Effects of Local Rural Land Use on Forest Habitats in Southern Ontario

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Foreword

The main focus of my plan of study is the various ways in which human land use interacts with pre-existing ecosystems, both positive and negative, and how to manage both the land use activities themselves and adjust conservation efforts and techniques as needed. A large part of this plan of study involved familiarization with the various ecosystems found throughout Canada and the world, and the various conservation challenges and primary stressors associated with them. This major paper specifically dealt with land use impacts in Canadian temperate forest, and offered an opportunity to observe how compositionally similar tracts of forest differed in the presence of various types of land use. It required me to become strongly familiar with the species and taxonomic groups found in forest environments, identify patterns among their composition, and link them to the various stressors associated with the trial disturbances, both physical and chemical. It also provided an opportunity to improve my research, data management, experimental design and scientific communication skills, which formed a component of my plan of study. In short, this paper directly and indirectly assisted with the completion of most of my plan of study's learning objectives, and represents a major advancement in my personal knowledge of the subject.

Abstract

Forest ecosystems are under severe threat from resource exploitation, fragmentation and disturbance. While the rate of forest loss has slowed in recent years, it is still ongoing, and what remains is increasingly degraded as human development continues. This study investigates some effects of human rural land use on adjacent forest habitats, measured by examining soil function and biodiversity/species composition. Biodiversity was surveyed on four occasions for each taxonomic group over the summer, using quadrat and transect sampling for plants, sweep net and pan trap sampling for insects, and unlimited-distance point counts for birds. Soil function was assessed by determining the rate of decomposition of leaf litter of five major tree species, and by investigating soil chemical content. There was evidence of nutrient loading near agricultural sites, which exhibited very high phosphorus, potassium, calcium and nitrogen relative to control. The agricultural sites also had the highest degree of leaf litter mass loss. Agricultural sites contained the highest incidence of invasive species, and the lowest insect and plant diversity. The trailside site contained similar plant diversity to the roadside site, but exhibited the lowest bird diversity. Bird diversity was highest at the control site, but also high at agricultural sites. It would appear the agriculture has the greatest effects on plant and insect life, and roads and trails have the greatest impact on bird communities, largely because the primary sources of disturbance from agriculture are chemical in nature, while the disturbance from trails and roads are based on noise and direct disturbance. But these conclusions are based on a limited sample. Overall, land use has significant effects on adjacent ecosystems, presenting difficult questions for ecological restoration in rural and urban environments.

Introduction

As the human population grows and the extent of human land use expands, terrestrial ecosystems find themselves under increasing encroachment and modification (Cebellos 2015). In Canada, forests are a particular concern, due to their wide historical coverage. As of 2008, 22% of the world's old growth forest remains intact and undisturbed (Hansen 2008), "intact" forest being defined as covering a contiguous area of 500 km^2 and having no visible signs of human development nearby (Potapov 2008). While the primary concern regarding this increasing forest modification is the issue of habitat loss and fragmentation, habitat degradation by direct and indirect disturbance also demands attention.

Forests are extremely important to both global biodiversity and human interests. Canada's temperate forests are home to a large portion of its native wildlife, many of whom associate with trees in some way. Forty-three species of arthropod depend solely on ash trees (*Fraxinus spp*), and another 30 generally associate with them (Gandhi 2010), to say nothing of nesting birds or plants growing in the shade. For humans, forests represent a valuable commodity in the form of lumber, as well as providing a number of extremely important ecosystem services. They represent a major carbon sink (Prescott 2010), reduce soil erosion (Sayer 2005), and present a positive psychological effect to both humans and animals (Seidl 2016). However, as forests become increasingly fragmented or degraded, the ecosystem services provided become less effective, and the wildlife that depends on them finds itself under greater and greater stress (Haddad 2015, Magrach 2014). In some cases, species may avoid edge habitat entirely, which in severely fragmented areas can reduce effective available habitat to zero (Fonderflick 2012).

In rural environments, a large portion of this disturbance to forests comes from agriculture, roads and recreational trails. These land use types represent strong sources of localized disturbance than can impact nearby ecosystems in various ways. Some impacts, such as chemical runoff, punctuated noise pollution and alteration of drainage regimes, are common to all three, albeit on different scales (Coffin 2007, Ware 2015, Ballantyne 2015, Godefroid 2004, Miller 1998, Foufoula-Georgiou 2015). Others are relatively unique to the disturbance type: road dust can clog plant gas exchange organs (Jones 2015) and cars can provide a vector for invasive plant dispersal (Pickering 2010). Trails can cause localized soil compaction and alter plant community composition through a composition of plant trampling and accidental invasive species dispersal (Ballantyne 2015). Excess fertilizers from agriculture contaminates groundwater, causes nutrient loading in the soil and in extreme cases, can lead to freshwater eutrophication (Sharpley 1996). Another potential concern of agriculture is the effect of pesticides on non-target insects, including pollinating species; varying sublethal effects by agricultural pesticides on a wide range of non-target arthropods has been observed (Pisa 2015).

