Terrestrial Long-term Monitoring Plot Program: Spatial Patterns and Temporal Trends in Terrestrial Biodiversity (2011-2020)

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SUMMARY

Toronto and Region Conservation Authority's (TRCA) Regional Watershed Monitoring Program was developed to track regional changes in the health of ecosystems and biodiversity. The Terrestrial Long-term Monitoring Plot Program supports this initiative by using standardized, scientifically-sound monitoring protocols to provide an in-depth understanding of terrestrial biodiversity across the jurisdiction. This report provides a summary of temporal changes and spatial patterns in the biodiversity of forests, wetlands, and meadows collected through the program between 2011 and 2020.

Forest vegetation changed significantly over the past 10 years reflecting the impact of regional disturbances including both the December 2013 ice storm and the widespread destruction by Emerald Ash Borer (*Agrilus planipennis*; EAB). These large-scale disturbances led to changes in tree community composition, increased tree mortality, snag production, decreased crown vigour, increased regeneration of woody species, and an increased production of ash seedlings.

Forest bird communities were also indirectly affected by EAB. Bark-foraging insectivores, such as woodpeckers and nuthatches, are those that feed on insects on or in the bark of trees. There was a significant increase in the total number of bark foragers across all stations likely reflecting an increased abundance of EAB as a food source.

Wetland and meadow ecosystems appeared to be relatively stable over time although several changes were apparent including increasing cover of invasives species, such as common buckthorn (*Rhamnus cathartica*), in wetlands and changes in habitat availability in meadows affecting species composition.

Species communities in all habitat types (forest, wetland, and meadow) continue to indicate the strong, negative effects of urbanization on these ecosystems. Monitoring stations in urban areas consisted of fewer species, lower abundances, and communities consisting of more generalists or tolerant species. Several factors related to urbanization continue to affect biodiversity including habitat loss, fragmentation, isolation, invasive species, and urban noise.

These results provide a greater understanding of regional factors affecting terrestrial biodiversity across the jurisdiction, along with the continued impact of urbanization, and can be used to predict changes likely to occur in the future as a result of future land development and climate change (new pests and increased frequency of ice storms). Results can also be used to inform various other initiatives such as invasive species management, forest management plans, watershed planning, and restoration.

INTRODUCTION

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 watersheds and natural heritage features. In 2008, this program was augmented with the addition of several terrestrial long-term fixed plots. The long-term monitoring plots represent an addition to 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).

TRCA 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, and meadow birds (Figure 1). Plots were placed in forest, wetland, and meadow habitat types using TRCA's Long-term Monitoring Program (LTMP) protocols (TRCA 2016a, 2021a-b, 2022a-d). 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, and amphibian communities. Using standardized scientific data collection protocols, the response of the terrestrial system to various landscape changes can be quantitatively documented. These data can be used to better guide regional or local management actions to improve overall biodiversity.

The purpose of this report is to 1) summarize spatial patterns by comparing selected biodiversity indicators between urban and rural land use zones, 2) summarize temporal trends in selected biodiversity indicators between 2011 and 2020, and 3) provide an update on the overall health of TRCA's wetland, forest, and meadow communities based on these analyses.

METHODS

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 (2016a, 2021a-b, 2022a-d).

Land use determination

For the purposes of this report, the TRCA jurisdiction has been divided into rural and urban zones, based on TRCA's 2017 land use layer. Sites were classed as urban or rural by determining the amount of urban land cover within a 2 km buffer of the plot/station (Appendix 1). If this amount was greater than 50% of the total area, then the station was classified as urban. If this amount was less than 50% of the total area, then the station was classified as rural. If two stations in the same site had opposite land use designations, an average of the total hectares of urban area was used. If this average was >50%, the site was designated as urban and if this average was <50% the site was designated as rural. This was a rare occurrence and only applied to one site for meadow bird surveys (Oak Ridges Moraine Corridor Park (ORMCP), MB-8). A summary of long-term monitoring plots found in each land use zone is shown in Table 1. It is important to note that while none of the plots changed from rural to urban since the last report (TRCA 2015), several plots did have development occurring nearby including West Gormley, Duffins Heights Trail, Boyd North, Kenpark, and Snelgrove and should be carefully examined in the future.

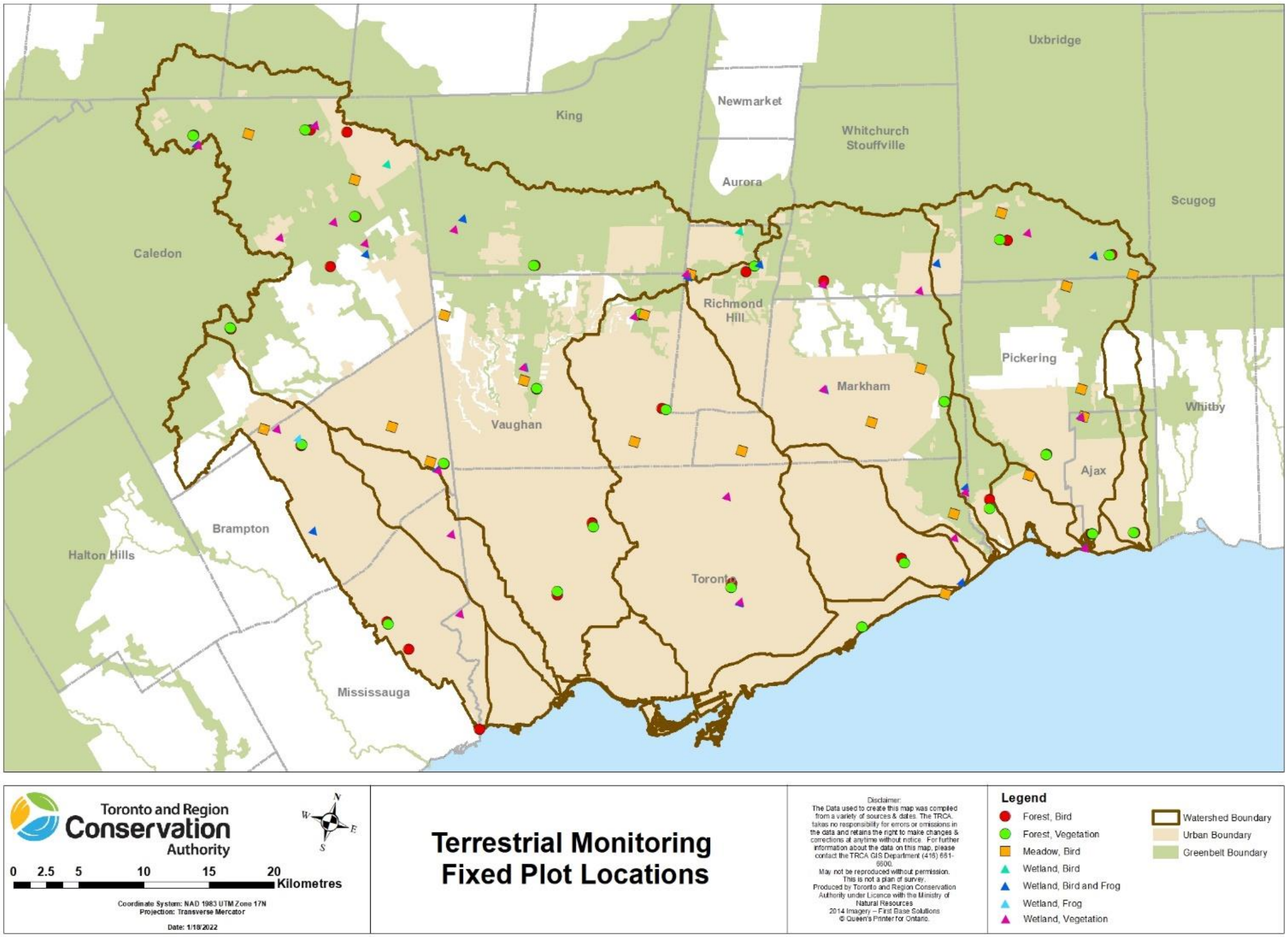


Figure 1. Terrestrial monitoring plots in TRCA's jurisdiction.

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

Plot type	Rural	Urban
Forest vegetation	11	13
Forest birds	14	15
Wetland vegetation	11	9
Wetland birds	9	10
Frogs	9	9
Meadow birds	8	8

Selection of biodiversity indicators

Long-term monitoring plots were established to identify the health and condition of key biological communities (i.e. vegetation, bird, frog) 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 TRCA's 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, and,
- 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.

Wetland monitoring plots were designed to:

- determine the health of wetlands in TRCA's 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, and,
- 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 TRCA's jurisdiction.

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
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

Assessing 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. TRCA has developed a local ranking system for flora and fauna species designed to reflect the ability of each species to thrive in the changing landscape of the Toronto region. The ranks range from extremely sensitive species (L1) to 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 2017). Species with ranks of L1 to L3 are considered to be of regional concern, 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 Appendix 2 and were produced primarily through staff understanding of the various species' nesting requirements.

Forest monitoring methodology

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 2010b).

Detailed information on plot set-up can be found in TRCA (2021a). 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.

Canopy cover estimates were added at forest plots in 2018 (TRCA 2021a). Note that while all pests and disease were monitored in all years, Emerald Ash Borer (*Agrilus planipennis*) monitoring for tree health started in 2014.

Forest bird stations

Forest birds were monitored using the Ontario Forest Bird Monitoring Program (FBMP) protocol designed by the Canadian Wildlife Service and now run by Birds Canada (TRCA 2022a). 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.

Wetland monitoring methodology

Wetland 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 2021b). 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 (2021b).

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.

Wetland bird stations

Monitoring stations were set-up following the Marsh Monitoring Program (MMP) protocol that was established by Bird Studies Canada (TRCA 2022b). 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.

Frog stations

Stations were set-up and monitored following the MMP in the same manner as wetland birds (TRCA 2022c). 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).

Meadow monitoring methodology

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 2022d). 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.

Data analysis

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 3.

Statistical power

The term "power" in statistics refers to your ability to detect a real difference or trend in the data when there is truly a difference or trend. Several factors can affect power including sample size, effect size (the magnitude of change you want to detect), and variation in the data set. In general, larger sample sizes, larger effect sizes, and less variation in the data set allow for higher power. An *a priori* power analysis was conducted using a surrogate data set at the outset of the program to set-up an adequate number of monitoring plots to achieve 80% power (Zorn 2008). A retrospective power analysis was conducted in 2015 using TRCA data. Since the number of monitoring sites has increased since the program started in 2008/2009, samples sizes are greater in later years, and particularly starting in 2011. Using 2011 as a baseline year, we determined through the retrospective power analysis that overall, TRCA's terrestrial LTMP program sample sizes are adequate to detect regional temporal trends and spatial patterns for the majority of high-level indicators.

Temporal trends

Trends were analyzed using the Mann-Kendall test: a non-parametric test for identifying trends in time series data and describes monotonic trends. All temporal trend analyses were conducted using R (R Core Team 2021).

Monotonic trends occur when a population of observations shifts over time. The detection of a monotonic trend does not imply that the trend is linear, occurs in one or more discrete steps, or in any pattern (Hirsch et al. 1991). The test is well-suited to data with missing values and to data that are truncated at upper and lower detection limits (Gilbert 1987). The data values are evaluated as an ordered time series. The initial value of the Mann-Kendall statistic, S, is assumed to be zero (e.g., no trend). A very high positive value of S is an indicator of an increasing trend, and a very low negative value indicates a decreasing trend. Mann-Kendall uses the z statistic to test for significance. A significance level of $p<0.05$ was used to determine if temporal trends were significant. All temporal trend graphs show means ± 1 standard error unless otherwise indicated. Tables showing statistical results for temporal trends can be found in Appendix 4.

Spatial patterns

Differences between urban and rural land use zones were analyzed using independent t-tests on the average value across all years at a specific site. All analyses were conducted using SAS JMP statistical software (SAS Institute Inc. 2008). 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 data transformations were attempted to improve normality. 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 for spatial patterns can be found in Appendix 5.

RESULTS

Forest monitoring

Forest vegetation

Temporal trends and spatial patterns in high level forest vegetation indicators

There were significant increases in the FQI regionally, in the urban zone, and in the rural zone; however, the mean CC score did not change over time in the urban zone or significantly declined regionally and in the rural zone (Figure 2). This suggests that the FQI may be increasing primarily due to an increase in the total number of species (species richness). The number of L1-L3 ranked species also increased in the urban zone however, the mean CC score suggests that the quality of these species (habitat specialist versus generalist) is decreasing. The percent of species that are native declined significantly regionally and in the rural zone suggesting that exotic species are becoming more prominent within the plots. Forest vegetation indicators strongly demonstrate the impact of urbanization with plots in the urban zone containing fewer species and species of conservation concern, a lower percentage of native species, and lower FQI and mean CC scores than stations in the rural zone.

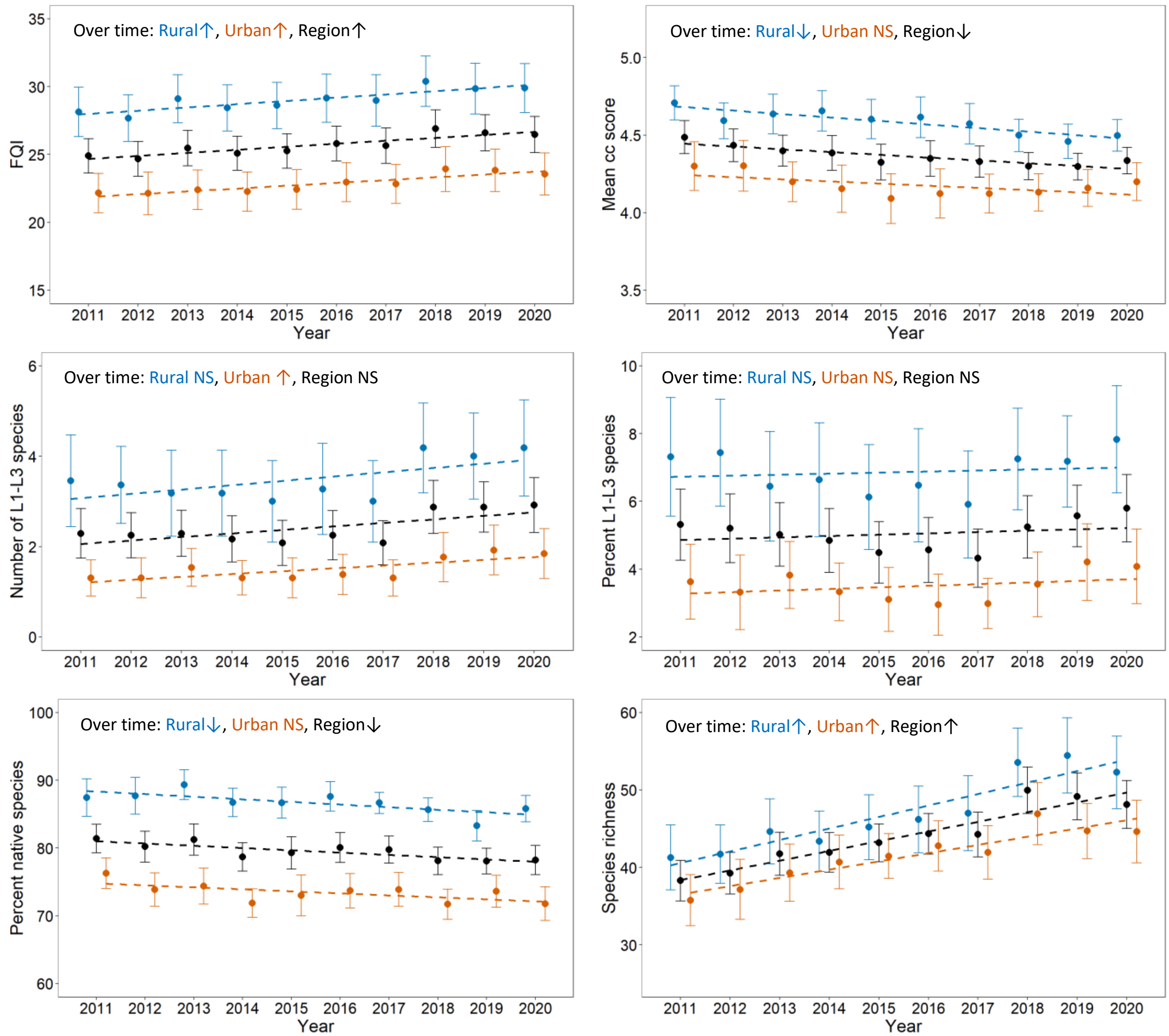


Figure 2. Spatial patterns and temporal trends in forest vegetation indicators between 2011 and 2020. An arrow pointing up (↑) represents a statistically significant increasing trend while an arrow pointing down (↓) represents a statistically significant decreasing trend. NS (non-significant) means there was no statistically significant trend.

Tree composition

As of 2020, a total of 519 live trees were being monitored in regional tree health plots. There were a total of 30 different tree species including five exotic species: common buckthorn (*Rhamnus cathartica*), Manitoba maple (*Acer negundo*), Chinese crab-apple (*Malus prunifolia*), 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 2015 Terrestrial Long-term Monitoring Program report (TRCA 2015), several species changed position based on relative abundance ranking although the differences were often minimal such as American beech (*Fagus grandifolia*) increasing from 7th to

6th. One notable change was the decrease in red ash (*Fraxinus pennsylvanica*) from 16th to 17th and white ash (*Fraxinus americana*) from 8th to 11th.

