Impacts of Anthropogenic Land Transformation on Habitat Amount, Fragmentation, and Connectivity in the Adirondack-to-Laurentians (A2L) Transboundary Wildlife Linkage: Implications for Conservation and Ecological Restoration

Jonathan Robert Cole

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This is to certify that the thesis

Prepared by: Jonathan Robert Cole

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complies with the regulations of the University and meets the accepted standards with respect to originality and quality. Signed by the final Examining Committee:

____________________________________Chair
Dr. Pedro Peres-Neto, Department of Biology, Concordia University

____________________________________Supervisor
Dr. Jochen A. G. Jaeger, Department of Geography, Planning & Environment, Concordia University

____________________________________Arms-Length Examiner
Dr. Robert Weladji, Department of Biology, Concordia University

____________________________________Examiner
Dr. Jeff Bowman, Department of Environmental & Life Sciences Graduate Program, Trent University/Ontario Ministry of Natural Resources and Forestry

____________________________________Examiner
Dr. Angela Kross, Department of Geography, Planning & Environment, Concordia University

____________________________________External examiner
Dr. Richard Pither, Landscape Ecology Research Section, Environment and Climate Change Canda

Approved by ____________________________________Graduate Program Director
Dr. Pascale Biron, Department of Geography, Planning & Environment

____________________________________Dean of Faculty
Dr. Pascale Sicotte, Faculty of Arts and Science

Date: Wednesday December 4th, 2024

Abstract

Impacts of anthropogenic land transformation on habitat amount, fragmentation, and connectivity in the Adirondack-to-Laurentians (A2L) transboundary wildlife linkage: Implications for conservation and ecological restoration

Jonathan R. Cole Ph.D.

Concordia University, 2024

Habitat loss and fragmentation, due to anthropogenic land transformation, is the leading cause of species declines and biodiversity loss worldwide. Habitat loss and fragmentation transform landscapes into a heterogeneous array of habitat fragments of smaller total habitat area, isolated from each other by a human-dominated matrix. This results in long-term changes in ecosystem structure and function, and an overall reduction in species abundance and movement ability between fragments. Globally, 56% of all terrestrial mammals have transboundary geographic ranges. In contrast, most conservation initiatives do not cross political boundaries. The Adirondack-to-Laurentians (A2L) transboundary wildlife linkage connects wilderness areas in the northeastern United States with southeastern Canada. Although the region contains many habitats of high ecological integrity and biodiversity; ceaseless anthropogenic land transformation within the A2L may be putting transboundary connectivity at risk. Changes in landscape structure, due to anthropogenic land transformation, that occurred within the A2L between 1992 and 2018 were quantified, and priority areas for conservation and restoration were identified. The results suggest that to achieve long-term functionality of the A2L, collaborative and coordinated measures will be necessary to preserve the integrity of the Québec portion, restore extensive habitat in eastern Ontario, and reestablish or maintain connectivity throughout

the linkage. The results can be used to inform conservation policy and land-use planning throughout the region. Left unaddressed, continued anthropogenic land transformation is likely to have additional detrimental effects on the ability of the A2L to function as a transboundary wildlife linkage.

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Author Contributions

I am the first author on all manuscript chapters. I led all aspects of each project including: the conception, study design, data collection, statistical analyses, interpretation of the results, and the writing of the manuscripts.

Chapter 2: This manuscript was co-authored with Angela Kross, and Jochen A. G. Jaeger. Jochen A. G. Jaeger contributed to the study conception and design. Jochen A. G. Jaeger and Angela Kross contributed to the methodology. All authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

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1. Introduction

Anthropogenic Land Transformation

The Earth is now experiencing what scientists are calling "the Anthropocene", a period where human activities are having a significant influence on all of Earth's vital systems (WWF, 2020). The human population is expected to reach 9.7 billion by 2050 (United Nations, 2019). Wild mammalian species, of which there are over 5000, only constitute 4% of the global mammalian biomass, while humans now make up 36%, and livestock 60% (Bar-On et al., 2018). This phenomenon is generating an increasing demand for land and resources. Over 50% of the Earth's terrestrial surface has now been altered by human activities (Hooke et al., 2012; Riggio et al., 2020).

Biodiversity, the diversity within species, between species, and between ecosystems, and the benefits they provide, are fundamental to human well-being and a healthy planet (IPBES, 2019). Yet human activities are causing biodiversity to decline faster than at any time in human history (Díaz et al., 2019). This rapid loss of species is estimated to be between 1,000 and 10,000 times higher than the average rate over the past 10 million years (Ceballos et al., 2015; De Vos et al., 2015). This rate is only expected to increase in the face of climate change (Urban, 2015). Globally, over 45,000 species are now assessed as threatened (IUCN, 2024), and it is estimated that around one million species face extinction, many within decades, unless considerable conservation intervention is taken to halt the drivers of biodiversity loss (IPBES, 2019). The direct drivers of this mass extinction (Barnosky et al., 2011) include changes in land- and sea-use (i.e. habitat loss and degradation), direct exploitation of organisms, climate change, pollution, invasion of alien species, and disease (IPBES, 2019; WWF, 2022).

The area of Earth's terrestrial surface devoted to cropland occupies >15 million km^2 (Foley et al., 2011) and is expected to expand 18% by 2050 (Tilman, 1999); and the area committed to urban development is predicted to triple to 1.8 million km^2 by 2030 (Seto et al., 2012). Deforestation over the past half century has resulted in the loss of more than a third of all forest cover worldwide (Williams, 2003; Hansen et al., 2013). Between 1993 and 2009, 3.3 million km^2 of terrestrial wilderness was lost globally to urban development, agriculture, forestry, mining, and other human modifications; and between 2000 and 2015, 1.3 million km^2 of native forests were lost (Watson et al, 2016). As of 2016, only 23.2 percent (30.1 million km^2) of the world's terrestrial areas met the definition of being wilderness (i.e., biologically and ecologically largely intact landscapes that are mostly free of human disturbance) and these areas are scattered around the globe; with the largest portions located in Russia, Canada, and Brazil (Watson et al., 2016; Pardini et al., 2017).

Canadian wilderness areas are not immune to land conversion. Since 2000, almost 5% (216,199 km^2) of Canada's intact forest landscapes (i.e., >500 km^2 in size and untouched by roads or other significant human activity) were fragmented by human activities (CBI, 2024). Québec, Alberta, Ontario, and British Columbia accounted for 71% of these human disturbances (CBI, 2024). Wetland ecosystems are also being transformed in North America and around the globe. Canada alone hosts 25% (1.3 million km^2) of the world's wetlands (Government of Canada, 2016a). In the last 200 years, Canada has lost roughly 15% of its wetland ecosystems, and the United States have lost 53%. These losses are mainly attributed to land conversion and water level control (Government of Canada, 2016b).

The global transportation network is immense, spanning over 40 million km, and is a major contributor to land transformation (van der Ree et al., 2011; Dulac, 2013). Since 2000, this network has grown by approximately 12 million km, with China and India accounting for more than 50% of this increase (Dulac, 2013). The global road network is expected to continue to increase dramatically by 2050, with estimates of total growth between 14-23% from 2015 to 2050 (Meijer et al., 2018), and 35-60% from 2010 to 2050 (Dulac, 2013). In Canada, the road network is also continuously increasing. In the province of Ontario, the major roads of southern Ontario increased five-fold from 7,133 km in 1935 to 35,637 km in 1995 (Fenech et al., 2005). By providing access to resources, jobs, and markets, road networks are significantly important to human socio-economic development; however, they are also recognized as a major cause of land transformation (Laurance et al., 2014; Meijer et al., 2018).

Fragmentation with Habitat Loss

Habitat loss is defined as the complete removal or modification of the environment where a species lives (Wilcove, 1986) and it has consistent negative effects on biodiversity (Fahrig, 2003). These negative effects include reductions in species richness, population abundance, distribution, and genetic diversity (Venier & Fahrig, 1996; Hanski et al., 1996; Best et al., 2001; Gibbs, 2001; Gurd et al., 2001). Habitat loss has also been shown to reduce trophic chain length (Komonen et al., 2000), alter species interactions (Taylor & Merriam, 1995), and reduce the number of large-bodied specialist species (Gibbs & Stanton, 2001). In addition, habitat loss negatively affects breeding and dispersal success, predation rate, and foraging success rate (Mahan & Yahner, 1999; Kurki et al., 2000; Bélisle et al., 2001; Bergin et al. 2000).

Anthropogenic land transformation typically results in both habitat loss and habitat fragmentation (hereafter referred to as habitat loss and fragmentation). Habitat fragmentation is defined as the transformation of the environment where a species lives into several smaller habitat patches of smaller total area, isolated from each other by a human modified matrix (Wilcove, 1986). Habitat loss and fragmentation occur concomitantly as a continuous process, and the loss of area, increase in isolation, and greater exposure to human land uses along fragment edges initiate long-term changes in landscape structure and function (van den Berg et al. 2001; Lindenmayer & Fischer, 2013; Haddad et al., 2015; Figure 1.1). The process of habitat loss and fragmentation generally results in four outcomes: (1) a reduction in habitat amount; (2) an increase in the number of habitat patches (unless entire habitat patches are lost in the process; Figure 1.2 A, C & E); (3) a decrease in habitat patch size; and (4) an increase in the distance between habitat patches (i.e. isolation) (Hagen et al., 2012; Figure 1.2). Consequently, habitat can be removed from a landscape in many different ways, resulting in many different spatial patterns (Fahrig, 2003; Figure 1.1).

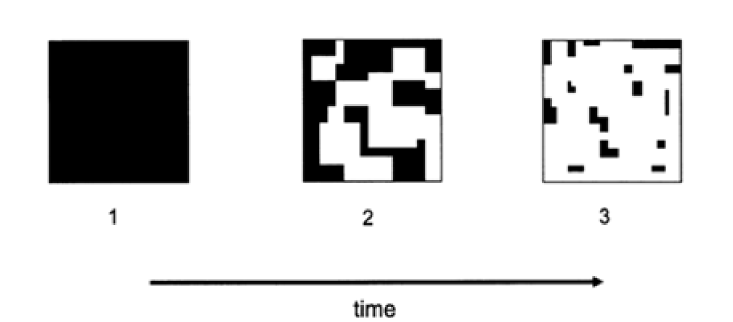

Figure 1.1 The process of habitat loss and fragmentation. A large expanse of habitat (1) is transformed into several smaller patches of smaller total area (2) and (3), isolated from each other by a human-modified matrix. Black areas represent habitat and white areas represent matrix. Taken from Fahrig (2003).

Roads are a major contributor to habitat loss and fragmentation, and their impacts on the surrounding landscape (described by the "road-effect zone"; Forman & Alexander, 1998) can extend up to several kilometers from the road edge, reducing the quality of adjacent habitats

(Benítez-López et al., 2010; Torres et al., 2016). Roads can act as barriers to movement and lead to resource inaccessibility (Forman & Alexander, 1998; Jaeger et al., 2005). Roads can also cause increased mortality due to animal-vehicle collisions (Forman & Alexander, 1998; Jaeger et al., 2005) and facilitate "contagious development" by providing access to previously isolated areas (Laurance & Balmford, 2013; Selva et al., 2015; Ibisch et al., 2016).

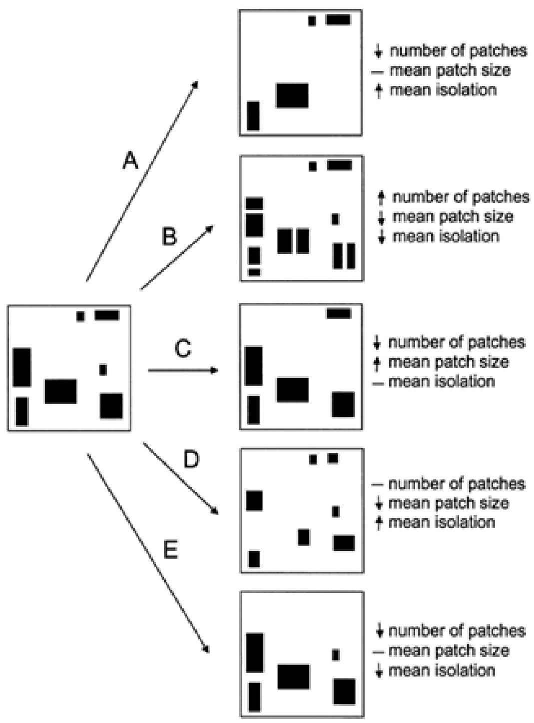

Figure 1.2 Habitat can be removed from a landscape in many ways resulting in different spatial patterns. Actual changes are indicated by arrows. Taken from Fahrig (2003).

Habitat loss and fragmentation have detrimental impacts on wildlife; including direct mortality, behavioral changes, disturbance effects such as increased noise and light, decreased dispersal capacity, and diminished genetic flow between meta-populations (Forman & Alexander, 1998; Ewers & Didham, 2006). Species with small and isolated populations (i.e., species at risk) are particularly vulnerable. These populations can become susceptible to inbreeding depression, reduced reproductive fitness, and loss of genetic diversity, reducing their ability to adapt to further environmental impacts (Traill, 2010).

Many studies have attempted to separate the effects of habitat loss and fragmentation (Gonzalez et al., 1998; 2000; Laurance et al., 2000; Ferraz et al., 2003; Collins et al., 2009; Vasconcelos et al., 2012). Haddad et al. (2015) reviewed and summarized these experiments. They found that fragmentation produced strong negative effects on biodiversity across experiments spanning numerous studies and ecosystems (Haddad et al., 2015). Consistently, all aspects of fragmentation (i.e. reduced fragment area, increased isolation, and increased edge), resulted in degraded ecosystems, reducing species persistence, species richness, nutrient retention, trophic dynamics, and, in more isolated fragments, movement (Haddad et al., 2015). Laurence et al. (2000) found that in tropical forests, reduced patch size and increased proportion of edge habitat resulted in the loss of old trees in favour of pioneer trees, with subsequent impacts on the insect community composition; while Cook et al. (2005) found that in grasslands, patch size also affected succession rate, where increased light penetration in smaller fragments impeded the rate of ecological succession relative to that of larger patches.

However, such experiments have been challenged for their ability to separate the effects of fragmentation from the effects of habitat loss (Fahrig, 2017, Fahrig et al., 2019). Studies of patch size effects and patch isolation effects do not provide evidence for effects of habitat fragmentation, because both patch size effects and patch isolation effects are inherently confounded with effects of habitat amount (Fahrig 2003, 2013). In addition, smaller patches have less habitat than larger patches, and more isolated patches are more isolated precisely because there is less habitat surrounding them than there is surrounding the less isolated patches (Fahrig, 2017). Fahrig and colleagues propose that the effects of habitat fragmentation can only be measured at the landscape scale by comparing effects of habitat configuration across multiple landscapes while controlling for the total amount of habitat within them (Fahrig, 2003; 2017).

Fragmentation without Habitat Loss

While habitat fragmentation is often thought of as a process involving both the loss and the breaking apart of habitat, habitat fragmentation independent of habitat loss (i.e., habitat fragmentation per se; Fahrig, 2003) constitutes a difference in pattern (or configuration) between landscapes (Fahrig, 2003). For a given amount of habitat, a more fragmented landscape has more, smaller habitat patches and contains a greater total length of habitat edge (Fahrig, 2017; Figure 1.3). While habitat fragmentation always includes habitat loss, habitat loss does not always entail habitat fragmentation. For example, if a single large area of habitat is diminished, this is not habitat fragmentation because the number of patches has not increased (i.e., the habitat has not been broken apart; Figure 1.3). Similarly, habitat fragmentation does not occur when a whole habitat patch is removed from a landscape, because the number of patches has not increased, but rather decreased (Fahrig, 2018).

The empirical evidence to date (as reviewed and summarised by Fahrig, 2003;2017) suggests that the effects of fragmentation per se are generally much weaker than the effects of habitat loss. However, unlike the effects of habitat loss, the effects of habitat fragmentation per se are more likely to be positive than negative (Fahrig, 2003; 2017). Fahrig (2003) suggested seven reasons for these positive effects of fragmentation per se: (1) subdivision of the same amount of habitat into smaller patches could enhance the persistence of predator-prey systems (Huffaker, 1958), by providing temporary refugia for the prey species where they can increase in numbers and disperse elsewhere before the predator or parasite finds them; (2) subdivision may

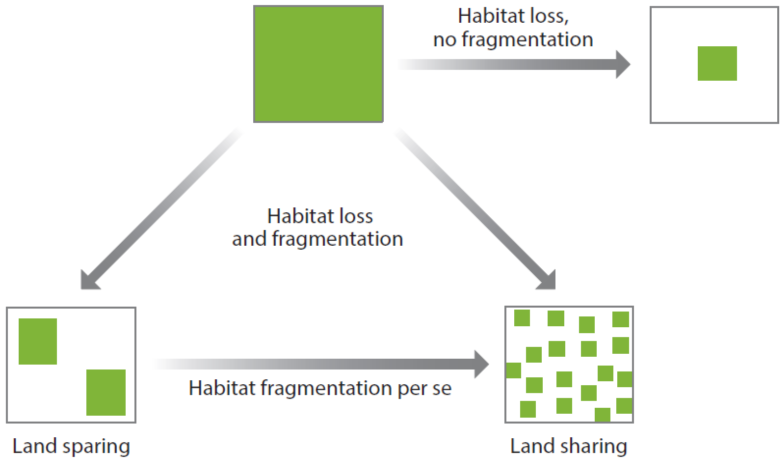

Figure 1.3 Habitat fragmentation per se is a difference in spatial pattern. For a given amount of habitat, a more fragmented pattern has more, smaller patches, with more total edge in the landscape. Taken from Fahrig (2017).

enhance the stability of two-species competition (Levin, 1974), where a trade-off between dispersal rate and competitive ability allows the inferior competitor (but superior disperser) to colonize empty patches first, before being later displaced by the superior competitor (Chesson, 1985); (3) subdivision may stabilize single-species population dynamics by reducing the probability of simultaneous extinction of the whole population (den Boer, 1981); (4) subdivision may enhance immigration rate when the landscape is made up of a larger number of smaller patches (higher fragmentation per se) than when it is comprised of a smaller number of larger patches (Bowman et al., 2002), especially in situations where immigration is an important determinant of population density; (5) subdivision may reduce patch isolation by creating smaller distances between habitat patches (Fahrig, 2003); (6) fragmented landscapes may have a greater variety of habitat types facilitating movement between different required resources (i.e., landscape complementation; Dunning et al., 1992); (7) fragmented landscapes contain more edge; thus, positive edge effects may be responsible for increases in abundance and distribution of species that exhibit positive edge effects (Laurance et al., 2001).

Another possible explanation for significant positive effects of habitat fragmentation per se is that the matrix quality may be less hostile (i.e., more wildlife friendly) in landscapes containing many small habitat patches than in landscapes containing a few large habitat patches (Fahrig, 2017). Such positive effects have been demonstrated in agricultural landscapes where crop fields form the matrix (Fahrig et al., 2011; 2015). However, the opposite could also be true when the matrix quality is more hostile (i.e., less wildlife friendly) in landscapes containing many small habitat patches than in landscapes containing a few large habitat patches (Fahrig, 2017). For example, if landscapes with many small patches contain more roads and development than landscapes with a smaller number of large patches this may result in negative responses to habitat fragmentation per se (Fahrig, 2017). Thus, considerable work is still required to determine the relationship between matrix quality and significant positive and negative responses to habitat fragmentation per se. Another important factor to keep in mind is what constituted a positive relationship in Fahrig's meta-analyses (Fahrig, 2003; 2017). Significant positive or negative relationships referred only to the direction of the relationship. Thus, increases in invasive species, pest species, or overabundant species were considered positive and then construed as being positive for biodiversity (Fahrig, 2017).

If the effects of fragmentation per se are more likely to be positive than negative (Fahrig, 2003; 2017) it would imply that conservation policies should emphasize land sparing over land sharing (Figure 1.3). However, it is important to keep in mind that approximately 24% of the significant responses to habitat fragmentation per se were negative (Fahrig, 2017). With more robust studies that aim to remove biases by controlling for such variables as landscape composition and configuration, matrix area and quality, types of biodiversity (i.e., alpha, beta and gamma diversity), dispersal capacities, and volant, non-volant and stationary species, this number may change significantly. Finally, the question must be considered, do the methods used to measure fragmentation per se explicitly separate the effects of fragmentation from habitat loss?

Connectivity

Movement is crucial for the long-term viability of wildlife populations. This includes within-home-range movements such as daily foraging, and between-home-range movements such as dispersal, migration, and range shifts in response to climate change (Zeller et al., 2012; Ament, 2014; Blazquez-Cabrera et al., 2016). Under the direct influence of climate change it has been estimated that species' ranges (including arthropods, birds, mammals, and plants) will shift to higher elevations at a median rate of 11 m per decade, and to higher latitudes at a median rate of 16.9 km per decade (Chen et al., 2011), placing increasing pressure on native biodiversity and indigenous ecosystems (Pecl et al., 2017). Dispersing individuals maintain long-term viability of meta-populations by colonizing new areas, re-colonizing sink populations, and maintaining genetic variation and gene flow (Ewers & Didham, 2006; Blazquez-Cabrera et al., 2016; Gonzalez-Saucedo, 2020).

The degree to which habitat patches within landscapes are connected influences the overall amount of movement taking place within and between local populations. Landscape connectivity is defined as "the degree to which the landscape impedes or facilitates movement of organisms among resource patches" (Taylor et al., 1993). It can influence individuals, populations, and communities through within-species, between-species, and between-ecosystem interactions (Wilson, 1999). These include predator-prey dynamics, pollination, seed dispersal, nutrient and energy flows, local adaptation, and the spread of diseases (Hanski, 1998; Rudnick et al., 2012; Ament et al., 2014; Correa Ayram et al., 2016). Movement between habitat patches is influenced by: (1) the quality of the matrix through which the organism must move; and (2) an organism's physical and mental attributes (Merriam, 1984; Ament et al., 2014). The matrix can be anything from urban development to agricultural land to human managed grasslands and forests (Kindlmann & Burel 2008). The matrix has the potential to afford habitat to some species yet the capacity to be a barrier to movement for others. Landscapes dominated by a matrix that facilitates movement will have high connectivity, while landscapes dominated by a matrix that impedes movement will have low connectivity (Kindlmann & Burel, 2008).

Because connected fragments retain more species than isolated ones, preserving or creating landscape connectivity has become increasingly recognized as a key strategy to protect biodiversity, maintain viable ecosystems and wildlife populations, and facilitate the movement and adaptation of wildlife populations in the face of increasing anthropogenic land conversion and climate change (Meiklejohn et al., 2009; Hagen, 2012; Ament et al, 2014). In heterogeneous landscapes, connectivity is attained through wildlife corridors and linkages. Wildlife corridors facilitate the movement of species between habitat patches, whereas wildlife linkages promote the movement of multiple species and ecological processes within a network of habitat patches across a large region (Beier et al., 2008; Meiklejohn et al., 2009). There are two ways to preserve and/or enhance connectivity: (1) conserve/restore areas that facilitate movement (i.e., ecological corridors and linkages); and (2) mitigate landscape features that impede movement, such as roads, railways, and other linear infrastructure (i.e., wildlife crossing structures), or areas of connectivity loss (i.e., ecological restoration). Studies have shown that implementation of both strategies together produce the most effective results (Ament et al, 2014).

Protected Areas

Protected areas have been an important tool for the conservation of biodiversity in North America. However, many protected areas are simply not large enough to support viable populations of species with large home ranges nor do they include the range of species, processes, and habitats necessary to fully conserve ecosystem integrity and biodiversity (Boyd et al., 2008; Pimm et al., 2014). Moreover, with the landscape between protected areas vulnerable to continued habitat loss and fragmentation, it is only a matter of time before protected areas become islands in a sea of human-modified landscape (Wilson & MacArthur, 1967).

As of December 2024, 17.62% of the Earth's terrestrial areas and inland water ecosystems have been protected, whereas only 13.83% of Canadian terrestrial areas and 12.95% of U.S. terrestrial areas were within protected areas and other effective area-based conservation measures (Protected Planet, 2024a/b/c). Nevertheless, it is estimated that to conserve global biodiversity and meet the area requirements for large-ranging species, a minimum of 44% of the earth's terrestrial areas require protection, including at least 64% of terrestrial areas in North America (Allan et al., 2022). To achieve such goals, it is now understood that active measures will need to be taken to not only create new and expand existing protected areas, but also maintain, enhance, or restore connectivity corridors between them; as studies have shown that interconnected protected areas (i.e., ecological networks) are much more effective at preserving biodiversity than disconnected ones (Hilty et al., 2020; Newmark et al., 2023). Ecological network-based conservation and restoration strategies are comprised of a system of protected areas interconnected by a network of protected ecological corridors that facilitate the interactions of species and ecosystems at large landscape scales while also providing appropriate opportunities for the sustainable use of natural resources (Bennett, 2004; Hilty et al. 2020).

Globally, 56% of all terrestrial mammals, 27% of all amphibians, and 69% of all birds, as well as 21% of all threatened species within these taxa, have transboundary (i.e. across national borders) geographic ranges (Mason et al., 2022). Although geographic ranges span political borders, conservation efforts usually do not, making "positive" conservation outcomes contingent on similar decisions being made across political boundaries (Kark et al., 2015; Mason et al., 2022). Transboundary conservation projects present a compelling opportunity to improve the protection of species with transboundary ranges (Vasilijević et al., 2015; Mason et al., 2022). The Convention on Biological Diversity (CBD), and the Convention on the Conservation of Migratory Species of Wild Animals (CMS) now promote the requirement for ecological connectivity across species' ranges and national borders (Trouwborst, 2012; CMS, 2019), and transboundary conservation is now a key component of the post-2020 global biodiversity framework (CBD, 2021).

The Study Area

The Adirondack-to-Laurentians (A2L) transboundary wildlife linkage encompasses the landscape between the Adirondack Mountains in New York and the Laurentian Mountains in Québec. It is one of five wildlife linkages identified by the Nature Conservancy of Canada (Fig 1.4). The area spans approximately 127,408 km^2 in size and includes 22 municipalités régionales de comté (MRCs) in Québec (58,867 km^2; 46%), five counties in Ontario (15,445 km^2; 12%), and 16 counties in New York (53,096 km^2; 42%) (Figure 1.4). The geology of the A2L is comprised of Canadian Shield (i.e., Precambrian igneous and high-grade metamorphic rocks) to the north, St. Lawrence Platform in the centre, and Precambrian to the south (Tardif, 2023), with the highest peak being Mount-Marcy in New York (1,629 m). The A2L is located in the northern forest and eastern temperate forest eco-regions and is home to 440 vertebrate species and 1600 vascular plant species (Tardif, 2005; CEC, 2023). Dominant tree species in the Québec and Ontario portions include sugar maple (*Acer saccharum*), American basswood (*Tilia americana*), white ash (*Fraxinus americana*), American hop-hornbeam (*Ostrya virginiana*), butternut (*Juglans cinerea*), yellow birch (*Betula alleghaniensis*), American beech (*Fagus grandifolia*), northern red oak (*Quercus rubra*), and eastern hemlock (*Tsuga canadensis*) (Tardif et al. 2005). Dominant tree species in the New York portion include spruce-fir, evergreen-northern hardwood, and mesic upland hardwoods including sugar maple, American beech, yellow birch, and oak (*Quercus spp.*) (Graves & Wang, 2012). As of 2016, the study area was home to over 6.8 million people and an overall population density of 54.0 per km^2. This is an increase of 1.1 million people since 1990 when the population density was 44.5 per km^2 (Statistics Canada, 1991; 2016; US Census bureau, 1990; 2016). Although this region still maintains habitats of high ecological integrity and biodiversity, increased anthropogenic land transformation could be impacting transboundary connectivity. Thus, there is growing concern about the need to protect the integrity and connectivity of the landscape to ensure the continued functionality of the transboundary linkage.

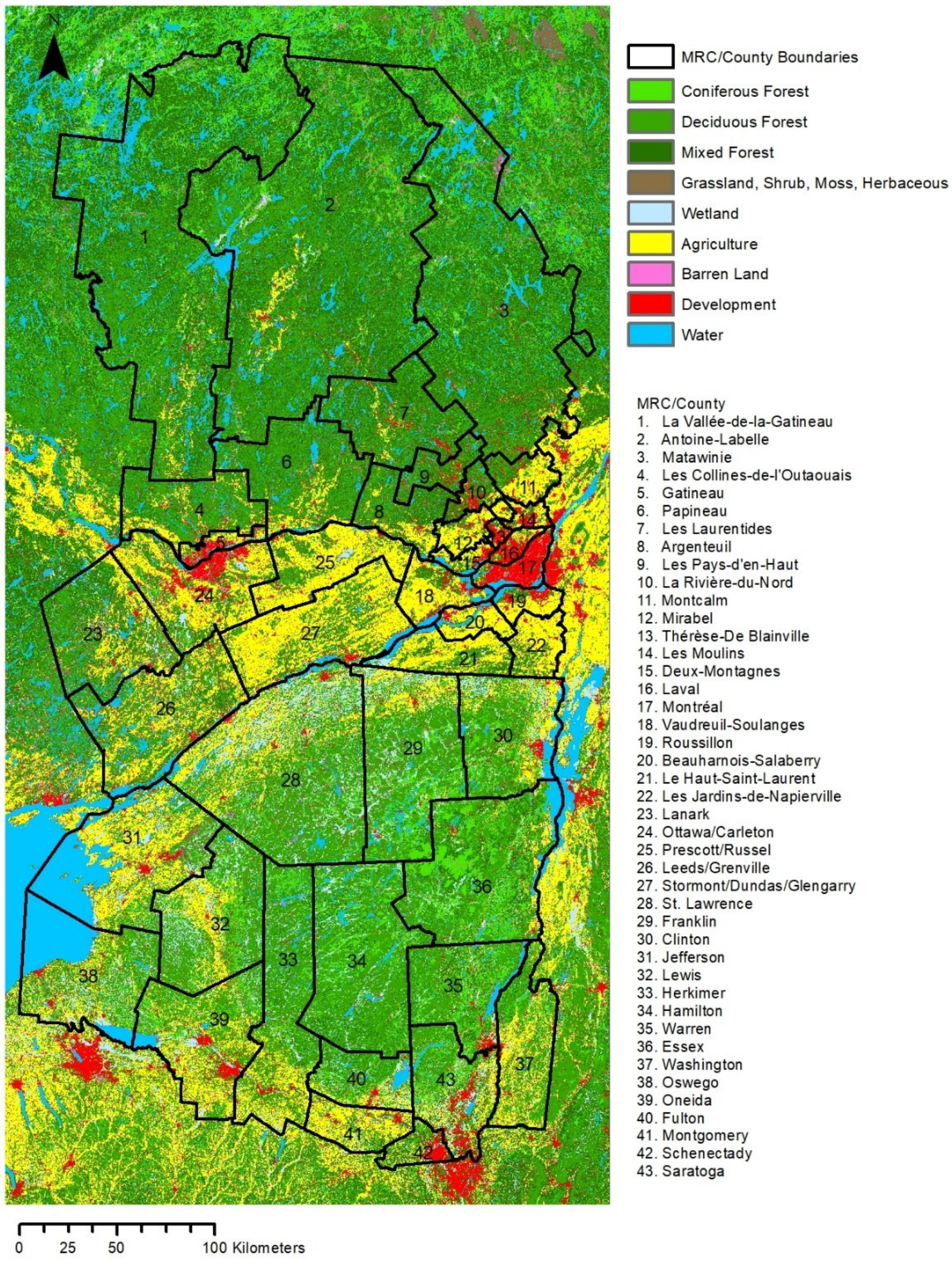

Figure 1.4 Land-cover map of the Adirondack-to-Laurentians (A2L) study area overlaid with municipalité régionale de comté (MRC)/county boundaries. MRC/county names are numbered and correspond to the numbers on the map.

The Structure of the Thesis

The research in this thesis is part of the Quebec Ecological Corridors Initiative created by the Nature Conservancy of Canada along with 9 partner organizations as a nature-based adaptation strategy that identifies and protects ecological corridors (Figure 1.5). Although there has been

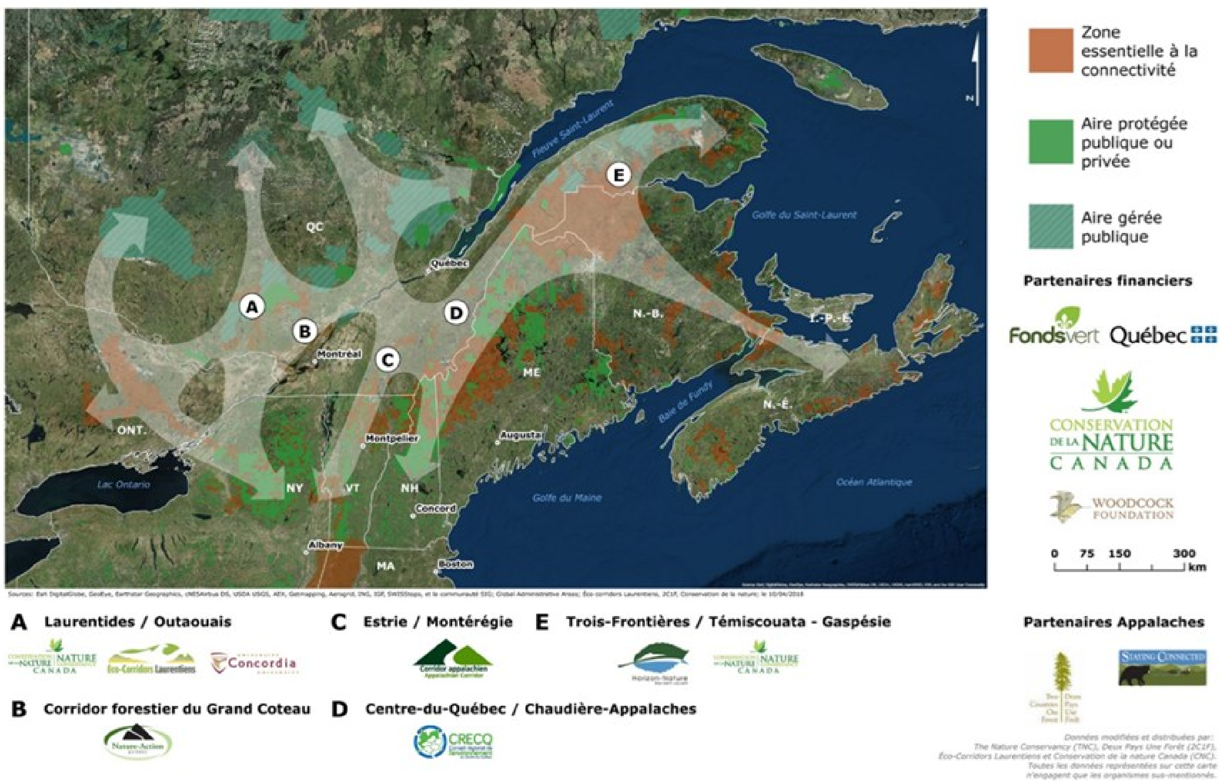

Figure 1.5 Map of wildlife linkages in Eastern Canada as determined by Nature Conservancy of Canada. A. The Adirondack-to-Laurentians (A2L) transboundary wildlife linkage. (Source: Nature Conservancy of Canada, 2021)

extensive research investigating other wildlife linkages in the region such as the Algonquin-to-Adirondack (A2A) (i.e., Koen et al., 2014; Garrah et al., 2015) and the St-Lawrence lowlands and Montérégie region (i.e., Gonzalez et al., 2013; Dupras et al., 2016; Albert et al., 2017; Meurant et al., 2018; Rayfield et al., 2018), to my knowledge no studies have documented the functionality of the A2L wildlife linkage (Figure 1.5). Therefore, within this thesis, I conduct an in-depth, multi-phase analysis to quantify the impacts of anthropogenic land transformation on (1) landscape structure and structural connectivity; (2) habitat amount, habitat fragmentation and functional connectivity; and (3) potential eastern wolf habitat amount and functional connectivity within the A2L transboundary wildlife linkage. The overall aim of this research was to determine the functionality of the A2L to act as a wildlife linkage, and to what extant it has been weakened due to land transformation since 1992.

This thesis is presented in a manuscript-based format. It consists of three manuscript chapters (Chapters 2-4) on which I am lead author. Each manuscript has been published in a peer-reviewed scientific journal. The style of each manuscript chapter follows the scientific journal it was published in, i.e., *Landscape Ecology* (Chapters 2 & 3), *Regional Environmental Change* (Chapter 4). Each manuscript chapter is the result of collaborations with other researchers, and I therefore use the plural "we" throughout these chapters. Each manuscript chapter was written to stand-alone and includes a brief introduction to the background literature relevant to that study. The thesis introduction (Chapter 1) therefore acts to expand on the underlying themes of the thesis and provides further rationale for the intent of the research. I use connecting statements to provide logical bridges between each manuscript chapter outlining how the chapters are linked. A fifth chapter serves as a synthesis of the thesis and provides general conclusions.

Chapter 2: Monitoring changes in landscape structure in the Adirondack-to-Laurentians (A2L) transboundary wildlife linkage between 1992 and 2018: Identifying priority areas for conservation and restoration. This chapter has been published by Jonathan R. Cole, Angela Kross, & Jochen A. G. Jaeger, (2023) *Landscape Ecology*, *38*, 383-408.

In the first phase of analysis, I quantify changes in landscape structure within the A2L transboundary wildlife linkage between 1992 and 2018. Landscape structure consists of landscape composition (the amount of each land-cover type) and landscape configuration (the spatial arrangement of land-cover elements) (Turner et al. 2001). I then use this information to identify priority areas for conservation and ecological restoration. I quantify landscape structure at three distinct hierarchical scales (study area, provincial/state portion, MRC/county) in order to: (1) visualize how land-cover change and landscape fragmentation were spatiotemporally distributed; (2) allow for the direct comparison and ranking between provincial/state portions and MRCs/counties; (3) determine priority areas for conservation and/or ecological restoration; and (4) provide multiple levels of governance with the information necessary to develop coordinated and collaborative local, regional, and transboundary conservation plans. To compare changes in landscape composition, I calculated the area and proportion of five grouped land-cover themes between 1992 and 2018, and to compare changes in landscape configuration I measured patch number, mean patch size, the effective mesh size, and road density, between 2000 and 2018.

This chapter addressed three research questions:

1. To what degree have land-cover change and landscape fragmentation occurred within the A2L transboundary wildlife linkage?
2. Are there spatiotemporal differences in land-cover change between grouped land-cover themes, and between scales?
3. Are there spatiotemporal differences in landscape fragmentation between the four fragmentation geometries, and between scales?

Chapter 3: Impacts of anthropogenic land transformation on species-specific habitat amount, fragmentation, and connectivity in the Adirondack-to-Laurentians (A2L) transboundary wildlife linkage between 2000 and 2015: Implications for conservation and ecological restoration. This chapter has been published by Jonathan R. Cole, Erin L. Koen, Eric J. Pedersen, John A. Gallo, Angela Kross, & Jochen A. G. Jaeger (2023) *Landscape Ecology*, *38*, 2591–2621.

In the second phase of analysis, I measured the impacts of anthropogenic land transformation on species-specific habitat amount, fragmentation, and connectivity in the Adirondack-to-Laurentians (A2L) transboundary wildlife linkage between 2000 and 2015. I developed suitable habitat and resistance models for four native species: American black bear (*Ursus americanus*), fisher (*Pekania pennanti*), moose (*Alces alces*), and white-tailed deer (*Odocoileus virginianus*) and identified suitable and optimal habitat patches for each species. I then quantified habitat amount, fragmentation, and connectivity, and used Linkage Mapper and Circuitscape to map corridors and pinch-points important for connectivity. I used this information to identify priority habitat patches and corridors for conservation and restoration.

This chapter addressed three research questions:

1. To what degree has habitat loss and fragmentation occurred within the study area for each species?
2. To what degree has connectivity changed for each species?
3. What percentage of habitat patches and corridors are under protection for each species?

Chapter 4: Land conversion and lack of protection significantly reduce suitable wolf habitat amount and functional connectivity in the Adirondack-to-Laurentians (A2L) transboundary wildlife linkage. This chapter has been published by Jonathan R. Cole, Marianne Cheveau, John A. Gallo, Angela Kross, Martin-Hugues St-Laurent, & Jochen A.G. Jaeger (2024) *Regional Environmental Change, 24*, 1-18.

While the coyote (*Canis latrans*) is ubiquitous throughout the A2L region, gray wolves (*Canis lupus*) and eastern wolves (*Canis lupus lycaon* - a threatened species in Ontario and a species of special concern federally) only occur within the Québec portion of the study area. In 2021, the federal government released a management plan for the eastern wolf in Canada (ECCC, 2021). The plan includes two primary conservation objectives: 1) achieve and maintain viable eastern wolf populations within the species' current range in Canada; and 2) achieve and maintain connectivity between occupied sites as well as potential suitable habitat sites to facilitate dispersal and maintain genetic diversity (ECCC, 2021). Potential suitable habitat and dispersal routes for the eastern wolf have not been re-examined since *circa* 2000 (Harrison & Chapin, 1998; Mladenoff & Sickley, 1998; Paquet et al., 1999; Carroll, 2003). Thus, there is an urgent need for updated information to achieve these objectives. Consequently, in this final phase of analysis, I explored the impacts of land conversion on wolf habitat amount, habitat fragmentation, and functional connectivity in the A2L transboundary wildlife linkage between 2000 and 2015. I identified potential suitable habitat patches, optimal habitat patches, and stepping-stone patches, and measured habitat fragmentation. I then applied Linkage Mapper and Circuitscape to the habitat network to map least-cost corridors and pinch-points important for functional connectivity. I used this information to identify priority habitat patches and corridors for conservation and restoration.

This chapter addressed three research questions:

1. To what degree have habitat loss and fragmentation occurred within the study area for the wolf?
2. To what degree has connectivity changed within the study area for the wolf?
3. What percentage of habitat patches and corridors are under protection?

It is my hope that the questions raised and addressed in this temporal analysis of the Adirondak-to-Laurentians transboundary wildlife linkage will not only prove useful to local land-use and conservation planners but will also interest practitioners of transboundary conservation globally.

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2. Monitoring Changes in Landscape Structure in the Adirondack-to-Laurentians (A2L) Transboundary Wildlife Linkage between 1992 and 2018: Identifying Priority Areas for Conservation and Restoration

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Abstract

Context

Although many species have transboundary geographic ranges, most conservation initiatives do not cross political boundaries. The landscape between the Adirondack Mountains in New York and the Laurentian Mountains in Québec includes one of three potential north–south transboundary wildlife movement linkages that connect wilderness areas in northeastern USA and southeastern Canada. Although this region still maintains habitats of high ecological integrity and biodiversity, increasing land-cover changes and fragmentation are putting landscape connectivity at risk.

Objectives

We measured changes in landscape composition and configuration within the Adirondack-to-Laurentians transboundary wildlife linkage (A2L) between 1992 and 2018 to identify priority areas for conservation and restoration.

Methods

Land-cover change was calculated by measuring area and proportion of land-cover classes, and landscape fragmentation was determined by measuring patch number, mean patch size, the effective mesh size, and road density, at three spatial scales and four fragmentation geometries (i.e., combinations of fragmenting elements).

Results

Extensive changes in land-cover and landscape fragmentation occurred within the A2L between 1992 and 2018. Forest areas declined by 1363 km^2 and wetlands declined by 1365 km^2 (69%). This was most pronounced in the Québec portion of the A2L where wetland areas declined by 872 km2 (88.5%). Forest areas in the Québec portion experienced the greatest amount of fragmentation with a $m_{\text{eff\_CUT}}$ decline of 3262.5 km^2 (58.5%) since 2000.

Conclusions

Coordinated and collaborative transboundary conservation efforts help improve protection of species with transboundary ranges. Monitoring land-cover changes and landscape fragmentation is an effective way to identify priority areas for conservation and support transboundary coordination. Strengthening conservation strategies that enhance landscape connectivity and protect ecosystems at the local level will help achieve post-2020 biodiversity commitments at the national and transboundary levels.

Keywords: Land-cover change · Landscape fragmentation · Patch number · Mean patch size · Effective mesh size · Road density · Transborder · Post-2020 Biodiversity Framework.

Introduction

Land conversion from natural areas to human modified uses is the leading cause of biodiversity loss worldwide (Hooke et al. 2012). Humans have altered greater than 50% of the Earth's terrestrial surface (77% excluding Antarctica) (Hooke et al. 2012; Allen et al. 2017; Watson et al. 2018). Land conversion results in habitat loss and fragmentation (i.e., "where a large expanse of habitat is transformed into a number of smaller patches of smaller total area, isolated from each other by a matrix of habitats unlike the original"; Wilcove et al. 1986), which contributes to long-term changes in landscape structure and function (Lindenmayer and Fischer 2013; Haddad et al. 2015). Landscape structure consists of landscape composition (the amount of each land-cover type present in the landscape) and landscape configuration (the spatial arrangement of land-cover elements) (Turner et al. 2001). Habitat loss and fragmentation then relate to both changes in landscape composition and configuration (Fletcher et al. 2016). Roads are a major contributor to habitat loss and fragmentation (van der Ree et al. 2011). The global road network spans over 40 million km (Dulac 2013). Since 2000 this network has grown by approximately 12 million km, and it is expected to continue to grow by more than 35% by 2050 (Dulac 2013).

Movement is crucial for long-term viability of wildlife populations including daily foraging, dispersal, migration, and range shifts in response to climate change (Ament et al. 2014). Dispersing individuals maintain long-term viability of populations by colonizing new areas, re-colonizing sink populations, and maintaining genetic variation and gene flow within meta-populations (Ewers and Didham 2006; Traill et al. 2010; McGuire et al. 2016; Blazquez-Cabrera et al. 2016). Animal movements and many other ecological processes require connectivity ("the degree to which the landscape facilitates or impedes movement among resource patches"; Taylor et al. 1993). Connectivity is subdivided into two main branches in terms of its measurement: "Structural connectivity" refers to the arrangement and contiguity of land-cover elements (Lindenmayer and Fischer 2013; Hilty et al. 2020), whereas "functional connectivity" is species-specific and is described as the product of both landscape structure and the responses of a species to that structure (i.e., the ability of a species to move between resource patches within a landscape) (Meiklejohn et al. 2009; Lindenmayer and Fischer 2013). In heterogeneous landscapes, connectivity is attained through wildlife corridors and linkages.

Wildlife corridors facilitate the movement of species between habitat patches, whereas wildlife linkages promote the movement of multiple species and ecological processes within a network of habitat patches across a large region (Beier et al. 2008; Meiklejohn et al. 2009).

Globally, 56% of all terrestrial mammals, 27% of all amphibians, and 69% of all birds, as well as 21% of all threatened species within these taxa, have transboundary geographic ranges (Mason et al. 2020). Although geographic ranges span political borders, conservation usually does not, making conservation outcomes contingent on similar decisions being made across multiple provincial/state or national boundaries (Kark et al. 2015; Mason et al. 2020). Transboundary conservation presents an opportunity to improve protection of species with transboundary ranges through coordinated and collaborative international conservation efforts (Vasilijević et al. 2015; Mason et al. 2020). The Convention on Biological Diversity (CBD) and the Convention on the Conservation of Migratory Species of Wild Animals (CMS) now promote the requirement for ecological connectivity across species' ranges and national borders (Trouwborst 2012; CMS 2019), and transboundary conservation is currently recognized as a key component in the post-2020 global biodiversity framework discussions (SCBD 2018; Díaz et al. 2020; Mason et al. 2020). The International Union for the Conservation of Nature (IUCN) - World Commission on Protected Areas (WCPA) recognizes three types of transboundary conservation (Vasilijević et al. 2015): type 1. Transboundary Protected Area (i.e., protected areas ecologically connected across one or more international boundaries; type 2. Transboundary Conservation Landscape and/or Seascape (i.e., ecologically connected areas that include both protected areas and multiple resource use areas across one or more international boundaries); and type 3. Transboundary Migration Conservation Area (i.e., wildlife habitats in two or more countries that are necessary to sustain populations of migratory species).

Subsequently, a special designation of Park for Peace (i.e., protected areas established for the conservation of biodiversity, cultural resources, and regional peace and stability; Sandwith et al 2001) can be applied to each of the three types (Vasilijević et al. 2015).

The landscape between the Adirondack Mountains in New York State, USA, and the Laurentian Mountains in Québec, Canada, (hereafter, referred to as the A2L) is one of three potential north–south transboundary wildlife movement linkages that connect natural areas in Northeastern USA with Southeastern Canada. This region boasts a wide variety of habitats that still maintain a high degree of ecological integrity and are rich in biodiversity (Tardif et al. 2005). However, land conversion due to urban and industrial development, agriculture, roads, and other infrastructure, have led to the current mosaic that includes a central band of intensive agricultural and urban areas running parallel to the St Lawrence and Ottawa rivers, while forest fragments dominate the northern and southern domains (Pan et al. 1999; Bélanger and Grenier 2002; Brisson and Bouchard 2003).

In this study we measure changes in landscape structure within the A2L transboundary wildlife linkage to identify priority areas for conservation and ecological restoration at three spatial scales (the complete study area; the three provincial/state portions; and the 43 municipalités régionales de comté (MRCs)/counties that reside within the transboundary wildlife linkage). To evaluate changes in landscape composition, we calculated and compared the area and proportion of five grouped land-cover themes between 1992 and 2018. To assess changes in landscape configuration (i.e., landscape fragmentation and structural connectivity), we measured and compared patch number, mean patch size, the effective mesh size, and road density, between 2000 and 2018. Calculating landscape fragmentation and structural connectivity requires the specification of the natural and anthropogenic landscape elements that cause fragmentation (i.e., roads, development, agriculture, waterbodies, etc.). The specific choices of these fragmenting elements define what is called the "fragmentation geometry" (Girvetz et al. 2008). In this study we analyze and compare four different fragmentation geometries, each representing different land-cover scenarios.

We asked the following research questions: (1) To what degree have land-cover change and landscape fragmentation occurred within the A2L transboundary wildlife linkage? (2) Are there spatiotemporal differences in land-cover change between grouped land-cover themes, and between scales? (3) Are there spatiotemporal differences in landscape fragmentation between the four fragmentation geometries, and between scales?

Methods

Study area

The A2L spans an area of over 127,000 km^2 from the Adirondack Mountains in New York, USA to the Laurentian Mountains in Québec, Canada (Fig. 2.1). It is within the Northern Forests ecoregion to the north and the Eastern Temperate Forests ecoregion to the south (EPA, 2024). This region is home to 440 vertebrate species and 1600 vascular plant species (Tardif et al. 2005). Its geology is comprised of Canadian Shield to the north and St. Laurence Platform

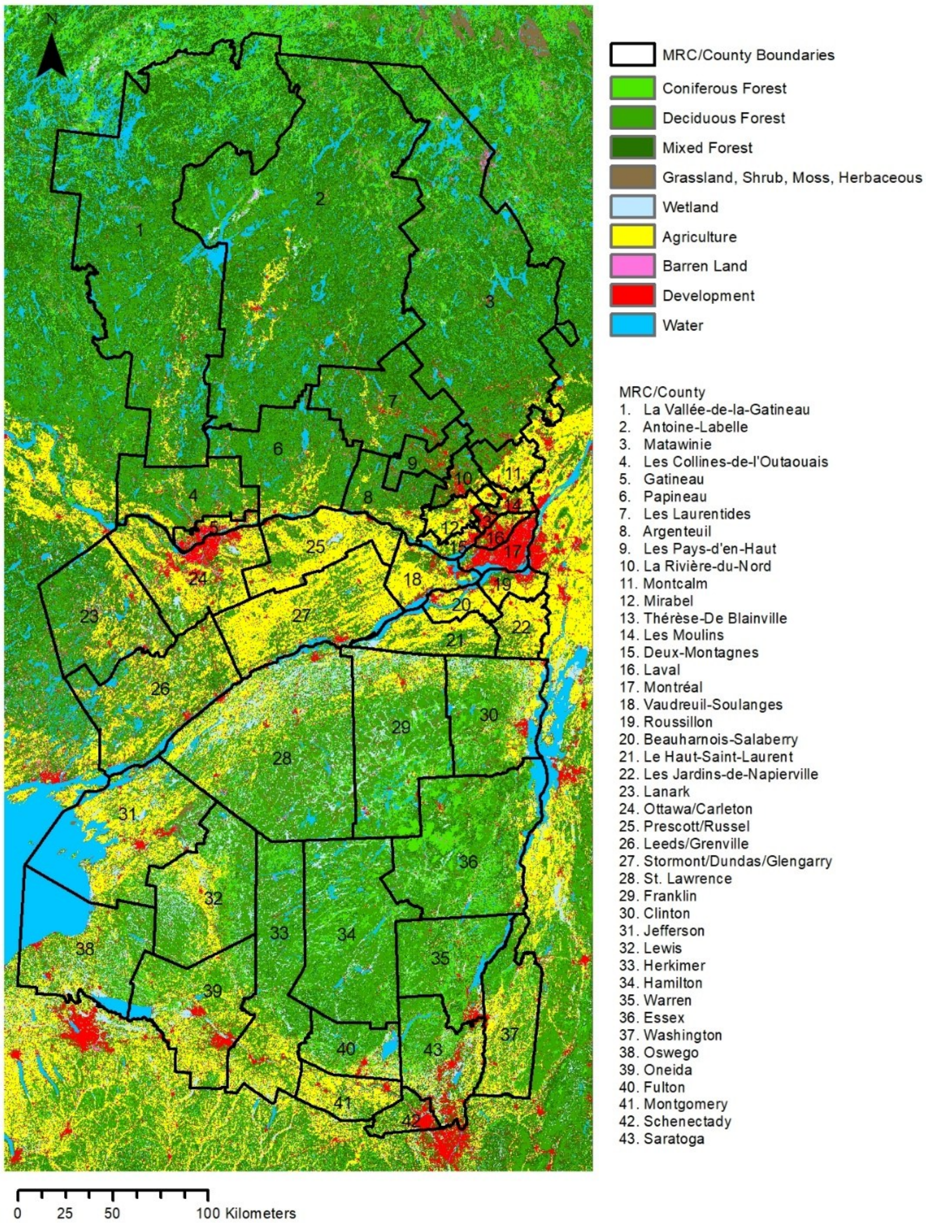

Figure 2.1 Land-cover map of the Adirondack-to-Laurentian (A2L) study area overlaid with municipalité régionale de comté (MRC)/county boundaries. MRC/county names are numbered and correspond to the numbers on the map.

to the south (Tardif et al. 2005). The highest peak is Mount-Marcy in the Adirondacks at 1629 m above sea level. The three bioclimatic domains within the A2L region include the maple/bitternut hickory which has the mildest climate and is made up of diverse forests containing butternut and shagbark hickories, hackberries, black maple, swamp white oak, rock elm, and pitch pine (Tardif et al. 2005). The maple/basswood domain further to the north and east contains forests of sugar maple, American basswood, white ash, American hophornbeam, and butternut (Tardif et al. 2005). The most northern is the maple/yellow birch domain. It is the least diverse and includes yellow birch, sugar maple, American basswood, American hophornbeam, American beech, northern red oak, and eastern hemlock (Tardif et al. 2005). As of 2016, the area was home to over 6.8 million people (54 per km^2), an increase of 1.1 million people since 1990 (45 per km^2) (Statistics Canada 1991; 2016; US Census bureau 1990; 2016).

Data collection

We used four, 300 m resolution, global land-cover maps from the European Space Agency Climate Change Initiative Land-Cover Project (ESA-CCI-LC). Each map contained 24 consistent land-cover classes based on the United Nations (UN) Land-Cover Classification System (LCCS) (Table 2.1 and S2.9). The ESA-CCI-LC dataset had higher classification accuracy (~73.9–74.2%) and stability across timepoints than any other existing dataset (i.e., MODIS annual series from 2001 to 2020 (500 m resolution), and GLASS Products annual series from 1982 to 2018 (5 km resolution) (Sun et al. 2022). The primary limitation of the dataset was some inaccuracy in land-cover classification which varied according to global region. Most classification errors were between classes within the same theme (i.e., broad-leaved forest vs. needle-leaved forest) (Santoro et al. 2017). However, North America was in a high-quality

Land-cover class/element	Land-cover themes					Fragmentation geometries			
	Natural and anthropogenic fragmentation elements	Forests	Non-forest vegetation	Wetlands	Combined habitats	FG-forests	FG-non-forest vegetation	FG-wetlands	FG-combined habitats
Development	✓					✓	✓	✓	✓
Bare areas	✓					✓	✓	✓	✓
Waterbodies	✓					✓	✓	✓	✓
Agricultural land	✓					✓	✓	✓	✓
Forests		✓			✓		✓	✓	
Grassland, shrub, moss, herbaceous cover			✓		✓	✓		✓	
Wetlands				✓	✓	✓	✓		
Primary roads (10m buffer)						✓	✓	✓	✓
Secondary roads (5m buffer)						✓	✓	✓	✓
Tertiary roads (3m buffer)						✓	✓	✓	✓

Table 2.1 Map categories included in each of the "Land-Cover Themes" and "Fragmentation Geometries". Agricultural land included the land-cover classes: cropland, rainfed; cropland, rainfed, herbaceous cover; mosaic cropland (>50%) / natural vegetation (tree, shrub, herbaceous cover) (<50%); mosaic natural vegetation (tree, shrub, herbaceous cover) (>50%) / cropland (<50%). Forests included the land-cover classes: broad-leaved evergreen closed to open tree / broad-leaved semi-deciduous closed to open trees; tree cover, broad-leaved, deciduous, closed to open (>15%); tree cover, broad-leaved, deciduous, closed (>40%); tree cover, broadleaved, deciduous, open (15-40%); tree cover, needle-leaved, evergreen, closed to open (>15%); tree cover, needle-leaved, evergreen, closed (>40%; tree cover, needle-leaved, evergreen, open (15-40%); tree cover, needle-leaved, deciduous, closed to open (>15%); tree cover, mixed-leaf type (broad-leaved and needle leaved); mosaic tree and shrub (>50%) / herbaceous cover (<50%). Grassland, shrub, moss, herbaceous cover included the land-cover classes: mosaic herbaceous cover; shrubland; grassland; lichens and mosses; sparse vegetation (tree, shrub, herbaceous cover) (<15%). Wetlands included the land-cover classes: tree cover, flooded, fresh or brackish water; shrub or herbaceous cover, flooded, fresh/saline/brackish water (✓) = included.

region, and we also grouped classes into themes (i.e., Forests, Non-Forest Vegetation, Wetlands, etc.) which would have considerably reduced any prevailing classification errors. All land-cover classes were subject to the resolution of the ESA-CCI-LC dataset. For example, a cell (300 m x 300 m) was classified as water if the cell contained greater than 50% water (Lamarche et al. 2017). Nevertheless, there are patches smaller than 90,000 m^2 in the fragmentation analysis because the surface area of the roads and their buffers were erased from the vectorized land-cover maps prior to the fragmentation calculations (see *Creating fragmentation geometry patches*).