Because of the highly visible nature of these stressors on local biodiversity, this study will largely focus on insects, herbaceous plants and birds. All provide their own services to the ecosystem, which interact and overlap across the taxonomic groups, including pollination, seed dispersal and pest predation. An imbalance in one of these groups can impact the services provided by the other two, reducing overall ecological productivity (Isbell 2011). Thus it is important to assess taxon-specific biodiversity as well as overall diversity. Of the ways of quantifying biodiversity in use, the most common is the Shannon-Wiener index, which measures both species richness and community evenness (Spellerberg 2003). In addition to measuring diversity within a community, a rough approximation of stress can be estimated by examining the community composition itself. In plants, a reduced amount of slow-growing plants versus known invasive species can indicate physical or chemical stress (Brundrett 1990, Gilliam 2015), while a reduced amount of interior forest specialist songbirds can indicate noise disturbance and fragmentation effects (Freemark 1992). Likewise, carabid beetles and spiders are considered a useful indicator of general forest health (Pearce 2015).

In addition to the diversity of visible species, there has been growing concern regarding the health and activity of soil microfauna. Due to their role in decomposing lignin, a major connective tissue in plants, they play a critical part of forest nutrient cycling (Berg 2000), and can be severely impacted by local disturbance (Hartmann 2012). Also of great importance is arbuscular mycorrhizal fungi (AMF), the hyphae of which penetrates plant roots, aiding in nutrient uptake through symbiotic relationships. This group of fungi is also sensitive to physical disturbance (Schneider 2015), as well as chemical disturbance via the toxic allelochemicals secreted by some invasive plant species (Gilliam 2015). Unlike macrofauna, however, it is difficult to reliably measure the diversity of soil microbes: some groups will outcompete others depending on the nutritional content, and the general practice utilizes ribosomal RNA analysis (Fierer 2007). Quantification of AMF is similarly involved (Sharma 2015). However, the rate of leaf litter decomposition has proven to be a good indicator for soil microbial activity (Mosseau 2014), and may provide a useful tool for roughly assessing it.

To that end, this study will investigate how the disturbance effects of rural agriculture, roads and recreational trails manifest in forest fragments bordering these land use types in southern Ontario. It will perform a comparative assessment of insect, herbaceous plant and bird biodiversity observed in these fragments, as well as a brief analysis of their respective community composition, and will also measure soil activity by measuring the rate of leaf litter decomposition. It is expected that the nutrient loading caused by agriculture would manifest in a higher rate of leaf litter mass loss at sites bordering agriculture of any scale, with higher mass loss near higher-intensity agriculture. The plants growing near agricultural sites are expected to be more vigorous, defined in this context by a higher number of plants per quadrat, but possibly with lower diversity. Absolute Plant abundance and diversity, as well as bird abundance diversity, is expected to be lpwer at trailside and roadside sites. There is expected to be a higher frequency of invasive species at trailside and roadside sites. Bird diversity is expected to be lower than the control at all trial sites. Insect communities at agricultural sites are expected to be composed of less pollinators and more parasitic wasps or pest species.

Methods

Site Descriptions

Five sites were selected in the York Region/Bradford area, based on an initial criteria of mixed deciduous forest immediately adjacent to one of the chosen stressors (Figure 1). The types of disturbance assessed were commercial agriculture at small and large scales, high-traffic dirt roads, and recreational trail use. The control site was located in a large stretch of old-growth forest in the Koffler Scientific Reserve. The trial sites were located, on average, approximately 20 metres into the forest from the source of disturbance, in an attempt to minimize fragmentation edge effects while maintaining direct exposure to the stressor in question. The exception was the trailside site, which was located 5 metres away from the trails.

The control site, roadside site and trailside site were located within the Koffler Scientific Reserve, a stretch of forest owned by the University of Toronto. The forest represents one of the largest tracts of old-growth forest in the local municipal area, and is bordered by agriculture, low-density residences and disused pine plantation. It is effectively split in two by an unpaved section of Dufferin St. The control site is located within the western section, several hundred metres away from some informal trails used only for scientific research in the area, close to a small stream. The roadside site is located just within the eastern section, downslope from the road, near a small strip that was cleared for a buried power line two years before the study. The trailside site is located farther into the eastern section, well beyond where a stretch of pine plantation transitions to deciduous forest. The trail it borders is surfaced by sand and well maintained, with frequent foot traffic. While it lies within an on-leash area, this is poorly enforced, and dogs are frequently allowed to run freely.

The first agricultural site borders a forty acre farm that provides a wide variety of crops, including corn, pumpkins and lettuce, and often receives major traffic during educational field trips. A small stretch of old-growth forest is nearby and is adjacent to three small corn fields. The farm owner, while not organic, commits to using low amounts of pesticides. The second industrial farm site is located just south of Cook's Bay, and is adjacent to an industrial carrot and onion farm. Pesticide use is heavy enough that I was advised to avoid field edges for several days after spraying. The soil in this region is wetter than the other sites, and the study site borders a large swamp. While it exhibits several species characteristic of wetlands, such as marsh marigold and swamp bedstraw, it appears to be more of a transition zone between wetland and forest ecosystems.