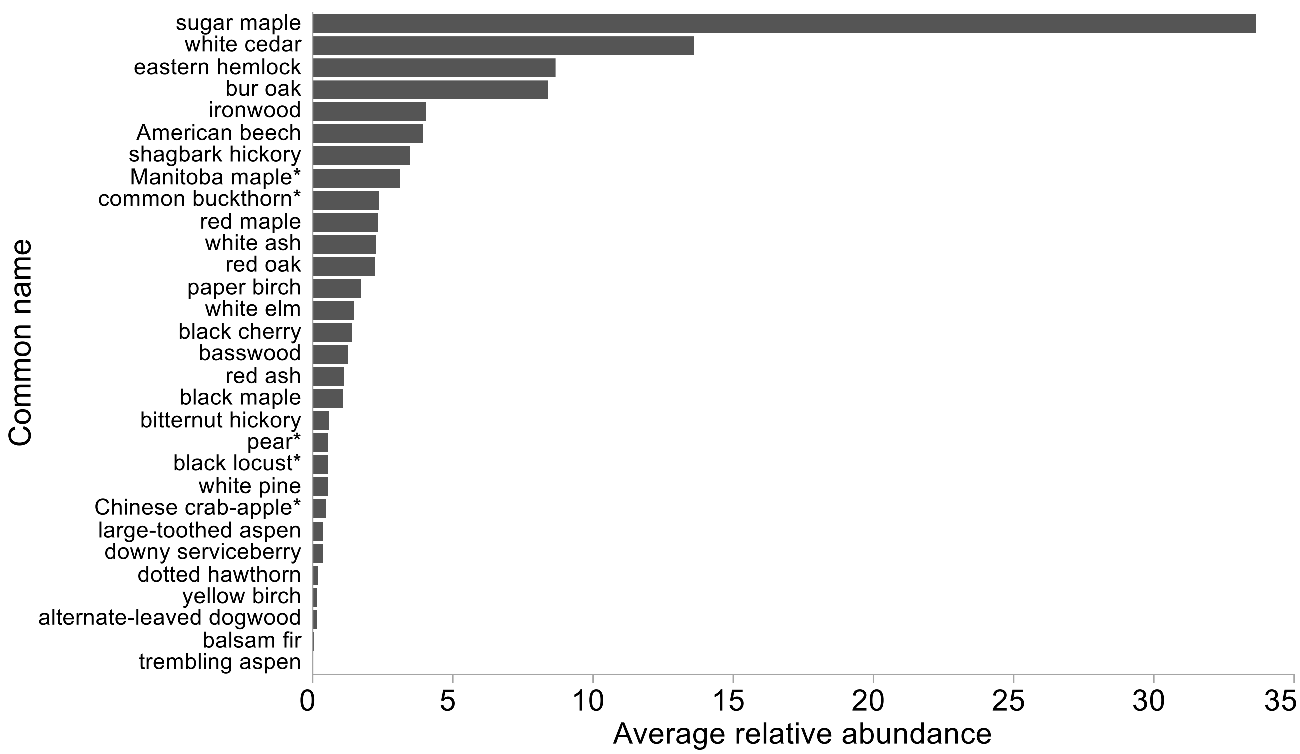
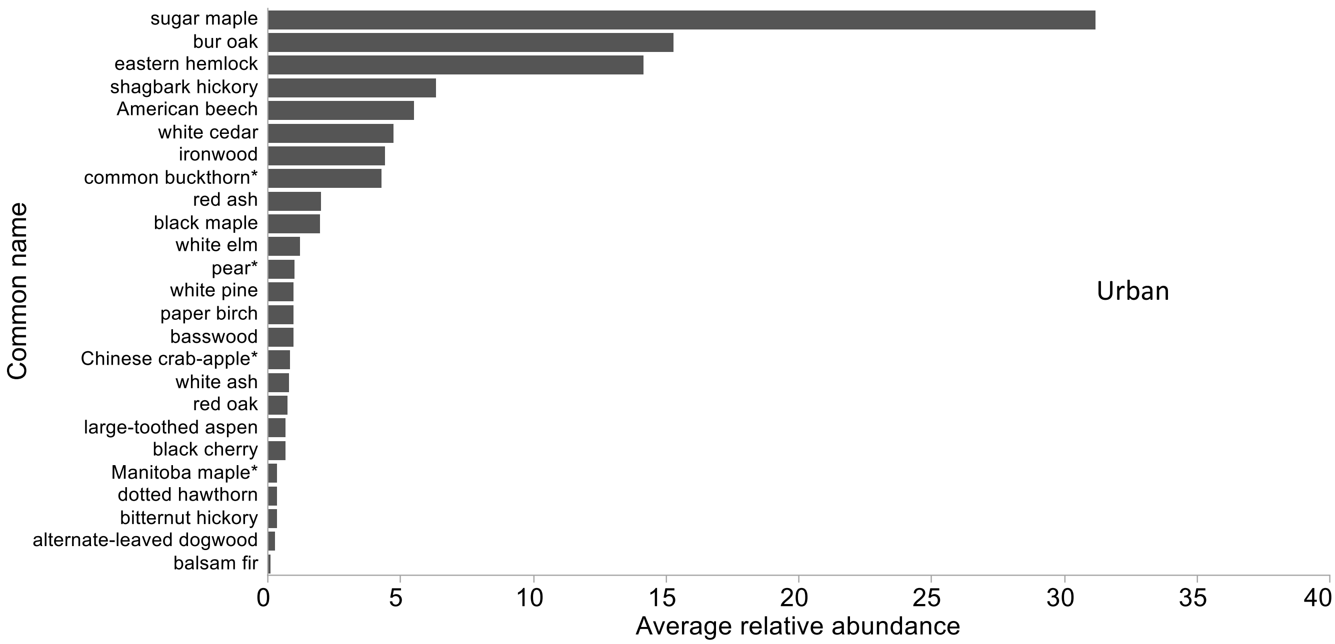


Figure 3. Average relative abundance of tree species in regional tree health plots (2011-2014). Exotic species are indicated with an asterisk (*).

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 (37%) compared to urban sites (31%). 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 changed in both the urban and rural zones since TRCA (2015). In the rural zone, white ash dropped from 4th to 6th and in the urban zone, ironwood moved from 5th to 7th.



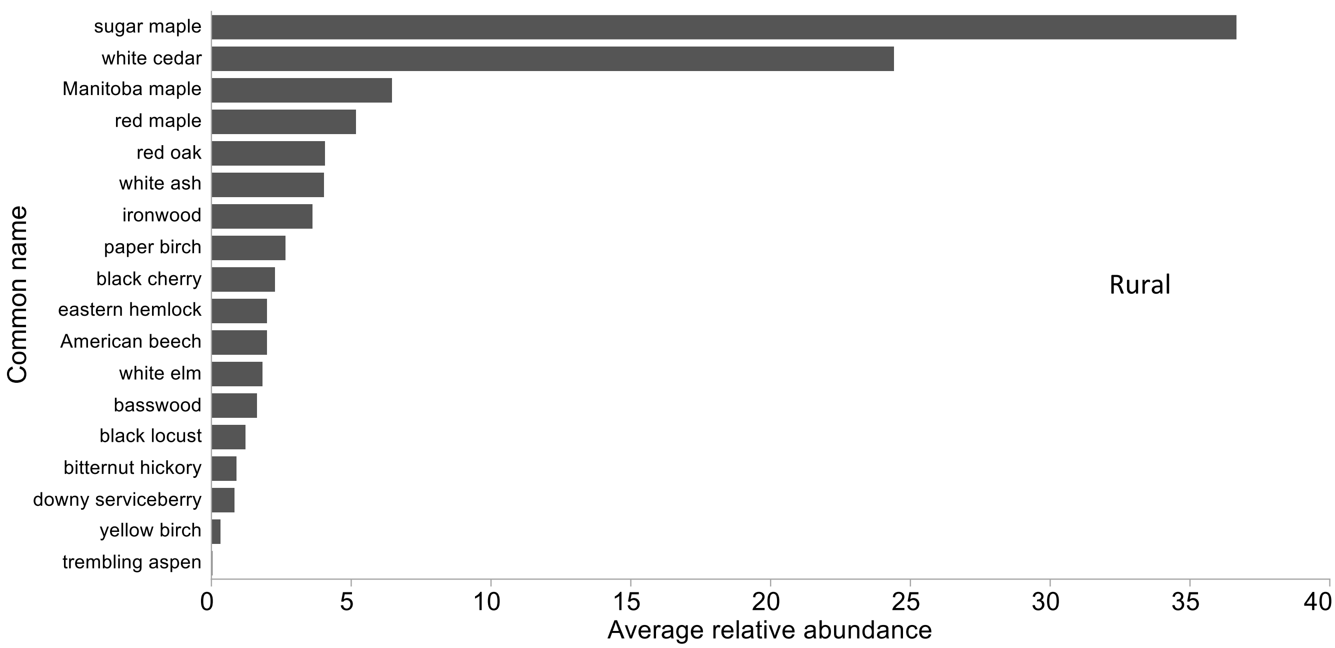


Figure 4. Average relative abundance of tree species in regional tree health plots in urban (top) or rural (bottom) areas (2011-2014). Exotic species are indicated with an asterisk (*).

Total quantity of woody regeneration

Density of woody regeneration (tree saplings, shrubs, and woody vines) varied across the plots (Figure 5). Between 2011 and 2020, the densest plot was at Heart Lake in the upper Etobicoke Creek watershed 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 although density was the highest in 2015 and 2016 and has yet to return to pre-2015 densities (Figure 5). This pattern was most pronounced in urban plots. Heart Lake strongly affected this pattern increasing from 171 woody stems in 2014 to 1942 stems in 2016. Even when Heart Lake was removed from this analysis, the general pattern remained, although was not as pronounced. Choke cherry (*Prunus virginiana* var. *virginiana*), one of the most important native species in the regeneration layer, decreased in both abundance and cover between 2011 and 2020 (Figure 5).

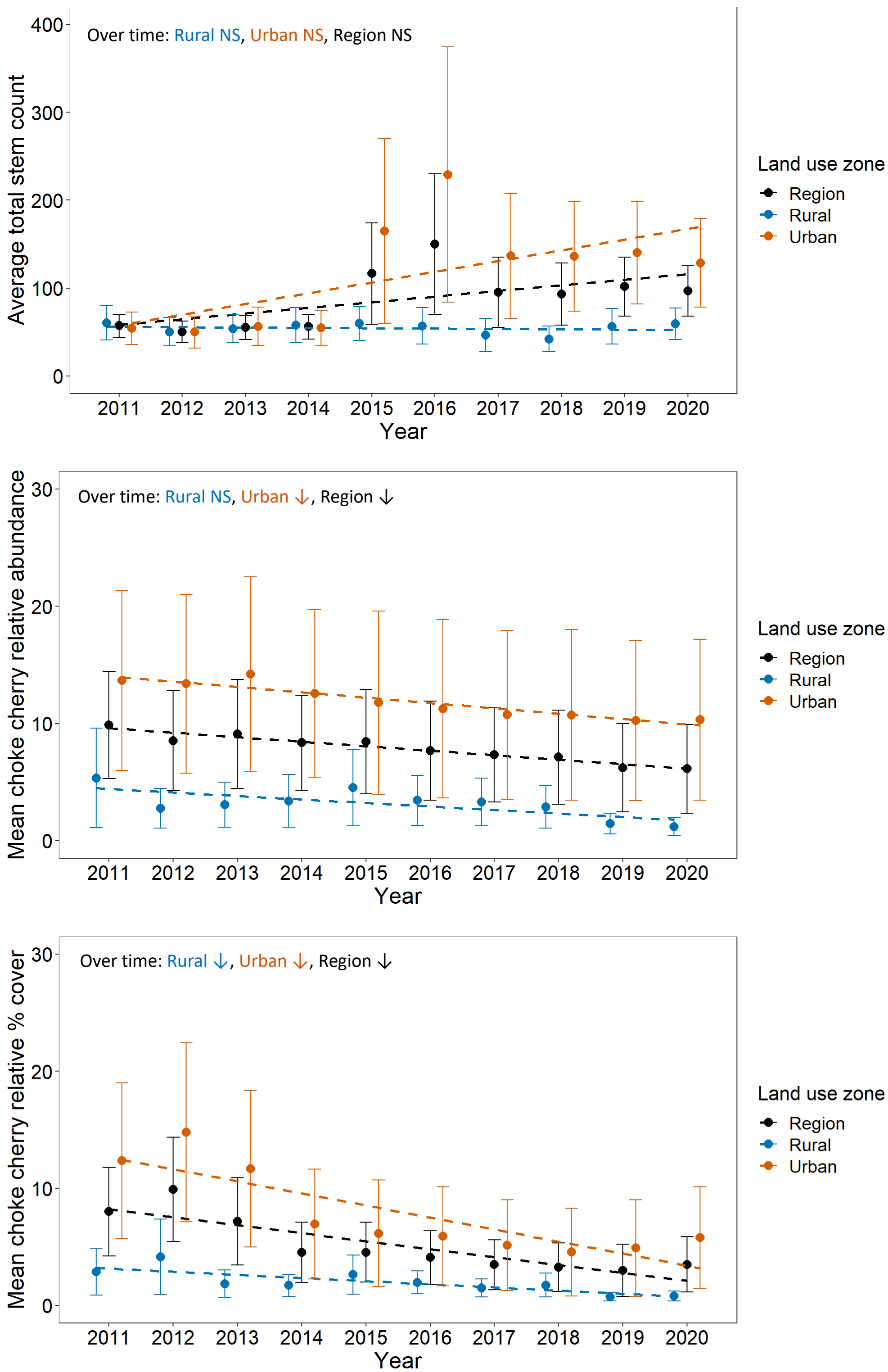


Figure 5. Spatial patterns and temporal trends in forest vegetation regeneration between 2011 and 2020 including the total number of stems (top), choke cherry relative abundance (middle), and choke cherry relative cover (bottom). An arrow pointing up (↑) represents a statistically significant increasing trend while an arrow pointing down (↓) represents a statistically significant decreasing trend. NS (non-significant) means there was no statistically significant trend.

Woody regeneration species composition

A total of 60 woody species were found in the regeneration layer across the region with an average of 6 species per site. A total of 41 species were found in the regeneration layer in the rural zone and 48 species in the urban zone (Figure 6). Portage Trail had the highest species richness of woody regeneration across the region with on average 15 species found at the site between 2011 and 2020. Reesor Rd & Hwy 7 had no regeneration.

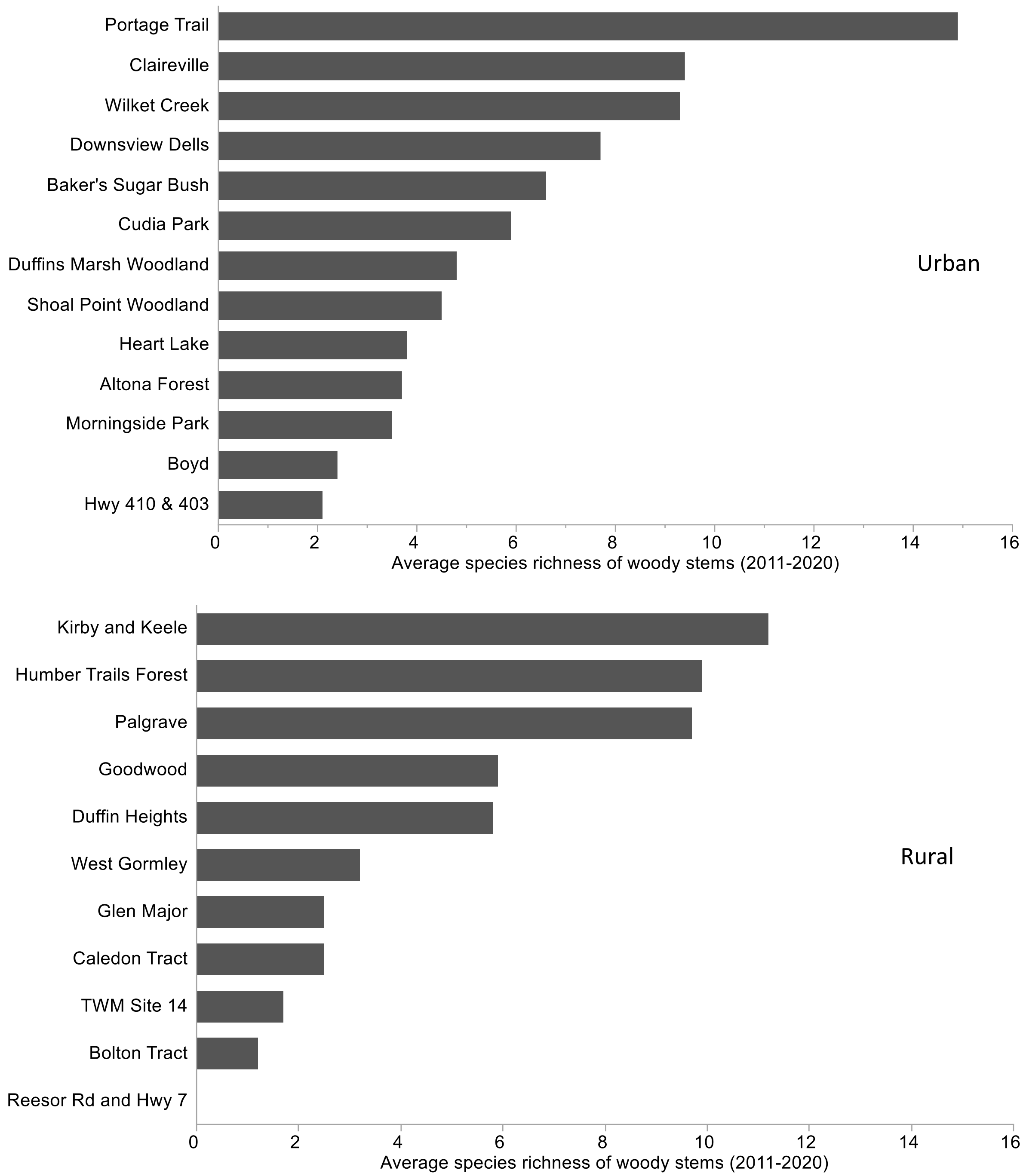
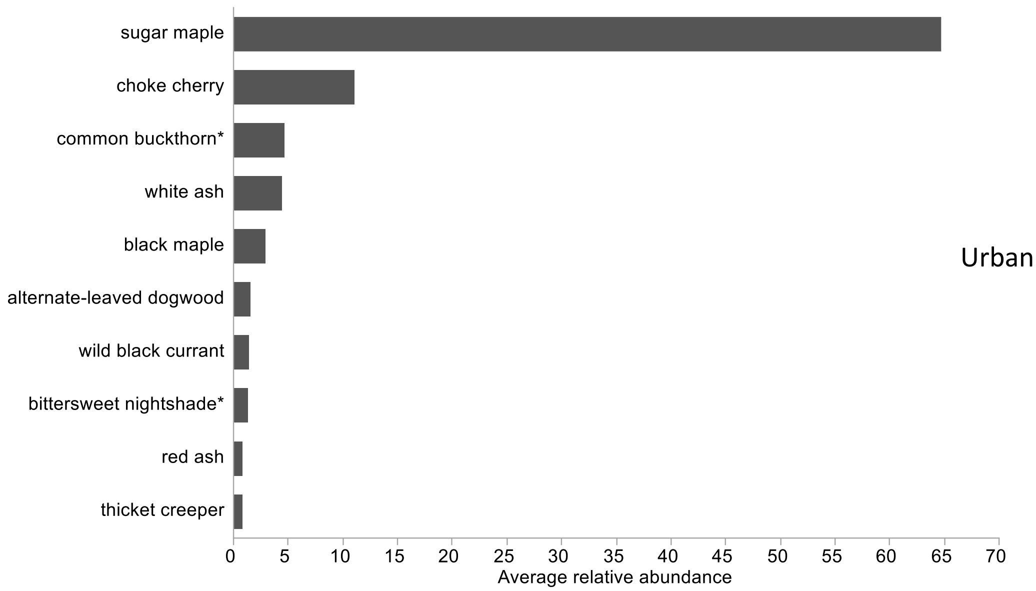


Figure 6. Species richness of woody vegetation in the regeneration layer by site measured as the average number of species per site between 2011 and 2020 in urban areas (top) and rural areas (bottom).

The regeneration community in both urban and rural areas was dominated by sugar maple based on the number of stems and percent cover (Figures 7 and 8). This is slightly different from TRCA (2015) where choke cherry dominated as relative abundance in urban areas. Common buckthorn, an invasive plant species, was the most dominant exotic species in the regeneration layer, found in a higher abundance and cover in plots in rural areas.



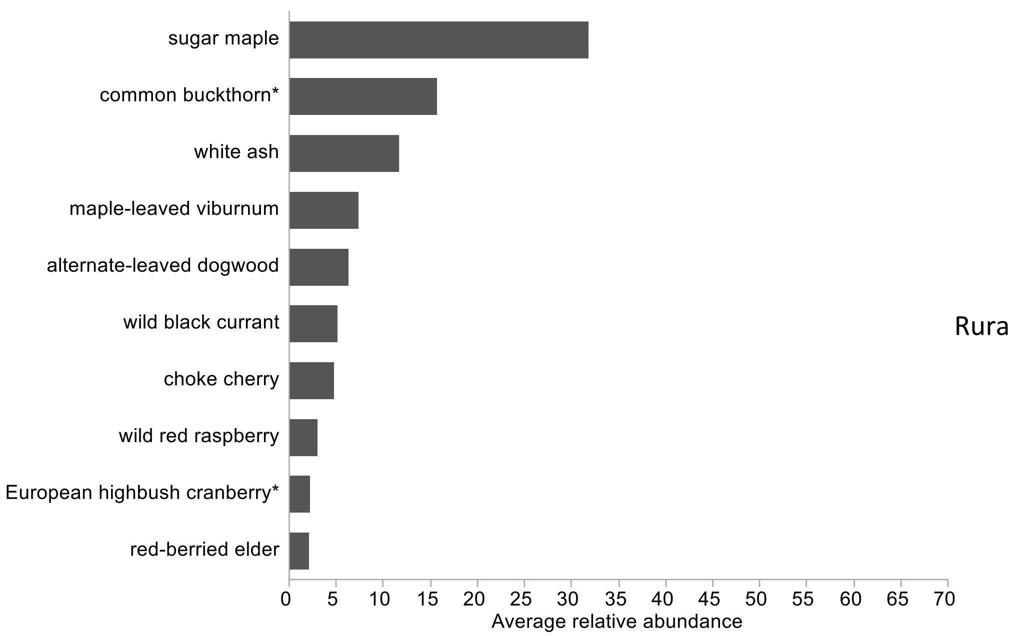


Figure 7. Average relative abundance of the 10 most common species of woody regeneration (2011-2020) in urban areas (top) and rural areas (bottom). Exotic species are indicated with an asterisk (*).