The ESA-CCI-LC maps did not contain a separate road category. To complete the landscape fragmentation analysis, a set of compatible road-network maps were required for Québec, Ontario, and New York. Road maps for Québec and Ontario were obtained from DMTI Spatial Inc., for the years 2000, 2010, and 2018. Road maps for New York State were obtained from New York State Information Technology Services for 2010 and 2018. Due to a lack of digital road maps for 1992 and inconsistencies in the maps for 2000, the landscape fragmentation analysis was performed for 2000, 2010, and 2018 in the Québec and Ontario portions, and for 2010 and 2018 in the New York portion. Road categories were reclassified into: (1) Primary Roads; (2) Secondary Roads; and (3) Tertiary Roads (Table S2.1).

The railway network was not considered in the landscape fragmentation analysis because compatible data for all provincial/state portions and timepoints were unavailable. However, railway density, traffic, and speed within the study area are considerably low, especially compared to European and Asian equivalents. Many railway tracks run parallel to other natural and anthropogenic fragmentation elements such as roads, development, waterbodies, barren areas, and agriculture, which would capture their fragmentation by proximity.

We analyzed land-cover change and landscape fragmentation at three spatial scales (study area, provincial/state portion, MRC/county). We utilized these distinct hierarchical scales to: (1) visualize how land-cover change and landscape fragmentation were spatiotemporally distributed; (2) allow for the direct comparison and ranking between provincial/state portions and MRCs/counties; (3) determine priority areas for conservation and/or ecological restoration; and (4) provide multiple levels of governance with the information necessary to develop coordinated and collaborative local, regional, and transboundary conservation plans.

Land-cover change

Land-cover classes were grouped into five themes: (1) “Natural and Anthropogenic Fragmentation Elements”, which included development, barren areas, waterbodies and agricultural land; (2) “Forests”, which contained all forest types; (3) “Non-Forest Vegetation”, which included all grassland, shrub, moss, and herbaceous land-cover types; (4) “Wetlands”, which included all wetland types; and (5) “Combined Habitats”, which included the combined themes of “Forests”, “Non-Forest Vegetation”, and “Wetlands” (Table 2.1and S2.9).

To quantify land-cover change over time, we calculated and compared the area and proportion of the five grouped land-cover themes between 1992, 2000, 2010, and 2018. Land-cover area was calculated by multiplying the cell count of each land-cover class (within the boundaries of the reporting unit; see below) by the area of a single cell (90,000 m^2), and then dividing by 1,000,000 m^2 /km^2 to convert to km^2. Land-cover proportion was calculated by dividing the land-cover area by the total area of the reporting unit, and then multiplying by 100%, to convert to percent.

Landscape fragmentation

Fragmentation geometries and reporting units

A fragmentation geometry specifies all the land-cover classes and elements that will be considered barriers in the fragmentation analysis. Including more barriers in a fragmentation geometry will increase the degree of fragmentation (Roch and Jaeger 2014). The justification for including, or not including, a specific barrier depends on the type of fragmentation being analyzed. For instance, if the goal is to quantify the overall degree of "forest" fragmentation, then every land-cover type that is not "forest" will be included as a barrier (Roch and Jaeger 2014).

The spatial boundaries in which land-cover change and the degree of landscape fragmentation are calculated are referred to as "reporting units" (Girvetz et al. 2008). Reporting units (i.e., political boundaries or ecological regions) can occur at a range of spatial scales and are often hierarchically organized (Girvetz et al. 2008). The reporting units in this study represent three scales of analysis: the entire study area, the Québec, Ontario, and New York portions, and 22 MRCs in Québec, 5 counties in Ontario, and 16 counties in New York (Fig. 2.1).

For this study, we used four different fragmentation geometries (Table 2.1 and S2.9) that complimented the range of grouped land-cover themes that were assessed for land-cover change: (1) "FG-Forests" included all land-cover classes and elements (i.e., all three road classes) that were not forest types. FG-Forests represented the patches of remaining forest cover within the study area; (2) "FG-Non-Forest Vegetation" included all land-cover classes and elements that were not grassland, shrub, moss, or herbaceous land-cover types. FG-Non-Forest Vegetation represented the remaining patches of grassland, shrub, moss, and herbaceous cover; (3) "FG-

Wetlands” included all land-cover classes and elements that were not wetlands. FG-Wetlands represented the patches of remaining patches of wetland areas; and (4) “FG-Combined Habitats” included all land-cover classes and elements that were not forest types, grassland, shrub, moss, herbaceous, or wetland cover types. FG-Combined Habitats represented the remaining patches of natural land-cover within the transboundary wildlife linkage. These four fragmentation geometries were created to represent land-cover themes that are potential habitats for species living within the transboundary linkage. Consequently, the results of the fragmentation analysis can be applied to a range of species, for example, habitat specialists, which inhabit a specific habitat type (i.e., “FG-Forests”, “FG-Non-Forest Habitats”, or “FG-Wetlands”) and/or habitat generalists, which inhabit a range of habitats (i.e., “FG-Combined Habitats”).

Creating fragmentation geometry patches

Each of the ESA-CCI-LC raster maps were reclassified in ArcGIS10 (Environmental Systems Research Institute, Redlands, CA) to represent each of the fragmentation geometry classifications (i.e., fragmenting elements/barriers=1; non-fragmenting elements/non-barriers=2), and then converted to vector using the ‘‘Raster to Polygon’’ function, with the parameter ‘‘no simplify’’, to ensure the resulting polygons matched their raster counterparts. The fragmenting elements were then removed from each map using “Select by Attributes” and selecting for the non-fragmenting elements. Next, each of the road classes were buffered to represent real-world widths. Primary roads were buffered by 10 m (on either side), secondary roads by 5 m, and tertiary roads by 3 m (Girvetz et al. 2008). The surface of the buffered road networks (for each timepoint) were erased from each fragmentation geometry map using the “Erase” function, resulting in vector maps of patches of the non-fragmenting elements, for each fragmentation geometry scenario.

Patch number and mean patch size

We calculated “patch number” (i.e., the number of patches within a reporting unit for a specific fragmentation geometry) and “mean patch size” (i.e., the sum of each patch area within a reporting unit of a specific fragmentation geometry divided by the number of patches) for each scale of the analysis using the “Feature Area” function in ArcGIS10. The combined use of these two metrics presents a simple approach for quantifying landscape fragmentation. In general terms, as landscape fragmentation increases within a reporting unit, patch number increases and mean patch size decreases (Santiago-Ramos and Feria-Toribio 2021).

Effective mesh size

The effective mesh size fragmentation metric is a more advanced approach for quantifying landscape fragmentation (Jaeger 2000; Moser et al. 2007). The effective mesh size is based on the average probability that any two randomly chosen points in the study area are structurally connected with one another (i.e., not separated by a fragmentation barrier) (Jaeger 2000). The effective mesh size therefore also serves as a measure of structural connectivity (i.e., the degree to which movement between different parts of the landscape is possible) (Jaeger et al. 2011). By multiplying this probability by the total area of the reporting unit, it is converted into a surface area: the effective mesh size. The more barriers fragmenting the landscape, the lower the probability that the two points are connected, and the lower the effective mesh size (Girvetz et al. 2008; Jaeger et al. 2011). Because the boundary of a reporting unit can profoundly influence the effective mesh size, two variations of the effective mesh size were used to quantify the degree of landscape fragmentation and structural connectivity.

The "cutting out" procedure ($m_{\text{eff\_CUT}}$) was used to measure fragmentation strictly within the boundaries of the reporting units, with

$$m_{\text{eff\_CUT}} = \frac{1}{A_{\text{total}}} \sum_{i=1}^{n} A_i^2 \quad ,$$

where n=the number of patches inside the reporting unit; A_i=the sizes of the n patches (i=1, …, n); and A_{total}=total terrestrial area of the reporting unit (excluding waterbodies). Patches that extend outside the boundary are "cut" along the border leaving only the portion that resides within the reporting unit to be measured. This procedure enables multiple reporting units to be compared on the basis of the fragmentation of the terrestrial area strictly within their borders (e.g., MRCs/counties). The value of $m_{\text{eff\_CUT}}$ varies between zero, when the reporting unit is completely fragmented (i.e., contains no habitat of interest), and the total area of the reporting unit, when there is no fragmentation (i.e., the reporting unit contains only habitat of interest).

The "cross-boundary connections" procedure ($m_{\text{eff\_CBC}}$) was used to include the area of patches that cross the boundaries of the reporting units. This metric allocates the area of the boundary-crossing patches to both reporting units (Moser et al. 2007), with

$$m_{\text{eff\_CBC}} = \frac{1}{A_{\text{total}}} \sum_{i=1}^{n} A_i \cdot A_i^{\text{cmpl}} \quad ,$$

where A_i=the size of patch i inside the boundary of the reporting unit (i=1, 2, 3, …, n) and A_i^{cmpl}

= the area of the complete patch that A_i is a part of (i.e., including the area on the other side of the boundary) and n and A_{total} as above. This procedure considers the overall fragmentation pattern in the landscape rather than just within each reporting unit (Moser et al. 2007).

Although the effective mesh size is typically calculated using the entire area of the reporting unit, the proportion of waterbodies within the reporting units in this study varied between 1% and 30%. Therefore, we followed the approach of Jaeger et al. (2007a; 2008) by comparing landscape fragmentation only between the terrestrial areas of the reporting units. Accordingly, for both $m_{\text{eff\_CUT}}$ and $m_{\text{eff\_CBC}}$, A_{total}=total terrestrial area of the reporting unit (i.e., excluding waterbodies).

Road density

Road length and density were measured for each timepoint (2000, 2010, and 2018) to determine by how much the road network had increased as well as its spatiotemporal pattern of increase. Road length was measured by summing the polyline lengths (in metres) of each road category within each reporting unit, and then dividing by 1000 m/km, to convert to km. Road density was calculated by dividing the road length by the area of the reporting unit and then dividing by 1,000,000 m^2/km^2 to convert to kilometres of road per km^2.

Priority areas for conservation and ecological restoration

We applied the following criteria to prioritize reporting units for conservation and/or restoration intervention. For all land-cover change and landscape fragmentation measurements, changes of less than 10% were considered low priority (i.e., of least concern), changes between 10% and 30% were considered medium priority, changes between 30% and 50% were considered medium–high priority, and changes >50% were considered high priority. Also, for reasons discussed in the "Recommendations" sub-section, we considered all reporting units with less than 30% combined habitats remaining, all habitat patches greater than 100 km^2, and all habitat patches shared by two or more reporting units as high priority for conservation and/or ecological restoration actions.

Results

Land-cover change

Proportions

Proportions of the grouped land-cover themes stayed fairly consistent between 1992 and 2018, with the exception of wetlands, which decreased from 1.2% of the study area down to 0.4% by 2018 (Table 2.2). This decline was seen in the Québec portion (from 1.2% down to 0.1%) and in the New York portion (from 1.4% down to 0.7%). In 2018, the average proportion of wetlands within each MRC/county was 0.4% (Table S2.2). Proportions of the grouped land-cover themes were not equivalent between the provincial/state portions. While the composition of the study area was roughly 75% combined habitats and 25% natural and anthropogenic fragmentation elements, within the Québec and New York portions this ratio was roughly 80% combined habitats and 20% natural and anthropogenic fragmentation elements, while in the Ontario portion this ratio was lower: 57% combined habitats and 43% natural and anthropogenic fragmentation elements (Table 2.2).

Land-Cover Theme	Area (Km²)				Proportion (%)				Land-Cover Change		
									Area (Km²)	Area (%)	Proportion (%)
Study Area	**1992**	**2000**	**2010**	**2018**	**1992**	**2000**	**2010**	**2018**			
Natural and Anthropogenic Fragmentation Elements	38683	39611	40416	40967	22.8	23.3	23.8	24.1	2284	5.9	1.4
Forests	121703	120512	121001	120340	71.7	71.0	71.3	70.9	-1363	-1.1	-0.8
Non-Forest Vegetation	7413	7643	7742	7857	4.4	4.5	4.6	4.6	444	6.0	0.3
Wetlands	1982	2015	622	617	1.2	1.2	0.4	0.4	-1365	-68.9	-0.8
Combined Habitats	131098	130170	129365	128813	77.2	76.7	76.2	75.9	-2284	-1.7	-1.3
Québec Portion											
Natural and Anthropogenic Fragmentation Elements	15400	15691	16015	16095	19.1	19.5	19.9	20.0	695	4.5	0.9
Forests	63858	63559	64103	63948	79.2	78.8	79.5	79.3	91	0.1	0.1
Non-Forest Vegetation	415	422	427	501	0.5	0.5	0.5	0.6	86	20.7	0.1
Wetlands	985	987	113	113	1.2	1.2	0.1	0.1	-872	-88.5	-1.1
Combined Habitats	65258	64967	64643	64563	80.9	80.5	80.1	80.0	-695	-1.1	-0.9
Ontario Portion											
Natural and Anthropogenic Fragmentation Elements	10958	11286	11456	11616	53.4	55.0	55.9	56.6	659	6.0	3.2
Forests	9350	9005	8878	8714	45.6	43.9	43.3	42.5	-637	-6.8	-3.1
Non-Forest Vegetation	132	132	129	130	0.6	0.6	0.6	0.6	-2	-1.0	0.0
Wetlands	67	83	43	47	0.3	0.4	0.2	0.2	-20	-30.5	-0.1
Combined Habitats	9549	9221	9050	8890	46.6	45.0	44.1	43.4	-659	-6.9	-3.2
New York Portion											
Natural and Anthropogenic Fragmentation Elements	12355	12666	12976	13287	18.0	18.4	18.9	19.4	932	7.5	1.4
Forests	48497	47949	48020	47678	70.6	69.8	69.9	69.5	-819	-1.7	-1.2
Non-Forest Vegetation	6866	7088	7187	7225	10.0	10.3	10.5	10.5	359	5.2	0.5
Wetlands	931	946	466	458	1.4	1.4	0.7	0.7	-472	-50.8	-0.7
Combined Habitats	56294	55983	55672	55362	82.0	81.6	81.1	80.6	-932	-1.7	-1.4

Table 2.2. Changes in land-cover area (km^2) and proportion (%) for each grouped land-cover theme between 1992 and 2018, at the scale of the study area and each provincial/state portion. Red text (>50% change), areas of high priority for conservation and ecological restoration.

Area

At the level of the study area, natural and anthropogenic fragmentation elements (i.e., development, barren areas, waterbodies, and agricultural lands) increased by 2284 km^2 between 1992 and 2018, (Table 2.2) with increases of 695 km^2 in the Québec portion, 659 km^2 in the Ontario portion, and 932 km^2 in the New York portion (Table 2.2); and in 42 of the 43 MRCs/counties (Table S2.2; Fig. 2.2 and S2.1). Agricultural lands increased by 964 km^2, with a net loss of 57 km^2 in the Québec portion, and net gains of 434 km^2 and 588 km^2 in the Ontario and New York portions, respectively) (Table S2.3). Forests decreased by 1363 km^2, with losses of 637 km^2 in the Ontario portion and 819 km^2 in the New York portion (Table 2.2), and declines in 34 of the 43 MRCs/counties (Table S2.2; Fig. 2.2 and S2.1). Non-forest vegetation (i.e., grassland, shrub, moss, and herbaceous land-cover types) increased by 444 km^2, with increases of 86 km^2 in the Ontario portion, and 359 km^2 in the New York portion (Table 2.2), and increases in 27 of the 43 MRCs/counties (Table S2.2; Fig. 2.2 and S2.1). Wetlands experienced an overall loss of 1365 km^2 (68.9%), with losses of 871.9 km^2 (88.5%) in the Québec portion, 20.4 km^2 (30.5%) in the Ontario portion, and 472.4 km^2 (50.8%) in the New York portion (Table 2.2), with 19 MRCs/counties losing more than 50%, 13 losing more than 75%, and 8 losing more than 90% of their wetlands since 1992 (Table S2.2; Fig. 2.2 and S2.1). Natural and anthropogenic fragmentation elements and non-forest vegetation experienced the greatest increases across the study area, while forests and wetlands suffered the greatest declines. While forest loss was gradual between 1992 and 2018, wetland loss occurred rapidly between 2000 and 2010, with the vast majority occurring in 5 MRCs/counties: La Vallée-de-la-Gatineau (− 269.5 km^2); Antoine-Labelle (− 268.8 km^2); Hamilton (− 249.3 km^2); Matawinie (− 187.8 km^2); and Herkimer (− 111.1 km^2) (Figs. 2.2 and S2.1).

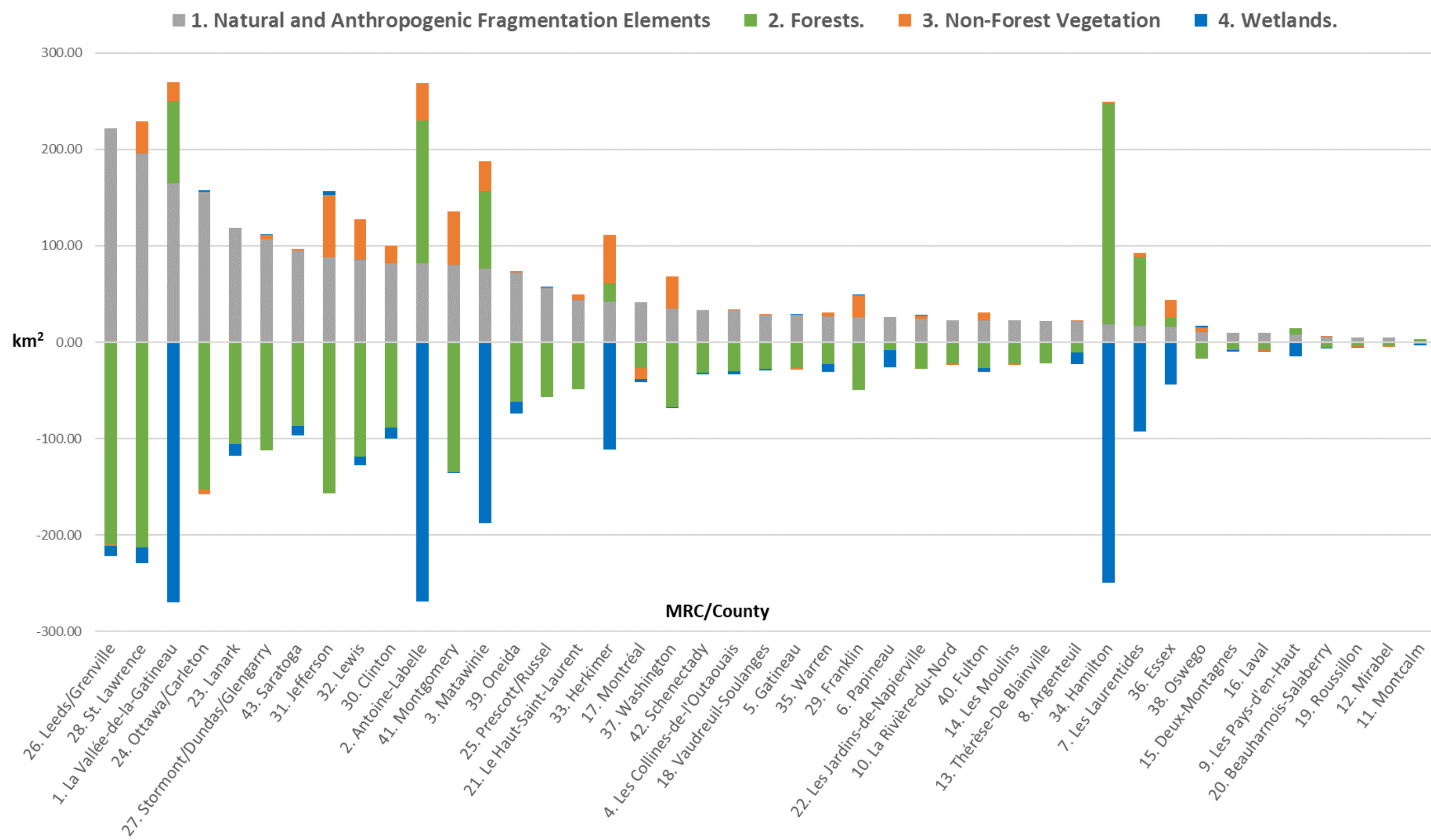

Figure 2.2 Changes in area (km^2) of the grouped land-cover themes in each MRC/county between 1992 and 2018. From left to right, MRC/counties are ranked from greatest to least amount of areal change in natural and anthropogenic fragmentation elements. MRC/county names are numbered and correspond to the numbers on the map in Figure 2.1.

Landscape fragmentation

Patch number and mean patch size

For the fragmentation geometries FG-forests, FG-wetlands, and FG-combined habitats, patch numbers increased, and mean patch size decreased between 2010 and 2018 indicating that landscape fragmentation had occurred (Table 2.3). For FG-non-forest vegetation, patch number decreased, and mean patch size increased, signifying growth in the fragmentation geometry similar to the growth seen in the non-forest vegetation land-cover theme (Tables 2.2 and 2.3).

In 2018, FG-forests were made up of 56,760 patches (Table 2.4). Of these patches, 49,910 were less than 1 km^2 (covering an area of 4455 km^2); 985 were greater than 10 km^2 (66,447 km^2); 99 were greater than 100 km^2 (42,998 km^2), with 55 located in Québec, 43 in New York, and 1 in Ontario; 13 were greater than 500 km^2 (28,347 km^2), with 5 in Québec, and 8 in New York; 7 were greater than 1000 km^2 (24,511 km^2), with 4 in Québec, and 3 in New York; 2 were greater than 5000 km^2 (16,445 km^2 , both in Québec) (Table 2.4; Fig. 2.3). FG-non-forest vegetation were made up of 32,145 patches (an area of 5986 km^2). Of these, 4 were greater than 10 km^2 (46 km^2), with 1 in Québec and 3 in New York (Table 2.4; Fig. 2.3). FG-wetlands were made up of 3716 patches (an area of 468 km^2). Of these, 61 were greater than 1 km^2 (127 km^2), with 11 in Québec, 2 in Ontario, and 47 in New York (Table 2.4; Fig. 2.3). Consequently, the land area of FG-combined habitats comprised 75.5% of the landscape (96,161 km^2) and was made up of 67,790 patches (Table 2.4; Fig. 2.3).

Effective mesh size

The effective mesh size ($m_{\text{eff\_CUT}}$), used to measure fragmentation strictly within the boundaries of the reporting units, decreased between 2010 and 2018 for each of the fragmentation

Reporting Unit	Year	FG - Forests		FG - Non-Forest Vegetation		FG - Wetlands		FG - Combined Habitats	
		Patch Number	Mean Patch Size (km²)	Patch Number	Mean Patch Size (km²)	Patch Number	Mean Patch Size (km²)	Patch Number	Mean Patch Size (km²)
Study Area	2010	53981	1.67	40933	0.172	3598	0.131	64114	1.51
	2018	56760	1.58	32145	0.186	3716	0.126	67790	1.42
Change		**2779**	**-0.09**	**-8788**	**0.014**	**118**	**-0.005**	**3676**	**-0.09**
Change (%)		**5.1**	**-5.4**	**-21.5**	**8.3**	**3.3**	**-4.0**	**5.7**	**-5.8**
Québec Portion	2000	17564	2.62	3571	0.085	2054	0.348	18420	2.57
	2010	19654	2.36	6573	0.088	752	0.110	20715	2.27
	2018	20802	2.23	3731	0.096	751	0.110	21779	2.16
Change		**3238**	**-0.40**	**160**	**0.012**	**-1303**	**-0.238**	**3359**	**-0.41**
Change (%)		**18.4**	**-15.1**	**4.5**	**13.6**	**-63.4**	**-68.4**	**18.2**	**-16.0**
Ontario Portion	2000	8440	0.80	1259	0.078	466	0.134	9112	0.76
	2010	9199	0.72	2147	0.060	329	0.099	9910	0.69
	2018	9548	0.68	1396	0.069	377	0.093	10261	0.65
Change		**1108**	**-0.12**	**137**	**-0.009**	**-89**	**-0.042**	**1149**	**-0.11**
Change (%)		**13.1**	**-14.5**	**10.9**	**-11.4**	**-19.1**	**-30.9**	**12.6**	**-14.4**
New York Portion	2010	25166	1.47	32230	0.196	2530	0.141	33537	1.28
	2018	26452	1.39	27032	0.205	2604	0.135	35805	1.19
Change		**1286**	**-0.08**	**-5198**	**0.008**	**74**	**-0.006**	**2268**	**-0.09**
Change (%)		**5.1**	**-5.5**	**-16.1**	**4.2**	**2.9**	**-4.6**	**6.8**	**-6.9**

Table 2.3 Changes in patch number and mean patch size (km^2) for each fragmentation geometry between 2000 and 2018, at the scale of the study area and each provincial/state portion. Yellow text (10% - 30% change), areas of medium priority for conservation and ecological restoration; orange text (30% - 50% change), areas of medium/high priority; and red text (>50% change), areas of high priority for conservation and ecological restoration.

Fragmentation Geometry	Proportion of Study Area (%)	Number of Remaining Patches							
		Total	<1 km^2	>1 km^2	>10 km^2	>100 km^2	>500 km^2	>1000 km^2	>5000 km^2
Study Area									
FG-Forests	70.3	56760	49910	6850	985	99	13	7	2
FG-Non-Forest Vegetation	4.7	32145	30840	1305	4	0	0	0	0
FG-Wetlands	0.4	3716	3655	61	0	0	0	0	0
FG-Combined Habitats	75.5	67790	59589	8201	1047	100	13	7	2
Québec Portion									
FG-Forests	78.3	20802	17614	1929	474	55	5	4	2
FG-Non-Forest Vegetation	0.6	3721	3694	77	1	0	0	0	0
FG-Wetlands	0.1	751	740	11	0	0	0	0	0
FG-Combined Habitats	80.3	21779	19832	1947	475	55	5	4	2
Ontario Portion									
FG-Forests	42.3	9549	8373	1175	111	1	0	0	0
FG-Non-Forest Vegetation	0.6	1396	1386	10	0	0	0	0	0
FG-Wetlands	0.2	377	375	2	0	0	0	0	0
FG-Combined Habitats	43.2	10261	9072	1189	114	1	0	0	0
New York Portion									
FG-Forests	69.0	26452	22697	3755	397	43	8	3	0
FG-Non-Forest Vegetation	10.4	27032	25774	1258	3	0	0	0	0
FG-Wetlands	0.7	2604	2557	47	0	0	0	0	0
FG-Combined Habitats	80.1	35805	30729	5076	454	44	8	3	0

Table 2.4. Relative proportion of each fragmentation geometry and size distribution of remaining patches in 2018, at the scale of the study area and each provincial/state portion.

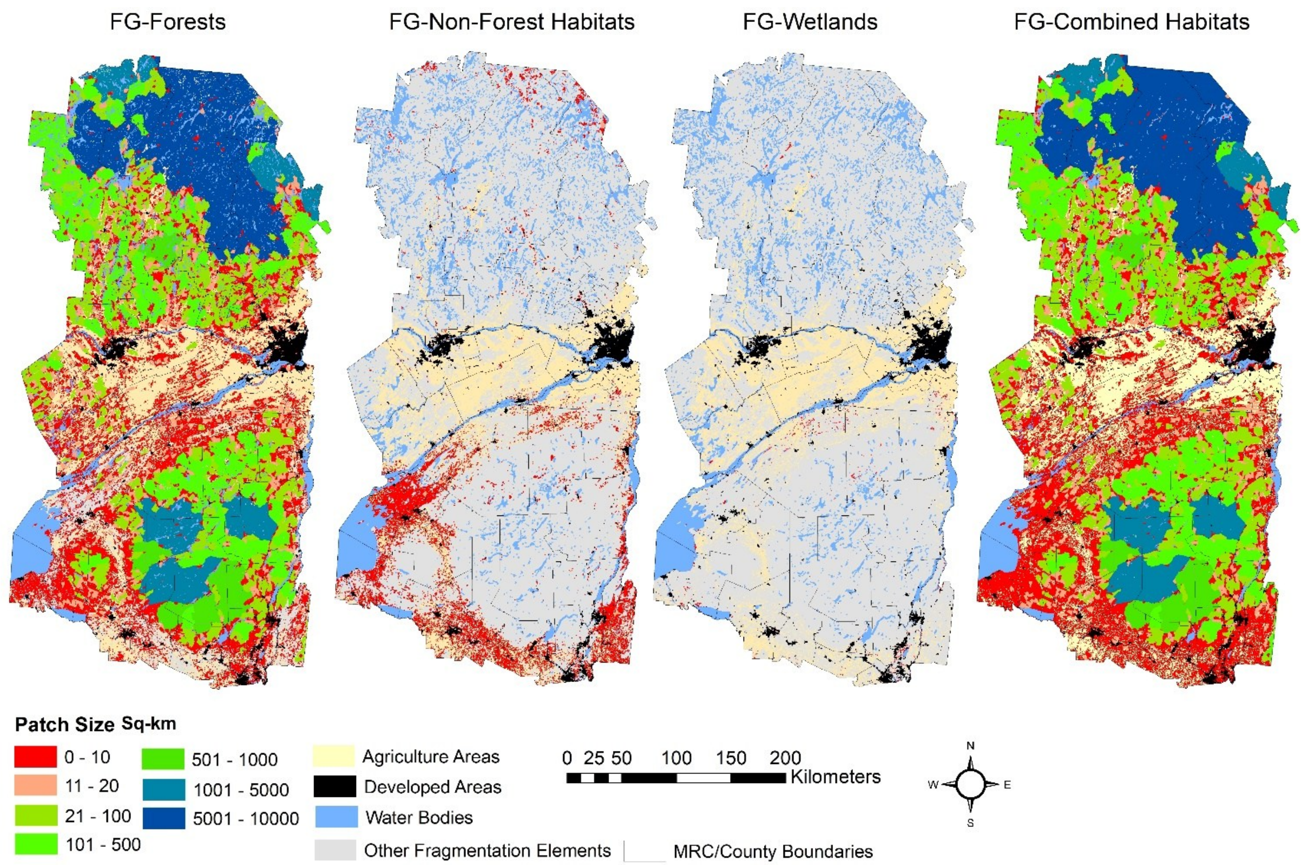

Figure 2.3. Size distribution of remaining patches for each fragmentation geometry in 2018. Municipalité régionale de comté (MRC)/county boundaries overlaid on each map.

geometries, indicating that landscape fragmentation had occurred (Table 2.5). Within the A2L, $m_{\text{eff\_CUT}}$ values ranged from 0.0036 km^2 (FG-wetlands) to 1468.8 km^2 (FG-combined habitats) in 2010; and from 0.0035 km^2 (FG-wetlands) to 1235.9 km^2 (FG-combined habitats) in 2018 (Table 2.5). Between 2000 and 2018, $m_{\text{eff\_CUT}}$ for FG-combined habitats decreased by 3726.1 km^2 (60.4%) within the Québec portion (Table 5). For FG-forests, FG-wetlands, and FG-combined habitats, the majority of fragmentation took place in the Québec portion of the study area, whereas for FG-non-forest vegetation, the majority of fragmentation occurred in the New York portion (Table 2.5). This pattern was also observed at the level of the MRC/county where the mean $m_{\text{eff\_CUT}}$ decreased for each of the fragmentation geometries (Table S2.4–2.7). The mean $m_{\text{eff\_CBC}}$, used to measure fragmentation that considered patches that "crossed" reporting unit boundaries, also decreased for each of the fragmentation geometries in the MRCs/counties (Table S2.4–2.7); and the mean $m_{\text{eff\_CBC}}$ - $m_{\text{eff\_CUT}}$, decreased for all of the fragmentation geometries indicating that the area of patches shared by multiple MRCs/counties also decreased between 2010 and 2018 (Table S2.4–2.7). In 2018, the lowest effective mesh values (highest fragmentation) for FG-forests were located in the MRCs/counties of the central region of the study area and along the west edge of the New York portion (Figure S2.2). The lowest effective mesh values for FG-non-forest vegetation were located in the MRCs/counties just north of the central region with the highest values (lowest fragmentation) occurring in the west edge of the New York portion (Figure S2.2). FG-wetlands had a very similar pattern to FG–non-forest vegetation, whereas FG-combined habitats had a near identical pattern to FG-forests (Figure S2.2).

Reporting Unit	Year	m_{eff_CUT} (km^2)			
		FG - Forests	FG - Non-Forest Vegetation	FG - Wetlands	FG - Combined Habitats
Study Area	2010	1428.7	0.092	0.0036	1468.8
	2018	1173.9	0.080	0.0035	1235.9
Change		**-254.8**	**-0.012**	**-0.0001**	**-232.9**
Change (%)		**-17.8**	**-12.9**	**-2.1**	**-15.9**
Québec Portion	2000	5572.5	0.003	0.0172	6167.9
	2010	2851.5	0.009	0.0015	2936.5
	2018	2310.0	0.007	0.0015	2441.8
Change		**-3262.5**	**0.005**	**-0.0156**	**-3726.1**
Change (%)		**-58.5**	**185.7**	**-91.0**	**-60.4**
Ontario Portion	2000	7.1	0.002	0.0031	7.3
	2010	6.4	0.003	0.0021	6.6
	2018	6.2	0.002	0.0021	6.4
Change		**-0.9**	**0.000**	**-0.0010**	**-0.9**
Change (%)		**-12.7**	**-3.9**	**-31.2**	**-12.3**
New York Portion	2010	264.8	0.210	0.0063	266.8
	2018	254.0	0.184	0.0061	256.4
Change		**-10.8**	**-0.027**	**-0.0002**	**-10.4**
Change (%)		**-4.1**	**-12.6**	**-3.2**	**-3.9**

Table 2.5. Changes in the Effective Mesh Size (m_{eff_CUT}) for each fragmentation geometry between 2000 and 2018, at the scale of the study area and each provincial/state portion. Yellow text (10% - 30% change), areas of medium priority for conservation and ecological restoration; orange text (30% - 50% change), areas of medium/high priority; and red text (>50% change), areas of high priority for conservation and ecological restoration.

Road Network	Road Length (km^2)				Road Density (km/km^2)				Change (%)
Study Area	**2000**	**2010**	**2018**	**Change**	**2000**	**2010**	**2018**	**Change**	
Primary roads (10m buffer)		4127	4566	439		0.02	0.03	0.003	10.6
Secondary roads (5m buffer)		20222	20772	551		0.12	0.12	0.003	2.7
Tertiary roads (3m buffer)		106112	107710	1598		0.63	0.64	0.009	1.5
Total		**130460**	**133048**	**2588**		**0.77**	**0.79**	**0.015**	**2.0**
Québec									
Primary roads (10m buffer)	1661	1783	2134	473	0.02	0.02	0.03	0.006	28.5
Secondary roads (5m buffer)	5859	6275	6442	583	0.07	0.08	0.08	0.007	10.0
Tertiary roads (3m buffer)	41546	46917	48174	6628	0.52	0.58	0.60	0.082	16.0
Total	**49066**	**54975**	**56750**	**7684**	**0.61**	**0.68**	**0.70**	**0.095**	**15.7**
Ontario									
Primary roads (10m buffer)	1077	1147	1218	141	0.05	0.06	0.06	0.007	13.1
Secondary roads (5m buffer)	4731	4829	4862	131	0.23	0.24	0.24	0.006	2.8
Tertiary roads (3m buffer)	14682	16450	16791	2109	0.72	0.80	0.82	0.103	14.3
Total	**20491**	**22426**	**22871**	**2380**	**1.00**	**1.09**	**1.12**	**0.116**	**11.6**
New York									
Primary roads (10m buffer)		1197	1213	16		0.02	0.02	0.000	1.4
Secondary roads (5m buffer)		9118	9468	350		0.13	0.14	0.005	3.8
Tertiary roads (3m buffer)		42745	42747	2		0.62	0.62	0.000	0.0
Total		**53059**	**53428**	**369**		**0.77**	**0.78**	**0.005**	**0.7**

Table 2.6. Changes in road length (km) and road density (km/km^2) for each road category between 2000 and 2018, at the scale of the study area and each provincial/state portion. Yellow text (10% - 30% change), areas of medium priority for conservation and ecological restoration.

Road density

Between 2010 and 2018, the length of the road network increased by 2588 km within the study area (Table 2.6). Primary roads increased by 439 km (10.6%), secondary roads increased by 551 km, and tertiary roads increased by 1598 km. These increases were spread out between the provincial/state portions. Since 2000, roads in the Québec portion expanded by 7684 km (15.7%), with primary roads increasing by 473 km (28.5%). Ontario roads expanded by 2380 km (11.6%), and New York roads increased by 369 km between 2010 and 2018. Accordingly, road density also increased throughout the study area between 2010 and 2018 (Table 2.6). As of 2018, the Ontario portion had the highest road density with 1.12 km/ km^2, followed by the New York portion with 0.78 km/km^2, and the Québec portion with 0.70 km/km^2. These increases were distributed across 39 of the 43 MRCs/counties with a mean road length increase of 60 km, and a mean road density increase of 0.04 km/ km^2 (Table S2.8).

Priority areas for conservation and ecological restoration

For forest losses between 1992 and 2018, there were 6 MRCs/counties at medium priority (10%–30% change) and 2 at medium–high priority (30%–50%) for conservation and ecological restoration intervention (Table S2.2); for non-forest vegetation losses, there were 4 MRCs/counties at medium priority, 1 at medium–high priority, and 2 at high priority (>50% change) (Table S2.2); for wetland losses there were 7 at medium, 2 at medium–high, and 19 at high priority (Table S2.2); and for combined habitat losses, 5 MRCs/ counties were at medium, and 2 were at medium–high priority (Table S2.2). There were also 11 MRCs/counties that had less than 30% combined habitats remaining, these were also given high priority for conservation and ecological restoration actions (Table S2.2).

For FG-forest fragmentation between 2000/2010 and 2018, there were 4 MRCs/counties at medium priority and 3 at medium–high priority for conservation and ecological restoration intervention when measured by $m_{\text{eff\_CUT}}$ (Table S2.4); and 8 at medium priority and 2 at medium–high priority when measured by $m_{\text{eff\_CBC}}$ (Table S2.4). For FG-non-forest vegetation fragmentation, there were 12 MRCs/counties at medium priority, 10 at medium–high priority, and 11 at high priority, as measured by both $m_{\text{eff\_CUT}}$ and $m_{\text{eff\_CBC}}$ (Table S2.5).For FG-wetlands fragmentation, there were 6 MRCs/counties at medium priority and 1 at medium–high priority when measured by $m_{\text{eff\_CUT}}$ (Table S2.6); and 7 at medium priority and 1 at medium–high priority when measured by $m_{\text{eff\_CBC}}$ (Table S2.6). For FG-combined habitats fragmentation, there were 5

MRCs/counties at medium priority and 3 at medium–high priority when measured by $m_{\text{eff\_CUT}}$ (Table S2.7); and 7 at medium priority and 2 at medium–high priority, when measured by $m_{\text{eff\_CBC}}$ (Table S2.7).

These MRCs/counties represent areas of medium priority, medium–high priority, and high priority within the A2L where continued monitoring, planning, and conservation and restoration actions are required to ensure habitats are restored and no further land-cover change and landscape fragmentation continues.

Discussion

Land-cover change and landscape fragmentation

Although there have been several "static" studies of the extent of landscape connectivity within the larger region, such as the Algonquin-to-Adirondacks (A2A) region (Quinby et al. 1999), Southeastern Canada/ Northeastern USA (Carroll 2003), and Montréal and the Saint Lawrence Lowlands (Mitchell et al. 2015; Albert et al. 2017; Rayfield et al. 2019; Gonzalez et al. 2019), this is the first "temporal" study of changes in landscape structure within one of the three potential north–south transboundary wildlife movement linkages that connect wilderness areas in northeastern USA and southeastern Canada.

Our results clearly show that extensive land-cover change and landscape fragmentation have occurred within the A2L between 1992 and 2018. These findings are in agreement with a proximal study of the Montréal Metropolitan Region (MMR) by Dupras et al. (2016), who reported that *"land-use changes which occurred in the MMR between 1966 and 2010 have in turn caused profound changes on both the structural (landscape patterns such as fragmentation)* *and functional (landscape processes such as barrier effects and ecological connectivity) properties of the landscape"* (p. 69).

Changes in land-cover varied between the grouped land-cover themes. Natural and anthropogenic fragmentation elements and non-forest vegetation experienced net increases in land-cover area, whereas forests and wetlands suffered net declines in land-cover area between 1992 and 2018 (Table 2). This pattern is striking at the MRC/county level where one can visualize how gains in natural and anthropogenic fragmentation elements, forests, and non-forest vegetation were the direct result of losses in forests, wetlands, or both (Fig. 2). These losses could be the result of wetland drainage. Wetland drainage for development and agriculture is the leading cause of wetland loss in Canada (Council of Canadian Academies 2013). Gains in forests and non-forest vegetation could indicate areas of "dried" wetlands (i.e., seasonally flooded forests, wooded swamps, and marshes), after the draining process. Gains in non-forest vegetation could also be the result of forest harvest. After forest harvesting, these areas would be in various stages of succession and constitute non-forest vegetation types (i.e., grassland, shrub, moss, and herbaceous cover types).

Landscape fragmentation, however, occurred within all fragmentation geometries (with FG–forests undergoing the greatest amount of fragmentation), as measured by the effective mesh size and road density. Patch number and mean patch size indicated fragmentation took place in FG-forests, FG-wetlands, and FG-combined habitats, but not in FG-non-forest vegetation, which was due to the loss of 8788 patches (Table 2.3) causing an overall increase in the mean patch size. Mean patch size can increase when small patches are lost due to habitat loss, even when landscape fragmentation has occurred (Jaeger et al. 2011). Consequently, mean patch size is not a suitable metric for landscape fragmentation on its own and is only valuable when used in combination with more appropriate metrics such as the effective mesh size (Jaeger 2000).

Priority areas for conservation and ecological restoration

We directly compared reporting units as well as ranked them in terms of land-cover change and landscape fragmentation. MRCs/counties with the lowest proportion of potential habitat (for each grouped land-cover theme) and/or the lowest effective mesh size for each fragmentation geometry (highest fragmentation) were identified (Tables S2.2, and S2.4–2.7). Reporting units were also prioritized for conservation and/or restoration intervention. Many MRCs/counties have reached or exceeded thresholds of habitat loss and fragmentation (see below) and to endure further changes in landscape structure would significantly jeopardize the overall integrity and connectivity of the transboundary wildlife linkage. These MRCs/counties should be given the highest priority for conservation and restoration actions within the A2L to ensure the functionality of the transboundary wildlife linkage.

The implications for land-use planning are clear. Development in these locations should be implemented strategically to avoid further habitat loss and fragmentation. Such tactics include: (1) limiting the area of urban and agricultural development, while promoting "up instead of out" development practices, salvaging brownfield sites, and adopting agro-ecological diversification techniques (Jaeger et al. 2011; Kremen and Merenlender 2018); (2) addition of "greenbelts" surrounding urban areas which have been shown to significantly reduce urban sprawl as well as provide habitat and maintain landscape connectivity (Pourtaherian and Jaeger 2022); (3) addition of wildlife crossing structures (WCS) to restore landscape connectivity; (4) preference to upgrading and widening of existing highways over construction of new highways at additional locations (Jaeger et al. 2011); and (5) bundling of transportation infrastructure (i.e., constructing roads and railways in parallel). Although these last two strategies will increase the barrier effect of each individual transportation route, they are still considered better options than the fragmentation of a much larger area; especially if WCSs can be placed strategically along the widened/bundled infrastructures so that they can be traversed all at once (Jaeger et al. 2011).

Recommendations

2020 conservation targets

In 2015, federal, provincial, and territorial governments established the "2020 Biodiversity Goals and Targets for Canada" to achieve its commitments to the United Nations Convention on Biological Diversity (CBD) "Strategic Plan for Biodiversity 2011–2020" and its global "Aichi Biodiversity Targets" (Environment and Climate Change Canada 2019). Target 1 declared that by 2020, at least 17% of terrestrial areas and inland water, and 10% of coastal and marine areas would be conserved through a network of protected areas and other conservation measures (Environment and Climate Change Canada 2019). The USA also signed the strategic plan for biodiversity; however, it was never ratified (CBD 2021). In 2016 the New England Governors and Eastern Canadian Premiers adopted "Resolution 40–3—Resolution on ecological connectivity, adaptation to climate change, and biodiversity conservation" (CICS 2016; Arkilanian et al. 2020). The objectives highlighted the necessity for its partners to work across landscapes and borders to restore and maintain ecological connectivity and for all levels of governance, especially municipalities, to incorporate habitat connectivity objectives into their regional land-use plans and policies (CICS 2016).

By the end of 2020, only 10.2% of Canadian terrestrial areas and 11.8% of USA terrestrial areas were under some level of protection (UNEP-WCMC, 2021a/b). Nevertheless, the province of Québec reached 17% (~257,000 km^2) of its terrestrial area protected (Environment and Climate Change Canada 2020), as did New York State, with approximately 17% (~24,000 km^2) of its terrestrial area protected (New York Protected Areas Database 2020), whereas Ontario achieved only 10.7% (115,593 km^2) of its terrestrial areas protected by the end 2020 (Ontario 2022). At the global level, none of the 20 Aichi Biodiversity Targets agreed by Parties to the CBD in 2010 have been fully achieved (IUCN 2022).

Post-2020 conservation targets

In 2019, the Trudeau government pledged to protect 25% of Canada's land and oceans by 2025 and 30% by 2030 (One Planet Summit 2021). In 2021, the Biden administration also committed to conserving at least 30% of U.S. lands and waters by 2030 (The White House 2021). In December 2022, members of the CBD will meet in Montréal, Canada for the 15th meeting of the Conference of the Parties (COP 15) to adopt a "Post-2020 Global Biodiversity Framework" which will act as a stepping-stone towards the 2050 vision of "living in harmony with nature" (IUCN 2022). Parties to the CBD aim to halt the loss of biodiversity by 2030 and achieve recovery and restoration by 2050 (IUCN 2022). The post-2020 strategy includes the expansion of protected areas and other effective area-based conservation measures (OECMs) to cover at least 30% of the planet by 2030 ("30×30"), while recognizing the rights and roles of Indigenous peoples and local communities (IUCN 2022). With conservation outcomes often conditional to decisions made across multiple boundaries, a key component in the post-2020 framework is a commitment to coordinated and collaborative international conservation at the transboundary level (SCBD 2018; Díaz et al. 2020; Mason et al. 2020).

Based on our findings, the following sub-sections propose recommendations for conservation at the local level (MRC/county) that will complement post-2020 commitments at the provincial/state, national, and transboundary levels (i.e., "Think globally, act locally").

Minimum 30% combined habitats

Although the proportion of remaining combined habitats (forests, grassland, shrub, moss, herbaceous cover, and wetlands) in the A2L was 75.9% in 2018 (Table 2.2), at the MRC/county level, this proportion ranged from 7.1% (MRC Beauharnois-Salaberry) to 99.7% (Hamilton County), with 11 of the 43 MRCs/counties having less than 30% combined habitats remaining within their borders (Table S2.2). Studies suggest that to conserve biodiversity and meet the area requirements for large-ranging species, up to 75% combined habitats within a landscape should be protected (Noss et al. 2012; Lovejoy and Nobre 2018; Mogg et al. 2019). While this value is dependent on a variety of landscape and species-specific characteristics (i.e., size of landscape, rate of habitat loss, degree of fragmentation, landscape connectivity, and matrix quality), simulation and empirical studies have suggested that with less than 30% habitat remaining, the ecological effects of habitat loss and fragmentation (including species richness and abundance) begin to increase exponentially and population extinctions become increasingly inevitable (Andrén 1994; Swift and Hannon 2010).

Increasing the number and total amount of protected areas has, thus far, been the most important tool for the conservation of biodiversity (at the provincial/state and national levels). However, many protected areas are simply not large enough to support viable populations of species with large home ranges nor do they include the range of species, processes, and habitats necessary to fully conserve ecosystem integrity and biodiversity (Boyd et al. 2008; Pimm et al. 2014). Because the remaining natural and semi-natural areas between protected areas are vulnerable to continued habitat loss and fragmentation, it is only a matter of time before protected areas become islands in a sea of human modified landscape (Wilson and MacArthur 1967).

According to the principle of subsidiarity, management issues should be dealt with at the most proximal level that is competent of resolution (Jefferies and Sawyer 2019). One such solution is to establish area-based conservation targets at the level of the MRC/county to help achieve federal and provincial biodiversity conservation objectives and ensure connectivity between protected areas. MRCs/counties are the primary planners of regional land-use. They are well positioned to assess local ecosystems and develop area-based conservation and restoration plans to protect biodiversity and maintain connectivity with surrounding MRCs/counties (Jefferies and Sawyer 2019). Goal A of the "2020 Biodiversity Goals and Targets for Canada" states that by 2020 Canada's lands and waters will be managed using an ecosystem approach to support biodiversity conservation outcomes at local, regional, and national scales; and Target 4 states that by 2020 biodiversity considerations will be integrated into municipal planning (Environment and Climate Change Canada 2019); however, neither goal nor target established area-based objectives for conservation at the MRC/county level. Similarly, there are no such area-based targets for conservation at the county level in the USA either. Therefore, in most cases, it is up to the discretion of the MRCs/counties themselves to implement any area-based conservation and restoration targets.

Accordingly, we recommend maintaining the A2L at, or above, 75% combined habitats, and ecologically restoring combined habitats to a "minimum" of 30% land area in MRCs/counties where they are already below this threshold. These restoration actions will offer additional habitats and resources, improve landscape connectivity by providing corridors and stepping-stones, and increase the overall integrity of the transboundary wildlife linkage (Kremen and Merenlender 2018; Locke et al. 2019; Garibaldi et al. 2020).

Wetland conservation and restoration

North America is home to 30% of the world's wetlands with 25% (~1.3 million km^2) solely in Canada (Government of Canada 2016a). In the last 200 years, Canada has lost 15% (~200,000 km^2) of its wetland ecosystems, while the U.S. have lost 53% (~473,000 km^2) (Dahl 1990). In 1981, Canada signed the "Ramsar Convention on Wetlands of International Importance", which was followed by the U.S. in 1986 (RCS 2016). Through international cooperation, policy creation, and technology transfer, the Ramsar Convention's aim is to halt the worldwide loss of wetlands and to conserve those that remain (RCS 2016). In 1986, the Canadian and U.S governments established the "North American Waterfowl Management Plan" to conserve declining waterfowl and migratory bird habitats in North America (Government of Canada 2016b); and in 1989, the U.S. Congress passed the "North American Wetlands Conservation Act", which authorizes grants to public–private partnerships in Canada, Mexico, and the United States, to protect, enhance, and/or restore, wetland ecosystems, consistent with the North American Waterfowl Management Plan (USFW 2022). Despite these global and international obligations, wetlands have still declined by 68.9% within the A2L since 1992 (Table 2.2).

In 1975, the New York State Legislature passed "The Freshwater Wetlands Act" with the intent to protect freshwater wetlands and their ecosystem benefits (DEC 2022). Regardless, wetlands within the New York portion have declined by 472 km^2 (50.8%) since 1992 (Table 2.2). In the same timeframe, the Québec portion experienced a critical loss of 872 km^2 (88.5%), and the Ontario portion lost 20 km^2 (30.5%) of wetland habitats (Table 2.2).

At the regional level, 19 of the 43 MRCs/counties lost more than 50%, 12 lost more than 80%, and 8 lost more than 90% of their wetlands since 1992 (Table S2.2). Consequently, in 2017, the National Assembly of Québec passed "An act respecting the conservation of wetlands and bodies of water" (Bill 132). This set of legislation, which includes a no-net-loss principle for both wetlands and bodies of water, affords the MRCs the responsibility of developing and implementing a "plan régional des milieux humides et hydriques (PRMHH)", a regional conservation and restoration plan for wetlands and water-bodies in their territories (Assemblée nationale du Québec 2017); and in 2021, the Ontario government invested $30 million in the "Wetlands Conservation Partner Program" to assist conservation organizations in conserving and restoring wetlands in priority areas across the province (Ontario 2022).

With such extensive wetland losses across the A2L, a no-net-loss policy is simply not enough. To ensure an abundance of wetland habitat for both local and migratory species, to safeguard the ecosystem services they provide (i.e., water purification, food and erosion control, groundwater recharge, etc.), to generate connectivity between wetland habitats, and to preserve ecosystem integrity and productivity, wetland losses need to be recovered. As a result, we recommend that all provincial/state wetland policies be based on a net gain of area (extent and quality of wetland habitats), and function (ecosystem services).

Species-appropriate effective mesh sizes

The more barriers fragmenting the landscape, the lower the effective mesh size (Jaeger et al. 2007b). In 2018, the effective mesh size $m_{\text{eff\_CUT}}$ for FG-combined habitats was 1235.9 km^2 within the A2L (Table 2.5). At the level of the MRC/county, this value ranged from 0.1 km^2 (Montréal and MRC Beauharnois-Salaberry) to 2863.6 km^2 (MRC Antoine-Labelle), with 14 out of 43 MRCs/counties with a $m_{\text{eff\_CUT}}$ size less than 2 km^2, and 13 with a $m_{\text{eff\_CBC}}$ size less than 2 km^2 (Table S2.7).

When the effective mesh size is smaller than the size of a species' home range then the likelihood decreases drastically that individuals of the species will be able to move freely in the landscape without encountering barriers (Jaeger et al. 2011). For example, an American black bear (*Ursus americanus*) has an average home range size of ~100 km^2 (Gantchoff et al., 2018), a fisher (*Pekania pennanti*) has an average home range size of ~8 km^2 (Koen et al., 2007), a white-tailed deer (*Odocoileus virginianus*) has an average home range size of ~2 km^2 (Nelson & Mech, 1981) and a moose (*Alces alces americana*) has an average home range size of ~40 km^2 (Murray et al., 2012). When effective mesh sizes are larger than the species with the largest home range size, all individuals have a higher probability of moving freely in the landscape without encountering barriers. We therefore recommend restoring landscape connectivity (i.e., reducing fragmentation) to accommodate effective mesh sizes that are appropriate for the species that inhabit the region or may move into (or through) the region following the transboundary wildlife linkage.

Protection of large roadless areas

Not only are roads a major contributor to habitat loss and fragmentation, their impacts on the surrounding landscape (described by the "road-effect zone"; Forman and Alexander 1998) can extend up to several kilometers from the road edge, reducing the quality of adjacent habitats (Benítez-López et al. 2010; Torres et al. 2016). For some species, roads can act as barriers to movement and lead to resource inaccessibility; for others, roads can cause increased mortality due to animal-vehicle collisions (Forman and Alexander 1998; Jaeger et al. 2005). Roads also facilitate "contagious development" by providing access to previously isolated areas (Laurance and Balmford 2013; Selva et al. 2015; Ibisch et al. 2016). Large roadless areas are characterized by high ecological value, integrity, and connectivity, making their safeguarding a significant contribution to the prevention of biodiversity loss (IENE 2015; Ibisch et al 2016). There is no legislation in place to protect the remaining large roadless areas in Canada. In the U.S., the "2001 Roadless Area Conservation Rule" established prohibitions on road construction, road reconstruction, and timber harvesting on 236,700 km^2 of inventoried roadless areas on "National Forest System" lands (IENE 2015; Coffin et al. 2021). However, since its inception, the roadless rule has been under threat from multiple states seeking their own special roadless rule exemptions. The "Roadless Area Conservation Act of 2021" (H.R.279; 117th Congress, 2020–2021), which has been introduced consecutively since 2018, would codify the protections provided by the 2001 roadless area conservation rule ensuring the protection of these public lands for future generations. As of November 2022, this act has not been passed by the U.S. Congress.

In 2018, there were only 100 large roadless areas (patches) of combined habitats (>43,000 km^2 in total area) remaining in the A2L (Fig. 2.3; Table 2.4), 13 of which were greater than 500 km^2 (5 in the Québec portion, 8 in the New York portion); 7 of which were greater than 1000 km^2 (4 in the Québec portion, 3 in the New York portion); and 2 larger than 5000 km^2 , both in the Québec portion. These locations represent the last large roadless areas within the A2L transboundary wildlife linkage and are vital to wide-ranging mammals and species vulnerable to habitat fragmentation. These large roadless areas should be given high priority for conservation within the A2L to ensure their persistence in the wildlife linkage.