Leaf Litter and Soil Chemistry

Soil activity was measured by depositing coarse-mesh bags containing 10 grams of dried leaf litter composed mostly of white birch, red pine, silver maple, red oak and trembling aspen at the sites. The leaf litter had all been collected the fall of 2015 from a forested area on my property and dried over the winter before being bagged in the spring of 2016. This followed Mosseau et al (2014), who utilized leaf litter decomposition as an indicator of soil microbial activity in irradiated environments. In normal environments, the primary predictors of leaf litter decomposition are chemical composition of the litter itself, macroclimate and presence of the plant species in question in the area (Cornwell 2008). These variables were controlled for by gathering leaf litter from a single location outside the study area and drawing all samples from the same pool, and by deliberately selecting forests with similar canopy composition. Coarse mesh was deliberately chosen to allow access to the leaves by decomposing arthropods. The bags were placed in on top of the soil substrate and covered with existing leaf litter, and allowed to decompose undisturbed for six months (May 13, 2016 to October 16, 2016). 186 bags were deposited in total, of which 181 were in recoverable condition at the end of the period (those that were unrecoverable appeared to have been torn open by raccoons). The bags were scattered across an area measuring approximately 25 m^2 at each site that was enclosed by rope to prevent human disturbance. Average sample size across sites was 36, and ranged from 34 to 39 (Table 1). After recovery, they were dried at 65ºC for 96 hours, and weighed to establish the amount of mass loss to decomposition.

Twenty soil cores were taken from each site to a depth of 25 cm at random locations within each site and sent to Guelph University for chemical analysis, assessing K, P, Mg and Ca concentrations, pH, % organic matter and cation exchange capacity. Trends from this soil analysis were compared to trends in leaf litter decomposition.

Biodiversity Assessment

A biodiversity assessment was performed over the summer of 2016, focusing on herbaceous plants, birds and insects. Plant diversity and abundance were sampled once a month from May to August around the middle of the month by semi-random placement of 2x2 metre quadrats (Pielou 1966). Twenty-nine quadrat samples were taken in total, averaging 7 quadrats placed per monthly visit. In the fourth month, an additional 20 metre transect was taken in a random direction, sampling plants 1 metre away from the transect.

Bird diversity was sampled once a month by twenty minute unlimited-radius point counts in the late morning, during which every bird that was heard was recorded in methods similar to the North American Breeding Bird Survey (Sauer 2013).

Insects were sampled by alternating treatments: months 1 and 3 sampled insects by overnight placement of red, yellow, blue and white pan traps. Months 2 and 4 sampled insects by sweep netting across a total of eight 20-meter transects. All insect specimens were preserved in ethanol and later identified to the family level. Trees were sampled by semi-random placement of five 20-meter transects on June 18, 2016, sampling every tree within a metre of the transect.

Data from across the four month sampling period were pooled and used to calculate overall (bird, tree and herbaceous plants) and taxon specific (bird, ground plant, insect familial diversity) Shannon-Weiner biodiversity indices and evenness (Spellerberg 2003). Data for birds, ground plants and trees was used to calculate Bray-Curtis dissimilarity indices (Clarke 2006) between the sites. Insect diversity data was not used for dissimilarity calculations or overall biodiversity due to insects being identified to the family instead of species level. A brief compositional analysis of plants was also performed, largely focusing on invasive species and sensitive, slow-growing plants, following Brundrett and Kendrick (1990). Compositional analysis of birds was based off classification as interior forest/edge specialists, generalists and tolerance to disturbance, following Freemark and Collins (1992).For each land use category, I compared the various measures between the control and treatment sites. For overall Bray-Curtis dissimilarity indexes, all study sites were compared to all other sites.

Fragmentation

QGIS was used to determine the relative degree of fragmentation present at the five sites. Polygon data on forest coverage was obtained from Land Information Ontario. The APr (area to perimeter ratio) of the forest fragment each site was contained in was then calculated by dividing perimeter in metres by area in square metres. The results were compared to benchmarks derived from APr of perfect circles with areas of 1 ha, 2.5 ha and 5 ha (Aurambout 2005).

Results

Leaf litter and soil analysis

Leaf litter mass loss at all trial sites showed some differences from mass loss at the control site (Table 1). The roadside site and both agricultural sites experienced higher mass loss, while the trailside site experienced lowered mass loss. Average mass loss at the small-scale agricultural site was nearly double that of the control; mass loss at the industrial agriculture site was more than double (Table 1).

