.8% of trees were alive and 9.2% of trees were snags. Urban and rural sites did not vary statistically in the percentage of snags, but qualitatively rural sites did have a lower percentage of live trees (90.1%) and a higher percentage of snags (9.9%) than urban sites (91.5% live, 8.5% snags; Table 17).



a)



b)

Figure 15. Percent of trees classified as snags or alive analyzed a) temporally and b) spatially.

Table 16. Statistical results for temporal trends in the percent snag vs. live trees (2008-2014).

Metric	Land use	S-value	P-value	Increase, decrease or stable?
Average % snags	Region	0.300	0.764	Stable
	Urban	0.901	0.368	Stable
	Rural	-0.601	0.548	Stable

Table 17. Statistical results for spatial trends in the percentage of snag or live trees.

Metric	t-value	P-value	Effect of urbanization
Average % snags	0.548	0.590	Non-significant

Snag Species Composition

White elm was the most abundant snag tree in the jurisdiction while balsam poplar (*Populus balsamifera*) snags dominated the rural zone and ironwood snags dominated the urban zone (Table 18). Since the baseline report, apple has increased in relative abundance in the jurisdiction moving into the list of the six most abundant snag species and white ash has decreased in relative abundance moving from 5th to 8th. Balsam poplar, sugar maple and apple are the three most abundant snag species in the rural zone and ironwood, white elm and common buckthorn are the three most abundant snag species in the urban zone.

Table 18. Composition of the six most abundant snag species (average % relative abundance) observed in TRCA regional plots between 2008 and 2014.

Region Total		Rural Zone		Urban Zone	
white elm	14.2%	balsam poplar	15.5%	ironwood	21.2%
sugar maple	9.9%	sugar maple	13.3%	white elm	18.0%
balsam poplar	9.3%	apple	12.4%	common buckthorn	9.0%
ironwood	8.9%	white cedar	12.1%	American beech	8.6%
white cedar	7.6%	white elm	11.1%	bur oak	8.4%
apple	7.4%	red maple	7.6%	eastern hemlock	6.3%

Pests and Disease

Incidences of Anthracnose were low across the jurisdiction with only 0.4% of trees affected in 2013 and 0.2% of trees affected in 2014 (Table 19). Affected tree species included one shagbark hickory (*Carya ovate*) and one paper birch (*Betula papyrifera*) in the urban zone and one American beech (*Fagus grandifolia*) in the rural zone (Table 20).

Gypsy moth (*Lymantria dispar*) was the most prevalent pest/disease in the region affecting a low of 0.7% of trees in 2010 and 2011 and a high of 7% in 2013. Tree species affected by gypsy moth were quite variable with sugar maple most commonly having eggs or larvae present. Gypsy moths were a greater problem in urban sites affecting 11% of trees compared to rural sites affecting 9% of trees.

Beech bark disease and the associated vector, beech scale (*Cryptococcus fagisuga*), affected 94% of beech trees in 2010 and 90% in 2014. Rural sites and urban sites across all years at some point had 100% of trees affected by the disease and/or scale.

Ash yellows affected between 0 and 8.7% of ash trees between 2010 and 2014 peaking in 2012. The only tree species affected by ash yellows were white ash and these were affected in equal proportions in rural compared to urban sites.

Table 19. Occurrence of four selected identified pests and diseases in TRCA forest plots between 2010 and 2014.

Pest/disease		2010	2011	2012	2013	2014
Anthracnose	# live stems affected	0	0	0	2	1
	% live stems affected	0	0	0	0.4	0.2
Gypsy moth	# live stems affected	4	4	14	39	9
	% live stems affected	0.7	0.7	2.5	7.0	1.6
Beech bark disease/scale	# live stems affected	16	16	19	18	18
	% of live beech stems affected	94	84	100	95	90
Ash yellows	# live stems affected	1	1	2	1	0
	% of live ash stems affected	4.4	4.6	8.7	4.4	0

Table 20. Occurrence of four selected pests and diseases in TRCA forest plots between urban and rural land use zones.

Pest/disease		Urban	Rural
Anthracnose	# live stems affected	2	1
	% live stems affected	0.7	0.4
Gypsy moth	# live stems affected	33	23
	% live stems affected	11	9
Beech bark disease/scale	# live stems affected	15	5
	% live beech stems affected	100	100
Ash yellows	# live stems affected	1	1
	% live ash stems affected	9.1	8.3

3.2.2 Forest Birds

Temporal Trends

Temporal trends were examined for the number L1-L3 bird species, species richness of forest-dependent birds and abundance of forest-dependent birds. Bird species were defined as forest-dependent based on their nesting requirements in forest or swamp habitat (Appendix Table A.1). Forest bird communities were relatively stable between 2008 and 2014 based on high-level indicators (Figure 16, Table 21). The number of L1-L3 forest bird species in the rural zone is showing signs of decline but this should be cautiously interpreted because the results were approaching significance but not significant at the $p<0.05$ level.

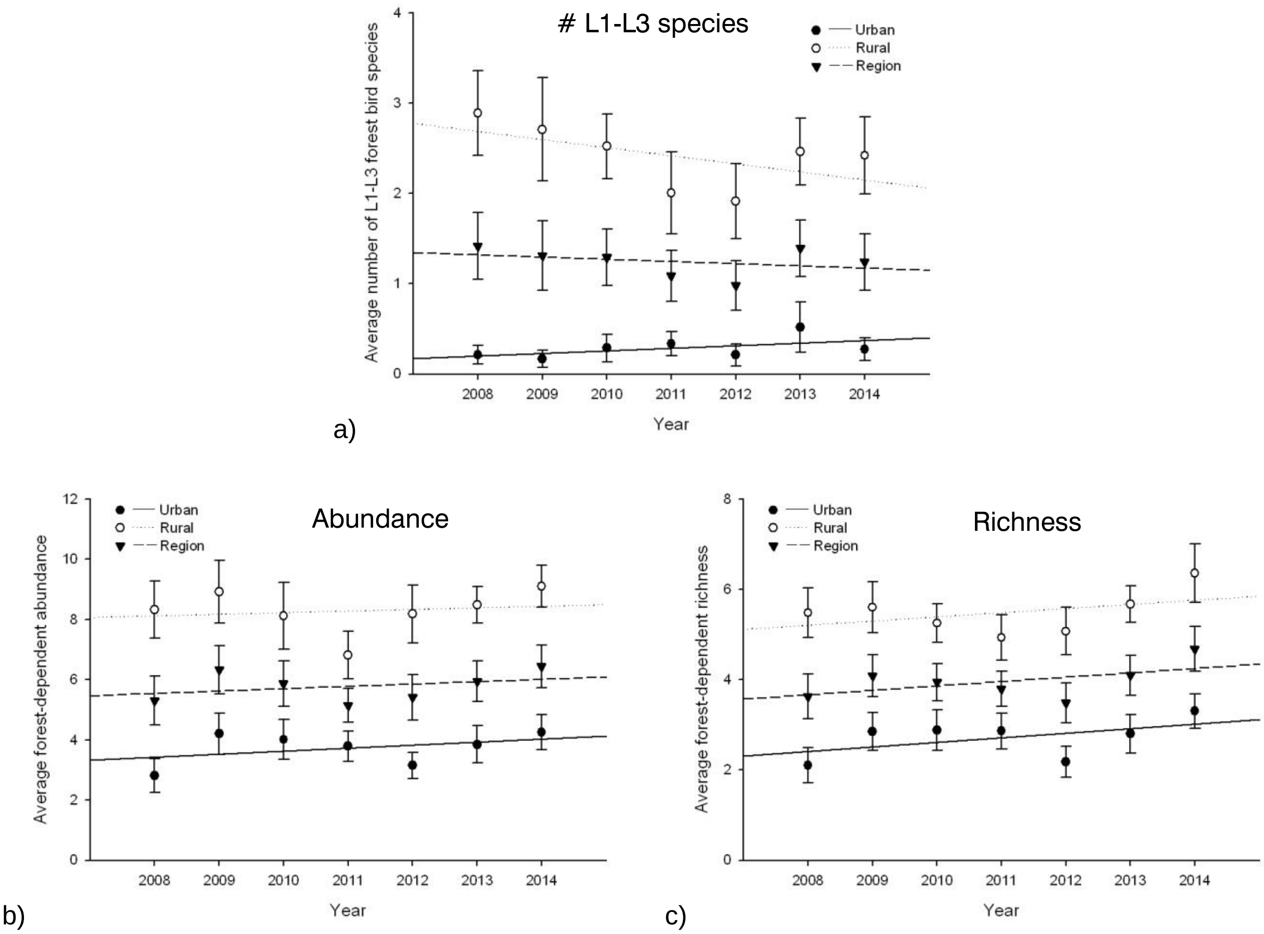


Figure 16. Temporal trends in forest bird high-level indicators a) number of L1-L3 bird species, b) forest-dependent bird abundance and c) forest-dependent bird richness. Trends are shown for the region and urban and rural land use zones.

Table 21. Statistical results for temporal trends in forest bird high-level indicators (2008-2014).

Metric	Land use	S-value	P-value	Increase, decrease or stable?
Average number of L1-L3 forest bird species	Region	-1.202	0.230	Stable
	Urban	1.051	0.293	Stable
	Rural	-1.802	0.072	↓ Caution
Average forest-dependent abundance	Region	0.901	0.368	Stable
	Urban	0.601	0.548	Stable
	Rural	0.601	0.548	Stable
Average forest-dependent richness	Region	0.901	0.368	Stable
	Urban	0.901	0.368	Stable
	Rural	0.601	0.548	Stable

Spatial Trends

Forest bird communities are showing significant impacts due to urbanization (Figure 17, Table 22). On average, rural sites contained two L1-L3 species while urban sites often contained zero or one L1-L3 species. There were also fewer forest-dependent individuals in urban sites (4.0) compared to rural sites (7.4). The number of forest-dependent species also changed with urbanization where urban sites contained on average 3.1 forest-dependent species and rural sites contained on average 5.1 forest-dependent species.

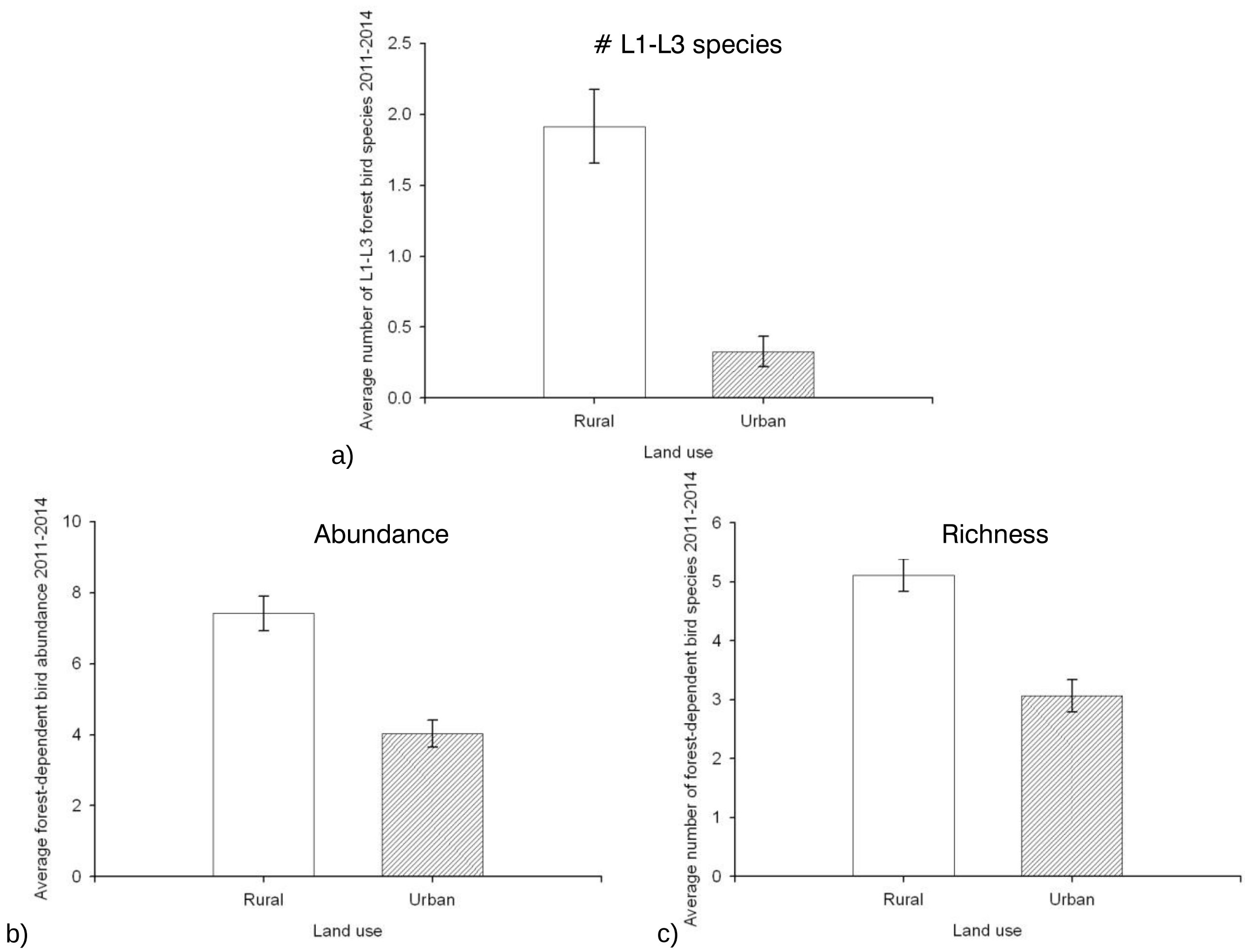


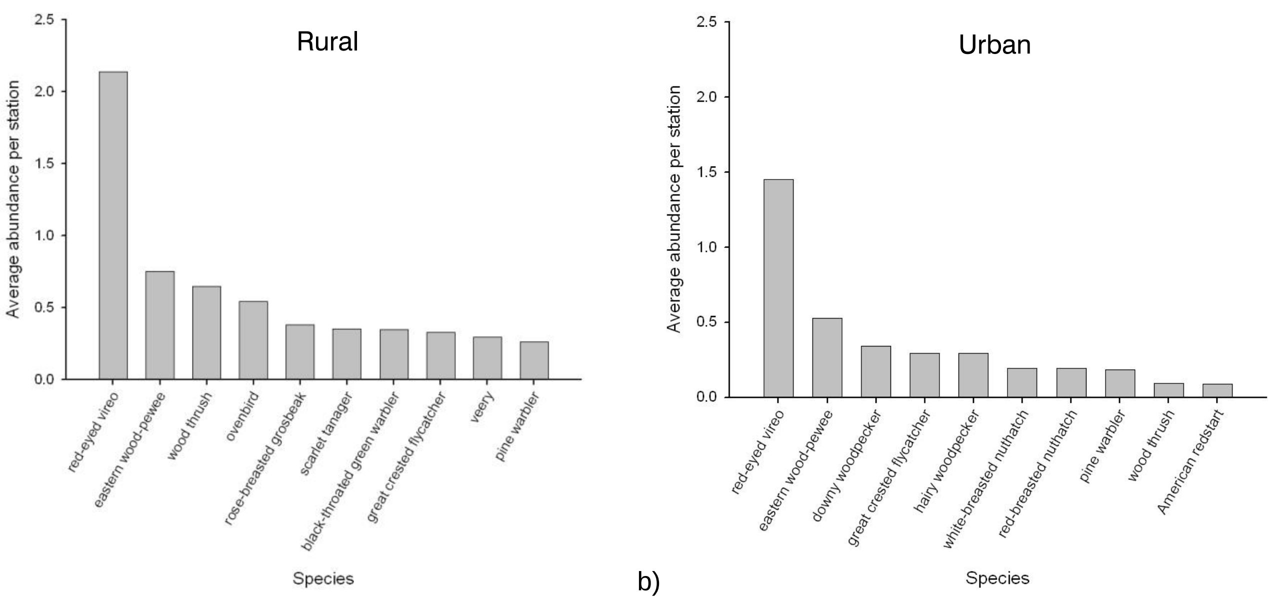
Figure 17. Spatial trends in forest bird high-level indicators a) number of L1-L3 bird species, b) forest-dependent bird abundance and c) forest-dependent bird richness. The number of L1-L3 forest bird species data were log+1 transformed due to large normality and equal variance violations; however, original values are presented.

Table 22. Statistical results for spatial trends in forest bird high-level indicators.

Metric	t-value	P-value	Effect of urbanization
Number of L1-L3 species	6.46	<0.0001	↓
Forest-dependent bird abundance	5.54	<0.0001	↓
Forest-dependent bird species richness	5.31	<0.0001	↓

Forest Bird Community Composition

Forest bird communities differed based on the 10 most abundant species per station (Figure 18). Red-eyed vireo (*Vireo olivaceus*) and eastern wood pewee (*Contopus virens*) were the most dominant forest bird species in both the rural and urban zones, although the abundance per station for both species was lower in urban forests. Ovenbird (*Seiurus aurocapillus*), an L2 species, was ranked 4th in abundance in the rural zone but was ranked 21st in abundance in the urban zone. Wood thrush (*Hylocichla mustelina*), an L3 species, ranked higher in abundance in the rural zone compared to the urban zone.



a) b)

Figure 18. Forest bird community composition based on average abundance per station of the 10 most abundant species in the a) rural land use zone and b) urban land use zone.

3.2.3 Forest Plethodontids

TRCA is one of only two known urban salamander monitoring programs in Ontario. The majority of salamander monitoring occurs in more pristine forests such as National Parks or other forest preserves throughout the province. Starting the salamander monitoring program in the TRCA region has been challenging and a learning experience. In the first years of monitoring, concerns were raised over the safety of salamanders using the double artificial cover boards. Since these concerns were raised, efforts were made to transition to monitoring using single boards (Table 23). By 2014, all sites had single cover boards for monitoring salamanders. In addition to these concerns, there were major issues with cover board disturbance but these disturbances were mostly isolated to 2011. During this year, 13 of 25 sites had many boards either moved or flipped by either racoons or humans and the disturbances were distributed evenly between urban and rural sites. Since 2011, there have been no further disturbances to that extent.