Protection of border-crossing patches

In 2018, 36 of the 43 MRCs/counties shared combined habitat patches with at least one other MRC/county (i.e., patches crossing MRC/county boundaries in Fig. 2.3) as calculated by the difference between the $m_{\text{eff\_CUT}}$ and $m_{\text{eff\_CBC}}$ values (Table S2.7). This was also the case with the other land-cover themes with 34 out of 43 MRCs/counties sharing forest patches, 32/43 sharing non-forest vegetation patches, and 22/43 sharing wetland patches (Tables S2.4–S2.6; Fig. 2.3). Because of the importance of these transboundary patches for landscape connectivity and their disproportionate risk of being reduced or fragmented, we recommend coordinated and collaborative conservation strategies between MRCs/counties to ensure that these patches continue to serve as vital habitats, connectivity corridors, and stepping-stones for a wide range of species within the A2L.

Inclusion of Ontario and New York in resolution 40-3

Resolution 40-3—Resolution on ecological connectivity, adaptation to climate change, and biodiversity conservation - promotes regional collaborations in order to identify priority habitat corridors that connect and expand existing protected areas; as well as the design and/or modification of transportation infrastructure to improve habitat connectivity including reducing the risk of wildlife-vehicle collisions (CICS, 2016). The members of the New England Governors and Eastern Canadian Premiers (NEG/ECP) include Québec, New Brunswick, Prince Edward Island, Nova Scotia, Newfoundland and Labrador, Maine, Vermont, New Hampshire, Massachusetts, Rhode Island, and Connecticut (Arkilanian et al. 2020). Ontario and New York are not included in the NEG/ ECP. However, not only do they share the A2L, but they also share another potential north–south transboundary linkage, the “Algonquin-to-Adirondack” (A2A) linkage (Algonquin to Adirondacks Collaborative 2016) and a potential east–west linkage, the “Adirondack Mountains to the Green Mountains” linkage (Staying Connected Initiative 2022). Ontario and New York are also the last westward province and state to share a land border before the natural barrier of Lake Ontario (Fig. 2.1). Consequently, it would be advantageous for both Ontario and New York to join the NEG/ECP and adopt Resolution 40-3 to benefit from collaborations between transportation and natural resource agencies that aim to improve habitat connectivity (CICS, 2016).

Continued monitoring of land-cover change and landscape fragmentation

Monitoring is an important requirement for transboundary conservation (Vasilijević et al. 2015). Land-cover change and landscape fragmentation are essential indicators of threats to biodiversity, sustainable land-use, and landscape quality; and the distribution of conservation and restoration resources is dependent on the knowledge of ongoing trends in landscape structure (Jaeger et al. 2011). The data presented here provides valuable information for land-use, transportation, and conservation planning and can be used as a baseline to evaluate the impacts of future land-use development scenarios. By applying the same parameters, the effects of multiple projects can be compared and the least intrusive can be selected. This same logic can also be applied for the continuous monitoring of the region. Accordingly, we strongly recommend continued monitoring within the A2L utilizing the same grouped land-cover themes and fragmentation geometries. Doing so will not only enable the detection of long- and short-term changes in landscape structure but will also allow monitoring agencies to determine whether past conservation targets are being achieved (Roch and Jaeger 2014).

Limitations

Transboundary analysis involves the inherent challenge of gathering and working with GIS data from multiple jurisdictions (i.e., provinces/states, countries, etc.). This challenge is only exacerbated when the study involves multiple timepoints (i.e., time-series, dynamic analysis). Data not only need to be compatible between maps (i.e., format, resolution, attributes, etc.), but also consistent over time. Although there were a variety of recent (circa~2015) Canadian, American, and continental North American land-cover maps available at resolutions down to 15 m, none of these GIS datasets had equivalent datasets for past years; and different datasets from earlier timepoints had both attribute and resolution disparities with the most recent maps. Thus, we opted for the ESA-CCI-LC global land-cover dataset that was updated yearly and had consistent map resolution (300 m) and attributes (24 land-cover categories) throughout the entire time period (1992–2018). The only drawback was that the ESA-CCI-LC dataset did not include a separate roads network category. To compensate, we initially applied the Census Road Network files from Statistics Canada and the U.S. Census Bureau, however, we found that not only were they incompatible between countries (i.e., road classifications), but were also incompatible between timepoints within each dataset (i.e., we found inconsistencies (increases and decreases) in the length and density of the road network over time, within both datasets). As a result, we opted for commercial datasets from DMTI Spatial Inc. (Québec and Ontario) and New York State Information Technology Services (New York State). Although both datasets were highly compatible and consistent over time, the data for Québec and Ontario only went back to 2000, and the data for New York only back to 2010. Thus, the trade-off for accuracy was that we could only do the landscape fragmentation analysis (which required roads) between 2000/2010 and 2018. We ran into similar difficulties when we tried to add additional map layers used in traditional static land-cover analyses, such as population density, forest attributes (i.e., age, height, density, etc.), and agricultural attributes (i.e., hedgerows, wooded areas, wetlands, natural pastures, etc.). Nevertheless, by selecting accuracy and compatibility over complexity, we were able to measure changes in landscape composition and configuration within the A2L and identify priority areas for conservation and ecological restoration.

Since the completion of this work, a new global land-cover dataset "GlobeLand30" was introduced. It includes the years 2000, 2010, and 2020, has a resolution of 30 m, and a very high classification accuracy (~86.7%) (Sun et al. 2022). Its major drawbacks are that it only utilizes 10 land-cover classes, and it also does not contain a separate road network category. However, it does give researchers a higher-resolution option for future multi-timepoint transboundary analysis.

Conclusion

Many MRCs/counties within the A2L have reached or exceeded thresholds of habitat loss and fragmentation and further changes in landscape structure will significantly jeopardize the integrity and connectivity of the transboundary wildlife linkage. These results highlight the necessity for coordinated and cooperative transboundary conservation efforts. Coordinated conservation across boundaries not only improves the protection of shared conservation features (i.e., ecosystems, species, and natural resources), but can also prove to be considerably cost-effective (Kark et al. 2015). Transboundary conservation is especially valuable when neighbours share common objectives and practices, socioeconomic networks, and information and technology (Bodin and Crona 2009; Kark et al. 2015). Canada and the USA are neighbours that possess these attributes, making an A2L transboundary conservation collaboration highly feasible (Mason et al. 2020). A prime example of the benefits of a transboundary collaboration between Canada and the USA is the Yellowstone-to-Yukon Conservation Initiative (Y2Y). The Y2Y is made up of a network of public, private, and Indigenous protected areas that are critical for the protection of large-ranging transboundary species (Graumlich and Francis 2010; Chester 2015). The strength of the Y2Y initiative comes from its diverse multiscale partnerships across the region. These partnerships have permitted the organization to develop policies for the construction of WCSs, to secure priority lands, and to prevent habitat loss and reduce fragmentation (Graumlich and Francis 2010; Chester 2015; Kark et al. 2015; Mason et al. 2020). Worldwide, there are now over 200 active cases of transboundary conservation (Vasilijević et al. 2015).

The post-2020 global biodiversity framework includes the expansion of protected areas and OECMs to cover at least 30% of the planet by 2030 (CBD 2021). Nevertheless, biodiversity will continue to decline if protected areas become isolated from one another by a landscape vulnerable to increasing habitat loss, fragmentation, and a rapidly changing climate (Kremen and Merenlender 2018). Measuring and monitoring land-cover changes and landscape fragmentation is an effective way to identify priority locations for conservation and ecological restoration and should be included in regional conservation planning and monitoring programs. We have offered seven recommendations for conservation at the local level that will increase habitats and resources and enhance landscape connectivity between protected areas. Strengthening conservation strategies that safeguard and restore landscape connectivity and protect local ecosystems at the MRC/county level will ultimately help achieve post-2020 biodiversity commitments at the provincial, national, and transboundary levels.

Supplemental Tables and Figures

Québec & Ontario 2000, 2010 & 2018				New York 2010 & 2018				Study Area Re-classification 2000, 2010,& 2018		
DMTI Inc.				New York State Information Technology Services						
Class	Category	Description	Re-Class	Class	Category	Description	Re-Class	Category	Description	Class
1	Expressway	400 series highways	1	1	North America/Continental	Largest/Longest highways Connect major/largest cities "Coast-to-coast" origin to destination Inter-state commerce/travel Intra-state commerce/travel	1	Primary roads	Major highways	1
2	Primary highway	Primary highways	2	2	State/Region Inter-metropolitan area	Long/Large highways Beltways/Secondary freeways Connect major cities Connect major suburbs with metro core Intra-state commerce Recreational travel	2	Secondary roads	Highways/Major roads	2
3	Secondary highway	Secondary Highway	2	3	Intra-state Intra-metropolitan	Medium highways US/State highway network Connect minor cities Intra-state commerce Recreational travel	2	Tertiary roads	All other roads	3
4	Major road	Major roads or arterial roads	2	4	City/County	Local arteries Retail commerce Recreational activities Initial route origin/final destination	2			
5	Local road	Subdivision road in a city or gravel road in a rural area	3	5	Neighborhood	Neighborhood/Community access Initial route origin/final destination	3			
				6	Residential	Intra-neighborhood travel Initial route origin/final destination	3			

Table S2.1 Reclassification of the road network maps. Maps from DMTI Inc. are for Québec and Ontario for the years 2000, 2010, and 2018. Maps from New York State Information Technology Services are for New York for the years 2010 and 2018.

Rank	MRC/County	Natural and Anthropogenic Fragmentation Elements			Forests			Non-Forest Vegetation			Wetlands			Combined Habitats		
		Proportion (%)			Proportion (%)			Proportion (%)			Proportion (%)			Proportion (%)		
		1992	2018	Change	1992	2018	Change	1992	2018	Change	1992	2018	Change	1992	2018	Change
1	Hamilton	4.1	4.4	7.6	91.8	95.5	4.1	0.01	0.04	170.02	4.109	0.018	-99.569	95.9	95.6	-0.3
2	Les Pays-d'en-Haut	5.1	5.9	15.4	92.8	93.5	0.8	0.64	0.60	-7.04	1.470	0.027	-98.160	94.9	94.1	-0.8
3	Warren	7.3	8.1	11.8	91.4	90.7	-0.8	1.04	1.16	11.60	0.310	0.049	-84.259	92.7	91.9	-0.9
4	Essex	8.1	8.4	3.1	88.5	88.6	0.2	2.60	2.88	10.81	0.816	0.145	-82.203	91.9	91.6	-0.3
5	Les Laurentides	8.4	8.8	5.6	88.7	90.6	2.2	0.42	0.52	24.40	2.556	0.020	-99.226	91.6	91.2	-0.5
6	Antoine-Labelle	9.6	9.9	3.8	88.6	89.2	0.7	0.50	0.68	34.55	1.366	0.169	-87.595	90.4	90.1	-0.4
7	Matawinie	9.9	10.4	5.4	87.9	88.5	0.6	0.80	1.01	27.36	1.394	0.089	-93.629	90.1	89.6	-0.6
8	Franklin	11.1	11.6	4.1	84.0	83.2	-1.0	4.08	4.47	9.45	0.779	0.796	2.187	88.9	88.4	-0.5
9	Herkimer	12.1	13.0	7.1	72.6	73.0	0.5	12.81	13.84	8.06	2.444	0.175	-92.852	87.9	87.0	-1.0
10	Lewis	11.7	13.6	16.7	81.9	79.2	-3.3	6.10	7.08	16.11	0.304	0.101	-66.667	88.3	86.4	-2.2
11	La Vallée-de-la-Gatineau	12.9	13.7	6.6	85.5	85.9	0.5	0.20	0.30	50.00	1.423	0.041	-97.145	87.1	86.3	-1.0
12	Washington	12.5	13.7	9.9	53.2	50.9	-4.5	33.39	34.56	3.50	0.882	0.854	-3.237	87.5	86.3	-1.4
13	Fulton	14.0	15.3	9.2	77.0	75.5	-1.9	8.18	8.63	5.56	0.814	0.566	-30.435	86.0	84.7	-1.5
14	Papineau	16.6	17.2	3.6	82.7	82.6	-0.2	0.19	0.15	-18.68	0.448	0.048	-89.352	83.4	82.8	-0.7
15	Saratoga	14.7	18.1	23.0	72.9	69.8	-4.2	9.64	9.70	0.63	2.754	2.393	-13.094	85.3	81.9	-4.0
16	Argenteuil	19.3	20.6	6.3	80.0	79.4	-0.8	0.00	0.03	0.00	0.659	0.005	-99.242	80.7	79.4	-1.5
17	La Rivière-du-Nord	17.0	20.6	21.7	82.1	78.5	-4.4	0.96	0.90	-5.97	0.000	0.000	0.000	83.0	79.4	-4.4
18	St. Lawrence	18.9	20.9	10.8	73.7	71.5	-3.0	5.91	6.26	5.84	1.461	1.292	-11.565	81.1	79.1	-2.5
19	Les Collines-de-l'Outaouais	20.8	21.9	5.4	78.8	77.8	-1.3	0.26	0.26	1.19	0.091	0.000	-100.000	79.2	78.1	-1.4
20	Clinton	20.3	22.5	10.5	74.1	71.8	-3.1	4.40	4.86	10.57	1.171	0.882	-24.649	79.7	77.5	-2.7
21	Oneida	29.4	31.1	5.8	54.0	52.5	-2.7	16.05	16.10	0.35	0.551	0.254	-53.876	70.6	68.9	-2.4
22	Lanark	30.0	32.7	9.3	69.3	66.9	-3.6	0.20	0.19	-6.25	0.503	0.211	-57.983	70.0	67.3	-4.0
23	Oswego	37.1	37.3	0.6	51.7	51.4	-0.7	10.01	10.11	1.05	1.166	1.190	2.131	62.9	62.7	-0.4
24	Jefferson	37.3	38.8	4.0	33.6	31.0	-7.8	28.53	29.61	3.77	0.578	0.651	12.565	62.7	61.2	-2.4
25	Schenectady	38.3	43.1	12.6	51.3	46.8	-8.8	8.70	8.63	-0.74	1.691	1.433	-15.267	61.7	56.9	-7.8
26	Montgomery	37.4	43.2	15.7	23.3	13.5	-42.2	38.44	42.47	10.47	0.876	0.817	-6.767	62.6	56.8	-9.4
27	Leeds/Grenville	42.8	47.2	10.4	55.4	51.2	-7.6	1.23	1.21	-1.77	0.605	0.380	-37.126	57.2	52.8	-7.8
28	Montcalm	57.3	57.1	-0.3	42.3	42.6	0.8	0.30	0.30	0.00	0.176	0.046	-73.684	42.7	42.9	0.5
29	Le Haut-Saint-Laurent	59.8	62.4	4.2	37.4	34.5	-7.6	2.30	2.61	13.57	0.515	0.515	0.000	40.2	37.6	-6.3
30	Ottawa/Carleton	61.1	65.1	6.6	38.2	34.3	-10.3	0.57	0.45	-20.82	0.100	0.147	46.512	38.9	34.9	-10.3
31	Gatineau	61.8	67.3	8.9	35.4	30.2	-14.7	2.72	2.37	-12.82	0.104	0.174	66.667	38.2	32.7	-14.3
32	Les Jardins-de-Napierville	66.0	68.3	3.5	29.2	26.6	-8.9	2.10	2.35	12.40	2.757	2.782	0.912	34.0	31.7	-6.8
33	Les Moulins	63.9	70.3	9.9	35.8	29.4	-17.7	0.30	0.27	-8.33	0.000	0.000	0.000	36.1	29.7	-17.6
34	Thérèse-De Blainville	62.8	70.6	12.4	36.8	29.1	-21.1	0.35	0.35	0.00	0.032	0.032	0.000	37.2	29.4	-20.9
35	Stormont/Dundas/Glengarry	68.9	71.3	3.4	30.1	27.7	-8.0	0.70	0.79	12.85	0.220	0.241	9.735	31.1	28.7	-7.4
36	Mirabel	73.1	73.8	1.0	26.5	25.9	-2.5	0.37	0.34	-7.41	0.000	0.000	0.000	26.9	26.2	-2.6
37	Prescott/Russel	71.9	74.0	2.8	27.7	25.7	-7.4	0.27	0.29	9.64	0.058	0.071	22.222	28.1	26.0	-7.2
38	Deux-Montagnes	75.6	77.9	3.0	23.7	21.7	-8.5	0.31	0.24	-21.43	0.419	0.199	-52.632	24.4	22.1	-9.4
39	Vaudreuil-Soulanges	78.1	80.2	2.7	21.5	19.5	-9.5	0.09	0.11	23.08	0.270	0.197	-26.829	21.9	19.8	-9.6
40	Roussillon	84.6	85.4	0.9	14.6	14.0	-3.8	0.34	0.19	-44.00	0.478	0.396	-17.143	15.4	14.6	-5.1
41	Laval	85.2	87.8	3.0	14.1	11.9	-15.6	0.30	0.12	-58.33	0.424	0.200	-52.941	14.8	12.2	-17.5
42	Montréal	88.0	92.9	5.6	9.6	6.4	-33.7	1.79	0.50	-71.86	0.632	0.257	-59.322	12.0	7.1	-40.7
43	Beauharnois-Salaberry	93.1	93.9	0.8	6.5	5.8	-11.1	0.06	0.07	20.00	0.306	0.269	-12.000	6.9	6.1	-10.9
Mean		**38.6**	**40.7**	**5.6**	**55.7**	**53.8**	**-3.4**	**4.84**	**5.06**	**4.57**	**0.877**	**0.422**	**-51.921**	**61.4**	**59.3**	**-3.5**

Table S2.2 Changes in land-cover proportion (%) for each grouped land cover theme between 1992 and 2018, at the scale of the MRC/county. Ranked from highest to lowest proportion of "Combined Habitats" in 2018. Yellow text (10% - 30% change), areas of medium priority for conservation and ecological restoration; orange text (30% - 50% change), areas of medium/high priority; and red text (>50% change), areas of high priority for conservation and ecological restoration.

Agricultural Land Cover Class	Area (Km2)				Proportion (%)				Change	
	1992	2000	2010	2018	1992	2000	2010	2018	Area (Km2)	Area (%)
Study Area										
Cropland, rainfed	504.4	502.7	484.3	480.5	0.3	0.3	0.3	0.3	-23.9	-4.7
Cropland, rainfed, herbaceous cover	13043.7	13318.5	13242.3	13321.9	7.7	7.8	7.8	7.8	278.2	2.1
Mosaic cropland (>50%) / natural vegetation (tree, shrub, herbaceous cover) (<50%)	3091.1	3149.1	3164.0	3159.7	1.8	1.9	1.9	1.9	68.7	2.2
Mosaic natural vegetation (tree, shrub, herbaceous cover) (>50%) / cropland (<50%)	6995.7	7116.5	7334.6	7637.0	4.1	4.2	4.3	4.5	641.3	9.2
Total	**23634.8**	**24086.8**	**24225.1**	**24599.2**	**13.9**	**14.2**	**14.3**	**14.5**	**964.3**	**4.1**
Québec Portion										
Cropland, rainfed	273.6	272.5	262.5	261.5	0.3	0.3	0.3	0.3	-12.2	-4.4
Cropland, rainfed, herbaceous cover	4575.2	4573.6	4445.8	4445.0	5.7	5.7	5.5	5.5	-130.1	-2.8
Mosaic cropland (>50%) / natural vegetation (tree, shrub, herbaceous cover) (<50%)	1786.7	1805.3	1801.9	1799.0	2.2	2.2	2.2	2.2	12.3	0.7
Mosaic natural vegetation (tree, shrub, herbaceous cover) (>50%) / cropland (<50%)	869.3	853.8	914.5	942.0	1.1	1.1	1.1	1.2	72.7	8.4
Total	**7504.7**	**7505.3**	**7424.7**	**7447.5**	**9.3**	**9.3**	**9.2**	**9.2**	**-57.2**	**-0.8**
Ontario Portion										
Cropland, rainfed	135.9	136.3	132.7	132.0	0.7	0.7	0.6	0.6	-3.9	-2.8
Cropland, rainfed, herbaceous cover	7374.2	7574.9	7605.0	7652.0	36.0	36.9	37.1	37.3	277.7	3.8
Mosaic cropland (>50%) / natural vegetation (tree, shrub, herbaceous cover) (<50%)	879.9	892.8	905.0	906.3	4.3	4.4	4.4	4.4	26.4	3.0
Mosaic natural vegetation (tree, shrub, herbaceous cover) (>50%) / cropland (<50%)	1185.9	1227.2	1240.6	1319.6	5.8	6.0	6.0	6.4	133.7	11.3
Total	**9576.0**	**9831.2**	**9883.3**	**10009.9**	**46.7**	**47.9**	**48.2**	**48.8**	**433.9**	**4.5**
New York Portion										
Cropland, rainfed	94.9	94.0	89.1	87.0	0.1	0.1	0.1	0.1	-7.8	-8.3
Cropland, rainfed, herbaceous cover	1094.4	1170.0	1191.6	1225.0	1.6	1.7	1.7	1.8	130.6	11.9
Mosaic cropland (>50%) / natural vegetation (tree, shrub, herbaceous cover) (<50%)	424.4	451.0	457.0	454.4	0.6	0.7	0.7	0.7	30.0	7.1
Mosaic natural vegetation (tree, shrub, herbaceous cover) (>50%) / cropland (<50%)	4940.5	5035.4	5179.5	5375.4	7.2	7.3	7.5	7.8	435.0	8.8
Total	**6554.2**	**6750.4**	**6917.2**	**7141.9**	**9.5**	**9.8**	**10.1**	**10.4**	**587.7**	**9.0**

Table S2.3 Changes in area (km^2) and proportion (%) of each agricultural land-cover class between 1992 and 2018, at the scale of the study area and each provincial/state portion.

Rank	MRC/County	2018					FG-Forests													
		Total Area (km^2)	Area of Habitat (km^2)	Proportion (%)	Patch Number	Mean Patch Size (km^2)	m_{eff_CUT} (km^2)				*Change (%)	m_{eff_CBC} (km^2)				*Change (%)	m_{eff_CBC} - m_{eff_CUT} (km^2)			
							2000	2010	2018	*Change		2000	2010	2018	*Change		2000	2010	2018	*Change
1	Antoine-Labelle	14971.4	14451.3	96.5	2040	7.08	6014.7	2788.2	2771.2	-17.0	-0.6	13427.0	6789.9	5530.4	-1259.5	-18.5	7412.3	4001.7	2759.1	-1242.6
2	Matawinie	9538.0	9203.3	96.5	2750	3.35	3588.2	3502.5	2037.9	-1464.6	-41.8	12981.9	8928.6	4924.7	-4003.9	-44.8	9393.7	5426.1	2886.8	-2539.3
3	La Vallée-de-la-Gatineau	12509.0	12089.6	96.6	1872	6.46	667.9	670.3	655.4	-14.9	-2.2	4204.6	1791.2	1765.3	-25.9	-1.4	3536.7	1120.9	1109.9	-11.0
4	Hamilton	4526.5	4511.9	99.7	511	8.83		521.5	520.3	-1.2	-0.2		1025.0	1010.7	-14.3	-1.4		503.5	490.4	-13.1
5	Essex	4707.6	4416.7	93.8	1477	2.99		355.9	355.5	-0.4	-0.1		426.3	425.8	-0.5	-0.1		70.4	70.4	0.0
6	Herkimer	3740.2	2773.7	74.2	1369	2.03		342.6	335.3	-7.3	-2.1		888.5	852.9	-35.6	-4.0		545.9	517.6	-28.3
7	Warren	2287.0	2197.9	96.1	1558	1.41		111.4	122.3	10.9	9.8		223.5	236.2	12.7	5.7		112.1	113.9	1.8
8	St. Lawrence	7031.3	5242.2	74.6	2996	1.75		102.2	88.8	-13.4	-13.1		233.9	204.0	-29.9	-12.8		131.7	115.2	-16.5
9	Les Laurentides	2518.6	2409.2	95.7	2077	1.16	106.6	106.1	84.2	-21.9	-20.6	3724.7	2586.5	1569.9	-1016.6	-39.3	3618.1	2480.4	1485.7	-994.7
10	Franklin	4265.3	3666.3	86.0	1392	2.63		81.3	77.6	-3.7	-4.6		156.8	151.8	-5.0	-3.2		75.5	74.2	-1.3
11	Papineau	2979.2	2635.0	88.4	1048	2.51	100.1	86.8	68.3	-18.5	-21.3	130.1	112.3	85.6	-26.7	-23.8	30.0	25.5	17.4	-8.1
12	Les Collines-de-l'Outaouais	2076.3	1695.8	81.7	1555	1.09	43.6	44.0	52.5	8.5	19.3	57.1	57.8	60.0	2.2	3.8	13.4	13.8	7.5	-6.3
13	Lewis	3358.2	2661.7	79.3	882	3.02		75.0	47.3	-27.7	-36.9		199.1	145.7	-53.4	-26.8		124.1	98.4	-25.7
14	Saratoga	2133.5	1529.9	71.7	4220	0.36		35.3	34.8	-0.5	-1.4		72.4	72.9	0.5	0.7		37.2	38.1	0.9
15	Fulton	1307.0	1050.0	80.3	945	1.11		37.9	32.9	-5.0	-13.2		107.1	102.1	-5.0	-4.7		69.2	69.2	0.0
16	Argenteuil	1268.2	1053.2	83.0	765	1.38	25.4	36.0	25.1	-10.9	-30.3	53.4	63.6	49.7	-13.9	-21.9	28.0	27.6	24.6	-3.0
17	Clinton	2712.3	2078.2	76.6	1320	1.57		24.6	24.3	-0.3	-1.2		28.5	28.0	-0.5	-1.8		4.0	3.7	-0.3
18	Les Pays-d'en-Haut	703.5	678.9	96.5	1286	0.53	18.3	23.1	23.2	0.1	0.4	32.1	40.1	40.4	0.3	0.7	13.8	17.0	17.1	0.1
19	Washington	2187.6	1121.5	51.3	1239	0.91		14.0	20.8	6.8	48.6		141.8	168.2	26.4	18.6		127.8	147.4	19.6
20	Lanark	3060.6	2136.1	69.8	1515	1.41	20.7	17.8	17.7	-0.1	-0.6	26.0	20.3	20.1	-0.2	-1.0	5.3	2.5	2.5	0.0
21	Oswego	2503.5	1872.8	74.8	2124	0.88		8.0	7.9	-0.1	-1.3		10.5	10.0	-0.5	-4.8		2.5	2.1	-0.4
22	La Rivière-du-Nord	455.2	359.9	79.1	1277	0.28	6.6	5.2	6.3	1.1	21.2	11.2	9.5	13.0	3.5	36.8	4.6	4.3	6.7	2.4
23	Montcalm	715.7	301.8	42.2	770	0.39	4.9	4.4	5.8	1.4	31.8	8.0	7.5	9.2	1.7	22.7	3.1	3.1	3.4	0.3
24	Oneida	3196.3	1719.2	53.8	2570	0.67		5.8	5.8	0.0	0.0		47.2	47.4	0.2	0.4		41.4	41.6	0.2
25	Leeds/Grenville	3398.2	1917.5	56.4	2812	0.68	5.4	4.9	4.8	-0.1	-2.0	5.7	5.3	5.2	-0.1	-1.9	0.4	0.4	0.4	0.0
26	Jefferson	3316.3	1410.5	42.5	2373	0.59		4.4	4.1	-0.3	-6.8		5.6	5.2	-0.4	-7.1		1.2	1.1	-0.1
27	Le Haut-Saint-Laurent	1177.4	447.5	38.0	558	0.80	6.4	3.0	2.9	-0.1	-3.3		4.5	4.4	-0.1	-2.2		1.5	1.5	0.0
28	Prescott/Russel	2012.7	528.5	26.3	868	0.61	1.9	1.9	1.9	0.0	0.0	2.0	2.0	1.9	-0.1	-5.0	0.0	0.0	0.0	0.0
29	Les Moulins	262.8	77.4	29.5	424	0.18	1.7	1.4	1.5	0.1	7.1	2.2	1.9	1.9	0.0	0.0	0.5	0.5	0.4	-0.1
30	Stormont/Dundas/Glengarry	3324.3	958.8	28.8	1590	0.60	1.5	1.4	1.3	-0.1	-7.1	1.6	1.5	1.4	-0.1	-6.7	0.0	0.0	0.0	0.0
31	Les Jardins-de-Napierville	807.5	213.2	26.4	312	0.68	2.0	1.3	1.3	0.0	0.0		1.5	1.5	0.0	0.0		0.2	0.2	0.0
32	Schenectady	541.9	253.9	46.8	1313	0.19		1.3	1.3	0.0	0.0		1.4	1.3	-0.1	-7.1		0.0	0.0	0.0
33	Ottawa/Carleton	2805.5	984.9	6.7	2852	0.35	1.2	1.2	1.1	-0.1	-8.3	1.2	1.2	1.1	-0.1	-8.3	0.0	0.0	0.0	0.0
34	Thérèse-De Blainville	208.5	60.5	29.0	417	0.14	1.1	1.2	1.1	-0.1	-8.3	1.8	1.8	1.6	-0.2	-11.1	0.6	0.6	0.5	-0.1
35	Vaudreuil-Soulanges	856.7	195.8	22.9	995	0.20	0.9	0.8	0.8	0.0	0.0	0.9	0.9	0.8	-0.1	-11.1	0.0	0.0	0.0	0.0
36	Roussillon	424.2	68.0	16.0	371	0.18	1.1	0.8	0.8	0.0	0.0	1.1	0.9	0.9	0.0	0.0	0.1	0.1	0.1	0.0
37	Deux-Montagnes	244.0	65.4	26.8	192	0.34	0.8	0.8	0.7	-0.1	-12.5	0.9	0.9	0.8	-0.1	-11.1	0.1	0.1	0.1	0.0
38	Mirabel	486.0	124.4	25.6	391	0.32	0.9	0.6	0.6	0.0	0.0	1.6	1.3	1.2	-0.1	-7.7	0.7	0.6	0.6	0.0
39	Gatineau	347.6	114.4	32.9	967	0.12	0.7	0.5	0.6	0.1	20.0	1.0	0.8	0.8	0.0	0.0	0.3	0.2	0.3	0.1
40	Laval	247.0	31.0	12.5	350	0.09	0.2	0.3	0.3	0.0	0.0	0.2	0.3	0.3	0.0	0.0	0.0	0.0	0.0	0.0
41	Montgomery	1070.1	143.3	13.4	625	0.23		0.2	0.2	0.0	0.0		0.2	0.2	0.0	0.0		0.0	0.0	0.0
42	Montréal	499.8	39.1	7.8	649	0.06	0.2	0.1	0.1	0.0	0.0	0.2	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0
43	Beauharnois-Salaberry	468.3	31.5	6.7	162	0.19	0.1	0.1	0.1	0.0	0.0	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0
Mean		**2726.8**	**2081.9**	**56.6**	**1343.7**	**1.4**	**N/A**	**209.9**	**173.2**	**-36.7**	**-1.9**	**N/A**	**558.1**	**408.3**	**-149.8**	**-4.7**	**N/A**	**348.2**	**235.1**	**-113.1**

Table S2.4 MRC/County summary data for FG-Forests fragmentation geometry including total area of the MRC (km^2), total area of the fragmentation geometry habitat (km^2), proportion of the fragmentation geometry within the MRC/county (%), patch number, and mean patch size (km^2) in 2018, as well as the Effective Mesh Size values (m_{eff_CUT}, m_{eff_CBC}, and m_{eff_CBC} - m_{eff_CUT}) for 2000, 2010 and 2018. Ranked from highest to lowest m_{eff_CUT} value (i.e., from lowest to highest amount of fragmentation). *Changes in Effective Mesh Size are from 2010 to 2018 only. Yellow text (10% - 30% change), areas of medium priority for conservation and ecological restoration; orange text (30% - 50% change), areas of medium/high priority; and red text (>50% change), areas of high priority for conservation and ecological restoration.

Rank	MRC/County	2018					FG-Non-Forest Vegetation													
		Total Area (km^2)	Area of Habitat (km^2)	Proportion (%)	Patch Number	Mean Patch Size (km^2)	m_{eff_CUT} (m^2)				*Change (%)	m_{eff_CBC} (m^2)					m_{eff_CBC} - m_{eff_CUT} (m^2)			
							2000	2010	2018	*Change		2000	2010	2018	*Change		2000	2010	2018	*Change
1	Jefferson	3316.3	1337.7	40.3	3966	0.34		1218142	1140880	-77262	-6.3		1221770	1144245	-77525	-6.3		3628	3365	-263
2	Washington	2187.6	758.4	34.7	2425	0.31		867546	749059	-118487	-13.7		1165045	828894	-336151	-28.9		297499	79836	-217663
3	Montgomery	1070.1	451.4	42.2	1844	0.24		790001	705506	-84495	-10.7		821885	735351	-86534	-10.5		31884	29845	-2039
4	Herkimer	3740.2	525.5	14.1	2143	0.25		283097	211240	-71857	-25.4		291316	217654	-73662	-25.3		8219	6413	-1806
5	Oswego	2503.5	365.4	14.6	2393	0.15		156788	129516	-27272	-17.4		156933	129632	-27301	-17.4		145	115	-30
6	St. Lawrence	7031.3	454.4	6.5	2216	0.21		122518	122251	-267	-0.2		124040	123674	-366	-0.3		1522	1423	-99
7	Oneida	3196.3	521.6	16.3	4184	0.12		173144	110517	-62627	-36.2		178726	114586	-64140	-35.9		5582	4069	-1513
8	Saratoga	2133.5	211.5	9.9	1740	0.12		177852	103007	-74845	-42.1		178679	103020	-75659	-42.3		827	13	-814
9	Lewis	3358.2	236.3	7.0	1204	0.20		89913	83796	-6117	-6.8		90670	84393	-6277	-6.9		757	597	-160
10	Fulton	1307.0	118.4	9.1	854	0.14		96269	76562	-19707	-20.5		103019	82521	-20498	-19.9		6750	5958	-792
11	Franklin	4265.3	193.9	4.6	1126	0.17		43225	48637	5412	12.5		43274	48675	5401	12.5		49	38	-11
12	Essex	4707.6	141.5	3.0	1030	0.14		46078	44518	-1560	-3.4		46130	44545	-1585	-3.4		52	27	-25
13	Clinton	2712.3	138.5	5.1	1228	0.11		45410	38927	-6483	-14.3		45481	38970	-6511	-14.3		71	43	-28
14	Matawinie	9538.0	104.2	1.1	534	0.20	8136	38276	32347	-5929	-15.5	9093	43207	37032	-6175	-14.3	957	4931	4685	-246
15	Schenectady	541.9	46.3	8.5	592	0.08		77234	28385	-48849	-63.2		83784	31305	-52479	-62.6		6550	2920	-3630
16	Le Haut-Saint-Laurent	1177.4	33.6	2.9	265	0.13	10040	11741	11260	-481	-4.1	13155	11972	11484	-488	-4.1	3115	231	224	-7
17	Warren	2287.0	27.8	1.2	283	0.10		15982	10873	-5109	-32.0		16809	11147	-5662	-33.7		826	274	-552
18	Les Jardins-de-Napierville	807.5	18.8	2.3	130	0.14	9681	10103	10213	110	1.1	9840	10333	10443	110	1.1	159	230	230	0
19	Gatineau	347.6	9.0	2.6	218	0.04	7553	16988	7158	-9830	-57.9	7811	17269	7437	-9832	-56.9	259	280	279	-1
20	Leeds/Grenville	3398.2	44.6	1.3	530	0.08	7214	6836	6741	-95	-1.4	7224	6851	6751	-100	-1.5	10	15	10	-5
21	Antoine-Labelle	14971.4	108.6	0.7	937	0.12	2601	6466	5004	-1462	-22.6	2601	6466	5005	-1461	-22.6	0	1	1	0
22	Stormont/Dundas/Glengarry	3324.3	26.8	0.8	405	0.07	1689	2002	2354	352	17.6	1744	2056	2408	352	17.1	55	55	55	0
23	La Vallée-de-la-Gatineau	12509.0	42.2	0.3	405	0.10	518	2085	2130	45	2.2	518	2085	2130	45	2.2	0	0	0	0
24	Mirabel	486.0	1.7	0.4	18	0.09	1817	2774	1800	-974	-35.1	1827	2805	1802	-1003	-35.8	10	30	2	-28
25	Prescott/Russel	2012.7	6.2	0.3	52	0.12	1359	2059	1321	-738	-35.8	1450	2149	1411	-738	-34.3	90	90	90	0
26	Les Laurentides	2518.6	13.3	0.5	341	0.04	871	1529	1025	-504	-33.0	875	1536	1028	-508	-33.1	3	7	3	-4
27	Deux-Montagnes	244.0	0.7	0.3	8	0.09	1024	1522	995	-527	-34.6	1024	1522	995	-527	-34.6	0	0	0	0
28	Les Pays-d'en-Haut	703.5	4.0	0.6	190	0.02	999	1524	910	-614	-40.3	1004	1530	916	-614	-40.1	5	5	6	1
29	La Rivière-du-Nord	455.2	3.9	0.9	137	0.03	1225	1013	812	-201	-19.8	1242	1029	823	-206	-20.0	16	17	11	-6
30	Thérèse-De Blainville	208.5	0.7	0.4	10	0.07	775	2853	737	-2116	-74.2	779	2968	737	-2231	-75.2	4	115	0	-115
31	Lanark	3060.6	6.0	0.2	98	0.06	474	644	510	-134	-20.8	509	679	545	-134	-19.7	35	35	35	0
32	Montréal	499.8	3.0	0.6	144	0.02	2186	7440	472	-6968	-93.7	2186	7460	472	-6988	-93.7	0	20	0	-20
33	Les Collines-de-l'Outaouais	2076.3	5.4	0.3	120	0.04	410	534	357	-177	-33.1	430	553	376	-177	-32.0	20	20	19	-1
34	Les Moulins	262.8	0.7	0.3	15	0.05	319	436	258	-178	-40.8	319	481	258	-223	-46.4	0	45	0	-45
35	Vaudreuil-Soulanges	856.7	1.1	0.1	15	0.07	200	609	218	-391	-64.2	200	610	218	-392	-64.3	0	2	0	-2
36	Roussillon	424.2	0.9	0.2	16	0.05	270	11592	193	-11399	-98.3	281	11592	200	-11392	-98.3	11	0	8	8
37	Papineau	2979.2	4.8	0.2	97	0.05	569	643	181	-462	-71.9	571	646	184	-462	-71.5	3	3	3	0
38	Montcalm	715.7	2.0	0.3	138	0.01	168	460	135	-325	-70.7	168	467	135	-332	-71.1	0	7	0	-7
39	Argenteuil	1268.2	0.5	0.0	3	0.16		134	130	-4	-3.0		134	130	-4	-3.0		0	0	0
40	Ottawa/Carleton	2805.5	12.5	0.1	313	0.04	267	538	121	-417	-77.5	267	540	122	-418	-77.4	0	1	0	-1
41	Laval	247.0	0.3	0.1	7	0.05	344	6130	110	-6020	-98.2	344	6243	110	-6133	-98.2	0	113	0	-113
42	Beauharnois-Salaberry	468.3	0.4	0.1	6	0.07	79	6677	89	-6588	-98.7	80	6683	90	-6593	-98.7	1	6	1	-5
43	Hamilton	4526.5	1.9	0.0	30	0.06		106	78	-28	-26.4		106	78	-28	-26.4		0	0	0
Mean		**2726.8**	**139.2**	**5.5**	**753.7**	**0.10**	**N/A**	**101091**	**85845**	**-15246**	**-33.3**	**N/A**	**109710**	**89115**	**-20595**	**-33.7**	**N/A**	**8619**	**3270**	**-5349**

Table S2.5 MRC/County summary data for FG-Non-Forest Vegetation fragmentation geometry including total area of the MRC (km^2), total area of the fragmentation geometry habitat (km^2), proportion of the fragmentation geometry within the MRC/county (%), patch number, and mean patch size (km^2) in 2018, as well as the Effective Mesh Size values (m_{eff_CUT}, m_{eff_CBC}, and m_{eff_CBC} - m_{eff_CUT}) for 2000, 2010 and 2018. Ranked from highest to lowest m_{eff_CUT} value (i.e., from lowest to highest amount of fragmentation). *Changes in Effective Mesh Size are from 2010 to 2018 only. Yellow text (10% - 30% change), areas of medium priority for conservation and ecological restoration; orange text (30% - 50% change), areas of medium/high priority; and red text (>50% change), areas of high priority for conservation and ecological restoration.

Rank	MRC/County	2018					FG-Wetlands													
		Total Area (km^2)	Area of Habitat (km^2)	Proportion (%)	Patch Number	Mean Patch Size (km^2)	m_{eff_CUT} (m^2)				*Change (%)	m_{eff_CBC} (m^2)				*Change (%)	m_{eff_CBC} - m_{eff_CUT} (m^2)			
							2000	2010	2018	*Change		2000	2010	2018	*Change		2000	2010	2018	*Change
1	Oswego	2503.5	43.8	1.75	287	0.15		27540	27607	67	0.2		28681	27607	-1074	-3.7		1141	0	-1141
2	Les Jardins-de-Napierville	807.5	22.4	2.78	105	0.21	27741	26095	25878	-217	-0.8	28040	26385	26167	-218	-0.8	299	290	290	0
3	Jefferson	3316.3	29.7	0.90	203	0.15		16935	16879	-56	-0.3		16939	16883	-56	-0.3		5	5	0
4	Saratoga	2133.5	52.4	2.46	532	0.10		17478	14893	-2585	-14.8		17569	14953	-2616	-14.9		91	60	-31
5	St. Lawrence	7031.3	94.3	1.34	504	0.19		10118	9954	-164	-1.6		10120	9956	-164	-1.6		2	2	0
6	Franklin	4265.3	34.9	0.82	170	0.21		9181	9196	15	0.2		9273	9287	14	0.2		92	92	0
7	Clinton	2712.3	25.4	0.94	143	0.18		9417	9100	-317	-3.4		9544	9227	-317	-3.3		127	127	0
8	Leeds/Grenville	3398.2	14.3	0.42	118	0.12	9967	7876	7974	98	1.2	9971	7876	7974	98	1.2	4	0	0	0
9	Washington	2187.6	18.6	0.85	165	0.11		7320	7244	-76	-1.0		10673	10596	-77	-0.7		3352	3352	0
10	Antoine-Labelle	14971.4	27.4	0.18	92	0.30	23800	4121	4164	43	1.0	23848	4121	4164	43	1.0	48	0	0	0
11	Schenectady	541.9	7.7	1.41	118	0.06		3549	3413	-136	-3.8		3615	3479	-136	-3.8		66	66	0
12	Montgomery	1070.1	8.5	0.80	122	0.07		2372	2334	-38	-1.6		2666	2530	-136	-5.1		294	196	-98
13	Oneida	3196.3	8.5	0.26	65	0.13		1853	1498	-355	-19.2		1855	1501	-354	-19.1		3	3	0
14	Le Haut-Saint-Laurent	1177.4	6.7	0.57	69	0.10	1561	1356	1393	37	2.7	2046	1677	1717	40	2.4	485	320	324	4
15	Fulton	1307.0	7.8	0.60	100	0.08		2114	1296	-818	-38.7		2353	1456	-897	-38.1		239	159	-80
16	Lanark	3060.6	6.8	0.22	61	0.11	3326	1097	1102	5	0.5	3362	1097	1102	5	0.5	36	0	0	0
17	Roussillon	424.2	2.1	0.48	45	0.05	421	767	819	52	6.8	426	775	819	44	5.7	5	8	0	-8
18	Herkimer	3740.2	6.7	0.18	46	0.15		689	629	-60	-8.7		689	629	-60	-8.7		0	0	0
19	Stormont/Dundas/Glengarry	3324.3	8.4	0.25	81	0.10	666	555	568	13	2.3	666	555	569	14	2.5	0	0	0	0
20	Lewis	3358.2	3.4	0.10	22	0.16		399	358	-41	-10.3		399	358	-41	-10.3		0	0	0
21	Matawinie	9538.0	9.1	0.10	78	0.12	20121	323	317	-6	-1.9	20681	323	317	-6	-1.9	560	0	0	0
22	Montréal	499.8	1.5	0.31	89	0.02	638	241	220	-21	-8.7	638	242	220	-22	-9.1	0	0	0	0
23	Essex	4707.6	7.2	0.15	112	0.06		238	216	-22	-9.2		240	217	-23	-9.6		1	1	0
24	Gatineau	347.6	0.7	0.19	16	0.04	193	204	204	0	0.0	195	207	207	0	0.0	2	3	3	0
25	Beauharnois-Salaberry	468.3	1.5	0.31	57	0.03	223	195	202	7	3.6	231	203	209	6	3.0	8	8	8	0
26	Papineau	2979.2	1.6	0.05	8	0.20	2764	196	177	-19	-9.7	3319	199	179	-20	-10.1	555	3	3	0
27	Vaudreuil-Soulanges	856.7	1.9	0.23	65	0.03	147	129	138	9	7.0	149	131	140	9	6.9	2	2	2	0
28	Deux-Montagnes	244.0	0.6	0.23	22	0.03	357	69	137	68	98.6	367	79	147	68	86.1	10	10	10	0
29	La Vallée-de-la-Gatineau	12509.0	5.7	0.05	51	0.11	27032	122	123	1	0.8	27249	122	123	1	0.8	217	0	1	1
30	Laval	247.0	0.5	0.21	22	0.02	272	154	122	-32	-20.8	273	155	123	-32	-20.6	1	1	1	0
31	Prescott/Russel	2012.7	1.3	0.06	24	0.05	84	80	81	1	1.3	84	80	81	1	1.3	0	0	0	0
32	Thérèse-De Blainville	208.5	0.2	0.09	7	0.03	22	21	78	57	271.4	33	33	78	45	136.4	12	12	0	-12
33	Montcalm	715.7	0.3	0.05	3	0.11	362	78	78	0	0.0	535	78	78	0	0.0	173	0	0	0
34	Warren	2287.0	1.1	0.05	48	0.02		42	39	-3	-7.1		47	44	-3	-6.4		5	5	0
35	Ottawa/Carleton	2805.5	4.1	0.03	93	0.04	105	35	34	-1	-2.9	105	35	34	-1	-2.9	0	0	0	0
36	Hamilton	4526.5	0.8	0.02	15	0.06		19	20	1	5.3		19	20	1	5.3		0	0	0
37	Les Pays-d'en-Haut	703.5	0.2	0.03	6	0.03	7978	18	15	-3	-16.7	8285	18	15	-3	-16.7	308	0	0	0
38	Les Laurentides	2518.6	0.5	0.02	21	0.02	29407	15	13	-2	-13.3	30135	15	13	-2	-13.3	728	0	0	0
39	Argenteuil	1268.2	0.1	0.01	2	0.03	3115	2	2	0	0.0	3244	2	2	0	0.0	129	0	0	0
40	Les Moulins	262.8	0.0	0.00	2	0.00	0	0	0	0	0.0	1	1	1	0	0.0	1	1	1	0
41	Les Collines-de-l'Outaouais	2076.3	0.0	0.00	0	0.00	416	0	0	0	0.0	416	0	0	0	0.0	0	0	0	0
42	Mirabel	486.0	0.0	0.00	0	0.00	0	0	0	0	0.0	0	0	0	0	0.0	0	0	0	0
43	La Rivière-du-Nord	455.2	0.0	0.00	0	0.00	0	0	0	0	0.0	0	0	0	0	0.0	0	0	0	0
Mean		**2726.8**	**10.9**	**0.50**	**88.1**	**0.10**	**N/A**	**3558**	**3454**	**-104**	**4.8**	**N/A**	**3700**	**3563**	**-137**	**1.2**		**141**	**110**	**-31**

Table S2.6 MRC/County summary data for FG-Wetlands fragmentation geometry including total area of the MRC (km^2), total area of the fragmentation geometry habitat (km^2), proportion of the fragmentation geometry within the MRC/county (%), patch number, and mean patch size (km^2) in 2018, as well as the Effective Mesh Size values (m_{eff_CUT}, m_{eff_CBC}, and m_{eff_CBC} - m_{eff_CUT}) for 2000, 2010 and 2018. Ranked from highest to lowest m_{eff_CUT} value (i.e., from lowest to highest amount of fragmentation). *Changes in Effective Mesh Size are from 2010 to 2018 only. Yellow text (10% - 30% change), areas of medium priority for conservation and ecological restoration; orange text (30% - 50% change), areas of medium/high priority; and red text (>50% change), areas of high priority for conservation and ecological restoration.

Rank	MRC/County	2018					FG-Combined Habitats													
		Total Area (km^2)	Area of Habitat (km^2)	Proportion (%)	Patch Number	Mean Patch Size (km^2)	m_{eff_CUT} (km^2)				*Change (%)	m_{eff_CBC} (km^2)				*Change (%)	m_{eff_CBC} - m_{eff_CUT} (km^2)			
							2000	2010	2018	*Change		2000	2010	2018	*Change		2000	2010	2018	*Change
1	Antoine-Labelle	14971.4	14614.9	97.6	2213	6.6	6575.9	2865.5	2863.6	-1.9	-0.1	14787.6	7003.6	6047.6	-956.0	-13.7	8211.8	4138.1	3184.0	-954.1
2	Matawinie	9538.0	9373.5	98.3	2652	3.5	4117.4	3671.2	2301.4	-1369.8	-37.3	14654.0	9349.0	5849.9	-3499.1	-37.4	10536.6	5677.8	3548.5	-2129.3
3	La Vallée-de-la-Gatineau	12509.0	12146.2	97.1	1924	6.3	709.9	674.6	661.8	-12.8	-1.9	4575.1	1821.7	1803.0	-18.7	-1.0	3865.1	1147.1	1141.2	-5.9
4	Hamilton	4526.5	4514.8	99.7	505	8.9		521.8	520.9	-0.9	-0.2		1026.3	1012.8	-13.5	-1.3		504.5	491.9	-12.6
5	Essex	4707.6	4567.0	97.0	1678	2.7		358.4	357.8	-0.6	-0.2		429.1	428.5	-0.6	-0.1		70.7	70.7	0.0
6	Herkimer	3740.2	3309.2	88.5	1948	1.7		344.4	337.0	-7.4	-2.1		891.8	856.5	-35.3	-4.0		547.4	519.5	-27.9
7	Warren	2287.0	2227.9	97.4	1556	1.4		111.9	123.1	11.2	10.0		224.4	237.6	13.2	5.9		112.5	114.5	2.0
8	St. Lawrence	7031.3	5796.3	82.4	4146	1.4		104.2	91.7	-12.5	-12.0		236.5	207.9	-28.6	-12.1		132.3	116.2	-16.1
9	Les Laurentides	2518.6	2437.1	96.8	2104	1.2	112.7	107.1	85.1	-22.0	-20.5	3978.5	2656.4	1772.6	-883.8	-33.3	3865.8	2549.3	1687.5	-861.8
10	Franklin	4265.3	3895.6	91.3	1830	2.1		82.9	79.7	-3.2	-3.9		158.8	154.2	-4.6	-2.9		75.9	74.5	-1.4
11	Papineau	2979.2	2644.4	88.8	1076	2.5	100.9	87.0	68.4	-18.6	-21.4	131.9	112.6	85.8	-26.8	-23.8	31.0	25.6	17.4	-8.2
12	Les Collines-de-l'Outaouais	2076.3	1708.0	82.3	1597	1.1	44.8	44.2	52.7	8.5	19.2	58.6	58.1	60.2	2.1	3.6	13.9	13.9	7.5	-6.4
13	Lewis	3358.2	2901.7	86.4	1489	1.9		75.8	48.0	-27.8	-36.7		200.2	146.8	-53.4	-26.7		124.4	98.7	-25.7
14	Saratoga	2133.5	1795.4	84.2	4610	0.4		35.8	35.3	-0.5	-1.4		73.0	73.4	0.4	0.5		37.2	38.1	0.9
15	Fulton	1307.0	1177.5	90.1	1207	1.0		38.4	33.4	-5.0	-13.0		107.7	102.6	-5.1	-4.7		69.3	69.3	0.0
16	Washington	2187.6	1898.5	86.8	1701	1.1		22.8	31.6	8.8	38.6		456.2	513.8	57.6	12.6		433.4	482.1	48.7
17	Argenteuil	1268.2	1054.8	83.2	776	1.4	25.8	36.0	25.2	-10.8	-30.0	54.2	63.7	49.8	-13.9	-21.8	28.3	27.7	24.6	-3.1
18	Clinton	2712.3	2243.4	82.7	1966	1.1		25.3	25.0	-0.3	-1.2		29.4	28.9	-0.5	-1.7		4.0	3.8	-0.2
19	Les Pays-d'en-Haut	703.5	685.2	97.4	1287	0.5	19.4	23.3	23.6	0.3	1.3	33.8	40.4	40.8	0.4	1.0	14.3	17.1	17.3	0.2
20	Lanark	3060.6	2154.7	70.4	1576	1.4	21.1	18.0	18.0	0.0	0.0	26.4	20.5	20.5	0.0	0.0	5.4	2.6	2.5	-0.1
21	Oswego	2503.5	2285.2	91.3	2232	1.0		9.0	8.9	-0.1	-1.1		11.6	11.0	-0.6	-5.2		2.6	2.1	-0.5
22	Jefferson	3316.3	2779.1	83.8	3619	0.8		8.1	7.5	-0.6	-7.4		9.3	8.7	-0.6	-6.5		1.3	1.1	-0.2
23	Oneida	3196.3	2250.2	70.4	4415	0.5		6.7	6.7	0.0	0.0		48.6	48.7	0.1	0.2		41.9	42.1	0.2
24	La Rivière-du-Nord	455.2	377.7	83.0	1251	0.3	7.2	5.5	6.6	1.1	20.0	11.9	9.8	13.4	3.6	36.7	4.7	4.3	6.8	2.5
25	Montcalm	715.7	315.1	44.0	812	0.4	5.2	4.7	6.2	1.5	31.9	8.6	7.9	9.7	1.8	22.8	3.4	3.2	3.5	0.3
26	Leeds/Grenville	3398.2	1985.6	58.4	3049	0.7	5.8	5.2	5.2	0.0	0.0	6.3	5.7	5.6	-0.1	-1.8	0.5	0.4	0.4	0.0
27	Le Haut-Saint-Laurent	1177.4	488.0	41.4	758	0.6	7.5	3.3	3.3	0.0	0.0		4.9	4.8	-0.1	-2.0		1.6	1.6	0.0
28	Les Jardins-de-Napierville	807.5	254.7	31.5	291	0.9	3.0	2.3	2.1	-0.2	-8.7		2.6	2.4	-0.2	-7.7		0.3	0.3	0.0
29	Prescott/Russel	2012.7	536.0	26.6	908	0.6	2.0	2.1	2.0	-0.1	-4.8	2.1	2.1	2.0	-0.1	-4.8	0.0	0.0	0.0	0.0
30	Les Moulins	262.8	78.3	29.8	433	0.2	1.7	1.4	1.5	0.1	7.1	2.2	1.9	2.0	0.1	5.3	0.5	0.5	0.5	0.0
31	Schenectady	541.9	307.9	56.8	1584	0.2		1.5	1.5	0.0	0.0		1.6	1.5	-0.1	-6.3		0.1	0.1	0.0
32	Stormont/Dundas/Glengarry	3324.3	994.0	29.9	1795	0.6	1.6	1.5	1.4	-0.1	-6.7	1.6	1.5	1.4	-0.1	-6.7	0.0	0.0	0.0	0.0
33	Montgomery	1070.1	603.3	56.4	1885	0.3		1.2	1.1	-0.1	-8.3		1.3	1.2	-0.1	-7.7		0.1	0.1	0.0
34	Ottawa/Carleton	2805.5	1002.0	35.7	3023	0.3	1.2	1.2	1.1	-0.1	-8.3	1.2	1.2	1.1	-0.1	-8.3	0.0	0.0	0.0	0.0
35	Thérèse-De Blainville	208.5	61.5	29.5	424	0.1	1.1	1.2	1.1	-0.1	-8.3	1.8	1.8	1.6	-0.2	-11.1	0.6	0.6	0.5	-0.1
36	Vaudreuil-Soulanges	856.7	198.8	23.2	1063	0.2	0.9	0.8	0.8	0.0	0.0	0.9	0.9	0.8	-0.1	-11.1	0.0	0.0	0.0	0.0
37	Roussillon	424.2	70.8	16.7	409	0.2	1.1	0.8	0.8	0.0	0.0	1.1	0.9	0.9	0.0	0.0	0.1	0.1	0.1	0.0
38	Deux-Montagnes	244.0	66.7	27.3	209	0.3	0.8	0.8	0.8	0.0	0.0	1.0	0.9	0.9	0.0	0.0	0.1	0.1	0.1	0.0
39	Gatineau	347.6	125.0	35.9	1079	0.1	0.8	0.6	0.7	0.1	16.7	1.1	0.9	0.9	0.0	0.0	0.3	0.2	0.3	0.1
40	Mirabel	486.0	127.2	26.2	415	0.3	0.9	0.7	0.6	-0.1	-14.3	1.7	1.3	1.3	0.0	0.0	0.7	0.6	0.6	0.0
41	Laval	247.0	31.8	12.9	368	0.1	0.2	0.3	0.3	0.0	0.0	0.2	0.3	0.3	0.0	0.0	0.0	0.0	0.0	0.0
42	Montréal	499.8	43.7	8.7	859	0.1	0.2	0.1	0.1	0.0	0.0	0.2	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0
43	Beauharnois-Salaberry	468.3	33.3	7.1	213	0.2	0.1	0.1	0.1	0.0	0.0	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0
Mean		**2726.8**	**2236.3**	**63.4**	**1603.1**	**1.3**	**N/A**	**216.5**	**182.4**	**-34.1**	**-2.4**	**N/A**	**583.2**	**456.1**	**-127.1**	**-4.1**	**N/A**	**366.7**	**273.7**	**-93**

Table S2.7 MRC/County summary data for FG-Combined Habitats fragmentation geometry including total area of the MRC (km^2), total area of the fragmentation geometry habitat (km^2), proportion of the fragmentation geometry within the MRC/county (%), patch number, and mean patch size (km^2) in 2018, as well as the Effective Mesh Size values (m_{eff_CUT}, m_{eff_CBC}, and m_{eff_CBC} - m_{eff_CUT}) for 2000, 2010 and 2018. Ranked from highest to lowest m_{eff_CUT} value (i.e., from lowest to highest amount of fragmentation). *Changes in Effective Mesh Size are from 2010 to 2018 only. Yellow text (10% - 30% change), areas of medium priority for conservation and ecological restoration; orange text (30% - 50% change), areas of medium/high priority; and red text (>50% change), areas of high priority for conservation and ecological restoration.