Soil chemical analysis showed strong imbalances in mineral salts at agricultural sites relative to the control site, particularly in Mg and Ca (Table 2). The roadside site also showed an unusually high concentration of Ca ions. P was likewise unusually high at the trailside and small farm sites, but approximately equivalent at the industrial farm site. Both agricultural sites and the roadside site were much more basic than the control site, with pH values of 7.0-7.3 and 5.9 respectively.

Biodiversity and species composition

The control site was most compositionally similar to the trailside site based on the Bray-Curtis dissimilarity index, and most different from the industrial farm and small farm sites (Table 3). Both agricultural sites were most compositionally similar to each other. The roadside and trailside sites were strongly dissimilar to both the agricultural sites and each other, while less dissimilar with the control site. Plant community composition varied widely between the sites (Table 4); the greatest similarity to the control site was expressed by the small farm site, with a similarity of 31.4%. The road and trailside sites had bird communities far more similar to the control site than the agricultural sites.

In addition, the control site expressed the highest overall biodiversity and the highest biodiversity for birds and ground plants (Table 5). The lowest overall biodiversity was expressed by the trailside site (Table 4). The lowest bird diversity was observed at the roadside site; the lowest plant diversity and evenness was recorded at the industrial farm site. The industrial and small farm sites expressed the highest bird diversity after the control site. Trends in evenness tended to follow trends in biodiversity, with some exceptions: plant species evenness was much lower at the trial sites than they were at the control site. The trailside site was observed to have the lowest overall species evenness, and the industrial farm site the next lowest.

The overwhelming majority of insects captured in pan traps were flies (Diptera: Dolichopodidae, Muscidae, Empididae, Phoridae) and parasitic wasps (Hymenoptera: Ichneumonidae, Braconidae, Chalcidoidea). Insects caught by net were overwhelmingly mosquitos (Diptera: Culicidae), leafhoppers (Hemiptera: Cicadellidae), common spiders (Arachnida: Dictynidae) and a small volume of leaf-eating beetles (Coleoptera: Curculionidae, Coccinellidae, Carabidae). Few insects that were recognizably pollinators (e.g. Lepidoptera, Apidae, Syrphidae) were caught at any of the sites by either method (Table 6). The highest familial diversity and evenness were observed at the control site, and the lowest observed at the small farm site. Familial diversity and evenness at the industrial farm site, which yielded the largest insect abundance, was similar to that of the trailside site, which yielded the lowest.

Plant species composition varied widely between sites. Some species, such as Herb Robert (*Geranium robertanium*) were present at all sites. Others, like jewelweed (*Impatiens capensis*), white baneberry (*Actaea pachypoda*) and Canada wood nettle (*Laportea canadensis*) were found at all but one of the sites. The control and roadside sites exhibited a greater abundance of sensitive, slow-growing plant species with limited dispersal and intolerance to disturbance (Table 7), such as white trillium (*Trillium grandiflorum*), trout lily (*Erythronium americanum*) and Solomon's plume (*Smilacina racemosa*), while the trailside and agricultural sites had higher abundances of invasive species and disturbance tolerant plants such as garlic mustard (*Alliaria petiolata*), dandelion (*Taraxacum spp.*) and lesser burdock (*Arctium minus*).

Fragmentation

The small agriculture site appeared to be located within the most heavily fragmented area; the site was the largest forest patch in the immediate area and did not meet the APr threshold of a 1 ha circle (Figure 1). The control site's fragment fell just short of the APr threshold of a 5 ha circle, while the fragment containing the roadside and trailside site exceeded it. The industrial farm site's APr also fell slightly short of the 5 ha threshold, but it was closely bordered by a large forest fragment which greatly exceeded it.

Tables and Figures

Table 1 - Average leaf litter mass loss

	Control	**Road**	**Trail**	**Small**	**Industrial**
Mass Loss	2.502 g	3.007 g	1.979 g	4.853 g	5.983 g
STDEV	0.496	1.0649	0.7599	1.186	0.854
n	35	36	34	39	37

Table 2 - Soil chemical test results

	Control	**Road**	**Trail**	**Small**	**Industrial**
pH (BpH)	5.9 (6.4)	7.3	5.4 (6.0)	7.0	7.2
Organic %	5.4	6.7	5.7	8.0	33.4
P (ppm)	11	13	42	21	9
K (ppm)	58	54	38	96	48

Mg (ppm)	109	154	74	191	392
Ca (ppm)	1499	4307	850	3667	7725

Table 3 - Overall Bray-Curtis dissimilarity indices between sites

	Control	**Road**	**Trail**	**Small**	**Industrial**
Control		0.590	0.717	0.695	0.918
Road	0.590		0.911	0.522	0.605
Trail	0.717	0.911		0.862	0.930
Small	0.695	0.522	0.862		0.492
Industrial	0.918	0.605	0.930	0.492	

Table 4 - Taxon-specific Bray Curtis dissimilarity indices.