Table 23. Modifications to cover board type based on site and year.

Years surveyed and cover board type	Site
Single boards 2009-2014	4 Downsview 7 TWM #14 8 Caledon 9 Bolton 12 Wilket 13 Bakers 15 Morningside 17 Ressor 19 Duffins heights 22 Duffins marsh 23 Bruces mill 27 South Kirby
Single boards 2012-2014	31-34 (Peel, Centennial, Claremont, Stouffville)
Double boards 2009-2013 (changed to single 2014)	2 Heart Lake 11 West Gormley 16 Altona 21 Glen Major
Double boards 2009-2011 (changed to single 2012)	6 Boyd
Double boards 2009 (changed to single 2010-2014)	24 Palgrave
Single boards 2014	35A Mayfield West
Single boards (2011-2014)	28 Eglinton and Tomken

No statistical analysis was done on salamander data collected from 2009-2014 to analyze temporal and spatial trends due to the methodological changes made over the years during the start-up of the program (mentioned above). These changes will make the salamander monitoring program more scientifically accurate and safe for salamanders using cover boards in the future. Even though there were methodological inconsistencies, TRCA data were used to determine baseline salamander abundance and variance required for power analysis.

Due to the methodological changes ceasing by 2014, this year should be used as a baseline year for monitoring trends in salamander populations (Table 24). Trends in salamander populations will be monitored over time using the average total number of salamanders under all ACO's at a single site. It was recommended by Zorn (2008) that an average baseline abundance of 20 salamanders was needed to be able to detect a 15% decline in salamander populations over five years. Based on these data, TRCA has an average baseline salamander abundance of 11.3 resulting in a decrease in statistical power to detect the same trends recommended by Zorn (2008). We are now able to detect a decline of 20% over 5 years with a statistical power of 85.6%.

Table 24. Baseline average number of red-backed salamanders in 2014. A total was first calculated for each of the five site visits in the spring and an average was taken to determine a final value per site.

Plot ID	Site	Average number of red-backed salamanders
FS-4	Downsview Dells	37.8
FS-17	Reesor Rd & Hwy #7	26.8
FS-9	Bolton Tract	24.4
FS-6	Boyd	24.2
FS-2	Heart Lake	18.8
FS-11	West Gormley	17.6
FS-15	Morningside Park	16.2
FS-35A	Mayfield West	16.2
FS-33	Claremont	15.4
FS-13	Bakers Sugar Bush	14.4
FS-23	Bruces Mill	9
FS-31	Peel Tract	8.8
FS-21A	Glen Major B	8.4
FS-8	Caledon Tract	8.4
FS-7	TVM #14	8
FS-34	Stouffville Reservoir	6.8
FS-21	Glen Major A	4.8
FS-32	Centennial Park	4.8
FS-16	Altona Forest	3.4
FS-19	Duffin Heights	2.8
FS-27	Kirby and Keele	2.4
FS-24	Palgrave	2.2
FS-12	Wilket Creek Park	0.4
FS-22	Duffins Marsh Woods	0.4
FS-28	Eglinton & Tomken	0.4

Now that there are several years of data, the data were explored to make changes to the program that will save resources while still maintaining data quality. Salamander monitoring involves visiting each site five times in the spring and obtaining an average abundance value based on these five visits. Data collected to date were used to answer the question: what would happen to average salamander abundance values if only four visits were made by excluding the first visit? When the first visit was removed, the data showed no major changes in average abundance in 2012, 2013 and 2014 (Figure 19).

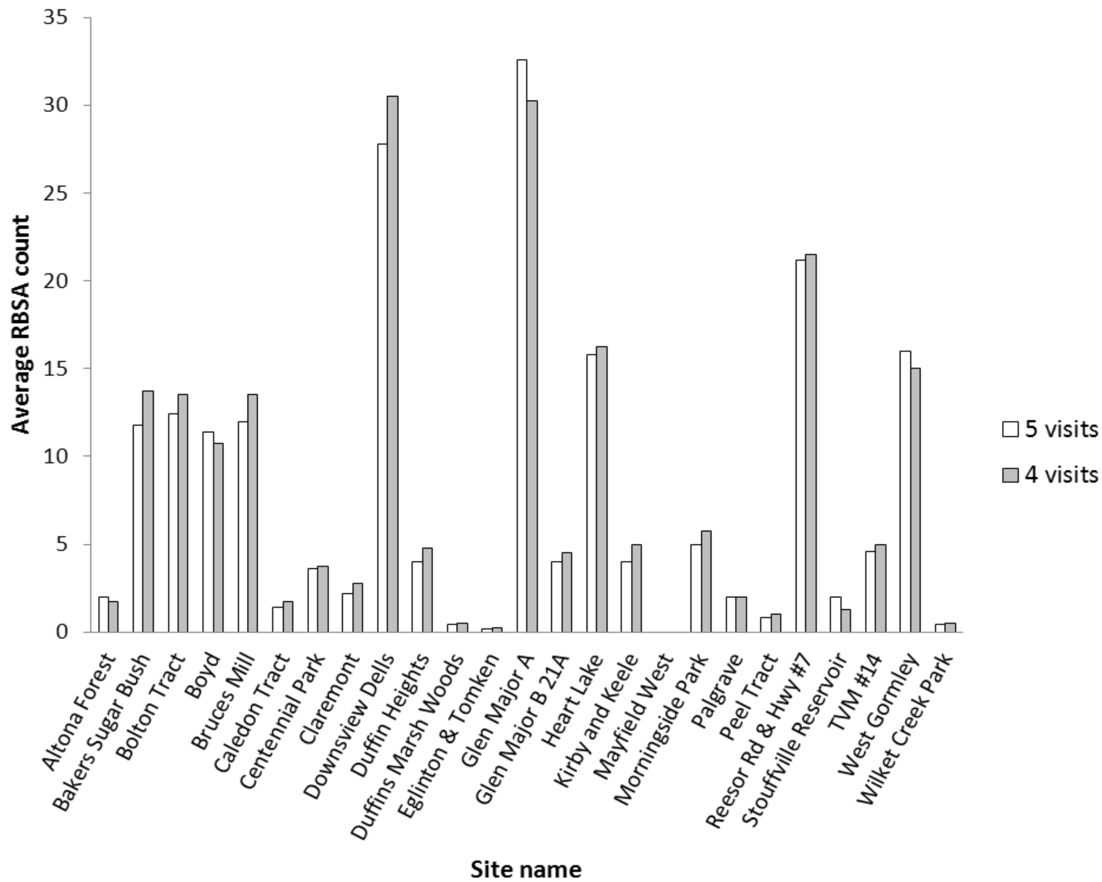


Figure 19. Average salamander abundance based on 4 vs. 5 visits to 25 sites across TRCA jurisdiction in 2012.

3.3 Wetland Monitoring

3.3.1 Wetland Vegetation

Similar to forest vegetation, we expect to have some variability temporally in the early years of monitoring wetland flora communities and therefore temporal trends were not examined in this report. A baseline for tracking temporal trends in these variables will be established once the vegetation identified has stabilized.

Spatial Trends

There were significant impacts of urbanization on wetland vegetation high-level indicators (Figure 20, Table 25). Both the number of L1-L3 species and the percentage of native wetland vegetation species was significantly lower in urban wetlands compared to rural wetlands. Urban wetlands on average had 2.3 species of concern (L1-L3) while rural wetlands supported 5.7 species of concern. Urban wetlands also had fewer native species (60.8%) than rural wetlands (78.3%). The FQI did not show statistically significant differences between urban and rural sites although there was a tendency for rural sites to have a higher FQI (23.2) than urban sites (16.9).

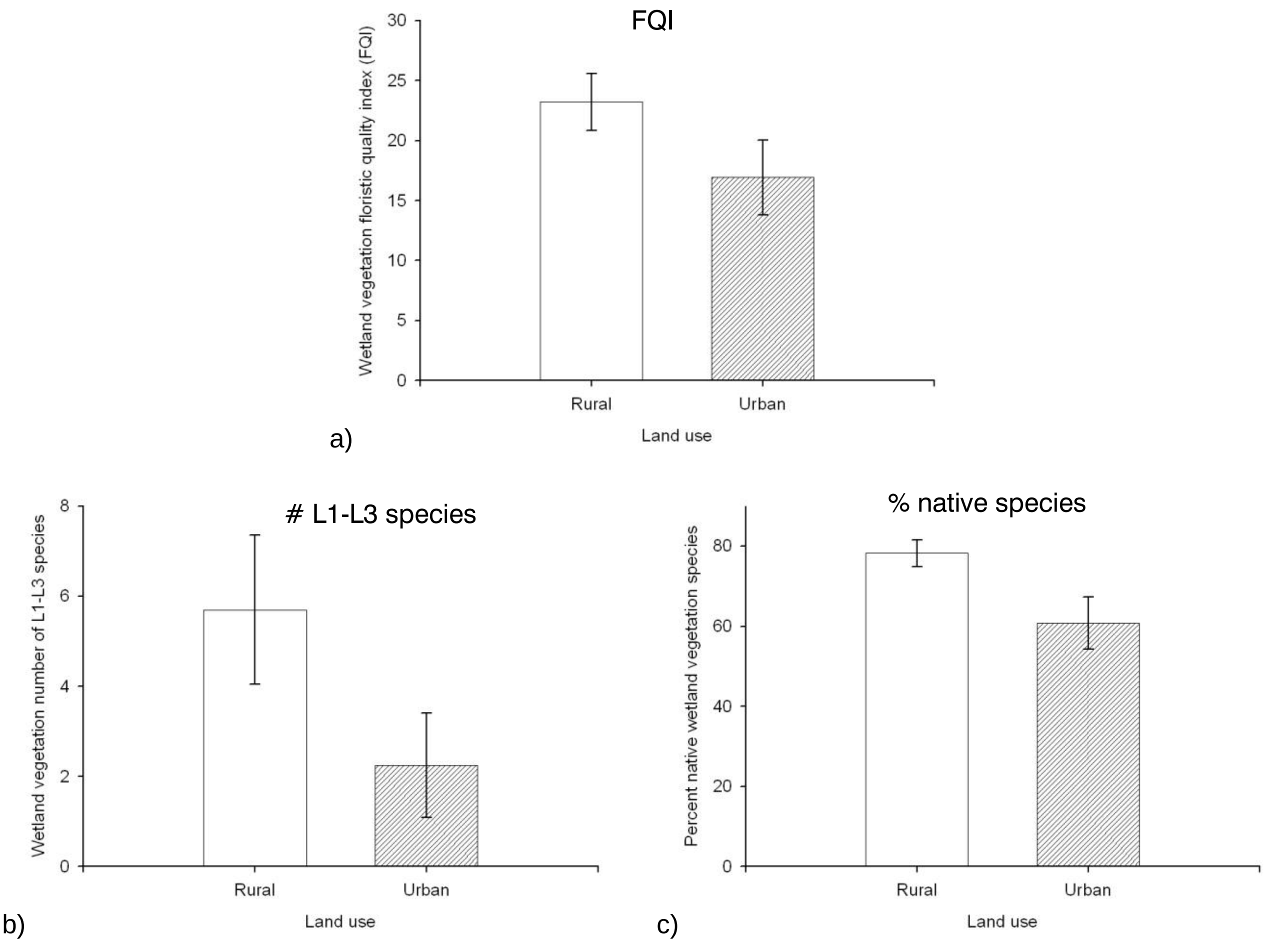


Figure 20. Spatial trends in wetland vegetation high-level indicators a) Floristic Quality Index (FQI), b) number of L1-L3 wetland vegetation species and c) percent native wetland vegetation species. The number of L1-L3 species was log-transformed to improve normality; however, original values are presented.

Table 25. Statistical results for spatial trends in wetland vegetation high-level indicators.

Metric	t-value	P-value	Effect of urbanization
Floristic Quality Index (FQI)	1.63	0.120	Non-significant
Number of L1-L3 species	2.76	0.013	↓
Percent native species	2.44	0.026	↓

Wetland Woody Regeneration Species Composition

Relative percent cover of wetland woody regeneration across the jurisdiction was dominated by speckled alder (*Alnus incana* ssp. *rugosa*; 15%), red osier dogwood (*Cornus stolonifera*; 13%) and balsam fir (*Abies balsamea*; 11%). These rankings changed when species composition was measured using stem counts (relative abundance). Red osier dogwood had the highest relative abundance (17%) following by common buckthorn (16%), winterberry (*Ilex verticillata*; 6.6%) and speckled alder (6.3%) as the top four species.

The urban and rural land use zones varied based on the species composition of woody regeneration (Figure 21). Relative cover in the rural zone was dominated by balsam fir (18%), red osier dogwood (17.7%) and speckled alder (9%). This is in contrast to the urban zone where speckled alder (25%), common buckthorn (12%) and sugar maple (6%) dominated the woody regeneration community.

Species composition based on relative abundance showed that the woody regeneration communities were dominated by one species that had a stem count that was quite higher than all other species. In the rural zone, wetlands were dominated by red osier dogwood (27%) while in urban zones wetlands were dominated by common buckthorn (24%).

A major concern is a large increase in both the relative percent cover and the relative abundance of common buckthorn between 2009 and 2014 in wetland vegetation plots throughout the Region (Table 26). Even when the driest sub-plots (1 and 2) were removed, these trends were still apparent.

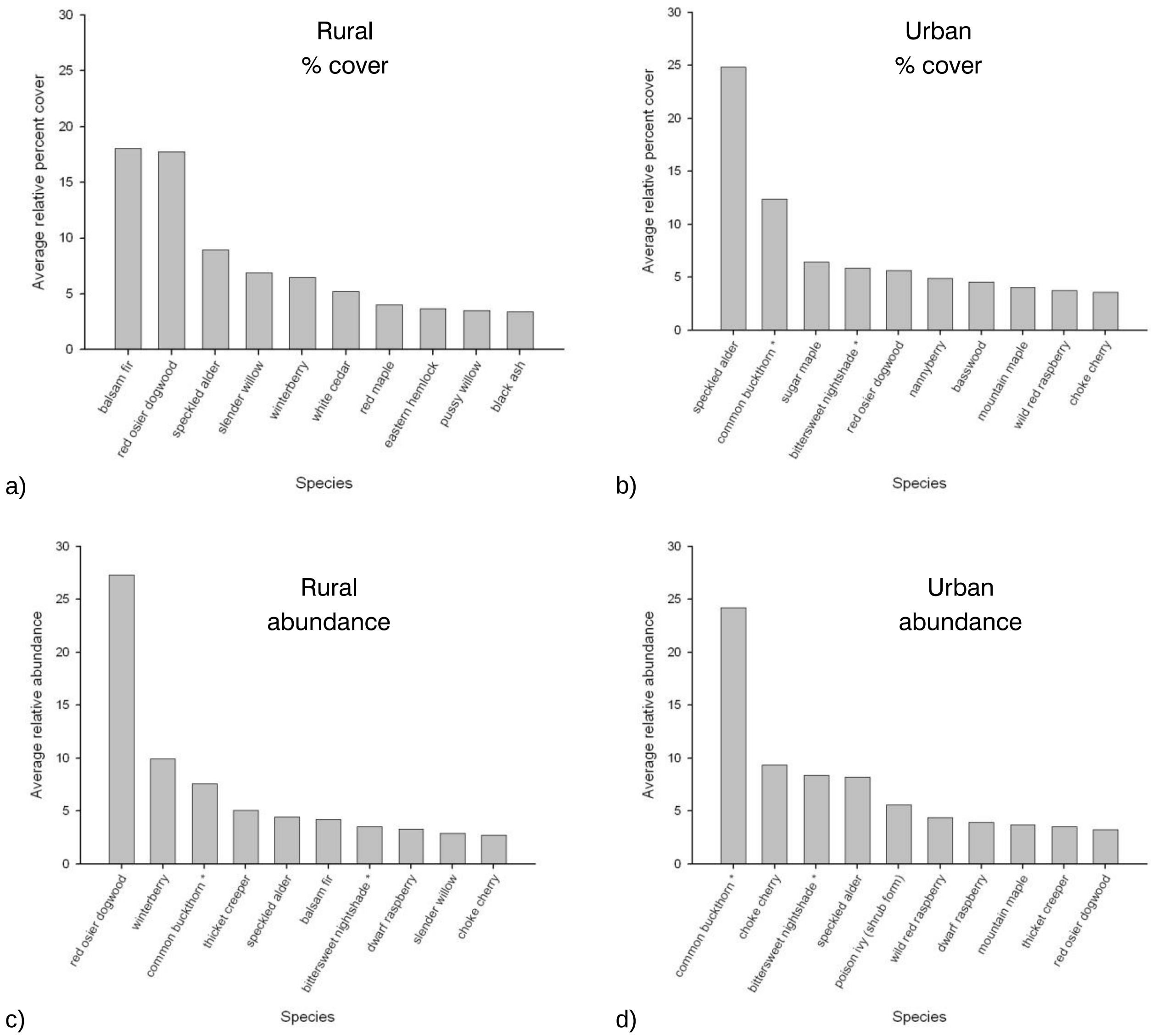


Figure 21. Average relative percent cover and abundance of the 10 most common wetland woody species in the rural zone (a, c) and in the urban zone (b, d). Stem count data were only available between 2009-2014. Exotic species are indicated with an asterisk (*).

Table 26. Changes in relative percent cover and relative abundance for common buckthorn in wetland woody regeneration plots throughout the Region.

Common buckthorn	2008	2009	2010	2011	2012	2013	2014
Relative % cover	1.4	3.0	3.9	7.3	7.4	9.4	10.1
Relative abundance	-	3.4	7.1	13.9	14.7	20.2	26.8

Wetland Ground Vegetation Species Composition

In 2013 flora biologists discovered that much of the previously identified common duckweed (*Lemna minor*) may have actually been turion duckweed (*Lemna turionifera)*. Therefore, in 2014 turion and common duckweed were distinguished in the data; however, prior to 2014 these were identified as the same species. Due to this variation in species identification, turion duckweed and common duckweed were grouped for this analysis.