Rank	MRC/County	Road Legnth (km^2)				Road Density (km/km^2)				Change (%)
		2000	2010	2018	Change	2000	2010	2018	Change	
1	Montréal	5558.1	5956.4	6009.9	53.5	6.61	7.09	7.15	0.064	0.9
2	Laval	1796.3	1981.1	2063.0	81.9	4.98	5.49	5.72	0.227	4.1
3	Thérèse-De Blainville	906.8	1063.8	1098.5	34.6	3.18	3.73	3.85	0.122	3.3
4	Gatineau	1400.2	1669.8	1746.5	76.7	2.71	3.23	3.38	0.148	4.6
5	Les Moulins	947.8	1093.5	1161.9	68.4	2.63	3.04	3.23	0.190	6.3
6	Schenectady		1825.3	1847.1	21.8		2.62	2.65	0.031	1.2
7	Ottawa/Carleton	5976.1	7001.1	7398.9	397.7	2.06	2.41	2.55	0.137	5.7
8	La Rivière-du-Nord	1178.2	1557.1	1574.4	17.4	1.87	2.47	2.50	0.028	1.1
9	Roussillon	1265.5	1422.9	1536.6	113.8	1.92	2.16	2.33	0.173	8.0
10	Deux-Montagnes	740.8	808.2	813.8	5.6	1.82	1.98	1.99	0.014	0.7
11	Les Pays-d'en-Haut	1498.0	1637.0	1752.7	115.7	1.50	1.64	1.76	0.116	7.1
12	Saratoga		4318.9	4278.2	-40.7		1.53	1.52	-0.014	-0.9
13	Vaudreuil-Soulanges	1715.8	1933.2	2039.2	106.1	1.25	1.41	1.49	0.078	5.5
14	Montgomery		1930.8	1938.3	7.5		1.41	1.42	0.006	0.4
15	Montcalm	1232.5	1301.3	1321.9	20.6	1.27	1.34	1.36	0.021	1.6
16	Oneida		5431.7	5473.7	42.0		1.29	1.30	0.010	0.8
17	Beauharnois-Salaberry	787.0	863.7	952.1	88.4	1.07	1.17	1.29	0.120	10.2
18	Mirabel	676.1	793.9	839.5	45.6	1.03	1.21	1.28	0.070	5.7
19	Washington		3014.0	2920.4	-93.6		1.06	1.03	-0.033	-3.1
20	Argenteuil	1514.5	1718.8	1782.7	63.8	0.84	0.95	0.99	0.035	3.7
21	Prescott/Russel	2622.1	2694.8	2722.4	27.6	0.94	0.97	0.98	0.010	1.0
22	Stormont/Dundas/Glengarry	4342.3	4487.1	4496.8	9.6	0.94	0.97	0.97	0.002	0.2
23	Leeds/Grenville	4407.0	4824.7	4817.3	-7.4	0.89	0.97	0.97	-0.001	-0.2
24	Les Jardins-de-Napierville	1031.9	1054.3	1038.7	-15.6	0.96	0.98	0.97	-0.015	-1.5
25	Fulton		1672.7	1678.4	5.6	0.00	0.94	0.94	0.003	0.3
26	Les Laurentides	2871.2	3232.6	3423.7	191.1	0.79	0.89	0.94	0.053	5.9
27	Les Collines-de-l'Outaouais	2102.6	2644.6	2713.6	69.0	0.71	0.90	0.92	0.023	2.6
28	Le Haut-Saint-Laurent	1305.0	1500.4	1498.9	-1.5	0.75	0.87	0.87	-0.001	-0.1
29	Clinton		3245.3	3262.4	17.1		0.85	0.85	0.004	0.5
30	Jefferson		4815.3	5020.6	205.3		0.81	0.84	0.035	4.3
31	Lanark	3143.6	3418.4	3436.2	17.8	0.74	0.80	0.81	0.004	0.5
32	Oswego		3665.8	3730.7	64.9		0.78	0.79	0.014	1.8
33	Warren		2716.5	2442.3	-274.2		0.87	0.78	-0.087	-10.1
34	Papineau	2762.0	3058.3	3171.0	112.7	0.64	0.71	0.73	0.026	3.7
35	Lewis		2875.1	3140.9	265.8		0.66	0.72	0.061	9.2
36	St. Lawrence		6299.2	6414.1	114.8		0.65	0.67	0.012	1.8
37	Franklin		3453.9	3582.4	128.5		0.59	0.62	0.022	3.7
38	Herkimer		2930.4	2974.6	44.1		0.60	0.61	0.009	1.5
39	Essex		3349.1	3168.2	-180.9		0.51	0.49	-0.028	-5.4
40	Matawinie	4527.5	5285.1	5827.7	542.6	0.31	0.37	0.41	0.038	10.3
41	Antoine-Labelle	7369.6	8164.5	8106.2	-58.4	0.33	0.36	0.36	-0.003	-0.7
42	La Vallée-de-la-Gatineau	5879.1	6234.6	6277.6	43.0	0.30	0.32	0.32	0.002	0.7
43	Hamilton		1544.1	1574.5	30.4		0.25	0.26	0.005	2.0
Mean		**N/A**	**3034.6**	**3094.6**	**60.0**	**N/A**	**1.49**	**1.53**	**0.040**	**2.3**

Table S2.8 Changes in road length (km) and road density (km/km^2) for each road category between 2000 and 2018, at the scale of the MRC/county. Ranked from highest to lowest road density in 2018. *Changes in Road Length and Road Density are from 2010 to 2018 only. Yellow text (10% - 30% change), areas of medium priority for conservation and ecological restoration.

Land Cover Class/Element	Land Cover Themes					Fragmentation Geometries			
	Natural and Anthropogeneic Fragmnetaion Elements	Forests	Non-Forest Vegatation	Wetlands	Combineed Habitats	FG -Forests	FG - Non-Forrst Vegation	FG - Wetlands	FG - Combined Habitats
Developed areas	✓					✓	✓	✓	✓
Bare areas	✓					✓	✓	✓	✓
Water bodies	✓					✓	✓	✓	✓
Cropland, rainfed	✓					✓	✓	✓	✓
Cropland, rainfed, herbaceous cover	✓					✓	✓	✓	✓
Mosaic cropland (>50%) / natural vegetation (tree, shrub, herbaceous cover) (<50%)	✓					✓	✓	✓	✓
Mosaic natural vegetation (tree, shrub, herbaceous cover) (>50%) / cropland (<50%)	✓					✓	✓	✓	✓
Broadleaved evergreen closed to open trees / Broadleaved semi-deciduous closed to open trees		✓			✓		✓	✓	
Tree cover, broadleaved, deciduous, closed to open (>15%)		✓			✓		✓	✓	
Tree cover, broadleaved, deciduous, closed (>40%)		✓			✓		✓	✓	
Tree cover, broadleaved, deciduous, open (15-40%)		✓			✓		✓	✓	
Tree cover, needleleaved, evergreen, closed to open (>15%)		✓			✓		✓	✓	
Tree cover, needleleaved, evergreen, closed (>40%)		✓			✓		✓	✓	
Tree cover, needleleaved, evergreen, open (15-40%)		✓			✓		✓	✓	
Tree cover, needleleaved, deciduous, closed to open (>15%)		✓			✓		✓	✓	
Tree cover, mixed leaf type (broadleaved and needleleaved)		✓			✓		✓	✓	
Mosaic tree and shrub (>50%) / herbaceous cover (<50%)		✓			✓		✓	✓	
Mosaic herbaceous cover			✓		✓	✓		✓	
Shrubland			✓		✓	✓		✓	
Grassland			✓		✓	✓		✓	
Lichens and mosses			✓		✓	✓		✓	
Sparse vegetation (tree, shrub, herbaceous cover) (<15%)			✓		✓	✓		✓	
Tree cover, flooded, fresh or brakish water				✓	✓	✓	✓		
Shrub or herbaceous cover, flooded, fresh/saline/brakish water				✓	✓	✓	✓		
Primary Roads (10m buffer)						✓	✓	✓	✓
Secondary Roads (5m buffer)						✓	✓	✓	✓
Tertiary Roads (3m buffer)						✓	✓	✓	✓

Table S2.9 Map categories included in each of the “Land-Cover Themes” and “Fragmentation Geometries”. (✓) = included.

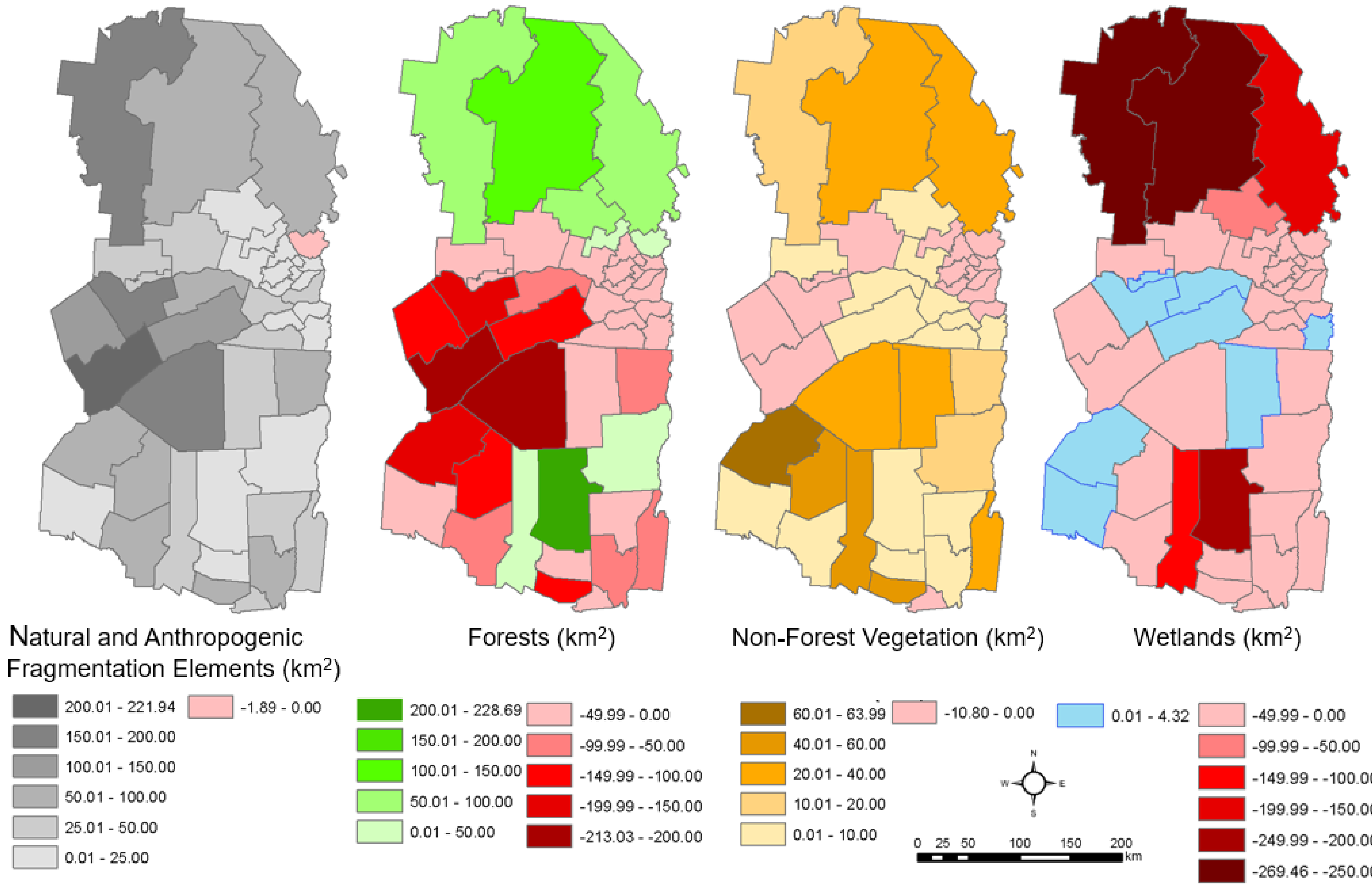

Figure S2.1 Changes in area (km^2) of grouped land-cover themes in each MRC/county between 1992-2018.

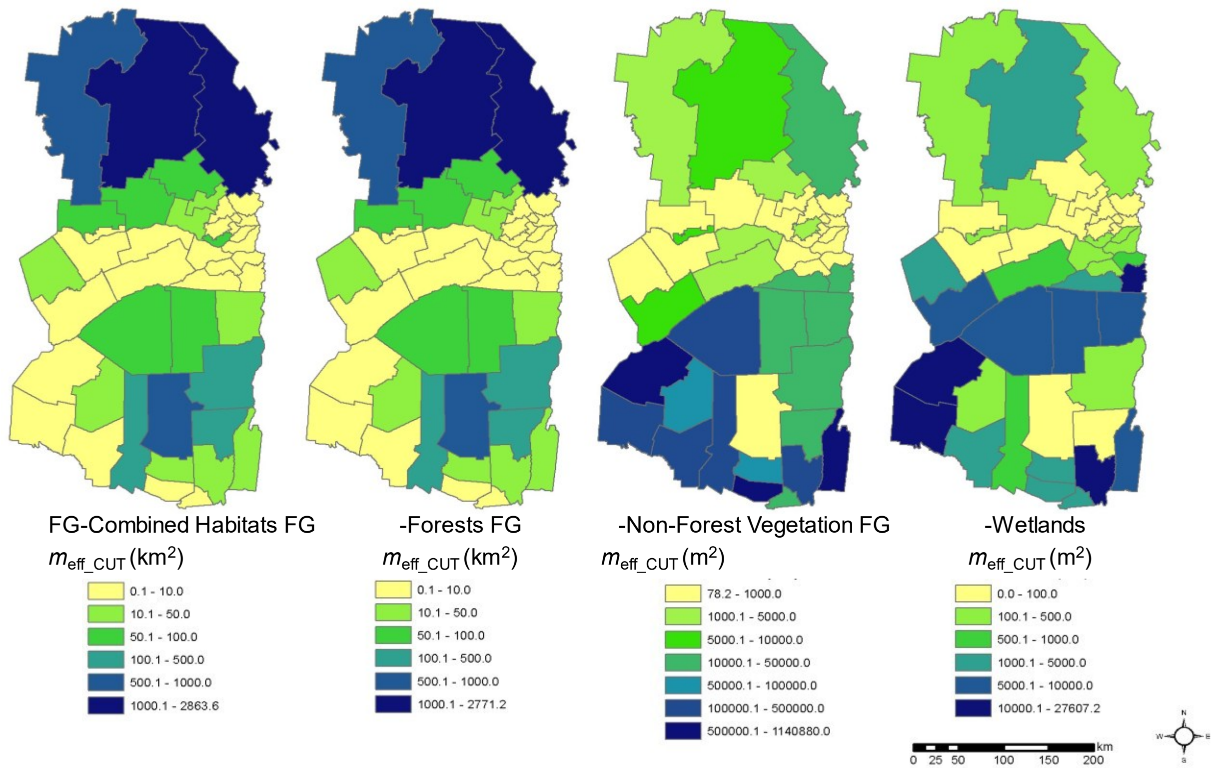

Figure S2.2. Degree of fragmentation in each MRC/county for each fragmentation geometry in 2018 as measured by effective mesh size $m_{\text{eff\_CUT}}$

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Connecting Statement 1.

Chapter 2 showed that extensive changes in landscape structure (composition and configuration) occurred within the A2L between 1992 and 2018. Animal movements and many other ecological processes require connectivity ("the degree to which the landscape facilitates or impedes movement among resource patches"; Taylor et al., 1993). There are two types of connectivity: (1) Structural connectivity, which refers to the arrangement, permeability, and contiguity of land-cover elements (Lindenmayer & Fischer, 2013; Hilty et al., 2020), and (2) functional connectivity, which is species-specific and is defined as the combination of both landscape structure and the responses of a species to that structure (i.e., the ability of a species to move between resource patches within a landscape) (Meiklejohn et al., 2009; Lindenmayer & Fischer, 2013).

In chapter 3, I continue this line of thought by exploring if these changes to landscape structure and structural connectivity, measured by the effective mesh size, had significant impacts on the species that inhabit or traverse the wildlife linkage. Using suitable habitat and resistance modeling, I quantify and compare species-specific habitat amount, habitat fragmentation, and functional connectivity for the American black bear (*Ursus americanus*), fisher (*Pekania pennanti*), moose (*Alces alces*), and white-tailed deer (*Odocoileus virginianus*) between 2000 and 2015, and then use this information to identify priority locations for conservation and ecological restoration.

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3. Impacts of Anthropogenic Land Transformation on Species-Specific Habitat Amount, Fragmentation, and Connectivity in the Adirondack-to-Laurentians Transboundary Wildlife Linkage between 2000 and 2015: Implications for Conservation and Ecological Restoration

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Abstract

Context

The Adirondack-to-Laurentians (A2L) transboundary wildlife linkage is one of three north–south movement linkages that connect natural areas in northeastern USA and southeastern Canada. This region still retains habitats of high ecological integrity and biodiversity; however, anthropogenic land transformation may be putting transboundary connectivity at risk.

Objectives

We measured the impacts of anthropogenic land transformation on species-specific habitat amount, fragmentation, and connectivity in the A2L between 2000 and 2015.

Methods

We developed suitable habitat and resistance models for the American black bear (*Ursus americanus*), fisher (*Pekania pennanti*), moose (*Alces alces*), and white-tailed deer (*Odocoileus virginianus*) to identify suitable and optimal habitat patches for each species. We quantified habitat amount, fragmentation, and connectivity, and used Linkage Mapper and Circuitscape to map corridors and pinch-points important for connectivity.

Results

Between 2000 and 2015, suitable and optimal habitat patch area declined considerably, fragmentation increased, and inter-patch connectivity decreased for each species. Moose and black bear habitat patches experienced the greatest habitat loss, fragmentation, and decline in inter-patch connectivity. The majority of habitat patch area loss and fragmentation occurred in the southern Québec and Ontario portions.

Conclusions

To achieve long-term functionality of the A2L, collaborative and coordinated measures will be necessary to preserve the integrity of the Québec mega-patch, restore extensive habitat in eastern Ontario, and reestablish or maintain connectivity throughout the linkage. Left unaddressed, continued anthropogenic land transformation is likely to have detrimental effects on the ability of the A2L to function as a transboundary wildlife linkage.

Keywords: Habitat loss, Effective mesh size, Linkage mapper, Least-cost corridors, Circuitscape, Black bear, Fisher, Moose, White-tailed deer

Introduction

Habitat loss and fragmentation due to anthropogenic land transformation is one of the leading causes of biodiversity declines worldwide (Haddad et al. 2015; Maxwell et al. 2016; Diaz et al. 2019). Habitat loss and fragmentation contribute to long-term changes in ecosystem structure and function (Lindenmayer and Fischer 2013) and can lead to an overall reduction in species abundance and movement ability between fragments (Haddad et al. 2015; Crooks et al. 2017). In North America, monitored vertebrate population sizes have declined by an average of 20% since

1970, with habitat loss and fragmentation being the main driver of these declines (WWF 2020, 2022).

Landscape connectivity can facilitate animal movement among resource patches (Taylor et al. 1993). Indeed, long-term viability of wildlife populations is linked to landscape connectivity which consists of intra-patch connectivity (i.e., movements such as daily foraging; Tischendorf and Fahrig 2000; Spanowicz and Jaeger 2019) and inter-patch connectivity (movements such as dispersal, migration, and range shifts in response to climate change; Tischendorf and Fahrig 2000; Ament et al. 2014; Spanowicz and Jaeger 2019).

Successful dispersal events help to maintain long-term viability of populations by colonizing new areas, rescuing sink populations, and maintaining genetic variation and gene flow within meta-populations (Kokko and López-Sepulcre 2006). Ecological corridors facilitate movement between habitat patches, whereas wildlife linkages promote the movement of multiple species and ecological processes within a network of habitat patches (Beier et al. 2008; Meiklejohn et al. 2009).

Globally, 56% of all terrestrial mammals have transboundary geographic ranges (Mason et al. 2020). On the contrary, conservation programs generally do not span political borders, making positive conservation outcomes contingent on the alignment of similar conservation values across multiple jurisdictions (Kark et al. 2015). Transboundary conservation presents an opportunity to improve protection of species with transboundary ranges through coordinated and collaborative international conservation efforts (Vasilijević et al. 2015; Mason et al. 2020).

The Adirondack-to-Laurentians (A2L) transboundary wildlife linkage is one of three north–south movement linkages that connect natural areas in northeastern USA and southeastern Canada. This region features habitats of high ecological integrity and biodiversity; however, continued anthropogenic land transformation is putting transboundary connectivity at risk (Cole et al. 2023). As a result, there is an urgent need for strategic conservation and restoration intervention, as well as the development of coordinated transboundary management plans between Canada and the USA, to limit further deterioration of the linkage.

Our aim was to assess the impact of land-cover change on species-specific habitat amount, fragmentation, and connectivity in the A2L transboundary wildlife linkage between 2000 and 2015. We created suitable habitat and resistance models for four species: American black bear (*Ursus americanus*), fisher (*Pekania pennanti*), moose (*Alces alces*), and white-tailed deer (*Odocoileus virginianus*). We modelled suitable habitat patches (SHPs), optimal habitat patches (OHPs), and stepping-stone patches (SSPs) for each species, and used Linkage Mapper and Circuitscape to map least-cost corridors (LCCs) and pinch-points important for connectivity. We then identified priority habitat patches and corridors for conservation and restoration. Specifically, we asked: (1) to what degree has habitat loss and fragmentation occurred within the study area for each species? (2) to what degree has connectivity changed for each species? (3) what percentage of habitat patches and corridors are under protection for each species?

Methods

Study area

Our ~127,000 km^2 A2L study area spans portions of Québec and Ontario in Canada, and New York in the USA; and includes 43 municipalités régionales decomté (MRCs) and counties. The A2L is bounded by the Laurentian Mountains in Québec to the north, the Adirondack Mountains in New York to the south, and the cities of Ottawa, Ontario and Montréal, Québec to the west and east, respectively (Fig. 3.1). The A2L is situated in the northern forest and eastern temperate forest eco-regions and is home to 440 vertebrate species and 1600 vascular plant species (Tardif et al. 2005; CEC 2023). Dominant tree species in the Québec and Ontario portions include sugar maple, American basswood, white ash, American hophornbeam, butternut, yellow birch, American beech, northern red oak, and eastern hemlock (Tardif et al. 2005); dominant tree species in the New York portion include spruce-fir, evergreen-northern hardwood, and mesic upland hardwoods including sugar maple, American beech, yellow birch, and oak (Graves and Wang 2012). The geology of the A2L is comprised of Canadian Shield to the north, Saint Lawrence Platform in the centre, and Precambrian to the south (Tardif et al. 2005). The highest peak is Mount-Marcy in the Adirondacks at 1629 m. As of 2016, the area was home to over 6.8 million people (54 per km^2) (Statistics Canada 2023; US Census Bureau 2023).

Data sources

We identified four focal species to represent the broad range of habitat and movement requirements of native terrestrial, non-volant mammals in the study region (Beier and Loe 1992): American black bear (*Ursus americanus*), fisher (*Pekania pennanti*), moose (*Alces alces*), and white-tailed deer (*Odocoileus virginianus*). These species differ in home range size, habitat preferences, and intra- and inter-patch movement capabilities. Identifying potential habitat and connectivity routes for these "umbrella species" will undoubtedly identify potential habitat and connectivity routes for a variety of other species that reside within the same ecological community (Frankel and Soulé, 1981).

We obtained land-cover, road-network, MRC/county boundaries, and protected area maps for Québec, Ontario, New York, Vermont, and Massachusetts for the years 2000 and 2015 (Table S3.1). We created 30 m resolution maps of the study area for each time-period by converting each source map from polygon/polyline to raster using the "Polygon to Raster" and "Polyline to Raster" tools in ArcGIS10.7 (Environmental Systems Research Institute, Redlands, CA). We then reclassified the land-cover and road-network maps into 10 common land-cover classes and 3 common road classes unifying the classification scheme across all input maps (Table S3.2; S3.3). We then used the re-classified land-cover and road-network maps to create four additional environmental variable layers to represent human disturbance levels: (1) distance from development; (2) distance from primary roads; (3) distance from secondary roads; and (4) distance from tertiary roads. We created each of these layers by generating three buffers of 0–500, 500–1000, and 1000+ meters around the land-cover element of interest (i.e., roads, development) using the "Euclidean Distance" function in ArcGIS10.7. Buffers that overlapped did not impact the individual buffers because overlapping buffer areas remain part of all the individual buffers. These distances represent the medium and maximum distances disturbance-avoidance behaviour (i.e., from human activity, development, and roads) is displayed by black bear and moose; whereas fisher and white-tailed deer exhibit negligible disturbance-avoidance behaviour (Arthur et al. 1989; Laurian et al. 2008; Munro et al. 2012). Analyses were performed at three spatial scales: "the study area" which included the surface area of all 43 MRCs/counties together; "the provincial/state portions" which included the surface area of each individual provincial/state portion only; and "the MRCs/counties", which included the surface area of each individual MRC/county only. The land area surrounding the outside of the study area (which included small areas of Vermont and Massachusetts) were not used in any of the analyses.

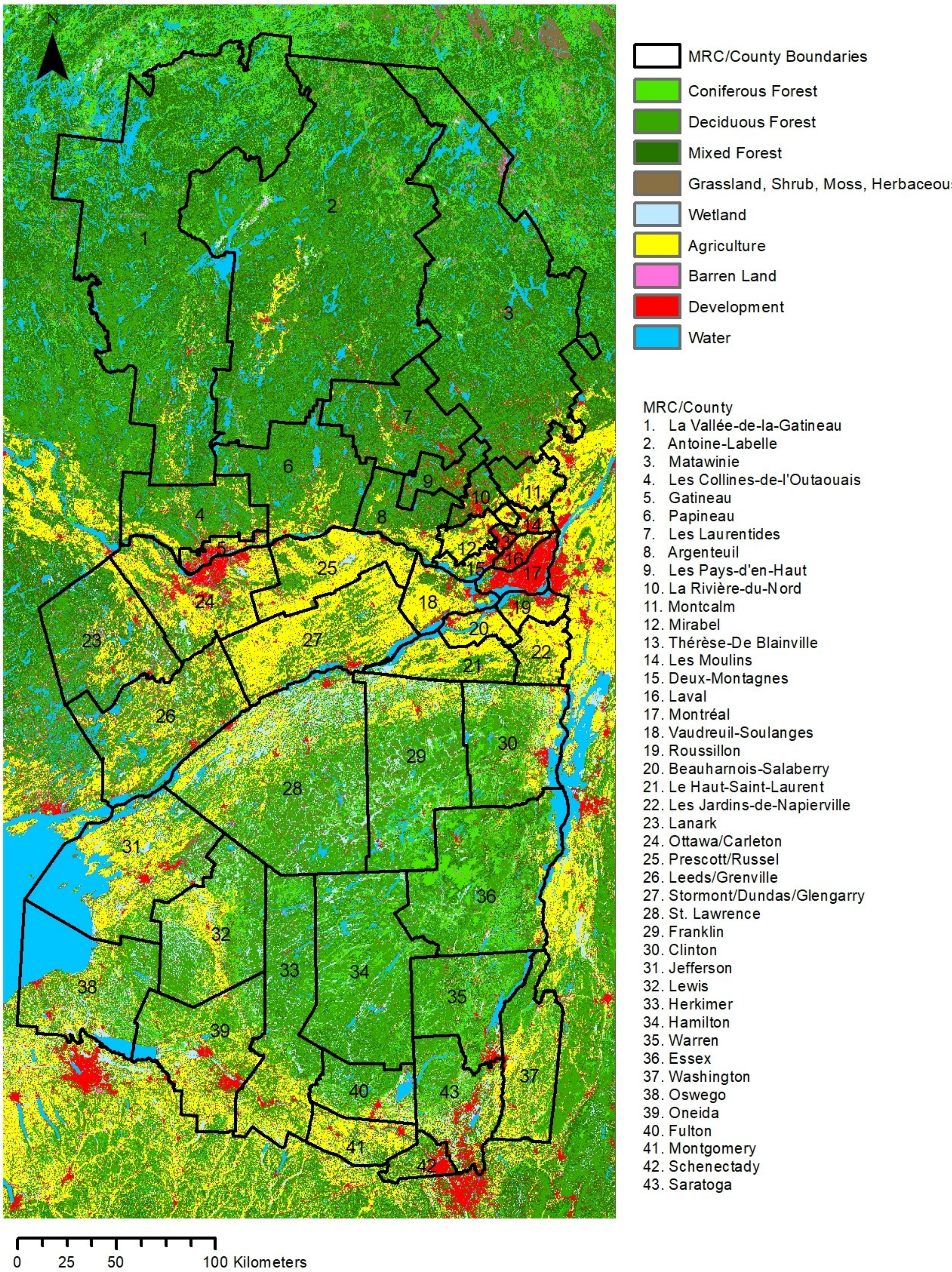

Figure 3.1 Land-cover map of the Adirondack-to-Laurentians (A2L) study area overlaid with municipalité régionale de comté (MRC)/county boundaries. MRC/county names are numbered and correspond to the numbers on the map.

This area was included as a buffer-zone to eliminate the overestimation of resistance values at artificial map boundaries during the least-cost path and Circuitscape analyses (Koen et al. 2010). We utilized these distinct hierarchical scales to: (1) allow for the direct comparison and ranking between provincial/state portions and MRCs/counties; and (2) provide stakeholders with the information necessary to develop coordinated and collaborative local, regional, and transboundary conservation plans.

Suitable habitat and resistance models

Suitable habitat

We estimated suitable habitat for each species by assigning relative values to our land-cover maps using a combination of previously published values, review of the literature, and expert opinion (Table S3.4). We then re-scaled the suitable habitat values for each species so that the values ranged between 0 and 1 using the following equation:

$$F(x) = (x - \min) / (\max - \min),$$

where x is the assigned relative suitability value for a 30 m grid cell, and min and max are the minimum and maximum suitable habitat values of the habitat suitability surface, respectively (Keeley et al. 2016; Table S3.5). Values near one represent the most suitable habitat conditions and values near zero represent the least suitable habitat (Keeley et al. 2016). For each species, we created one aggregated suitable habitat map by overlaying all six layers (i.e., land cover, roads, distance from development; distance from primary roads; distance from secondary roads; and distance from tertiary roads; Figure S5) in ArcGIS10.7, using Gnarly Landscape Utilities: Resistance and Habitat Calculator toolset (McRae et al. 2013), and retaining the minimum suitability value for each 30 m cell across all input layers.

Resistance layers

We created movement resistance layers for each species that represent the relative probability that the species will avoid a particular land-cover. Thus, we derived resistance values for each of the six raster layers for each species (24 raster layers total; Table S3.5) by calculating the linear inverse of our suitable habitat values (Koen et al. 2012; Keeley et al. 2016) using the following equation:

$$F(x) = 1 - (x/100),$$

where x is the habitat suitability value. We then used Gnarly Landscape Utilities: Resistance and Habitat Calculator toolset (McRae et al. 2013) in ArcGIS10.7 to overlay all six resistance layers; we created a single aggregated resistance layer for each species by retaining the maximum resistance value for each cell across all six input layers (McRae et al. 2013). We added a value of one to each cell, such that habitats with a relatively low movement cost had a value of 1, and habitats with a high cost had values up to a maximum of 101. Bowman et al. (2020) found that landscape connectivity models tend to be insensitive to absolute cost values, provided that the rank order is correct.

Species-specific habitat patches

To identify species-specific habitat patches, we used our aggregated suitable habitat and resistance layers and the software Gnarly Landscape Utilities: Core Mapper toolset (Shirk and McRae 2013) in ArcGIS10.7. Suitable habitat patches (SHPs) were identified as patches with an average suitable habitat value ≥ 0.6 within a circular moving window with a radius based on home range size (Appendix 1). Patches that fell below the species' minimum habitat patch cut-off size (Appendix 1) were removed. Because animals move among multiple habitat patches to obtain the resources they need within their home ranges, we expanded habitat patches outwards up to a total cost-weighted distance equal to each species' minimum home range radius (Tables S3.6 and S3.7) to potentially link proximate patches into larger aggregates, simulating intra-patch connectivity (Spanowicz and Jaeger 2019). Habitat patches still separated at this point require movements that exceed twice the species' cost-weighted mean minimum home range radius and were considered dispersal distances (i.e., inter-patch connectivity).

We identified optimal habitat patches (OHPs) by performing the same steps as above, however, we did not expand the patches, and we removed all raster cells with suitable habitat values ≤ 0.4 (black bear, fisher, and moose), and ≤ 0.2 (white-tailed deer) (Table S3.7) to exclude non-habitat types (i.e., roads, development, agriculture, etc.), leaving patches that represented the most suitable habitat. This removal of non-habitat fragmented the original patches further, creating significantly more patches. However, many of these patches fell below the species' minimum habitat patch cut-off size (Appendix 1) and were removed.

We identified stepping-stone patches (SSPs) by performing the same steps as above, however, this time we identified patches that were smaller than the species' minimum habitat patch cut-off size (Appendix 1) but still large enough to serve as a refuge area during dispersal (≥ 10 km^2 for black bear and moose, ≥ 5 km^2 for fisher, and ≥ 1 km^2 for white-tailed deer; Table S3.7).

Evaluation of the suitable habitat and habitat patch models

We used two empirical datasets collected in the Québec and Ontario portions of the study area to evaluate our suitable habitat layers: (1) unpublished trapping/harvest data for black bear, moose, and white-tailed deer provided by the Québec Ministère des Forêts, de la Faune et des Parcs, consisting of n=131,039 trapping/harvest GPS locations collected in 1998–2002 and 2014–2019; and (2) previously published radiotelemetry data for fishers (Koen et al. 2007, 2014), consisting of n=1083 locations obtained by triangulation for 26 adult fishers (10 M, 16 F) between 2003 and 2004. Hereafter, we refer to these two datasets together as "evaluation points". We evaluated the fisher suitable habitat layers for the year 2000 only, because we did not have evaluation points for this area for 2015. We were also unable to obtain similar evaluation points for the New York portion, as the New York State - Department of Conservation does not collect harvest data with high-resolution GPS locations. We did not use data points obtained from citizen science programs (e.g., iNaturalist) in our validation process because of the potential for spatial biases that may be present in the GPS locations (see Dickinson et al. 2010). The consequence of not using citizen science data was that we were only able to validate our suitable habitat maps for a portion of the study area.

Typically, species move across the landscape differently during winter months when snow is on the ground (i.e., some can cross frozen lakes in winter that they cannot cross in summer), or not at all (hibernating black bear). Thus, we used only evaluation points obtained between April 1st and November 30th of each year to characterize movement ability during spring, summer, and fall. Since the evaluation points only covered subsections of the study area, we delineated these subsections by creating a 100% minimum convex polygon (MCP) around all the data points for each species (Koen et al. 2007; Brodeur et al., 2008) using the "Convex Hull" function in ArcGIS10.7 (Figures S3.1-S3.4).

We used three metrics to assess the performance of each map of suitable habitat (SH) (i.e., how well each map predicted suitable habitat for each species within the local landscape). First, we used the absolute validation index (*AVI*; Hirzel and Arlettaz 2003), calculated as the proportion of evaluation points that were located on raster cells with an SH value > 0.5 (Hirzel et al. 2006; Guisan et al. 2017):

AVI = Number of evaluation points found on raster cells with a SH value > 0.5 within the MCP ∕ Number of evaluation points within the MCP

Values for the AVI ranged between 0 (weak performance) and 1 (strong performance). Second, we used the contrast validation index (CVI; Hirzel et al. 2004, 2006), calculated as the AVI minus the AVI of a random chance model predicting presence within the MCP (Hirzel et al. 2006; Guisan et al. 2017):

$CVI = AVI -$ Number of raster cells within the MCP with a SH value > 0.5 ∕ Number of raster cells within the MCP

Values for the CVI range between −0.5 (weak performance) and 0.5 (strong performance). Finally, we used the *Boyce Index* (Boyce et al. 2002; Hirzel et al. 2006; Guisan et al. 2017), whereby we calculated two frequencies for each of the 6 suitable habitat classes (i.e., 1, 0.8, 0.6, 0.4, 0.2, 0): (1) the proportion of observed evaluation points found in each SH class within the MCP (P); and (2) the expected proportion of evaluation points found in each SH class within the MCP (E) (Boyce et al. 2002; Hirzel et al. 2006). We then used these values to calculate the P/E ratio for each class. If the SH model predicted suitable habitat well, then a low SH class should contain fewer evaluation points than expected by chance (i.e., a P/E ratio > 1; Hirzel et al. 2006; Guisan et al. 2017). The *Boyce Index* can then be calculated using the Spearman rank correlation coefficient between the SH value and the P/E ratio (Boyce et al. 2002; Hirzel et al. 2006). A SH model that performs well is expected to show a monotonically increasing relationship between the SH value and the P/E ratio (Hirzel et al. 2006; Guisan et al. 2017). *Boyce Index* values ranged between −1 (an incorrect model) and 1 (a model whose predictions are consistent with the evaluation dataset); values close to zero indicate the model is no different from a chance model (Hirzel et al. 2006; Guisan et al. 2017).

To measure the performance of the habitat patch (HP) models, we applied variations of the AVI and CVI metrics. We used the AVI_{patch} to calculate the proportion of evaluation points that were located within SHPs and OHPs as follows,

AVI_{patch} = Number of evaluation points in HPs within the MCP ∕ Number of evaluation points within the MCP

Values for the AVI_{patch} ranged between 0 (weak performance) and 1 (strong performance). We used the CVI_{patch} as follows:

$$CVI_{\text{patch}} = AVI_{\text{patch}} - \text{Area of HPs within the MCP (km}^2) / \text{Area of MCP (km}^2)$$

Values for the CVI_{patch} ranged between -0.5 (weak performance) and 0.5 (strong performance).

Land-cover change

To quantify land-cover change, we measured and compared the area of three groups of land-cover classes between 2000 and 2015: (1) Natural land-cover, which included coniferous forest, deciduous forest, mixed forest, grassland, shrub, moss, and herbaceous vegetation, and wetlands (2) Agriculture, which included all agriculture classes, and (3) Development, which included all development classes. Land-cover area was calculated by multiplying the cell count of each land-cover class within the boundaries of the reporting unit (i.e., study area, provincial/ state portion, MRC/county) by the area of a single cell (900 m^2), and then dividing by 1,000,000 m^2/km^2 to convert to km^2.

Species-specific habitat amount and fragmentation

To quantify changes in species-specific habitat amount, we measured and compared the area of SHPs and OHPs between 2000 and 2015 within each reporting unit in ArcGIS10.7. We calculated habitat proportion by dividing the habitat area by the total area of the reporting unit.

The effective mesh size is based on the average probability that any two randomly chosen points in the study area are connected (i.e., not separated by a fragmentation barrier; Jaeger 2000; Moser et al. 2007). The effective mesh size also serves as a measure of structural connectivity, or the degree to which movement between different parts of the landscape is possible (Spanowicz and Jaeger 2019). Because the value of the effective mesh size can be profoundly influenced by the boundary of a reporting unit, we used two variations of the effective mesh size. First, we used the "cutting out" procedure ($m_{\text{eff\_CUT}}$) to measure fragmentation strictly within the boundaries of the reporting units. Second, we used the "cross-boundary connections" procedure ($m_{\text{eff\_CBC}}$) to include patches that crossed borders into adjacent reporting units. We performed all measurements using the effective mesh size tool from the ZonalMetrics ArcGIS toolbox (Wetzel 2019).

Species-specific connectivity

Least-cost paths and least-cost corridors

When modeling inter-patch connectivity, we assumed that each species would take the lowest-resistance path between two patches (i.e., the least cost path (LCP)). This gives one best-case measure of connectivity between patches across the resistance surface (Adriaensen et al. 2003). LCPs assume that animals can determine the single optimal path (Fletcher and Fortin 2018) and the method can be sensitive to the specific choice of resistance values used (Rayfield et al. 2010). As such, we also calculated least-cost corridors (LCCs) between patches. The LCC method relaxes the assumption of single best paths by calculating corridors representing similarly low-cost movement (Pinto and Kiett, 2009; Fletcher and Fortin 2018). We used the Linkage Pathways tool of the Linkage Mapper ArcGIS Toolbox (McRae and Kavanagh 2011) to create species-specific LCPs and LCCs. For the calculation of LCCs with this software, we calculated adjacency using both cost-weighted and Euclidean distances, dropped corridors that intersected other habitat patches (i.e., to reduce corridor lengths and identify the ideal least-cost path between each pair of patches), put no limit on the number of linkages originating from each habitat patch, and truncated the width of LCCs to 200 cost-weighted km. As a rule of thumb, ecological corridors should be at least 2 km wide, except at unavoidable bottlenecks such as wildlife crossing structures (Beier et al., 2008; Beier, 2012; Brost & Beier 2012; Beier 2018). We used a least-cost corridor cut-off width of 200 cost-weighted km to ensure that even when corridors traversed regions with the highest resistance values (101), corridors would still maintain a width of at least 2 km wide (i.e., if the lowest resistance = 1 cost-weighted km, 202 cost-weighted km = 2.02 km on the ground).

Prior to running the Linkage Pathways software, we increased the cell size of the suitable habitat and resistance layers to 90 m×90 m using the “Raster cell size coarsener” tool in Gnarly Landscape Utilities: Resistance and Habitat Calculator (Shirk and McRae 2013), to reduce computing time and memory use. The tool increased habitat and resistance raster cell sizes by smoothing grid cell values (i.e., taking the average in the NxN window) and then coarsening the result to a larger cell size (i.e., taking the average value of smoothed values in the NxN window; Shirk and McRae 2013).

Calculation of dispersal distance

To determine if any of the LCCs were too long to be considered dispersal corridors, we needed to define species-specific median and maximum dispersal distances; these data were not available for our focal species in our study area. Instead, we estimated median and maximum dispersal distances for each species based on the relationship between home range size (derived from the literature; Table S3.6) and dispersal distance (Table S3.8; Bowman et al. 2002),

Median dispersal distance = 7 · (linear dimension of home range)

Maximum dispersal distance = 40 · (linear dimension of home range)

Identifying pinch-points

Pinch-points are narrow sections within a corridor where movement is restricted due to natural or anthropogenic landscape features (McRae and Shah 2011; Pelletier et al. 2014). Pinch-points can be conservation and restoration priorities as their loss can disproportionately disrupt connectivity (McRea et al. 2008; Dickson et al. 2013). To identify pinch-points within the connectivity corridors, we used the Pinch-Point Mapper tool of the Linkage Mapper ArcGIS Toolbox (McRae 2012). Pinch-Point Mapper uses Circuitscape (McRae and Shah 2011) to simulate the path of electric current through the LCCs, based on local resistances along the LCCs. This method assumes that individuals follow random walks through the LCCs, with a probability of moving into a location from a neighbouring one, dependent on the resistance of the location (McRea et al. 2008). We used the "pairwise" mode within the Linkage Mapper Toolbox to identify pinch-points between SHPs (i.e., using the centre of the SHPs as a node). We used the "all to one" mode within the Linkage Mapper Toolbox, where current flows from all source nodes (i.e., SHPs) iteratively to each ground node, to produce cumulative current density maps where areas of high current flow were identified as pinch-points critical for maintaining connectivity for the entire network of SHPs (McRea et al. 2008; Dutta et al. 2016).

Measuring connectivity

To quantify changes in species-specific connectivity between 2000 and 2015, we compared the values for Euclidean distance, cost-weighted distance, least-cost path length, and effective resistance (output from the Circuitscape runs) between SHPs for each species. We assumed that if connectivity had diminished for a specific species, then these distances would have increased significantly. We compared the distances between time points using a Welch two-sample *t*-test to account for unequal variances. A paired *t*-test was not appropriate because some of the SHPs disappeared in 2015 due to habitat loss. We used Cohen's effect size to further assess the change in connectivity between 2000 and 2015 (d=0.2 represents a small effect size, d=0.5 represents a medium effect size, and d=0.8 represents a large effect size; Cohen 1988) using the rstatix package (Kassambara 2023) in R Studio.

Habitat patches and corridors under protection

To determine the percentage of species-specific SHPs, OHPs, and LCCs under legal protection, we obtained maps of government protected areas and private protected areas secured by Nature Conservancy of Canada/The Nature Conservancy (Table S3.1). We measured the proportion of each SHP, OHP, or LCC currently under protection in ArcGIS10.7. To determine which habitat patches and corridors were used by all species, we overlaid the species-specific SHP, OHP, and LCC layers in ArcGIS10.7 to create an intersect map. Our assumption was that conservation in these portions would be beneficial to all species.

Results

Evaluation of the suitable habitat and habitat patch models

Our maps of suitable habitat performed well at predicting suitable habitat for each species within the local landscape. *AVI* values ranged from 0.8 to 0.9 in 2000 and 2015 (Table 3.1), *CVI* values were 0.2 in 2000, and ranged from 0.1 to 0.3 in 2015 (Table 3.1), and *Boyce Index* values ranged from 0.8 to 0.9 in 2000 and 2015 (Table 3.1).

The suitable habitat patch (SHP) models performed well at predicting SHPs for each species within the local landscape. AVI_{patch} values ranged from 0.8 to 0.9 in 2000, and from 0.7 to 0.9 in 2015 (Table 3.1), and CVI_{patch} values ranged from 0.08 to 0.1 in 2000, and from 0.06 to 0.3 in 2015 (Table 3.1). The optimal habitat patch (OHP) models also performed well for black bear, moose, and white-tailed deer: AVI_{patch} values were 0.7 in 2000, and ranged from 0.5 to 0.7 in 2015, and CVI_{patch} values ranged from 0.03 to 0.2 in 2000, and 0.05 to 0.3 in 2015 (Table 3.1). However, the OHP model for fisher showed weak performance; the AVI_{patch} value was 0.1 and the CVI_{patch} value was − 0.03. Because the CVI_{patch} value was negative, we did not use the fisher OHPs in any calculations. For R code and calculations see Appendix 1.

Land-cover change

Between 2000 and 2015, natural land-cover (i.e., coniferous forest, deciduous forest, mixed forest, grassland, shrub, moss, and herbaceous vegetation, and wetlands) decreased by 1457 km^2 across the study area; with losses of 587 km^2 in the Québec portion, 966 km^2 in Ontario portion, and a gain of 96 km^2 in the New York portion (Table 3.2). Agriculture also decreased across the study area; with losses of 201 km^2 in the Québec portion, 148 km^2 in the New York portion, and a gain of 194 km^2 in the Ontario portion (Table 3.2). Development

Table 1A

Species	2000				2015			
	AVI	*CVI*	*Boyce Index*	*p*-value	*AVI*	*CVI*	*Boyce Index*	*p*-value
Black Bear	0.9	0.2	0.9	0.03	0.9	0.2	0.8	0.1
Fisher	0.9	0.2	0.9	0.02	-	-	-	-
Moose	0.9	0.2	0.8	0.06	0.9	0.3	0.9	0.03
White-Tailed Deer	0.8	0.2	0.9	0.02	0.8	0.1	0.9	0.02

Table 1B

Species	Model	2000		2015	
		AVI_{patch}	CVI_{patch}	AVI_{patch}	CVI_{patch}
Black Bear	SHP	0.9	0.09	0.8	0.1
	OHP	0.7	0.08	0.5	0.08
Fisher	SHP	0.8	0.1	-	-
	OHP	0.1	-0.03	-	-
Moose	SHP	0.9	0.1	0.9	0.3
	OHP	0.7	0.2	0.7	0.3
White-Tailed Deer	SHP	0.9	0.08	0.7	0.06
	OHP	0.7	0.03	0.6	0.05

Table 3.1 A) Results of suitable habitat model validation for each species in 2000 and 2015, and **B)** Results of habitat patch model validation for each species in 2000 and 2015. *AVI* = Absolute validation index, AVI_{patch} = Absolute validation index for patches, *CVI* = Contrast validation index, CVI_{patch} = Contrast validation index, SHP = Suitable habitat patch, OHP = Optimal habitat patch.

Location	2000 (km^2)	2015 (km^2)	2015-2000 (km^2)	Percent Change (%)
Study Area				
Natural Land Cover	93210	91752	-1457	-2
Agriculture	17726	17570	-155	-1
Development	4332	5743	1410	33
Quebec Portion				
Natural Land Cover	46290	45702	-587	-1
Agriculture	4749	4549	-201	-4
Development	1361	2194	833	61
Ontario Portion				
Natural Land Cover	8202	7236	-966	-12
Agriculture	5629	5823	194	3
Development	564	1010	445	79
New York Portion				
Natural Land Cover	38718	38814	96	0.2
Agriculture	7347	7199	-148	-2
Development	2407	2539	132	5

Table 3.2 Changes in land-cover area (km^2) and (%) for each set of grouped land-cover categories between 2000 and 2015, at the scale of the study area and each provincial/state portion.

increased by 1410 km^2 across the study area; with increases of 833 km^2 in the Québec portion, 445 km^2 in the Ontario portion, and 132 km^2 in the New York portion (Table 3.2). As of 2015, there were 17 MRCs/counties (13 in the Québec portion, 3 in the Ontario portion, and 1 in the New York portion) that had > 50% of their surface areas dedicated to agriculture and development (Table S3.9). The proportion of natural land-cover within these MRCs/counties ranged from 11% (Montréal) to 44% (MRC Montcalm; Table S3.9). In the three Ontario counties, natural land-cover decreased by 289 km^2 (Stormont/Dundas/Glengarry), 232 km^2 (Ottawa/Carleton), and 96 km^2 (Prescott/Russel; Table S9); whereas natural land-cover increased in all the Québec MRCs, except MRC Gatineau (-34 km^2), and Montgomery County (-1 km^2) in New York (Table S9). Agriculture increased in Stormont/ Dundas/Glengarry (154 km^2), Ottawa/Carleton (52 km^2), Prescott/Russel (4km^2) and Gatineau (17 km^2; Table S9); and development increased in all MRCs/ counties, except Montréal and MRC Gatineau in Québec (Table S3.9).

Species-specific habitat amount

We detected net losses of both suitable and optimal habitat patch area for all four species between 2000 and 2015. The greatest SHP area loss within the study area occurred for moose, with a reduction of 16,842 km^2 (26%), followed by black bear with a reduction of 8894 km^2 (11%) (Table 3.3, Figs. 3.2 and 3.3). Most of these losses took place in the Québec portion of the study area where SHP area for moose was reduced by 13,382 km^2 (28%) and SHP area for black bear was reduced by 6891 km^2 (14%) (Table 3.3, Figs. 3.2 and 3.3). The greatest OHP area loss also occurred for moose with a reduction of 6832 km^2 (17%), followed by black bear with a reduction of 4369 km^2 (9%) (Table 3.3, Figs. 3.2 and 3.3). The vast majority of these losses also took place in the Québec portion of the study area, where OHP area for moose was reduced by 6148 km^2 (21%) and OHP area for black bear was reduced by 4487 km^2 (14%) (Table 3.3, Figs. 3.2 and 3.3). The Ontario portion of the study area had the lowest proportion of suitable and optimal habitat patch area for each species in 2000, and the greatest relative reductions of habitat patch area between 2000 and 2015. In Ontario, SHP area was reduced by 95% for moose, 62% for black bear, 38% for fisher, and 30% for white-tailed deer. OHP area in Ontario was reduced by 89% for moose, 65% for black bear, and 30% for white-tailed deer (Table 3.3 and Figs. 3.2 and 3.3). Whereas in New York, SHP area increased by 26 km^2 for black bear, 83 km^2 for white-tailed deer, and OHP area increased by 390 km^2 for black bear (Table 3.3 and Figs. 3.2 and 3.3).

Species-specific fragmentation

We detected a net increase in suitable and optimal habitat patch fragmentation for all species within the study area between 2000 and 2015. The highest level of SHP fragmentation was for moose, with an 8674 km^2 (46%) reduction in $m_{\mathrm{eff\_CUT}}$ size, and an 11,918 km^2 (42%) reduction in $m_{\mathrm{eff\_CBC}}$ size (Table 3.4). In the Québec portion, moose also had the highest level of SHP fragmentation with an 18,672 km^2 (49%) reduction in $m_{\mathrm{eff\_CUT}}$ size and a 25,606 km^2 (44%) reduction in $m_{\mathrm{eff\_CBC}}$ size (Table 3.4). This was followed by black bear with a 10,578 km^2 (26%) reduction in $m_{\mathrm{eff\_CUT}}$ size and a 13,949 km^2 (23%) reduction in $m_{\mathrm{eff\_CBC}}$ size. In the Ontario portion, moose had a 298 km^2 (99%) reduction in $m_{\mathrm{eff\_CUT}}$ size and a 622 km^2 (99%) reduction in $m_{\mathrm{eff\_CBC}}$ size (Table 3.4). For black bear, we detected a 236 km^2 (81%) reduction in $m_{\mathrm{eff\_CUT}}$ size

A.

Location/Species	SHP area in 2000 (km²)	SHP area in 2015 (km²)	SHP area 2015-2000 (km²)	Proportion of SHP area (2000, %)	Proportion of SHP area (2015, %)	Percent change 2000 to 2015 (%)
Study Area						
Black Bear	77690	68795	-8895	61	54	-11
Fisher	86348	84555	-1793	68	66	-2
Moose	64266	47424	-16842	50	37	**-26**
White-Tailed Deer	89303	86147	-3156	70	68	-4
Québec Portion						
Black Bear	48615	41724	-6891	83	71	-14
Fisher	48978	48858	-119	83	83	-0.2
Moose	47283	33902	-13382	80	58	**-28**
White-Tailed Deer	49564	48488	-1076	84	82	-2
Ontario Portion						
Black Bear	3286	1256	-2030	21	8	**-62**
Fisher	3971	2447	-1524	26	16	**-38**
Moose	3575	174	-3401	23	1	**-95**
White-Tailed Deer	7127	4964	-2163	46	32	**-30**
New York Portion						
Black Bear	25789	25816	26	49	49	0.1
Fisher	33400	33250	-150	63	63	-0.4
Moose	13408	13348	-59	25	25	-0.4
White-Tailed Deer	32613	32696	83	61	62	0.3

B.

Location/Species	OHP area in 2000 (km²)	OHP area in 2015 (km²)	OHP area 2015-2000 (km²)	Proportion of OHPs in reporting unit (2000, %)	Proportion of OHPs in reporting unit (2015, %)	Percent change 2000 to 2015 (%)
Study Area						
Black Bear	46344	41975	-4369	36	33	-9
Fisher	-	-	-	-	-	-
Moose	40717	33884	-6832	32	27	-17
White-Tailed Deer	77361	73979	-3383	61	58	-4
Québec Portion						
Black Bear	31124	26637	-4487	53	45	-14
Fisher	-	-	-	-	-	-
Moose	29796	23648	-6148	51	40	**-21**
White-Tailed Deer	42682	41461	-1222	73	70	-3
Ontario Portion						
Black Bear	416	144	-271	3	1	**-65**
Fisher	-	-	-	-	-	-
Moose	340	39	-301	2	0.3	**-89**
White-Tailed Deer	5729	3991	-1738	37	26	**-30**
New York Portion						
Black Bear	14804	15194	390	28	29	3
Fisher	-	-	-	-	-	-
Moose	10581	10197	-383	20	19	-4
White-Tailed Deer	28950	28527	-423	55	54	-1

Table 3.3 A) Changes in suitable habitat patch (SHP), and **B)** Optimum habitat patch (OHP) area (km^2) and proportion (%) for each species between 2000 and 2015, at the scale of the study area and each provincial/state portion. Bold numbers = Greater than 25% reduction between 2000 and 2015.