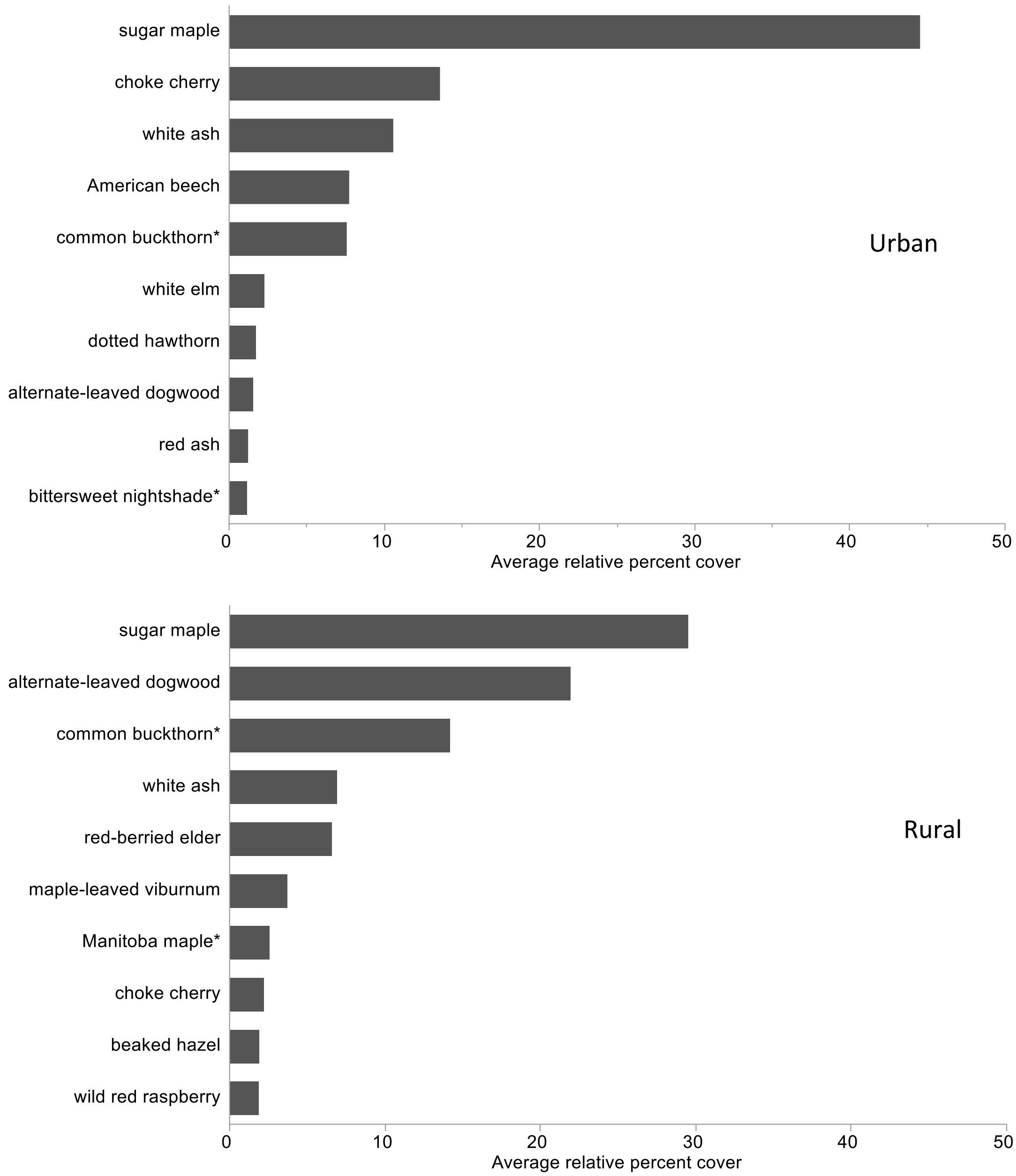


Figure 8. Average relative percent cover of the 10 most common species of woody regeneration (2011-2020) in urban areas (top) and rural areas (bottom). Exotic species are indicated with an asterisk (*).

The relative abundance and cover of native species was considerably higher than exotic species in both the rural and urban land use zones (Table 3). This demonstrates that the woody regeneration layer in our forests is still dominated by native species.

Table 3. Relative abundance and cover of native and exotic woody regeneration species by land use zone (2011-2020).

Measure	Rural		Urban	
	Relative abundance (%)	Relative cover (%)	Relative abundance (%)	Relative cover (%)
Native	81.7	81.8	92.8	89.7
Exotic	18.3	18.2	7.2	10.3

While we did not examine changes over time for all species in the regeneration layer, we did notice increases in the stem count of white ash between 2011 and 2020 (Figure 9). These increases were statistically significant at both rural and urban sites.

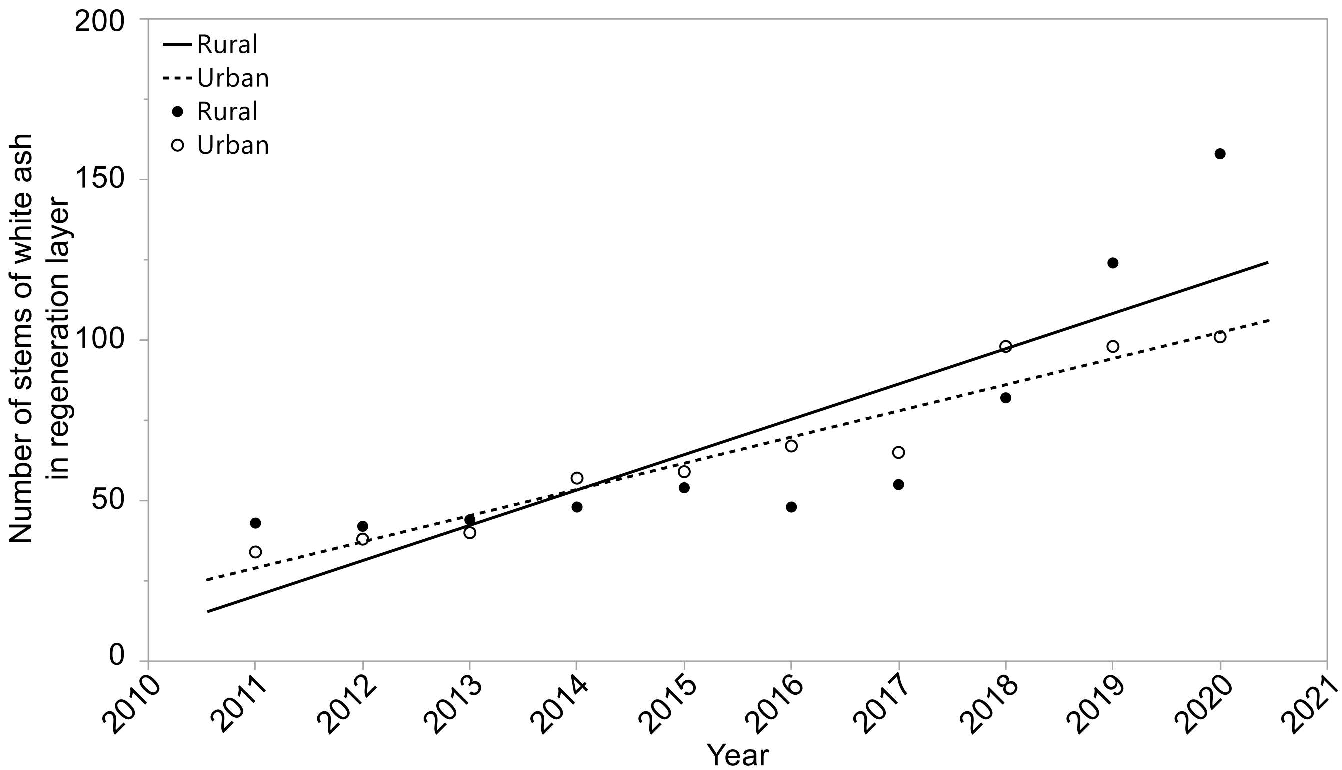


Figure 9. Change in stem count of white ash in the regeneration layer of forests between 2011 and 2020.

Ground vegetation composition

Across the jurisdiction, 218 species were identified in ground vegetation plots excluding species that could only be identified to genus. Of the 218 species, 50 were exotic (23%) and 168 were native (77%).

Yellow trout lily (*Erythronium americanum* ssp. *Americanum*; 12%), orange touch-me-not (*Impatiens capensis*; 9.6%), sugar maple (8.3%), garlic mustard (*Alliaria petiolata*; 6.5%), and ostrich fern (*Matteuccia struthiopteris* var. *pensylvanica*; 6%) had the top five highest relative covers in the jurisdiction.

Relative cover of each species varied by zone with rural sites dominated by yellow trout lily, ostrich fern, and orange touch-me-not and urban sites dominated by garlic mustard, yellow trout lily, and sugar maple (Figure 10). In TRCA (2015), 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; however, squirrel-corn is now ranked 13th in the rural zone with no L3 species occurring in the top t10.

There were no exotic species ranking in the top 10 at rural sites; however, garlic mustard had the highest relative percent cover of all species in urban sites. This is a cause for concern and suggests that garlic mustard is dominating the ground vegetation layer in urban sites. Relative cover of dog-strangling vine (*Cynanchum rossicum*) ranked 8th in the urban zone. Garlic mustard was found at 12 of 14 urban sites and only 4 of 11 rural sites. Sites with particularly high average cover of garlic mustard between 2011 and 2020 were Morningside Park (FV-15) at 36%, Cudia Park (FV-14) at 28%, and Downsview Dells (FV-4) at 21% relative cover. Maximum relative cover of exotic species was higher in urban sites (25%) compared to rural sites (8%; Table 4).

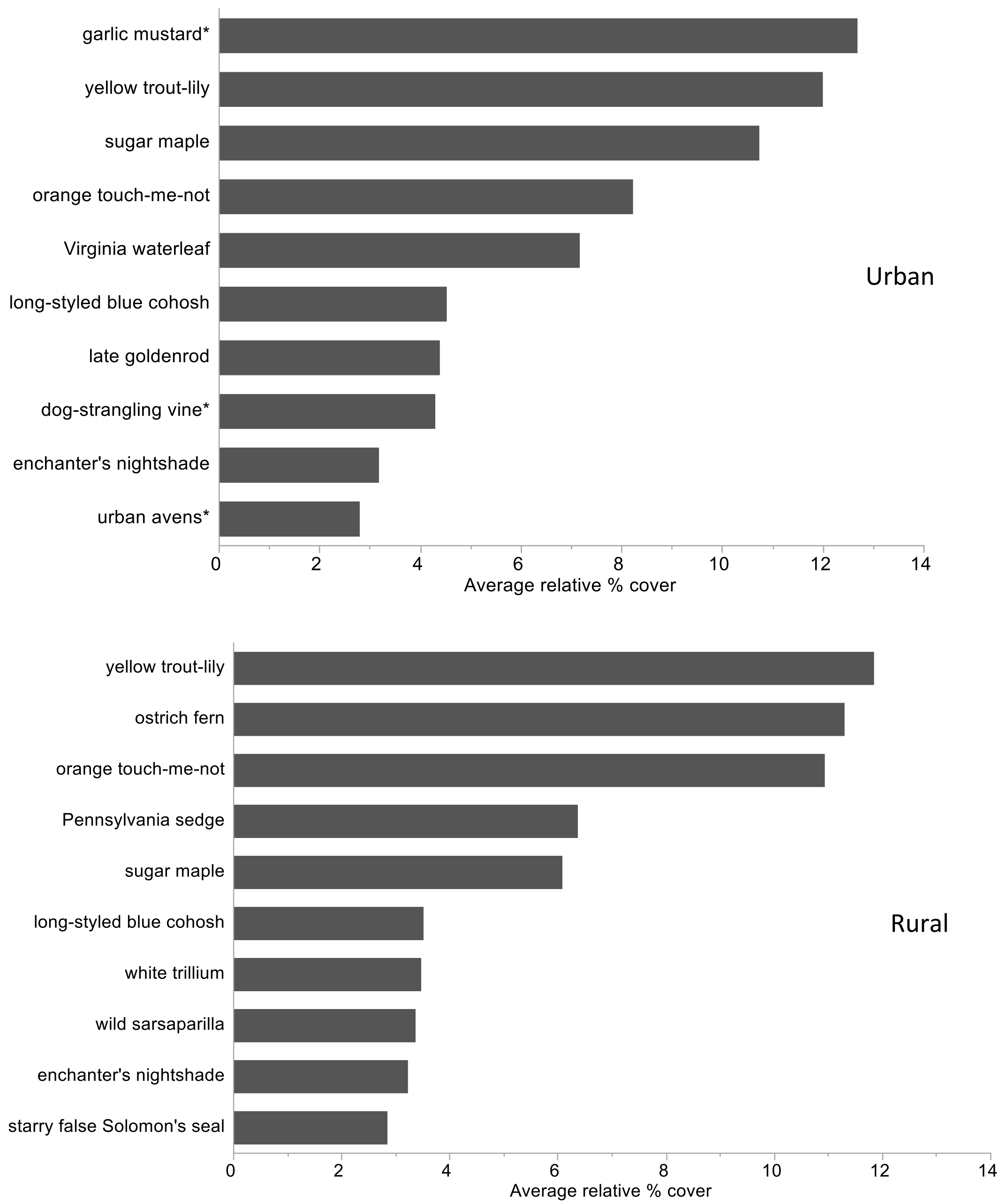


Figure 10. Average relative percent cover of the 10 most common ground vegetation species (2011-2020) in urban areas (top) and rural areas (bottom). Exotic species are indicated with an asterisk (*).

Table 4. Relative maximum % cover of native and exotic species in the rural and urban zones.

	Relative percent cover	
	Rural	**Urban**
Native	92.1	74.9
Exotic	7.9	25.1

Spread of exotic and/or invasive species

We measured the spread of invasive exotic and exotic species by examining changes in the relative cover in the ground vegetation layer for all exotic species combined and garlic mustard independently. We examined the spread of common buckthorn in the regeneration layer.

There was an increase in the relative percent cover of all exotic species combined in the ground vegetation layer in urban sites and regionally between 2011 and 2020 (Figure 11). In fact, there appeared to be an increase in cover starting in 2015, peaking in 2016, followed by a decrease, although not returning to pre-2015 levels. Garlic mustard when analyzed separately appeared to demonstrate a similar pattern both at the regional scale and at urban sites although the results were only approaching significance (p=0.11) likely due to the low cover in 2019. The relative cover and abundance of common buckthorn in forests increased at sites in rural areas and this pattern in rural sites contributed to a statistically significant increase regionally for percent cover (Figure 11). 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.

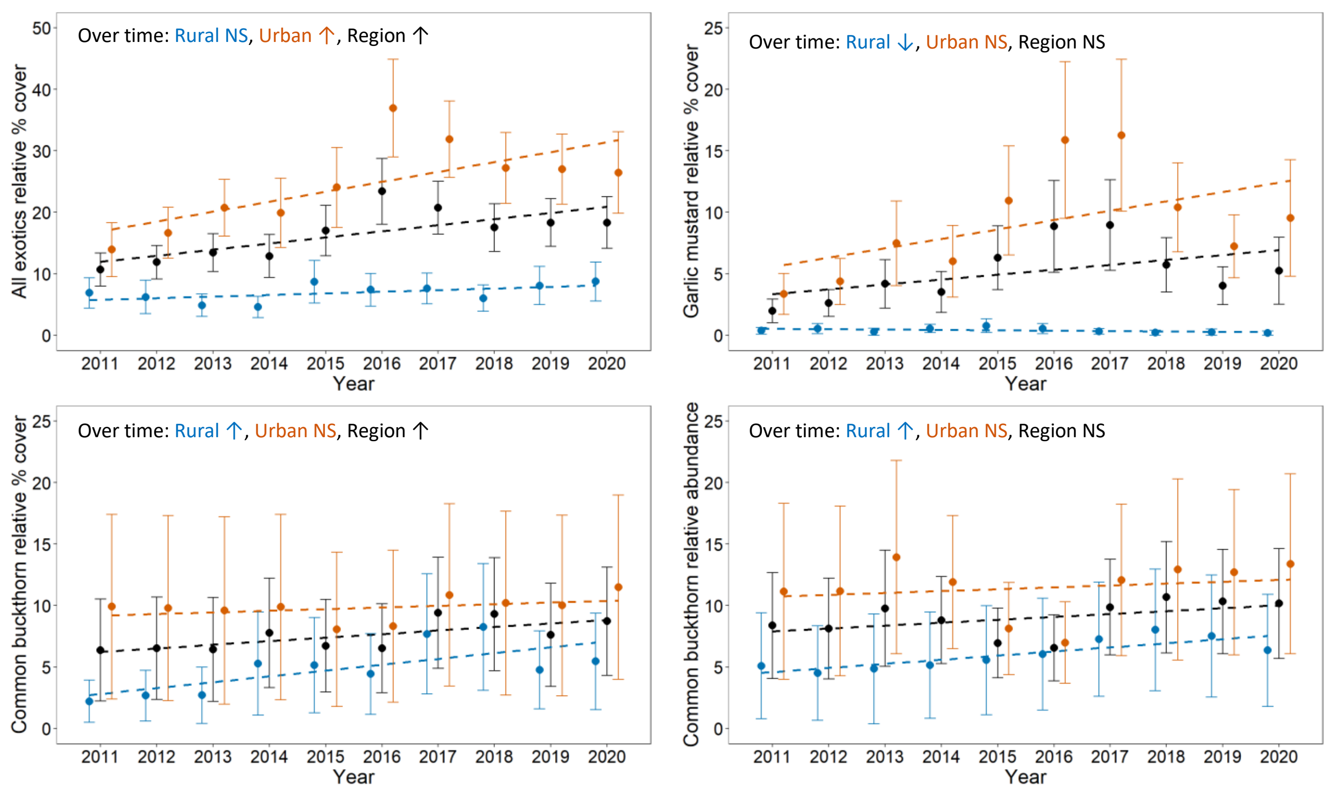
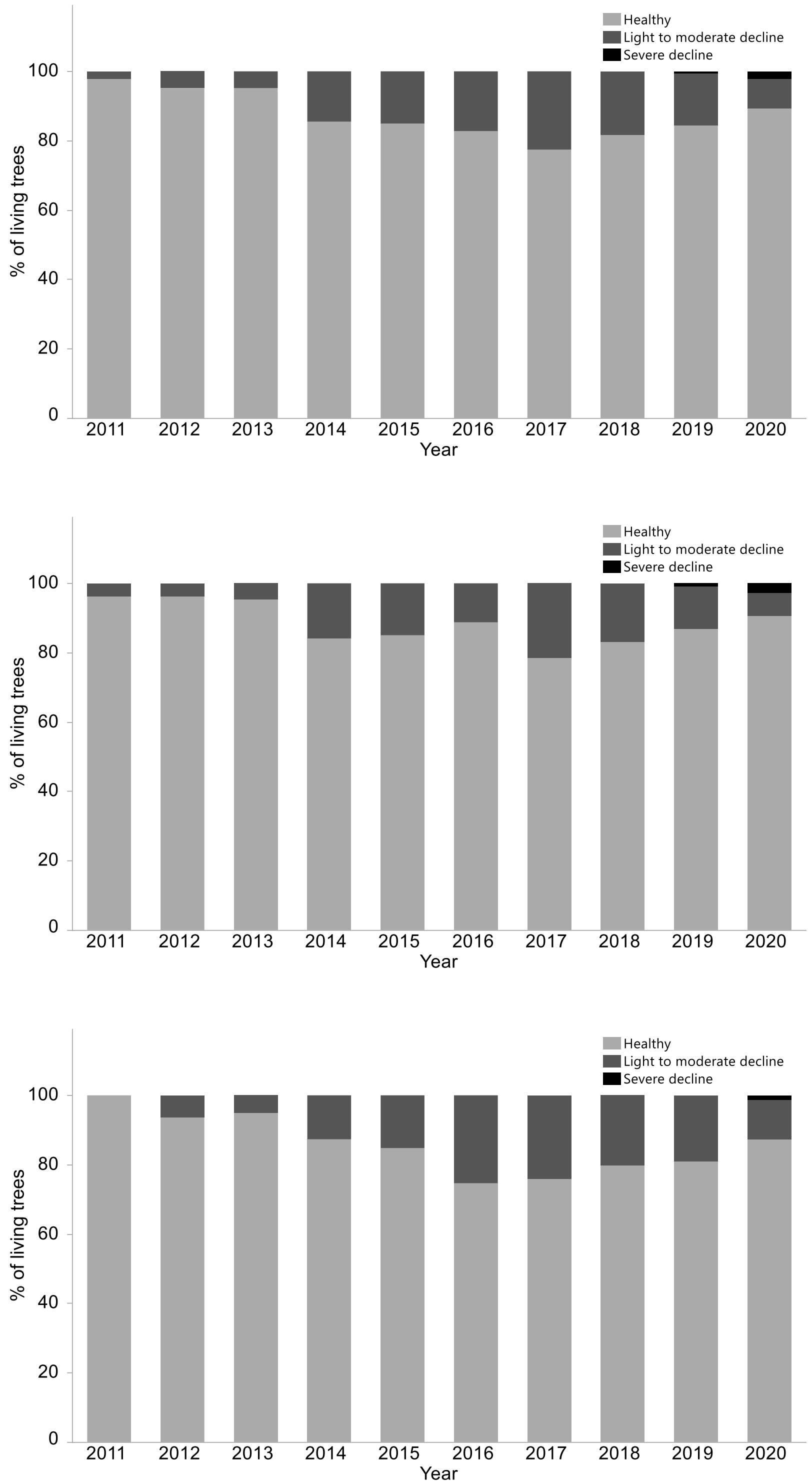


Figure 11. Temporal changes in relative percent cover of all exotic species (top-left), relative percent cover of garlic mustard (top-right), relative percent cover of common buckthorn (bottom-left), and relative abundance of common buckthorn (bottom-right). An arrow pointing up (↑) represents a statistically significant increasing trend while an arrow pointing down (↓) represents a statistically significant decreasing trend. NS (non-significant) means there was no statistically significant trend.