Wetland ground vegetation was dominated by hybrid cattail (*Typha* x *glauca*; 19%) at the regional level followed by turion/common duckweed (13%), reed canary grass (*Phalaris arundinacea*; 6%) and common reed (*Phragmites australis* ssp. *australis*; 5%). Rural sites had a higher species richness of species with a relative cover of >1% (18 species) than urban sites (11 species; Figure 22). Emergent vegetation in rural wetlands was dominated by common reed (10%) while urban sites were dominated by hybrid cattail (33%). Turion/common duckweed was prominent in both rural (13%) and urban wetlands (13%).

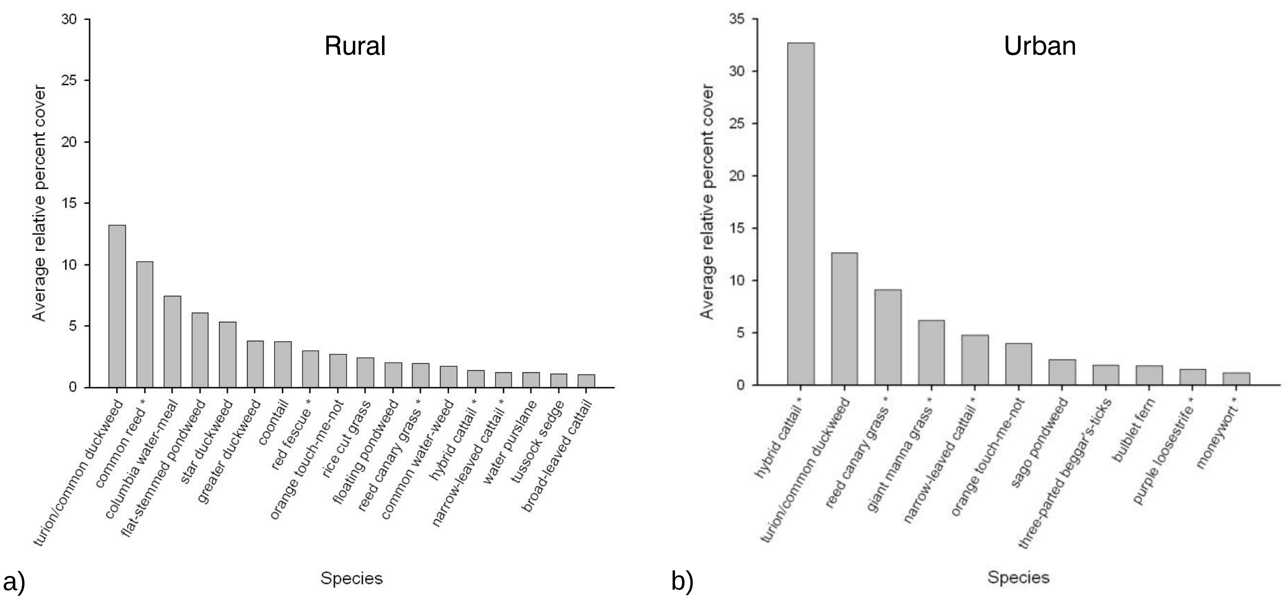


Figure 22. Average relative percent cover of ground vegetation (herbaceous) species with more than 1% of total a) in the rural zone and b) in the urban zone.

3.3.2 Wetland Birds

Temporal Trends

Wetland bird communities were stable between 2009 and 2014 (Figure 23, Table 27). None of the wetland bird high-level indicators showed any sign of decline in either land use zone or across the region. Regionally, there was an increase in the number of L1-L3 wetland bird species but it was not statistically significant ($p>0.05$).

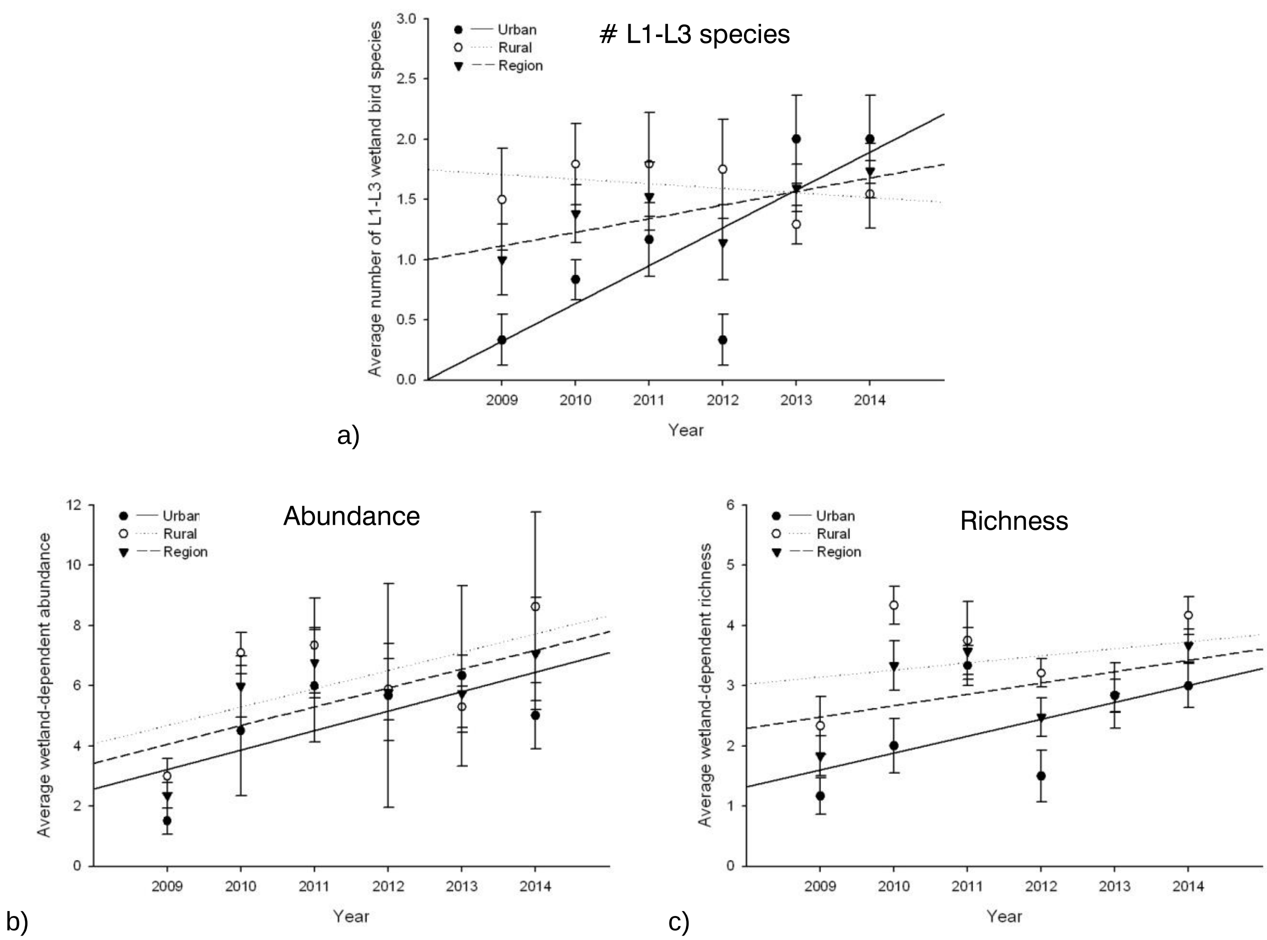


Figure 23. Temporal trends in wetland bird high-level indicators a) number of L1-L3 bird species, b) wetland-dependent bird abundance and c) wetland-dependent bird richness. Trends are shown for the region and urban and rural land use zones.

Table 27. Statistical results for temporal trends in wetland bird high-level indicators (2009-2014).

Metric	Land use	S-value	P-value	Increase, decrease or stable?
Average number of L1-L3 wetland bird species	Region	1.879	0.060	↑ Caution
	Urban	1.503	0.133	Stable
	Rural	-0.564	0.573	Stable
Average wetland-dependent abundance	Region	0.751	0.452	Stable
	Urban	1.127	0.260	Stable
	Rural	0.751	0.452	Stable
Average wetland-dependent richness	Region	1.127	0.260	Stable
	Urban	1.127	0.260	Stable
	Rural	0.000	1.000	Stable

Spatial Trends

There were no significant impacts of urbanization on wetland bird high-level indicators (Figure 24, Table 28). Even though the results were not statistically significant, the number of L1-L3 species along with bird abundance and richness, all had lower mean values in urban wetlands compared to rural wetlands.

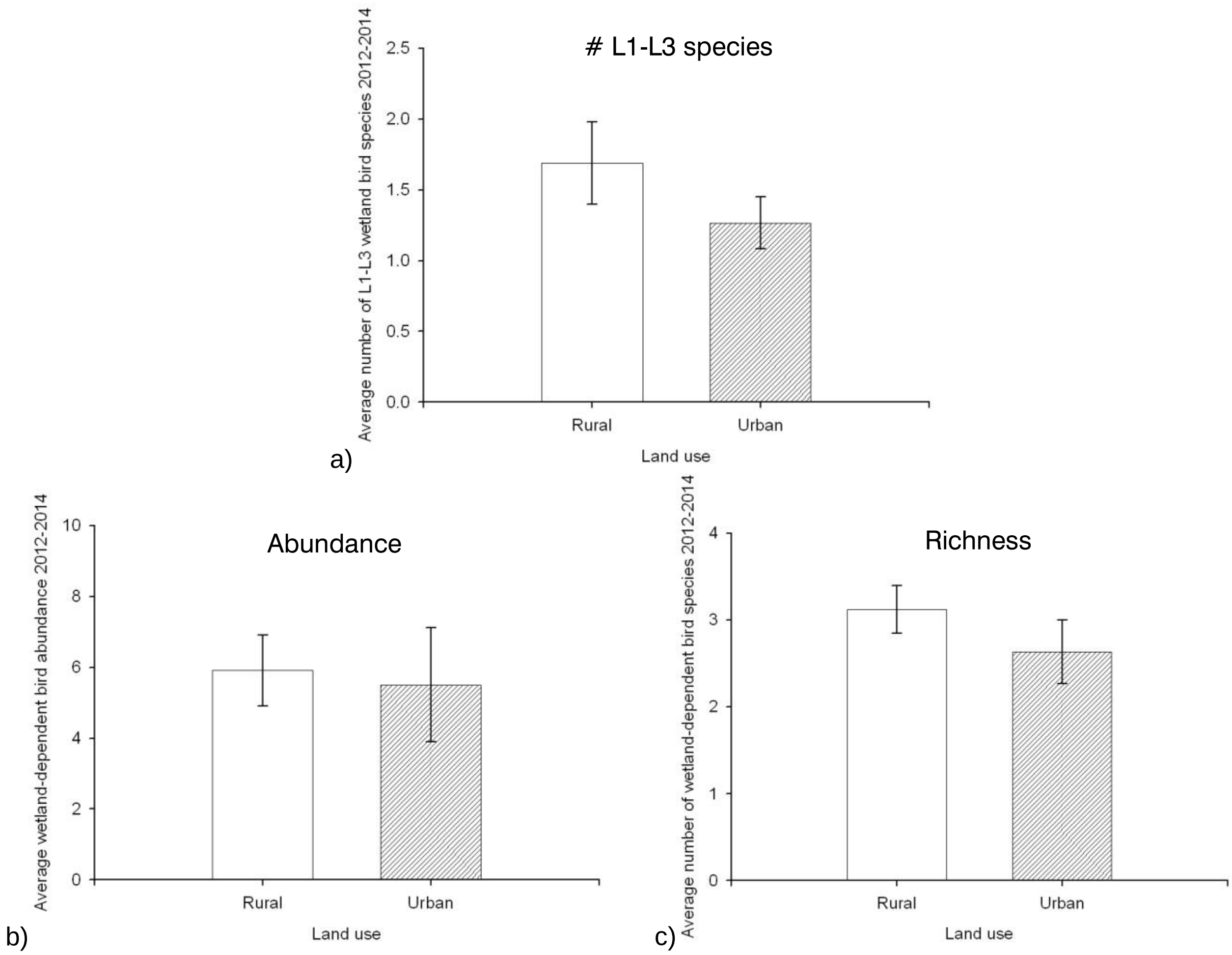


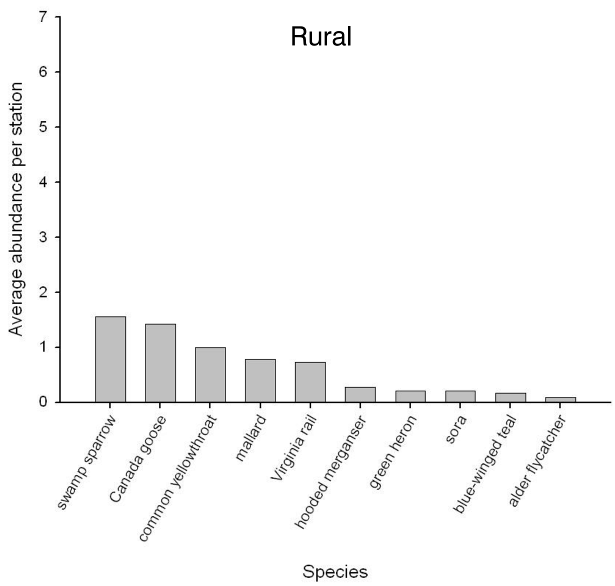
Figure 24. Spatial trends in wetland bird high-level indicators a) number of L1-L3 bird species, b) wetland-dependent bird abundance and c) wetland-dependent bird richness.

Table 28. Statistical results for spatial trends in wetland bird high-level indicators.

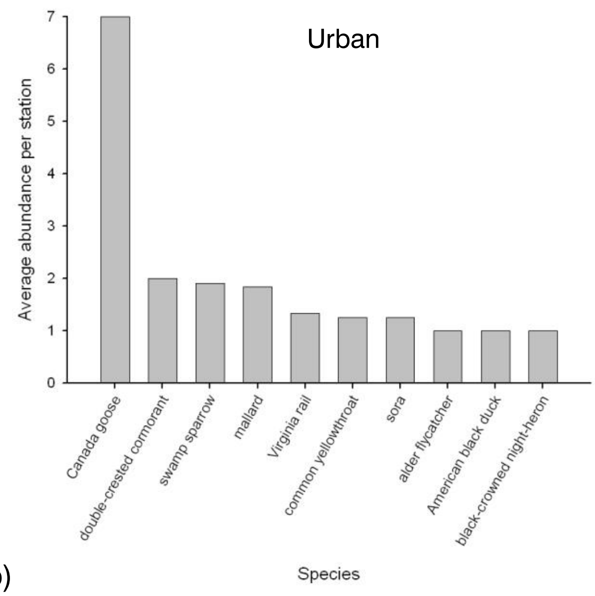
Metric	t-value	P-value	Effect of urbanization
Number of L1-L3 species	1.23	0.239	Non-significant
Wetland-dependent bird abundance	0.217	0.831	Non-significant
Wetland-dependent bird species richness	1.08	0.297	Non-significant

Wetland Bird Community Composition

Wetland bird communities varied based on land use zone (Figure 25). Canada goose (*Branta canadensis*) was over three times more abundant in the urban zone compared to the rural zone. Virginia rail (*Rallus limicola*) and sora (*Porzana carolina*), two L3 species, were found both in the urban and rural zones.



a)



b)

Figure 25. Wetland bird community composition based on average abundance per station of the 10 most abundant species in the a) rural land use zone and b) urban land use zone.

3.3.3 Frogs

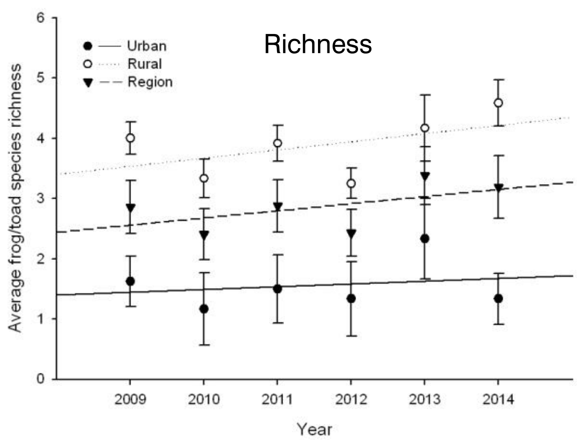
Frog Species Composition

Eight frog species were detected in regional plots across the jurisdiction between 2009 and 2014 including green frog (*Lithobates clamitans*), spring peeper (*Pseudacris crucifer crucifer*), wood frog (*Lithobates sylvatica*), tetraploid grey treefrog (*Hyla versicolor*), northern leopard frog (*Lithobates pipiens*), American toad (*Anaxyrus americanus*), chorus frog (*Pseudacris triseriata*) and bullfrog (*Lithobates catesbeiana*; in descending order of abundance measured as the percent of sites occupied). Bullfrog was only detected in 2013 (Duffins Marsh, Palgrave and Bolton Tract West) and chorus frog was only detected in a limited number of sites each year (Wildwood Park and Claireville) and were therefore excluded from statistical analysis.

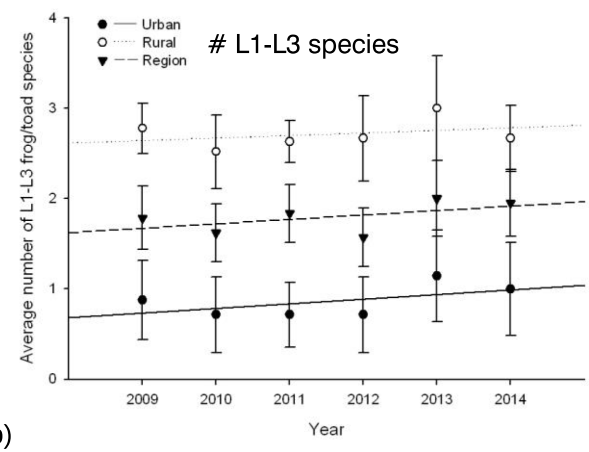
Temporal Trends

Temporal trends for frog communities were measured using three high-levels indicators: species richness, number of L1-L3 species and the proportion of sites occupied. Proportion of sites occupied is used as a surrogate measure of abundance since abundance estimates cannot be determined using MMP protocols. Temporal trends in the proportion of sites occupied were analyzed for all frog species combined and by individual species. Temporal trends were also determined for “no species” which is recorded by the field surveyor in the event that there were no frogs detected at a site.