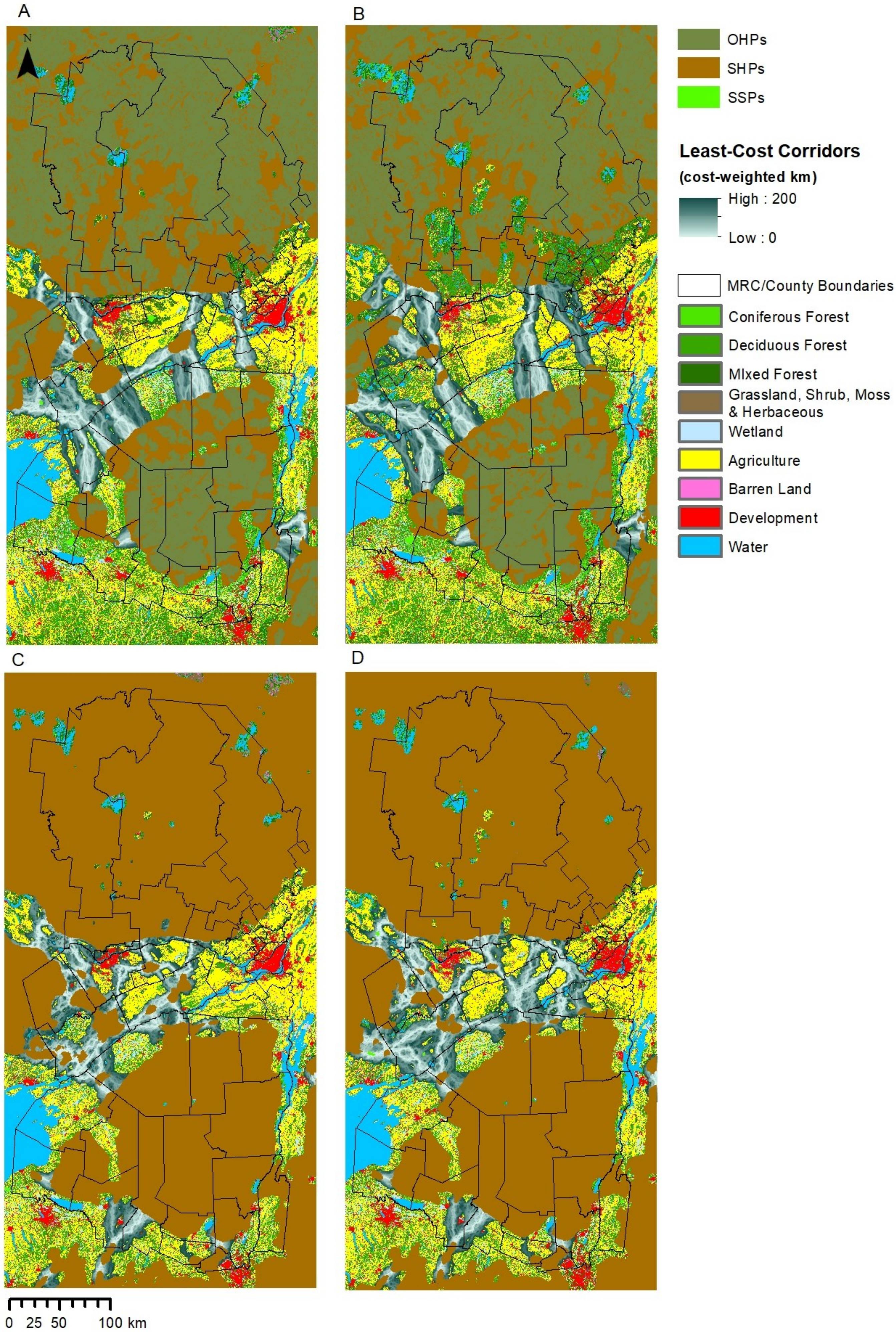

Figure 3.2 Changes in suitable habitat patches (SHP), optimal habitat patches (OHP), stepping-stone patches (SSP), and least-cost corridors (LCC) between 2000 and 2015. A) Black bear habitat 2000, B) Black bear habitat 2015, C) Fisher habitat 2000, D) Fisher habitat 2015.

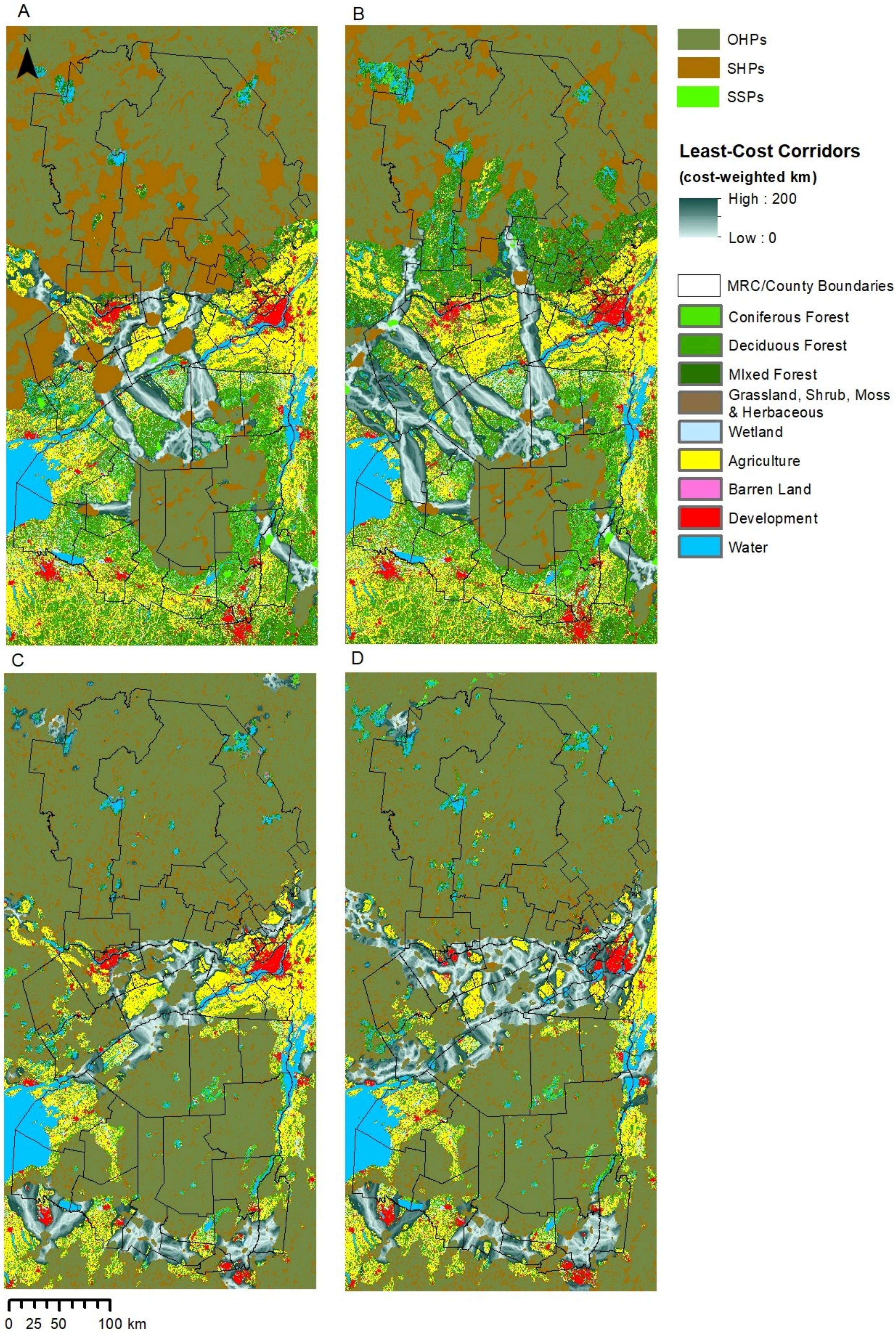

Figure 3.3 Changes in suitable habitat patches (SHP), optimal habitat patches (OHP), stepping-stone patches (SSP), and least-cost corridors (LCC) between 2000 and 2015. A) Moose habitat 2000, B) Moose habitat 2015, C) White-tailed deer habitat 2000, D) White-tailed deer habitat 2015.

and a 432 km^2 (73%) reduction in $m_{\mathrm{eff\_CBC}}$ size. For white-tailed deer, we found a 1051 km^2 (52%) reduction in $m_{\mathrm{eff\_CUT}}$ size and a 28,170 km^2 (94%) reduction in $m_{\mathrm{eff\_CBC}}$ size (Table 3.4). The lowest level of SHP fragmentation for each species was in New York, with $m_{\mathrm{eff\_CUT}}$ and $m_{\mathrm{eff\_CBC}}$ size reductions of less than 3%, with the exception being white-tailed deer, which had a 3619 km^2 (17%) reduction in $m_{\mathrm{eff\_CBC}}$ size (Table 3.4). At the level of the MRC/county, mean values for $m_{\mathrm{eff\_CUT}}$, $m_{\mathrm{eff\_CBC}}$, and the difference between the $m_{\mathrm{eff\_CUT}}$ and $m_{\mathrm{eff\_CBC}}$ (a measure of patch sharing between adjacent reporting units) decreased for each species between 2000 and 2015, except for fisher, for which we detected an increase in both the mean $m_{\mathrm{eff\_CBC}}$ and the difference between the $m_{\mathrm{eff\_CUT}}$ and $m_{\mathrm{eff\_CBC}}$ values (Tables S3.10-S3.16).

The highest level of OHP fragmentation that we detected was for moose, with a 1592 km^2 (71%) reduction in $m_{\mathrm{eff\_CUT}}$ size and an 1865 km^2 (71%) reduction in $m_{\mathrm{eff\_CBC}}$ size (Table 3.4). This was followed by black bear with a 1623 km^2 (70%) reduction in $m_{\mathrm{eff\_CUT}}$ size and a 1908 km^2 (70%) reduction in $m_{\mathrm{eff\_CBC}}$ size (Table 3.4). This same pattern was also observed in the Québec and Ontario portions of the study area. In the New York portion, we found that white-tailed deer had the highest level of OHP fragmentation with a 283 km^2 (59%) reduction in $m_{\mathrm{eff\_CUT}}$ size, and a 284 km^2 (59%) reduction in $m_{\mathrm{eff\_CBC}}$ size. (Table 3.4). At the level of the MRC/county, mean values for $m_{\mathrm{eff\_CUT}}$, $m_{\mathrm{eff\_CBC}}$, and the difference between the $m_{\mathrm{eff\_CUT}}$ and $m_{\mathrm{eff\_CBC}}$ decreased for each species between 2000 and 2015 (Tables S3.10-S3.16).

Species-specific connectivity

Least-cost paths and least-cost corridors

For black bear, 8 of the 14 LCPs between SHPs in 2000 were longer than the median female dispersal distance of 40.1 km and 3 of the 14 LCPs were longer than the median male dispersal distance of 91.7 km (Table S3.8). However, all the LCPs were less than both the maximum female and male dispersal distances of 229.1 km and 523.8 km, respectively (Table S3.8). In 2015, 6 of the 12 LCPs were longer than the median female black bear dispersal distance, and 3 LCPs were longer than the median male black bear dispersal distance; however, all the LCPs were less than the female and male maximum dispersal distances (Table S3.8). For fisher, 13 of the 31 LCPs in 2000 were longer than the median female dispersal distance of 26.6 km (Table S3.8), and 10 of the 31 were longer than the median male dispersal distance of 33.9 km.

A.

Species/Location	m_{eff_CUT} 2000 (km²)	m_{eff_CUT} 2015 (km²)	Change in m_{eff_CUT} 2015-2000 (km²)	m_{eff_CBC} 2000 (km²)	m_{eff_CBC} 2015 (km²)	Change in m_{eff_CBC} 2015-2000 (km²)	m_{eff_CBC} - m_{eff_CUT} 2015 (km²)
Study Area							
Black Bear	23181	18275	-4907	33000	26510	-6491	8235
Fisher	27243	27026	-217	38417	38312	-105	11287
Moose	18799	10125	-8674	28326	16408	-11918	6283
White-Tailed Deer	31180	25655	-5524	45018	35213	-9805	9557
Québec Portion							
Black Bear	40017	29439	-10578	61212	47263	-13949	17824
Fisher	40677	40151	-526	61580	61533	-47	21382
Moose	37979	19308	-18672	58509	32903	-25606	13595
White-Tailed Deer	40912	38752	-2159	69924	59361	-10563	20609
Ontario Portion							
Black Bear	293	57	-236	595	162	-432	105
Fisher	345	196	-150	742	493	-249	297
Moose	300	2	-298	627	5	-622	3
White-Tailed Deer	2025	974	-1051	29965	1794	-28170	820
New York Portion							
Black Bear	11091	11102	11	11134	11152	19	50
Fisher	20106	19969	-137	23678	23552	-126	3583
Moose	2907	2884	-23	2907	2884	-23	0
White-Tailed Deer	18121	17618	-503	21763	18144	-3619	526

B.

Species/Location	m_{eff_CUT} 2000 (km²)	m_{eff_CUT} 2015 (km²)	Change in m_{eff_CUT} 2015-2000 (km²)	m_{eff_CBC} 2000 (km²)	m_{eff_CBC} 2015 (km²)	Change in m_{eff_CBC} 2015-2000 (km²)	m_{eff_CBC} - m_{eff_CUT} 2015 (km²)
Study Area							
Black Bear	2334	711	-1623	2729	821	-1908	110
Fisher	-	-	-	-	-	-	-
Moose	2237	645	-1592	2615	750	-1865	104
White-Tailed Deer	2406	981	-1425	2724	1145	-1578	165
Québec Portion							
Black Bear	4848	1366	-3482	5702	1603	-4099	237
Fisher	-	-	-	-	-	-	-
Moose	4669	1252	-3417	5487	1478	-4009	226
White-Tailed Deer	4773	1945	-2829	5459	2301	-3158	357
Ontario Portion							
Black Bear	2.4	0.8	-2	3.2	1.0	-2	0.2
Fisher	-	-	-	-	-	-	-
Moose	2.0	0.1	-2	2.7	0.2	-2	0.2
White-Tailed Deer	7.8	5.8	-2	8.7	6.2	-3	0.4
New York Portion							
Black Bear	225	191	-33	225	191	-33	0.04
Fisher	-	-	-	-	-	-	-
Moose	189	160	-29	189	160	-29	0
White-Tailed Deer	478	195	-283	479	195	-284	0.1

Table 3.4 A) Changes in Effective Mesh Size for suitable habitat patches (SHPs) and **B)** Optimum habitat patches (OHPs) for each species between 2000 and 2015, at the scale of the study area and each provincial/state portion.

However, all the LCPs were less than both the maximum female and male dispersal distances of 151.8 km and 193.5 km, respectively (Table S3.8). In 2015, 13 of the 27 LCPs were longer than the median female and male fisher dispersal distances; however, all the LCPs were less than the female and male maximum dispersal distances (Table S3.8). For moose, 8 of the 16 LCPs in 2000 and 9 of the 12 LCPs in 2015 were longer than the median moose dispersal distance of 44.6 km; however, all the LCPs in 2000 and 2015 were less than the maximum moose dispersal distance of 260.5 km (Table S3.8). In contrast, for white-tailed deer, 37 of the 72 LCPs in 2000, and 77 of the 114 LCPs in 2015, were longer than the median white-tailed deer dispersal distance of 10.1 km; and 1 of the 72 LCPs in 2000, and 4 of the 114 LCPs in 2015, were longer than the maximum dispersal distance of 58 km (Table S3.8). The number of LCCs decreased for each species between 2000 and 2015, except for white-tailed deer, which had an increase of 42 LCCs (Table 3.5, Figs. 3.2 and 3.3).

Changes in connectivity

Euclidean distances between SHPs increased for each species between 2000 and 2015, with fisher, moose, and white-tailed deer distances being statistically significant (*p*-values=0.03, 0.03, 0.06 respectively; Table 3.5). The greatest change in Euclidean distances was for moose, with a mean increase of 41 km (*df*=16, 95% *CI*=-77 - -5, *Cohen's d*=0.9; Table 5). Cost-weighed distances between SHPs increased for each species between 2000 and 2015: increases in fisher, moose, and white-tailed deer cost-weighted distances were significant (*p*-values=0.02, 0.03, 0.08 respectively; Table 3.5). The greatest change in cost-weighted distance was for moose, with a mean increase of 1954 cost-weighed km (*df*=16, 95% *CI*=-3634 - -275, *Cohen's d*=0.9; Table 3.5). LCP distances also increased for each species, with significant increases for fisher, moose, and white-tailed deer (*p*-values=0.05, 0.03, 0.04 respectively; Table 3.5). The largest increase in LCP distance was for moose with a mean increase of 58 km (*df*=16, 95% *CI*=-107 - -8, *Cohen's d*=0.9; Table 3.5). Effective resistance values also increased for each species, with significant increases for fisher, moose, and white-tailed deer (*p*-values=0.05, 0.05, 0.02 respectively; Table 3.5). The largest increase in effective resistance was for moose, with a mean increase of 437 Ohms (*df*=17, 95% *CI*=-8776 - 33, *Cohen's d*=0.8; Table 3. 5).

Pinch-points

Most pinch-points changed locations or disappeared between 2000 and 2015 for all species. Pairwise current flow between black bear SHPs in 2015 identified a very narrow bottleneck of high current flow density between the patch shared by the counties of Lanark and Ottawa/Carleton in Ontario and the patch located in the county of Leeds/Grenville in Ontario (Fig. 3.4). This pinch-point became even more pronounced when current was run in the "all to one" mode in Circuitscape (i.e., a measure of current flow centrality) highlighting its importance in maintaining connectivity across the entire network of suitable habitat patches in the study area (Figure S5). Pairwise current flow for the fisher in 2015 identified a long pinch-point in the LCC traversing the county of Ottawa/Carleton (Fig. 3.4). However, when current was run "all to one" this pinch-point disappeared, indicating its lower relevance in maintaining connectivity across the entire network (Figure S3.5). Instead, pinch-points in Leeds/Grenville in Ontario, and St. Lawrence and Saratoga counties in New York are more important in maintaining overall network connectivity for the fisher. Pairwise current flow in 2015 for moose identified two main pinch-points in the LCCs traversing Lanark County in Ontario, MRC Les Collines-de-l'Outaouais in Québec, and Warren and Washington counties in New York with less pronounced pinch-points throughout many of the remaining LCCs (Fig. 3.5). These less pronounced pinch-points disappeared when current was run "all to one", indicating that only the pinch-points in Lanark County in Ontario, MRC Les Collines-de-l'Outaouais in Québec, and Warren and Washington counties in New York are important for overall network connectivity (Figure S3.6).

Species/Year	Number of LCPs	Mean EucD (km)	Mean CWD (weighted km)	Mean LCP (km)	Effective Resistance (Ohms)
Black Bear					
2000	14	42.5	1420.9	58.9	3525.0
2015	12	44.4	1683.6	60.8	3580.1
2015-2000	**-2**	**2**	**263**	**2**	**55**
t-value		-0.1	-0.5	-0.1	-0.96
df		21	20	21	21
p-value		0.9	0.6	0.9	0.5
95% CI		-29 - 25	-1411 - 885	-43 - 39	-2138 - 2028
Cohen's d		0.06	0.2	0.04	0.02
Fisher					
2000	31	16.0	353.8	25.3	1620.3
2015	27	26.7	702.5	41.1	3022.4
2015-2000	**-4**	**11**	**349**	**16**	**1402**
t-value		-2.3	-2.4	-2.0	-2.0
df		39	34	39	36
p-value		0.03	0.02	0.05	0.05
95% CI		-20 - -1	-645 - -52	-32 - 0.19	-2790 - -14
Cohen's d		0.6	0.7	0.5	0.6
Moose					
2000	16	38.0	1271.8	52.6	4378.0
2015	14	78.9	3226.3	110.5	8749.4
2015-2000	**-2**	**41**	**1954**	**58**	**4371**
t-value		-2.4	-2.5	-2.5	-2
df		16	16	16	17
p-value		0.03	0.03	0.03	0.05
95% CI		-77 - -5	-3634 - -275	-107 - -8	-8776 - 33
Cohen's d		0.9	0.9	0.9	0.8
White-Tailed Deer					
2000	69	11.4	352.3	15.5	1431.3
2015	111	14.7	460.9	20.9	1950.5
2015-2000	**42**	**3**	**109**	**5**	**519**
t-value		-1.9	-1.7	-2.1	-2.3
df		177	178	178	177
p-value		0.06	0.08	0.04	0.02
95% CI		-6.7 - 0.14	-232 - 15	-11 - -0.3	-961 - -77
Cohen's d		0.3	0.3	0.3	0.3

Table 3.5 Changes in number of least-cost paths (LCPs), mean Euclidean distance (EucD, km), mean cost-weighted distance (CWD, weighted km), mean least-cost path length (LCP, km), and effective resistance (Ohms), for each species between 2000 and 2015.

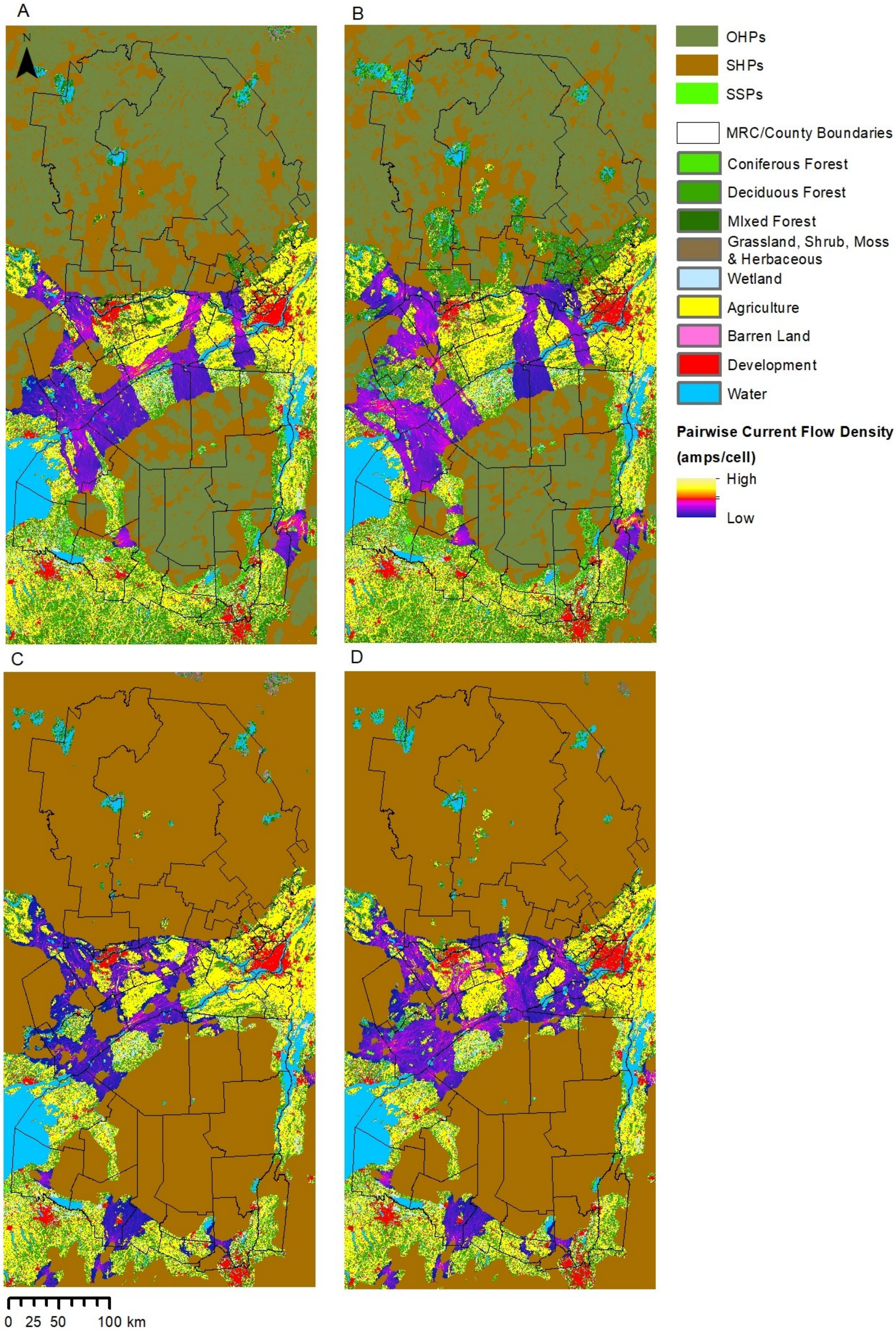

Figure 3.4 Changes in pairwise current flow density within LCCs between 2000 and 2015. A) Black bear habitat 2000, B) Black bear habitat 2015, C) Fisher habitat 2000, D) Fisher habitat 2015.

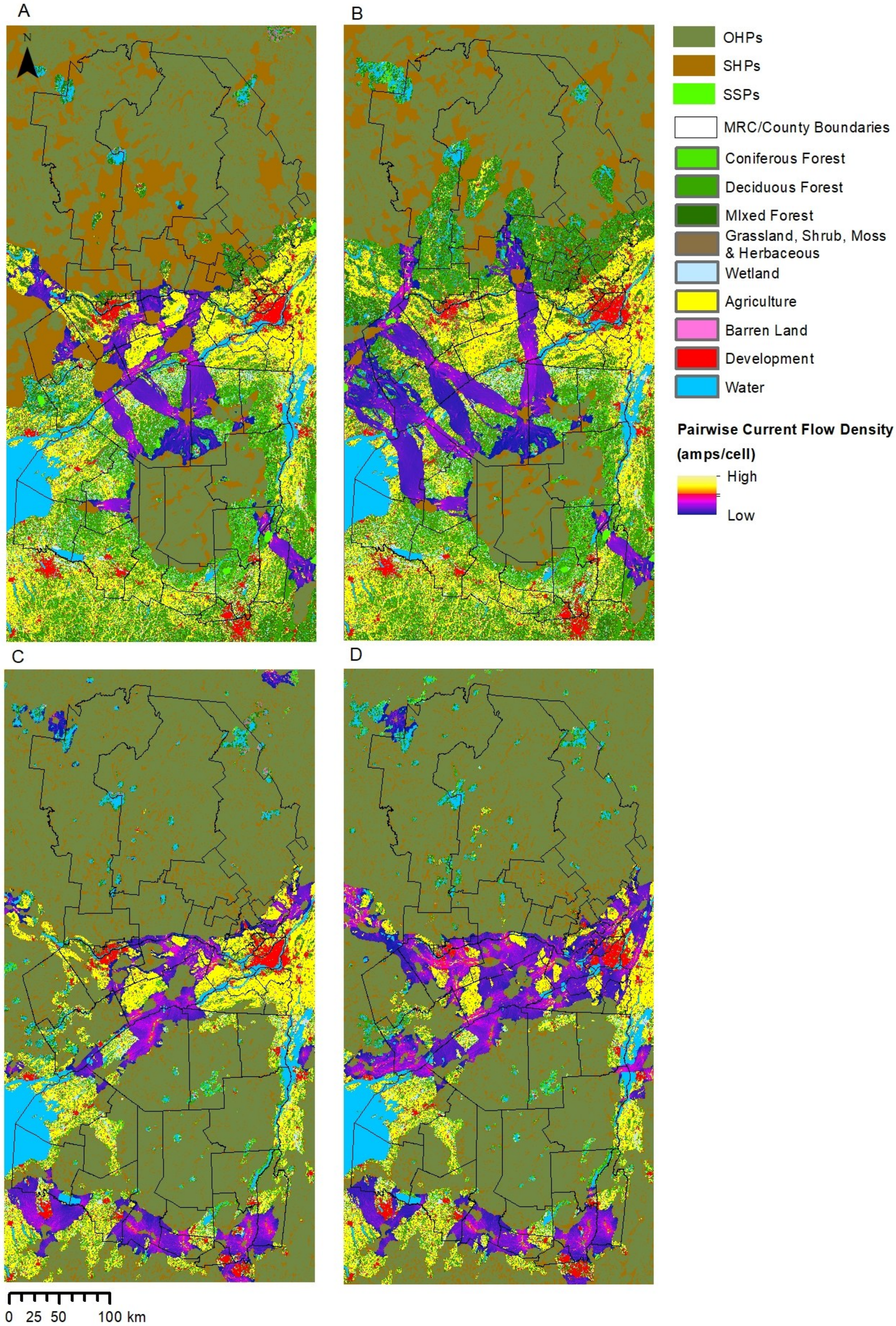

Figure 3.5 Changes in pairwise current flow density within LCCs between 2000 and 2015. A) Moose habitat 2000, B) Moose habitat 2015, C) White-tailed deer habitat 2000, D) White-tailed deer habitat 2015.

Pairwise current flow between white-tailed deer SHPs in 2015 revealed a high concentration of pinch-points in the county of Ottawa/Carleton in Ontario (Fig. 3.5). This concentration disappeared when current was run "all to one" and was replaced by a concentration of pinch-points between the SHPs traversing the counties of Leeds/ Grenville in Ontario and St. Lawrence in New York (Figure S3.6).

Stepping-stone patches

In 2000, there was one stepping-stone patch (SSP) for black bear within the LCC just west of Leeds/Grenville County in Ontario. However, this patch disappeared in 2015 (Fig. 3.2). There were no SSPs for fisher in 2000, however, in 2015 there were four; two in the LCC west of MCR Les Collines-de-l'Outaouais in Québec, one in the LCC west of Leeds/Grenville County in Ontario, and one in a LCC inside Leeds/ Grenville County, Ontario (Fig. 3.2). There were four SSPs for moose in 2000, one within Stormont/Dundas/Glengarry County in Ontario, one within St. Lawrence County in New York, one within Washington County in New York, and one other just east of Washington County in New York. In 2015, number of SSPs for moose increased to 6: one in MRC Papineau, Québec, one shared by the MRCs Les Laurentides and Papineau, Québec, one in Lanark County, Ontario, one west of Leeds/Grenville County, Ontario, one within Washington County, New York, and one just east of Washington County, New York (Fig. 3.3). There were five SSPs for white-tailed deer in 2000: one in Prescott/Russel County, Ontario, three in St. Lawrence County, New York, and one in Herkimer County, New York. In 2015, there was one SSP for white-tailed deer in Ottawa/Carleton County, Ontario, one east of Montréal in Québec, one in Prescott/Russel County, Ontario, one in St. Lawrence County, New York, and one in Herkimer County, New York (Fig. 3.3).

Habitat patches and corridors under protection

In 2015, the proportion of SHP area under protection ranged from 21% (white-tailed deer) to 29% (moose); the proportion of OHP area under protection ranged from 23% (white-tailed deer) to 33% (moose); and the proportion of LCC area under protection ranged from 3% (fisher) to 14% (moose) (Table 3.6; Fig. 3.6). Protection was not equally distributed across the study area. The average SHP area under protection was 9.5% in the Québec portion, 0.2% in the Ontario portion, and 54% in the New York portion; the average OHP area under protection was 10% in the Québec portion, 0.1% in the Ontario portion, and 67% in the New York portion; and the average LCC area under protection was 12% in the Québec portion, 2% in the Ontario portion, and 10% in the New York portion (Table 3.6; Fig. 3.6).

When SHP, OHP, and LCC layers were overlaid to create an intersect map to identify which habitat patches and corridors could be used by all species in 2015, we identified that three of the north–south LCCs could potentially be utilized by all four species (Fig. 3.7).

Discussion

Evaluation of the suitable habitat and habitat patch models

Our results showed that AVI_{patch} values were higher for suitable habitat patches (SHPs) than optimal-habitat patches (OHPs); whereas CVI_{patch} values were similar for SHPs and OHPs (Table 3.1). This can be explained by the fact that the AVI_{patch} value for SHPs is the percentage of evaluation points that fall within SHPs divided by the number of evaluation points within the MCP (Hirzel and Arlettaz 2003); whereas the AVI_{patch} value for OHPs is the percentage of evaluation points that fall within OHPs divided by the number of evaluation points within the MCP (Hirzel and Arlettaz 2003). SHPs have an average suitable habitat value of > 0.5 and have been expanded to include adjacent patches, whereas OHPs have only suitable habitat values > 0.5 and have not been expanded; therefore, SHPs are much bigger than OHPs and usually contain multiple OHPs within them.

The fact that AVI_{patch} values are less for OHPs compared to SHPs demonstrates that the majority of evaluation points fall within OHPs; however, when we evaluate the larger and expanded SHPs, we get slightly more evaluation points (i.e., OHP AVI_{patch} value=0.7, SHP AVI_{patch} value=0.9 for black bear and white-tailed deer in 2000; Table 3.1).

The CVI_{patch} formula, which is the AVI_{patch} value minus the area of habitat patches within the MCP divided by the area of the MCP, takes into consideration the difference in SHP and OHP proportions within the MCP (Hirzel et al. 2004, 2006). Consequently, SHP and OHP CVI_{patch} values are very similar, or the same, for each species and time-point, which effectively validates how well the models fit the evaluation points (Table 1). Only the fisher and the white-tailed deer in 2000 had discrepancies between their SHP and OHP CVI_{patch} values. For the fisher we rejected this model because the discrepancy was very large, whereas the white-tailed deer was much less and there was no discrepancy in 2015.

Location/Species	SHP area (km^2)	SHP area protected (km^2)	Proportion of SHP area Protected (%)	OHP area (km^2)	OHP area protected (km^2)	Proportion of OHP area Protected (%)	LCC area (km^2)	LCC area protected (km^2)	Proportion of LCC area Protected (%)
Study Area									
Black bear	68796	17817	26	41975	13620	32	15630	1488	10
Fisher	84555	18820	22	-	-	-	14598	438	3
Moose	47424	13786	29	33884	11190	33	16249	2346	14
White-tailed deer	86148	18176	21	73979	17048	23	28666	1246	4
Québec Portion									
Black bear	41724	3972	10	26637	2801	11	3055	559	18
Fisher	48858	4410	9	-	-	-	2462	203	8
Moose	33902	3368	10	23648	2539	11	1719	246	14
White-tailed deer	48488	4285	9	41461	3819	9	9012	589	7
Ontario Portion									
Black bear	1256	3	0.2	144	0	0	5713	140	2
Fisher	2447	4.6	0.2	-	-	-	7777	158	2
Moose	174	0	0	39	0	0	5433	94	2
White-tailed deer	4964	17	0.3	3991	13	0.3	9778	305	3
New York Portion									
Black bear	25816	13842	54	15194	10819	71	6861	790	12
Fisher	33250	14405	43	-	-	-	4360	77	2
Moose	13348	10418	78	10197	8651	85	9097	2007	22
White-tailed deer	32696	13874	42	28527	13216	46	9876	352	4

Table 3.6 Amount of suitable habitat patch (SHP) area, optimum habitat patch (OHP) area, and least-cost corridor (LCC) area protected in 2015, at the scale of the study area and each provincial/state portion. Area = total area of patches; Protected = total area of patches protected; Number of PAs = number of Protected Areas; Mean PA Size = Mean size of Protected Area; and Proportion = Proportion of total area of patches protected.

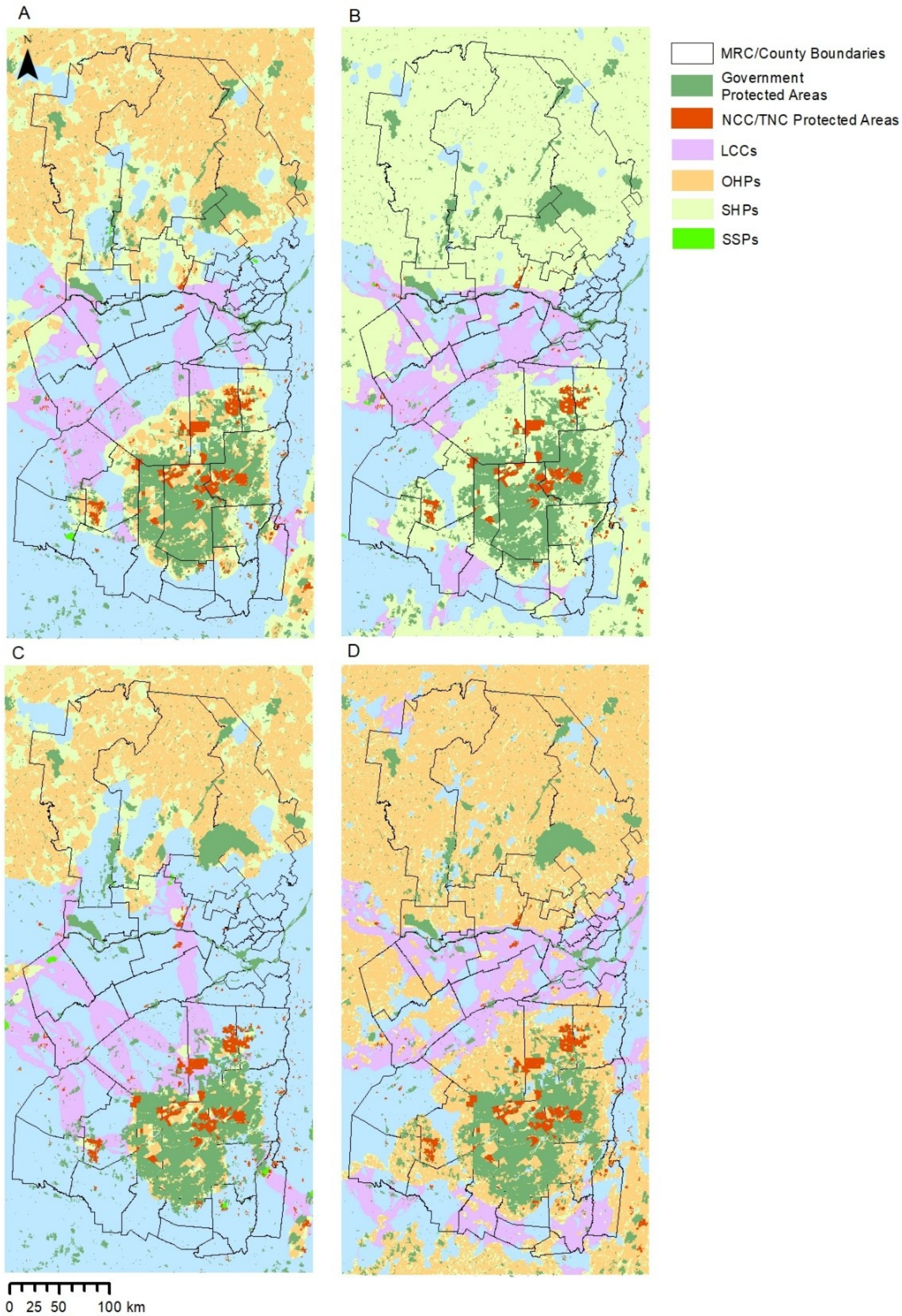

Figure 3.6 Suitable habitat patches (SHP), optimal habitat patches (OHP), stepping-stone patches (SSP), and least-cost corridors (LCC) in 2015, with protected areas superimposed. A) Black bear habitat, B) Fisher habitat, C) Moose habitat, D) White-tailed deer habitat.

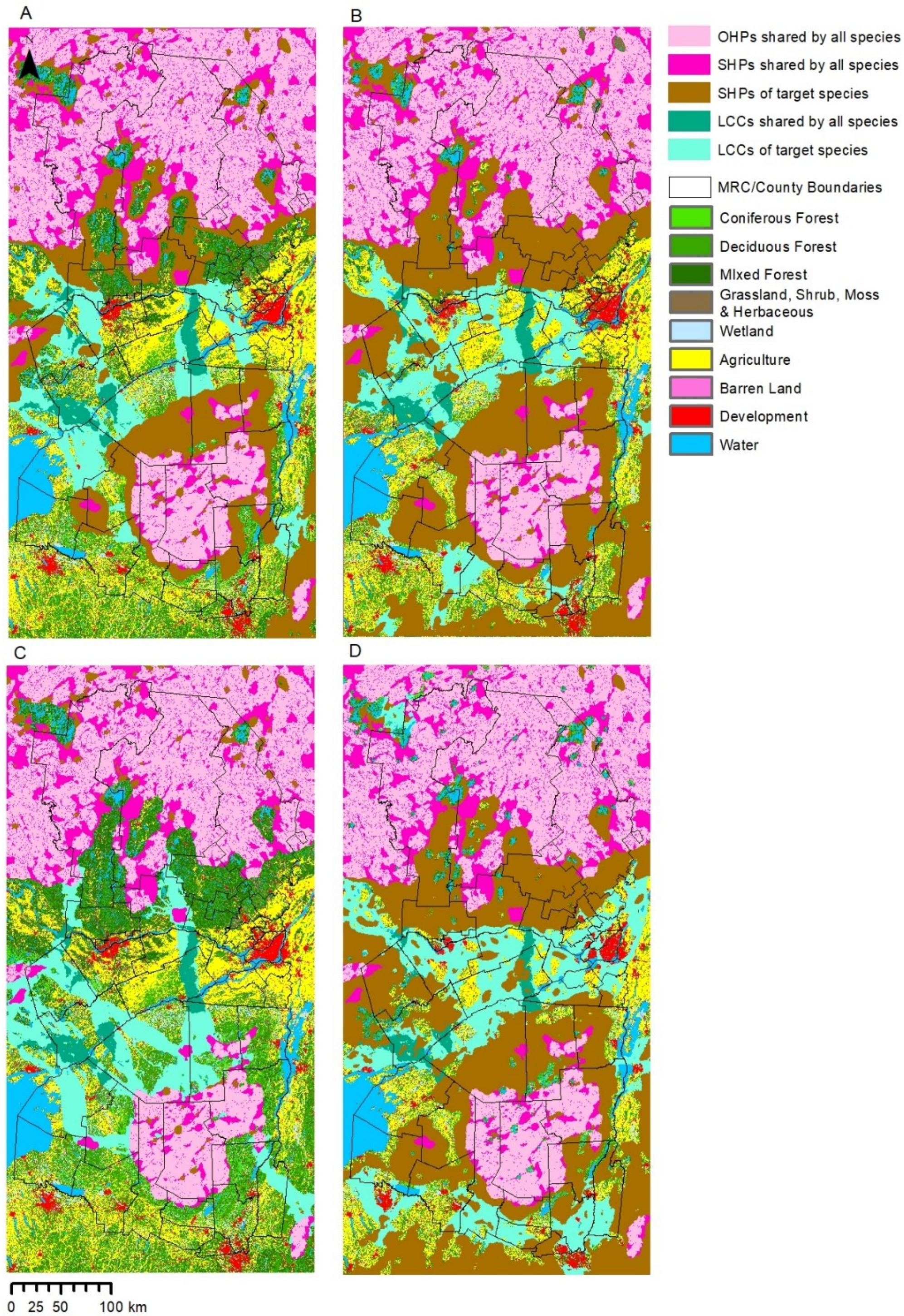

Figure 3.7 Suitable habitat patches (SHPs) and least-cost corridors (LCCs) in 2015, with SHPs, OHPs and LCCs shared by all species in 2015 superimposed. A) Black bear habitat, B) Fisher habitat, C) Moose habitat, D) White-tailed deer habitat.

Land-cover change

As of 2015, there were 17 MRCs/counties that had > 50% of their surface area dedicated to agriculture and development (13 in the Québec portion, 3 in the Ontario portion, and 1 in the New York portion; Table S3.9). These MRCs/counties could be considered "working lands" (Kremen and Merenlender 2018) or "C1- cities and farms regions" according to the "3Cs" framework (i.e., three global conditions for biodiversity conservation and sustainable use; Locke et al. 2019). These MRCs/counties are distinct from the wilderness areas of the Québec and Adirondack mega-patches in that they have been, and continue to be, highly modified by humans (i.e., development, agriculture, roads, etc.) and thus require specialized management and conservation strategies. Most working lands still contain natural land-cover areas (i.e., hedgerows, wooded areas, wetlands, natural pastures, etc.) for which Garibaldi et al. (2021) use the term "native habitats within working landscapes". If these working lands are appropriately managed with area-based conservation and restoration approaches they could provide essential habitat for patches and corridors between the Québec and Adirondack mega-patches. Such an approach would require government and non-government agencies and organizations providing monetary incentives to farmers and ranchers to create, improve upon, and/or maintain natural land-cover habitats on their agricultural lands (i.e., nature-based solutions; ALUS Canada 2023). Nature-based solutions such as wetland restoration, riparian buffers, shelterbelts, afforestation, and grassland restoration would provide habitat, cleaner air and water, carbon sequestration, climate resiliency and many other ecosystem services to the region (ALUS Canada 2023).

Lancaster et al. (2008) found that forest area increased from 29 to 40% within Stormont/Dundas/Glengarry and Leeds/Grenville counties in Ontario between 1934 and 1995, which they proposed significantly contributed to the recovery of fisher populations in the region.

We found that between 2000 and 2015 this trend continued in the counties of Leeds/Grenville and Ottawa/Carleton with net gains of 94 km^2 and 82 km^2 in forest areas, respectively. Conversely, the adjacent counties of Stormont/Dundas/Glengarry and Prescott/Russell had net losses of 243 km^2 and 11 km^2 in forest area, respectively. However, natural land-cover area (which includes all forest types) decreased, and agriculture and development increased, in all of these Ontario counties (Table S3.9). This may be the reason for the underperformance of our fisher OHP models. With such significant losses in natural habitat areas and equivalent increases in agriculture and development, fishers in these locations may presently be (1) settling for sub-optimal habitat to survive, (2) more habitat generalist than previously understood, or (3) a sink population. This trend was not the case for the 12 working lands in the Québec portion, which all experienced increases in natural land-cover area and decreases in agricultural area between 2000 and 2015 (Table S3.9).

Species-specific habitat amount

Our results demonstrate that anthropogenic land transformation between 2000 and 2015 yielded drastic changes in habitat amount for the four focal species. The greatest suitable and optimal habitat patch area losses occurred for moose, with a reduction of 16,842 km^2 (26%) SHP area, and 6832 km^2 (17%) OHP area, followed by black bear with a reduction of 8894 km^2 (11%) SHP area, and 4369 km^2 (9%) OHP area (Table 3.3, Figs. 3.2 and 3.3). However, these significant losses do not translate into actual land-cover loss. We found that natural land-cover area decreased by 1457 km^2, agriculture area decreased by 155 km^2, and development increased by 1410 km^2 within the A2L between 2000 and 2015 (Table 3.2). What these results reveal is that the majority of moose and black bear habitat patch decline was the result of indirect habitat loss due to limiting habitat constraints.

Moose and black bear require considerably large territories, ranging from 25 km^2 to 63 km^2 (Moose; Table S3.6), and 18 km^2 to 290 km^2 (Black bear; Table S3.6). In this study, we used an area of 75 km^2 (Moose) and 68 km^2 (Black bear) as the minimum habitat patch cut-off size for each species, where patches smaller than these cut-off sizes were not considered suitable moose and black bear habitat patches, respectively. As a result, small amounts of land transformation, between 2000 and 2015, caused habitat patches with areas close to the minimum cut-off size, to fall below, and be eliminated as suitable or optimal moose and black bear habitat. We recognized the ecological importance of these smaller-sized patches, however, by identifying stepping-stone patches ≥ 10 km^2 (Table S3.7).

Moose and black bear also exhibit significant avoidance behavior of up to 1 km from human activity, including human presence, urban and industrial development, agriculture, and roads (Jalkotzy et al. 1997; Laurian et al. 2008). This disturbance distance, also referred to as the "zone of influence" and the "road effect zone", cause avoidance of, or displacement from, preferred habitats due to disturbances such as noise, light, pollutants, habitat degradation and other anthropogenic alterations (Forman and Alexander 1998; Benítez-López et al. 2010; Polfus et al., 2011). Between 2000 and 2018, the length of the road network within the Québec portion of the study area increased by 7684 km (16%), with primary road length increasing by 29%; and in the Ontario portion, the length of the road network increased by 2380 km (12%), with primary road length increasing by 13% (Cole et al. 2023). With a road effect zone of up to 1 km, each new kilometer of road added to the landscape has the potential to create a 2 km^2 area of degraded moose and/or black bear habitat.

These direct and indirect habitat losses can be seen in the southern portion of the Québec mega-patch and the entire Ontario portion in 2015 as compared to 2000 (Fig. 3.2; Table 3.3) and have the potential to severely influence long-term transboundary connectivity within the A2L. Consequently, moose populations are declining in the southern portions of Québec and Ontario (Environmental Commissioner of Ontario 2016; Québec 2022), and the situation is being exacerbated by climate change. As temperatures rise, white-tailed deer populations are expanding poleward and sharing more landscapes with moose (Kennedy-Slaney et al. 2018). White-tailed deer are a host to many parasite species including winter ticks (*Dermacentor albipictus*), liver fluke (*Fascioloides magna*), and meningeal worm (*Parelaphostrongylus tenuis*), that can be transmitted to moose with detrimental and sometimes lethal effects (Murray et al. 2006).

Species-specific fragmentation

Anthropogenic land transformation between 2000 and 2015 also caused substantial habitat fragmentation for the four focal species within the A2L. The greatest suitable and optimal habitat patch fragmentation occurred for moose, with a SHP $m_{\text{eff\_CUT}}$ size reduction of 46%, and a SHP $m_{\text{eff\_CBC}}$ size reduction of 42%, and an OHP $m_{\text{eff\_CUT}}$ size, and a OHP $m_{\text{eff\_CBC}}$ size reduction of 71% (Table 3.4). This was followed by black bear with a SHP $m_{\text{eff\_CUT}}$ size reduction of 21%, and a SHP $m_{\text{eff\_CBC}}$ size reduction of 20%, and an OHP $m_{\text{eff\_CUT}}$ size, and an OHP $m_{\text{eff\_CBC}}$ size reduction of 70% (Table 3.4). Mammals with large area requirements are especially vulnerable to the effects of fragmentation (Woodrofe and Ginsberg 1998). As new roads, infrastructure, agriculture, and development are added to the landscape, suitable and optimal habitat patches are increasingly fragmented, producing smaller and more isolated patches; with some being reduced below the species-specific minimum habitat patch size, and others being lost altogether. Future road development plans should be evaluated for their exacerbating effects on habitat amount and fragmentation in this already degraded and sensitive landscape. One solution would be to decrease the overall density of the road network. Road density can be reduced through several measures, including (1) closing low-traffic roads, (2) upgrading and widening existing roads over construction of new ones, and (3) bundling roads and other transportation infrastructure close together (i.e., constructing roads and railways in parallel) (Jaeger et al. 2006, 2011). To reduce the barrier effect of these strategies, wildlife crossing structures and fencing can be placed strategically along the widened/bundled infrastructures allowing animals access to both sides and maintaining connectivity (Rytwinski et al. 2016).

Many suitable and optimal habitat patches cross political borders, and their land area is thus shared by multiple MRCs/counties (Figs. 3.2 and 3.3). However, the number of patches and the amount of patch area shared between MRCs/counties considerably decreased between 2000 and 2015 (Tables S3.10-S3.16). Because of the importance of these transboundary patches for species-specific habitat amount and connectivity, and their disproportionate risk of being reduced or fragmented, we recommend collaborative conservation strategies between MRCs/counties to ensure that these patches continue to serve as vital habitats for a wide range of species within the A2L.

Species-specific connectivity

Within the study area, inter-patch connectivity decreased for each species, as measured by increases in mean Euclidean distance, mean least-cost path, mean cost-weighted distance, and effective resistance (Table 3.5). Euclidean distance increased because habitat patches were either reduced in size, fragmented into multiple smaller patches, or completely lost due to land conversion which resulted in greater distances between patches in 2015 (Table 3.5, Figs. 3.2 and 3.3). Whereas increases in least-cost path, cost-weighted distance, and effective resistance were due to a combination of land conversion reducing suitable habitat values (i.e., increasing resistance values) in the matrix and the increased distance between patches (Table 3.5, Figs. 3.2 and 3.3). Accordingly, each species must now travel farther between suitable habitat patches and the cost of travelling these distances is higher. This could potentially translate into a reduction in the probability of successful dispersals. Inter-patch and intra-patch connectivity are essential for ecosystem functioning, and a landscape-wide decrease in functional connectivity could have severe negative consequences on key ecosystem processes such as seed dispersal, food web interactions, metapopulation dynamics, and disease transmission (Gonzalez et al. 1998; Bauer and Hoye 2014; Tucker et al. 2018; Plowright et al. 2021).

Dispersal movements between the three provincial/state portions requires the crossing of at least one of two large rivers, the Ottawa River that separates the Québec portion from the Ontario portion, and the St. Lawrence River that separates the Ontario portion from the New York portion (Fig. 3.1). Although both rivers have swift-moving currents, sections of the Ottawa River between Montréal and Ottawa, and sections of the St. Lawrence River between Montréal and Lake Ontario freeze-over in the winter months permitting crossing; with some locations less than 1 km wide (Koen et al. 2015; ECCC 2023). Over the past 20 years, there have been many sightings/reports of animal movement across the rivers. Alice the moose, the Algonquin-to-Adirondack Collaborative’s animal inspiration was a female moose collared and released into the Adirondack Park, New York in 1998. Alice left the Adirondack Park in 2000, and after crossing both the St. Lawrence River and highway 401 in Ontario, ended up in Algonquin Provincial Park in Ontario (A2A 2023). Genetic analysis confirms that fishers have been crossing the St. Lawrence River from the Adirondack region to recolonize eastern Ontario since the 1950s (Carr et al. 2007); and black bear have been reported swimming across the rivers, whereas white-tailed deer have been spotted crossing the ice during winter (Ottawa Citizen 2020, 2022). Other large mammals such as lynx (Koen et al. 2015) and eastern wolves (McAlpine et al. 2015) have also been reported crossing the rivers. Nevertheless, while both rivers are almost certainly a major deterrent to long-distance dispersal, they are not complete barriers to animal movement. We recommend further detailed study to identify priority locations where these focal species are crossing the Ottawa and St. Lawrence rivers within the LCCs, and where applicable, the expansion and protection of these sites.

Pinch-points represent narrow sections within LCCs where movement is restricted due to natural or anthropogenic landscape features and alternative pathways are limited (McRae and Shah 2011; Pelletier et al. 2014). Pinch points can be critical for both facilitating movement between habitat patches as well as contributing to the long-term maintenance of functional connectivity throughout the linkage (McRae et al. 2008). We identified multiple pinch-points within LCCs where movement could become increasingly limited for each species (Figs. 3.4, 3.5, S3.5 and S3.6). These pinch-point locations should be prioritized for both conservation and restoration interventions: (1) to ensure that additional habitat loss does not further restrict the pinch-point, and (2) to decrease the constrictive severity of the pinch-point and increase its connectivity potential. In addition, pinch-points that intersect primary and secondary roads should be further evaluated for their potential as locations for wildlife crossing structures and fencing (Nussey and Noseworthy 2018; Spanowicz et al. 2020).

SSPs are small habitat patches that offer refuge to individuals as they travel through the matrix between SHPs (Baum et al. 2004). These small patches can disproportionately contribute to species-specific connectivity when distances between SHPs are greater than the maximum dispersal distance of a species (Dutta et al. 2016). We located several SSPs within LCCs that could offer shelter and resources to individuals as they travel through the LCCs (Figs. 3.2 and 3.3). These patches should be considered a high priority for conservation and ecological restoration as they have the potential to facilitate movement between habitat patches and contribute to maintaining connectivity throughout the A2L.

Habitat patches and corridors under protection

In 2015, only 2.2% of the Ontario portion was protected, whereas 8.7% of the Québec portion, and 28% of the New York portion was protected. This sizable protection of 14,914 km^2 in the New York portion includes 54% SHP, 71% OHP, and 12% LCC area protected (black bear); 43% SHP and 2% LCC area protected (fisher); 78% SHP, 85% OHP, and 22% LCC area protected (moose); and 42% SHP, 46% OHP, and 4% LCC area protected (white-tailed deer), and is the primary reason for the observed stability of habitat amount and fragmentation in this region between 2000 and 2015 (Table 3.6, Fig. 3.6). Consequently, to ensure functional connectivity between the Québec and Adirondack mega-patches, increasing protection while synchronously increasing habitat restoration in the southern Québec and Ontario portions (Currie et al. 2023) will be crucial to reduce or eliminate further habitat loss and fragmentation.

Protected areas have been an important tool for the conservation of biodiversity in North America. However, many protected areas are simply not large enough to support viable populations of species with large home ranges nor do they include the range of species, processes, and habitats necessary to fully conserve ecosystem integrity and biodiversity (Boyd et al. 2008; Pimm et al. 2014). For example, in 2015, 20,389 km^2 (16%) of the study area was under protection. This was made up of 1314 sites protected by the Canadian/U.S. Government, and 1278 sites protected by Nature Conservancy of Canada/The Nature Conservancy (Fig. 3.6). However, the average Canadian/U.S. Government protected area size was 13 km^2, and the average Nature Conservancy of Canada/The Nature Conservancy protected area size was 2 km^2, which are considerably below the average home range size of black bear (101 km^2), moose (42 km^2), fisher (19 km^2), and many other large-ranging species. The average proportion of SHP area (25%) was lower than the average OHP area protected (29%). This was due to the fact that SHP area contains non-habitat land-cover classes such as roads, development, and agriculture lands which would not be included in protected areas, whereas OHPs only contain natural habitat types such as forests, grasslands and wetlands.

The average proportions of SHP and OHP area protected, however, were both higher than the average proportion of LCC area protected (8%) (Table 3.6). This result highlights that to achieve long-term transboundary connectivity for large ranging species, active measures should be taken to not only create new and expand existing protected areas within the A2L, but also restore, maintain, enhance, and protect connectivity corridors between them (Hilty et al. 2020).

While protected area conservation has been around since the founding of Yellowstone National Park in 1872, conservation of connectivity corridors is a relatively new idea (National Park Service 2022). In April 2022, Parks Canada launched the “National Program for Ecological Corridors”, the first of its kind in Canada. The program will invest $60.6 million over five years to help support other jurisdictions and organizations develop better ecological connections between protected areas (Government of Canada 2022). In the USA, nearly 50 corridor conservation policies have been released from different levels of government since 2007 (Breuer et al. 2022; Conservation Corridor 2022). One of the most significant is the Wildlife Corridors Conservation Act of 2019, which establishes a “National Wildlife Corridors System” to designate corridors on federal public lands as well as provide funding for states, tribes, and other entities to protect wildlife corridors on non-federal lands (116th Congress, 2019–2020). Protection of connectivity corridors will require cooperation at the MRC/county, provincial/state and transboundary levels to develop an ecological network-based conservation and restoration strategy. Such an approach should comprise of a system of protected areas inter-connected by a network of protected ecological corridors that would enhance ecosystem integrity, biodiversity, and connectivity (Hilty et al. 2020). Such harmonized efforts will improve the protection of shared conservation features (i.e., species, ecosystems, and natural resources), as well as reduce the financial costs for each cooperating member (Kark et al. 2015); and managing transboundary conservation for these umbrella species will simultaneously conserve and restore connectivity for a variety of other species that utilize the A2L linkage.