	Control	**Road**	**Trail**	**Small**	**Industrial**
Ground Plants		0.631	0.745	0.549	0.842
Birds		0.530	0.575	0.703	0.714

Table 5 - Overall Shannon biodiversity indices and evenness (insects not included)

	Control	**Road**	**Trail**	**Small Farm**	**Industrial**
Shannon	2.929	2.951	2.312	2.587	2.505
Evenness	0.962	0.766	0.562	0.622	0.609

Table 6 - Taxon-specific Shannon biodiversity indices and evenness

	Control	**Road**	**Trail**	**Small Farm**	**Industrial**
Birds	2.969	1.895	2.346	2.579	2.602
Evenness	0.911	0.862	0.889	0.952	0.938
Plants	2.286	2.164	2.251	2.173	2.149
Evenness	0.710	0.608	0.708	0.607	0.590
Insects	3.038	2.829	2.771	2.586	2.701
Evenness	0.869	0.816	0.756	0.727	0.779

Table 7 - Relative abundances of common insect groups/groups of interest

	Control	Road	Trail	Small Farm	Industrial
Ichneumonidae	5	28	66	50	17
Mymaridae	9	13	5	2	6
Apidae	4	1	1	0	2
Braconidae	1	2	3	5	0
Culicidae	21	19	32	20	66
Arachnida: Dictynidae	13	11	11	12	16
Dolichopodidae	7	13	29	8	12
Phoridae	10	12	33	26	20
Empididae	4	11	28	3	41
Muscidae	16	15	22	32	26
Cicadellidae	6	7	10	10	26
Formicidae	0	11	6	76	48
Syrphidae	3	5	1	1	0
Geometridae	2	0	4	3	0
Carabidae	3	1	0	0	0
Total	164	191	286	294	337

Table 8 - Relative abundances of common plant species/species of interest

	Control	Road	Trail	Small Farm	Industrial
White Trillium	145	31	16	23	0
Garlic Mustard	0	8	0	160	80
Lesser Burdock	0	1	0	6	7
Trout Lily	205	500	0	363	0
Solomon’s Plume	18	2	47	0	0

Canada Goldenrod	0	19	0	0	26
Herb Robert	37	60	10	31	6
Jewelweed	86		294	180	482
Canada Wood Nettle	17	46	1	23	0
White Baneberry	7	0	7	9	1
Wild Black Currant	0	10	11	11	6
Common Wood Sedge	13	4	6	13	27
Long-Awned Wood Grass	4	0	0	2	27
Canada Mayflower	70	38	106	29	0
Total	828	1647	302	979	973

Table 9 - Relative abundances of common bird species/species of interest

	Control	**Road**	**Trail**	**Small Farm**	**Industrial**
Black-Capped Chickadee	9	5	6	1	2
American Goldfinch	1	8	2	2	3
Red-Eyed Vireo	6	9	7	4	1
American Robin	3	0	0	0	5
Red-Winged Blackbird	0	0	0	3	6
Eastern Wood-Peewee	3	0	2	2	0
Black-Throated Green Warbler	4	2	1	0	1
Hermit Thrush	4	0	0	0	1
Ovenbird	2	2	1	1	3
Brown Creeper	2	0	0	0	0
Wood Thrush	1	0	0	0	0
Total	57	41	23	24	41