Frog communities were stable between 2009 and 2014 based on measurements using the high-level indicators (Figure 26). There were no statistically significant increases or decreases in frog species richness or the number of L1-L3 species across the region or in the urban or rural zones (Table 29).



a)



b)

Figure 26. Temporal trends in frog high-level indicators a) number of frog species and b) number of L1-L3 species. Trends are shown for the region and urban and rural land use zones.

Table 29. Statistical results for temporal trends in frog high-level indicators (2009-2014).

Metric	Land use	S-value	P-value	Increase, decrease or stable?
Average frog/toad species richness	Region	1.127	0.260	Stable
	Urban	0.564	0.573	Stable
	Rural	0.751	0.452	Stable
Average number of L1-L3 frog/toad species	Region	0.751	0.452	Stable
	Urban	1.127	0.260	Stable
	Rural	0.939	0.348	Stable

Percent of Sites Occupied

A total of 86% of sites were occupied by at least one frog species between 2009 and 2012. This rose to 93% of sites occupied between 2013 and 2014; however, this increase was not statistically significant ($S = 1.32$, $p = 0.19$). The percent of sites occupied by specific frog species was relatively stable between 2009 and 2014 (Figure 27, Table 30). There were decreases in the percent of sites occupied by wood frogs and tetraploid grey tree frogs between 2009 and 2014 although these results were not statistically significant. Even though the decrease was not statistically valid, it is important to use this qualitative observation as a potential warning of declines in these species.

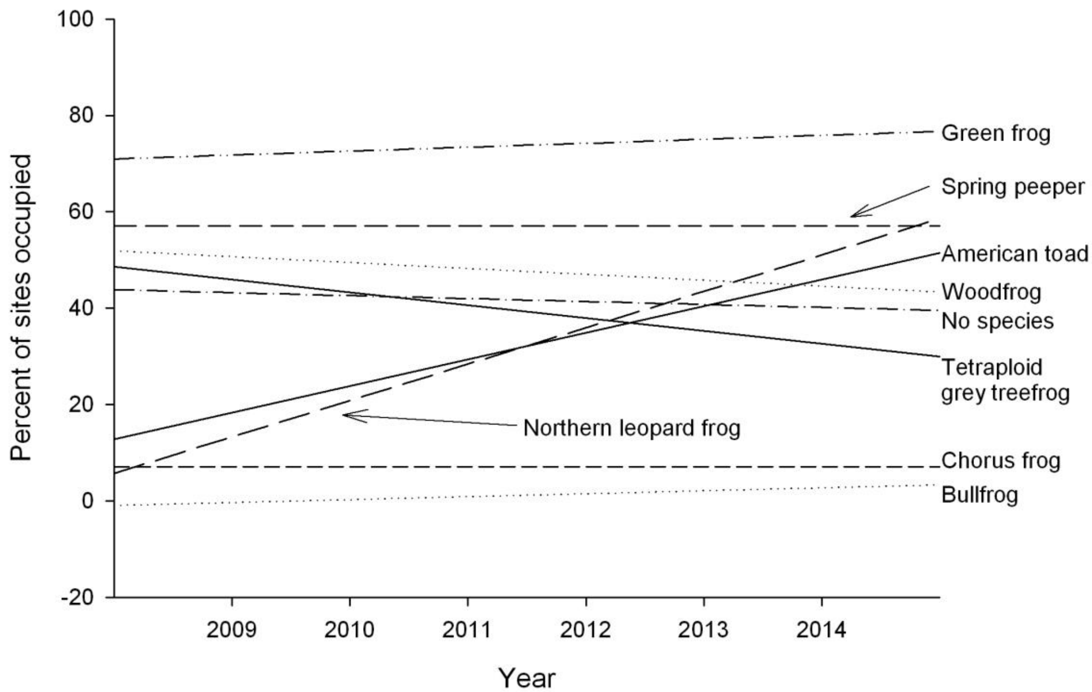


Figure 27. Temporal trends in the percent of sites occupied by specific frog species. "No species" indicates the percent of sites that the field surveyor recorded no species present.

Table 30. Statistical results for temporal trends in individual frog species (2009-2014).

Frog/toad species	S-value	P-value	Increase, decrease or stable?
American toad	1.691	0.091	↑ (Caution)
Green frog	0.000	1.000	Stable
Northern leopard frog	1.879	0.060	↑ (Caution)
Spring peeper	0.000	1.000	Stable
Tetraploid grey treefrog	-0.751	0.452	Stable
Wood frog	-0.939	0.348	Stable
No species	0.000	1.000	Stable

Spatial Trends

Frog high-level indicators were significantly affected by urbanization (Figure 28, Table 31). Frog species richness was significantly lower in urban wetlands (2.2 species) compared to rural wetlands (4.5 species). The number of L1-L3 frog species was also significantly lower in urban wetlands (1.2 species) compared to rural wetlands (2.9 species).

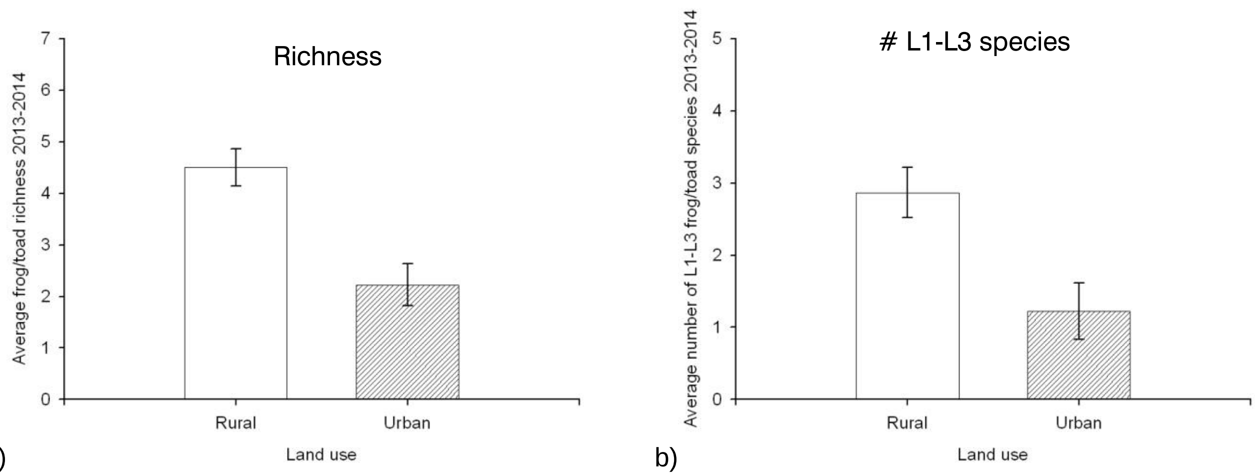


Figure 28. Spatial trends in frog high-level indicators a) number of frog species and b) number of L1-L3 species.

Table 31. Statistical results for spatial trends in frog high-level indicators.

Metric	t-value	P-value	Effect of urbanization
Frog species richness	4.17	<0.001	↓
Number of L1-L3 frog species	3.15	<0.01	↓

In general, percent site occupancy was high between urban and rural sites with 90% of urban sites containing at least one frog species and 100% of rural sites containing at least one frog species. Ninety percent of urban sites (9/10) and 10%(1/10) rural sites detected no species during at least one visit between 2013 and 2014. Spring peeper was extremely sensitive to urbanization with significantly higher occupancy rates in rural sites (90%) compared to urban sites (30%; Figure 29, Table 32). All other species had higher occupancy rates at rural compared to urban sites; however, these results were not statistically significant.

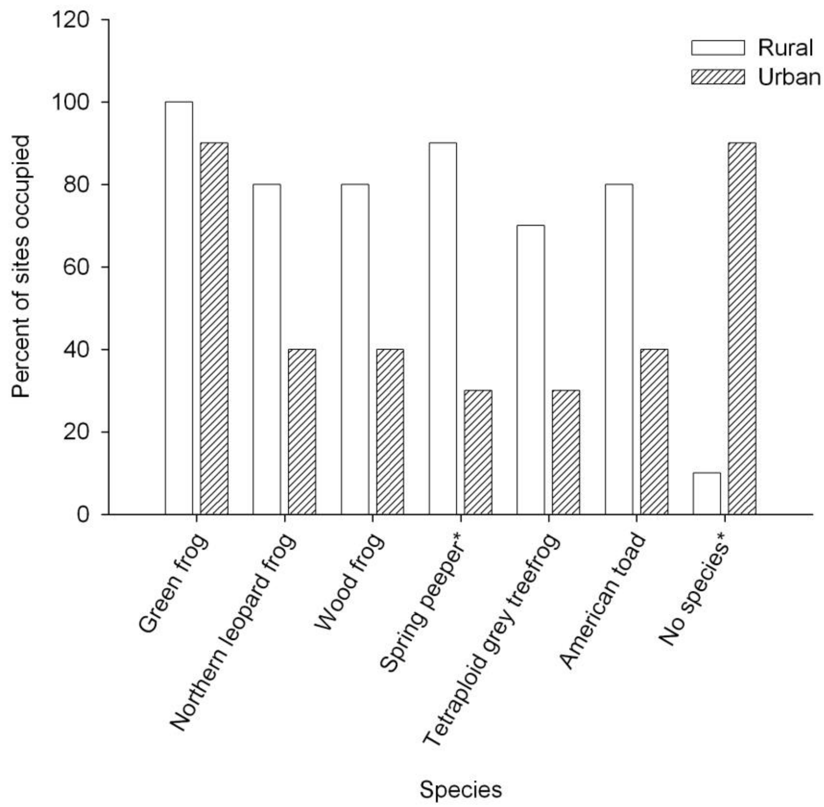


Figure 29. Percent of sites occupied by species in urban and rural land use zones. Significant differences between zones are indicated with an asterisk (*).

Table 32. Fisher's Exact Test results for frog species-specific differences in proportion of sites occupied between urban and rural land uses.

Species	Percent of urban sites occupied	Percent of rural sites occupied	Fisher's p-value	Effect of urbanization
Green frog	90	100	0.500	Non-significant
Northern leopard frog	40	80	0.085	↓ Caution
Wood frog	40	80	0.085	↓ Caution
Spring peeper	30	90	0.01	↓
Grey treefrog	30	70	0.089	↓ Caution
American toad	40	80	0.085	↓ Caution
No species	90	10	<0.001	↑

3.4 Meadow Monitoring

3.4.1 Meadow Birds

Temporal Trends

Meadow bird communities were relatively stable between 2008 and 2014 (Figure 30, Table 33). There was a significant increase in the number of L1-L3 species in the rural land use zone meaning that over time the community is containing a higher average number of L1-L3 ranked species; however, this is contradictory to a potential decrease in the abundance of meadow birds in all zones.

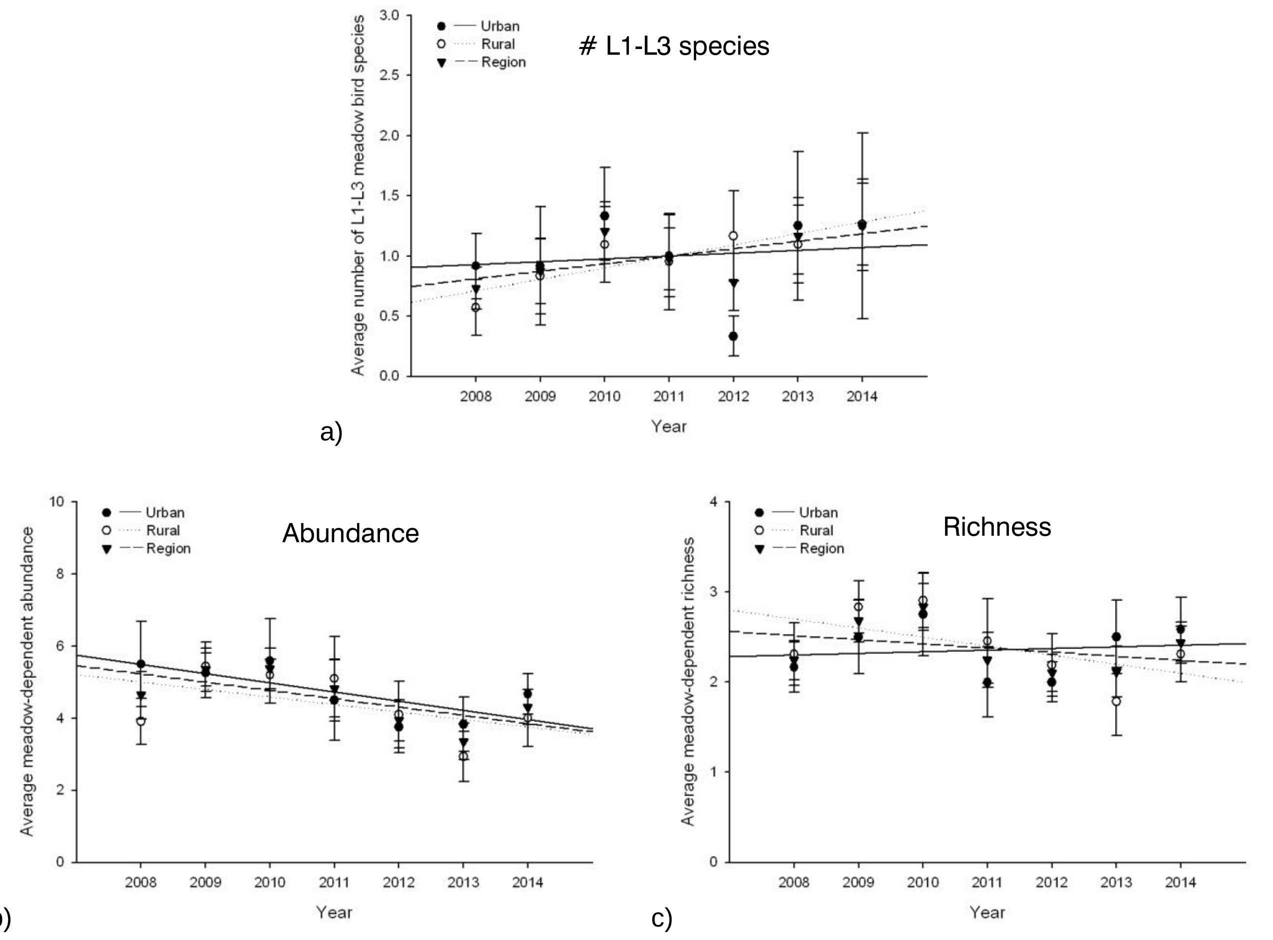


Figure 30. Temporal trends in meadow bird high-level indicators a) number of L1-L3 bird species, b) meadow-dependent bird abundance and c) meadow-dependent bird richness. Trends are shown for the region and urban and rural land use zones. Meadow birds L1-L3 species data were log+1 transformed for statistical analysis; however, original values are presented.

Table 33. Statistical results for temporal trends in meadow bird high-level indicators (2008-2014).

Metric	Land use	S-value	P-value	Increase, decrease or stable?
Average number of L1-L3 meadow bird species	Region	1.502	0.133	Stable
	Urban	0.601	0.548	Stable
	Rural	2.403	0.016	↑
Average meadow-dependent abundance	Region	-1.202	0.230	Stable
	Urban	-1.202	0.230	Stable
	Rural	-1.202	0.230	Stable
Average meadow-dependent richness	Region	-0.451	0.652	Stable
	Urban	0.300	0.764	Stable
	Rural	-1.051	0.293	Stable

Spatial Trends

Similar to wetland birds, meadow birds did not show any statistically significant effects of urbanization (Figure 31, Table 34). Even though there were no significant differences, there was a tendency for all high-level indicators to be lower in urban meadows. The largest difference was a decrease in the average number of species of concern (L1-L3) from 1.5 species in rural meadows to 0.7 species in urban meadows.

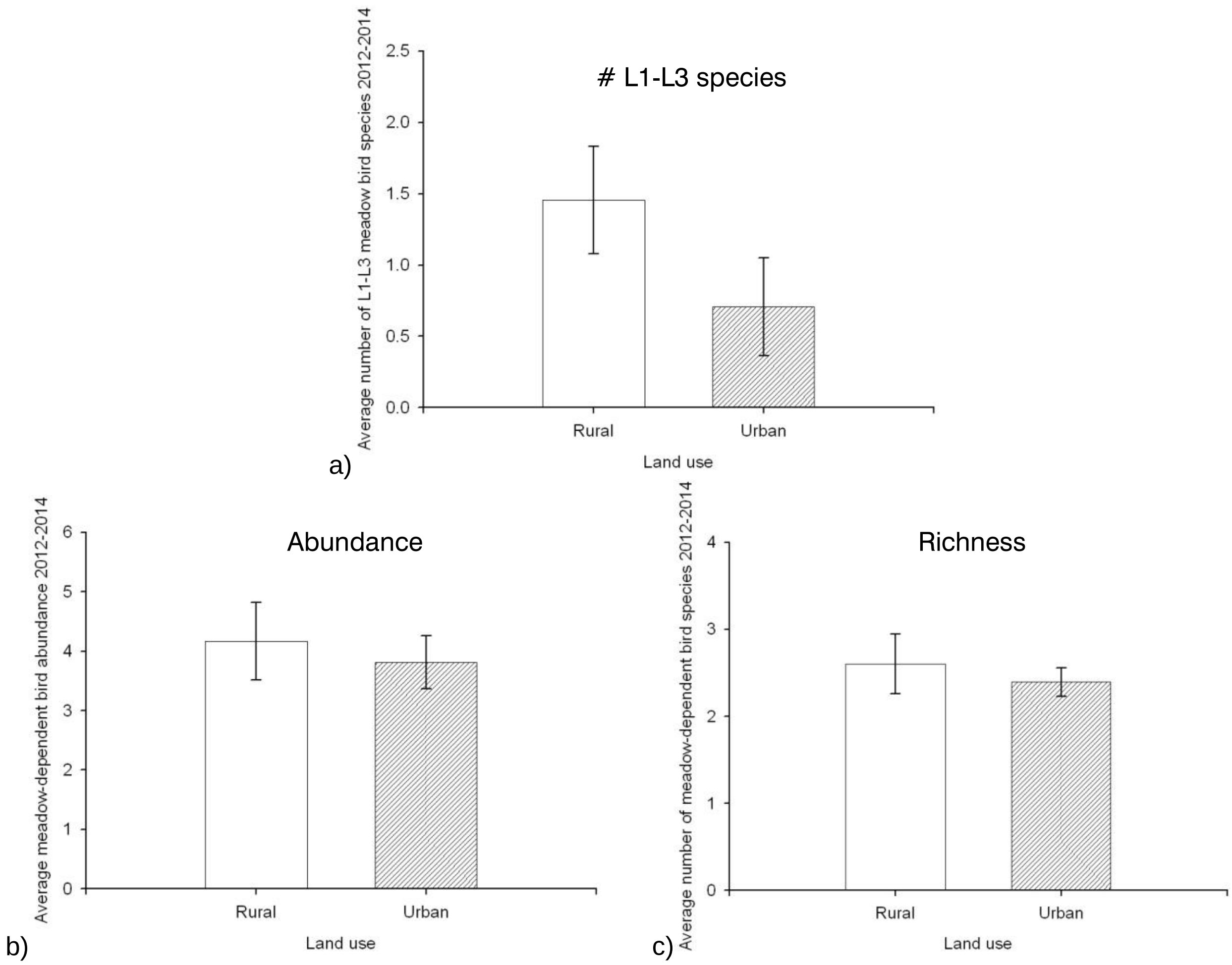


Figure 31. Spatial trends in meadow bird high-level indicators a) number of L1-L3 bird species, b) meadow-dependent bird abundance and c) meadow-dependent bird richness.