Conclusion

Our results highlight the degree to which anthropogenic land transformation has impacted species-specific habitat amount, fragmentation, and connectivity in the A2L transboundary wildlife linkage between 2000 and 2015. Suitable and optimal habitat patch area decreased for each species with moose suitable and optimal habitat patch area declining by 26% and 17%, respectively. This was followed by black bear with SHP area losses of 11%, and OHP area losses of 9% (Table 3.3). Habitat fragmentation increased for each species with moose experiencing an OHP effective mesh size decrease of 71% ($m_{\mathrm{eff\_CUT}}$ and $m_{\mathrm{eff\_CBC}}$); and black bear experiencing an OHP effective mesh size decrease of 70% ($m_{\text{eff\_CUT}}$ and $m_{\text{eff\_CBC}}$). Inter-patch connectivity also decreased significantly for fisher, moose, and white-tailed deer (Table 3.5). Consequently, to achieve long-term functionality of the A2L, collaborative and coordinated measures will be necessary to preserve the integrity of the Québec mega-patch, restore extensive habitat in eastern Ontario, and reestablish or maintain connectivity throughout the linkage. Left unaddressed, continued anthropogenic land transformation is likely to have detrimental effects on the ability of the A2L to function as a transboundary wildlife linkage.

Appendix 1

Values used to identify species-specific habitat patches

Values for black Bear (Ursus americanus)

For the value of the moving window radius, we used a radius of 9602.8 m (Table S7) of the maximum male home range size of 289.7 km^2 (Table S6), assuming that this was the maximum area needed to sustain at least 1 male and 3 or more female black bear. We used the radius 2413.5 m (Table S7) of the average minimum female home range size 18.3 km^2 (Table S6) as the maximum distance to expand habitat patches outward to simulate intra-patch connectivity (WHCWG, 2010; Spanowicz & Jaeger, 2019). We used a value of 68.1 km^2 (0.75 x average minimum male home range size; Table S6; Table S7) as the minimum habitat patch cut-off size necessary to sustain at least 1 male and 3 or more female black bear. Habitat patches smaller than this value were removed from the SHP layer.

Values for Fisher (Pekania pennanti)

For the value of the moving window radius, we used a radius of 5970.8 m (Table S7) of four times the maximum male home range size of 28 km^2 (Table S6), assuming that this was the maximum area needed to sustain at least 1 male and 3 or more female fishers. We used the radius 1595.8 m (Table S7) of the average minimum female home range size 8.0 km^2 (Table S6) as the maximum distance to expand habitat patches outward to simulate intra-patch connectivity (WHCWG, 2010; Spanowicz & Jaeger, 2019). We used a value of 24.0 km^2 (0.75 x average minimum female home range size x 4; Table S6; Table S7) as the minimum habitat patch cut-off size necessary to sustain at least 1 male and 3 or more female fishers. Habitat patches smaller than this value were removed from the SHP layer.

Values for moose (Alces alces)

For the value of the moving window radius, we used a radius of 8920.6 m (Table S7) of four times the average maximum home range size 62.5 km^2 (Table S6), assuming that this was the maximum area needed to sustain at least 1 male and 3 or more female moose. We used the radius 2820.9 m (Table S7) of the average minimum home range size 25.0 km^2 (Table S6) as the maximum distance to expand habitat patches outward to simulate intra-patch connectivity (WHCWG, 2010; Spanowicz & Jaeger, 2019). We used a value of 75.0 km^2 (0.75 x average minimum home range size x 4; Table S6; Table S7) as the minimum habitat patch cut-off size necessary to sustain at least 1 male and 3 or more female moose. Habitat patches smaller than this value were removed from the SHP layer.

Values for white-tailed deer (Odocoileus virginianus)

For the value of the moving window radius, we used a radius of 4126.7 m (Table S7) which was 17 times the average maximum home range size of 3.1 km^2 (Table S6), assuming that this was the maximum area needed to sustain 12 or more family members and 5 or more males over one year of age. We used the radius 535.2 m (Table S7) of the average minimum home range size 0.9 km^2 (Table S6) as the maximum distance to expand habitat patches outward to simulate intra-patch connectivity (WHCWG, 2010; Spanowicz & Jaeger, 2019). We used a value of 2.7 km^2 (0.75 x average minimum home range size x 4; Table S6; Table S7) as the cut-off minimum habitat patch size necessary to sustain 2 or more family members and 2 or more males over one year of age. Habitat patches smaller than this value were removed from the SHP layer.

R code and calculations

```
#### comparing the ED, CWD, LCP, and ER means between 2000 and 2015 for the black bear
var.test(BB_ED~Year, data = BB_R)
qqnorm(lm(BB_ED~Year, data = BB_R)$residuals)
abline(0,1)
t.test(BB_ED~Year, alternative = 'less', var.equal = FALSE, data = BB_R)

var.test(BB_CWD~Year, data = BB_R)
qqnorm(lm(BB_CWD~Year, data = BB_R)$residuals)
abline(0,1)
t.test(BB_CWD~Year, alternative = 'less', var.equal = FALSE, data = BB_R)

var.test(BB_LCP~Year, data = BB_R)
qqnorm(lm(BB_LCP~Year, data = BB_R)$residuals)
abline(0,1)
t.test(BB_LCP~Year, alternative = 'less', var.equal = FALSE, data = BB_R)

var.test(BB_ER~Year, data = BB_R)
qqnorm(lm(BB_ER~Year, data = BB_R)$residuals)
abline(0,1)
t.test(BB_ER~Year, alternative = 'less', var.equal = FALSE, data = BB_R)


var.test(BB_ED~Year, data = BB_R)

        F test to compare two variances

data:  BB_ED by Year
F = 0.64597, num df = 13, denom df = 11, p-value = 0.4493
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.1904555 2.0654967
sample estimates:
ratio of variances
       0.6459731

> qqnorm(lm(BB_ED~Year, data = BB_R)$residuals)
> abline(0,1)
> t.test(BB_ED~Year, alternative = 'less', var.equal = FALSE, data = BB_R)
```

```
        Welch Two Sample t-test

data:  BB_ED by Year
t = -0.14263, df = 21.084, p-value = 0.444
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
95 percent confidence interval:
    -Inf 20.31575
sample estimates:
mean in group 2000 mean in group 2015
        42.51479           44.35133

> var.test(BB_CWD~Year, data = BB_R)

        F test to compare two variances

data:  BB_CWD by Year
F = 0.55778, num df = 13, denom df = 11, p-value = 0.3153
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.1644544 1.7835145
sample estimates:
ratio of variances
        0.5577847

> qqnorm(lm(BB_CWD~Year, data = BB_R)$residuals)
> abline(0,1)
> t.test(BB_CWD~Year, alternative = 'less', var.equal = FALSE, data = BB_R)

        Welch Two Sample t-test

data:  BB_CWD by Year
t = -0.47727, df = 20.138, p-value = 0.3192
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
95 percent confidence interval:
    -Inf 686.4088
sample estimates:
mean in group 2000 mean in group 2015
        1420.908           1683.645

> var.test(BB_LCP~Year, data = BB_R)

        F test to compare two variances

data:  BB_LCP by Year
```

```
F = 0.62076, num df = 13, denom df = 11, p-value = 0.41
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.1830222 1.9848830
sample estimates:
ratio of variances
         0.6207616

> qqnorm(lm(BB_LCP~Year, data = BB_R)$residuals)
> abline(0,1)
> t.test(BB_LCP~Year, alternative = 'less', var.equal = FALSE, data = BB_R)

	Welch Two Sample t-test

data:  BB_LCP by Year
t = -0.097084, df = 20.83, p-value = 0.4618
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
95 percent confidence interval:
     -Inf 31.78128
sample estimates:
mean in group 2000 mean in group 2015
          58.87679           60.77633

> var.test(BB_ER~Year, data = BB_R)

	F test to compare two variances

data:  BB_ER by Year
F = 0.68799, num df = 13, denom df = 11, p-value = 0.5158
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.2028437 2.1998474
sample estimates:
ratio of variances
         0.6879906

> qqnorm(lm(BB_ER~Year, data = BB_R)$residuals)
> abline(0,1)
> t.test(BB_ER~Year, alternative = 'less', var.equal = FALSE, data = BB_R)

	Welch Two Sample t-test

data:  BB_ER by Year
t = -0.054881, df = 21.479, p-value = 0.4784
```

```
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
95 percent confidence interval:
    -Inf 1669.203
sample estimates:
mean in group 2000 mean in group 2015
        3525.004          3580.054

#### comparing the ED, CWD, LCP, and ER means between 2000 and 2015 for the fisher
var.test(FI_ED~Year, data = FI_R)
qqnorm(lm(FI_ED~Year, data = FI_R)$residuals)
abline(0,1)
t.test(FI_ED~Year, alternative = 'less', var.equal = FALSE, data = FI_R)

var.test(FI_CWD~Year, data = FI_R)
qqnorm(lm(FI_CWD~Year, data = FI_R)$residuals)
abline(0,1)
t.test(FI_CWD~Year, alternative = 'less', var.equal = FALSE, data = FI_R)

var.test(FI_LCP~Year, data = FI_R)
qqnorm(lm(FI_LCP~Year, data = FI_R)$residuals)
abline(0,1)
t.test(FI_LCP~Year, alternative = 'less', var.equal = FALSE, data = FI_R)

var.test(FI_ER~Year, data = FI_R)
qqnorm(lm(FI_ER~Year, data = FI_R)$residuals)
abline(0,1)
t.test(FI_ER~Year, alternative = 'less', var.equal = FALSE, data = FI_R)
> var.test(FI_ED~Year, data = FI_R)

          F test to compare two variances

data:  FI_ED by Year
F = 0.3009, num df = 30, denom df = 26, p-value = 0.001905
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.1395319 0.6356286
sample estimates:
ratio of variances
       0.3009042

> qqnorm(lm(FI_ED~Year, data = FI_R)$residuals)
> abline(0,1)
> t.test(FI_ED~Year, alternative = 'less', var.equal = FALSE, data = FI_R)

          Welch Two Sample t-test
```

```
data:  FI_ED by Year
t = -2.249, df = 39.087, p-value = 0.01511
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
95 percent confidence interval:
    -Inf -2.68666
sample estimates:
mean in group 2000 mean in group 2015
         15.99577           26.70441

> var.test(FI_CWD~Year, data = FI_R)

        F test to compare two variances

data:  FI_CWD by Year
F = 0.18334, num df = 30, denom df = 26, p-value = 1.759e-05
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.08501676 0.38728850
sample estimates:
ratio of variances
        0.1833409

> qqnorm(lm(FI_CWD~Year, data = FI_R)$residuals)
> abline(0,1)
> t.test(FI_CWD~Year, alternative = 'less', var.equal = FALSE, data = FI_R)

        Welch Two Sample t-test

data:  FI_CWD by Year
t = -2.3906, df = 34.211, p-value = 0.01123
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
95 percent confidence interval:
     -Inf -102.0908
sample estimates:
mean in group 2000 mean in group 2015
         353.8358           702.5197

> var.test(FI_LCP~Year, data = FI_R)

        F test to compare two variances

data:  FI_LCP by Year
F = 0.30368, num df = 30, denom df = 26, p-value = 0.002056
```

```
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.1408181 0.6414880
sample estimates:
ratio of variances
        0.303678

> qqnorm(lm(FI_LCP~Year, data = FI_R)$residuals)
> abline(0,1)
> t.test(FI_LCP~Year, alternative = 'less', var.equal = FALSE, data = FI_R)

        Welch Two Sample t-test

data:  FI_LCP by Year
t = -1.9981, df = 39.196, p-value = 0.02634
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
95 percent confidence interval:
     -Inf -2.472231
sample estimates:
mean in group 2000 mean in group 2015
        25.32284          41.08515

> var.test(FI_ER~Year, data = FI_R)

        F test to compare two variances

data:  FI_ER by Year
F = 0.21958, num df = 30, denom df = 26, p-value = 0.0001091
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.1018202 0.4638355
sample estimates:
ratio of variances
        0.219578

> qqnorm(lm(FI_ER~Year, data = FI_R)$residuals)
> abline(0,1)
> t.test(FI_ER~Year, alternative = 'less', var.equal = FALSE, data = FI_R)

        Welch Two Sample t-test

data:  FI_ER by Year
t = -2.0496, df = 35.762, p-value = 0.02389
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
```

```
95 percent confidence interval:
     -Inf -246.9754
sample estimates:
mean in group 2000 mean in group 2015
         1620.308          3022.416
```

```
#### comparing the ED, CWD, LCP, and ER means between 2000 and 2015 for the moose
var.test(MO_ED~Year, data = MO_R)
qqnorm(lm(MO_ED~Year, data = MO_R)$residuals)
abline(0,1)
t.test(MO_ED~Year, alternative = 'less', var.equal = FALSE, data = MO_R)

var.test(MO_CWD~Year, data = MO_R)
qqnorm(lm(MO_CWD~Year, data = MO_R)$residuals)
abline(0,1)
t.test(MO_CWD~Year, alternative = 'less', var.equal = FALSE, data = MO_R)

var.test(MO_LCP~Year, data = MO_R)
qqnorm(lm(MO_LCP~Year, data = MO_R)$residuals)
abline(0,1)
t.test(MO_LCP~Year, alternative = 'less', var.equal = FALSE, data = MO_R)

var.test(MO_ER~Year, data = MO_R)
qqnorm(lm(MO_ER~Year, data = MO_R)$residuals)
abline(0,1)
t.test(MO_ER~Year, alternative = 'less', var.equal = FALSE, data = MO_R)
```

```
> var.test(MO_ED~Year, data = MO_R)

	F test to compare two variances

data:  MO_ED by Year
F = 0.14215, num df = 15, denom df = 13, p-value = 0.000617
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.04656468 0.41577111
sample estimates:
ratio of variances
        0.1421486

> qqnorm(lm(MO_ED~Year, data = MO_R)$residuals)
> abline(0,1)
> t.test(MO_ED~Year, alternative = 'less', var.equal = FALSE, data = MO_R)

	Welch Two Sample t-test
```

```
data:  MO_ED by Year
t = -2.4045, df = 16.218, p-value = 0.01424
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
95 percent confidence interval:
     -Inf -11.23095
sample estimates:
mean in group 2000 mean in group 2015
         38.00100          78.91271

> var.test(MO_CWD~Year, data = MO_R)

	F test to compare two variances

data:  MO_CWD by Year
F = 0.1224, num df = 15, denom df = 13, p-value = 0.0002559
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.04009492 0.35800332
sample estimates:
ratio of variances
        0.1223983

> qqnorm(lm(MO_CWD~Year, data = MO_R)$residuals)
> abline(0,1)
> t.test(MO_CWD~Year, alternative = 'less', var.equal = FALSE, data = MO_R)

	Welch Two Sample t-test

data:  MO_CWD by Year
t = -2.47, df = 15.777, p-value = 0.01266
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
95 percent confidence interval:
    -Inf -571.775
sample estimates:
mean in group 2000 mean in group 2015
         1271.786           3226.256

> var.test(MO_LCP~Year, data = MO_R)

	F test to compare two variances

data:  MO_LCP by Year
F = 0.14092, num df = 15, denom df = 13, p-value = 0.0005868
```

```
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.04616148 0.41217096
sample estimates:
ratio of variances
        0.1409178

> qqnorm(lm(MO_LCP~Year, data = MO_R)$residuals)
> abline(0,1)
> t.test(MO_LCP~Year, alternative = 'less', var.equal = FALSE, data = MO_R)

        Welch Two Sample t-test

data:  MO_LCP by Year
t = -2.4737, df = 16.19, p-value = 0.0124
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
95 percent confidence interval:
    -Inf -17.0638
sample estimates:
mean in group 2000 mean in group 2015
         52.59819          110.49421

> var.test(MO_ER~Year, data = MO_R)

        F test to compare two variances

data:  MO_ER by Year
F = 0.17285, num df = 15, denom df = 13, p-value = 0.001855
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.05662031 0.50555678
sample estimates:
ratio of variances
        0.1728456

> qqnorm(lm(MO_ER~Year, data = MO_R)$residuals)
> abline(0,1)
> t.test(MO_ER~Year, alternative = 'less', var.equal = FALSE, data = MO_R)

        Welch Two Sample t-test

data:  MO_ER by Year
t = -2.0949, df = 16.895, p-value = 0.02578
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
```

```
95 percent confidence interval:
     -Inf -740.0713
sample estimates:
mean in group 2000 mean in group 2015
        4378.019          8749.356
```

```
#### comparing the ED, CWD, LCP, and ER means between 2000 and 2015 for the white-tailed deer
var.test(WTD_ED~Year, data = WTD_R)
qqnorm(lm(WTD_ED~Year, data = WTD_R)$residuals)
abline(0,1)
t.test(WTD_ED~Year, alternative = 'less', var.equal = FALSE, data = WTD_R)

var.test(WTD_CWD~Year, data = WTD_R)
qqnorm(lm(WTD_CWD~Year, data = WTD_R)$residuals)
abline(0,1)
t.test(WTD_CWD~Year, alternative = 'less', var.equal = FALSE, data = WTD_R)

var.test(WTD_LCP~Year, data = WTD_R)
qqnorm(lm(WTD_LCP~Year, data = WTD_R)$residuals)
abline(0,1)
t.test(WTD_LCP~Year, alternative = 'less', var.equal = FALSE, data = WTD_R)

var.test(WTD_ER~Year, data = WTD_R)
qqnorm(lm(WTD_ER~Year, data = WTD_R)$residuals)
abline(0,1)
t.test(WTD_ER~Year, alternative = 'less', var.equal = FALSE, data = WTD_R)

> var.test(WTD_ED~Year, data = WTD_R)

          F test to compare two variances

data:  WTD_ED by Year
F = 0.43824, num df = 68, denom df = 110, p-value = 0.000328
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.2880701 0.6815967
sample estimates:
ratio of variances
       0.4382372

> qqnorm(lm(WTD_ED~Year, data = WTD_R)$residuals)
> abline(0,1)
> t.test(WTD_ED~Year, alternative = 'less', var.equal = FALSE, data = WTD_R)

          Welch Two Sample t-test
```

```
data:  WTD_ED by Year
t = -1.8946, df = 177.26, p-value = 0.02989
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
95 percent confidence interval:
      -Inf -0.4177766
sample estimates:
mean in group 2000 mean in group 2015
         11.44139           14.72445

> var.test(WTD_CWD~Year, data = WTD_R)

           F test to compare two variances

data:  WTD_CWD by Year
F = 0.4144, num df = 68, denom df = 110, p-value = 0.000132
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.2724041 0.6445297
sample estimates:
ratio of variances
        0.4144047

> qqnorm(lm(WTD_CWD~Year, data = WTD_R)$residuals)
> abline(0,1)
> t.test(WTD_CWD~Year, alternative = 'less', var.equal = FALSE, data = WTD_R)

           Welch Two Sample t-test

data:  WTD_CWD by Year
t = -1.7356, df = 177.76, p-value = 0.04219
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
95 percent confidence interval:
     -Inf -5.139132
sample estimates:
mean in group 2000 mean in group 2015
         352.2956           460.9298

> var.test(WTD_LCP~Year, data = WTD_R)

           F test to compare two variances

data:  WTD_LCP by Year
F = 0.34604, num df = 68, denom df = 110, p-value = 5.31e-06
```

```
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.2274680 0.5382073
sample estimates:
ratio of variances
         0.3460439

> qqnorm(lm(WTD_LCP~Year, data = WTD_R)$residuals)
> abline(0,1)
> t.test(WTD_LCP~Year, alternative = 'less', var.equal = FALSE, data = WTD_R)

        Welch Two Sample t-test

data:  WTD_LCP by Year
t = -2.1004, df = 177.55, p-value = 0.01855
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
95 percent confidence interval:
      -Inf -1.153584
sample estimates:
mean in group 2000 mean in group 2015
          15.47649           20.89772

> var.test(WTD_ER~Year, data = WTD_R)

        F test to compare two variances

data:  WTD_ER by Year
F = 0.32977, num df = 68, denom df = 110, p-value = 2.101e-06
alternative hypothesis: true ratio of variances is not equal to 1
95 percent confidence interval:
 0.2167681 0.5128907
sample estimates:
ratio of variances
         0.3297665

> qqnorm(lm(WTD_ER~Year, data = WTD_R)$residuals)
> abline(0,1)
> t.test(WTD_ER~Year, alternative = 'less', var.equal = FALSE, data = WTD_R)

        Welch Two Sample t-test

data:  WTD_ER by Year
t = -2.3194, df = 177.06, p-value = 0.01076
alternative hypothesis: true difference in means between group 2000 and group 2015 is less than
0
```

```
95 percent confidence interval:
     -Inf -149.0408
sample estimates:
mean in group 2000 mean in group 2015
         1431.330          1950.464
```

Literature Cited

Spanowicz, A. G., & Jaeger, J. A. (2019). Measuring landscape connectivity: on the importance of within-patch connectivity. *Landscape Ecology*, 34(10), 2261-2278.

Washington Wildlife Habitat Connectivity Working Group (WHCWG). (2010). Washington Connected Landscapes Project: Statewide Analysis. Washington Departments of Fish and Wildlife, and Transportation, Olympia, WA.

Supplemental Tables and Figures

Maps	2000	Resolution	Source	Link to Dataset
Land-cover				
Quebec Ontario	Canadian Land Cover, Circa 2000 (Vector) - GeoBase Series, 1996-2005	Vector	Natural Resources Canada	https://open.canada.ca/data/en/dataset/97126362-5a85-4fe0-9dc2-915464cfdbb7
NY/VT/MA	NLCD 2001 Land Cover (CONUS)	30m	U.S. Geological Survey (USGS)	https://www.mrlc.gov/data?f%5B0%5D=category%3Aland%20cover&f%5B1%5D=region%3Aconus
Roads				
Quebec Ontario	DMTI 2000	Polyline	DMTI	Received from Concordia University Library - Concordia Licence
NY/VT/MA	TigerLine 2000	Polyline	United States Census Bureau	https://www.census.gov/geographies/mapping-files/time-series/geo/tiger-line-file.html
MRC/County Boundaries				
Quebec	Civil Boundaries	Vector	Quebeec Government	Recived from the Nature Conservancy of Québec
Ontario	Civil Boundaries	Vector	Ontario Government	https://data.ontario.ca/
NY/VT/MA	TigerLine Shapefiles	Vector	United States Census Bureau	https://www.census.gov/geographies/mapping-files/time-series/geo/tiger-line-file.html
Maps	**2015**	**Resolution**	**Source**	**Link to Dataset**
Land-cover				
Quebec Ontario NY/VT/MA	North America Land Cover (Landsat and RapidEye)	30m	Commission for Environmental Cooperation (CEC).	http://www.cec.org/north-american-environmental-atlas/land-cover-30m-2015-landsat-and-rapideye/
Roads				
Quebec Ontario	DMTI 2018	Polyline	DMTI	Received from Concordia University Library - Concordia Licence
NY/VT/MA	TigerLine 2015	Polyline	United Staes Census Bureau	https://catalog.data.gov/dataset/2015-roads-national-geodatabase
MRC/County Boundaries				
Quebec	Civil Boundaries	Vector	Quebeec Government	Recived from the Nature Conservancy of Québec
Ontario	Civil Boundaries	Vector	Ontario Government	https://data.ontario.ca/
NY/VT/MA	TigerLine Shapefiles	Vector	United States Census Bureau	https://www.census.gov/geographies/mapping-files/time-series/geo/tiger-line-file.html
Proteted Areas				
Quebec	Protected areas (Canada)	Vector	Protected Planet	https://www.protectedplanet.net/country/CAN
	Areas protected by NCC (Québec)		Nature Conservancy of Québec	Recived from the Nature Conservancy of Québec
Ontario	Protected areas (Canada)		Protected Planet	https://www.protectedplanet.net/country/CAN
	Areas protected by NCC (Ontario)		Nature Conservancy of Québec	Recived from the Nature Conservancy of Québec
NY/VT/MA	Protected areas (USA)		Protected Planet	https://www.protectedplanet.net/country/USA
	Areas protected by TNC (USA)		The Nature Conservancy	TNC Lands

Table S3.1 Map layers used for this study and their sources.

Canadian Land Cover, Circa 2000 NRS		
Class#	Category	Reclass#
0	No Data	0
11	Cloud	0
12	Shadow	0
20	Water	9
30	Barren/Rock	7
32	Rock/Rubble	7
33	Non-Veg Surfaces	7
34	Developed	8
40	Bryoids	4
50	Shrubland	4
51	Shrubland tall	4
52	Shrubland short	4
80	Wetland	5
81	Wetland - Treed	5
82	Wetland - Shrub	5
83	Wetland - Herb	5
100	Herb (Grasses/Crop)	4
110	Grassland	4
121	Annual Crops	6
122	Perrenial Crop land and Pasture	6
210	Coniferous Forest	1
211	Coniferous - Dense	1
212	Coniferous - open	1
213	Coniferous - Sparse	1
220	Deciduous Forest	2
221	Deciduous - dense	2
222	Deciduous - open	2
230	Mixed Forest	3
231	Mixed - Dense	3
232	Mixed - Open	3
233	Mixedwood Sparse	3

NLCD 2001 Land Cover (CONUS) USGS		
Class#	Category	Reclass#
0	Unclassified	0
11	Open Water	9
21	Development	8
22	Developed, Low Intensity	8
23	Developed, Medium Intensity	8
24	Developed, High Intensity	8
31	Barren Land	7
41	Deciduous Forest	2
42	Coniferous forest	1
43	Mixed Forest	3
52	Shrub/Scrub	4
71	Herbaceuous	4
81	Hay/Pasture	6
82	Cultivated Crops	6
90	Woody Wetlands	5
95	Emergent Herbaceuous Wetlands	5

North America Land Cover 2015 (Landsat and RapidEye)		
Class#	Category	Reclass#
0	Background	0
1	Temperate or sub-polar needleleaf forest	1
5	Temperate or sub-polar broadleaf deciduous forest	2
6	Mixed Forest	3
8	Temperate or sub-polar shrubland	4
10	Temperate or sub-polar grassland	4
12	Sub-polar or polar grassland-lichen-moss	4
13	Sub-polar or polar barren-lichen-moss	4
14	Wetland	5
15	Cropland	6
16	Barren Lands	7
17	Urban and Built-up	8
18	Water	9

Final 2000 & 2015 Reclass All Maps	
Class#	Category
0	NoData
1	Coniferous forest
2	Deciduous forest
3	Mixed forest
4	Grassland, Shrub, Moss, Herbaceous
5	Wetland
6	Agriculture
7	Barren Lands
8	Developed
9	Water

Table S3.2 Re-classification of the land-cover maps used in this study.

DMTI Roads Canada 2000 & 2018			
CARTO #	Category	Description	Re-Class
1	Expressway	400 series highways	1
2	Primary highway	Primary highways	2
3	Secondary highway	Secondary highway	2
4	Major road	Major road or arterial road	2
5	Local road	Subdivision road in a city or gravel road in a rural area	3

USA roads 2000 US Census Bureau			
CFCC #	Category	Description	Re-Class
A00	Primary highway with limited access	Interstate highways and some toll highways are in this category (A1) and are distinguished by the presence of interchanges.	1
A18			1
A21	Primary road without limited access	Nationally and regionally important highways that do not have limited access	2
A28			2
A31	Secondary and connecting road	State highways, but may include some county highways that connect smaller towns, subdivisions, and neighborhoods.	2
A38			2
A41	Local, neighborhood, and rural road	Local, Neighborhood, and rural road	3
A48	Vehicular trail	4WD access only dirt rural	3
A51			3
A53			3
A60	Road with special characteristics	Special road feature	3
A61	Cul de-sac		3
A62	Traffic circle		3
A63	Access ramp	The portion of a road that forms a cloverleaf or limited-access interchange	2
A64	Service drive	The road that provides access to businesses, facilities, and rest areas along a limited-access highway	2

USA roads 2015 US Census Bureau			
MTFCC #	Category	Description	Re-Class
S1100	Primary road	Primary roads are limited-access highways that connect to other roads	1
S1200	Secondary road	Secondary roads are main arteries that are not limited access, usually in the U.S. highway, state highway, or county highway systems.	2
S1400	Local	Local Neighborhood Road, Rural Road, City Street, Gravel, Dirt	3
S1500	Vehicular trail (4WD)	An unpaved dirt trail where a four-wheel drive vehicle is required	3
S1630	Ramp	A road that allows controlled access from adjacent roads onto a limited access highway, often in the form of a cloverleaf interchange.	1
S1640	Service	Service Drive usually along a limited access highway	1
S1730	Alley	It is located at the rear of buildings and properties and is used for deliveries	3
S1740	Private road	Private roads for service vehicles (logging, oil fields, ranches, etc.)	3

Final 2000 & 2015 North American Road Network Re-Class		
Class#	Category	Description
1	Primary road	All major highways
2	Secondary road	All secondary highways/major roads
3	Tertiary road	All other roads

Table S3.3 Re-classification of the road-network maps used in this study.

Previously published habitat suitability and resistance values for each species.		
Species	**Reference**	**Location**
Ursus americanus	WHCWG, 2010	Washington
	Graves & Wang, 2012	Adirondacks
	Albert et al., 2017	Québec/Ontario
	Gantchoff & Belant, 2017	Southcentral USA
	Nussey & Noseworthy, 2018	New Brunswick/Nova Scotia
	Steckler & Brickner-Wood, 2019	New Hampshire
Pekania pennanti	Garroway et al., 2011	Ontario
	Graves & Wang, 2012	Adirondacks
	Rohweder et al. 2012	Maine
	Nussey & Noseworthy, 2018	New Brunswick/Nova Scotia
	Steckler & Brickner-Wood, 2019	New Hampshire
Alces alces americanus	Graves & Wang, 2012	Adirondacks
	Rohweder et al. 2012	Maine
	Nussey & Noseworthy, 2018	New Brunswick/Nova Scotia
	Laliberté & St-Laurent, 2020	Québec
Odocoileus virginianus	Albert et al., 2017	Québec/Ontario
	Laliberté & St-Laurent, 2020	Québec

Table S3.4 List of published habitat suitability and resistance values used for this study.

Habitat Suitablity and Resisitance Values										
			Black Bear		Fisher		Moose		White-Tailed Deer	
Data Layer	Class ID	Class Description	HS	R	HS	R	HS	R	HS	R
Land	0	NoData	0	100	0	100	0	100	0	100
Land	1	Coniferous Forest	0.8	20	1	0	0.8	20	0.6	40
Land	2	Deciduous Forest	1	0	0.8	20	1	0	0.8	20
Land	3	Mixed Forest	1	0	1	0	1	0	1	0
Land	4	Grassland, Shrub, Moss, Herbaceous	0.6	40	0.4	60	0.6	40	0.8	20
Land	5	Wetland	0.8	20	0.6	40	0.6	40	0.6	40
Land	6	Agriculture	0.4	60	0.2	80	0.4	60	0.4	60
Land	7	Barren Lands	0.4	60	0	100	0.2	80	0.2	80
Land	8	Development	0.2	80	0.4	60	0.2	80	0.2	80
Land	9	Water	0	100	0	100	0	100	0	100
Roads	0	NoData	1	0	1	0	1	0	1	0
Roads	1	Primary	0	100	0	100	0	100	0	100
Roads	2	Secondary	0	100	0	100	0	100	0	100
Roads	3	Tertiary	0	100	0	100	0	100	0	100
Dist_Roads1	1	0-500m	0.4	60	1	0	0.4	60	1	0
Dist_Roads1	2	500-1000m	0.6	40	1	0	0.6	40	1	0
Dist_Roads1	3	1000-	1	0	1	0	1	0	1	0
Dist_Roads2	1	0-500m	0.6	40	1	0	0.6	40	1	0
Dist_Roads2	2	500-1000m	0.8	20	1	0	0.8	20	1	0
Dist_Roads2	3	1000-	1	0	1	0	1	0	1	0
Dist_Roads3	1	0-500m	0.8	20	1	0	1	0	1	0
Dist_Roads3	2	500-1000m	1	0	1	0	1	0	1	0
Dist_Roads3	3	1000-	1	0	1	0	1	0	1	0
Dist_Dev	1	0-500m	0.6	40	1	0	0.4	60	1	0
Dist_Dev	2	500-1000m	0.8	20	1	0	0.6	40	1	0
Dist_Dev	3	1000-	1	0	1	0	1	0	1	0

Table S3.5 List of suitable habitat and resistance values used for this study.

Species	Home Range (km^2)			Reference	Location
Ursus americanus	**Min**	**Average**	**Max**		
F	15.0	20.0	25.0	Schenk et al., 1998	Ontario
F		28.3		Koehler & Pierce, 2003	Washington
F		18.0		Koehler & Pierce, 2003	Washington
F		25.9		Koehler & Pierce, 2003	Washington
F	21.6	37.1	58.9	Lyons et al., 2003	Washington
F		65.1		Brodeur et al., 2008	Réserve faunique des Laurentides, QC
F		35.0		Gantchoff et al., 2018	Michigan/Missouri/Mississippi
Average	**18.3**	**32.8**	**42.0**		
M		125.5		Koehler & Pierce, 2003	Washington
M		**90.8**		Koehler & Pierce, 2003	Washington
M		**289.7**		Lyons et al., 2003	Washington
M		180.0		Gantchoff et al., 2018	Michigan/Missouri/Mississippi
Average	**90.8**	**171.5**	**289.7**		
Pekania pennanti					
F		16.3		Arthur et al., 1989	Maine
F	2.9	7.6	11.1	Fuller et al., 2001	Massachusetts
F	13.2	29.9	56.5	Tully, 2006	Algonquin Provincial Park, ON
F		3.6		Koen et al., 2007	Ontario
Average	**8.0**	**14.4**	**33.8**		
M		30.9		Arthur et al., 1989	Maine
M	6.5	10.0	16.6	Fuller et al., 2001	Massachusetts
M	38.1	38.7	39.3	Tully, 2006	Algonquin Provincial Park, ON
M		14.0		Koen et al., 2007	Ontario
Average	**22.3**	**23.4**	**28.0**		
Alces alces					
M/F		**25.0**		Leptich & Gilbert, 1989	Maine
M/F		**62.5**		Labonte et al., 1998	Quebec
M/F		53.9		Laurian et al., 2008	Laurentides Wildlife Reserve, QC
F		27.6		Murray et al., 2012	Algonquin Provical Park, ON
F		42.9		Murray et al., 2012	Wildlife Management Unit No. 49, ON
Average	**25.0**	**42.4**	**62.5**		
Odocoileus virginianus					
M/F		2.8		Drolet, 1976	New Brunswick
M/F	0.8	2.0	3.2	Nelson & Mech, 1981	Minnesota
M/F		2.3		Tierson et al., 1985	New York
M/F		0.7		Mooty et al., 1987	Minnesota
M/F	1.7	3.3	4.9	Van Deelen, 1995	Michigan
M/F	0.2	0.8	1.4	Franci, 2008	North America
Average	**0.9**	**2.1**	**3.1**		

Table S3.6 Species home range values from literature review used for this study: M = Male, F = Female

Core Mapper parameters for each species									
Run Name- will define beginning of core area file names	**Moving Window Radius** (smaller values result in larger numbers of more-detailed core areas)	**Min Average Habitat Value** (avg habitat value in the moving window around a pixel must be greater than this for the pixel to be considered 'core')	**Min Habitat Value Per Pixel** (Pixel value must be greater than this to be 'core')	**OPTIONAL: Expand cores by this CWD value** (Grows cores outward after minimum habitat values applied. Enter 0 to skip)	**Trim back expanded cores** (After CWD expansion, trim back cores by eliminating pixels using moving window average habitat values. This eliminates "halos" around core areas)	**Min Core Area size** (squared map units. Core areas smaller than this will be eliminated at end)	**Exclude nonhabitat from core size calcs** (Exclude pixels below per-pixel cutoffs from core area calculation when cores have been expanded. This makes core area calculation more conservative and eliminates cores with low amounts of habitat.)	**Append core stats** (can take extra time with very large numbers of cores)	**Delete Temporary Files**
Bear_SHP	9602.8	0.6	0	2413.5	Yes	68100000	No	Yes	Yes
Bear_SSP	9602.8	0.6	0	2413.5	Yes	10000000	No	Yes	Yes
Bear_OHP	9602.8	0.6	0.4	0	No	68100000	No	Yes	Yes
Fisher_SHP	5970.8	0.6	0	1595.8	Yes	24000000	No	Yes	Yes
Fisher_SSP	5970.8	0.6	0	1595.8	Yes	5000000	No	Yes	Yes
Fisher_OHP	5970.8	0.6	0.4	0	No	24000000	No	Yes	Yes
Moose_SHP	8920.6	0.6	0	2820.9	Yes	75000000	No	Yes	Yes
Moose_SSP	8920.6	0.6	0	2820.9	Yes	10000000	No	Yes	Yes
Moose_OHP	8920.6	0.6	0.4	0	No	75000000	No	Yes	Yes
Deer_SHP	4127.0	0.6	0	535.0	Yes	2660000	No	Yes	Yes
Deer_SSP	4127.0	0.6	0	535.0	Yes	1000000	No	Yes	Yes
Deer_OHP	4127.0	0.6	0.2	0	No	2660000	No	Yes	Yes

Table S3.7 Core Mapper parameters used for each species: SHP = Suitable habitat patches, SSP = Stepping-stone patches, OHP = Optimum habitat patches

Species	Average HR size (km^2)	Average HR size squared (km) = Linear dimension of HR	Median dispersal distance (km)	Maximum dispersal distance (km)
Black Bear - F	32.8	5.7	40.1	229.1
Black Bear - M	171.5	13.1	91.7	523.8
Fisher - F	14.4	3.8	26.6	151.8
Fisher - M	23.4	4.8	33.9	193.5
Moose - F+M	42.4	6.5	45.6	260.5
White-Tailed Deer - F+M	2.1	1.4	10.1	58.0

Table S3.8 Median and maximum dispersal distances for each species: F = Female, M = Male

MRC/County	2000						2015						2015-2000					
	Natural Land Cover (km^2)	Proportion (%)	Agriculture (km^2)	Proportion (%)	Development (km^2)	Proportion (%)	Natural Land Cover (km^2)	Proportion (%)	Agriculture (km^2)	Proportion (%)	Development (km^2)	Proportion (%)	Natural Land Cover (km^2)	Percent Change (%)	Agriculture (km^2)	Percent Change (%)	Development (km^2)	Percent Change (%)
Stormont/Dundas/Glengarry	1389	40	1856	53	68	2	1100	32	2010	58	182	5	-289	-21	154	8	114	166
Ottawa/Carleton	1381	48	1035	36	321	11	1150	40	1087	37	415	14	-232	-17	52	5	94	29
Prescott/Russel	708	34	1258	61	35	2	612	29	1261	61	106	5	-96	-14	4	0.3	71	203
Gatineau	158	41	70	18	113	29	123	32	87	22	106	27	-34	-22	17	24	-7	-6
Montgomery	430	40	543	50	87	8	430	40	541	50	90	8	-1	0	-2	0	3	3
Les Moulins	85	32	109	41	60	23	86	32	98	37	71	27	1	1	-11	-10	11	18
Montcalm	312	44	346	48	20	3	314	44	329	46	59	8	3	1	-17	-5	39	195
Deux-Montagnes	78	26	121	40	40	13	82	27	112	37	47	15	4	5	-9	-7	7	18
Thérèse-De Blainville	63	30	74	35	66	31	69	32	69	32	66	31	6	9	-5	-7	0	0
Mirabel	131	27	320	66	24	5	138	28	297	61	44	9	7	5	-23	-7	20	83
Les Jardins-des-Napierville	214	27	549	68	23	3	228	28	537	66	36	4	14	7	-12	-2	13	57
Beauharnois-Salaberry	48	9	370	67	41	7	73	13	342	62	49	9	25	52	-28	-8	8	20
Roussillon	63	13	272	55	80	16	88	18	239	49	86	17	25	40	-33	-12	6	8
Laval	24	9	91	34	129	48	57	21	48	18	133	50	33	137	-43	-47	4	3
Vaudreuil-Soulanges	194	19	559	55	80	8	235	23	509	50	104	10	41	21	-50	-9	24	30
Montréal	18	3	41	7	438	70	67	11	19	3	396	63	49	265	-22	-54	-42	-10
Le Haut-Saint-Laurent	457	35	676	52	20	2	511	39	611	47	48	4	53	12	-65	-10	28	140

Table S3.9 Changes in land-cover area (km^2) and proportion (%) for each set of grouped land-cover categories between 2000 and 2015, for the 17 MRCs/counties that had > 50% of their surface area dedicated to agriculture and development.

Black bear suitable habitat fragmentation (km^2)										
Rank	MRC/County	2000			2015			2015-2000		
		m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}	m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}	m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}
1	Antoine-Labelle	15890.5	73379.0	57488.5	13987.4	61957.2	47969.8	-1903.1	-11421.8	-9518.7
2	La Vallée-de-la-Gatineau	13170.4	71686.0	58515.6	10977.4	58898.5	47921.1	-2193.0	-12787.4	-10594.4
3	Matawinie	9007.9	68940.6	59932.8	8091.3	58802.3	50711.0	-916.6	-10138.4	-9221.8
4	Hamilton	4733.4	24305.1	19571.7	4733.2	24332.8	19599.6	-0.2	27.7	28.0
5	Essex	3270.4	19645.8	16375.4	3256.2	19626.1	16369.9	-14.1	-19.6	-5.5
6	Franklin	2306.3	17547.9	15241.6	2285.5	17488.9	15203.4	-20.8	-59.0	-38.2
7	Papineau	2667.1	67617.6	64950.5	1537.5	46203.6	44666.1	-1129.5	-21414.0	-20284.5
8	Herkimer	1511.5	15285.0	13773.5	1517.9	15334.7	13816.9	6.3	49.7	43.4
9	St Lawrence	1483.6	10915.5	9431.9	1485.3	10934.5	9449.2	1.7	19.0	17.3
10	Warren	1105.0	16356.9	15251.9	1045.0	15925.2	14880.2	-60.0	-431.7	-371.7
11	Clinton	904.5	13550.8	12646.3	1006.7	14312.7	13306.0	102.2	761.9	659.7
12	Les Laurentides	2569.5	72677.2	70107.8	924.9	39242.3	38317.4	-1644.5	-33434.9	-31790.4
13	Lewis	661.5	9641.4	8979.9	669.8	9643.4	8973.6	8.3	2.0	-6.3
14	Fulton	291.1	11084.0	10792.9	286.2	11003.0	10716.8	-4.9	-80.9	-76.0
15	Les Collines-de-l'Outaouais	1538.3	62112.9	60574.6	276.0	23678.1	23402.1	-1262.3	-38434.8	-37172.5
16	Lanark	873.7	2414.5	1540.8	257.6	769.0	511.4	-616.1	-1645.5	-1029.4
17	Saratoga	202.6	7347.2	7144.6	192.3	7166.6	6974.3	-10.3	-180.5	-170.3
18	Argenteuil	642.3	51413.2	50770.9	84.0	16728.9	16644.9	-558.3	-34684.3	-34126.0
19	Oswego	43.9	164.3	120.4	48.8	175.7	126.9	4.9	11.4	6.5
20	Washington	42.2	3367.9	3325.8	42.1	3369.7	3327.6	-0.1	1.7	1.8
21	Oneida	30.6	1722.9	1692.3	26.4	1664.9	1638.5	-4.2	-58.1	-53.9
22	Jefferson	8.5	859.8	851.3	9.6	940.7	931.2	1.0	80.9	79.8
23	Le Haut-Saint-Laurent	4.8	1479.3	1474.5	6.2	1679.7	1673.5	1.4	200.4	199.1
24	Leeds/Grenville	140.3	264.5	124.2	3.6	3.6	0.0	-136.6	-260.8	-124.2
25	Ottawa/Carleton	21.0	109.5	88.5	3.5	8.3	4.8	-17.6	-101.2	-83.6
26	Les Pays-d'en-Haut	392.4	54133.5	53741.1	0.1	753.7	753.6	-392.3	-53379.8	-52987.5
27	Les Jardins-de-Napierville	0.0	0.0	0.0	0.0	146.0	146.0	0.0	146.0	146.0
28	Gatineau	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
29	La Rivière-du-Nord	32.8	19683.4	19650.6	0.0	0.0	0.0	-32.8	-19683.4	-19650.6
30	Montcalm	36.0	16621.1	16585.1	0.0	0.0	0.0	-36.0	-16621.1	-16585.1
31	Mirabel	0.0	237.4	237.4	0.0	0.0	0.0	0.0	-237.4	-237.4
32	Thérèse-De Blainville	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33	Les Moulins	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
34	Deux-Montagnes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
35	Laval	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
36	Montréal	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
37	Vaudreuil-Soulanges	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
38	Roussillon	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
39	Beauharnois-Salaberry	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
40	Prescott/Russel	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
41	Stormont/Dundas/Glengarry	33.3	33.3	0.0	0.0	0.0	0.0	-33.3	-33.3	0.0
42	Montgomery	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
43	Schenectady	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Average		**1479.4**	**16618.5**	**15139.1**	**1226.8**	**10716.1**	**9489.2**	**-252.6**	**-5902.5**	**-5649.9**

Table S3.10 Suitable habitat patch (SHP) effective mesh size values (m_{eff_CUT}, m_{eff_CBC}, and m_{eff_CBC} - m_{eff_CUT}) for black bear in 2000 and 2015. MRCs/Counties ranked from highest to lowest 2015 m_{eff_CUT} value (i.e., from lowest to highest amount of fragmentation).

Black bear optimal habitat fragmentation (km²)										
Rank	MRC/County	2000			2015			2015-2000		
		m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}	m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}	m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}
1	Antoine-Labelle	4872.7	10761.6	5888.9	1586.0	3070.8	1484.8	-3286.7	-7690.8	-4404.0
2	Matawinie	3727.1	11756.8	8029.7	1282.6	3229.8	1947.2	-2444.5	-8527.0	-6082.5
3	Hamilton	419.1	852.5	433.4	389.7	729.9	340.2	-29.4	-122.7	-93.2
4	La Vallée-de-la-Gatineau	340.6	2007.5	1666.9	366.2	507.0	140.7	25.6	-1500.5	-1526.1
5	Essex	302.1	427.0	124.9	302.3	363.0	60.7	0.2	-64.0	-64.2
6	Herkimer	296.0	730.3	434.3	256.7	629.3	372.6	-39.3	-101.0	-61.7
7	Franklin	70.5	177.3	106.8	61.0	124.1	63.1	-9.4	-53.1	-43.7
8	Les Laurentides	98.5	3466.9	3368.4	60.5	1310.9	1250.4	-38.0	-2156.0	-2118.0
9	St Lawrence	46.1	128.3	82.2	59.8	143.4	83.6	13.7	15.1	1.4
10	Warren	72.3	147.0	74.8	58.6	107.6	49.0	-13.6	-39.4	-25.8
11	Lewis	55.7	147.1	91.4	36.3	103.5	67.2	-19.4	-43.6	-24.2
12	Papineau	79.4	108.7	29.3	22.1	26.7	4.6	-57.4	-82.0	-24.7
13	Fulton	19.9	75.7	55.8	19.9	76.0	56.1	0.0	0.3	0.4
14	Saratoga	23.0	30.5	7.6	16.3	20.8	4.5	-6.7	-9.8	-3.1
15	Clinton	15.6	17.9	2.3	11.7	14.5	2.8	-3.9	-3.4	0.5
16	Washington	5.4	5.8	0.4	5.2	5.6	0.4	-0.2	-0.2	0.0
17	Les Collines-de-l'Outaouais	32.1	48.9	16.8	4.5	8.4	3.9	-27.6	-40.5	-12.9
18	Argenteuil	6.3	24.3	18.0	3.9	6.2	2.3	-2.4	-18.1	-15.7
19	Lanark	9.7	13.8	4.1	3.9	5.0	1.1	-5.9	-8.8	-3.0
20	Oswego	1.8	2.9	1.1	1.8	3.2	1.4	0.0	0.3	0.3
21	Oneida	1.3	37.6	36.3	0.8	10.4	9.6	-0.5	-27.3	-26.7
22	Le Haut-Saint-Laurent	0.4	1.4	1.0	0.4	1.6	1.2	0.0	0.2	0.2
23	Jefferson	0.4	0.8	0.5	0.3	0.9	0.5	0.0	0.1	0.1
24	Les Pays-d'en-Haut	5.2	10.0	4.8	0.0	0.4	0.4	-5.2	-9.5	-4.4
25	Leeds/Grenville	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
26	Ottawa/Carleton	1.9	1.9	0.0	0.0	0.0	0.0	-1.9	-1.9	0.0
27	Les Jardins-de-Napierville	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
28	Gatineau	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
29	La Rivière-du-Nord	0.2	1.5	1.3	0.0	0.0	0.0	-0.2	-1.5	-1.3
30	Montcalm	1.5	4.1	2.6	0.0	0.0	0.0	-1.5	-4.1	-2.6
31	Mirabel	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
32	Thérèse-De Blainville	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33	Les Moulins	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
34	Deux-Montagnes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
35	Laval	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
36	Montréal	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
37	Vaudreuil-Soulanges	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
38	Roussillon	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
39	Beauharnois-Salaberry	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
40	Prescott/Russel	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
41	Stormont/Dundas/Glengarry	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
42	Montgomery	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
43	Schenectady	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Average		**244.3**	**720.7**	**476.4**	**105.8**	**244.2**	**138.3**	**-138.5**	**-476.5**	**-338.0**

Table S3.11 Optimum habitat patch (OHP) effective mesh size values (m_{eff_CUT}, m_{eff_CBC}, and m_{eff_CBC} - m_{eff_CUT}) for black bear in 2000 and 2015. MRCs/Counties ranked from highest to lowest 2015 m_{eff_CUT} value (i.e., from lowest to highest amount of fragmentation).

Fisher suitable habitat fragmentation (km²)											
Rank	MRC/County	2000			2015			2015-2000			
		m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}	m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}	m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}
1	Antoine-Labelle	15639.0	72642.3	57003.3	15240.2	71972.3	56732.1	-398.8	-670.0	-271.2
2	La Vallée-de-la-Gatineau	13105.5	71358.2	58252.6	13088.0	71571.2	58483.2	-17.5	213.1	230.6
3	Matawinie	9052.1	68963.7	59911.6	9479.5	70831.0	61351.6	427.4	1867.3	1439.9
4	Hamilton	4733.4	38422.5	33689.1	4733.4	38353.0	33619.5	0.0	-69.5	-69.5
5	Essex	3992.7	34315.5	30322.8	3975.7	34180.6	30204.9	-16.9	-134.8	-117.9
6	Franklin	2873.5	30962.5	28089.0	2881.9	30951.4	28069.5	8.4	-11.1	-19.5
7	Les Laurentides	2678.4	74045.6	71367.2	2677.4	74302.5	71625.1	-1.0	256.9	257.9
8	St Lawrence	2572.8	22720.1	20147.4	2567.9	22658.1	20090.2	-4.9	-62.0	-57.2
9	Papineau	2561.0	66119.2	63558.3	2246.4	62151.8	59905.4	-314.6	-3967.4	-3652.8
10	Warren	2139.0	35976.5	33837.5	2114.6	35706.0	33591.4	-24.4	-270.5	-246.1
11	Lewis	1920.3	28974.4	27054.2	1898.9	28760.5	26861.6	-21.4	-213.9	-192.5
12	Herkimer	1753.1	26022.7	24269.6	1748.2	25938.6	24190.5	-5.0	-84.1	-79.1
13	Clinton	1310.8	25788.7	24477.8	1421.8	26809.7	25387.9	111.0	1021.1	910.1
14	Les Collines-de-l'Outaouais	1667.0	64521.5	62854.6	1333.9	57928.4	56594.5	-333.0	-6593.1	-6260.0
15	Lanark	1032.4	2995.2	1962.7	829.7	2253.7	1424.1	-202.8	-741.4	-538.6
16	Les Pays-d'en-Haut	737.3	74045.6	73308.3	737.3	74316.3	73579.0	0.0	270.7	270.7
17	Argenteuil	739.3	55041.2	54301.9	701.8	53825.9	53124.0	-37.4	-1215.4	-1177.9
18	Fulton	619.5	25561.7	24942.2	612.9	25378.9	24766.0	-6.6	-182.8	-176.2
19	Oneida	585.2	16173.4	15588.2	579.7	16068.8	15489.1	-5.4	-104.6	-99.1
20	Saratoga	610.2	20206.1	19595.9	579.6	19651.1	19071.6	-30.7	-555.0	-524.3
21	Washington	524.7	18679.2	18154.5	422.5	16730.7	16308.2	-102.2	-1948.5	-1846.3
22	Oswego	330.7	11494.5	11163.8	327.4	11417.7	11090.3	-3.3	-76.8	-73.5
23	La Rivière-du-Nord	169.3	44624.0	44454.7	244.4	53805.2	53560.8	75.0	9181.1	9106.1
24	Schenectady	211.5	2566.3	2354.8	184.7	2351.2	2166.5	-26.8	-215.1	-188.3
25	Montcalm	47.4	19040.4	18993.0	62.2	21894.3	21832.1	14.8	2853.9	2839.0
26	Jefferson	58.4	4330.9	4272.5	59.3	4357.5	4298.2	0.9	26.6	25.7
27	Leeds/Grenville	140.4	371.4	231.0	22.9	80.8	57.8	-117.4	-290.6	-173.2
28	Le Haut-Saint-Laurent	1.2	1148.1	1146.9	21.1	4882.9	4861.9	19.9	3734.9	3714.9
29	Ottawa/Carleton	22.5	111.2	88.7	10.2	23.2	13.0	-12.3	-88.0	-75.7
30	Les Jardins-de-Napierville	0.0	225.7	225.6	7.3	3644.2	3636.9	7.3	3418.6	3411.3
31	Montgomery	7.6	348.6	341.0	6.8	321.7	315.0	-0.9	-26.8	-26.0
32	Prescott/Russel	8.4	9.1	0.7	1.6	1.6	0.0	-6.8	-7.5	-0.7
33	Mirabel	0.0	602.1	602.1	0.3	1992.6	1992.2	0.3	1390.5	1390.1
34	Stormont/Dundas/Glengarry	27.8	27.8	0.0	0.0	0.0	0.0	-27.8	-27.8	0.0
35	Les Moulins	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
36	Thérèse-De Blainville	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
37	Montréal	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
38	Laval	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
39	Deux-Montagnes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
40	Gatineau	0.5	2711.0	2710.5	0.0	0.0	0.0	-0.5	-2711.0	-2710.5
41	Vaudreuil-Soulanges	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
42	Roussillon	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
43	Beauharnois-Salaberry	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Average		**1671.5**	**22352.3**	**20680.8**	**1647.0**	**22444.5**	**20797.5**	**-24.5**	**92.3**	**116.7**

Table S3.12 Suitable habitat patch (SHP) effective mesh size values (m_{eff_CUT}, m_{eff_CBC}, and m_{eff_CBC} - m_{eff_CUT}) for fisher in 2000 and 2015. MRCs/Counties ranked from highest to lowest 2015 m_{eff_CUT} value (i.e., from lowest to highest amount of fragmentation).

Moose suitable habitat fragmentation (km^2)										
Rank	MRC/County	2000			2015			2015-2000		
		m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}	m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}	m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}
1	Antoine-Labelle	15696.4	71593.0	55896.6	11266.9	47840.0	36573.1	-4429.5	-23753.0	-19323.5
2	La Vallée-de-la-Gatineau	13085.5	70145.1	57059.6	8447.4	44451.1	36003.7	-4638.1	-25694.1	-21055.9
3	Matawinie	8548.3	65928.3	57380.0	6089.2	43886.4	37797.3	-2459.1	-22041.9	-19582.8
4	Hamilton	4675.5	12326.7	7651.2	4666.8	12265.8	7599.0	-8.8	-60.9	-52.1
5	Essex	1708.3	7245.6	5537.3	1679.1	7154.6	5475.5	-29.2	-91.1	-61.8
6	Herkimer	1171.5	6866.7	5695.2	1170.3	6835.7	5665.4	-1.2	-31.0	-29.8
7	Warren	222.0	3741.3	3519.3	220.4	3713.1	3492.6	-1.6	-28.3	-26.7
8	Papineau	2297.6	61609.8	59312.1	216.4	14563.2	14346.7	-2081.2	-47046.6	-44965.4
9	Franklin	111.9	1498.1	1386.2	109.9	1481.6	1371.7	-2.0	-16.4	-14.5
10	Les Laurentides	2127.2	64915.7	62788.5	77.2	9752.9	9675.7	-2050.0	-55162.8	-53112.8
11	Fulton	61.8	2605.5	2543.7	61.0	2577.8	2516.8	-0.8	-27.7	-26.8
12	Lewis	44.2	1369.1	1324.8	44.9	1359.1	1314.2	0.6	-10.0	-10.6
13	St Lawrence	39.6	887.4	847.9	38.9	880.7	841.8	-0.7	-6.7	-6.0
14	Clinton	15.2	276.3	261.1	15.2	266.6	251.4	0.1	-9.7	-9.7
15	Lanark	926.9	2569.1	1642.2	9.4	25.7	16.3	-917.5	-2543.5	-1626.0
16	Saratoga	5.5	619.6	614.1	5.1	590.6	585.5	-0.5	-29.0	-28.6
17	Oneida	1.1	228.3	227.2	1.1	222.5	221.5	0.0	-5.8	-5.8
18	Oswego	0.7	2.3	1.5	0.8	2.6	1.8	0.0	0.3	0.3
19	Les Collines-de-l'Outaouais	1402.0	58210.2	56808.2	0.3	638.1	637.8	-1401.7	-57572.2	-56170.5
20	Argenteuil	526.4	45692.1	45165.6	0.0	0.9	0.9	-526.4	-45691.2	-45164.8
21	Jefferson	0.0	0.1	0.0	0.0	0.1	0.1	0.0	0.0	0.0
22	Stormont/Dundas/Glengarry	71.7	71.7	0.0	0.0	0.0	0.0	-71.7	-71.7	0.0
23	Ottawa/Carleton	18.2	93.3	75.0	0.0	0.0	0.0	-18.2	-93.3	-75.0
24	Leeds/Grenville	120.6	234.7	114.1	0.0	0.0	0.0	-120.6	-234.7	-114.1
25	Prescott/Russel	14.0	14.7	0.7	0.0	0.0	0.0	-14.0	-14.7	-0.7
26	Montgomery	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
27	Schenectady	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
28	Washington	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
29	Les Pays-d'en-Haut	237.4	41333.8	41096.4	0.0	0.0	0.0	-237.4	-41333.8	-41096.4
30	Montcalm	22.6	12949.8	12927.2	0.0	0.0	0.0	-22.6	-12949.8	-12927.2
31	La Rivière-du-Nord	2.9	5727.7	5724.8	0.0	0.0	0.0	-2.9	-5727.7	-5724.8
32	Les Moulins	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33	Thérèse-De Blainville	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
34	Mirabel	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
35	Montréal	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
36	Laval	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
37	Deux-Montagnes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
38	Gatineau	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
39	Vaudreuil-Soulanges	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
40	Roussillon	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
41	Beauhamois-Salaberry	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
42	Les Jardins-de-Napierville	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
43	Le Haut-Saint-Laurent	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Average		1236.2	12529.2	11293.0	793.5	4616.5	3823.0	-442.7	-7912.7	-7470.0

Table S3.13 Suitable habitat patch (SHP) effective mesh size values (m_{eff_CUT}, m_{eff_CBC}, and m_{eff_CBC} - m_{eff_CUT}) for moose in 2000 and 2015. MRCs/Counties ranked from highest to lowest 2015 m_{eff_CUT} value (i.e., from lowest to highest amount of fragmentation).