Crown vigour

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, we only considered trees with crown classes of dominant and co-dominant (classes 1 and 2) for the analysis. Due to methodological inconsistencies, we did not include crown vigour category 4 (dead) and only live trees were included. We excluded trees with a missing crown class or a crown class of zero.

Crown vigour of dominant and co-dominant trees consisted primarily of healthy trees (88.5% on average between 2011 and 2020 across the region; Figure 12). On average 11.1% of trees were in light to moderate decline and 0.4% were in severe decline. Regionally, between 2011 and 2020 the percent of live trees with healthy crowns appeared to decline until 2017, after which there was an apparent increase. There was no significant increasing or decreasing trend for the entire time period although the only trees observed in severe decline were in 2019 and 2020. Sites in the urban and rural land use zones had a similar distribution of crown vigour categories although there appeared to be a higher percentage of trees with light to moderate decline at rural sites in 2016 compared to urban sites in 2016.



Region

Urban

Rural

Figure 12. Temporal changes in crown vigour of living trees with crown classes 1 and 2 (dominant and co-dominant) regionally (top), urban sites (middle), rural sites (bottom).

There was variation in which tree species showed signs of decline (Figure 13). American beech, white ash, and black cherry (*Prunus serotina*) all appeared to have had declines in crown vigour over time that were greater than those of other species which either only had a small decrease in crown vigour or variable crown vigour.

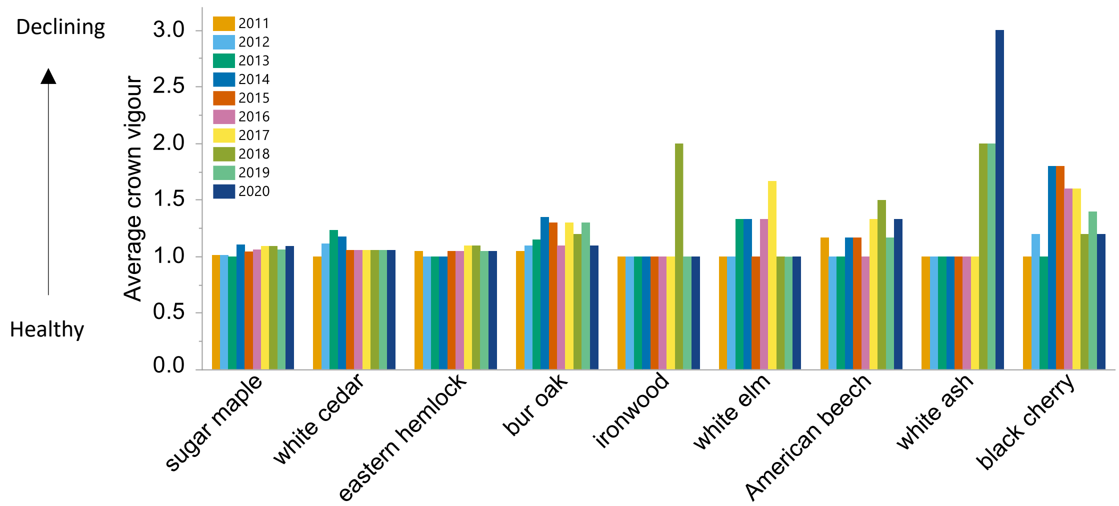


Figure 13. Temporal trends in average crown vigour (dominant and co-dominant) for selected tree species between 2011 and 2014. For each species, bars on the graph run left to right chronologically by year.

In addition to crown vigour, we measured mortality rates between years. Trees displaying decreased crown vigour could die in the year in which they displayed decreased crown vigour or a in subsequent 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 2011 and 2020 were included in this analysis (n=213 trees). In general, there were low mortality rates with only one or two trees dying per year between 2011 and 2015; however, mortality rates increased starting in 2016 (Table 5).

Many of the ash trees that died showed signs of EAB (e.g. exit holes) and all showed declines in crown vigour before dying. Most trees, other than ash, 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 and 2013 with epicormic growth and missing bark, with Dutch elm disease suspected.

The yellow birch that died between 2018 and 2019 was an old tree and succumbed to several ailments including rot, root girdling, and carpenter ants. Other species showed signs of decline before dying including exit holes, dieback, rot, open wounds, or carpenter ants.

There was no statistically significant change over time in mortality. Of the 25 trees that died between 2011 and 2020, 18 were in the rural zone and 7 were in the urban zone.

Table 5. Annual tree mortality between 2011 and 2020 based on tree status.

Years	Mortality (% of trees that died)	Species
2011-2012	0.47	trembling aspen
2012-2013	0.94	sugar maple, bitternut hickory
2013-2014	0.95	sugar maple, white elm
2014-2015	0	-
2015-2016	2.4	red maple, red ash, white ash (3)
2016-2017	1.5	basswood, black cherry, white ash
2017-2018	3	Eastern hemlock (2), sugar maple, red maple, white pine, white ash
2018-2019	3.1	yellow birch, red ash, white ash (4)
2019-2020	0	-

Snags

We included dead broken (DB), dead standing (DS), and dead leaning (DL) trees as snags and all live trees were included in the alive category. The percentage of trees classified as snags or alive was relatively constant between 2011 and 2020 but there appeared to be a slight increase in the number of snags starting in 2016 regionally and in rural sites (Figure 14). Across the region on average 89.5% of trees were alive and 10.5% of trees were snags.

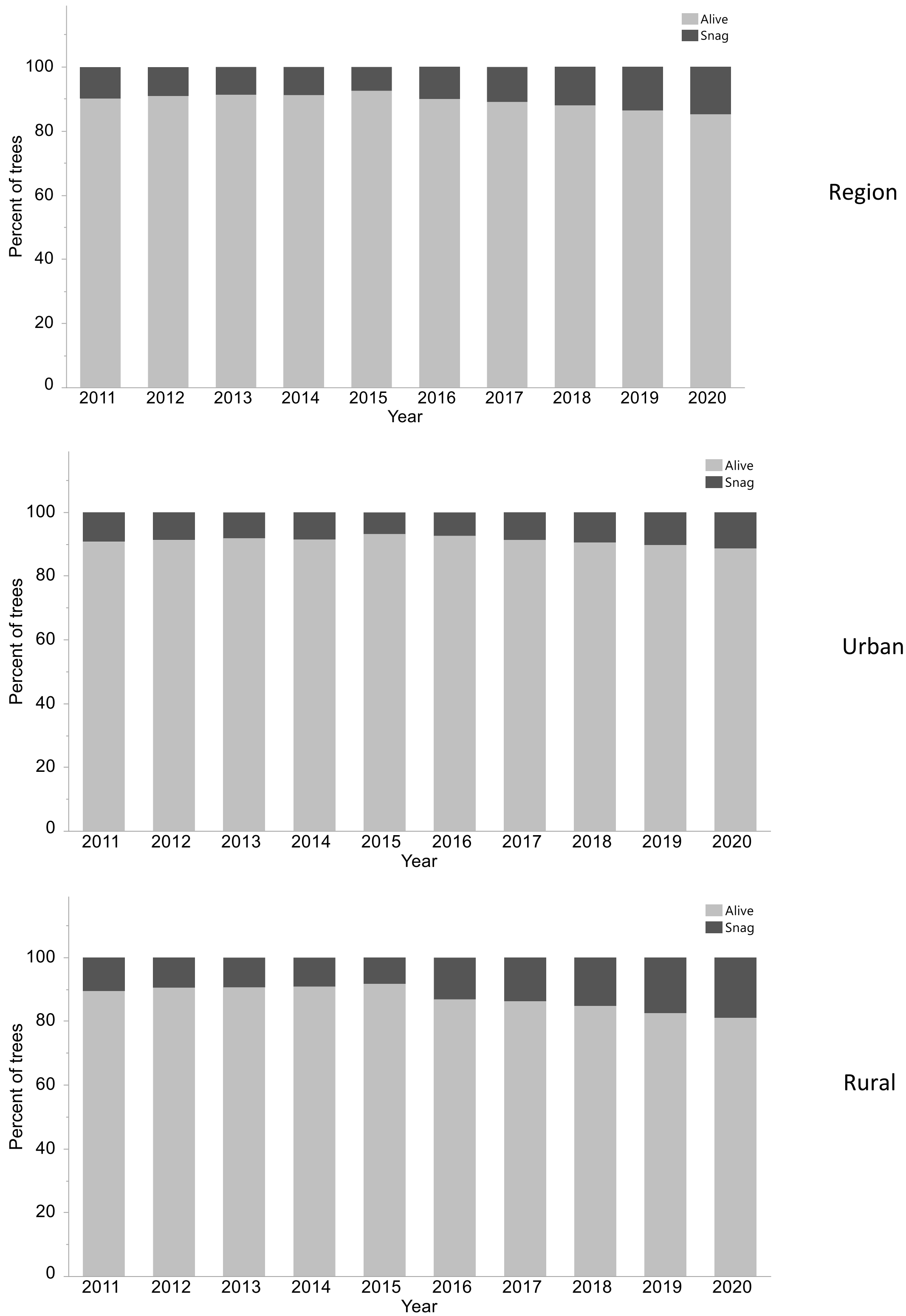


Figure 14. The percentage of trees classified as alive or snags regionally (top), at urban sites (middle), and at rural sites (bottom).

Sugar maple was the most abundant snag tree regionally and in rural areas, while white elm snags dominated the urban areas (Table 6). Since TRCA (2015), snags of sugar maple, white cedar, white ash, and bur oak have increased in relative abundance in the jurisdiction. There were no white ash snags in the top six in TRCA (2015); however, white ash is now ranked 4th regionally, 6th in urban areas, and 4th in rural areas.

Table 6. Composition of the six most abundant snag species (average % relative abundance) observed in TRCA regional plots between 2011 and 2020.

Region		Urban		Rural	
sugar maple	17.4%	white elm	15.9%	sugar maple	23.1%
white elm	12.9%	bur oak	14.8%	white cedar	15.5%
white cedar	9.4%	ironwood	14.4%	white elm	10.8%
white ash	8.9%	American beech	9.6%	white ash	9.5%
bur oak	6.1%	sugar maple	9.3%	red maple	8.7%
ironwood	6.0%	white ash	8.2%	apple	7.9%

Canopy cover

Canopy cover assessments were added to surveys in 2018. Between 2018 and 2020, percent canopy cover ranged from 28% to 96% with an average of 89%. The single plot with 28% cover suffered from severe defoliation due to Spongy Moth (formerly gypsy moth; *Lymantria dispar dispar*) changing from 90% cover in both 2018 and 2019 to 28% in 2020. Rural sites appeared to have a slightly lower percent canopy cover than rural sites (Table 7).

Table 7. Average percent canopy cover in urban and rural plots between 2018 and 2020.

Land use	Percent canopy cover		
	2018	2019	2020
Rural	90.8	87.3	82.9
Urban	91.6	89.1	90.8

Pests and disease

Incidences of Anthracnose were generally low across the jurisdiction peaking in 2016 (Figure 15). Anthracnose affected a range of tree species but predominantly sugar maple, bur oak, basswood, and American beech. The percentage of trees affected was low in both urban and rural sites (Table 8).

Spongy Moth was the most prevalent pest/disease in the region affecting a low of 0.7% of trees in 2011 and a high of 34.3% in 2020 (Figure 15). Spongy Moth affected many different tree species although sugar maple, red oak, and bur oak were most heavily affected. Urban sites appeared to be more heavily affected (9.1% of trees) compared to rural sites (5.7% of trees) (Table 8).

EAB monitoring started in 2014 and its prevalence increased over time. The percent of trees affected was likely much higher in later years but since trees started dying, they were no longer monitored for pests and disease

(Figure 15). EAB affected a similar number of stems in urban and rural areas although the percentage of ash stems affected appeared to be higher in urban plots (Table 8). Shoal Point Woodlot (FV-18), Humber Trails Forest & Wildlife Area (FV-10), Bolton Tract (FV-9), and Boyd (FV-6) contain a large proportion of the ash trees in regional plots and were heavily affected by mortality related to EAB.

Beech bark disease and the associated vector, Beech Scale (*Cryptococcus fagisuga*), affected almost every beech tree between 2011 and 2020 (range 84-100%) (Figure 15). Trees in urban and rural areas appeared to be almost equally affected (Table 8).

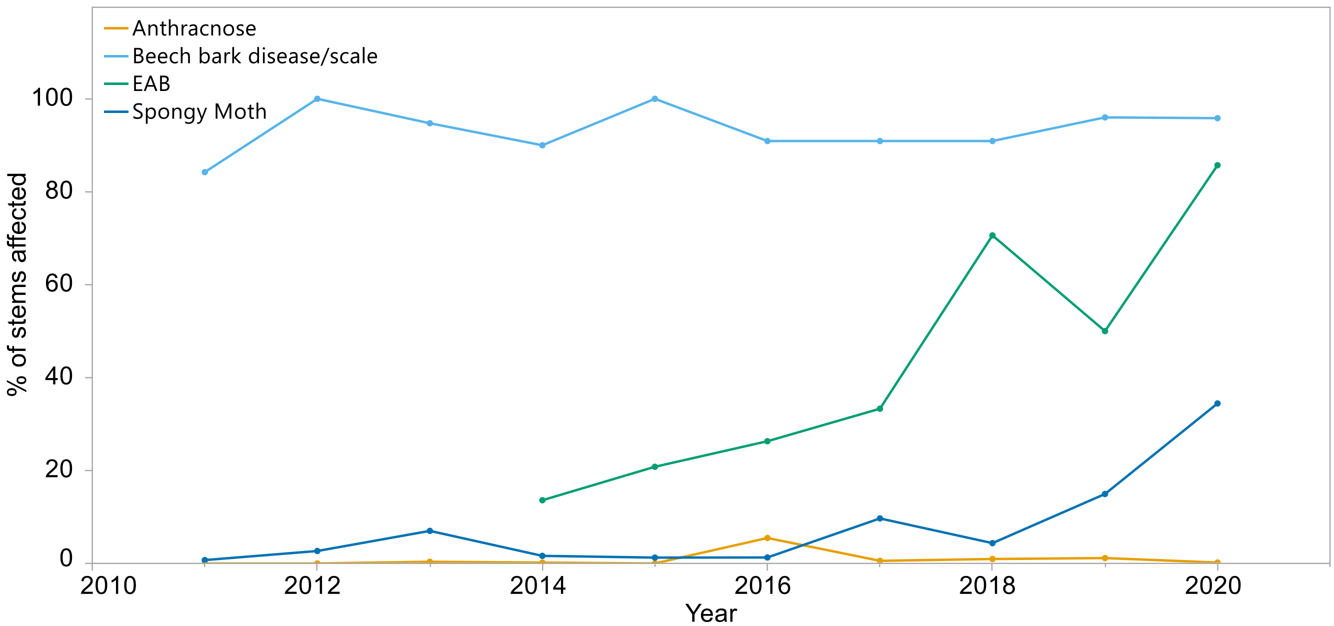


Figure 15. Occurrence of select pests and diseases in TRCA forest plots between 2011 and 2020. Emerald Ash Borer monitoring started in 2014.

Table 8. Occurrence of select pests and diseases in TRCA forest plots in urban and rural areas between 2011 and 2020.

Pest/disease		Urban	Rural
Anthracnose	# live stems	17	31
	% live stems	0.6	1.3
Spongy Moth	# live stems	274	138
	% live stems	9.1	5.7
EAB	# live stems	21	20
	% live stems	40	32
Beech bark disease/scale	# live stems	155	45
	% live beech stems	93.9	91.8

Forest birds

In general, there was either no change or an increase in forest bird indicators between 2011 and 2020 (Figure 16). There were significant increases in the abundance of forest-dependent birds regionally, in the urban zone, and in the rural zone; however, the number of forest-dependent bird species only increased in the urban zone. There was no change in the number of L1-L3 ranked forest-dependent bird species or abundance. Forest bird indicators strongly demonstrate the impact of urbanization with stations in the urban zone containing fewer forest-dependent individuals and species than stations in the rural zone with L1-L3 ranked species demonstrating this pattern more strongly.

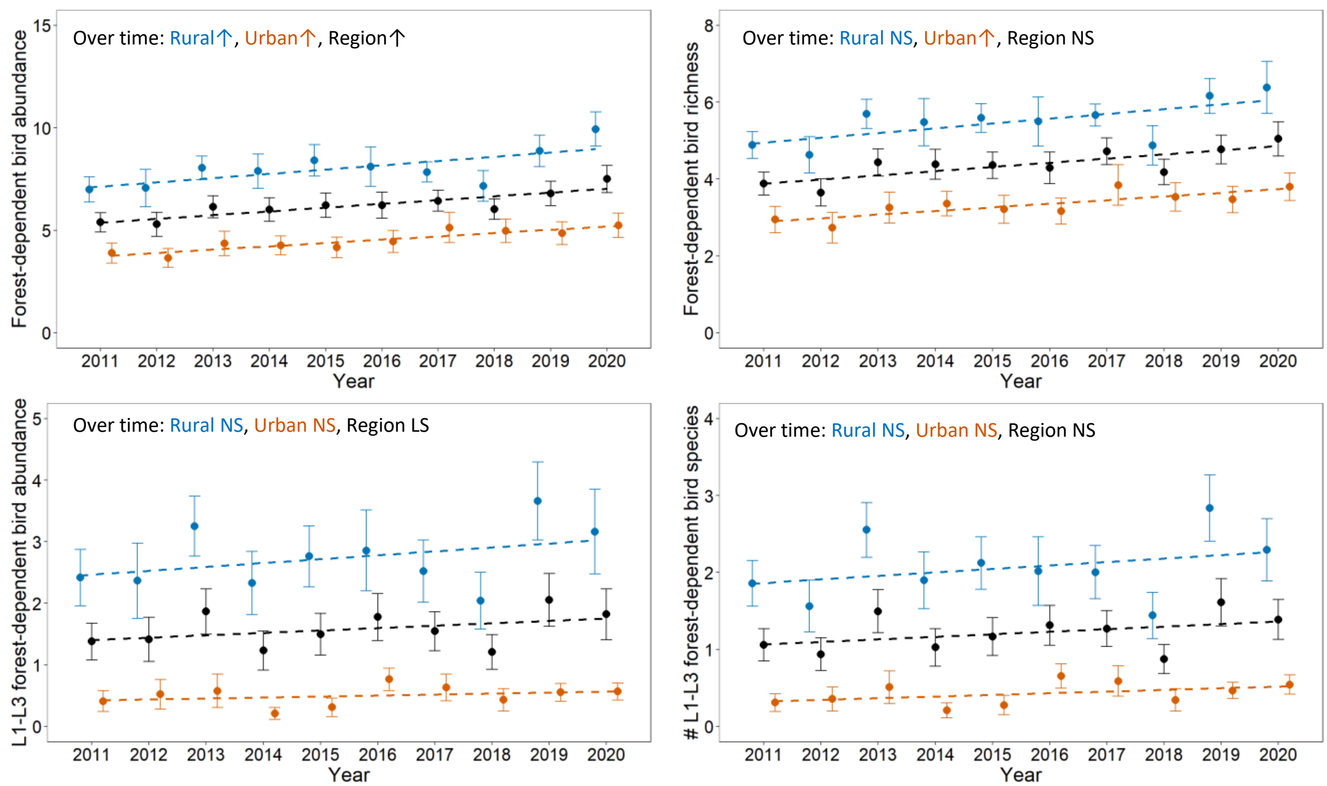
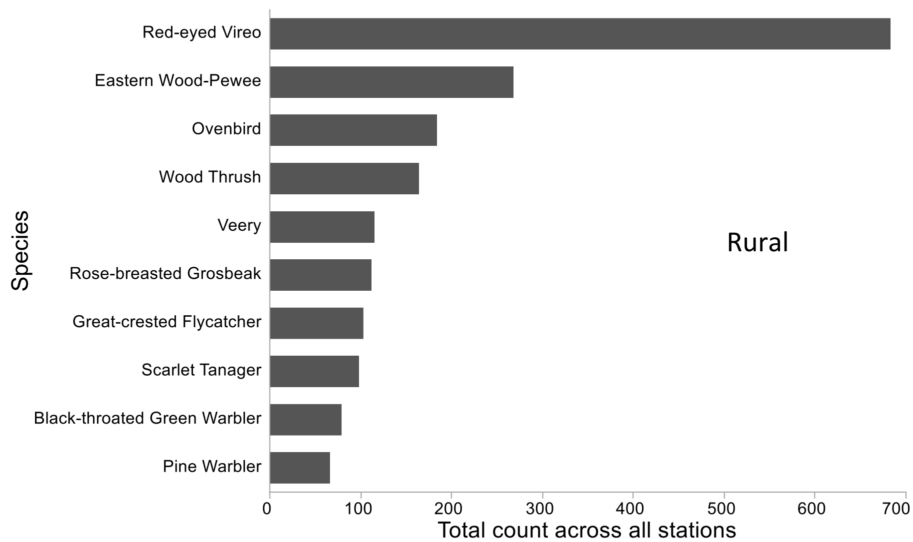


Figure 16. Spatial patterns and temporal trends in forest bird indicators between 2011 and 2020. An arrow pointing up (↑) represents a statistically significant increasing trend while an arrow pointing down (↓) represents a statistically significant decreasing trend. NS (non-significant) means there was no statistically significant trend.