Table 34. Statistical results for spatial trends in meadow bird high-level indicators.

Metric	t-value	P-value	Effect of urbanization
Number of L1-L3 species	1.74	0.10	↓ Caution
Meadow-dependent bird abundance	0.455	0.655	Non-significant
Meadow-dependent bird species richness	0.537	0.601	Non-significant

Meadow Bird Community Composition

Meadow bird communities in the urban and rural zones had similar species composition (Figure 32). The only differences were grasshopper sparrow (*Ammodramus savannarum*; an L2 species) in the rural zone and spotted sandpiper in the urban zone. Savannah sparrow (*Passerculus sandwichensis*) clearly dominated the urban land use zone while both eastern kingbird (*Tyrannus tyrannus*) and field sparrow (*Spizella pusilla*) dominated the rural land use zones. Bobolink (*Dolichonyx oryzivorus*), an L2 species, was present in both the urban and rural land use zones.

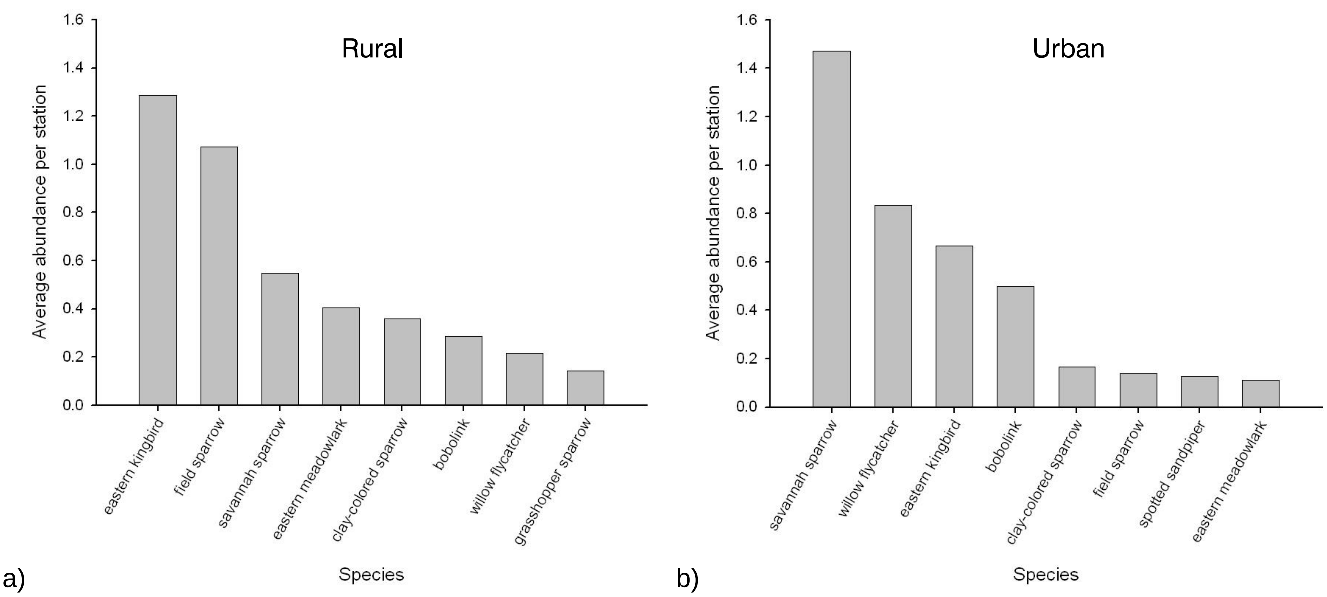


Figure 32. Meadow bird community composition based on average abundance per station in the a) rural land use zone and b) urban land use zone.

4.0 Discussion

4.1 Forest Monitoring

4.1.1 Forest Vegetation

Forest Vegetation Species Composition

Tree composition in TRCA forests has remained relatively consistent since the baseline report. Forest plots continue to be dominated by sugar maple and contain the same five non-native species: Manitoba maple, common buckthorn, apple, pear and black locust. Based on the species composition in the woody regeneration layer, it is expected that our forests will continue to be dominated by sugar maple.

The species composition of woody regeneration has also remained relatively the same since the baseline report in 2012 and continues to be dominated by native species. Regeneration in the rural zone continues to be dominated by sugar maple while chokecherry and sugar maple continue to dominate the urban zone. In the urban zone, choke cherry had a higher stem count and sugar maple had a higher percent cover so it is likely that as these forests mature, choke cherry will be out-competed by sugar maple through natural processes. The relative cover and abundance of native and exotic regeneration species has also remained relatively consistent since the 2012 baseline report in both the urban and rural zones.

The ground vegetation layer has seen the most changes in species composition based on position in the top 10 species (relative cover) since the baseline report suggesting that this layer is the most dynamic. This layer also had the most even distribution of relative cover values among species whereas the regeneration and tree layers showed dominance by primarily one or two species.

Urban Impacts and Temporal Changes

Urbanization had a significant impact on all three forest high-level indicators causing a decline in the FQI, the number of L1-L3 species and the percentage of native species in urban forests. Floristic quality assessments are an effective way to measure biological degradation based on plant species composition and are commonly used throughout North America (Spyreas 2014). The decline in the FQI in urban zone is consistent with previous studies and monitoring programs. CVC (2010b) found an increase in the mean FQI score when moving from the lower to upper physiographic zones. In the CVC watershed, the lower physiographic zone consists of 57% urban land use while the upper physiographic zone contains 15% urbanization.

Impacts of urbanization on floristic quality in forests have also been shown in the scientific literature. Gerken Golay *et al.* (2013) found that urban forests had lower floristic quality compared to preserved forests in Iowa. Urban forests in this study were located in the urban matrix and preserved forests fell in both the urban and agricultural zone but were state owned, providing the best representation of natural conditions. Preserved forests in this study also had lower variability in floristic quality among sites demonstrating an increased homogeneity. This result is not consistent with our data which showed approximately equal measures of variability in each land use zone.

The lower number of L1-L3 forest vegetation species in the urban zone indicates that urbanization is limiting the distribution of sensitive species. This result also demonstrates the robustness of TRCA's scoring and ranking system (TRCA 2010). Flora species with a rank of L1 are those that have the highest regional conservation concern status due to rarity, stringent habitat needs and/or threat to habitat (TRCA 2010). Four factors contribute to a flora species' rank including local distribution, population trend, habitat dependence and sensitivity to development. Species ranking as L1 have a low occurrence across the region, are showing signs of population decline, have the ability to occupy a low number of vegetative community types and have a negative response to development.

Since sensitivity to development is one factor involved in assigning L-ranks to species, we would expect that higher ranking species (L1-L3) should respond negatively to urbanization. One interesting finding is the larger variability in the number of L1-L3 species in rural sites suggesting that even some rural sites may be impaired. Alternatively, it would be worthwhile to look at the characteristics of urban sites that contain more L1-L3 species than on average.

Urban forests had a lower percentage of native forest vegetation species than rural forests indicating that exotic plant species are more prevalent in urban forests. This result supports existing literature suggesting that urban areas are heavily impacted by introduced plant species (Duguay *et al.* 2007, Gerken Golay *et al.* 2013). A study in the Ottawa region comparing small forest fragments in urban, agricultural and forested landscapes also found that forests in urban areas contained a significantly higher percentage of introduced plant species (40% more) compared to the other two landscape types (Duguay *et al.* 2007). This study also suggested several mechanisms by which introduced species enter and establish populations in forest fragments. Seeds may be transferred inadvertently by people using urban forests, soil disturbance could favour the growth of introduced species and seeds may disperse by natural mechanisms to forests from urban ornamental gardens (Duguay *et al.* 2007).

In addition to containing a higher percentage of exotic species, urban forests had a higher maximum percent cover of exotics in the ground vegetation layer. This trend is likely being driven by the dominance of garlic mustard in this layer which also had significantly higher cover in urban forests. The higher percent cover of garlic mustard in the urban zone has also previously been found in the literature. Credit Valley Conservation (2010b) found that the likelihood of occurrence of garlic mustard was higher in forests with greater urban cover. In the TRCA region, garlic mustard was found in extremely low amounts in the rural sites but percent cover did show a lot of variation in the urban sites suggesting that not all urban sites are heavily impacted by garlic mustard.

Common buckthorn, dog-strangling vine, shrub honeysuckle (*Lonicera* x *bella*) and European fly honeysuckle (*Lonicera xylosteum*) are other exotic species of concern in the region. Common buckthorn did have a higher mean relative abundance and cover in urban forests compared to rural forests; however, there was a lot of variation in the data suggesting that certain urban sites may have similar relative cover values to rural sites. This is consistent with the findings of Duguay *et al.* (2013) who also found that common buckthorn was not associated with a specific landscape (urban, agricultural or forested).

Temporal analysis of the percent cover of garlic mustard in the regional plots showed signs for concern. Garlic mustard showed an increasing trend in percent cover between 2009 and 2014 at both the regional scale and in the urban zone. Garlic mustard is a biennial herb that has spread quickly throughout Canada and the eastern United States since the late 1800's (Nuzzo 1999). It is a species that spreads quickly to other areas by establishing satellite populations which then continue to spread themselves. A study conducted by Nuzzo (1999) in relatively undisturbed forests of northern Illinois between 1987 and 1999 found that once established and in the absence of disturbance, garlic mustard becomes a permanent part of the forest and only fluctuates mildly in both cover and density between years. Nuzzo (1999) also suggests that the relatively stable cover and density values may not represent the rate of spread in highly disturbed forests. Since garlic mustard is spread exclusively by seed, natural disturbances such as floods or wind storms can facilitate spread and humans carrying seeds also facilitate dispersal and spread of the species. The increasing trend in garlic mustard cover in the regional plots is cause for concern because garlic mustard cover was measured as the maximum relative percent cover. Since this is a relative measure, the increases over time found in this report suggest that other species are declining in relative cover while garlic mustard increases.

Tree Health

The living crown of TRCA forests consists primarily of healthy trees (91.6%) with far fewer trees in light to moderate decline (6.7%) and even fewer in severe decline (1.6%). Based on Sajan (2006), if greater than 25% of trees show signs of severe decline then the forest is severely impaired. This value is well above the 1.6% of trees in severe decline found in the TRCA region indicating that TRCA forests are healthy with respect to crown vigour. These results are similar to CVC forests between 2005 and 2009, where the proportion of healthy trees ranged from 74-95%, light to moderate decline 4-21% and severe decline 1-3% (CVC 2010b).

Even though the results of this report fall within acceptable ranges for forest health and are similar to neighbouring jurisdictions, it is important to focus on the decline in healthy crowns and the increase in crowns showing light to moderate decline between 2013 and 2014 in the regional plots. This change is potentially due to the ice storm of December 21, 2013 that hit Toronto and the surrounding region. The storm produced widespread power outages due to downed power lines from fallen trees and tree limbs. In 1998, a similar ice storm hit eastern Canada and the northeastern United States and a research study was initiated in the sugar maple dominated forests of Gatineau Park, Quebec to track recovery over a 10-20 year period (Pisaric *et al.* 2008). In this study, trees were assessed in 1998 and again in 2003 based on measures of crown health, growth and mortality. Based on the findings of Pisaric *et al.* (2008), forests in the TRCA jurisdiction should see an initial improvement in tree health followed by decreased growth rates and/or increased mortality of trees that were severely damaged by the storm. Trees that experienced little damage should show increased growth due to canopy release (gaps in the canopy created by dying or fallen trees).

There was variation in crown vigour over time based on tree species. The most abundant species in our jurisdiction was sugar maple and it showed very little change over time in average crown vigour, although there was a decline between 2013 and 2014. White cedar, bur oak, ironwood, American beech and black cherry also showed declines in health between 2013 and 2014 but the degree of the decline was greater for these species than sugar maple. These results are consistent with the findings of Pisaric *et al.* (2008) who also found that sugar maple was more resilient to storm damage than other species. Red and white ash both consistently had more trees showing light to moderate decline or severe decline between 2008 and 2014. This could be due to emerald ash borer (*Agrilus planipennis*), an invasive wood-boring insect attacking ash trees that was first detected in Detroit in 2002 and has since spread throughout southern Ontario (TRCA 2012b). One of the first signs of emerald ash borer is crown dieback and this sign is apparent in our regional plots.

In general, TRCA regional plots had low levels of tree mortality with on average 0.5% (range: 0 - 1.1%) of trees dying each year between 2009 and 2014. The values of mortality found in this study are very similar to those found in CVC's jurisdiction which ranged from 0.3% to 1.5% between 2005 and 2009 (CVC 2010b). The values in this report are also consistent with those found by the Canadian Forest Service in Sajan (2006) which range from 0.5-4% in maple-dominated forests. The mortality rates in this report can be used as a baseline to track changes in the future and to look for the higher mortality rates which are expected due to the ice storm.

Snags are a vital component of eastern deciduous forests because they are used by over 45 birds and mammals for nesting, feeding, roosting and denning (Keddy and Drummond 1996). In the TRCA region, 9.2% of trees were classified as snags (dead standing trees, dead leaning trees or dead standing trees with a broken crown). This is lower than the 12% average characteristic of eastern deciduous forests of North America although the range for old growth and mature stands was found to be 5-22% (Keddy 1994). In order to maintain a healthy overall forest community by managing dead trees, the following guidelines are recommended by the Ministry of Natural Resources and Forestry in their land manager guidelines. Maintain as many cavity trees and snags as possible by harvesting only those standing dead trees that are a safety hazard. Where snags are felled, leave these on site to operate as downed logs (OMNR 2011) which provide habitat for other species and to retain forest nutrients.

Beech bark disease continues to affect a large proportion of American beech trees in the regional plots. Since the 2012 baseline report, the percent of affected trees has fluctuated between 84% and 100%. Of the 21 live trees monitored between 2008 and 2014, 1 died and 3 others showed declines in crown vigour or had fluctuating crown vigour values between the healthy and the light to moderate decline categories. These would be important trees to track in the future because if they succumb to the disease, mortality rates would be high for this species and close to the 5% per year cautionary level (Sajan 2006). Even though the disease is of great concern, in some areas where the disease has persisted for long periods of time, beech remain a component of the forest but no longer dominate the canopy (Lovett *et al.* 2006). In other areas, beech are showing signs of resistance with 19% of mature beech trees (>30 cm diameter) not substantially affected even though they were showing signs of the disease (Griffin *et al.* 2003).

Gypsy moth was the prevalent pest in regional plots affecting 0.7-7% of trees between 2010 and 2014. It is important to note that gypsy moth records in all cases represented the presence of egg masses or larvae and reported defoliation caused by gypsy moths was rare (only 0.9% of trees). Defoliation of trees does remain an issue in TRCA forests with 28% of trees at some point between 2008 and 2014 showing some degree of defoliation although the causative agent in the majority of cases remains unknown. The defoliation may be due to gypsy moth since 31% of affected trees are bur oak and gypsy moth are known to prefer oak (Lovett *et al.* 2006) although percentages were found for other species as well. Trees suffer primarily short-term effects from defoliation including loss of productivity and seed production, but trees initially in good condition produce new leaves when conditions improve. Stressed trees on the other hand, may die after a single defoliation event and this may be important to monitor given the increased stress of many trees since the ice storm and in the years to come.

4.1.2 Forest Birds

Urban Impacts and Temporal Changes

Forest bird communities are showing consistent negative impacts of urbanization. The number of forest-dependent individuals, the number of forest-dependent species and the number of L1-L3 species were all significantly lower in the urban land use zone compared to the rural land use zone. Also, rural forests contained a higher abundance of area sensitive forest bird species including wood thrush, ovenbird and scarlet tanager (*Piranga olivacea*). Urban impacts on birds are prevalent in developing landscapes and there are several mechanisms by which forest birds could be impacted. Woodlots tended to be smaller in urban areas and this excludes area sensitive forest bird species that need large tracts of forest for breeding and foraging (Austen *et al.* 2001). Predator communities are different in urban areas and areas with higher housing densities are known to contain a higher abundance of blue jay (*Cyanocitta cristata*), domestic cats (*Felis catus*), raccoons (*Procyon lotor*) and opossum (*Didelphis virginiana*; Haskell *et al.* 2001). This increase in nest predators is an important consideration for breeding songbirds because nest predation is the leading cause of nest failure in birds and this affects recruitment to the population (Martin 1995). Urban noise is another issue for forest birds because urban noise can interfere with avian communication methods and lead to lower densities of breeding birds near roads (Reijnen *et al.* 1995).