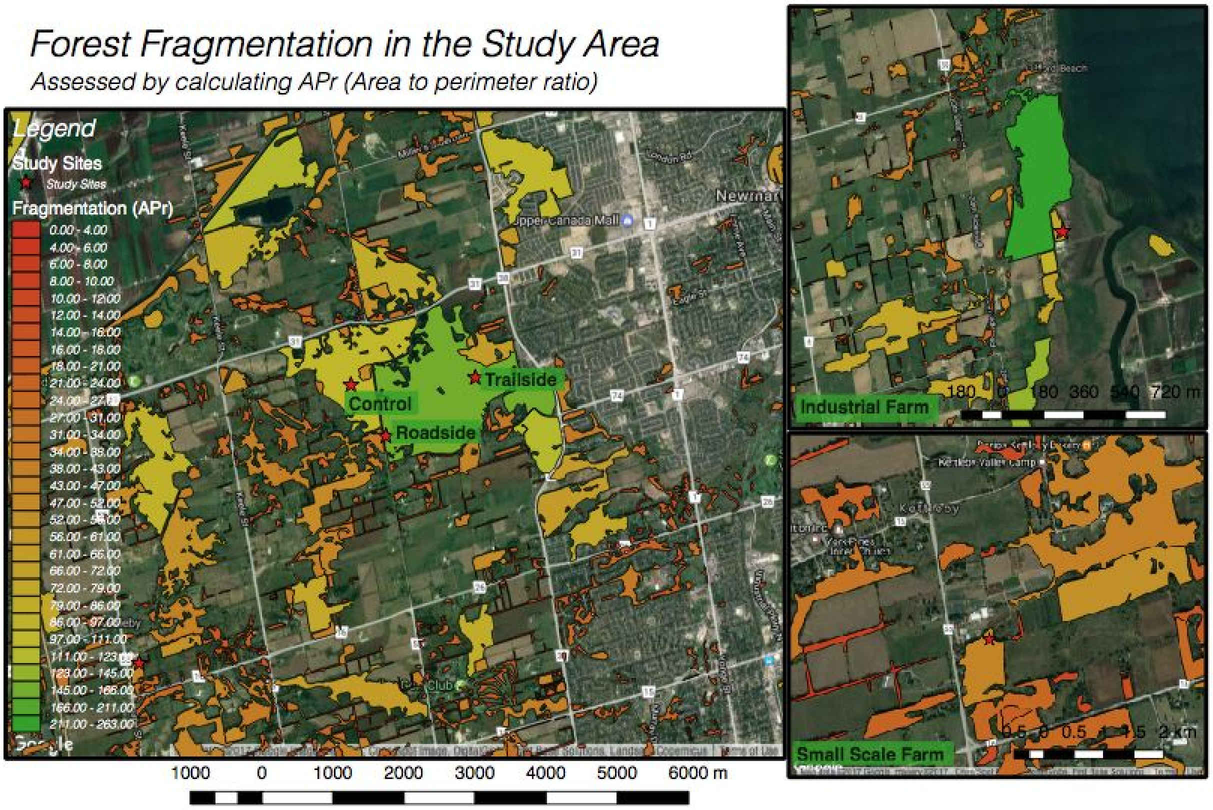

Figure 1 - Fragmentation analysis of forested areas surrounding study sites, performed by calculating APr (Area to perimeter ratio). Forest coverage shapefiles courtesy of Land Information Ontario. Satellite overlay from Google Maps

GPS Coordinates

Control – 44º2'7.85", -79º31'58.95"
Roadside – 44º1'44.63", -79º31'36.80"
Small Farm – 44º0'1.46", -79º43'12.58"
Industrial Farm – 44º12.8'45.68", -79º31'45.68"
Trailside – 44º2'11.35", -79º30'40.18"

Discussion

Leaf Litter Decomposition

In this study, leaf litter mass loss was much higher in agricultural environments than in control samples (Table 1), with a mean mass loss more than double that of the control. Given that the decomposition of leaf litter is not linear (Berg 2000), but instead follows an asymptotic curve, decomposition at the small and industrial farm sites is therefore many times faster than at the non-agricultural sites. In leaf decomposition, soluble minerals and carbohydrates tend to decompose very quickly, and is mostly dependent on local moisture (Donnelly 1990). The main limiting factor in leaf litter decomposition is the degradation of lignin and lignified carbohydrates, which comprise a large portion of the leaf mass. The primary predictors for the rate of lignin decomposition are generally the chemical composition of the leaf itself and the local plant community structure (Taylor 1989). Plant community structure varied quite drastically between the control and industrial farm sites (Table 3), but less so between the control sites and small farm sites. It appears unlikely that community composition is the sole driver of this large discrepancy in decomposition rates. It is therefore likely that nutrient loading, demonstrated by relatively high mineral salt concentrations observed in soil samples (Table 2), is a contributor to this increased rate of decomposition. Adding nitrogen to forest soil has been demonstrated to increase the rate of leaf litter humidification (Prescott 2010); while nitrogen content was not tested, % organic matter present in soil was much higher at the agricultural sites, especially the industrial site, than at the control site, indicating an increased rate of conversion to humus. Calcium loading from road salt runoff, manifesting in a calcium concentration nearly triple that observed at the control site (Table 2), was likely responsible in some degree for the slightly increased rate of decomposition also observed at the roadside site (Baribault 2010).

This increased decomposition rate would directly manifest as a much thinner leaf litter layer, which was observed upon initial visits to the agricultural sites. A robust leaf litter layer serves several important ecological roles in a forest, most notably in terms of nutrient cycling and storage: long-term experimental removal of leaf litter from forest sites depleted soil nitrogen and cations, and increased soil erosion in the process (Sayer 2005). Leaf litter decomposition into humus also functions as a carbon trap, leading to research into modifying decomposition rates to favour humus production over total decomposition (Prescott 2010). Leaf litter also plays a role in modifying the local plant community through a combination of manual blocking and shading; seedlings of shade-intolerant plants tend to have dramatically decreased success in areas with robust litter layers (Facelli 1991). The agricultural sites in this study contained a noticeably thinner leaf litter layer; the absence of this blocking effect appears to have manifested in the presence of greater abundances of grass and sedge than were observed at the other sites, where the litter layer was more robust. This effect on leaf litter decomposition by land use merits further study.