Forest bird community composition varied between stations in the urban and rural zones (Figure 17). The two most abundant species in both the urban and rural zone were Red-eyed Vireo (*Vireo olivaceus*) and Eastern Wood-Pewee (*Contopus virens*) although the overall abundance was lower in the urban zone compared to the rural zone. There were several other similarities including Wood Thrush (*Hylocichla mustelina*), Rose-breasted Grosbeak (*Pheucticus ludovicianus*), Great-crested Flycatcher (*Myiarchus crinitus*), and Pine Warbler (*Setophaga pinus*); however, Ovenbird (*Seriurus aurocapillus*), Veery (*Catharus fuscescens*), Scarlet Tanager (*Piranga olivacea*), and Black-throated Green Warbler (*Setophaga virens*) were only found in the top 10 in the rural zone. Ovenbird, Veery and Scarlet Tanager are area-sensitive making them less likely to be found in smaller forest fragments in urban areas.



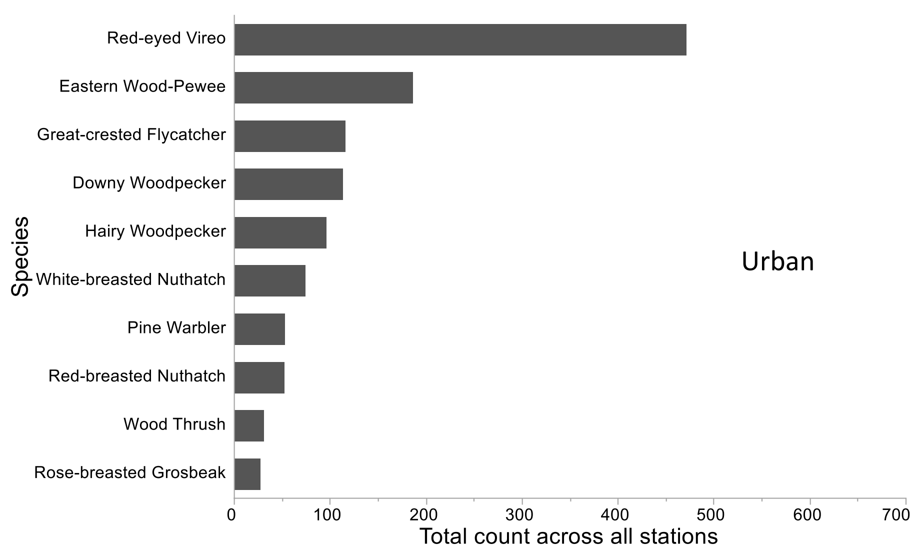


Figure 17. Forest bird community composition of the 10 most abundant species in the rural and urban zone.

With EAB moving through the jurisdiction in the mid-2010s, we examined if bark-foraging forest bird species increased in abundance over this time. We included Downy Woodpecker (*Picoides pubescens*), Hairy Woodpecker (*Picoides villosus*), Red-bellied Woodpecker (*Melanerpes carolinus*), and White-breasted Nuthatch (*Sitta carolinensis*) in this analysis as the primary bark-foraging species based on Flower et al. (2014) and their work on predation of EAB by bark-foraging birds. We found a significant increase regionally in the total number of bark-foragers across all stations (Figure 18).

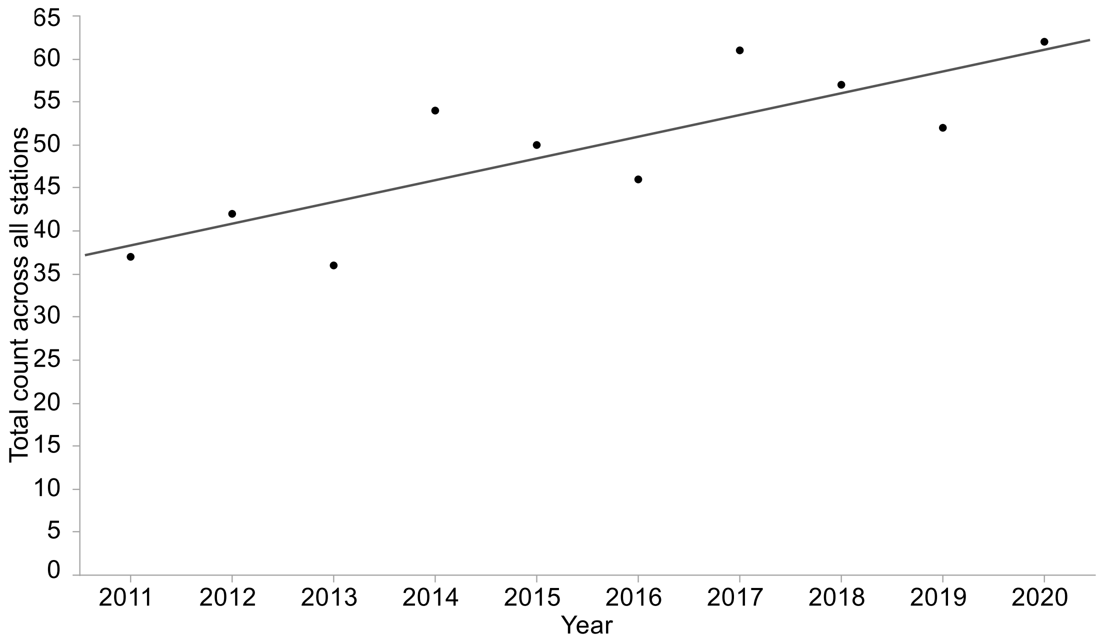


Figure 18. Temporal trends in the average abundance of bark-foraging forest bird species regionally between 2011 and 2020.

Wetland monitoring

Wetland vegetation

In general, there was either no change or an increase in wetland vegetation indicators between 2011 and 2020 (Figure 19). There were significant increases in the FQI regionally, in the urban zone, and in the rural zone; however, the mean CC score did not change over time. This suggests that the FQI may be increasing primarily due to an increase in the total number of species (species richness). The number of L1-L3 ranked species also increased regionally and in the rural zone; however, there were no other significant changes in wetland vegetation indicators. Wetland vegetation indicators strongly demonstrate the impact of urbanization with transects in the urban zone containing fewer species and species of conservation concern, a lower percentage of native species, and lower FQI and mean CC scores than stations in the rural zone.

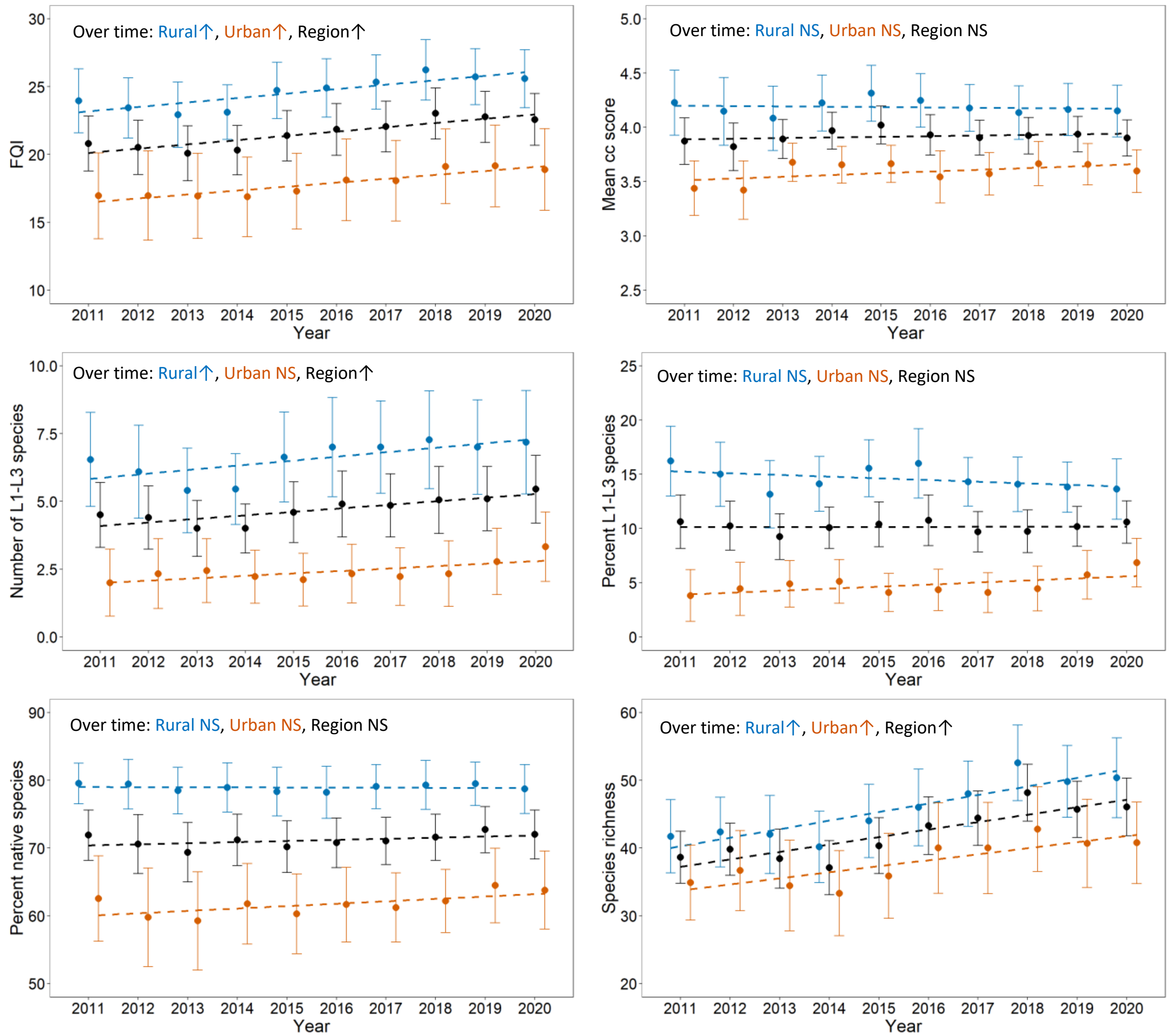


Figure 19. Spatial patterns and temporal trends in wetland vegetation indicators between 2011 and 2020. An arrow pointing up (↑) represents a statistically significant increasing trend while an arrow pointing down (↓) represents a statistically significant decreasing trend. NS (non-significant) means there was no statistically significant trend.

Wetland woody regeneration species composition

Relative percent cover of wetland woody regeneration across the jurisdiction was dominated by speckled alder (*Alnus incana* ssp. *rugosa*; 14%), red osier dogwood (*Cornus stolonifera*; 11%), common buckthorn (9%), bittersweet nightshade (*Solanum dulcamara*; 9%), and balsam fir (*Abies balsamea*; 6.6%). These rankings changed when species composition was measured using stem counts (relative abundance). Common buckthorn had the highest relative abundance (21%) followed by red osier dogwood (12%), bittersweet nightshade (11%), winterberry (*Ilex verticillata*; 6%) and speckled alder (6%) as the top five species.

Urban and rural land use zones varied based in the species composition of woody regeneration (Figure 20). Relative cover in the rural zone was dominated by red osier dogwood (15%), balsam fir (13%), and winterberry (8%). This is in contrast to the urban zone where speckled alder (20%), bittersweet nightshade (14%), and common buckthorn (14%) dominated the woody regeneration community.

Based on stem count, woody regeneration communities in rural areas were dominated by red osier dogwood (22%), common buckthorn (14%), and winterberry (10%). In urban areas, wetlands were dominated by common buckthorn (28%), bittersweet nightshade (15%), and poison ivy (shrub form) (7%).

One major concern is the continued increase in relative abundance of common buckthorn between 2011 and 2020 in wetland vegetation plots throughout the region. The relative abundance of common buckthorn has increased from 13.9 in 2011 to 23.7 in 2020.

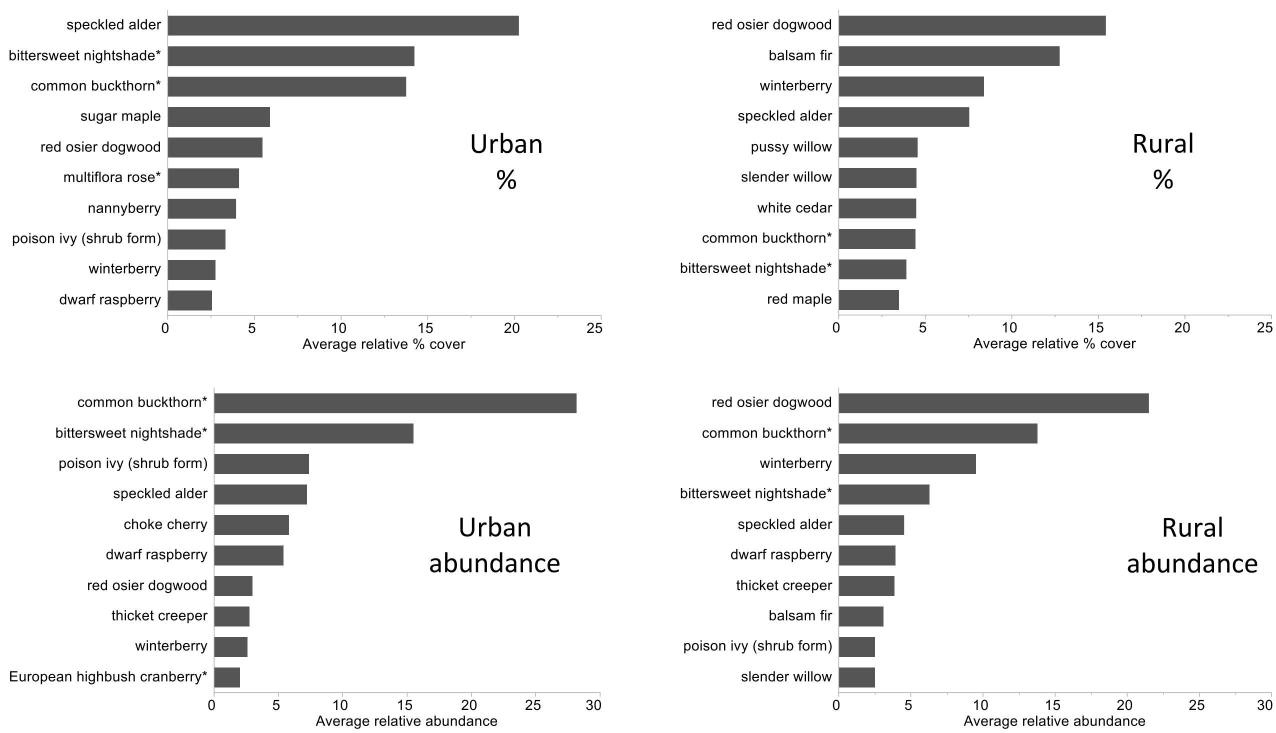


Figure 20. Average relative percent cover and abundance of the 10 most common wetland woody species in the rural zone (right) and urban zone (left). Exotic species are indicated with an asterisk.

Ground vegetation 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, and similar to TRCA (2015), turion duckweed and common duckweed were grouped for this analysis.

Wetland ground vegetation was dominated by hybrid cattail (*Typha* x *glauca*; 14%) at the regional level followed by turion/common duckweed (8%), reed canary grass (*Phalaris arundinacea*; 7%) and star duckweed (*Lemna trisulca*; 6%). Rural sites were dominated by star duckweed (10%), common reed (*Phragmites australis*; 9%), and turion/common duckweed (7%) while urban sites contained predominantly hybrid cattail (*Typha* x *glauca*; 27%), turion/common duckweed (10%), and reed canary grass (7%) (Figure 21).

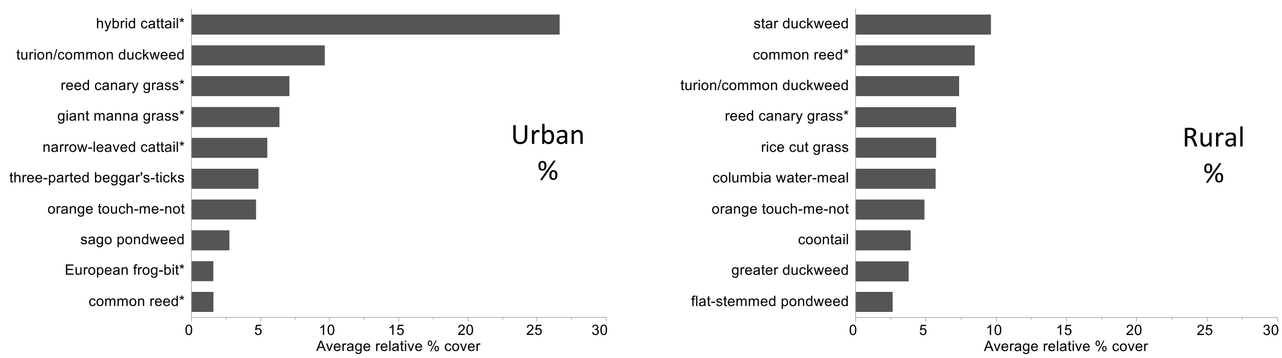


Figure 21. Average relative percent cover of the 10 most common wetland herbaceous species in the rural zone (right) and urban zone (left). Exotic species are indicated with an asterisk.

Wetland birds

There were significant increases in the abundance of wetland-dependent birds regionally and in the rural zone between 2011 and 2020 (Figure 22). There was no change in wetland-dependent bird species richness or the number of L1-L3 wetland-dependent bird species; however, the number of L1-L3 ranked wetland-dependent bird species in wetlands decreased in the urban zone. Wetland bird indicators strongly demonstrate the impact of urbanization with stations in the urban zone containing fewer wetland-dependent individuals and species than stations in the rural zone.