The finding that urban land use zones had fewer L1-L3 bird species suggests that the scoring and ranking system developed for fauna is also performing well. For fauna, L-ranks are assigned based on the following criteria: local occurrence, population trends, habitat dependence, area sensitivity, mobility and sensitivity to development. Higher L-rank scores (e.g. L1-L3) are assigned to rare species with declining population trends, species that are highly specialized for a specific habitat type, those that require large tracts of habitat, and species that are sensitive to development. Based on these criteria, we would expect urban areas to have fewer L1-L3 species. Opposed to forest flora communities, the variability within rural sites and urban sites was lower suggesting that the L-rank system for forest birds is better at separating the effects of urbanization. It also suggests that urban sites consistently had fewer L1-L3 bird species while rural sites consistently had more L1-L3 bird species.

Temporally, forest bird communities have remained relatively stable between 2008 and 2014 based on the number of individual birds and the number of species, although there may be declines in the number of L1-L3 species in the rural land use zone. These results were not significant at the $p<0.05$ level and so the results should be considered primarily a warning sign that the number of more sensitive forest bird species is declining in regional plots. One possible explanation for this trend is that forest bird species of concern are no longer using some or all of the forests but they are being replaced by less sensitive forest bird species (L4 or L5 species) thus maintaining a stable species richness over time.

4.1.3 Plethodontids

The salamander monitoring component of the program has undergone several changes since monitoring commenced in 2009. Monitoring has been refined in three main ways: 1) all boards have been switched to single boards to ensure salamander safety, 2) cover boards are now more thoroughly covered/disguised to avoid human and animal disturbance, and 3) data have been used to make the best use of staff time in the field. Data were used to assess the effect of staff only conducting four salamander surveys instead of five on the abundance of salamanders recorded. When the first visit was removed there was no change in salamander abundance. In 2015, only four visits were used and this saved staff time and resources.

As of 2014, all sites had only single cover boards to maximize salamander safety and since sampling methodology is now consistent among sites and years with the single boards, 2014 can be used as a baseline year to track temporal trends in red-backed salamander populations. A retrospective power analysis resulted in an increase in the magnitude of population decline we are able to detect (15% to 20%) with sufficient power (>80%). This increase in magnitude was due to lower than expected numbers of salamanders found under cover boards. One next step for the program is to determine why salamander numbers have been low. In 2015, a survey of coarse woody debris was conducted to answer a specific question: Does the amount of coarse woody debris in a forest affect the number of red-backed salamanders using artificial cover boards in the same forest? One hypothesis is that forests with more coarse woody debris provide more natural habitat for salamanders and therefore salamanders do not need to use the artificial cover boards that are set up for monitoring (resulting in low numbers). This hypothesis is based on previous research suggesting that salamander abundance is related to the density, volume and decay class of coarse woody debris (McKenny *et al.* 2006).

Overall the salamander monitoring component of the program has been refined for salamander safety, to minimize disturbance and to increase efficiency; however; questions remain about the region's salamander populations and these will be answered in the years ahead.

4.2 Wetland Monitoring

4.2.1 Wetland Vegetation

Wetland Vegetation Species Composition

The species composition of woody vegetation in wetlands has not changed greatly since the baseline report although rankings based on relative abundance and cover have changed for some species. One of the most notable changes was an increase in the relative abundance rank of common buckthorn, an exotic species, in both the urban and rural land use zones. In the urban zone, common buckthorn moved from 3rd in the baseline report to 1st in this report and in the rural zone common buckthorn entered the top 10 in 3rd position. Bittersweet nightshade (*Solanum dulcamara*), another exotic species, moved from 5th to 3rd based on relative abundance in the urban zone and entered the top 10 in 7th position in the rural zone. In the rural zone the relative cover of these species was low, not ranking in the top 10; however, in the urban zone common buckthorn increased from 3rd to 2nd position resulting in a relative cover of 12.4% and bittersweet nightshade moved from 10th to 4th position resulting in a relative cover of 5.9% in the urban zone.

These species are more likely to be found in the terrestrial portions of the wetland transect; however, it is still concerning that these exotic species are increasing in both abundance and cover in regional wetland plots. The movement of common buckthorn into the position of highest relative abundance means that choke cherry (a native regeneration species) has decreased in relative abundance. These results are consistent with the findings of Mills *et al.* (2009) in a study conducted examining the impacts of the invasion of glossy buckthorn on an undisturbed wetland in Wisconsin. In this study, the cover of other shrub species decreased as buckthorn cover increased; however, the overall impact on the native plant community over 15 years was minimal. The findings of Mills *et al.* (2009) are in stark contrast to studies on other invasive shrubs that generally show that native species in invaded sites show decreased growth, increased mortality and declines in overall richness (Gould and Gorchow 2000, Hutchinson and Vankat 1997). It is important to continue to monitor our wetland regional plots to determine if we expect the same result in more disturbed wetlands in urban areas.

The species composition of herbaceous vegetation in wetlands has also remained relatively stable in terms of the relative cover of dominant species. In the urban zone, changes generally involved moving one or two ranks. Garlic mustard moved from 9th position in the baseline report to 15th position in this report but unfortunately moneywort (another exotic species) entered the top 10 in 10th position. Other exotic species in urban wetlands only moved one or two ranks. In the rural zone, turion/common duckweed now has the highest relative cover while common reed (an exotic species) decreased from 1st to 2nd position. The increase in turion/common duckweed is not likely a true increase but an artifact of changing identification methodology and therefore the decrease in common reed should be interpreted cautiously. In the baseline report rural wetlands contained four exotic species with >1% relative cover (common reed, red fescue (*Festuca rubra* ssp. *rubra*), reed canary grass and hybrid cattail) and in this report narrow-leaved cattail (*Typha angustifolia*) was added to the list of exotic species with >1% relative cover.

Urban Impacts and Temporal Changes

Urbanization had a strong effect on wetland flora high-level indicators. The number of L1-L3 species and the percentage of native species were significantly lower in urban wetlands. The FQI for wetland flora did not show a significant difference although the direction of the relationship indicated lower values in urban wetlands and the result was approaching significance at a $p<0.10$ level. Urban impacts on floristic quality in wetlands have previously been found in the literature. The FQI in coastal wetlands of the Great Lakes region has been shown to decrease with increasing population density in the watershed (Bourdaghs *et al.* 2006). The decrease in FQI in urban areas means that these wetlands on average have fewer native plant species or plant species that are less sensitive. The number of L1-L3 species was also lower in urban areas. This again shows the ability of TRCA's scoring and ranking system to allow species to be the indicators of potential disturbance. It also shows that urban wetlands generally contain wetland plants that have more generalist attributes and are more tolerant of degraded conditions. The increase in exotic species in urban wetlands found in this report is also consistent with the literature. Urban wetlands have also previously been found to have a higher occurrence of exotic species than undeveloped wetlands (Magee *et al.* 1999).

Declines in FQI, percent native species and number of L1-L3 species in urban wetlands could be due to many factors including changes in water quality, competition, and spatial attributes of the landscape. Decreased water quality through increased siltation or road salts from storm water run-off could make the wetland inhospitable for species that cannot tolerate high salt or turbid environments (Lougheed *et al.* 2001). The higher abundance of exotic species in urban wetlands could be due to competition and replacement of native species by exotic species, such as common buckthorn (Mills *et al.* 2009). Many urban wetlands are more isolated from other wetlands or natural areas and species with limited dispersal may show population declines in isolated urban wetlands (Matthews *et al.* 2005).

4.2.2 Wetland Birds

Urban Impacts and Temporal Changes

There were no significant impacts of urbanization on the number of L1-L3 species, wetland-dependent bird abundance or wetland dependent bird richness. While there were no significant differences, there was a trend towards lower values of each variable in urban wetlands. The lack of significant results are in contrast to the literature on urban impacts on wetland birds. For example, Smith and Chow-Fraser (2010) found a significant decline in the number of wetland-dependent birds in coastal marshes surrounded by urban land uses compared to coastal marshes surrounded by rural land uses in the Great Lakes Region. DeLuca *et al.* (2004) found marshes in urban areas had fewer wetland-dependent birds than marshes in rural landscapes of Chesapeake Bay, USA. It is important to note that while the results in this report were not statistically significant, the trend observed was consistent with these studies.

Potential reasons for the non-significant findings of this report could be based on species-specific habitat preferences. For example, some species may be dependent on open water areas for foraging (e.g. double-crested cormorant (*Phalacrocorax auritus*) and therefore selecting a wetland closer to Lake Ontario may mean less travel to foraging sites, and not a preference for urban areas. An alternative explanation for finding no difference between urban and rural sites could be that wetlands, as opposed to forests or meadows, may be more protected from outside influences because of their aquatic nature. The presence of water may limit the accessibility of the wetland to potential predators. Water depth has been shown to be an important factor predicting nest predation rates for marsh birds because deeper water is less accessible for predators such as raccoons (Jobin and Picman 1997). Given that marsh birds can find deep water sites, they may be better protected from nest predation in the urban areas by non-aquatic mammalian predators compared to their forest and meadow-nesting counterparts.

4.2.3 Frogs

Urban Impacts and Temporal Changes

Overall, urban and rural sites did not vary greatly in the percent of sites occupied by at least one frog species, but the only sites that had no frogs detected were in the urban land use zone. When examining species-specific differences, all frog species occupied a higher percent of rural sites than urban sites. Some species were more evenly distributed across land use zones than others including the green frog which only showed a small decline in the percent of occupied sites in the urban zone. Alternatively, spring peepers were heavily impacted by urbanization and occurred in a significantly lower percent of sites in the urban zone compared to the rural zone. In addition to these species-specific trends, both the total number of frog species and the number of L1-L3 frog species were significantly lower in urban wetlands. This means that overall, fewer urban sites were occupied by frogs, the sites that were occupied are being used by less sensitive species and contained a more homogeneous frog species composition.

Frogs have previously shown a strong negative relationship with increased urbanization (Knutson *et al.* 1999). Urban areas are generally less favourable environments for frogs because of the increased density of roads and lack of important adjacent habitat (Knutson *et al.* 1999). Mortality caused by vehicular traffic is a major concern for frogs and roads with higher traffic volume leading to higher mortality rates (Bouchard *et al.* 2009). Noise associated with roads, and with urban areas in general, has been shown to affect frog populations. Anthropogenic noise can interfere with communication and thus affect breeding success and survival (Lengagne 2008). The complete removal of adjacent natural habitat is especially important for amphibian species that need adjacent habitat for feeding, overwintering and nesting (Semlitsch and Bodie 2003). The American toad, chorus frog, spring peeper, tetraploid grey treefrog and wood frog use more terrestrial areas at some point in their life cycle (Ontario Nature 2015). In general, anuran populations benefit from habitats that provide a variety of natural habitat types including upland forests, wet forests and emergent wetlands (Knutson *et al.* 1999).

Temporal analysis of the percent of sites occupied showed primarily stable trends between 2009 and 2014 for all species. The only species that showed slight, although non-significant declines were wood frog and tetraploid grey treefrog. These declines may just be natural temporal variation or due to variability based on weather conditions during surveys. Regardless, it is important to continue to monitor these trends in the future.

4.3 Meadow Monitoring

4.3.1 Meadow Birds

Urban Impacts and Temporal Changes

The number of L1-L3 meadow bird species was significantly lower in the urban land use zone compared to the rural land use zone. There was no significant difference in the number of meadow-dependent individuals or the number of meadow-dependent species between the rural and urban land use zones. One reason for a lack of a difference in richness could be that one of the urban sites is larger than any of the rural sites and there is a greater opportunity to encounter new species in larger sites. Abundance should not be affected by meadow size since average abundance per point count was used for the analysis. These results suggest that urban meadows have a similar number of meadow-dependent individuals and species although urban meadows contain fewer sensitive species (L1-L3).

Temporally, meadow birds are showing increases in the number of L1-L3 species in the rural zone and declines in meadow bird abundance across the region. The decline in abundance is cause for concern and may be mirroring the declines in grassland breeding birds that have been seen in North America since the 1960's in the North American Breeding Bird Survey. These include declines of -2.8% per year for grasshopper sparrow, -3.3% per year for eastern meadowlark (*Sturnella magna*), -2% per year for bobolink and -1.3% per year for savannah sparrow (Sauer *et al.* 2014). The underlying causes of these large scale declines are not completely known but low reproductive output is suggested as the reason for European declines (Roodbergen *et al.* 2012). Low reproductive output in birds could be due to many factors but nest predation is the leading cause of nest failure (Martin 1995) and habitat fragmentation in rural landscapes is contributing to increased nest predation rates especially in small meadow fragments in North America (Herkert *et al.* 2003). Another concern for meadow birds, and specifically bobolink, include their perception and control as a pest species and pesticide use on over-wintering grounds (Renfrew and Saavedra 2007).

5.0 Conclusions

The TRCA regional forest flora communities are relatively healthy and compare well to other eastern deciduous forests, although both urbanization and the ice storm have negatively impacted flora communities. Urbanization has led to more generalist forest flora communities containing fewer sensitive species and more exotic species such as garlic mustard. The ice storm has caused declines in crown vigour, although mortality due to the storm may not be seen for several years. Forest bird communities are showing strong, consistent negative impacts of urbanization in the TRCA region. Temporal trends are relatively stable for forest birds but there may be a decline in the number of more sensitive species in the rural land use zone. Wetland plant communities are showing strong, negative impacts of urbanization and the further spread of common buckthorn needs close monitoring. Wetland bird populations are stable temporally and are not showing signs of urban impacts, although this may be due to microhabitat preferences. Temporal trends for frogs are stable; however, fewer urban sites are occupied by frogs, the sites that are occupied are being used by less sensitive species and have a more homogeneous frog species composition. Meadow birds are showing moderate impacts of urbanization with fewer sensitive species in urban meadows and temporally are showing potential declines in the number of individuals across the region.

Overall, this report shows that while many of the high-level indicators are stable over time, almost all indicators are showing impairment due to urbanization. If the jurisdiction foresees the conversion of more rural land to urban land uses, it is expected that there will be drastic declines in both flora and fauna communities in the years to come. The finding that many urban natural areas have lower values for high-level indicators does not mean urban natural areas are of less importance; they continue to provide habitat for generalist species and those of concern in urban areas. Monitoring should continue in both the urban and rural land use zones and should also continue to track the temporal impacts of development on natural areas that are currently changing from rural to urban land uses. In addition, more research needs to be done on development methods and land use planning that maintain habitat for sensitive species while still meeting the demands of a growing population to ensure the health and persistence of natural areas in TRCA's watershed.

Terrestrial Long Term Monitoring Program Next Steps

- Continue monitoring. Assess temporal trends on an annual basis to identify emerging concerns and report on temporal and urban/rural spatial trends every five years.
- Conduct a retrospective power analysis to determine the power of current sample sizes.
- Include a timed walkthrough for forest flora regional plots starting in 2015 and determine if this method sufficiently captures the forest flora community compared to the currently used un-timed walkthrough.
- Survey existing coarse woody debris at a sub-set of sites in 2015 to determine if it is an important predictor of salamander abundance.

This data set presents many opportunities to address additional questions outside those which have been answered in this report. Answering these additional questions will provide a better understanding of other factors affecting biota in the region and will provide opportunities for future monitoring and directions for mitigating urban impacts. The formation of new partnerships or the acquisition of additional funding may be required to answer these questions.

Questions to Answer Using Terrestrial Long Term Monitoring Program Data

- Have frog population sizes and distributions across the region changed since before the Terrestrial Long-Term Monitoring Program began? Historical data could be obtained from the Marsh Monitoring Program for frogs in the TRCA region.
- What aspects of urbanization are causing frogs, and specifically spring peepers, to be absent in urban areas?
- Are breeding songbirds differentially affected by urbanization based on the height at which they nest (e.g. are forest low-level nesters more affected than other forest nesting guilds)?

- Since urbanization has such a strong effect on almost all high-level indicators, what is the threshold of % urban cover at which each high-level indicator is affected?
- What other factors contribute to the urban/rural differences in biodiversity (e.g. trail density, prevalence of off-leash dogs, red ants)?
- Evaluate the prevalence of dog-strangling vine among the LTMP bird plots and evaluate its potential effect on ground-nesting bird abundance.
- Is there consistency in point count occupancy among years by specific species of breeding birds in the data set (e.g. are the same species and number of individuals of these species documented from year to year at the exact same point count location)? Is there variation in occupancy among years by specific species of breeding birds based on the quality of the site? If differences are found, what characteristics of poor quality sites cause variation in occupancy by the same species among years?
- Can TRCA's scoring and ranking system be applied to a novel data set (e.g. CVC data) to accurately assess land use disturbance?
- Given the variability within the urban zone for the results of numerous high-level indicators, what characteristics make a high quality urban site? Is it patch size? A buffer around the site?
- Disturbance is a natural process in communities helping to maintain diversity and promote succession (DeSteven *et al.* 1991); however, the frequency of these disturbances needs to be considered. Pisaric *et al.* (1998) suggests that return times for events such as the 1998 ice storm are on the order of 20 to 100 years. Will climate change affect the return times of severe storms and will this affect our forests more heavily than historically?
- Are there region-wide changes in populations of flora and fauna with more southern and more northern affinities in response to climate change?
- Is there a lag time for various taxa to be affected in LTMP plots that are currently undergoing increased urbanization in the surrounding landscape? If lag times are identified, how long are they and do they vary based on taxa?

6.0 References

Austen, M. J. W., C. M. Francis, D. M. Burke, and M. S. W. Bradstreet. 2001. Landscape context and fragmentation effects onf forest birds in southern Ontario. The Condor 103:701-714.

Bouchard, J., A. T. Ford, F. E. Eigenbrod, and L. Fahrig. 2009. Behavioral responses of northern leopard frogs (*Rana pipiens*) to roads and traffic: implications for population persistence. Ecology and Society 14:23.