Moose optimal habitat fragmentation (km^2)										
Rank	MRC/County	2000			2015			2015-2000		
		m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}	m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}	m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}
1	Antoine-Labelle	4746.7	10413.7	5667.1	1528.4	2915.3	1387.0	-3218.3	-7498.4	-4280.1
2	Matawinie	3518.4	11193.8	7675.5	1167.6	2980.1	1812.4	-2350.7	-8213.8	-5863.0
3	Hamilton	418.4	809.4	391.1	387.6	696.2	308.6	-30.8	-113.3	-82.5
4	La Vallée-de-la-Gatineau	336.8	1964.7	1628.0	273.8	388.2	114.4	-63.0	-1576.5	-1513.6
5	Essex	260.0	370.7	110.7	257.7	312.4	54.6	-2.3	-58.3	-56.0
6	Herkimer	274.0	655.0	381.1	236.7	572.1	335.4	-37.3	-82.9	-45.6
7	Warren	57.6	119.0	61.5	44.0	83.8	39.8	-13.6	-35.3	-21.7
8	Les Laurentides	92.5	3277.9	3185.4	43.0	1120.5	1077.5	-49.5	-2157.4	-2108.0
9	Franklin	33.5	125.2	91.7	21.6	76.3	54.7	-11.9	-48.9	-37.0
10	St Lawrence	18.3	78.0	59.7	19.8	78.6	58.7	1.6	0.6	-1.0
11	Fulton	12.5	64.0	51.5	12.5	64.4	51.9	0.0	0.4	0.4
12	Papineau	78.6	105.3	26.6	10.4	11.0	0.6	-68.3	-94.3	-26.0
13	Lewis	17.9	73.5	55.5	9.9	54.8	44.9	-8.0	-18.7	-10.7
14	Saratoga	5.2	9.4	4.2	2.6	5.1	2.5	-2.6	-4.3	-1.8
15	Clinton	2.5	3.1	0.6	2.6	3.6	1.0	0.0	0.4	0.4
16	Lanark	9.5	13.1	3.6	0.5	1.2	0.7	-9.1	-11.9	-2.8
17	Oneida	0.4	18.6	18.3	0.1	8.8	8.7	-0.3	-9.9	-9.6
18	Les Collines-de-l'Outaouais	24.5	36.4	11.9	0.1	0.4	0.4	-24.5	-36.0	-11.5
19	Oswego	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
20	Argenteuil	6.1	23.7	17.6	0.0	0.0	0.0	-6.1	-23.7	-17.6
21	Jefferson	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
22	Stormont/Dundas/Glengarry	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
23	Ottawa/Carleton	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
24	Leeds/Grenville	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
25	Prescott/Russel	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
26	Montgomery	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
27	Schenectady	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
28	Washington	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
29	Les Pays-d'en-Haut	0.2	1.8	1.6	0.0	0.0	0.0	-0.2	-1.8	-1.6
30	Montcalm	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
31	La Rivière-du-Nord	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
32	Les Moulins	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33	Thérèse-De Blainville	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
34	Mirabel	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
35	Montréal	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
36	Laval	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
37	Deux-Montagnes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
38	Gatineau	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
39	Vaudreuil-Soulanges	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
40	Roussillon	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
41	Beauharnois-Salaberry	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
42	Les Jardins-de-Napierville	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
43	Le Haut-Saint-Laurent	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Average	**230.5**	**682.7**	**452.2**	**93.5**	**218.0**	**124.5**	**-137.1**	**-464.7**	**-327.7**

Table S3.14 Optimum habitat patch (OHP) effective mesh size values (m_{eff_CUT}, m_{eff_CBC}, and m_{eff_CBC} - m_{eff_CUT}) for moose in 2000 and 2015. MRCs/Counties ranked from highest to lowest 2015 m_{eff_CUT} value (i.e., from lowest to highest amount of fragmentation).

White-tailed deer suitable habitat fragmentation (km^2)										
Rank	MRC/County	2000			2015			2015-2000		
		m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}	m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}	m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}
1	Antoine-Labelle	15305.7	81086.9	65781.3	14510.1	68775.3	54265.3	-795.6	-12311.6	-11516.0
2	La Vallée-de-la-Gatineau	12759.0	79444.7	66685.7	12031.6	67203.2	55171.7	-727.4	-12241.4	-11514.0
3	Matawinie	8903.4	77172.7	68269.4	8941.0	67367.8	58426.8	37.6	-9804.9	-9842.5
4	Hamilton	4468.6	36093.9	31625.3	4424.0	30187.4	25763.4	-44.6	-5906.5	-5861.9
5	Essex	3762.5	32206.7	28444.3	3750.3	27027.8	23277.5	-12.2	-5178.9	-5166.7
6	Les Laurentides	2677.0	83526.0	80849.0	2614.5	71905.8	69291.4	-62.5	-11620.2	-11557.7
7	St Lawrence	2504.5	21646.2	19141.7	2552.1	18364.1	15812.0	47.6	-3282.1	-3329.7
8	Franklin	2452.9	27660.2	25207.3	2496.4	23455.5	20959.1	43.5	-4204.7	-4248.2
9	Papineau	2769.9	77588.7	74818.8	2478.9	63939.0	61460.1	-291.0	-13649.8	-13358.7
10	Lewis	1871.2	27653.1	25782.0	1878.9	23292.2	21413.3	7.8	-4360.9	-4368.7
11	Lanark	1886.3	64032.1	62145.8	1765.1	5293.2	3528.1	-121.2	-58738.8	-58617.7
12	Herkimer	1700.0	24758.0	23058.0	1681.8	20693.4	19011.7	-18.2	-4064.5	-4046.3
13	Les Collines-de-l'Outaouais	1918.0	78092.9	76174.8	1562.5	61398.8	59836.3	-355.6	-16694.1	-16338.5
14	Warren	1564.5	29747.9	28183.4	1518.2	24668.7	23150.4	-46.3	-5079.2	-5032.9
15	Clinton	1276.9	24608.8	23331.9	1325.0	21071.0	19746.0	48.1	-3537.8	-3585.9
16	Argenteuil	823.1	65531.7	64708.6	787.8	55849.4	55061.6	-35.3	-9682.3	-9647.0
17	Les Pays-d'en-Haut	737.3	83548.6	82811.3	664.8	69111.7	68446.9	-72.5	-14436.9	-14364.5
18	Oswego	357.5	10962.9	10605.3	587.1	12484.0	11897.0	229.5	1521.2	1291.6
19	Oneida	518.8	14794.1	14275.3	519.9	12463.8	11943.9	1.1	-2330.3	-2331.4
20	Fulton	490.1	21980.2	21490.1	493.1	18533.0	18039.9	3.0	-3447.2	-3450.3
21	Leeds/Grenville	1331.5	49614.0	48282.5	388.5	2281.4	1892.9	-943.0	-47332.6	-46389.6
22	Washington	340.5	14547.9	14207.5	326.7	2451.3	2124.6	-13.8	-12096.7	-12082.9
23	Saratoga	306.2	13887.4	13581.2	294.8	11476.8	11182.0	-11.4	-2410.6	-2399.1
24	La Rivière-du-Nord	309.9	68112.8	67802.9	292.2	57620.1	57327.9	-17.6	-10492.6	-10475.0
25	Schenectady	222.9	3642.4	3419.5	218.4	3620.3	3401.9	-4.5	-22.1	-17.6
26	Le Haut-Saint-Laurent	72.1	8749.4	8677.3	123.5	9562.1	9438.6	51.4	812.7	761.3
27	Jefferson	91.4	5225.5	5134.1	95.1	4477.7	4382.6	3.7	-747.8	-751.5
28	Montcalm	83.2	28434.9	28351.7	85.7	25127.8	25042.1	2.5	-3307.1	-3309.6
29	Stormont/Dundas/Glengarry	105.8	1626.0	1520.3	58.5	58.5	0.0	-47.3	-1567.5	-1520.3
30	Les Jardins-de-Napierville	21.1	6012.2	5991.1	47.3	7559.7	7512.4	26.2	1547.5	1521.3
31	Ottawa/Carleton	219.7	22037.5	21817.9	25.2	644.3	619.1	-194.4	-21393.2	-21198.8
32	Montgomery	15.0	674.7	659.7	14.9	675.7	660.8	-0.1	1.0	1.1
33	Prescott/Russel	103.0	116.9	13.8	11.1	12.3	1.2	-91.9	-104.6	-12.7
34	Vaudreuil-Soulanges	1.1	1.1	0.0	5.6	5.6	0.0	4.5	4.5	0.0
35	Mirabel	1.3	4316.1	4314.8	3.8	6435.4	6431.6	2.5	2119.3	2116.8
36	Gatineau	19.8	18921.7	18901.9	0.1	1255.5	1255.3	-19.7	-17666.2	-17646.5
37	Thérèse-De Blainville	0.4	3697.8	3697.4	0.0	414.7	414.7	-0.4	-3283.1	-3282.7
38	Les Moulins	0.0	200.6	200.6	0.0	0.0	0.0	0.0	-200.6	-200.6
39	Montréal	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
40	Laval	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
41	Deux-Montagnes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
42	Roussillon	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
43	Beauharnois-Salaberry	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Average		**1674.2**	**28185.0**	**26510.8**	**1594.8**	**20855.0**	**19260.2**	**-79.5**	**-7330.0**	**-7250.5**

Table S3.15 Suitable habitat patch (SHP) effective mesh size values ($m_{\text{eff\_CUT}}$, $m_{\text{eff\_CBC}}$, and $m_{\text{eff\_CBC}}$ - $m_{\text{eff\_CUT}}$) for white-tailed deer in 2000 and 2015. MRCs/Counties ranked from highest to lowest 2015 $m_{\text{eff\_CUT}}$ value (i.e., from lowest to highest amount of fragmentation).

White-tailed deer optimal habitat fragmentation (km^2)											
Rank	MRC/County	2000			2015			2015-2000			
		m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}	m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}	m_{eff_CUT}	m_{eff_CBC}	m_{eff_CBC} - m_{eff_CUT}
1	Matawinie	3606.2	11147.2	7541.0	2329.5	5431.0	3101.5	-1276.7	-5716.1	-4439.5
2	Antoine-Labelle	4839.9	10339.7	5499.9	1828.7	3982.4	2153.7	-3011.2	-6357.3	-3346.2
3	La Vallée-de-la-Gatineau	342.2	1943.1	1600.8	465.0	653.7	188.8	122.7	-1289.3	-1412.0
4	Hamilton	580.1	1550.9	970.8	380.2	711.2	331.0	-199.9	-839.7	-639.8
5	Essex	651.1	1206.5	555.4	308.4	366.7	58.3	-342.7	-839.9	-497.1
6	Herkimer	390.9	1025.9	635.0	255.2	631.1	375.9	-135.7	-394.8	-259.0
7	Les Laurentides	110.6	3358.2	3247.6	77.5	1677.9	1600.4	-33.1	-1680.3	-1647.2
8	Warren	126.1	572.0	446.0	67.3	115.1	47.8	-58.7	-457.0	-398.2
9	Franklin	183.8	467.2	283.4	65.4	125.5	60.1	-118.4	-341.7	-223.3
10	St Lawrence	108.5	276.2	167.7	64.5	144.1	79.6	-44.0	-132.1	-88.1
11	Lewis	144.5	514.7	370.2	62.7	144.5	81.8	-81.8	-370.2	-288.4
12	Les Collines-de-l'Outaouais	44.0	61.2	17.2	45.9	53.0	7.0	2.0	-8.2	-10.2
13	Papineau	87.7	117.5	29.8	44.2	53.5	9.3	-43.4	-64.0	-20.5
14	Fulton	27.7	90.5	62.7	27.2	83.7	56.5	-0.5	-6.8	-6.3
15	Clinton	36.9	43.5	6.6	20.8	24.1	3.2	-16.1	-19.4	-3.4
16	Saratoga	44.2	81.6	37.5	18.6	23.1	4.6	-25.6	-58.5	-32.9
17	Lanark	20.8	25.4	4.7	17.5	19.5	2.0	-3.3	-5.9	-2.6
18	Argenteuil	18.1	38.7	20.6	17.1	25.2	8.1	-0.9	-13.4	-12.5
19	Les Pays-d'en-Haut	15.1	24.4	9.3	14.9	26.7	11.8	-0.2	2.3	2.5
20	Washington	18.5	48.3	29.8	7.7	8.8	1.0	-10.7	-39.5	-28.8
21	La Rivière-du-Nord	5.5	9.7	4.2	5.4	11.0	5.6	-0.1	1.3	1.4
22	Oneida	7.6	67.2	59.6	4.7	14.7	10.0	-2.9	-52.5	-49.6
23	Montcalm	4.5	7.6	3.1	4.5	7.3	2.8	0.0	-0.4	-0.4
24	Oswego	12.6	29.2	16.6	4.5	6.0	1.6	-8.2	-23.2	-15.0
25	Ottawa/Carleton	7.2	7.3	0.1	4.2	4.3	0.1	-2.9	-3.0	0.0
26	Leeds/Grenville	6.0	6.2	0.2	4.0	4.1	0.1	-2.0	-2.1	-0.1
27	Le Haut-Saint-Laurent	2.4	3.7	1.3	3.2	4.6	1.5	0.8	0.9	0.1
28	Jefferson	2.1	7.7	5.6	2.7	3.5	0.8	0.6	-4.2	-4.8
29	Les Jardins-de-Napierville	1.9	2.1	0.2	2.7	3.0	0.3	0.8	0.9	0.1
30	Schenectady	9.3	9.9	0.6	2.2	2.3	0.1	-7.1	-7.6	-0.5
31	Stormont/Dundas/Glengarry	1.5	1.5	0.0	1.0	1.0	0.0	-0.5	-0.5	0.0
32	Prescott/Russel	1.3	1.3	0.0	0.5	0.5	0.0	-0.8	-0.8	0.0
33	Montgomery	0.6	0.9	0.2	0.5	0.5	0.1	-0.1	-0.3	-0.2
34	Vaudreuil-Soulanges	0.2	0.2	0.0	0.3	0.3	0.0	0.2	0.2	0.0
35	Mirabel	0.2	0.7	0.6	0.3	0.8	0.6	0.1	0.1	0.0
36	Thérèse-De Blainville	0.3	0.8	0.5	0.0	0.1	0.1	-0.3	-0.7	-0.4
37	Gatineau	0.5	0.8	0.3	0.0	0.1	0.1	-0.5	-0.8	-0.3
38	Les Moulins	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
39	Montréal	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
40	Laval	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
41	Deux-Montagnes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
42	Roussillon	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
43	Beauharnois-Salaberry	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Average		**266.5**	**769.5**	**503.0**	**143.2**	**334.1**	**190.8**	**-123.3**	**-435.5**	**-312.2**

Table S3.16 Optimum habitat patch (OHP) effective mesh size values ($m_{\mathrm{eff\_CUT}}$, $m_{\mathrm{eff\_CBC}}$, and $m_{\mathrm{eff\_CBC}}$ - $m_{\mathrm{eff\_CUT}}$) for white-tailed deer in 2000 and 2015. MRCs/Counties ranked from highest to lowest 2015 $m_{\mathrm{eff\_CUT}}$ value (i.e., from lowest to highest amount of fragmentation).

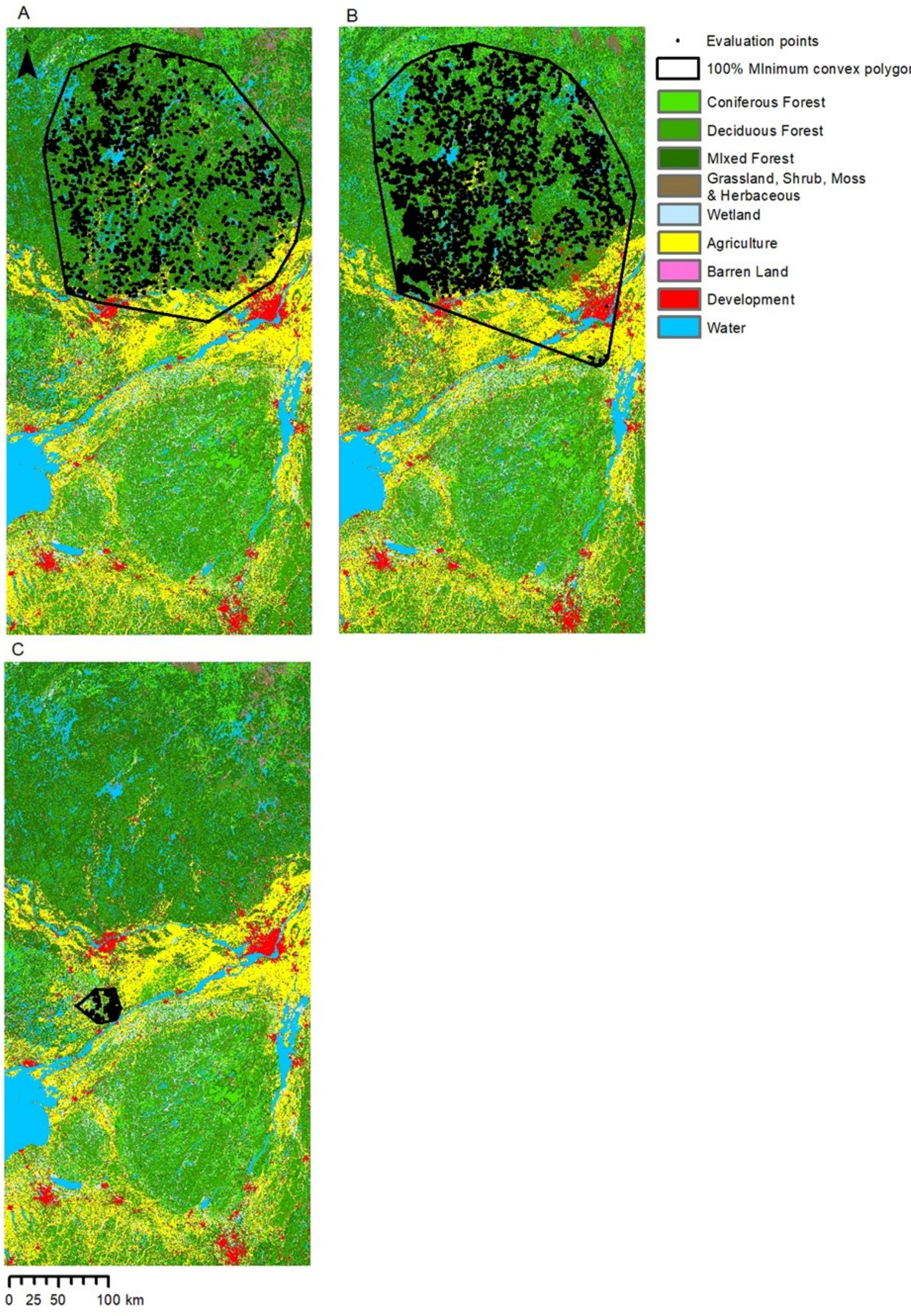

Figure S3.1 Evaluation points within 100% minimum convex polygons relative to habitat suitability. A) Black bear 2000, B) Black bear 2015, C) Fisher 2000

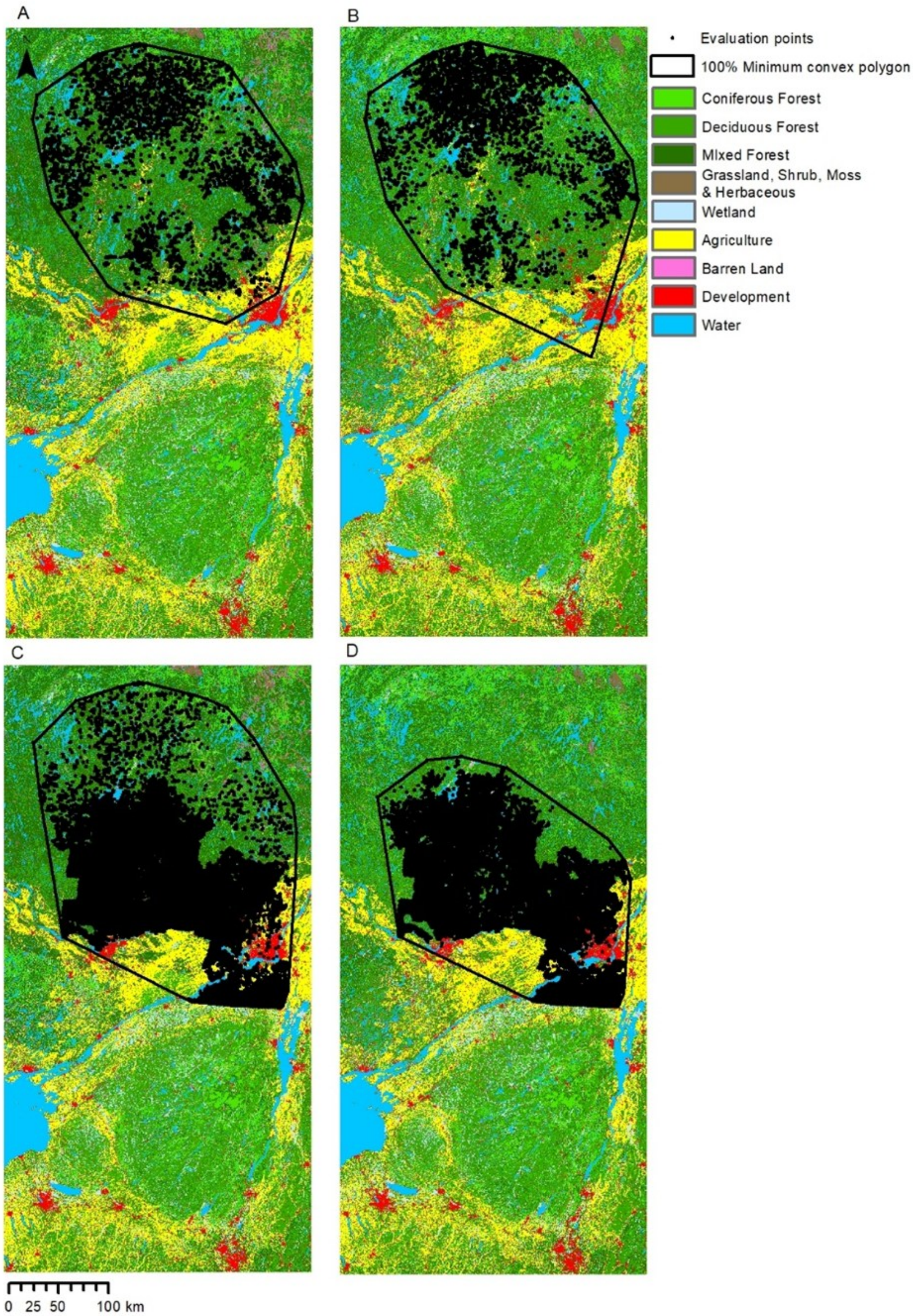

Figure S3.2 Evaluation points within 100% minimum convex polygons relative to habitat suitability. A) Moose 2000, B) Moose 2015, C) White-tailed deer 2000, D) White-tailed deer 2015.

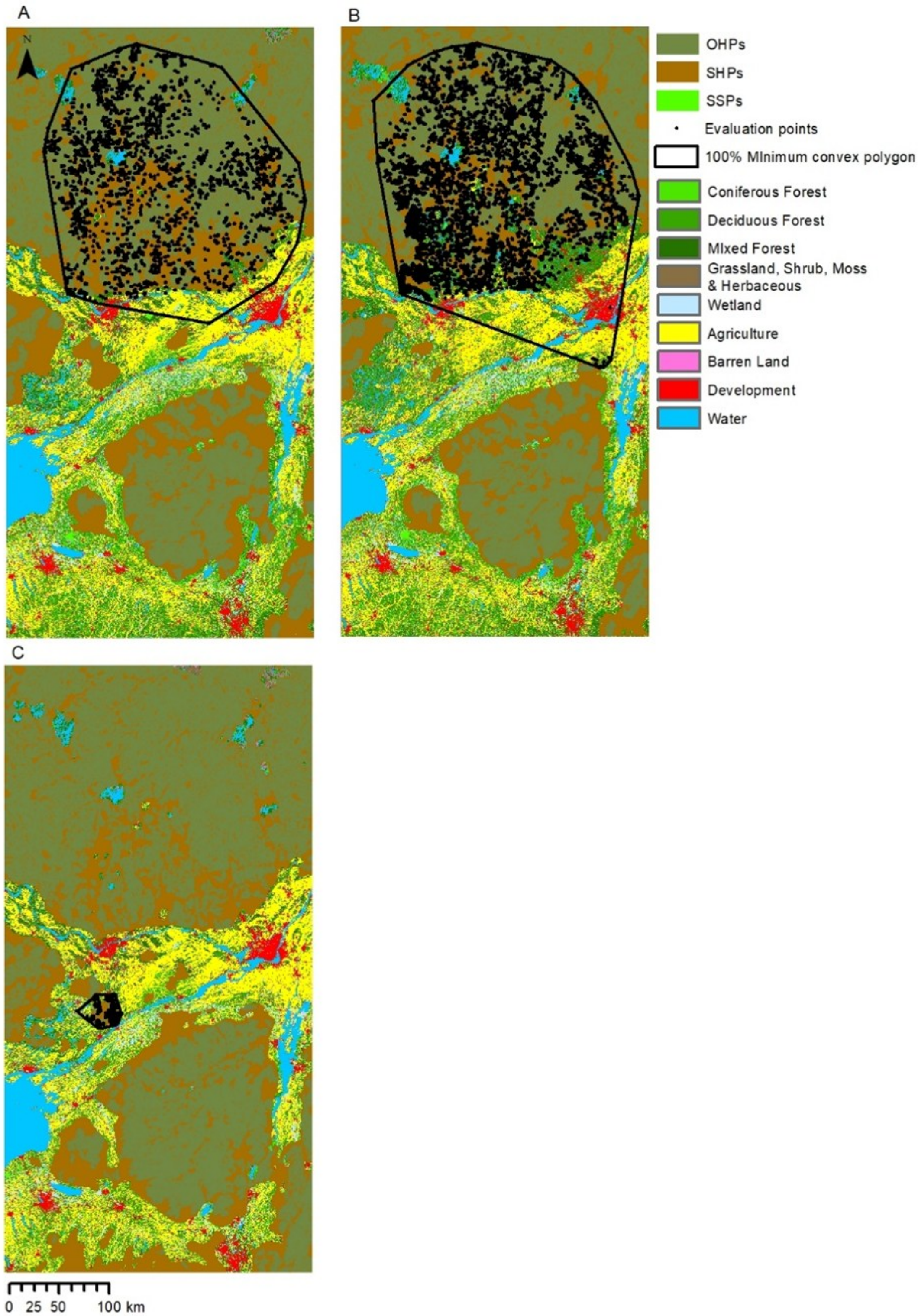

Figure S3.3 Evaluation points within 100% minimum convex polygons relative to habitat patches. A) Black bear 2000, B) Black bear 2015, C) Fisher 2000.

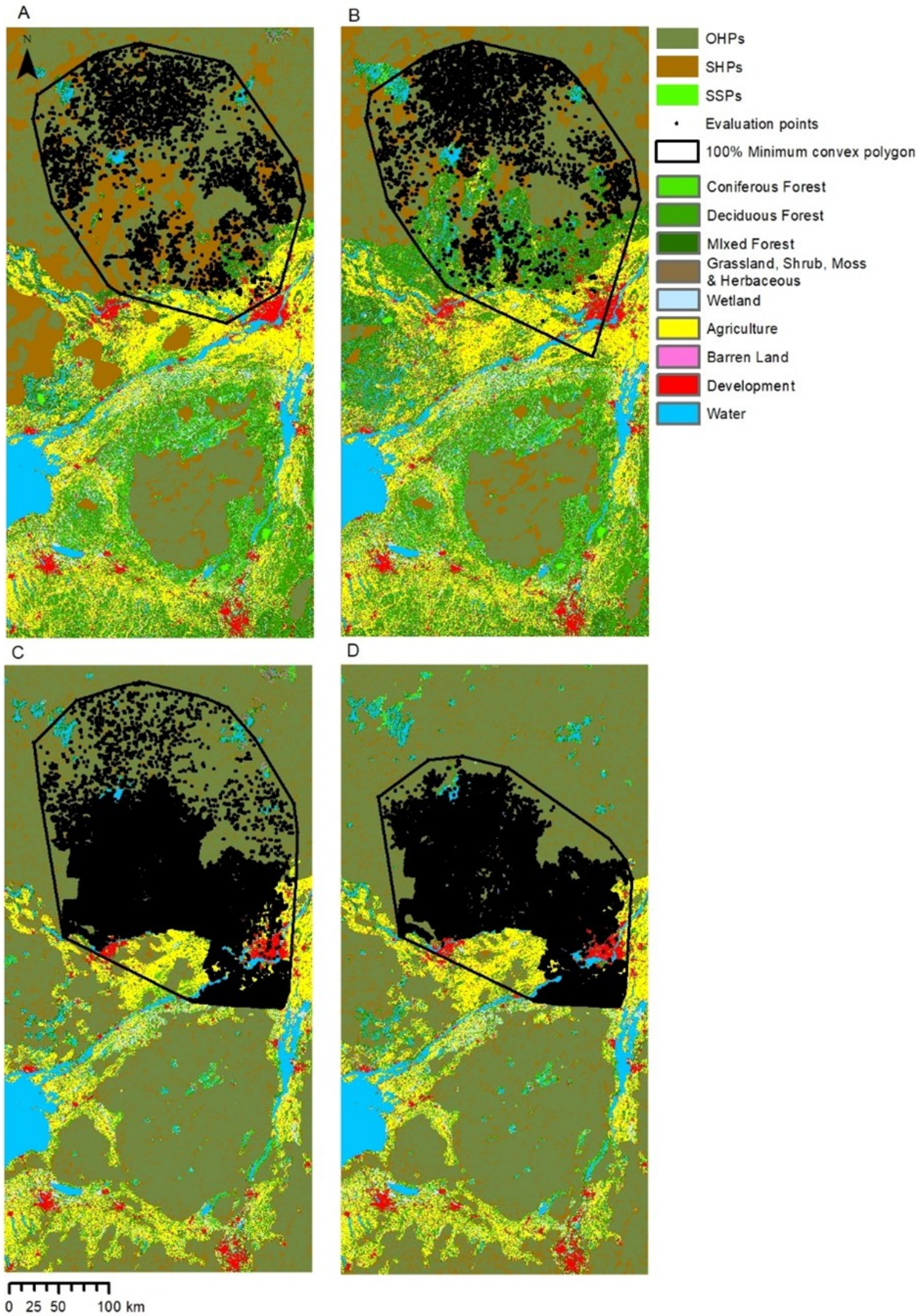

Figure S3.4 Evaluation points within 100% minimum convex polygons relative to habitat patches. A) Moose 2000, B) Moose 2015, C) White-tailed deer 2000, D) White-tailed deer 2015.

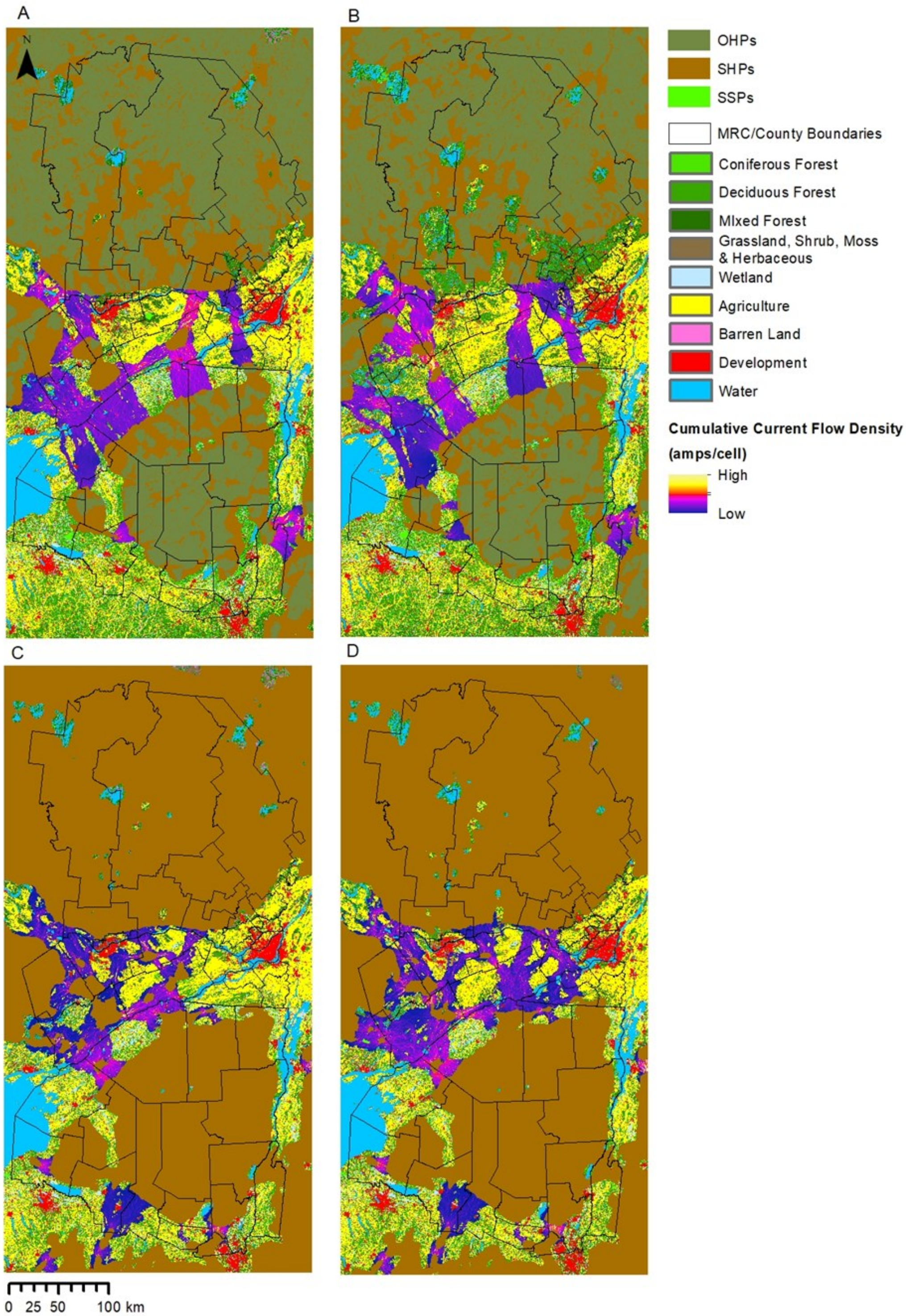

Figure S3.5 Changes in cumulative current flow density (i.e., all-to-one mode in Circuitscape) within LCCs between 2000 and 2015. A) Black bear habitat 2000, B) Black bear habitat 2015, C) Fisher habitat 2000, D) Fisher habitat 2015.

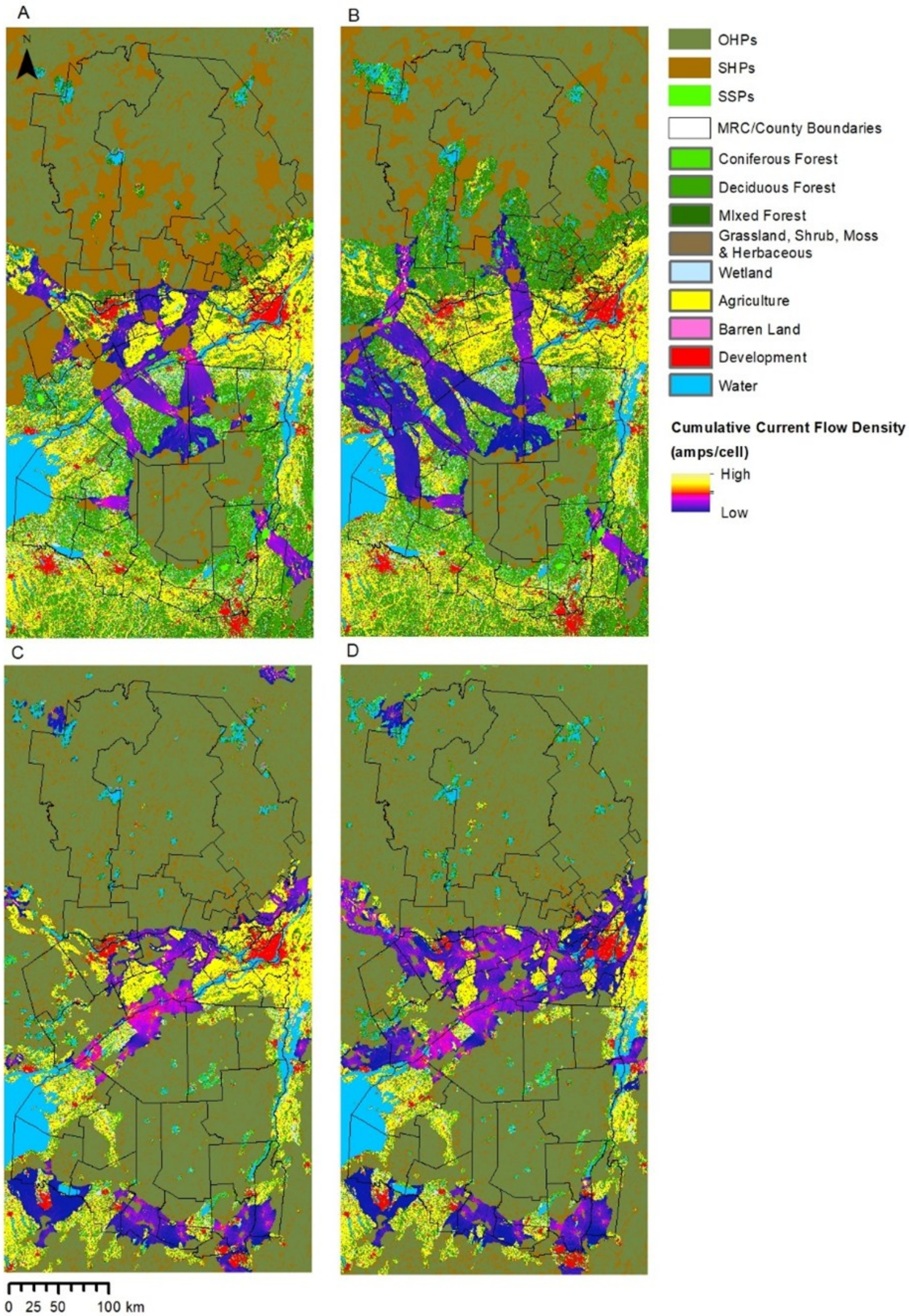

Figure S3.6 Changes in cumulative current flow density within LCCs (i.e., all-to-one mode in Circuitscape) between 2000 and 2015. A) Moose habitat 2000, B) Moose habitat 2015, C) White-tailed deer habitat 2000, D) White-tailed deer habitat 2015.

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Connecting Statement 2.

Chapter 3 showed that between 2000 and 2015, anthropogenic land transformation severely impacted species-specific habitat amount, fragmentation, and connectivity in the A2L transboundary wildlife linkage. Suitable and optimal habitat patch area declined, fragmentation increased, and inter-patch connectivity decreased for each species. Moose and black bear habitat patches experienced the greatest habitat loss, fragmentation, and decline in inter-patch connectivity, with the majority of habitat patch area loss and fragmentation occurred in the southern Québec and Ontario portions.

In chapter 4, I continue this exploration of the changing landscape by examining the impacts of anthropogenic land transformation on the potential habitat and functional connectivity of the eastern wolf (*Canis lupus lycaon*), an at-risk species in the A2L. The eastern wolf once inhabited most of the mixed and deciduous forests of eastern United States and southeastern Canada (Wilson et al. 2000; Kyle et al. 2006; Rutledge et al. 2010). However, from the onset of European colonization, direct human persecution, fur harvesting, and habitat loss has extirpated the eastern wolf from most of its historical range (Nowak, 2002). Today, the eastern wolf inhabits only a few, primarily protected, areas in Ontario and Québec (Figure 1 in ECCC, 2021).

There has been substantial debate concerning the genetic origins of the eastern wolf in North America. According to the 3-species hypothesis (Wilson et al., 2000), there are three large canid species present in North America: the gray wolf (*Canis lupus*), the eastern wolf, and the coyote (*Canis latrans*). Hybridization between the gray wolf and the eastern wolf has produced the Great Lakes wolf (*Canis lupus x lycaon*), and hybridization between the eastern wolf and coyote has produced the eastern coyote or "coywolf" (*Canis lycaon x latrans*) (Wilson et al., 2000; Wheeldon et al., 2010). The 2-species hypothesis proposes that there are just two canid species present in North America: the gray wolf and the coyote, and that all other canids are hybrids of these two species (Wayne & Jenks, 1991). However, recent genetic research (Wilson et al. 2000; Kyle et al. 2006; Rutledge et al. 2012, 2015) strongly indicates that the eastern wolf is a distinct species in accordance with the 3-species hypothesis (ECCC, 2021).

In 2015, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) recommended that the eastern wolf be recognized as a unique species and re-classified their federal conservation status to "Threatened" due to low population numbers and their restricted geographic distribution (COSEWIC, 2015; Benson et al., 2017). In 2016, the eastern wolf was renamed the "Algonquin Wolf" (*Canis sp.*) by the Committee on the Status of Species at Risk in Ontario (COSSARO) and their provincial conservation status was re-classified to "Threatened" under Ontario's Endangered Species Act (BELW 2000 Consulting, 2018). In Québec, only the grey wolf is officially recognized by the provincial government and this species is not listed under the Act Respecting Threatened or Vulnerable species (C.Q.L.R., c. E-12.01) (ECCC, 2021).

In this chapter I examine the changes in potential habitat amount, habitat fragmentation, and functional connectivity for wolves in general (i.e. eastern and gray wolves), and I determine priority areas for conservation and ecological restoration within the A2L transboundary wildlife linkage.

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4. Land Conversion and Lack of Protection Significantly Reduce Suitable Wolf Habitat Amount and Functional Connectivity in the Adirondack-to-Laurentians (A2L) Transboundary Wildlife Linkage

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Abstract

The Adirondack-to-Laurentians (A2L) transboundary wildlife linkage connects wilderness areas in the northeastern United States with southeastern Canada. However, land conversion is putting wolf habitat amount and functional connectivity at risk. With the exception of protected areas, hunting and trapping of wolves and coyotes are permitted within the Québec and Ontario portions; while hunting and trapping coyotes are permitted within the New York portion where wolves have been extirpated. Thus, the fear of humans strongly influences wolf habitat selection in this region. We assessed the impact of land conversion on wolf habitat amount, habitat fragmentation, and functional connectivity in the A2L between 2000 and 2015 and identified potential suitable habitat patches and corridors for protection. Suitable habitat patch area decreased by 18,245 km^2 (27%), with losses of 28% in the Québec portion, 95% in the Ontario portion, but only 0.3% in the New York portion. Habitat fragmentation, as measured by the effective mesh size, substantially increased in the Québec and Ontario portions, but only slightly in the New York portion. Functional connectivity significantly decreased, with mean distances and the cost of traveling these distances more than doubling. We propose nine recommendations centered on extensive habitat restoration and protected area expansion in the Québec and Ontario portions of the study area. Wolf recovery within the A2L will require collaborative and

coordinated transboundary conservation and the protection of suitable habitat patches and corridors, or the legal protection of both wolves and coyotes within the suitable habitat patches and corridors, to ensure that wolves are not harvested as they disperse and colonize new locations.

Keywords: Eastern wolf, Gray wolf, Habitat loss, Effective mesh size, Linkage mapper, Circuitscape

Introduction

The majority of large terrestrial carnivores have experienced substantial population declines and geographic range contractions over the past two centuries (Wolf and Ripple 2017). Large carnivores face a wide variety of anthropogenic threats including persecution, hunting and trapping, habitat loss and degradation, and depletion of prey base (Crooks et al. 2011; Ripple et al. 2014; Wolf and Ripple 2016). Consequently, large carnivore populations are small, restricted to isolated habitat fragments, and predominantly occur only within protected areas (Woodroffe and Ginsberg 1998).

The wolf, once ranging across most of North America, Europe, and Asia, exhibited the largest geographical range of any terrestrial mammal other than humans (Mech and Boitani 2003). However, persecution, hunting and trapping, and habitat loss reduced their range considerably (Young and Goldman 1944, Mech 1995). In North America, the wolf was extirpated from most of southern Canada, Mexico, and the 48 contiguous United States, except for northern Minnesota, by 1970 (Mech and Boitani 2003). Today, large wolf populations (i.e., greater than 5000 individuals) are only found in Canada and Alaska (Musiani and Paquets 2004).

However, in Europe and the United States, wolves are re-colonizing their former range in regions where they and their habitat have been granted legal protection (Chapron et al. 2014; Smith et al. 2016). This re-colonization of parts of their historical range could potentially restore the important regulatory role wolves and other large carnivores play within food webs and ecosystems (Estes et al. 2011; Ripple et al. 2014).

Wolves regulate ecosystem structure and function through both density-mediated and behaviorally mediated effects on prey and meso-predator populations and their associated trophic cascades (Estes et al. 2011; Ripple et al. 2014). Wolves typically occupy areas with high prey density, i.e., moose (*Alces alces*), white-tailed deer (*Odocoileus virginianus*), and beaver (*Castor canadensis*), and low human-caused mortality (Fuller et al. 2003; Benson et al. 2024). In addition, wolves typically select forest and wetland areas for denning and rendezvous site locations (Benson et al. 2015; Sazatornil et al. 2016). In general, wolves spatially avoid humans (Carricondo-Sanchez et al. 2020). However, this behavior is modulated by the history of coexistence and persecution (i.e., stronger avoidance behavior in areas where they are harvested, such as North America, weaker avoidance behavior where they are protected, such as Europe) (Sazatornil et al. 2016). Even in low human-modified landscapes in North America, wolves typically avoid areas of human activity (Bubnicki et al. 2019). For example, Malcolm et al. (2020) showed that wolves avoided human-modified areas (i.e., housing structures, campsites, and park facilities), suggesting that wolves perceived them as a risk. This fear of humans resembles the “landscape of fear” (Laundré et al. 2001; 2010) that wolves impose on their prey species (Gaynor et al. 2019). Humans as “super-predators” (Darimont et al. 2015) directly influence food-chain dynamics (i.e., predators, meso-predators, and prey populations) by affecting their densities (i.e., hunting and trapping), their behavior (by creating a landscape of fear), and landscape structure (loss of habitat and connectivity) (Kuijper et al. 2016). These influences limit wolf population sizes and reduce their ecological effectiveness in unprotected landscapes compared to protected or remote wilderness areas (Suraci et al. 2019; Kuijper et al. 2019; 2024).

The Adirondack-to-Laurentians (A2L) transboundary wildlife linkage connects wilderness areas in the northeastern United States with southeastern Canada and includes portions of Québec, Ontario, and New York (Fig. 4.1). This region contains habitats of high ecological integrity and biodiversity; however, anthropogenic land transformation is putting habitat amount and transboundary connectivity at risk (Cole et al. 2023a, 2023b). While the coyote *(Canis latrans*) is ubiquitous throughout the A2L region, gray wolves (*Canis lupus*), and eastern wolves (*Canis lupus lycaon*) only occur within the Québec portion of the study area (Mainguy et al. 2017).

In 2015, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) recommended that the eastern wolf be recognized as a unique species *(Canus lycaon*) and its federal conservation status be re-classified to "Threatened," due to its low abundance and restricted geographic distribution (COSEWIC 2015; Benson et al. 2017). However, as of January 2024, the official scientific name of the eastern wolf remains a subspecies of the gray wolf "*Canis lupus lycaon*" and its legal conservation status remains "Species of Special Concern" under Canada's Species at Risk Act 2002 (ECCC 2021). In 2016, the eastern wolf was renamed the "Algonquin Wolf" (Canis sp.) by the Committee on the Status of Species at Risk in

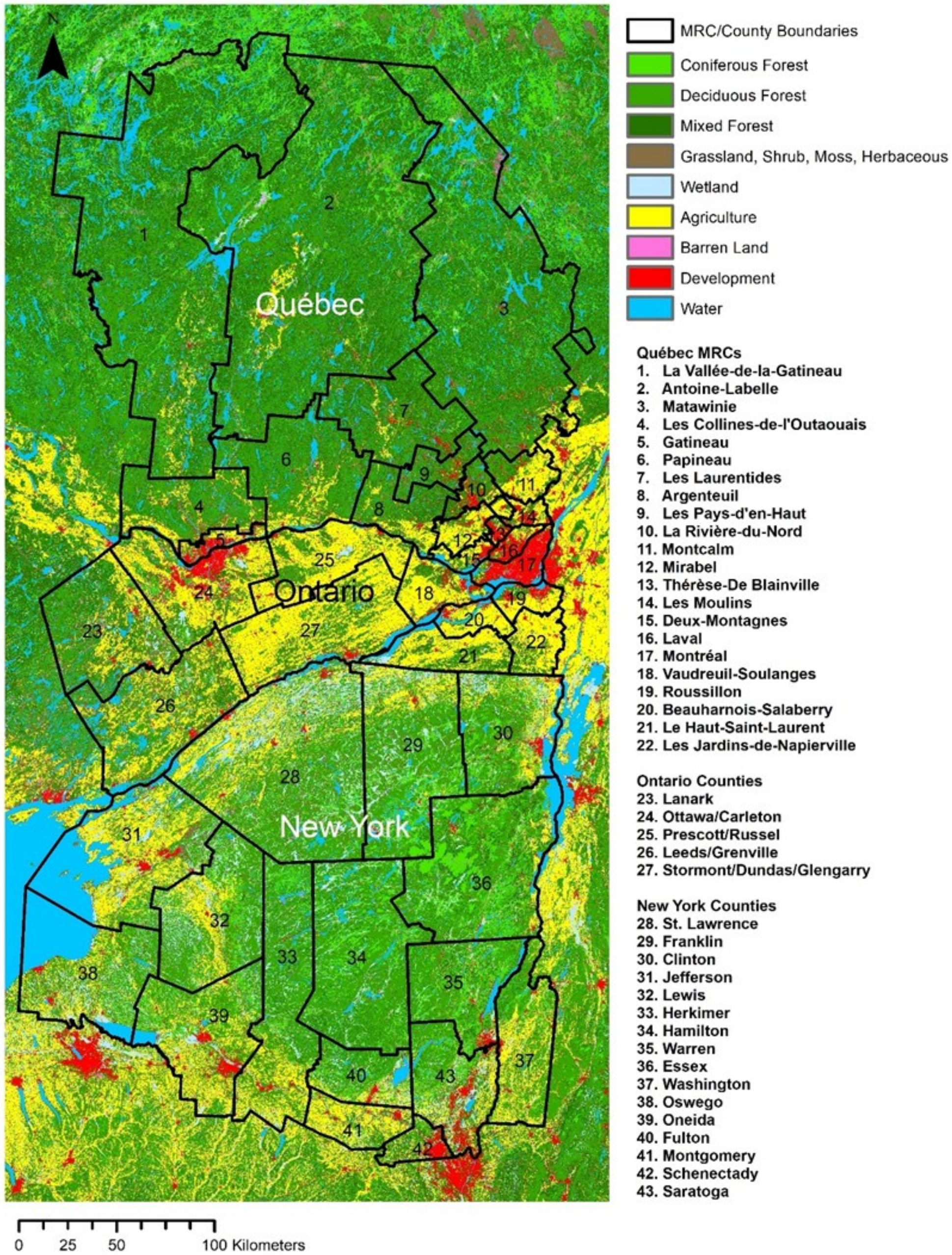

Figure 4.1 Land-cover map of the Adirondack-to-Laurentians (A2L) study area overlaid with municipalité régionale de comté (MRC)/county boundaries. MRC/county names are numbered and correspond to the numbers on the map.

Ontario (COSSARO) and their provincial conservation status was re-classified to “Threatened” under Ontario’s Endangered Species Act 2007 (BELW 2000 Consulting, 2018). In 2018, the Ontario government released a recovery strategy for the Algonquin Wolf in Ontario (BELW 2000 Consulting, 2018). However, its recovery strategy is focused on the area in and around Algonquin Provincial Park and does not include the entire province, nor the portion within the A2L. In 2021, the Canadian government released a management plan for the eastern wolf in Canada (ECCC 2021). The plan includes two primary conservation objectives: (1) achieve and maintain viable eastern wolf populations within the species’ current range in Canada, and (2) achieve and maintain connectivity between occupied sites as well as potential suitable habitat sites to facilitate dispersal and maintain genetic diversity (ECCC 2021). Potential suitable habitat and dispersal routes for the eastern wolf have not been re-examined since circa 2000 (Harrison and Chapin 1998; Mladenof and Sickley 1998; Paquet et al. 1999; Carroll 2003). Thus, there is an urgent need for updated information to achieve these objectives.

With the exception of protected areas, hunting and trapping of gray wolves and coyotes are permitted within the Québec portion of the study area between October and March each year (Québec 2023), all-year-round in the Ontario portion (Ontario 2023a), and although gray wolves have been extirpated from New York State since 1893, they are still protected under both the federal Endangered Species Act 1973 and New York’s Endangered and Threatened Species Regulations (NYS-DEC 2023a), while hunting and trapping of coyotes are permitted between October and March (NYS-DEC 2023b). In Ontario, eastern wolves are protected from hunting and trapping under Ontario’s Endangered Species Act 2007, while in Québec and New York they are simply recognized as gray wolves. However, despite this “protected” status, their similar size and appearance to gray wolves and coyotes, as well as the indiscriminate nature of trapping, leave them extremely vulnerable to death by mistaken identity when they venture outside of protected areas (Benson et al. 2014).

The Adirondack region has been identified as a location with suitable habitat for wolf re-colonization or re-introduction (Harrison and Chapin 1998; Paquet et al. 1999; Carroll 2003; van den Bosch et al. 2022). However, both natural re-colonization and re-introduction would require numerous long-distance dispersal events from existing populations in Ontario and Québec to establish new territories and facilitate gene flow and maintain genetic diversity (Harrison and Chapin 1998). In regions where wolves are protected, they are highly capable of long-distance dispersal through human-modified landscapes (Chapron et al. 2014; Kuijper et al. 2016). However, in regions where wolves are unprotected, leaving safe protected areas significantly increases their mortality risk (i.e., hunting, trapping, collisions with vehicles, and conflicts with humans), especially in highly fragmented landscapes (Crooks et al. 2011). Thus, the potential for wolves to successfully disperse into the Adirondack region or expand their range into unprotected suitable habitats within the A2L is unlikely without the implementation of legislation to protect wolves outside of protected areas (Rutledge et al. 2017; Benson et al. 2024). Identifying and protecting large areas of suitable habitat with sufficient prey density, and ecological corridors that interconnect them, may provide the greatest potential to maximize the ecological role that wolves play in ecosystem structure and function, while expanding the range and number of wolves in the region.

In this study, we created wolf habitat and resistance models to identify potential suitable habitat patches (HPs), optimal HPs, and stepping stone patches. Hunting and trapping strongly influences habitat selection. We then applied Linkage Mapper and Circuitscape to the habitat network to map least-cost corridors and pinch points important for functional connectivity. The aim was to assess the impact of land conversion on wolf habitat amount, habitat fragmentation, and functional connectivity in the A2L transboundary wildlife linkage between 2000 and 2015, and identify potential suitable habitat patches and corridors for protection.