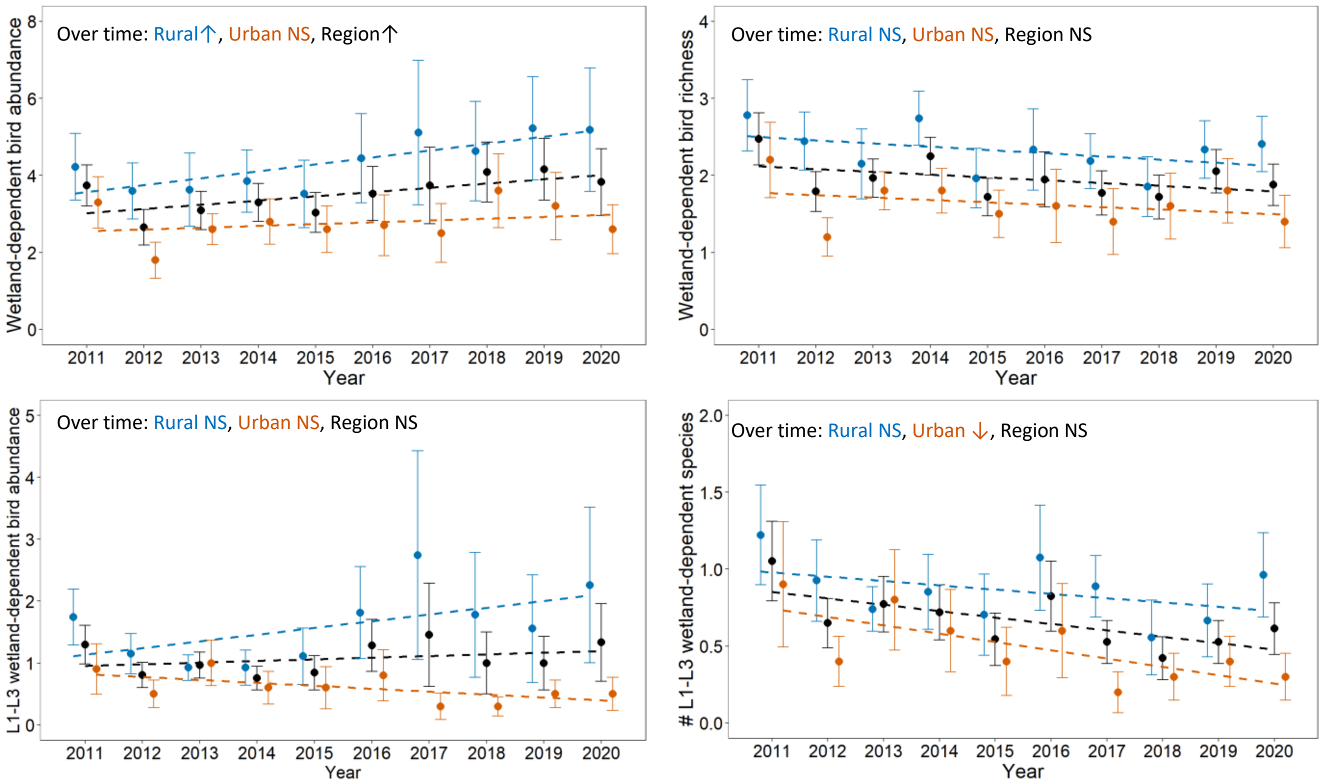


Figure 22. Spatial patterns and temporal trends in wetland bird indicators between 2011 and 2020. An arrow pointing up (↑) represents a statistically significant increasing trend while an arrow pointing down (↓) represents a statistically significant decreasing trend. NS (non-significant) means there was no statistically significant trend.

Wetland bird community composition varied between stations in the urban and rural zones (Figure 23). The three most abundant species in both the urban and rural zone were Swamp Sparrow (*Melospiza georgiana*), Common Yellowthroat (*Geothlypis trichas*), and Virginia Rail (*Rallus limicola*) although the overall abundance was lower in the urban zone compared to the rural zone. There were six other similar species including Great Blue Heron (*Ardea herodias*), Trumpeter Swan (*Cygnus buccinator*), Alder Flycatcher (*Empidonax alnorum*), Pied-billed Grebe (*Podilymbus podiceps*), and Green Heron (*Butorides virescens*). Hooded Merganser (*Lophodytes cucullatus*) was only found in the rural zone while Black-crowned Night-heron (*Nycticorax nycticorax*) was only found in the urban zone.

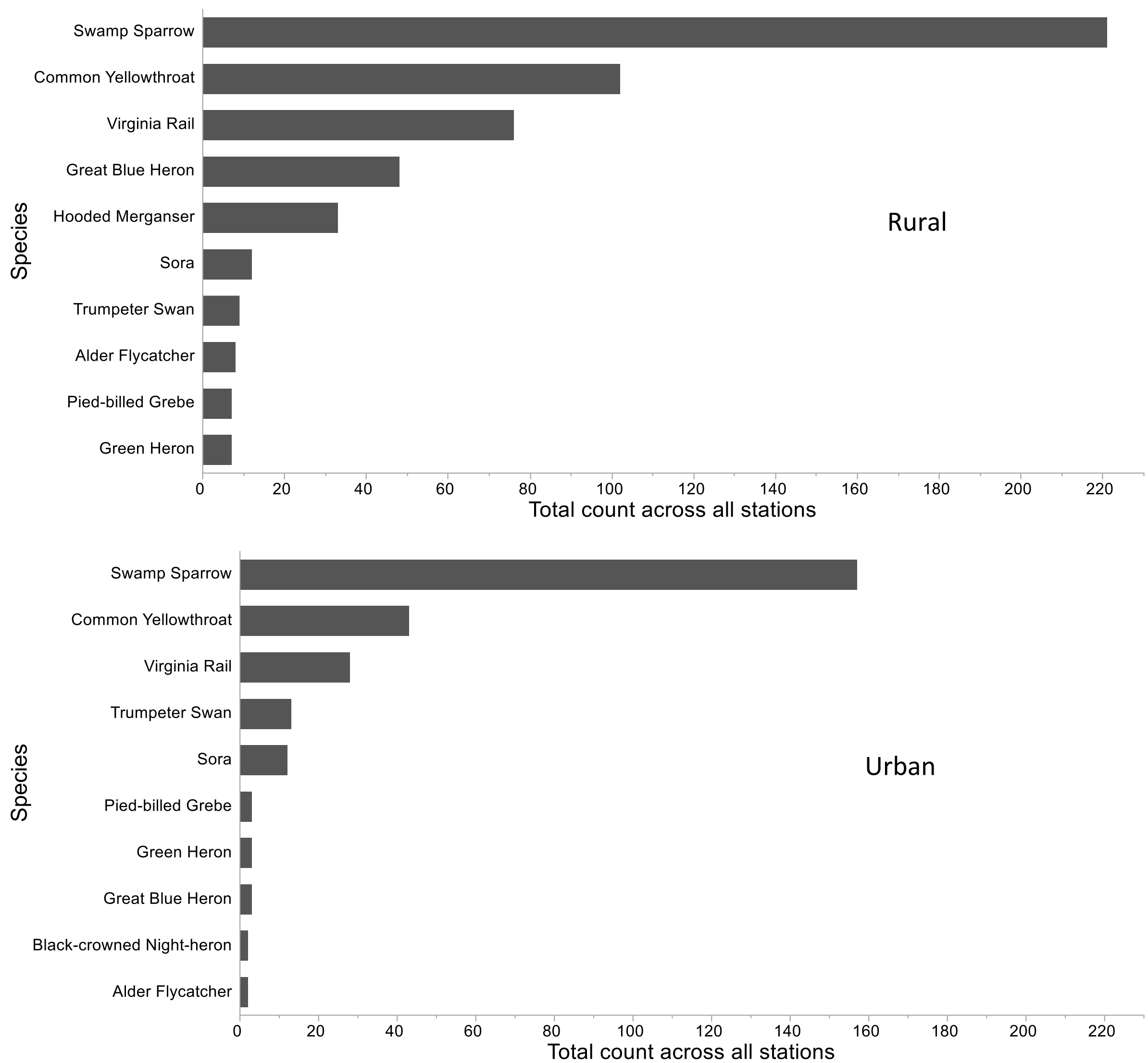


Figure 23. Wetland bird community composition of the 10 most abundant species in the rural and urban zone.

Frogs

Frog species composition

Eight frog species were detected in regional plots across the jurisdiction between 2011 and 2020 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).

Frog high level indicators

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 if there were no frogs detected at a site.

Frog communities were stable between 2011 and 2020 based on measurements using the high-level indicators (Figure 24). 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.

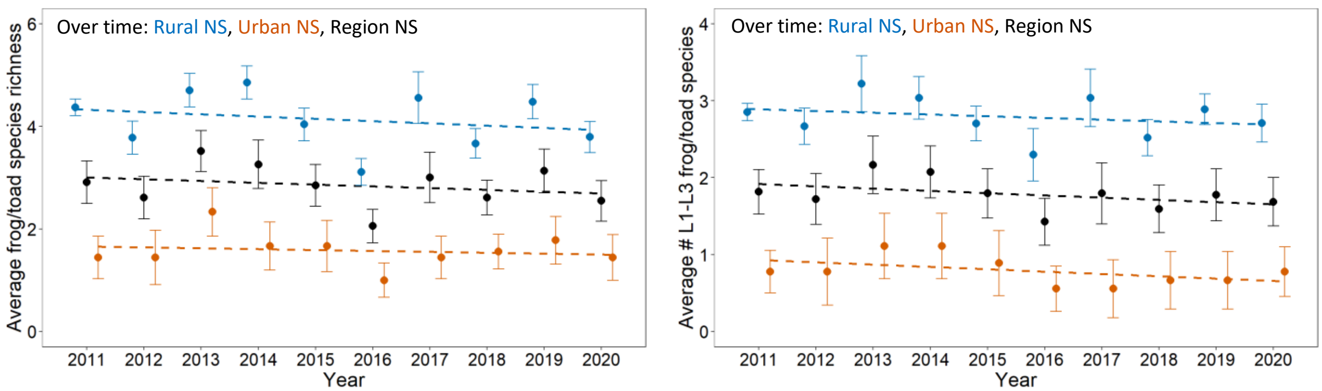


Figure 24. Temporal trends in frog high/level indicators a) number of frog species, and b) number of L1-L3 species. Trends are shown for the region and urban/rural sites. An arrow pointing up (↑) represents a statistically significant increasing trend while an arrow pointing down (↓) represents a statistically significant decreasing trend. NS (non-significant) means there was no statistically significant trend.

Percent of sites occupied by species and urban/rural differences

In 2020, 79% of sites were occupied by at least one frog species. The percent of sites occupied ranged from 78% to 95% between 2011 and 2020. The percent of sites occupied by specific frog species was relatively stable between 2011 and 2020 for the majority of species (Figure 25). There was no significant difference in the percent of sites occupied between urban and rural sites for American Toad, Green Frog, and Northern Leopard Frog (Figure 26). A significantly higher percent of sites were occupied in rural areas compared to urban areas for Spring Peeper, Tetraploid Grey Treefrog, and Wood Frog. All urban sites (9/9) and 56% (5/6) of rural sites detected no species during at least one visit between 2011 and 2020.

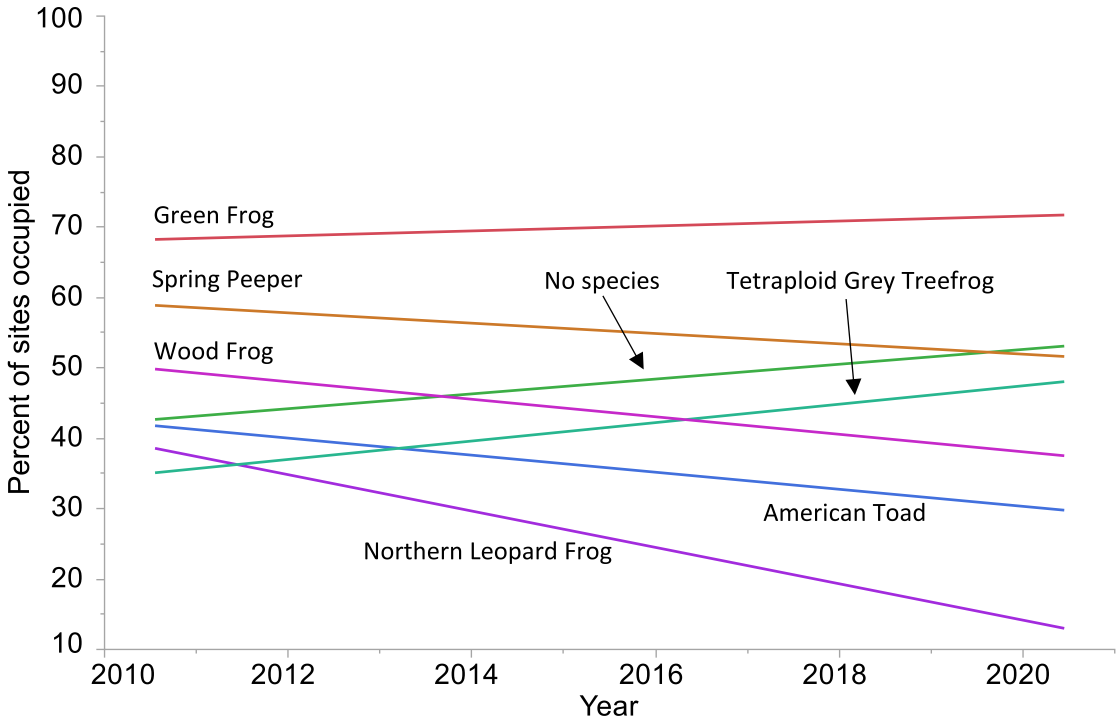


Figure 25. Temporal trends in the percent of sites occupied by specific frog species. "No species" indicates the percent of sites that the field surveyor recorded no species present.

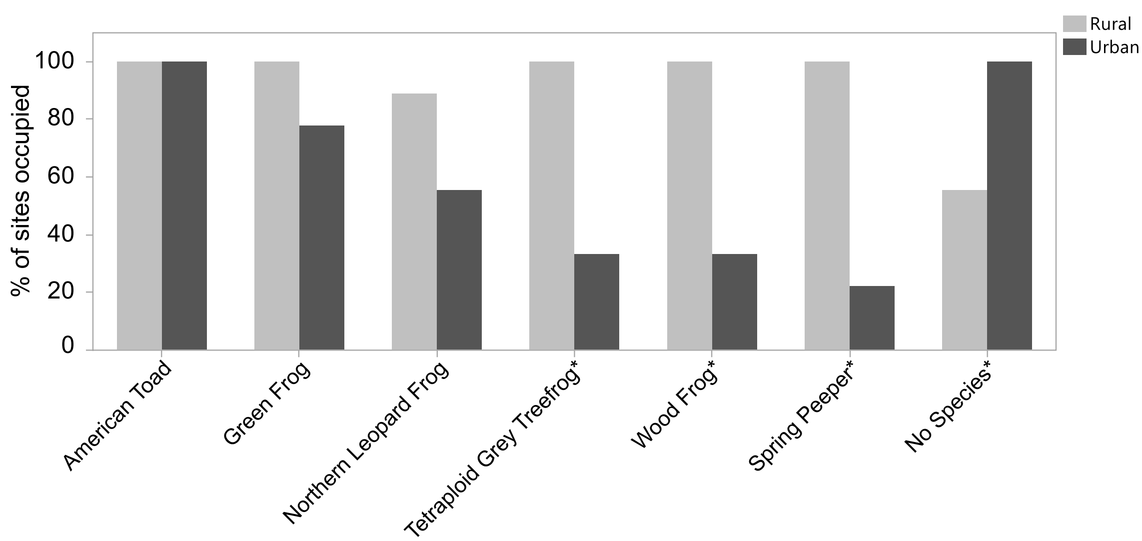


Figure 26. Percent of sites occupied by species at urban and rural sites. Significant differences between urban and rural are indicated with an asterisk (*).

Meadow birds

There was no significant change in meadow-dependent bird abundance, richness, or the number of L1-L3 meadow-dependent birds between 2011 and 2020 (Figure 27). In the last terrestrial LTMP report (TRCA 2015) and more recently in Cartwright et al. (2021), there were significant declines in these indicators. This difference is due to including data from a different time period in this report (2011-2020) compared to TRCA (2015) (2008-2014) and Cartwright et al. (2021). This suggests that the majority of the decline occurred between 2008 and 2011; however, meadow-dependent bird abundance and richness continues to decline at primarily two stations: ORMCP (MB-8) and Glen Major (MB-14).

Meadow bird indicators did not vary significantly between urban and rural stations although both the abundance and richness of L1-L3 ranked meadow bird species appeared to be lower in urban areas (approaching significance).

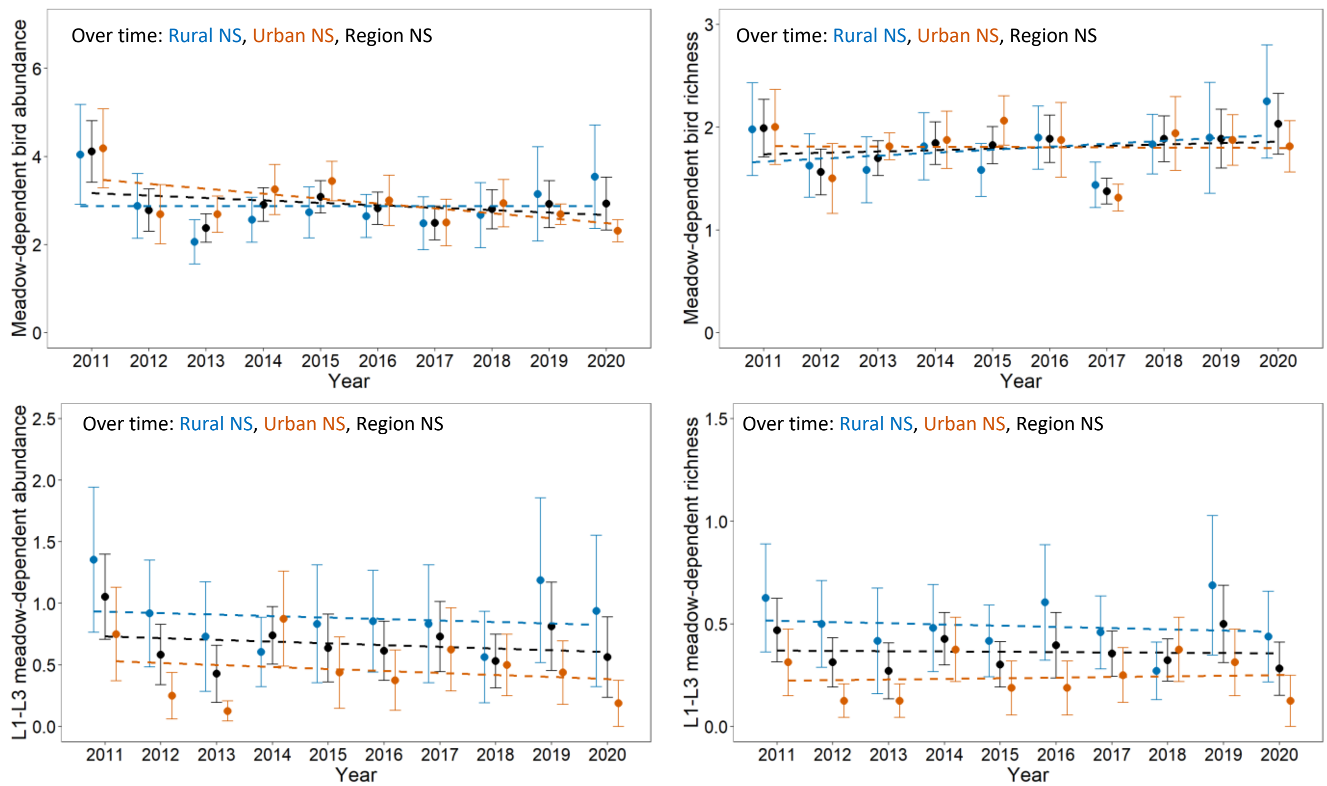


Figure 27. Spatial patterns and temporal trends in meadow bird indicators between 2011 and 2020. An arrow pointing up (↑) represents a statistically significant increasing trend while an arrow pointing down (↓) represents a statistically significant decreasing trend. NS (non-significant) means there was no statistically significant trend.

Meadow bird community composition varied between stations in the urban and rural zones (Figure 28). These species represent a complete list of the meadow-dependent species observed (not only the top ten). Species composition was similar between the rural and urban zone although there was variation in abundance.