Bourdaughs, M., C. A. Johnston, and R. R. Regal. 2006. Properties and performance of the floristic quality index in Great Lakes coastal wetlands. Wetlands:718-735.

CVC. 2010a. Credit Valley Conservation Integrated Watershed Monitoring Program 2008 Summary Draft Report. Credit Valley Conservation. January 2010.

CVC. 2010b. Monitoring Forest Integrity within the Credit River Watershed. Chapter 4: Forest Vegetation 2005-2009. Credit Valley Conservation. ix + 101 p. (P. 72 and 74).

DeLuca, W. V., C. E. Studds, L. L. Rockwood, and P. P. Marra. 2004. Influence of land use on theintegrity of marsh bird communities of Chesapeake Bay, USA. Wetlands 24:837-847.

Duguay, S., F. Eigenbrod, and L. Fahrig. 2007. Effects of surrounding urbanization on non-native flora in small forest patches. Landscape Ecology 22:589-599.

EMAN 2004a. Tree Health – EMAN Monitoring Protocols and Standards. Ecological Monitoring and Assessment Network (EMAN). Environment Canada. Available on-line at: http://eman-rese.ca/eman/ecotools/protocols/terrestrial/tree_health/tree_health.pdf [Accessed 6 January 2010].

EMAN 2004b. Regeneration and Sapling Survey – EMAN Monitoring Protocols and Standards. Ecological Monitoring and Assessment Network (EMAN), Environment Canada. Available on-line at: http://eman-rese.ca/eman/ecotools/protocols/terrestrial/sapling/sapling.pdf [Accessed 6 January 2010].

Gerken Golay, M. E., J. R. Thompson, C. M. Mabry, and R. K. Kolka. 2013. An investigation of water nutrient levels associated with forest vegetation in highly altered landscapes. Journal of Soil and Water Conservation 68:361-371.

Gould, A. M. A. and Gorchov, D. L. 2000. Effects of the exotic invasive shrub *Lonicera maackii* on the survival and fecundity of three species of native annuals. American Midland Naturalist 144:36-50.

Grandin, U. 2011. Understorey vegetation stability and dynamics in unmanaged boreal forests along a deposition gradient in Sweden. AMBIO 40:867-877.

Griffin, J. M., G. M. Lovett, M. A. Arthur, K. C. Weathers. 2003. The distribution and severity of beech bark disease in the Catskill Mountains, NY. Canadian Journal of Forest Research 33:1754-1760.

Haskell, D. G., A. M. Knupp, and M. C. Schneider. 2001. Nest predator abundance and urbanization. *In* Avian ecology and conservationin an urbanizing world (J. M. Marzluff, R. Bowman, and R. Donnelly, Eds.). Kluwar Academic Publishers, Norwell, Massachusetts, USA.

Herkert, J. R., D. L. Reinking, D. A. Wiedenfeld, M. Winter, J. L. Zimmerman, W. E. Jensen, E. J. Finck, R. R. Koford, D. H. Wolfe, S. K. Sherrod, M. A. Jenkins, J. Faaborg, and S. K. Robinson. 2003. Effects of prairie fragmentation on the nest success of breeding birds in the midcontinental United States. Conservation Biology 17:587-594.

Hutchinson, T. F. and J. L. Vankat. 1997. Invasibility and effects of Amur honeysuckle in southwestern Ohio forests. Conservation Biology 11:1117-1124.

Jobin, B. and J. Picman. 1997. Factors affecting predation on artificial nests in marshes. The Journal of Wildlife Management 61:792-800.

Lengagne, T. 2008. Traffic noise affects communication behaviour in a breeding anuran, *Hyla arborea*. Biological Conservation 141:2023-2031.

Lougheed, V. L., B. Crosbie, and P. Chow-Fraser. 2001. Primary determinants of macrophyte community structure in 62 marshes across the Great Lakes basin: latitude, land use, and water quality effects. Canadian Journal of Fisheries and Aquatic Sciences 58:1603-1612.

Lovett, G. M., C. D. Canham, M. A. Arthur, K. C. Weathers, and R. D. Fitzhugh. 2006. Forest ecosystem responses to exotic pests and pathogens in eastern North America. Bioscience 56:395-405.

Keddy, C. 1994. Forest structure in eastern North America. Ecological Woodlands Restoration project of the Eastern Ontario Model Forest. October 1994.

Keddy, P. A. and C. G. Drummond. 1996. Ecological properties for the evaluation, management, and restoration of temperate deciduous forest ecosystems. Ecological Applications 6:748-762.

Knutson, M. G., J. R. Sauer, D. A. Olsen, M. J. Mossman, L. M. Hemesath, and M. J. Lannoo. 1999. Effects of landscape composition and wetland fragmentation on frog and toad abundance and species richness in Iowa and Wisconsin, U.S.A. Conservation Biology 13:1437-1446.

Magee, T. K., T. L. Ernst, M. E. Kentula, and K. A. Dwire. 1999. Floristic comparison of freshwater wetlands in an urbanizing environment. Wetlands 19:517-534.

Martin, T. E. 1995. Avian life history evolution in relation to nest sites, nest predation, and food. Ecological Monographs 65:101-107.

Masters L. A. 1997. Monitoring vegetation. In Packard S. and Mutel C.F. (eds) 1997. The Tallgrass Restoration Handbook. Washington, Island Press, pp. 279-301.

Matthews, J. W., P. A. Tessene, S. M. Wiesbrook, and B. W. Zercher. 2005. Effect of area and isolation on species richness and indices of floristic quality in Illinois, USA wetlands. Wetlands 25:607-615.

McKenny, H. C., W. S. Keeton, and T. M. Donovan. 2006. Effects of structural complexity enhancement on eastern red-backed salamander (*Plethodon cinereus*) populations in northern hardwood forests. Forest Ecology and Management 230:186-196.

Mills, J. E., J. A. Reinartz, G. A. Meyer, E. B. Young. 2009. Exotic shrub invasion in an undisturbed wetland has little community-level effect over a 15-year period. Biological Invasions 11:1803-1820.

Nuzzo, V. 1999. Invasion pattern of the herb garlic mustard (*Allioria petiolata*) in high quality forests. Biological Invasions 1:169-179.

Oldham M.J., Bakowsky W.D. and D.A. Sutherland. 1995. Floristic Quality Assessment System for Southern Ontario. Natural Heritage Information Centre, Ontario Ministry of Natural Resources, Peterborough.

OMNR. 2011. A land manager's guide to conserving habitat for forest birds in southern Ontario. Retrieved16 March 2015 from: http://www.ont-woodlot-assoc.org/pdf/August%202011%20-%20Land_Manager's_Guide_2011.pdf.

Ontario Nature. 2015. Ontario reptile and amphibian atlas. Available on-line at: http://www.ontarionature.org [Accessed 16 March 2015].

Pisaric, M.F.J. , D.J. King, A.J.M. MacIntosh and R. Bemrose. 2008. Impact of the 1998 ice storm on the health and growth of sugar maple (Acer saccharum Marsh) dominated forests in Gatineau Park, Quebec. *Journal of the Torrey Botanical Society* . 135: 530-539.

Pope, Shealagh, L. Fahrig and H.G. Merriam. 2000. Landscape complementation and metapopulation effects on leopard frog populations. *Ecology*. 81: 2498-2508.

Reijnen, R. R. Foppen, C. Ter Braak, and J. Thissen. 1995. The effects of car traffic on breeding bird populations in woodland. III. Reduction of density in relation to the proximity of main roads. The Journal of Applied Ecology 32:187-202.

Renfrew, R. B. and A. M. Saavedra. 2007. Ecology and conservation of bobolinks (*Dolichonyx oryzivorus*) in rice production regions of Bolivia. Ornithologia Neotropical 18:61-73.

Roberts-Pichette P. and Gillespie L. 1999. Terrestrial vegetation biodiversity monitoring protocols: Ground Vegetation Biodiversity Monitoring Protocols. Ecological Monitoring and Assessment (EMAN) Occasional Paper Series. Report Number 9. 142 pp. Available on-line at: http://www.eman-rese.ca/eman/ecotools/protocols/terrestrial/vegetation/e_veg_protocol.pdf [Accessed 6 January, 2010].

Roodbergen, M., B. van der Werf, and H. Hotker. 2012. Revealing the contributions of reproduction and survival to the Europe-wide decline in meadow birds: review and meta-analysis. Journal of Ornithology 153:53-74.

Sajan R. 2006. Tree Health Data Analysis – EMAN Monitoring Protocols and Standards. Ecological Monitoring and Assessment Network. Environment Canada. Available on-line at: http://www.eman-rese.ca/eman/ecotools/protocols/terrestrial/tree_health/Tree%20health%20threshold_final.pdf [Accessed 6 January 2010].

SAS Institute Inc. 2008. SAS Jmp Version 8.0.

Sauer, J. R., J. E. Hines, J. E. Fallon, K. L. Pardieck, D. J. Ziolkowski, Jr., and W. A. Link. 2014. *The North American Breeding Bird Survey, Results and Analysis 1966 - 2013. Version 01.30.2015* USGS Patuxent Wildlife Research Center*, Laurel, MD.*

Semlitsch, R. D. and J. R. Bodie. 2003. "Biological criteria for buffer zones around wetlands and riparian habitats for amphibians and reptiles." Conservation Biology 17(5): 1219-28.

Smith, L. A. and P. Chow-Fraser. 2010. Impact of adjacent land use and isolation on marsh bird communities. Environmental Management 45:1040-1051.

Spyreas, G. 2014. An examination of temporal trends, regional variation, and habitat-type differences in site-level floristic quality, and their implications for the use of floristic quality assessment. Ph.D. thesis. Natural Resources and Environmental Sciences. University of Illinois.

TRCA 2007. Terrestrial Natural Heritage Program Data Collection Methodology. Toronto Region Conservation Authority.

TRCA 2010. Vegetation Community and Species Ranking and Scoring Method. Toronto Region Conservation Authority.

TRCA 2011a. Forest Vegetation Monitoring Protocol – Terrestrial Long-term Fixed Plot Monitoring Program – Regional Watershed Monitoring and Reporting. Toronto and Region Conservation Authority.

TRCA 2011b. Forest Bird Monitoring Protocol – Terrestrial Long-term Fixed Plot Monitoring Program – Regional Watershed Monitoring and Reporting. Toronto and Region Conservation Authority.

TRCA 2011c. Plethodontid Monitoring Protocol - Terrestrial Long-term Fixed Plot Monitoring Program – Regional Watershed Monitoring and Reporting. Toronto and Region Conservation Authority.

TRCA 2011d. Wetland Vegetation Monitoring Protocol - Terrestrial Long-term Fixed Plot Monitoring Program – Regional Watershed Monitoring and Reporting. Toronto and Region Conservation Authority.

TRCA 2011e. Wetland Bird Monitoring Protocol - Terrestrial Long-term Fixed Plot Monitoring Program – Regional Watershed Monitoring and Reporting. Toronto and Region Conservation Authority.

TRCA 2011f. Wetland Amphibian Monitoring Protocol - Terrestrial Long-term Fixed Plot Monitoring Program – Regional Watershed Monitoring and Reporting. Toronto and Region Conservation Authority.

TRCA 2011g. Meadow Bird Monitoring Protocol -Terrestrial Long-term Fixed Plot Monitoring Program – Regional Watershed Monitoring and Reporting. Toronto and Region Conservation Authority.

TRCA. 2011h. Regional Watershed Monitoring Program: Surface Water Quality Summary 2006-2010. 49 pp + appendices.

TRCA. 2012a. Terrestrial Long-term Monitoring – Baseline Conditions Report. Toronto and Region Conservation Authority.

TRCA. 2012b. Recommended Approach for the Management of Emerald Ash Borer. TRCA Forest Health Working Group.

Zar, J. H. 1999. Biostatistical analysis, 4th edition. New Jersey, USA.

Zorn P. 2008. *A Priori* Power Analysis for Toronto Region Conservation Authority's Regional Watershed Monitoring Program. Report prepared for TRCA. Ottawa, ON.

7.0 Appendix

Table A.1. Nesting habitat preferences for bird species. Preferences were used to determine the dependence of a species on a particular habitat (forest, wetland or meadow). Swamp nesters were included as forest-dependent.

Common name	Scientific name	Forest	Edge	Wetland	Meadow	General
Acadian flycatcher	*Empidonax virescens*					
barred owl	*Strix varia*					
black and white warbler	*Mniotilta varia*					
Blackburnian warbler	*Setophaga fusca*					
black-throated blue warbler	*Setophaga caerulescens*					
black-throated green warbler	*Setophaga virens*					
blue-grey gnatcatcher	*Polioptila caerulea*					
blue-headed vireo	*Vireo solitarius*					
broad-winged hawk	*Buteo platypterus*					
brown creeper	*Certhia americana*					
canada warbler	*Cardellina canadensis*					
cerulean warbler	*Setophaga cerulea*					
Cooper's hawk	*Accipiter cooperii*					
eastern screech-owl	*Megascops asio*					
eastern wood-pewee	*Contopus virens*					
golden-crowned kinglet	*Regulus satrapa*					
great-crested flycatcher	*Myiarchus crinitus*					
hairy woodpecker	*Picoides villosus*					
hermit thrush	*Catharus guttatus*					
hooded warbler	*Setophaga citrina*					
long-eared owl	*Asio otus*					
magnolia warbler	*Setophaga magnolia*					
merlin	*Falco columbarius*					
northern saw-whet owl	*Aegolius acadicus*					
nothern goshawk	*Accipiter gentilis*					
olive-sided flycatcher	*Contopus cooperi*					
ovenbird	*Seiurus aurocapillus*					
pileated woodpecker	*Dryocopus pileatus*					
pine siskin	*Carduelis pinus*					
pine warbler	*Setophaga pinus*					
red-breasted nuthatch	*Sitta canadensis*					
red-eyed vireo	*Vireo olivaceus*					
red-shouldered hawk	*Buteo lineatus*					
ruffed grouse	*Bonasa umbellus*					
scarlet tanager	*Piranga olivacea*					

Common name	Scientific name	Forest	Edge	Wetland	Meadow	General
sharp-shinned hawk	*Accipiter striatus*					
veery	*Catharus fuscescens*					
whip-poor-will	*Caprimulgus vociferus*					
white-breasted nuthatch	*Sitta carolinensis*					
white-winged crossbill	*Loxia leucoptera*					
winter wren	*Troglodytes hiemalis*					
wood duck	*Aix sponsa*					
wood thrush	*Hylocichla mustelina*					
worm-eating warbler	*Helmitheros vermivorus*					
yellow-bellied sapsucker	*Sphyrapicus varius*					
yellow-throated vireo	*Vireo flavifrons*					
American redstart	*Setophaga ruticilla*					
American woodcock	*Scolopax minor*					
black-billed cuckoo	*Coccyzus erythropthalmus*					
blue-winged warbler	*Vermivora pinus*					
brown thrasher	*Toxostoma rufum*					
chestnut-sided warbler	*Setophaga pensylvanica*					
downy woodpecker	*Picoides pubescens*					
eastern bluebird	*Sialia sialis*					
eastern towhee	*Piplio erythrophthalmus*					
golden-winged warbler	*Vermivora chrysoptera*					
indigo bunting	*Passerina cyanea*					
least flycatcher	*Empidonax minimus*					
mourning warbler	*Geothlypis philadelphia*					
Nashville warbler	*Oreothlypis ruficapilla*					
purple finch	*Carpodacus purpureus*					
red-bellied woodpecker	*Melanerpes carolinus*					
red-headed woodpecker	*Melanerpes erythrocephalus*					
ring-necked pheasant	*Phasianus colchicus*					
rose-breasted grosbeak	*Pheucticus ludovicianus*					
ruby-throated hummingbird	*Archilochus colubris*					
white-throated sparrow	*Zonotrichia albicollis*					
wild turkey	*Meleagris gallopavo*					
yellow-billed cuckoo	*Coccyzus americanus*					
yellow-breasted chat	*Icteria virens*					
yellow-rumped warbler	*Setophaga coronata*					
alder flycatcher	*Empidonax alnorum*					
American bittern	*Botaurus lentiginosus*					
American black duck	*Anas rubripes*					
American coot	*Fulica americana*					

Common name	Scientific name	Forest	Edge	Wetland	Meadow	General
black tern	*Chlidonias niger*					
black-crowned night-heron	*Nycticorax nycticorax*					
blue-winged teal	*Anas discors*					
Canada goose	*Branta canadensis*					
canvasback	*Aythya valisineria*					
Caspian tern	*Sterna caspia*					
common gallinule	*Gallinula galeata*					
common tern	*Sterna hirundo*					
common yellowthroat	*Geothlypis trichas*					
double-crested cormorant	*Phalacrocorax auritus*					
gadwall	*Anas strepera*					
great black-backed gull	*Larus marinus*					
great blue heron	*Ardea herodias*					
great egret	*Casmerodius albus*					
green heron	*Butorides virescens*					
green-winged teal	*Anas crecca*					
herring gull	*Larus argentatus*					
hooded merganser	*Lophodytes cucullatus*					
least bittern	*Ixobrychus exilis*					
mallard	*Anas platyrhynchos*					
marsh wren	*Cistothorus palustris*					
mute swan	*Cygnus olor*					
osprey	*Pandion haliaetus*					
pied-billed grebe	*Podilymbus podiceps*					
ring-billed gull	*Larus delawarensis*					
sora	*Porzana carolina*					
swamp sparrow	*Melospiza georgiana*					
trumpeter swan	*Cygnus buccinator*					
Virginia Rail	*Rallus limicola*					
Wilson's snipe	*Gallinago delicata*					
bobolink	*Dolichonyx oryzivorus*					
clay-coloured sparrow	*Spizella pallida*					
eastern kingbird	*Tyrannus tyrannus*					
eastern meadowlark	*Sturnella magna*					
field sparrow	*Spizella pusilla*					
grasshopper sparrow	*Ammodramus savannarum*					
Henslow's sparrow	*Ammodramus henslowii*					
horned lark	*Eremophila alpestris*					
loggerhead shrike	*Lanius ludovicianus*					
northern harrier	*Circus cyaneus*					
savannah sparrow	*Passerculus sandwichensis*					

Common name	Scientific name	Forest	Edge	Wetland	Meadow	General
sedge wren	*Cistothorus platensis*					
short-eared owl	*Asio flammeus*					
spotted sandpiper	*Actitis macularia*					
upland sandpiper	*Bartramia longicauda*					
vesper sparrow	*Pooecetes gramineus*					
western meadowlark	*Sturnella neglecta*					
willow flycatcher	*Empidonax traillii*					
American Crow	*Corvus brachyrhynchos*					
American goldfinch	*Carduelis tristis*					
American kestrel	*Falco sparverius*					
American robin	*Turdus migratorius*					
Baltimore oriole	*Icterus galbula*					
barn swallow	*Hirundo rustica*					
black-capped chickadee	*Parus atricapillus*					
blue jay	*Cyanocitta cristata*					
Carolina wren	*Thryothorus ludovicianus*					
cedar waxwing	*Bombycilla cedrorum*					
chimney swift	*Chaetura pelagica*					
chipping sparrow	*Spizella passerina*					
cliff swallow	*Petrochelidon pyrrhonota*					
common grackle	*Quiscalus quiscula*					
common nighthawk	*Chordeiles minor*					
eastern phoebe	*Sayornis phoebe*					
European starling	*Sturnus vulgaris*					
great-horned owl	*Bubo virginianus*					
grey catbird	*Dumetella carolinensis*					
house finch	*Carpodacus mexicanus*					
house sparrow	*Passer domesticus*					
house wren	*Troglodytes aedon*					
killdeer	*Charadrius vociferus*					
mourning dove	*Zenaida macroura*					
northern cardinal	*Cardinalis cardinalis*					
northern flicker	*Colaptes auratus*					
northern mockingbird	*Mimus polyglottos*					
orchard oriole	*Icterus spurius*					
peregrine falcon	*Falco peregrinus*					
red-tailed hawk	*Buteo jamaicensis*					
red-winged blackbird	*Agelaius phoeniceus*					
rock dove	*Columba livia*					
song sparrow	*Melospiza melodia*					
tree swallow	*Tachycineta bicolor*					

Common name	Scientific name	Forest	Edge	Wetland	Meadow	General
warbling vireo	*Vireo gilvus*					
yellow warbler	*Setophaga petechia*					
northern waterthrush	*Parkesia noveboracensis*					
prothonotary warbler	*Protonotaria citrea*					
bank swallow	*Riparia riparia*					
belted kingfisher	*Ceryle alcyon*					
brown-headed cowbird	*Molothrus ater*					
northern rough-winged swallow	*Stelgidoptery x serripennis*					
purple martin	*Progne subis*					
turkey vulture	*Cathartes aura*					

Table A.2. List of woody regeneration species.