Plant Communities

The agricultural sites were noted to be home to large numbers of the invasive species Garlic Mustard, *Alliaria petiolata* (Table 8). This species is known to be a particularly aggressive invader of northern forest groundcover, and is capable of quickly reducing ecosystems to total monocultures. One of the adaptations that makes it so invasive is its ability to add toxic allechemicals into the soil, which disrupt the hyphal network of mycorrhizal symbionts and cause major nutrient stress to any other plant species in the area (Gilliam 2015). Despite the invasive success and high dispersive potential of this and other plant invaders, they are not commonly found in undisturbed, interior forest environments (Luken 2014). In a separate paper, Gilliam (2006) proposes that nitrogen loading of the soil, which is known to increase the leaching potential of mineral salts, reducing their bioavailability, can on its own represent a source of disturbance great enough to significantly increase the invasibility of a forest habitat. Evidence of nitrogen loading at the agricultural sites has already been demonstrated by the increased organic matter and mineral salt concentrations, meaning that forest flora bordering agricultural environments is very vulnerable to invasive species. This is demonstrated not only by the presence of Garlic Mustard, but also the presence of Dandelion and Lesser Burdock, non-native species commonly associated with disturbed habitat (Pickering 2010). While not considered actively invasive, the presence of even these plants in an area undisturbed by human activity save for agriculture is concerning.

In contrast, the control and roadside sites were home to large volumes of White Trillium, Trout Lily and Solomon's Plume, species which tend to disperse and mature slowly and are therefore more intolerant of physical disturbance (Brundrett 1990). While the small farm site did contain several stands of trillium, the farm owner claimed to have planted them years prior. The five sites did not differ much in plant diversity, but their composition varied widely. It is unclear how much of this variance is the result of The plants at the roadside site in particular were very commonly covered in a visible film of road dust, which likely contributed to stress and decreased vigor (Jones 2016). While the trailside site exhibited a similar level of biodiversity to the control site, there was a visible lack of plant cover close to the trail, which is likely the result of trampling (Ballantyne 2015). Plant cover increased noticeably in vigor a small distance from the trail. Far more invasive and disturbed-habitat species were observed at trailheads than at the study site, which was located nearly a kilometre along the trail; seeds from these species were likely tracked in accidentally by hikers (Pickering 2010). Human-mediated weed dispersal appears to decrease with distance along the trail, but the presence of invasive plant communities presents potential for further dispersal into interior forest.

Insect Communities

Insect diversity and evenness was lowest at the agricultural sites, although the difference was not as large as expected. Insect diversity at the industrial farm site being higher than at the small farm site was especially surprising, given the evidence for sublethal effects of industrial insecticides on non-target arthropods (Pisa 2015). While the non-agricultural sites did contain more insect families that were recognizably pollinators (Apidae, Syrphidae), the low abundance of pollinators captured at any of the sites makes conclusions difficult. Overall, the attempted capture of pollinating insects by pan traps was considered a failure in this study, due to the overwhelming dominance of egg-swollen flies and parasitic wasps caught in them. It is likely that the flies were simply attempting to lay eggs in what appeared to be stagnant water, and the wasps arrived to parasitize the flies. The main notable differences (Table 6) appear to be a higher quantity of ants (Formicidae) observed at the agricultural sites, a much higher incidence of parasitic wasps (Ichneumonidae, Mymaridae, Braconidae) observed at the trial sites, and a higher incidence of leafhoppers (Cicadellidae) observed at the industrial farm site. The possible implications of these differences are less clear; Pearce and Venier (2006) proposed that spiders and carabid beetles are useful indicator species for forest health at local scales, yet there is little difference in the abundance of the former and little recorded incidence of the latter at any of the sites. It is possible that the high abundance of parasitoid wasps at agricultural sites is a result of spillover from the surrounding farms, a phenomenon that can occur in fragmented forest habitats (Frost 2015), but this is difficult to confirm without species-level insect identification, which was beyond the scope of the study.

Bird Communities

Among the trial sites, bird diversity was lowest at the trailside and roadside sites, and highest at the agricultural sites. Absolute abundance of birds observed followed the same patterns. This fits into the established theory that birds are more sensitive to the physical and audio disturbance associated with roads and trails than they are to the chemical disturbance associated with agriculture (Miller 1998). Further impacting bird diversity at the roadside site is the potential for increased mortality by way of collisions with passing cars (Jack 2015). Unsurprisingly, the control site contained the highest bird diversity, richness and abundance, and contained several sensitive interior forest specialists (Table 8), including the brown creeper, black-throated green warbler and wood thrush. While insensitive generalists such as the black-capped chickadee and American goldfinch were also observed at the control site, they were observed with larger frequency at trial sites. Also observed at agricultural sites were edge specialists such as the red-winged blackbird and American crow, which were likely a result of the large open area taken up by the nearby fields. While it is unlikely that there would be antagonistic relationships between these two groups on the scale of the brown headed cowbird, the presence of open-field species could possibly present a minor source of competition for food in addition to existing stress from disturbance effects.