Grasshopper Sparrow (*Ammodramus savannarum*) was only observed in the rural zone and Spotted Sandpiper (*Actitis macularius*) was only observed in the urban zone.

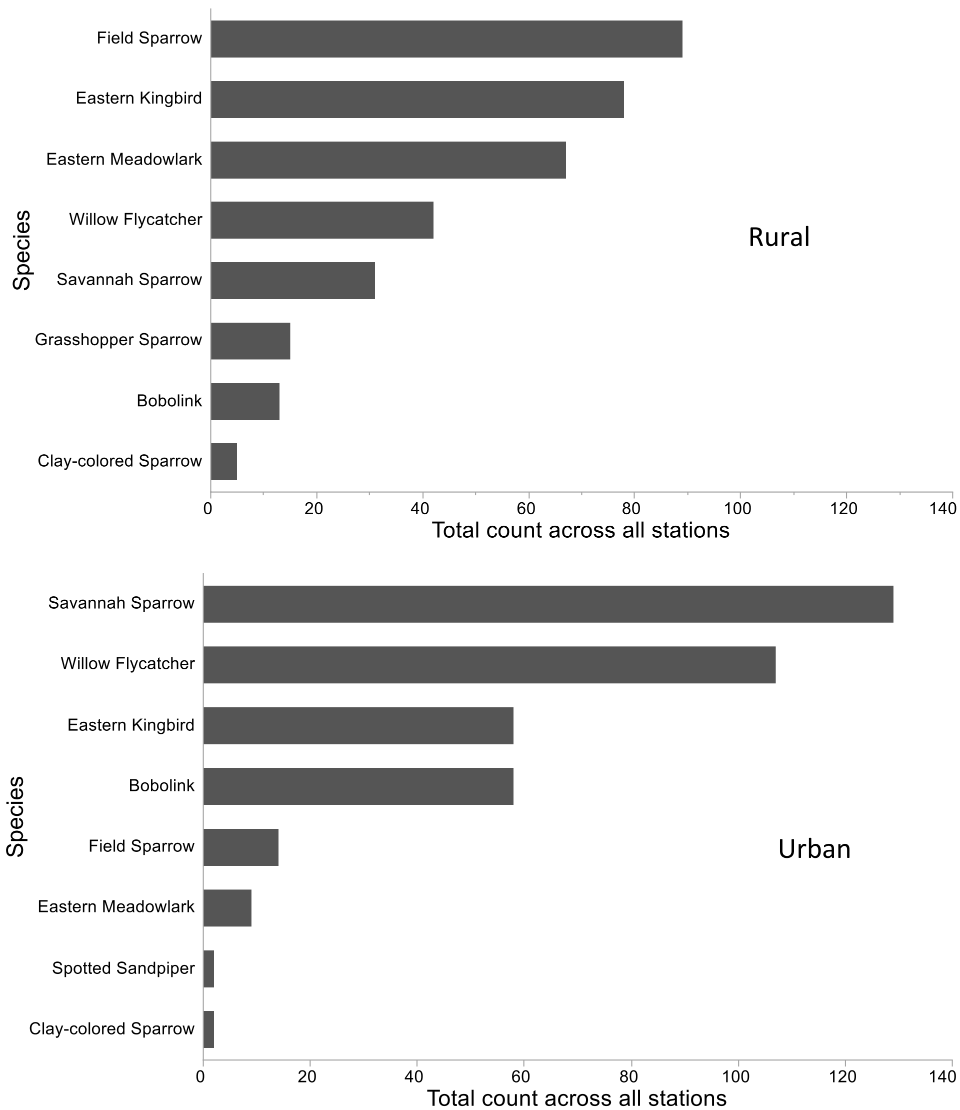


Figure 28. Meadow bird community composition in the rural and urban zone.

DISCUSSION

TRCA's Regional Watershed Monitoring Program was developed to track changes in the health of ecosystems and biodiversity across the region. This report summarized data on forests, wetlands, and meadows collected over the past 10 years. Overall, we observed temporal changes in forest, wetland, and meadow ecosystems along with the continued impact of urbanization on native species and biodiversity.

Forest vegetation changed significantly over the past 10 years. The FQI increased significantly although these changes appear to be caused largely by an increase in native species richness in forest plots. The largest increase in species richness occurred between 2017 and 2018 and the cause for this increasing pattern remains unknown although could be partially linked to EAB (more discussion below). Even though the FQI increased (again largely due to changes in species richness), the mean CC score decreased significantly both regionally and in the rural zone. This suggests that native plant communities are shifting from more conservative, sensitive species to less sensitive species. The percent of forest species that are native declined significantly regionally and in the rural zone suggesting that exotic species are becoming more prominent within the plots.

Forest vegetation indicators strongly demonstrate the impact of urbanization with plots in the urban zone containing fewer species and species of conservation concern, a lower percentage of native species, and lower FQI and mean CC scores than stations in the rural zone.

Forest vegetation monitoring reflects the impact of regional disturbances that have affected forest ecosystems over the past 10 years. These include both the December 2013 ice storm and EAB moving through the jurisdiction causing the most dieback between 2016 and 2017. These large-scale disturbances led to changes in community composition, increased tree mortality, snag production, decreased crown vigour, increased regeneration of woody species, and an increased production of ash seedlings. While several of these changes can be related to either the ice storm or EAB, some of these changes might be additive, although several distinctions can be made.

We observed decreases in crown vigour for several species occurring in 2014, and these changes are likely related to damage from the December 2013 ice storm. Tree species vary in their susceptibility to ice storm damage based on bark type, branching pattern, crown structure, and surface area of lateral branches (Hauer et al. 1994). Species or individual trees with an increased surface area of lateral branches, broad or imbalanced crowns, decaying or dead branches or inset bark tend to be more susceptible. Species such as bur oak and black cherry were especially affected in forest plots since these species have an intermediate or high susceptibility to storm damage (Hauer et al. 1994). Many species, when grown in the open, form broad crowns increasing susceptibility. Trees growing at forest edges are also more susceptible due to imbalanced crowns while those in the interior tend to have fewer, lower lateral branches decreasing susceptibility. This suggests that maintaining, protecting, and restoring large intact forests will help increase the resilience of forest ecosystems to the predicted impacts of climate change.

There was an increase in the total stem count of woody species regenerating in forest plots starting in 2015 and reaching a peak in 2016. This was largely due to increased sugar maple recruitment in plots particularly in the urban zone although a similar increase was found in rural plots although to a lesser extent. For example, at Heart Lake (an urban plot), the number of sugar maple stems in the regeneration layer increased from 235 in 2014, to 1604 in 2015, to 2402 in 2016. This plot had severe damage from the ice storm and the open canopy likely facilitated germination. In addition, several other species increased in stem count with a similar pattern although to a lesser extent including shagbark hickory (*Carya ovata*), wild black current (*Ribes americanum*), red ash (*Fraxinus pennsylvancia*), and black maple (*Acer saccharum ssp. nigrum*).

Community composition changed in forest plots due to the death of ash trees related to EAB with both red ash and white ash decreasing in relative abundance. Ash tree deaths started in late 2015 and continued until 2019 and these deaths led to an increase in the number of ash snags. In addition to deaths, we also noticed a significant increase in ash recruitment in plots. The stem count of white ash in the regeneration layer increased from 43 to 158 in rural plots and from 34 to 101 in urban plots between 2011 and 2020. This increase could indicate a stress response by adult trees where large amounts of seed are produced when under periods of extreme stress. This large seed crop helps to ensure gene transfer and persistence of the species. These results are similar to those of Kashian (2016) who also found that new seedling establishment reached high levels over the period of EAB infestation.

EAB is considered to be one of the most devastating invasive pests to be introduced to North America in recent years (Cappaert et al. 2005). It has caused nearly 100% mortality of ash species in the forests closest to its introduction and has continued to disperse (Burr and McCullough 2014). It is difficult to predict if the seed bank is sufficient to support continued seedling establishment as mortality of mature trees increases or plateaus although research suggests that ash may be able to persist, despite high mortality, due to its physiology and capacity to regenerate (e.g. natural tendency for sprouting and stable levels of seed production and seedling establishment) (Kashian 2016, Klooster et al. 2018).

The impacts of EAB on forest ecosystems also indirectly affected forest bird communities. Forest birds can be grouped into various guilds based on their foraging ecology (Graaf et al. 1985). Bark-foraging insectivores, such as woodpeckers and nuthatches, are those that feed on insects on or in the bark of trees. The significant increase in the total number of bark foragers across all stations likely reflects an increased abundance of EAB as a food source. These findings are consistent with those of Flower et al. (2014) that studied various relationships between bark foraging birds and EAB. Flower et al. (2014) found increased foraging on ash trees compared to non-ash trees along with a preference to forage on ash trees demonstrating canopy decline. Also, foraging significantly reduced EAB densities of up to 85% and intensity of foraging increased with EAB infestation levels. These results suggest that bark-foraging birds may be helping to regulate EAB populations within regional forests.

Other changes in forests include a significant decrease in both the abundance and cover of choke cherry across the region in the regeneration layer, an increase in the relative percent cover of exotic species in the ground vegetation layer, and severe defoliation at one plot due to Spongy Moth.

The decreasing cover of choke cherry is likely due to black knot fungus (*Apiosporina morbosa*), which is a common disease affecting *Prunus* spp. An increase in the occurrence or extent of the fungus can be influenced by environmental conditions including wet conditions and lower soil temperatures such as moist spring conditions over successive wet years (Stewart and Weber 1984).

The increasing cover of exotic species in the ground vegetation layer is a concern. Increases appear to be related again to either the ice storm or EAB with increasing covers starting in 2015, peaking in 2016, and decreasing thereafter although not returning to pre-2015 levels. Increased light levels due to gap formation are likely facilitating increased cover of exotic species (Hoven et al. 2017, Klooster et al. 2018).

The number of trees affected by Spongy Moth increased between 2017 and 2020. Spongy Moth infestation generally follows a cyclical pattern although periods of non-cyclicality have also been identified (Allstadt et al. 2013). Canopy cover decreased in response to defoliation and the extent of defoliation and infestation was particularly high at several sites including TVM 14 (in the upper reaches of Etobicoke Creek watershed), Palgrave forest, South Kirby and East of Keele, Portage Trail (located in the City of Toronto in the Humber watershed), and Heart Lake. Cyclical outbreaks of Spongy Moth typically last 4-5 years or 8-10 years and further monitoring will help to identify the end of the current cycle.

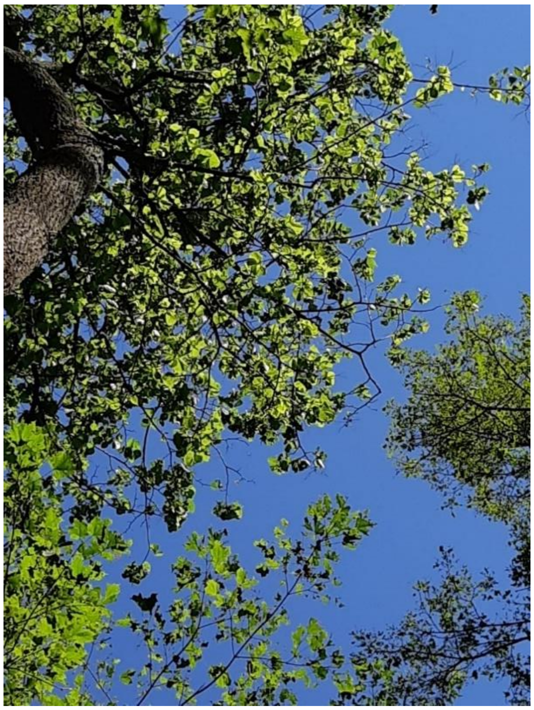


Figure 29. Palgrave forest in July 2020. All live basswood trees had a second flush of leaves since the first was entirely consumed.

Forest bird communities were generally stable over time across the jurisdiction; however, there were increases in abundance and richness of forest birds. These changes appear to be driven by 2019 and 2020, two years with particularly high numbers, with some variation in earlier years. Variation (higher or lower values) over time appeared to be greater at urban sites compared to rural sites. These general patterns could be related to EAB and the general increase in food availability (Long 2013).

Forest bird communities also indicated the strong impact of urbanization with a lower abundance of forest-dependent birds, species, and species of regional conservation concern in urban plots. 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, Calvert et al. 2013). 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, Remes et al. 2012). 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, Goodwin and Shriver 2011).

Similar to forest vegetation plots, the FQI for wetland vegetation transects increased over time. Again, this was a result of an increase in the number of native species over time although there was no change in the CC score. The reasons for this remain unknown although could be related to various factors such as changes in water quality, species composition, or water levels.

Common buckthorn, an invasive species, had the highest relative abundance of woody species in wetland vegetation transects and stem counts appear to be increasing. Common buckthorn has many traits that make it an effective invasive species including shade tolerance, high growth and photosynthetic rates, allelopathy, and tolerance for a wide range of environmental conditions (Knight et al. 2007). It also has the potential to completely displace native, woody wetland species, such as red osier dogwood and speckled alder, due to these properties.

Wetland vegetation communities in urban areas had lower FQI scores, mean CC scores, fewer species of regional concern, fewer native species, and lower species richness. 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 increased abundance of wetland birds could be related to wetland water levels since both 2017 and 2019 were very wet years. Wetland bird abundance has been correlated to lake water levels and these wetter years may also affect water levels in inland wetlands (Timmermans et al. 2008, Baschuk et al. 2012).

Wetland bird indicators strongly demonstrated the impact of urbanization with stations in the urban zone containing fewer wetland-dependent individuals and species than stations in the rural zone. These results are consistent with 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.

Frog communities did not change over time although did vary based on land use surrounding the station. A significantly higher percent of sites were occupied in rural areas compared to urban areas for Spring Peeper, Tetraploid Grey Treefrog, and Wood Frog suggesting that these species are less likely to occur in urban areas. 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 detrimental for amphibian species that need adjacent habitat for feeding or overwintering (Semlitsch and Bodie 2003).

Grassland birds have had the largest decline in total population size since 1970 compared to birds breeding in other habitats, with an estimated loss of more than 700 million individuals across North America (Rosenberg et al. 2019). In the last terrestrial LTMP report (TRCA 2015) and more recently in Cartwright et al. (2021), there were significant declines in meadow bird indicators while this report did not detect regional changes. This difference is due to including data from a different time period in this report (2011-2020) compared to TRCA (2015) (2008-2014) and Cartwright et al. (2021). This suggests that the majority of the decline occurred between 2008 and 2011. Even though declines were not detected in the regional dataset, meadow-dependent bird abundance and richness continues to decline at primarily two stations: ORMCP (MB-8) and Glen Major (MB-14).

The landscape surrounding the ORMCP stations has been developed and tree planting has occurred, and the plantings are maturing. Also, the vegetation at these stations appears to have changed from grasses to more of a wet area and could be due to run-off/drainage from the new residential development. Extensive reforestation efforts have occurred at Glen Major with plantings maturing as well. These changes in habitat suggest that these stations may no longer be suitable as meadow habitat and as such should be removed from the terrestrial LTMP meadow bird monitoring site roster. Nonetheless, the conversion of meadow habitat to early successional scrub/shrubland provides an opportunity to examine changes in bird species communities as these sites undergo natural succession and it may be beneficial to continue to monitor these stations to understand long-term changes in bird communities as they mature and measuring the performance of forest restoration. For example, at Glen Major there have been declines in meadow-dependent species such as Field Sparrow (*Spizella pusilla*), Savannah Sparrow (*Passerculus sandwichensis*), and Eastern Kingbird (*Tyrannus tyrannus*) while several new L3-ranked species have been observed only post-2012 including Magnolia Warbler (*Setophaga magnolia*), Blue-winged Warbler (*Vermivora cyanoptera*), Mourning Warbler (*Geothlypis philadelphia*), Nashville Warbler (*Oreothlypis ruficapilla*), Chestnut-sided Warbler (*Setophaga pensylvanica*), and Eastern Towhee (*Piplio erythrophthalmus*) all of which use early successional, second-growth habitats. Changes similar to these may also be contributing to increases in the number of L1-L3-ranked species at meadow bird stations regionally and in the rural zone as succession occurs across the landscape.

In summary, broad-scale disturbance events such as EAB and the December 2013 ice storm caused multiple changes in forest ecosystems affecting forest structure, species composition, and had wider impacts on food webs. Wetland and meadow ecosystems appeared to be relatively stable over time although several changes were apparent related to invasive species and changes in habitat availability affecting species composition. Species communities in all habitat types (forest, wetland, and meadow) continue to reflect the strong, negative effects of urbanization on these ecosystems. These results provide a greater understanding of regional (and local) factors affecting terrestrial biodiversity across the jurisdiction and can be used to predict changes likely to occur in the future as a result of climate change (new pests and increased frequency of ice storms), or inform invasive species management, forest management plans, and watershed planning.

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APPENDIX

Appendix 1. 2017 land cover layer codes and corresponding land use category for classifying stations as urban or rural.

Land use code (2017 layer)	Land use category
Beach/Bluff	Natural
Conservation Lands	Natural
Forest	Natural
Lacustrine	Natural
Meadow	Natural
Riverine	Natural
Successional Forest	Natural
Water	Natural
Wetland	Natural
Aggregate Extraction	Rural
Agricultural	Rural
Estate Residential	Rural
Farm	Rural
Golf Course	Rural
Hydro Corridor	Rural
Open Space	Rural
Park	Rural
Recreational	Rural
Recreational/Open Space	Rural
Rural Residential	Rural
Vacant Land	Rural
Airport	Urban
Cemetery	Urban
Commercial	Urban
High Density Residential	Urban
Industrial	Urban
Institutional	Urban
Landfill	Urban
Medium Density Residential	Urban
Mixed Commercial Entertainment	Urban
Railway	Urban
Residential High	Urban
Residential LowMed	Urban
Road (ROW)	Urban
Roads	Urban
Transportation	Urban

Appendix 2. Bird species and related nesting guilds.