Common name	Scientific name
alternate-leaved dogwood	*Cornus alternifolia*
American beech	*Fagus grandifolia*
American bittersweet	*Celastrus scandens*
apple	*Malus pumila*
balsam fir	*Abies balsamea*
basswood	*Tilia americana*
beaked hazel	*Corylus cornuta*
bitternut hickory	*Carya cordiformis*
bittersweet nightshade	*Solanum dulcamara*
black cherry	*Prunus serotina*
black maple	*Acer saccharum ssp. nigrum*
bur oak	*Quercus macrocarpa*
cherry sp.	
choke cherry	*Prunus virginiana var. virginiana*
common blackberry	*Rubus allegheniensis*
common buckthorn	*Rhamnus cathartica*
dotted hawthorn	*Crataegus punctata*
downy serviceberry	*Amelanchier arborea*
European fly honeysuckle	*Lonicera xylosteum*
European highbush cranberry	*Viburnum opulus ssp. opulus*
European mountain-ash	*Sorbus aucuparia*
fly honeysuckle	*Lonicera canadensis*
garden red currant	*Ribes rubrum*

garden red raspberry	*Rubus idaeus ssp. idaeus*
grey dogwood	*Cornus foemina ssp. racemosa*
ironwood	*Ostrya virginiana*
Manitoba maple	*Acer negundo*
maple-leaved viburnum	*Viburnum acerifolium*
multiflora rose	*Rosa multiflora*
Norway maple	*Acer platanoides*
poison ivy (shrub form)	*Toxicodendron radicans var. rydbergii*
prickly gooseberry	*Ribes cynosbati*
purple-flowering raspberry	*Rubus odoratus*
red ash	*Fraxinus pennsylvanica*
red maple	*Acer rubrum*
red oak	*Quercus rubra*
red-berried elder	*Sambucus racemosa ssp. pubens*
riverbank grape	*Vitis riparia*
round-leaved dogwood	*Cornus rugosa*
running strawberry-bush	*Euonymus obovatus*
shagbark hickory	*Carya ovata*
shrub honeysuckle	*Lonicera x bella*
sugar maple	*Acer saccharum ssp. saccharum*
thicket creeper	*Parthenocissus inserta*
trembling aspen	*Populus tremuloides*
virgin's bower	*Clematis virginiana*
white ash	*Fraxinus americana*
white cedar	*Thuja occidentalis*
white elm	*Ulmus americana*
wild black currant	*Ribes americanum*
wild black raspberry	*Rubus occidentalis*
wild honeysuckle	*Lonicera dioica*
wild red raspberry	*Rubus idaeus ssp. strigosus*
witch-hazel	*Hamamelis virginiana*
yellow birch	*Betula alleghaniensis*

Table A.3. Sites used to analyze trends over time (2009-2014) for frog richness, #L1-L3 species and proportion of sites occupied.

Site code	Site name	Land use
WF-1	Centennial Park	Urban
WF-2	Kenpark	Urban
WF-3	Claireville	Urban
WF-4	Kortright	Rural
WF-5	Caledon Tract	Rural
WF-6	Cold Creek	Rural
WF-7	ORMCP	Rural
WF-8	Toogood Pond	Urban
WF-9	ET Seton Park (OSC)	Urban
WF-10	East Don Parkland	Urban
WF-11	Finch & Pickering Townline	Rural
WF-14	Greenwood	Rural
WF-16	Palgrave	Rural
WF-17	Albright	Rural

Table A.4. Sites used to analyze differences between urban and rural sites for frog richness and #L1-L3 species. Site data were averaged between 2013 and 2014.

Site code	Site name	Land use
WF-1	Centennial Park	Urban
WF-2	Kenpark	Urban
WF-3	Claireville	Urban
WF-4	Kortright	Rural
WF-5	Caledon Tract	Rural
WF-6	Cold Creek	Rural
WF-7	ORMCP	Rural
WF-8	Toogood Pond	Urban
WF-9	ET Seton Park (OSC)	Urban
WF-10	East Don Parkland	Urban
WF-11	Finch & Pickering Townline	Rural
WF-14	Greenwood	Rural
WF-16	Palgrave	Rural
WF-17	Albright	Rural
WF-20	Bolton Tract West	Rural
WF-22	Wildwood Park	Urban
WF-24	Stouffville Reservoir	Urban
WF-25	Bolton Tract South	Rural
WF-28	Highland Creek	Urban
WF-32	Heart Lake	Urban

Table A.5. Sites used to analyze trends over time (2009-2014) for wetland-dependent bird richness and #L1-L3 species.

Site code	Site name	Land use
WB-1	Centennial Park	Urban
WB-2	Kenpark	Urban
WB-3	Claireville	Urban
WB-4	Kortright	Rural
WB-5	Caledon Tract	Rural
WB-6	Cold Creek	Rural
WB-7	ORMCP	Rural
WB-8	Toogood	Urban
WB-9	ET Seton Park/OSC	Urban
WB-10	East Don Parkland	Urban
WB-11	Finch and Pickering Townline	Rural
WB-14	Greenwood	Rural
WB-16	Palgrave	Rural
WB-17	Albright	Rural

Table A.6. Sites used to analyze differences between urban and rural sites for wetland-dependent bird richness and #L1-L3 species. Site data were averaged between 2012 and 2014.

Site code	Site name	Land use
WB-1	Centennial Park	Urban
WB-2	Kenpark	Urban
WB-3	Claireville	Urban
WB-4	Kortright	Rural
WB-5	Caledon Tract	Rural
WB-6	Cold Creek	Rural
WB-7	ORMCP	Rural
WB-8	Toogood	Urban
WB-9	ET Seton Park/OSC	Urban
WB-10	East Don Parkland	Urban
WB-11	Finch and Pickering Townline	Rural
WB-14	Greenwood	Rural
WB-16	Palgrave	Rural
WB-17	Albright	Rural
WB-20	Bolton Resource Mgmt Tract	Rural
WB-22	Wildwood Park	Urban
WB-23	Old Church Roads Land	Urban
WB-24	Stouffville Reservoir	Urban
WB-25	Bolton Tract South	Rural
WB-28	Highland Creek Mouth	Urban

Table A.7. Sites used to analyze trends over time (2008-2014) for forest-dependent bird richness and #L1-L3 species.

Site code	Site name	Land use
FB-2	Heart Lake	Urban
FB-3	Portage Trail	Urban
FB-4	Downsview Dells	Urban
FB-5	Claireville	Urban
FB-6	Boyd	Urban
FB-7	TVM Site #14	Rural
FB-8	Caledon Tract	Rural
FB-9	Bolton Tract	Rural
FB-10	Humber Trails	Rural
FB-11	West Gormley	Rural
FB-12	Wilket Creek	Urban
FB-13	Bakers Sugar Bush	Urban
FB-14	Cudia Park	Urban
FB-15	Morningside Park	Urban
FB-16	Altona Forest	Urban
FB-17	Reesor Rd and Hwy 7	Rural
FB-18	Shoal Point Woodland	Urban
FB-19	Duffin Heights	Rural
FB-20	Goodwood	Rural
FB-21	Glen Major	Rural

Table A.8. Sites used to analyze differences between urban and rural sites for forest-dependent bird richness and #L1-L3 species. Site data were averaged between 2011 and 2014.

Site code	Site name	Land use
FB-1	Hwy 410 & 403	Urban
FB-2	Heart Lake	Urban
FB-3	Portage Trail	Urban
FB-4	Downsview Dells	Urban
FB-5	Claireville	Urban
FB-6	Boyd	Urban
FB-7	TVM Site #14	Rural
FB-8	Caledon Tract	Rural
FB-9	Bolton Tract	Rural
FB-10	Humber Trails	Rural
FB-11	West Gormley	Rural
FB-12	Wilket Creek	Urban
FB-13	Bakers Sugar Bush	Urban
FB-14	Cudia Park	Urban
FB-15	Morningside Park	Urban
FB-16	Altona Forest	Urban
FB-17	Reesor Rd and Hwy 7	Rural
FB-18	Shoal Point Woodland	Urban
FB-19	Duffin Heights	Rural
FB-20	Goodwood	Rural
FB-21	Glen Major	Rural
FB-22	Duffins Marsh Woodland	Urban
FB-23	Bruces Mill	Rural
FB-24	Palgrave	Rural
FB-27	Kirby and Keele	Rural
FB-28	Eglinton and Hwy 403	Urban
FB-29	Marie Curtis Park	Urban
FB-30	Gibson Lake	Rural
FB-31	Peel Tract	Rural

Table A.9. Sites used to analyze trends over time (2008-2014) for meadow-dependent bird richness and #L1-L3 species.

Site code	Site name	Land use
MB-2	Claireville	Urban
MB-3	Boyd North	Urban
MB-4	Upland Sandpiper	Rural
MB-5	Bolton Tract	Rural
MB-6	South of Hwy 407 East of Keele	Urban
MB-7	South Kirby East of Keele	Rural
MB-8	ORMCP	Urban
MB-9	East Point Park	Urban
MB-10	N. Twyn Rivers Dr. East of Meadowvale	Rural
MB-11	Milne	Urban
MB-12	Duffins Trail	Rural
MB-13	Greenwood	Rural
MB-14	Glen Major	Rural

Table A.10. Sites used to analyze differences between urban and rural sites for meadow-dependent bird richness and #L1-L3 species. Site data were averaged between 2012 and 2014.

Site code	Site name	Land use
MB-2	Claireville	Urban
MB-3	Boyd North	Urban
MB-4	Upland Sandpiper	Rural
MB-5	Bolton Tract	Rural
MB-6	South of Hwy 407 East of Keele	Urban
MB-7	South Kirby East of Keele	Rural
MB-8	ORMCP	Urban
MB-9	East Point Park	Urban
MB-10	N. Twyn Rivers Dr. East of Meadowvale	Rural
MB-11	Milne	Urban
MB-12	Duffins Trail	Rural
MB-13	Greenwood	Rural
MB-14	Glen Major	Rural
MB-16	Glen Haffy	Rural
MB-17	Snelgrove Property (Mayfield)	Urban
MB-18	Ebenezer Tract	Urban
MB-19	Claremont	Rural
MB-20	Goodwood Meadow	Rural

Table A.11. Sites used to analyze differences between urban and rural sites for forest vegetation FQI, % native species, # L1-L3 species, proportion alive vs. snags, crown vigour, regeneration and the spread of invasive species in ground vegetation layer. Site data were averaged between 2010 and 2014.

Site code	Site name	Land use
FV-1	Hwy 410 & 403	Urban
FV-2	Heart Lake	Urban
FV-3	Portage Trail	Urban
FV-4	Downsview Dells	Urban
FV-5	Claireville	Urban
FV-6	Boyd	Urban
FV-7	TWM Site 14	Rural
FV-8	Caledon Tract	Rural
FV-9	Bolton Tract	Rural
FV-10	Humber trails Forest and Wildlife Area	Rural
FV-11	West Gormley	Rural
FV-12	Wilket Creek	Urban
FV-13	Baker's Sugar Bush	Urban
FV-14	Cudia Park	Urban
FV-15	Morningside Park	Urban
FV-16	Altona Forest	Urban
FV-17	Reesor Rd and Hwy 7	Rural
FV-18	Shoal Point Woodland	Urban
FV-19	Duffin Heights	Rural
FV-20	Goodwood	Rural
FV-21	Glen Major	Rural
FV-22	Duffins Marsh Woodland	Urban
FV-24	Palgrave	Rural
FV-27	South Kirby East of Keele	Rural

Table A.12. Sites used to analyze trends over time (2008-2014) for the proportion of live forest trees vs. snags, crown vigour and regeneration. This list of sites was also used to look at temporal trends in the spread of invasive species in the ground vegetation layer (2009-2014).

Site code	Site name	Land use
FV-1	Hwy 410 & 403	Urban
FV-2	Heart Lake	Urban
FV-3	Portage Trail	Urban
FV-4	Downsview Dells	Urban
FV-5	Claireville	Urban
FV-6	Boyd	Urban
FV-7	TWM Site 14	Rural
FV-8	Caledon Tract	Rural
FV-9	Bolton Tract	Rural
FV-10	Humber trails Forest and Wildlife Area	Rural
FV-11	West Gormley	Rural
FV-12	Wilket Creek	Urban
FV-13	Baker's Sugar Bush	Urban
FV-14	Cudia Park	Urban
FV-15	Morningside Park	Urban
FV-16	Altona Forest	Urban
FV-17	Reesor Rd and Hwy 7	Rural
FV-18	Shoal Point Woodland	Urban
FV-19	Duffin Heights	Rural
FV-20	Goodwood	Rural
FV-21	Glen Major	Rural

Table A.13. Sites used to analyze differences between urban and rural sites for wetland vegetation FQI, % native species and # L1-L3 species. Site data were averaged between 2011 and 2014.

Site code	Site name	Land use
WV-1	Centennial Park	Urban
WV-2	Kenpark	Urban
WV-3	Claireville	Urban
WV-4	Kortright	Rural
WV-5	Caledon Tract	Rural
WV-6	Cold Creek	Rural
WV-7	ORMCP	Rural
WV-8	Toogood Pond	Urban
WV-9	ET Seton Park/OSC	Urban
WV-10	East Don Parkland	Urban
WV-11	Finch and Pickering Townline	Rural
WV-12	Bruce's Mill	Rural
WV-13	Duffins Marsh	Urban
WV-14	Greenwood	Rural
WV-15	Secord	Rural
WV-16	Palgrave	Rural
WV-22	Wildwood Park	Urban
WV-25	Bolton Tract South	Rural
WV-26	Bob Hunter Park	Urban

Table A.14. List of exotic species included in the analysis of temporal and spatial trends in the ground vegetation layer.

Common name	Scientific name
apple	*Malus pumila*
balsam bitter cress	*Cardamine impatiens*
bittersweet nightshade	*Solanum dulcamara*
black medick	*Medicago lupulina*
common buckthorn	*Rhamnus cathartica*
common burdock	*Arctium minus*
common lilac	*Syringa vulgaris*
common speedwell	*Veronica officinalis*
creeping Charlie	*Glechoma hederacea*
creeping thistle	*Cirsium arvense*
dame's rocket	*Hesperis matronalis*
dandelion	*Taraxacum officinale*
dog-strangling vine	*Cynanchum rossicum*
European highbush cranberry	*Viburnum opulus ssp. opulus*
European mountain-ash	*Sorbus aucuparia*
European spindle-tree	*Euonymus europaeus*
garlic mustard	*Alliaria petiolata*
hedge-parsley	*Torilis arvensis*
helleborine	*Epipactis helleborine*
hemp-nettle	*Galeopsis tetrahit*
hound's tongue	*Cynoglossum officinale*
Japanese yew	*Taxus cuspidata*
Kentucky blue grass	*Poa pratensis ssp. pratensis*
moneywort	*Lysimachia nummularia*
Morrow's honeysuckle	*Lonicera morrowii*
motherwort	*Leonurus cardiaca ssp. cardiaca*
multiflora rose	*Rosa multiflora*
nipplewort	*Lapsana communis*
Norway maple	*Acer platanoides*
prickly lettuce	*Lactuca serriola*
purple loosestrife	*Lythrum salicaria*
Queen Anne's lace	*Daucus carota*
shrub honeysuckle	*Lonicera x bella*
urban avens	*Geum urbanum*
winter cress	*Barbarea vulgaris*
woodland spear grass	*Poa nemoralis*
herb Robert	*Geranium robertianum*
Manitoba maple	*Acer negundo*