More concerning, however, is the apparent avoidance by birds of the trailside and roadside sites, which exhibited both the highest dominance by insensitive generalists and the lowest absolute bird abundance. Due to the nature of the sampling (auditory instead of visual or banding), it is possible that there was an abundance of birds in the area approaching that of the control site that was simply remaining silent. Birdsong is both a way of attracting a mate and establishing or defending territory, and birds are known to behave much less territorially in disturbed or suboptimal habitats (Fort 2004). In either case, reduced singing activity severely impacts the chances of finding a mate, and in cases where a mate is secured, increases the chances of extra-pair mating, reducing individual reproductive success and reducing overall population recruitment (Stutchbury 2007).

Fragmentation Effects

Fragmentation effects appeared to be most prominent at the roadside and agricultural sites (Figure 1). While recreational trails are a major contributor to local habitat fragmentation (Ballantyne 2014), the effects of trampling and soil compaction appeared to be more significant at the trailside site, evidenced by the apparent lack of characteristically open-habitat species. The control site was specifically chosen to minimize fragmentation effects. The small farm site, located in the most heavily fragmented area, predictably exhibited the fewest interior specialists, while also including the second-lowest overall biodiversity and the lowest overall evenness. It is difficult to draw a causative line between these two factors, as habitat fragmentation has unpredictable and chaotic interactions with biodiversity on the local scale, resulting in increases as often as decreases (Fahrig 2003). Despite the thinned leaf litter layer as a result of nutrient loading, it does not express the same conflict between forest and open-habitat plant species observed at the industrial farm site, where patches of goldenrod and grass appeared to be encroaching on other groundcover.

The roadside site exhibited more visible fragmentation effects. Despite being located in one of the more intact forest fragments covered by the study, it was exposed to the largest proportion of edge habitat by consequence of a nearby cleared strip that had been cut into the forest to bury a hydro line years before. Open-field species, including goldenrod (*Solidago spp.*) and *Phlox spp.*, while not within the study area, were observed nearby even in areas with dense leaf litter.

Limitations and Sources of Error

This study, while fairly comprehensive, suffers from several methodological caveats. The lack of site replication is a concern, and the author acknowledges that several control replicates at the least would have been more appropriate to establish a firm baseline. The placement of the agricultural sites is also somewhat problematic; after the field season, it was discovered that the small farm site was in a much more fragmented area than had originally been thought, raising questions as to how much of the disparity observed there was from land use versus fragmentation effects. The industrial farm site was also situated in a far more swamp-like section of forest than the other sites, causing a much higher incidence of marsh indicator species. Data for insects, plants and birds was not separated by season, and was instead lumped together; likewise, the relative abundance table did not account for discrepancies in total sample size between sites. An additional shortcoming was the study's inability to identify insects to the species level, which prevented inclusion in biodiversity and dissimilarity calculations. Finally, the bird point counts would have been better served operating on the Breeding Bird Survey's standard of three minutes (Sauer 1966) rather than the performed twenty, due to the risk of hearing the same bird multiple times and recording it as multiple individuals. Further study on the subject should take note of these drawbacks upon attempting replication.

Conclusions

Overall, it is difficult to rank the different types of rural land use studied as more or less degrading than others. Recreational trails have only local impacts on plant communities, but severely impact the perceived quality of the habitat for birds. Dirt roads negatively impact both plant and bird communities with a combination of physical, noise and chemical disturbance. Agriculture has potential to cause species overflow from open-field habitats and increases the invasibility of the forest by virtue of nutrient loading, which, while far more significant at industrial farm sites than smaller operations, is still significant at the latter.

The impacts of trails on local bird populations is particularly troubling, as trails are often considered an excellent medium for exposure to and education about nature. It is possible that this impact can be lessened by more intensive management of the trails in question, preventing one single trail from having a disruptive level of traffic. What level of foot traffic can be considered disruptive is, however, a subject for future study. A more feasible and immediate solution could be to attempt to better educate hikers to stay on the trail, refrain from disruptive behaviour, and to keep dogs on leash.

The other types of land use, however, are as necessary as they are difficult to rectify. Roads will continue to exist as long as people have places to go, and agriculture continues to be necessary to feed them. While ecological intensification (a process involving allowing a section of farmfield to grow wild) is a potential way to provide extra meadow habitat, it risks worsening the problem of invasibility and species spillover into forests.

Ecosystem restoration seeks to restore, repair and enhance degraded ecosystem services, but it always appears to operate under the assumption that those ecosystem services remain exploitable. Since even a land use type as innocuous as a trail has major consequences on the local ecosystem, it is impossible to say there is a kind of land use that does not degrade those services all over again. That is not to say ecosystem restoration is a wasted effort; it is often the only activity that brings greenery back to urban spaces. However, in rural spaces, it needs a change in focus. Spaces subject to restoration should not be quantified in terms of ecosystem services gained, available, or exploitable. It should not be quantified at all. Restored ecosystems in rural environments should be left alone, free from human disturbance. In economic terms, this is completely unsustainable and represents a net loss of resources. But in the age-old prisoner dilemma, one party must sacrifice for mutual gain. And if we must sacrifice the effort of restoration now for the sole benefit of an improved biosphere in the future, the so be it.

After all, nature needs its ecosystem services too.

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