Common Name	Guild
Acadian Flycatcher	forest-dependent
American Redstart	forest-dependent
American Woodcock	forest-dependent
Barred Owl	forest-dependent
Black-and-white Warbler	forest-dependent
Black-billed Cuckoo	forest-dependent
Blue-grey Gnatcatcher	forest-dependent
Blue-headed Vireo	forest-dependent
Blackburnian Warbler	forest-dependent
Brown Creeper	forest-dependent
Brown Thrasher	forest-dependent
Black-throated Blue Warbler	forest-dependent
Black-throated Green Warbler	forest-dependent
Broad-winged Hawk	forest-dependent
Blue-winged Warbler	forest-dependent
Canada Warbler	forest-dependent
Cerulean Warbler	forest-dependent
Cooper's Hawk	forest-dependent
Chestnut-sided Warbler	forest-dependent
Downy Woodpecker	forest-dependent
Eastern Bluebird	forest-dependent
Eastern Screech-Owl	forest-dependent
Eastern Towhee	forest-dependent
Eastern Wood-Pewee	forest-dependent
Whip-poor-will	forest-dependent
Great-crested Flycatcher	forest-dependent
Golden-crowned Kinglet	forest-dependent
Golden-winged Warbler	forest-dependent
Hairy Woodpecker	forest-dependent
Hermit Thrush	forest-dependent
Hooded Warbler	forest-dependent
Indigo Bunting	forest-dependent
Least Flycatcher	forest-dependent
Long-eared Owl	forest-dependent
Magnolia Warbler	forest-dependent
Merlin	forest-dependent
Mourning Warbler	forest-dependent
Nashville Warbler	forest-dependent
Northern Goshawk	forest-dependent
Northern Waterthrush	forest-dependent

Common Name	**Guild**
Northern Saw-whet Owl	forest-dependent
Olive-sided Flycatcher	forest-dependent
Ovenbird	forest-dependent
Pine Siskin	forest-dependent
Pine Warbler	forest-dependent
Pileated Woodpecker	forest-dependent
Prothonotary Warbler	forest-dependent
Purple Finch	forest-dependent
Rose-breasted Grosbeak	forest-dependent
Red-breasted Nuthatch	forest-dependent
Red-bellied Woodpecker	forest-dependent
Ruby-crowned Kinglet	forest-dependent
Red-eyed Vireo	forest-dependent
Red-headed Woodpecker	forest-dependent
Ring-necked Pheasant	forest-dependent
Red-shouldered Hawk	forest-dependent
Ruby-throated Hummingbird	forest-dependent
Ruffed Grouse	forest-dependent
Scarlet Tanager	forest-dependent
Sharp-shinned Hawk	forest-dependent
Veery	forest-dependent
White-breasted Nuthatch	forest-dependent
Worm-eating Warbler	forest-dependent
Wild Turkey	forest-dependent
Winter Wren	forest-dependent
Wood Duck	forest-dependent
Wood Thrush	forest-dependent
White-throated Sparrow	forest-dependent
White-winged Crossbill	forest-dependent
Yellow-breasted Chat	forest-dependent
Yellow-billed Cuckoo	forest-dependent
Yellow-bellied Sapsucker	forest-dependent
Yellow-rumped Warbler	forest-dependent
Yellow-throated Vireo	forest-dependent
American Crow	generalist
American Goldfinch	generalist
American Kestrel	generalist
American Robin	generalist
Baltimore Oriole	generalist
Barn Swallow	generalist

Common Name	Guild
Black-capped Chickadee	generalist
Blue Jay	generalist
Carolina Wren	generalist
Cedar Waxwing	generalist
Chipping Sparrow	generalist
Chimney Swift	generalist
Cliff Swallow	generalist
Common Grackle	generalist
Common Nighthawk	generalist
Eastern Phoebe	generalist
European Starling	generalist
Great-horned Owl	generalist
Grey Catbird	generalist
House Finch	generalist
House Sparrow	generalist
House Wren	generalist
Killdeer	generalist
Mourning Dove	generalist
Northern Cardinal	generalist
Northern Flicker	generalist
Northern Mockingbird	generalist
Orchard Oriole	generalist
Peregrine Falcon	generalist
Red-tailed Hawk	generalist
Red-winged Blackbird	generalist
Song Sparrow	generalist
Tree Swallow	generalist
Warbling Vireo	generalist
Yellow Warbler	generalist
Rock Dove	generalist
Bobolink	meadow-dependent
Clay-coloured Sparrow	meadow-dependent
Eastern Kingbird	meadow-dependent
Eastern Meadowlark	meadow-dependent
Field Sparrow	meadow-dependent
Grasshopper Sparrow	meadow-dependent
Henslow's Sparrow	meadow-dependent
Horned Lark	meadow-dependent
Loggerhead Shrike	meadow-dependent
Northern Harrier	meadow-dependent

Common Name	**Guild**
Savannah Sparrow	meadow-dependent
Short-eared Owl	meadow-dependent
Sedge Wren	meadow-dependent
Spotted Sandpiper	meadow-dependent
Upland Sandpiper	meadow-dependent
Vesper Sparrow	meadow-dependent
Western Meadowlark	meadow-dependent
Willow Flycatcher	meadow-dependent
Bank Swallow	special case
Belted Kingfisher	special case
Brown-headed Cowbird	special case
Northern Rough-winged Swallow	special case
Purple Martin	special case
Turkey Vulture	special case
American Black Duck	wetland-dependent
Green-winged Teal	wetland-dependent
Alder Flycatcher	wetland-dependent
American Bittern	wetland-dependent
American Coot	wetland-dependent
Black Tern	wetland-dependent
Blue-winged Teal	wetland-dependent
Canada Goose	wetland-dependent
Canvasback	wetland-dependent
Caspian Tern	wetland-dependent
Common Tern	wetland-dependent
Common Yellowthroat	wetland-dependent
Double-crested Cormorant	wetland-dependent
Gadwall	wetland-dependent
Great Black-backed Gull	wetland-dependent
Great Blue Heron	wetland-dependent
Great Egret	wetland-dependent
Green Heron	wetland-dependent
Herring Gull	wetland-dependent
Hooded Merganser	wetland-dependent
Least Bittern	wetland-dependent
Mallard	wetland-dependent
Marsh Wren	wetland-dependent
Mute Swan	wetland-dependent
Osprey	wetland-dependent
Pied-billed Grebe	wetland-dependent

Common Name	Guild
Ring-billed Gull	wetland-dependent
Redhead	wetland-dependent
Sora	wetland-dependent
Swamp Sparrow	wetland-dependent
Trumpeter Swan	wetland-dependent
Virginia Rail	wetland-dependent
Wilson's Snipe	wetland-dependent
Common Moorhen	wetland-dependent
Black-crowned Night Heron	wetland-dependent

Appendix 3. Terrestrial Long-term Monitoring Program plots used for analyses 2011-2020.

Forest vegetation

Site	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

Forest birds

Site	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

Wetland vegetation

Site	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-20	Bolton Resource Mgmt Tract	Rural
WV-22	Wildwood Park	Urban
WV-25	Bolton Tract South	Rural
WV-26	Bob Hunter Park	Urban

Wetland birds

Site	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-14	Greenwood	Rural
WB-16	Palgrave	Rural
WB-17	Albright	Rural
WB-20	Bolton Resource Mgmt Tract	Rural
WB-21	South Queen St East of Main	Urban
WB-22	Wildwood Park	Urban
WB-23	Old Church Roads Land	Urban
WB-24	Stouffville Reservoir	Urban
WB-25	Bolton Tract South	Rural

Frogs

Site	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-14	Greenwood	Rural
WF-16	Palgrave	Rural
WF-17	Albright	Rural
WF-20	Bolton Tract West	Rural
WF-21	South Queen St and East of Main	Urban
WF-22	Wildwood Park	Urban
WF-24	Stouffville Reservoir	Urban
WF-25	Bolton Tract South	Rural

Meadow birds

Site	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

Appendix 4. Mann-Kendall statistics.

Plot type	Indicator	Land use	z	S	tau	p
FV	FQI	Rural	2.68	31	0.689	0.007
FV	FQI	Urban	2.86	33	0.733	0.004
FV	FQI	Region	2.68	31	0.689	0.007
FV	mean cc	Rural	-2.68	-31	-0.689	0.007
FV	mean cc	Urban	-0.716	-9	-0.2	0.474
FV	mean cc	Region	-2.86	-33	-0.733	0.004
FV	# L1-L3 spp	Rural	0.634	8	0.184	0.526
FV	# L1-L3 spp	Urban	2.11	23	0.58	0.035
FV	# L1-L3 spp	Region	0.909	11	0.256	0.363
FV	% L1-L3 spp	Rural	0	1	0.022	1
FV	% L1-L3 spp	Urban	0.358	5	0.111	0.721
FV	% L1-L3 spp	Region	0.179	3	0.067	0.858
FV	% native spp	Rural	-2.5	-29	-0.644	0.012
FV	% native spp	Urban	-1.79	-21	-0.467	0.074
FV	% native spp	Region	-2.5	-29	-0.644	0.012
FV	spp rich	Rural	3.4	39	0.867	<0.001
FV	spp rich	Urban	3.22	37	0.822	0.001
FV	spp rich	Region	3.22	37	0.822	0.001
WV	FQI	Rural	2.5	29	0.644	0.012
WV	FQI	Urban	2.5	29	0.644	0.012
WV	FQI	Region	2.5	29	0.644	0.012
WV	mean cc	Rural	-0.537	-7	-0.156	0.592
WV	mean cc	Urban	0.716	9	0.2	0.474
WV	mean cc	Region	0.894	11	0.244	0.371
WV	# L1-L3 spp	Rural	2.45	28	0.644	0.014
WV	# L1-L3 spp	Urban	1.82	21	0.489	0.068
WV	# L1-L3 spp	Region	2.78	32	0.719	0.005
WV	% L1-L3 spp	Rural	-1.61	-19	-0.422	0.107
WV	% L1-L3 spp	Urban	1.61	19	0.422	0.107
WV	% L1-L3 spp	Region	0	1	0.022	1
WV	% native spp	Rural	0.09	2	0.045	0.928
WV	% native spp	Urban	-1.71	-20	0.45	0.088
WV	% native spp	Region	1.61	19	0.422	0.107
WV	spp rich	Rural	2.86	33	0.733	0.004
WV	spp rich	Urban	2.42	28	0.629	0.015
WV	spp rich	Region	2.68	31	0.689	0.007
FB	forest-dependent bird abundance	Rural	1.97	23	0.511	0.049
FB	forest-dependent bird abundance	Urban	2.68	31	0.689	0.007
FB	forest-dependent bird abundance	Region	2.68	31	0.689	0.007

FB	forest-dependent bird richness	Rural	1.79	21	0.467	0.074
FB	forest-dependent bird richness	Urban	2.15	25	0.556	0.032
FB	forest-dependent bird richness	Region	1.79	21	0.467	0.074
FB	L1-L3 forest-dependent bird abundance	Rural	0.716	9	0.2	0.474
FB	L1-L3 forest-dependent bird abundance	Urban	0.894	11	0.244	0.371
FB	L1-L3 forest-dependent bird abundance	Region	1.07	13	0.289	0.283
FB	L1-L3 forest-dependent bird richness	Rural	0.894	11	0.244	0.371
FB	L1-L3 forest-dependent bird richness	Urban	1.07	13	0.289	0.283
FB	L1-L3 forest-dependent bird richness	Region	1.07	13	0.289	0.283
FB	number of L1-L3 bird species	Rural	0.894	11	0.244	0.371
FB	number of L1-L3 bird species	Urban	1.07	13	0.289	0.283
FB	number of L1-L3 bird species	Region	1.07	13	0.289	0.283
FB	number of L1-L4 bird species	Rural	1.79	21	0.467	0.074
FB	number of L1-L4 bird species	Urban	2.33	27	0.6	0.02
FB	number of L1-L4 bird species	Region	1.97	23	0.511	0.049
FB	total count of bark-foragers across all stations	Region	2.33	27	0.6	0.02
WB	wetland-dependent bird abundance	Rural	2.33	27	0.6	0.02
WB	wetland-dependent bird abundance	Urban	0.272	4	0.092	0.785
WB	wetland-dependent bird abundance	Region	2.25	26	0.584	0.025
WB	wetland-dependent bird richness	Rural	-0.988	-12	-0.27	0.323
WB	wetland-dependent bird richness	Urban	-0.824	-10	-0.236	0.41
WB	wetland-dependent bird richness	Region	-0.988	-12	-0.27	0.323
WB	L1-L3 wetland-dependent bird abundance	Rural	1.35	16	0.36	0.178
WB	L1-L3 wetland-dependent bird abundance	Urban	-1.56	-18	-0.424	0.12
WB	L1-L3 wetland-dependent bird abundance	Region	1.17	14	0.315	0.243
WB	L1-L3 wetland-dependent bird richness	Rural	-1.07	-13	-0.289	0.283
WB	L1-L3 wetland-dependent bird richness	Urban	-2.11	-24	-0.566	0.035
WB	L1-L3 wetland-dependent bird richness	Region	-1.89	-22	-0.494	0.059
WB	number of L1-L3 bird species	Rural	-0.808	-10	-0.225	0.419
WB	number of L1-L3 bird species	Urban	-0.64	-8	-0.189	0.522
WB	number of L1-L3 bird species	Region	-1.35	-16	-0.36	0.178
WB	number of L1-L4 bird species	Rural	-1.79	-21	-0.467	0.074
WB	number of L1-L4 bird species	Urban	-1.64	-19	-0.442	0.101
WB	number of L1-L4 bird species	Region	-1.97	-23	-0.511	0.049
MB	meadow-dependent bird abundance	Rural	0.358	5	0.111	0.721
MB	meadow-dependent bird abundance	Urban	-1.73	-20	-0.46	0.085
MB	meadow-dependent bird abundance	Region	0.179	3	0.067	0.858
MB	meadow-dependent bird richness	Rural	0.721	9	0.205	0.471
MB	meadow-dependent bird richness	Urban	-0.182	-3	-0.07	0.855
MB	meadow-dependent bird richness	Region	1.18	14	0.322	0.238
MB	L1-L3 meadow-dependent bird abundance	Rural	-0.09	-2	-0.045	0.928

MB	L1-L3 meadow-dependent bird abundance	Urban	-0.449	-6	-0.135	0.653
MB	L1-L3 meadow-dependent bird abundance	Region	-0.358	-5	-0.111	0.721
MB	L1-L3 meadow-dependent bird richness	Rural	-0.629	-8	-0.18	0.53
MB	L1-L3 meadow-dependent bird richness	Urban	0.368	5	0.119	0.713
MB	L1-L3 meadow-dependent bird richness	Region	-0.179	-3	-0.067	0.858
FV	total number of woody stems (shrub sapling regen)	Rural	-0.537	-7	-0.156	0.592
FV	total number of woody stems (shrub sapling regen)	Urban	1.43	17	0.378	0.152
FV	total number of woody stems (shrub sapling regen)	Region	1.61	19	0.422	0.107
FV	choke cherry relative abundance (shrub sapling regen)	Rural	-1.79	-21	-0.467	0.074
FV	choke cherry relative abundance (shrub sapling regen)	Urban	-3.4	-39	-0.867	<0.001
FV	choke cherry relative abundance (shrub sapling regen)	Region	-3.58	-41	-0.911	<0.001
FV	choke cherry relative cover (shrub sapling regen)	Rural	-2.5	-29	-0.644	0.012
FV	choke cherry relative cover (shrub sapling regen)	Urban	-3.04	-35	-0.778	0.002
FV	choke cherry relative cover (shrub sapling regen)	Region	-3.22	-37	-0.822	0.001
FV	all exotics cover (ground vegetation)	Rural	1.43	17	0.378	0.152
FV	all exotics cover (ground vegetation)	Urban	1.97	23	0.511	0.049
FV	all exotics cover (ground vegetation)	Region	2.5	29	0.644	0.012
FV	garlic mustard cover (ground vegetation)	Rural	-1.97	-23	-0.511	0.049
FV	garlic mustard cover (ground vegetation)	Urban	1.61	19	0.422	0.107
FV	garlic mustard cover (ground vegetation)	Region	1.61	19	0.422	0.107
FV	common buckthorn cover (shrub sapling regen)	Rural	2.32	27	0.6	0.02
FV	common buckthorn cover (shrub sapling regen)	Urban	1.25	15	0.333	0.211
FV	common buckthorn cover (shrub sapling regen)	Region	2.15	25	0.555	0.032
FV	common buckthorn stems (shrub sapling regen)	Rural	2.86	33	0.733	<0.01
FV	common buckthorn stems (shrub sapling regen)	Urban	1.25	15	0.333	0.211
FV	common buckthorn stems (shrub sapling regen)	Region	1.43	17	0.378	0.152
FV	tree mortality	Region	1.26	13	0.366	0.21
FV	crown vigour (healthy)	Region	-1.35	-16	-0.36	0.178
FV	crown vigour (lightmoderate)	Region	1.35	16	0.36	0.178

FV	crown vigour (severe)	Region	2.07	17	0.615	0.038
WF	frog species richness	Rural	-0.537	-7	-0.156	0.592
WF	frog species richness	Urban	0	0	0	1
WF	frog species richness	Region	-0.808	-10	-0.225	0.419
WF	# L1-L3 frog species	Rural	-0.449	-6	-0.135	0.653
WF	# L1-L3 frog species	Urban	-0.919	-11	-0.263	0.358
WF	# L1-L3 frog species	Region	-1.53	-18	-0.405	0.127
WF	American Toad	Region	-0.368	-5	-0.119	0.713
WF	Green Frog	Region	0.091	2	0.046	0.928
WF	No species	Region	1.26	14	0.358	0.21
WF	Northern Leopard Frog	Region	-1.01	-12	-0.283	0.314
WF	Spring Peeper	Region	-1.34	-13	-0.404	0.179
WF	Tetraploid Grey Treefrog	Region	1.01	12	0.283	0.314
WF	Wood Frog	Region	-1.35	-15	-0.389	0.177

Appendix 5. T-tests, median tests, Fisher's Exact test

Plot type	Variable	Test statistic	p	Land use	Mean	SE	N
FV	FQI	-2.708	0.013	Rural	29.02	1.68	11
				Urban	22.85	1.54	13
FV	meancc	-2.295	0.032	Rural	4.58	0.13	11
				Urban	4.18	0.12	13
FV	# L1-L3 species	-2.016	0.056	Rural	3.48	0.72	11
				Urban	1.5	0.666	13
FV	% L1-L3 species	-1.945	0.065	Rural	6.86	1.27	11
				Urban	3.5	1.17	13
FV	% native species	-4.24	<0.001	Rural	86.7	2.31	11
				Urban	73.4	2.12	13
FV	species richness	-1.03	0.313	Rural	46.96	3.87	11
				Urban	41.53	3.56	13
FV	total # stems	0.995	0.331	Rural	54.3	17.4	11
				Urban	115.1	53.4	13
FV	choke cherry abund	0.283	0.777	Rural	3.13	2.01	11
				Urban	11.89	7.39	13
FV	choke cherry cover	0.283	0.777	Rural	2	1.2	11
				Urban	7.82	4.83	13
FV	all exotics cover	3.54	<0.01	Rural	6.91	2.36	11
				Urban	24.5	5.18	13
FV	garlic mustard % cover	2.4	0.0254	Rural	0.404	2.68	11
				Urban	9.14	2.47	13
FV	buckthorn % cover	-2	0.045	Rural	4.86	3.37	11
				Urban	9.81	7.2	13
FV	buckthorn abund	-2	0.045	Rural	6.04	4.43	11
				Urban	11.44	5.97	13
FB	Forest dependent richness	-6.65	<0.0001	Rural	5.48	0.233	14
				Urban	3.33	0.225	15
FB	Forest dependent abund	-5.89	<0.0001	Rural	8.03	0.431	14
				Urban	4.5	0.417	15
FB	For dep # individuals	-5.08	<0.0001	Rural	2.74	0.317	14
				Urban	0.499	0.306	15
FB	For dep # species	-6.04	<0.0001	Rural	2.06	0.195	14
				Urban	0.427	0.188	15
WV	FQI	-1.91	0.072	Rural	24.6	2.38	11
				Urban	17.8	2.63	9
WV	mean cc	-1.83	0.085	Rural	4.19	0.22	11
				Urban	3.59	0.24	9
WV	# L1-L3 spp	-3.11	<0.01	Rural	6.6	1.64	11

				Urban	2.41	1.14	9
WV	% L1-L3 species	-2.94	<0.01	Rural	14.7	2.26	11
				Urban	4.78	2.5	9
WV	% native spp	-2.73	0.014	Rural	79	4.25	11
				Urban	61.7	4.7	9
WV	species richness	-0.992	0.334	Rural	45.7	5.27	11
				Urban	37.9	5.83	9
WB	wetland depdenent bird abundance	-1.47	0.16	Rural	4.34	0.775	9
				Urban	2.77	0.735	10
WB	wetland dependent bird richness	-1.63	0.122	Rural	2.32	0.307	9
				Urban	1.63	0.291	10
WB	L1-L3 wet dep abund	-2.03	0.059	Rural	1.6	0.358	9
				Urban	0.6	0.34	10
WB	L1-L3 wet dep rich	-1.86	0.081	Rural	0.859	0.144	9
				Urban	0.49	0.137	10
WF	species richness	-6.07	<0.0001	Rural	4.14	0.298	9
				Urban	1.58	0.298	9
WF	# L1-L3 species	5.41	<0.0001	Rural	2.8	0.262	9
				Urban	0.789	0.262	9
MB	meadow dependent abundance	0.127	0.901	Rural	2.88	0.522	8
				Urban	2.97	0.522	8
MB	meadow dependent bird richness	0.057	0.955	Rural	1.79	0.206	8
				Urban	1.81	0.206	8
MB	mead dep L1-L3 abund	-0.75	0.466	Rural	0.211	0.071	8
				Urban	0.136	0.071	8
MB	mead dep L1-L3 richness	-1.21	0.245	Rural	0.49	0.147	8
				Urban	0.238	0.147	8

Species	Presence/absence	Rural	Urban	Fisher's Exact 2-tailed p-value	Right-tailed
American Toad	Present	9	9	1	
	Absent	0	0		
Green Frog	Present	9	7	0.471	
	Absent	0	2		
No Species	Present	5	9	0.082	0.041
	Absent	4	0		
Northern Leopard Frog	Present	8	5	0.294	
	Absent	1	4		
Spring Peeper	Present	9	2	0.002	
	Absent	0	7		
Tetraploid Grey Treefrog	Present	9	3	0.009	
	Absent	0	6		
Wood Frog	Present	9	3	0.009	
	Absent	0	6		

Right-tailed test result used in one case meaning probability > for land use=urban than rural.

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