Methods

Study Area

The A2L study area is approximately 127,408 km^2 in size and is made up of 22 municipalités régionales de comté (MRCs) in Québec (58,867 km^2; 46%), five counties in Ontario (15,445 km^2; 12%), and sixteen counties in New York (53,096 km^2; 42%) (Fig. 4.1). The A2L is located in the northern forest and eastern temperate forest eco-regions and is home to 440 vertebrate species and 1600 vascular plant species (Tardif et al. 2005; CEC 2023). The geology of the A2L is comprised of Canadian Shield to the north, St. Lawrence Platform in the centre, and Precambrian to the south (Tardif et al. 2005), with the highest peak being Mount-Marcy in New York (1629 m). In 2016, the region was home to over 6.8 million people (54 per km^2) (Statistics Canada 2023; US Census Bureau 2023).

Suitable habitat and resistance models

Land cover and road network maps were re-classified into ten common land cover classes and three common road network classes unifying the classification scheme across all input maps (Table S4.1; S4.2). In unprotected landscapes where mortality risk is high due to hunting and trapping, wolves can exhibit significant avoidance behavior of up to 1 km from human activity (including human presence, development, agriculture, and roads; Singleton 1995; Paquet et al. 1996). To incorporate this landscape of fear, we generated buffers of 0–500 and 500–1000 m around roads and development using the "Euclidean Distance" function in ArcGIS10.7 to represent the median and maximum distances from roads and development at which avoidance behaviors are displayed. This created four additional environmental variable layers that incorporated wolf avoidance behavior: (1) distance from development; (2) distance from primary roads; (3) distance from secondary roads; and (4) distance from tertiary roads (Figure S4.3). Because prey density is adequate throughout the forest and wetland regions of the A2L, this model assumes that suitable wolf habitat is concentrated in large forest and wetland areas with sufficient prey density to accommodate at least one wolf pack.

We computed a habitat suitability index by assigning relative values to the land cover maps using a combination of previously published values, literature review, and expert opinion (Table S3). Previously published values were rescaled so that the values ranged between 0 and 1 using the following equation:

$$F(x) = (x - \min) / (\max - \min),$$

where x is the assigned relative suitability value for a 30 m grid cell, and min and max are the minimum and maximum suitability values of the habitat suitability surface, respectively (Keeley et al. 2016). Values near 1 represent the relative highest habitat suitability in the area, and values near 0 represent the relative lowest habitat suitability (Keeley et al. 2016). We created one aggregate suitable habitat map by overlaying all six layers in ArcGIS10.7, using Gnarly Landscape Utilities: Resistance and Habitat Calculator toolset (McRae et al. 2013), and retaining the minimum suitability value for each 30 m × 30 m cell across all input layers. Thus, each spatial layer received equal weighting (i.e., effect size), and the same relative importance to wolf habitat selection. This was because (1) all layers were derivative of the land cover layer, (2) all values, across all layers, where relative to ideal wolf habitat on the land cover layer, and (3) all values were obtained from previous studies, literature review, and expert option, where equal weighing was implied (Singleton 2002; Carroll et al. 2012; WWHCWG 2010; 2012).

We derived resistance values for each of the six raster layers by calculating the linear inverse of our suitable habitat values (Table S3; Koen et al. 2012, Keeley et al. 2016) using the following equation:

$F(x) = 1 - (x/100)$,

where x is the habitat suitability value. A single aggregate resistance surface was created by overlaying all six layers in ArcGIS10.7, using Gnarly Landscape Utilities: Resistance and Habitat Calculator toolset (McRae et al. 2013), and retaining the maximum resistance value for each 30 m × 30 m cell across all six input layers (McRae et al. 2013). We added a value of one to each cell, such that habitats with a relatively low movement cost had a value of 1, and habitats with a high cost had values up to a maximum of 101. Bowman et al. (2020) found that landscape connectivity models tend to be insensitive to absolute cost values, provided that the rank order is correct.

Identifying suitable and optimal habitat patches

To identify potential habitat patches, we used the aggregated suitable habitat and resistance layers and the software Gnarly Landscape Utilities: Core Mapper toolset (Shirk and McRae 2013) in ArcGIS10.7. Suitable habitat patches were identified as patches with an average habitat value ≥ 0.6 (WWHCWG (2010; 2012), within a circular moving window with a radius of 9788.3 m (i.e., the radius of the average maximum home range size of 301 km^2; Tables S4.4 & S4.5). This ensured that habitat patches contained no more than 50% unsuitable habitat types, i.e., agriculture, development, water, and forest and wetland areas less than 500 m from development and primary roads. This step generated a surface layer representing where the largest concentrations of suitable habitat occurred (WWHCWG 2010; 2012). To correct for the variability in minimum home range sizes within the literature we multiplied the average minimum home range size of 93.5 km^2 (Table S4.4; S4.5) by 0.75 to compensate for the fact that wolves can occur within smaller home range sizes when resource patches are of high quality (Loveless 2010). This reduced value of 70.1 km^2 was used as the minimum habitat patch cutoff size to ensure smaller potentially suitable habitat patches were not overlooked. Patches that fell below the minimum habitat patch cutoff size were removed (WWHCWG 2010; 2012). This is in agreement with Fuller et al. (2003), that state that even at the highest prey densities (i.e., 15 deer or 3 moose/km^2), an individual pack of four wolves would still require a territory of at least 75 km^2 to meet its nutritional requirements. We then expanded habitat patches outwards up to a total cost-weighted distance of 5455.5 m (i.e., the radius of the average minimum home range size of 93.5 km^2; Tables S4.4 & S4.5) to potentially link proximate patches into larger aggregates, simulating intra-patch connectivity (WWHCWG 2010; Spanowicz and Jaeger 2019). Habitat patches still separated at this point require movements that exceed twice the cost-weighted distance of the mean minimum home range radius and were considered dispersal distances (i.e., inter-patch connectivity). We identified optimal habitat patches by performing the same steps as above; however, we did not expand the patches, and we removed all raster cells within the habitat patches with values ≤ 0.4 (consistent with Cole et al. (2023b); Table S4.3) to exclude unsuitable habitat types, i.e., agriculture, development, water, and forest and wetland areas less than 500 m from development and primary roads, leaving habitat patches devoid of anthropogenic transformations. Stepping stone patches were identified as suitable HPs that were smaller than the 70.1 km^2 minimum habitat patch cutoff size, but still large enough to serve as a refuge area during dispersal (≥ 10 km^2; Table S4.5). We chose not to use a traditional species distribution model such as MaxEnt for three reasons: (1) there were no occurrence data available for the majority of the study area due to the extirpation of wolves from the Ontario and New York portions of the A2L. However, populations still inhabit the Québec portion of the study area and we used GPS location data from one of these populations to validate our suitable habitat patch models; (2) one of our main goals was to quantify the degree of habitat fragmentation within the study area; thus, we needed a model that could delineate potential suitable habitat patches (i.e., allowing for the incorporation of minimum home range size, minimum habitat patch size, and intra- and inter-patch connectivity into the model), and not just identify habitat suitability; and (3) we wanted to integrate avoidance behavior distances into the model. Therefore, by applying the Core Mapper toolset (Shirk and McRae 2013), we were able to incorporate all of these elements into the model and identify suitable and optimal habitat patches (also called habitat concentration areas (HCAs); WWHCWG 2010; 2012).

Validation of suitable habitat and habitat patch models

To validate our suitable habitat and habitat patch models, we used previously published telemetry data collected between 2015 and 2017 (Malcolm et al. 2020) from a canid population in the Québec portion of the study area (i.e., Parc National du Mont-Tremblant and adjacent areas) that contained gray wolves, eastern wolves, and coyotes. The dataset consisted of 24,550 GPS locations, hereafter referred to as “validation points,” obtained from five adult males and five adult females fitted with telemetry collars that were programmed to acquire location coordinates every 3 h for a period of 12 months (Malcolm et al. 2020). Because of changes in movement ability and behavior in the winter months (i.e., ability to cross frozen lakes; nomadic period), only GPS locations acquired between April 1st and November 30th were used for validation. We were unable to obtain wolf validation points for circa 2000; therefore, suitable habitat and habitat patches were only validated for 2015. Since the validation points only covered a subsection of the study area, we delineated this subsection by creating a 100% minimum convex polygon (MCP) around all the validation points (Koen et al. 2007; Brodeur et al. 2008) using the "Convex Hull" function in ArcGIS10.7 (Figure S1). We validated the performance of the suitable habitat model (i.e., how well the model predicted wolf suitable habitat) using three validation metrics. First, we used the absolute validation index (*AVI*), calculated as the proportion of validation points that were located on raster cells with a habitat value ≥ 0.6 within the MCP (Hirzel and Arlettaz 2003; Hirzel et al. 2006; Guisan et al. 2017). Values for the *AVI* range between 0 and 1. Second, we used the contrast validation index (*CVI*), calculated as the *AVI* minus the proportion of raster cells with a habitat value of ≥ 0.6 within the MCP (Hirzel et al. 2004; Hirzel et al. 2006; Guisan et al. 2017). Values for the *CVI* range between -0.5 and 0.5. Finally, we used the *Boyce Index* (Boyce et al. 2002; Hirzel et al. 2006; Guisan et al. 2017), using two calculated frequencies for each of the 6 habitat classes (i.e., 1, 0.8, 0.6, 0.4, 0.2, 0): (1) the proportion of observed validation points found in each habitat class within the MCP (*P*) and (2) the expected proportion of validation points found in each habitat class within the MCP (*E*) (Boyce et al. 2002; Hirzel et al. 2006). We then calculated the *P/E* ratio for each class. If the model predicted suitable habitat well, then a low habitat class should contain fewer validation points than expected by chance (i.e., a *P/E* ratio < 1), whereas a high habitat class should contain more validation points than expected by chance (i.e., a *P/E* ratio > 1; Hirzel et al. 2006; Guisan et al. 2017). The *Boyce Index* was then calculated using Spearman's rank correlation coefficient between the habitat value and the *P/E* ratio (Boyce et al. 2002; Hirzel et al. 2006). *Boyce Index* values range between −1 (incorrect model) and 1 (a highly consistent model); values close to zero indicate no difference from chance (Hirzel et al. 2006; Guisan et al. 2017). To measure the performance of the habitat patch model, we applied variations of the *AVI* and *CVI* metrics. We used the AVI_{patch} to calculate the proportion of validation points that were located within suitable HPs and optimal HPs, calculated as the number of validation points in HPs within the MCP divided by the number of validation points within the MCP. Values for the AVI_{patch} ranged between 0 (weak performance) and 1 (strong performance). Next, we used the CVI_{patch}, calculated as the AVI_{patch} – the area of HPs within the MCP (km^2) divided by the area of the MCP (km^2). Values for the CVI_{patch} ranged between − 0.5 (weak performance) and 0.5 (strong performance).

Habitat amount and fragmentation

We measured the area of suitable HPs and optimal HPs in 2000 and 2015 using ArcGIS10.7. We calculated proportion by dividing the HP area by the total area of the reporting unit (i.e., study area, provincial/state portion). To quantify fragmentation, we used the effective mesh size, which is based on the average probability that any two randomly chosen points in the study area are connected, i.e., not separated by some barrier (Jaeger, 2000). Because the boundary of a reporting unit can influence the value of the effective mesh size, two variations of the effective mesh size were used. The "cutting out" procedure ($m_{\text{eff\_CUT}}$) was used to measure fragmentation strictly within the boundaries of the reporting units, while the "cross-boundary connections" procedure ($m_{\text{eff\_CBC}}$) was used to include patches that cross boundaries into adjacent reporting units (Moser et al., 2007). All measurements were performed using the effective mesh size tool from the ZonalMetrics ArcGIS toolbox (Wetzel, 2019). We measured the road density of each suitable HP by dividing the total length of roads within a patch by the area of the patch.

Functional connectivity

We mapped functional connectivity between the suitable HPs using the Linkage Pathways tool of the Linkage Mapper ArcGIS Toolbox (McRae & Kavanagh, 2011). We calculated adjacency using both cost-weighted and Euclidean distances, omitted corridors that intersected other HPs, put no limit on the number of linkages originating from each HP, and truncated the width of least-cost corridors to 200 cost-weighted km. It is recommended that least-cost corridors should be at least 2 km wide (i.e., accommodate a wide-variety of species, reduce edge-effects, allow for recreational use; Beier, 2018). Thus, we used a cut-off width of 200 cost-weighted km to ensure that even when corridors navigated regions with the highest resistance values (101), corridors would still maintain a width of at least 2 km. Prior to running Linkage Pathways, the resistance layers were coarsened by three times to reduce computing time and memory use, which resulted in a final resistance layer resolution of 90 m.

To identify pinch-points within the least-cost corridors, we used the Pinch-Point Mapper tool of the Linkage Mapper ArcGIS Toolbox (McRae, 2012). Pinch-Point Mapper uses Circuitscape (McRae & Shah, 2011) to simulate a path of electric current through the least-cost corridors. We ran Circuitscape in both “pairwise” and “all to one” mode to identify pinch-points important for connectivity between pairs of suitable HPs and for maintaining connectivity for the entire network of suitable HPs (Dutta et al., 2016).

To quantify changes in connectivity, we compared Euclidean distance, cost-weighted distance, least-cost path length, and effective resistance between suitable HPs in 2000 and 2015.

We assumed that if there was a decline in functional connectivity then these distances would have increased. We compared the distances between time points with a two-sided Welch's *t*-test to account for unequal variances. We measured the effect size of the differences in distances with Cohen's effect size (d = 0.2 represents a small effect size, d = 0.5 represents a medium effect size, and d = 0.8 represents a large effect size; Cohen, 1988).

Proportion of habitat patches and least-cost corridors under protection

To determine the percentage of suitable HPs, optimal HPs, and least-cost corridors under protection, we obtained maps of government protected areas and private protected areas secured by Nature Conservancy of Canada/The Nature Conservancy (Table S4.6). We measured the proportion of suitable HP area, optimal HP area, and least-cost corridor area currently under protection.

Results

Validation of the suitable habitat and habitat patch models

The suitable habitat model performed well at predicting non-winter suitable wolf habitat within the local landscape as measured by the absolute validation index (*AVI*) with a value of 0.8 (Hirzel et al., 2006), whereas, the contrast validation index (*CVI*) gave a value of 0.07 indicating that although 80% of the validation points were located on suitable habitat, the amount of available suitable habitat was only slightly less (73%). The *Boyce Index* value of 0.89, however, suggests a stronger performance as it signifies then low habitat classes contained fewer validation points than expected by chance and that high habitat classes contained more validation points than expected by chance (Hirzel et al., 2006; Guisan et al., 2017). The habitat patch models also performed well at predicting suitable and optimal HPs within the local landscape with AVI_{patch} values of 0.93 and 0.71, respectively (Hirzel et al., 2006), whereas CVI_{patch} values were 0.09 for suitable HPs and 0.07 for optimal HPs. Consequently, although 93% and 71% of the validation points were located on suitable and optimal HPs respectively, overall HP area was only slightly less (suitable HP area 84% & optimal HP area 64%).

Habitat amount and fragmentation

Suitable HP area decreased by 18,245 km^2 (27%), and optimal HP area decreased by 7082 km^2 (17%) between 2000 and 2015 for the whole study area (Table 4.1, Figure 4.2). The majority of these losses took place in the Québec portion of the study area where suitable HP area was

Location	SHP area in 2000 (km^2)	Proportion of SHP area in reporting unit in 2000 (%)	SHP area in 2015 (km^2)	Proportion of SHP area in reporting unit in 2015 (%)	SHP area 2015-2000 (km^2)	Percent change 2015 - 2000 (%)
Study Area	67878	53	49633	39	-18245	**-27**
Québec Portion	48047	82	34679	59	-13369	**-28**
Ontario Portion	5098	33	269	2	-4830	**-95**
New York Portion	14732	28	14686	28	-46	-0.3
	OHP area in 2000 (km^2)	**Proportion of OHP area in reporting unit in 2000 (%)**	**OHP area in 2015 (km^2)**	**Proportion of OHP area in reporting unit in 2015 (%)**	**OHP area 2015-2000 (km^2)**	**Percent change 2015 - 2015 (%)**
Study Area	42516	33	35435	28	-7082	-17
Québec Portion	30996	53	24682	42	-6314	-20
Ontario Portion	439	3	40	0.3	-399	**-91**
New York Portion	11081	21	10712	20	-369	-3

Table 4.1 Land-cover map of the Adirondack-to-Laurentians (A2L) study area overlaid with municipalité régionale de comté (MRC)/county boundaries. MRC/county names are numbered and correspond to the numbers on the map.

reduced by 13,369 km^2 (28%) and optimal HP area was reduced by 6314 km^2 (20%) (Table 4.1, Figure 4.2). The Ontario portion showed the lowest amount of both suitable and optimal HP area in 2000, and the greatest relative losses in 2015, with a suitable HP area reduction of 4830 km^2 (95%) and an optimal HP area reduction of 399 km^2 (91%) (Table 4.1, Figure 4.2). In contrast, the New York portion had a suitable HP area loss of 46 km^2 (0.3%), while optimal HP area loss was 369 km^2 (3.3%), due to 323 km^2 of optimal HP area being degraded to suitable HP area (Table 4.1, Figure 4.2).

Substantial habitat fragmentation occurred across the study area (Table 4.2). For suitable HPs, $m_{\mathrm{eff\_CUT}}$ size decreased by 45%, and $m_{\mathrm{eff\_CBC}}$ size decreased by 41%; for optimal HPs, both $m_{\mathrm{eff\_CUT}}$ size and $m_{\mathrm{eff\_CBC}}$ size decreased by 71% (Table 4.2). Fragmentation was most pronounced in the Ontario portion of the study area, whereas the New York portion experienced the least amount of fragmentation. At the MRC/county level, mean suitable HP $m_{\mathrm{eff\_CUT}}$ size decreased by 479 km^2, and mean suitable HP $m_{\mathrm{eff\_CBC}}$ size decreased by 8460 km^2 (Table S4.7). This same pattern was seen with optimal HPs at the MRC/county level (Table S4.8).

In 2000, 27 of the 43 MRCs/counties shared suitable HPs with at least one other MRC/county, as identified by $m_{\mathrm{eff\_CUT}}$ - $m_{\mathrm{eff\_CBC}}$ values > 0 (Table S4.7). However, in 2015, only 22 MRCs/counties shared suitable HPs with at least one other MRC/county. This was also the case with optimal HPs, where in 2000, 20 MRCs/counties shared optimal HPs with at least one other MRC/county, and in 2015, only 18 MRCs/counties shared optimal HPs (Table S4.8). Not only are fewer patches being shared, the average amount of patch sharing between MRCs/counties also declined: The mean difference in suitable HP $m_{\mathrm{eff\_CBC}}$ - $m_{\mathrm{eff\_CBC}}$, a measure of habitat sharing between MRCs/counties, decreased by 7981 km^2 (Table S4.7), and mean difference in optimal HP $m_{\mathrm{eff\_CBC}}$ - $m_{\mathrm{eff\_CBC}}$ decreased by 350 km^2 (Table S4.8).

In 2015, road density was highest in the Papineau patch in Québec and the patch east of Washington County (New York), with road densities of 0.67 km/km^2 and 0.66 km/km^2,

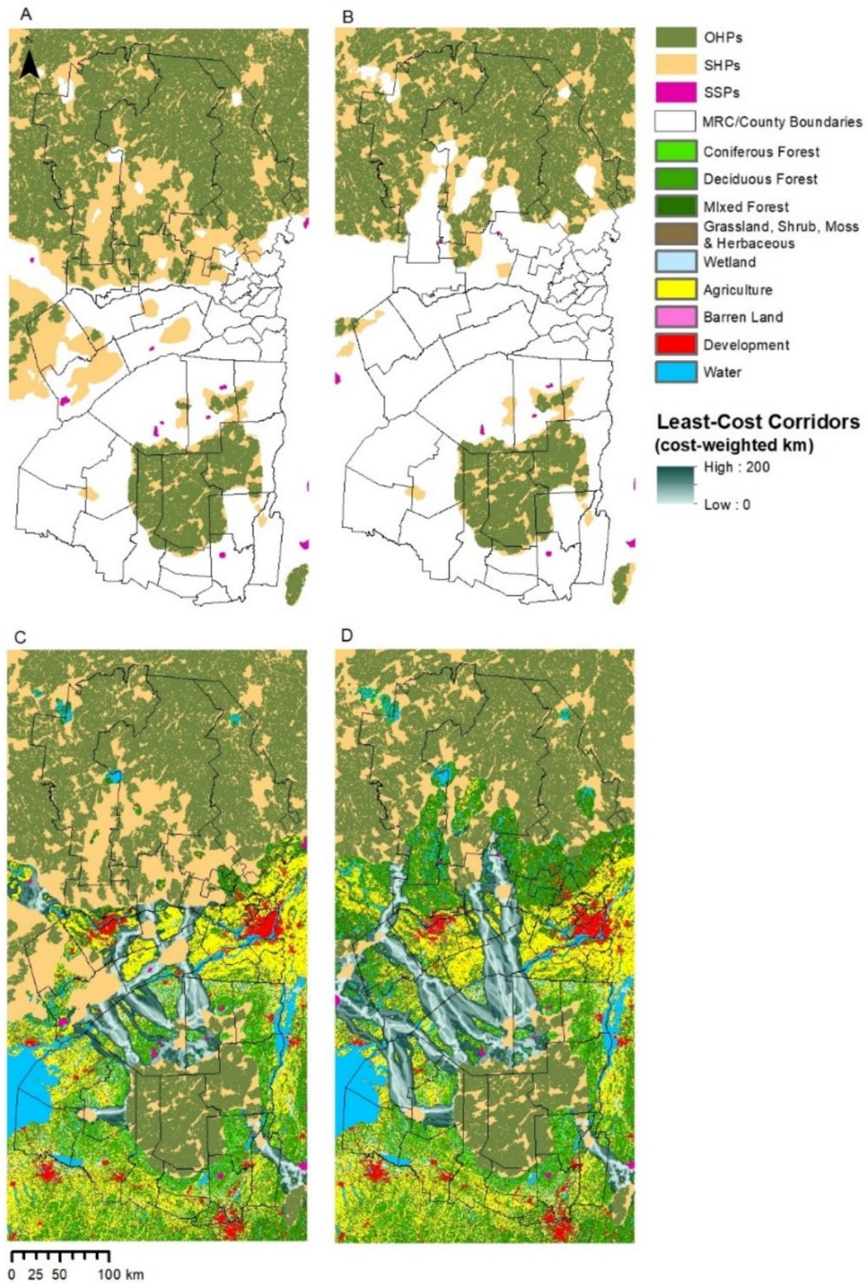

Figure 4.2 Wolf habitat patches and least-cost corridors (LCCs). A) Habitat patches in 2000, B) Habitat patches in 2015, C) Habitat patches and least-cost corridors in 2000, and D) Habitat patches and least-cost corridors in 2015. SHPs = Suitable habitat patches, OHPs = Optimal habitat patches, SSPs = Stepping-stone patches.

Location	SHPs 2000 m_{eff_CUT} (km^2)	SHPs 2015 m_{eff_CUT} (km^2)	SHPs 2015-2000 m_{eff_CUT} (km^2)	SHPs 2015-2000 m_{eff_CUT} (%)
Study Area	19822	10886	-8937	**-45**
Québec	39216	20205	-19012	**-48**
Ontario	1200	5	-1195	**-99.6**
New York	3729	3714	-15	-0.4
	SHPs 2000 m_{eff_CBC} (km^2)	**SHPs 2015 m_{eff_CBC} (km^2)**	**SHPs 2015-2000 m_{eff_CBC} (km^2)**	**SHPs 2015-2000 m_{eff_CBC} (%)**
Study Area	29842	17485	-12357	**-41**
Québec	60638	34484	-26154	**-43**
Ontario	2189	11	-2178	**-99.5**
New York	3729	3714	-15	-0.4
	OHPs 2000 m_{eff_CUT} (km^2)	**OHPs 2015 m_{eff_CUT} (km^2)**	**OHPs 2015-2000 m_{eff_CUT} (km^2)**	**OHPs 2015-2000 m_{eff_CUT} (%)**
Study Area	2342	681	-1661	**-71**
Québec	4889	1323	-3567	**-73**
Ontario	3	0.1	-3	**-96**
New York	198	167	-31	-15
	OHPs 2000 m_{eff_CBC} (km^2)	**OHPs 2015 m_{eff_CBC} (km^2)**	**OHPs 2015-2000 m_{eff_CBC} (km^2)**	**OHPs 2015-2000 m_{eff_CBC} (%)**
Study Area	2755	789	-1967	**-71**
Québec	5783	1555	-4227	**-73**
Ontario	3	0.3	-3	**-92**
New York	198	167	-31	-15

Table 4.2 Changes in the effective mesh size between 2000 and 2015 for suitable habitat patches (SHPs) and optimal habitat patches (OHPs), at the scale of the study area and in each provincial/state portion. Values in bold represent changes greater than 20%

respectively (Table S4.9). The suitable HP west of Lanark County in Ontario had the lowest road density at 0.30 km/km^2 (Table S4.9). Suitable HPs with the highest primary and secondary road densities were the Warren-Washington patch in New York (0.06 km/km^2), the Adirondack mega-patch in New York (0.07 km/km^2), and the patch east of Washington County in New York (0.11 km/km^2) (Table S4.9).

Functional connectivity

Although the number of least-cost corridors remained at 14, distances between suitable HPs increased (Figure 4.2). The mean Euclidean distance between suitable HPs increased by 46 km (*df* = 16, *p*-value = 0.02, 95% *CI* = -82.9 to -9.9 km, *Cohen's d* = 1.0); the mean cost-weighted distance increased by 2189 cost-weighted km (*df* = 16, *p*-value = 0.01, 95% *CI* = -3836.5 to -541.8 km, *Cohen's d* = 1.1); the mean least-cost path length increased by 63 km (*df* = 16, *p*-value = 0.01, 95% *CI* = -110.6 to -14.4 km, *Cohen's d* = 1.0); and the effective resistance increased by 4997 Ohms (*df* = 17, *p*-value = 0.03, 95% *CI* = -9331.0 to -663.7 Ohms, *Cohen's d* = 0.9).

Pinch-points were evident in most corridors, however, due to considerable suitable HP loss, the locations changed considerably between 2000 and 2015 (Figure 4.3). When Circuitscape was run in the "pairwise" mode, we identified areas of high current flow as pinch-points critical for movement between pairs of suitable HPs. Of particular importance were pinch-points located in the MRC Les Collines-de-l'Outaouais, Québec; Leeds/Grenville, and Stormont/Dundas/Glengarry, Ontario, and Jefferson County, St. Lawrence County, and Franklin County in New York (Figure 4.3B). When Circuitscape was run in the "all to one" mode, we identified areas of high current flow as pinch-points critical for maintaining connectivity for the entire network of suitable HPs. Although there were considerably less pinch-points produced by this method, the pinch-point in MRC Les Collines-de-l'Outaouais, Québec was still prominent, as was the pinch-point in Stormont/Dundas/Glengarry, Ontario (Figure 4.3D).

In 2000, 6 stepping-stone patches were identified within the least-cost corridors connecting suitable HPs in the study area. In 2015, there were 5 stepping-stone patches identified (Figure 4.2). Of particular importance is the patch shared by MRC Papineau ad MRC Les

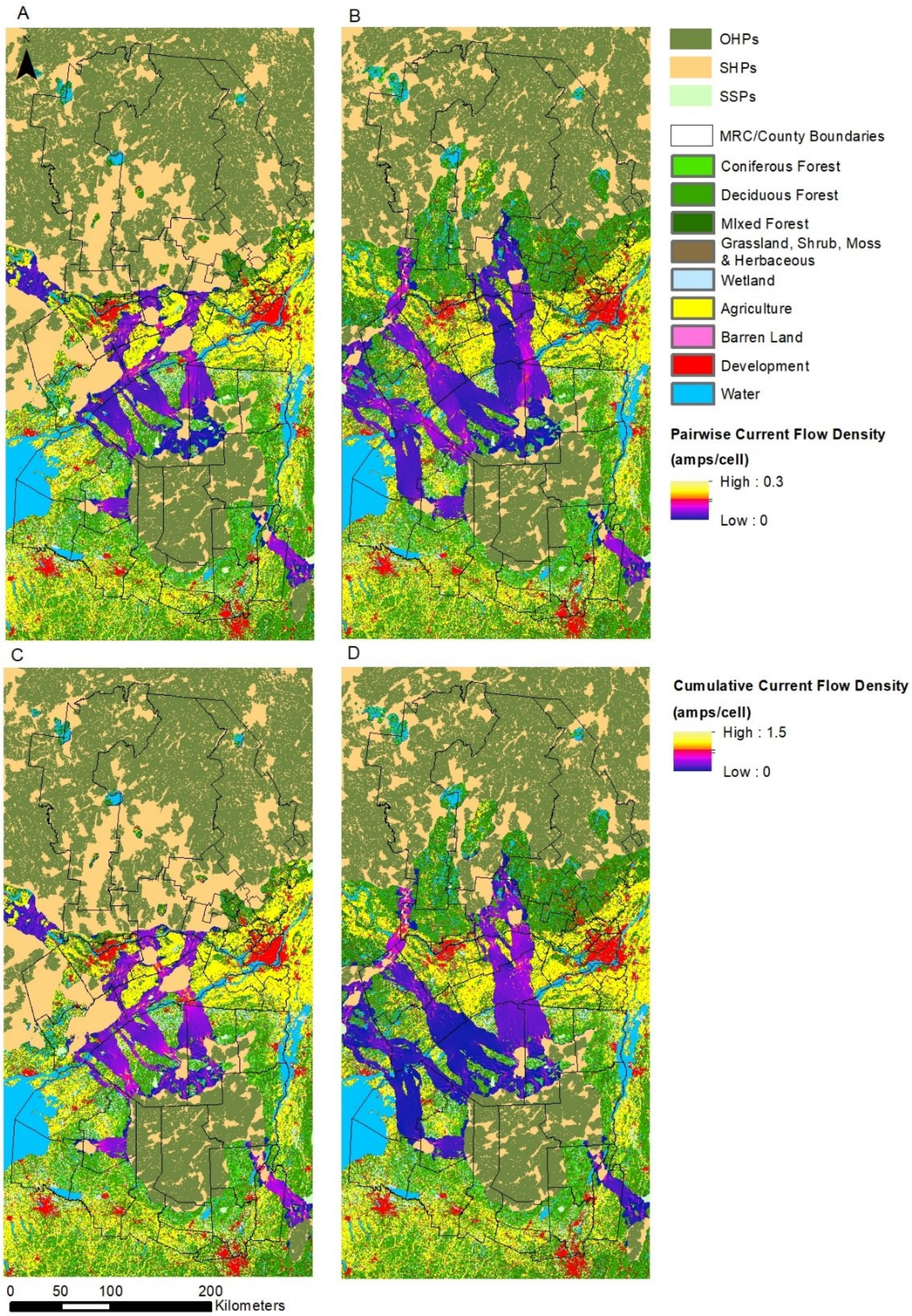

Figure 4.3 Pinch-points in the least-cost corridors (LCCs). A) Pairwise current flow density in 2000, B) Pairwise current flow density in 2015, C) Cumulative current flow density in 2000, D) Cumulative current flow density in 2015. SHPs = Suitable habitat patches, OHPs = Optimal habitat patches, SSPs = Stepping-stone patches.

Laurentides, Québec, as well as the patches in St. Lawrence County and Franklin County, New York (Figure 4.2).

Proportion of habitat patches and least-cost corridors under protection

In the A2L, 19% of suitable HP area, 22% of optimal HP area, and 9% of least-cost corridor area were protected by Canadian/U.S. government agencies and Nature Conservancy of Canada/The Nature Conservancy in 2015 (Figure S4.2; Table S4.10). However, this protection was not evenly distributed across the study area. In the Québec portion, 10% of suitable HP area, 11% of optimal HP area, and 14% of least-cost corridor area were protected; in the Ontario portion, no suitable nor optimal HP area was protected, and only 2% of least-cost corridor area; whereas in the New York portion, 76% of suitable and 85% of optimal HP area and 14% of least-cost corridor area were protected in 2015 (Figure S4.2; Table S4.10).

Discussion

Habitat amount

Wolf suitable HP area decreased by 27% and optimal HP area decreased by 17%. However, these declines in HP area were not equivalent to land cover loss. Natural land cover area (i.e., coniferous forest, deciduous forest, mixed forest, grassland, shrub, moss, herbaceous vegetation, and wetlands) only decreased by 1457 km^2 (2%) within the A2L between 2000 and 2015 (Cole et al. 2023b). Thus, the majority of HP area decline was due to suitable habitat becoming less desirable to wolves. In unprotected landscapes where mortality risk is high due to hunting and trapping, wolves can exhibit significant avoidance behavior of up to 1 km from human activity (including human presence, development, agriculture, and roads; Singleton 1995, Paquet et al.

1996). Thus, each new kilometer of anthropogenic land conversion between 2000 and 2015 created a 2 km^2 area of degraded habitat, reducing the size as well as eliminating many suitable and optimal HPs. The greatest amount of suitable and optimal HP area loss took place in the Québec and Ontario portions of the study area in response to increases in development and road network length. In the Québec portion, development increased by 833 km^2 and the length of the road network increased by 7684 km; in the Ontario portion, development increased by 445 km^2 and the length of the road network increased by 2380 km (Cole et al. 2023a, 2023b).

Although wolves have only been documented within the Québec mega-patch (Rogic et al. 2014; Mainguy et al. 2017; Hénault 2019; ECCC 2021), we identified 8 additional (potentially unoccupied) suitable HPs, with the largest being the Adirondak mega-patch in New York. Our results substantiate earlier studies that identified suitable wolf habitat in the Adirondack region. We identified 14,732 km^2 of suitable HP area within the New York portion in 2000. This agrees with estimates by Mladenof and Sickley (1998) who reported 16,020 km^2, and Harrison and Chapin (1998) who reported 14,618 km^2 of suitable wolf habitat in the New York region. For 2015, we identified 14,686 km^2 of suitable HP area in the New York portion, which was considerably smaller than the 22,847 km^2 estimated by van den Bosch (2022). This discrepancy could be due to the differences in map resolution used (i.e., 30 m vs 1 km), and/or how the two regions were delineated (habitat patch area vs habitat area). The nine suitable HPs identified within the A2L ranged in size from 137 km^2 to over 58,000 km^2. It has been estimated that gray wolf populations require an area of at least 12,800 km^2 for the persistence of an immigration-dependent population, and over 25,000 km^2 for a viable long-term independent population (USFWS 1992). Using these criteria, of the nine suitable HPs, only the Québec mega-patch is large enough to contain a viable long-term independent population, whereas the Adirondack mega-patch would be large enough to maintain an immigration-dependent population; the remaining seven suitable HPs would be considered sink populations. However, Fritts and Carbyn (1995) suggest that a protected area of at least 3000 km^2 with a sufficient prey base would be adequate to maintain a viable population in complete isolation; whereas Woodroffe and Ginsberg (1998) proposed that a critical reserve size of 766 km^2 would be necessary for a gray wolf population to have a long-term viability of 50%. Under these criteria, both the Québec mega-patch and the Adirondack mega-patch would be large enough to contain viable long-term independent wolf populations.

Fragmentation

Habitat fragmentation increases the probability of encounters and conflicts with humans. This increased pervasiveness of human presence in more fragmented landscapes reduces the potential for wolves to be ecologically effective (Kuijper et al. 2016). Habitat fragmentation significantly increased throughout the study area. Similar to the declines in HP area, the majority of habitat fragmentation occurred in the Québec and Ontario portions of the A2L. This result correlates with the increases in development and road network length in both regions (Cole et al. 2023a; 2023b). Although wolves can travel up to three times faster along tertiary roads and typically select these to increase prey encounter rates (Dickie et al. 2017; Muhly et al. 2019), primary and secondary roads contribute to habitat loss (due to avoidance behavior), increase mortality (due to collisions with vehicles), and can act as complete barriers to movement (due to fencing and traffic) leading to resource inaccessibility (Forman and Alexander 1998; Benson et al. 2015, 2024). Thus, wolves generally select habitats with low road density (i.e., < 0.3–0.7 km of roads per km^2, with the density of primary and secondary roads being < 0.02 km of roads per km^2;

Fuller et al. 1992; Wydeven et al. 1998; Rateaud et al. 2001). In 2015, all suitable HPs had road densities higher than 0.3 km/km^2, but less than 0.7 km/km^2. However, the Lanark, Adirondack mega-patch, the Warren-Washington, and the East of Washington patches all had combined primary and secondary road densities ≥ 0.02 km/km^2 which could deter wolf re-colonization of these habitat patches. In contrast, optimal HPs do not contain roads. In 2015, there were 101 optimal HPs (total area of 35,435 km^2; 68 in Québec, 1 in Ontario, and 32 in New York). They constitute the remaining large roadless areas > 70 km^2. Large roadless areas generally represent relatively undisturbed ecosystems with high ecological value, making their safeguarding important for the preservation of biodiversity and ecosystem services (Ibisch et al. 2016). Other than the "2001 Roadless Area Conservation Rule" which tentatively protects 236,700 km^2 of roadless areas on U.S. National Forest System lands, there is no legally binding legislation in place to protect large roadless areas in Canada and the U.S. (Coffin et al. 2021). Consequently, large roadless areas are scarcely considered in regional land development and transportation infrastructure planning (Selva et al. 2015).

Functional connectivity

Functional connectivity (i.e., the ability to move between resource patches within a landscape; Lindenmayer and Fischer 2013) is crucial for facilitating dispersal events between fragmented habitat patches. Dispersing individuals maintain long-term viability of populations by colonizing new areas, re-colonizing sink populations, and maintaining genetic variation and gene flow within meta-populations (Gonzalez et al. 1998; Kokko and López-Sepulcre 2006; Crooks et al. 2017). Re-colonization of suitable habitat patches within the A2L will require functional connectivity. However, functional connectivity is reduced when mortality risk outside of protected areas is high and habitat fragmentation increases the probability of encounters with humans. Between 2000 and 2015, functional connectivity among suitable HPs significantly decreased as measured by increases in mean Euclidean distance, mean least-cost path, mean cost-weighted distance, and mean effective resistance. Mean Euclidean distance increased directly, due to anthropogenic land conversion, and indirectly, due to the addition of avoidance buffers around these new land cover elements, which degraded adjacent habitat (i.e., habitat patches were reduced in size or completely lost) and resulted in greater distances between suitable HPs in 2015. Increases in mean least-cost path, mean cost-weighted distance, and mean effective resistance were due to anthropogenic land conversion, and the addition of avoidance buffers, which degraded adjacent habitat (i.e., increasing resistance values) and resulted in an overall increase in the cost of traveling between suitable HPs. Consequently, wolves in occupied sites in the Québec mega-patch will need to travel farther through less suitable habitat to recolonize unoccupied suitable HPs in the A2L, and the cost of traveling these distances will be higher. This reduced landscape-level connectivity may translate into longer time spent and farther distances traveled in both human-modified and unprotected landscapes (i.e., increased mortality and interactions with humans) during the transience stage and an overall reduction in the probability of dispersal success (Morales-González et al. 2022). This result suggests that protecting suitable HPs and the corridors that interconnect them may be critical for successful dispersal (Chapron et al. 2014) and expansion of wolf populations in the A2L. These declines in functional connectivity are consistent with other large mammal species within the A2L (fisher, moose, and white-tailed deer; Cole et al. 2023b). We identified fourteen least-cost corridors that interconnected the suitable HPs in both 2000 and 2015. In 2000, these corridors did not exceed 100 km in length. However, by 2015, three corridors were longer than 100 km, and four corridors were longer than 200 km. Although wolves have been recorded dispersing distances of up to 800 km (Linnell et al. 2005), typical dispersal events in the Great Lakes region range from 20 to 100 km (Treves et al. 2009). Thus, distance alone may reduce the probability of successful long-distance dispersal events within the A2L. Long-distance dispersal events between occupied sites in the Québec mega-patch and unoccupied sites in the Ontario and New York portions will require wolves to cross multiple primary and secondary roads. Road mortality is the second highest source of wolf fatality after hunting and trapping (Hebblewhite and Whittington 2020; ECCC 2021). Locations where least-cost corridors and pinch points intersect primary and secondary roads could be further evaluated as potential locations for wildlife passages and fencing to reduce mortality and increase landscape connectivity (Nussey and Noseworthy 2018; Spanowicz et al. 2020). Wolves may also need to traverse at least one of two large rivers (i.e., the Ottawa and St. Lawrence Rivers). While both rivers are major deterrents to long-distance dispersal, they are not complete barriers for wolves. Sections of the rivers freeze in the winter months permitting crossing, with some locations less than 1 km wide (Koen et al. 2015; ECCC 2023). Over the past 20 years, multiple wolves have been reported south of the St. Lawrence River, demonstrating that they are capable of crossing the rivers (Villemure and Jolicoeur 2004; McAlpine et al. 2015; Maine Wolf Coalition 2024).

Proportion of habitat patches and least-cost corridors under protection

Where wolves have been granted legal protection, they have been highly successful at recolonizing their former range, even in human-dominated landscapes (Linnell et al. 2001; Chapron et al. 2014; Smith et al. 2016). However, wolf recovery in unprotected landscapes is extremely challenging due to high rates of human-caused mortality (i.e., hunting, trapping, and conflicts with humans) when they venture outside of protected areas (Rutledge et al. 2017; Benson et al. 2024). For example, hunting and trapping outside park boundaries accounted for ~ 62% of annual mortality of wolf populations in Algonquin Park, Ontario (Theberge et al. 1996), and Benson et al. (2014) found that wolf survival declined outside of Algonquin Park as hunting and trapping access increased. In Parc National de la Mauricie, Québec, Villemure and Festa-Bianchet (2002) found that 88% of radio-collared wolf mortality occurred outside park boundaries. In Banf National Park, Alberta, wolves experienced up to 12.7 times higher daily risk of mortality when they ventured outside the park in winter during the hunting and trapping season (Hebblewhite and Whittington 2020). Therefore, wolf expansion into the Adirondack region or other suitable habitats within the A2L is unlikely without the enactment of legislation to protect wolves outside of protected areas (Rutledge et al. 2017; Benson et al. 2024). However, despite legal protection in New York under the Endangered Species Act 1973 (NYS-DEC 2023a), all wolves that have been reported within the region were killed by hunters or trappers mistaking them for coyotes (Villemure and Jolicoeur 2004; McAlpine et al. 2015; Maine Wolf Coalition 2024). Consequently, since wolves and coyotes are almost indistinguishable without genetic assessment (Vilaça et al. 2023), wolf expansion in the A2L would necessitate protection of both wolves and coyotes within the region. A similar ruling was passed in North Carolina to protect the critically endangered red wolf (*Canis rufus*) from mistaken identification by coyote hunters and trappers (Murray et al. 2015). Thus, identifying and protecting large areas of suitable habitat with sufficient prey density and ecological corridors that interconnect them would provide the greatest potential to maximize the ecological role that wolves play in ecosystem structure and function, while expanding the range and number of wolves in the region.

Only large, protected areas reduce mortality risk for wolves when human-caused mortality is high within adjacent landscapes (Larivière et al. 2000; Benson et al. 2024). However, most protected areas are simply too small to support viable populations of large-ranging species (Pimm et al. 2014; Williams et al. 2022). For example, in 2015, 14,605 km^2 (19%) of suitable HP area was protected, comprising 1056 Canadian/United States government sites and 381 Nature Conservancy of Canada/The Nature Conservancy sites. However, the average Canadian/United States government-protected area size was 13.3 km^2, and the average Nature Conservancy of Canada/The Nature Conservancy protected area size was 5.6 km^2. With the average regional wolf home range size being ~ 182 km^2 (Potvin 1988; Loveless 2010; Benson and Patterson 2015), and the area required to accommodate a viable long-term independent population being 25,600 km^2 (USFWS 1992), protected area sizes within the A2L are thus orders of magnitude too small, requiring wolves to inhabit large areas of unprotected land where hunting and trapping are permitted.

In 2015, the proportion of suitable and optimal HP area under protection was not evenly distributed across the A2L study area. While 10% of suitable HP area and 11% of optimal HP area were protected in Québec, zero suitable and optimal HP area were protected in the Ontario portion where the losses have been most pronounced. This was in stark contrast to the 76% of suitable and 85% of optimal HP area protected in the New York portion. This much more substantial amount of protection explains the stability in habitat amount and habitat fragmentation in the New York portion between 2000 and 2015. The amount of habitat patch area protected was considerably higher than the amount of corridor area protected. This result highlights the necessity to not only establish new and expand existing protected areas within the A2L, but also to restore and protect connectivity corridors between them (Hilty et al. 2020).

Conclusion

Although land conversion has diminished habitat amount, increased habitat fragmentation, and eroded functional connectivity between 2000 and 2015, we identified nine suitable HPs in the A2L, with the Québec and Adirondack mega-patches having the potential to accommodate long-term viable wolf populations. We also identified 14 least-cost corridors that interconnect the suitable HPs that have the potential to facilitate long-distance dispersals. However, with the region under high development pressure from a diversity of economic sectors (including agriculture, forestry, and urban development), it is unlikely that habitat loss and habitat fragmentation will subside without considerable conservation intervention.

Based on our findings, we propose the following nine recommendations to recover wolf habitat and functional connectivity within the A2L: (1) Commence extensive habitat restoration and protected area expansion, predominantly within the Québec and Ontario portions; (2) within the suitable HPs, maintain primary and secondary road density below 0.02 km of roads/km^2 and total road density below 0.7 km of roads/km^2; (3) avoid transportation development within the 101 optimal habitat patches that were identified as the last remaining large roadless areas; (4) develop collaborative conservation strategies to ensure that cross-border habitat patches, shared by multiple MRCs/counties, remain intact; (5) enhance and protect connectivity corridors between suitable HPs; (6) expand and protect stepping stone patches and pinch points within corridors to facilitate movement between suitable HPs; (7) determine priority locations for wildlife crossing structures to reduce road mortality and increase landscape connectivity; (8) maintain riparian access to ensure connectivity across waterways; and (9) although prey densities within the nine suitable HPs are adequate to accommodate wolf populations presently (Boucher et al. 2004; Hinton et al. 2022; NYS-DEC 2023c; Ontario 2023b; Rosenblatt et al. 2023), monitor prey densities as wolves re-colonize these locations. However, to facilitate wolf recovery within the A2L, either the protection of suitable habitat patches and corridors or the legal protection of both wolves and coyotes within the suitable habitat patches and corridors will be required to ensure that wolves are not harvested as they disperse and colonize new locations. This will necessitate collaborative and coordinated transboundary conservation between Québec, Ontario, and New York. However, beyond protecting habitats, corridors, and species, expansion and persistence within the A2L will ultimately depend on the willingness of humans to share the landscape with the wolf (van den Bosch et al. 2022).

Supplemental Tables and Figures

Canadian Land Cover, Circa 2000 NRS		
Class#	Category	Reclass#
0	No Data	0
11	Cloud	0
12	Shadow	0
20	Water	9
30	Barren/Rock	7
32	Rock/Rubble	7
33	Non-Veg Surfaces	7
34	Developed	8
40	Bryoids	4
50	Shrubland	4
51	Shrubland tall	4
52	Shrubland short	4
80	Wetland	5
81	Wetland - Treed	5
82	Wetland - Shrub	5
83	Wetland - Herb	5
100	Herb (Grasses/Crop)	4
110	Grassland	4
121	Annual Crops	6
122	Perrenial Crop land and Pasture	6
210	Coniferous Forest	1
211	Coniferous - Dense	1
212	Coniferous - open	1
213	Coniferous - Sparse	1
220	Deciduous Forest	2
221	Deciduous - dense	2
222	Deciduous - open	2
230	Mixed Forest	3
231	Mixed - Dense	3
232	Mixed - Open	3
233	Mixedwood Sparse	3

NLCD 2001 Land Cover (CONUS) USGS		
Class#	Category	Reclass#
0	Unclassified	0
11	Open Water	9
21	Development	8
22	Developed, Low Intensity	8
23	Developed, Medium Intensity	8
24	Developed, High Intensity	8
31	Barren Land	7
41	Deciduous Forest	2
42	Coniferous forest	1
43	Mixed Forest	3
52	Shrub/Scrub	4
71	Herbaceuous	4
81	Hay/Pasture	6
82	Cultivated Crops	6
90	Woody Wetlands	5
95	Emergent Herbaceuous Wetlands	5

North America Land Cover 2015 (Landsat and RapidEye)		
Class#	Category	Reclass#
0	Background	0
1	Temperate or sub-polar needleleaf forest	1
5	Temperate or sub-polar broadleaf deciduous forest	2
6	Mixed Forest	3
8	Temperate or sub-polar shrubland	4
10	Temperate or sub-polar grassland	4
12	Sub-polar or polar grassland-lichen-moss	4
13	Sub-polar or polar barren-lichen-moss	4
14	Wetland	5
15	Cropland	6
16	Barren Lands	7
17	Urban and Built-up	8
18	Water	9

Final 2000 & 2015 Reclass All Maps	
Class#	Category
0	NoData
1	Coniferous forest
2	Deciduous forest
3	Mixed forest
4	Grassland, Shrub, Moss, Herbaceous
5	Wetland
6	Agriculture
7	Barren Lands
8	Developed
9	Water

Table S4.1 Re-classification of the land-cover maps used in this study.

DMTI Roads Canada 2000 & 2018			
CARTO #	Category	Description	Re-Class
1	Expressway	400 series highways	1
2	Primary highway	Primary highways	2
3	Secondary highway	Secondary highway	2
4	Major road	Major road or arterial road	2
5	Local road	Subdivision road in a city or gravel road in a rural area	3

USA roads 2000 US Census Bureau			
CFCC #	Category	Description	Re-Class
A00	Primary highway with limited access	Interstate highways and some toll highways are in this category (A1) and are distinguished by the presence of interchanges.	1
A18			1
A21	Primary road without limited access	Nationally and regionally important highways that do not have limited access	2
A28			2
A31	Secondary and connecting road	State highways, but may include some county highways that connect smaller towns, subdivisions, and neighborhoods.	2
A38			2
A41	Local, neighborhood, and rural road	Local, Neighborhood, and rural road	3
A48	Vehicular trail	4WD access only dirt rural	3
A51			3
A53			3
A60	Road with special characteristics	Special road feature	3
A61	Cul de-sac		3
A62	Traffic circle		3
A63	Access ramp	The portion of a road that forms a cloverleaf or limited-access interchange	2
A64	Service drive	The road that provides access to businesses, facilities, and rest areas along a limited-access highway	2

USA roads 2015 US Census Bureau			
MTFCC #	Category	Description	Re-Class
S1100	Primary road	Primary roads are limited-access highways that connect to other roads	1
S1200	Secondary road	Secondary roads are main arteries that are not limited access, usually in the U.S. highway, state highway, or county highway systems.	2
S1400	Local	Local Neighborhood Road, Rural Road, City Street, Gravel, Dirt	3
S1500	Vehicular trail (4WD)	An unpaved dirt trail where a four-wheel drive vehicle is required	3
S1630	Ramp	A road that allows controlled access from adjacent roads onto a limited access highway, often in the form of a cloverleaf interchange.	1
S1640	Service	Service Drive usually along a limited access highway	1
S1730	Alley	It is located at the rear of buildings and properties and is used for deliveries	3
S1740	Private road	Private roads for service vehicles (logging, oil fields, ranches, etc.)	3

Final 2000 & 2015 North American Road Network Re-Class		
Class#	Category	Description
1	Primary road	All major highways
2	Secondary road	All secondary highways/major roads
3	Tertiary road	All other roads

Table S4.2 Re-classification of the road-network maps used in this study.

Raster Layer	Category	Suitable habitat value	Resistance value	Literature reference
Land-cover	Coniferous Forest	1	0	Singleton et al., 2002; Carroll et al., 2011
	Deciduous Forest	1	0	Singleton et al., 2002; Carroll et al., 2011
	Mixed Forest	1	0	Singleton et al., 2002; Carroll et al., 2011
	Grassland, etc.	0.8	20	Carroll et al., 2011
	Wetland	0.8	20	Singleton et al., 2002; expert opinion
	Agriculture	0.4	60	Singleton et al., 2002
	Barren Lands	0.6	40	Singleton et al., 2002
	Development	0.2	80	Singleton et al., 2002; Stricker et al., 2019
	Water	0	100	Singleton et al., 2002; Stricker et al., 2019
Road-network	Primary	0	100	Singleton et al., 2002; Carroll et al., 2011; expert opinion
	Secondary	0	100	
	Tertiary	0	100	
Primary roads	0-500m	0.4	60	Chapman, 1977; Thurber et al., 1994; Singleton, 1995,
	500-1000m	0.6	40	Paquet et al. 1996, 1999; Kaartinen et al., 2005;
	1000-	1	0	Carroll et al., 2011, expert opinion
Secondary roads	0-500m	0.6	40	Chapman, 1977; Thurber et al., 1994; Singleton, 1995,
	500-1000m	0.8	20	Paquet et al. 1996, 1999; Kaartinen et al., 2005;
	1000-	1	0	Carroll et al., 2011, expert opinion
Tertiary roads	0-500m	1	0	Chapman, 1977; Thurber et al., 1994; Singleton, 1995,
	500-1000m	1	0	Paquet et al. 1996, 1999; Kaartinen et al., 2005;
	1000-	1	0	Carroll et al., 2011, expert opinion
Development	0-500m	0.4	60	Paquet et al. 1999; expert opinion
	500-1000m	0.6	40	
	1000-	1	0	

Table S4.3 Suitable habitat and resistance values for each raster layer

Species	Home Range (km^2)			Reference	Location	Home Range Delineation Method
	Min	Average	Max			
Canis lupus	85.0	204.5	324.0	Potvin, 1987	Papineau-Labelle Wildlife Reserve, QC	95% Minimum convex polygon
Canis lupus	102.0	190.0	278.0	Loveless, 2010	Algonquin Provincial Park, ON	90% Fixed kernel density
Canis lycaon		239.4		Benson & Patterson, 2015	Algonquin Provincial Park, ON	90% Fixed kernel density
Canis lycaon		243.7		Benson & Patterson, 2015	Algonquin Provincial Park, ON	90% Fixed kernel density
Canis lycaon		168.4		Benson & Patterson, 2015	Algonquin Provincial Park, ON	90% Fixed kernel density
Canis lycaon		138.5		Benson & Patterson, 2015	Algonquin Provincial Park, ON	90% Fixed kernel density
Canis lycaon		314.0		Benson & Patterson, 2015	Algonquin Provincial Park, ON	90% Fixed kernel density
Canis lycaon		236.0		Benson & Patterson, 2015	Algonquin Provincial Park, ON	90% Fixed kernel density
Canis lycaon		104.4		Benson & Patterson, 2015	Kawartha Highlands, ON	90% Fixed kernel density
Canis lycaon		115.8		Benson & Patterson, 2015	Kawartha Highlands, ON	90% Fixed kernel density
Canis lycaon		114.4		Benson & Patterson, 2015	Wildlife Management Unit 49, ON	90% Fixed kernel density
Canis lycaon		115.4		Benson & Patterson, 2015	Wildlife Management Unit 49, ON	90% Fixed kernel density
Average	**93.5**	**182.0**	**301.0**			

Table S4.4 Regional home range values from literature used for this study.

Core Mapper Parameters									
Run Name- will define beginning of core area file names	**Moving Window Radius** (smaller values result in larger numbers of more-detailed core areas)	**Min Average Habitat Value** (avg habitat value in the moving window around a pixel must be greater than this for the pixel to be considered 'core')	**Min Habitat Value Per Pixel** (Pixel value must be greater than this to be 'core')	**OPTIONAL: Expand cores by this CWD value** (Grows cores outward after minimum habitat values applied. Enter 0 to skip)	**Trim back expanded cores** (After CWD expansion, trim back cores by eliminating pixels using moving window average habitat values. This eliminates "halos" around core areas)	**Min Core Area size** (squared map units. Core areas smaller than this will be eliminated at end)	**Exclude nonhabitat from core size calcs** (Exclude pixels below per-pixel cutoffs from core area calculation when cores have been expanded. This makes core area calculation more conservative and eliminates cores with low amounts of habitat.)	**Append core stats** (can take extra time with very large numbers of cores)	**Delete Temporary Files**
Wolf_SHP	9788.3	0.6	0	5455.5	Yes	70100000	No	Yes	Yes
Wolf_SSP	9788.3	0.6	0	5455.5	Yes	10000000	No	Yes	Yes
Wolf_OHP	9788.3	0.6	0.4	0	No	70100000	No	Yes	Yes

Table S4.5 Core Mapper parameters used for this study: SHP = Suitable habitat patches, SSP = Stepping-stone patches, OHP = Optimum habitat patches

Maps	2000	Resolution	Source	Link to Dataset
Land-cover				
Quebec Ontario	Canadian Land Cover, Circa 2000 (Vector) - GeoBase Series, 1996-2005	Vector	Natural Resources Canada	https://open.canada.ca/data/en/dataset/97126362-5a85-4fe0-9dc2-915464cfdbb7
NY/VT/MA	NLCD 2001 Land Cover (CONUS)	30m	U.S. Geological Survey (USGS)	https://www.mrlc.gov/data?f%5B0%5D=category%3Aland%20cover&f%5B1%5D=region%3Aconus
Roads				
Quebec Ontario	DMTI 2000	Polyline	DMTI	Received from Concordia University Library - Concordia Licence
NY/VT/MA	TigerLine 2000	Polyline	United States Census Bureau	https://www.census.gov/geographies/mapping-files/time-series/geo/tiger-line-file.html
MRC/County Boundaries				
Quebec	Civil Boundaries	Vector	Quebeec Government	Recived from the Nature Conservancy of Québec
Ontario	Civil Boundaries	Vector	Ontario Government	https://data.ontario.ca/
NY/VT/MA	TigerLine Shapefiles	Vector	United States Census Bureau	https://www.census.gov/geographies/mapping-files/time-series/geo/tiger-line-file.html

Maps	2015	Resolution	Source	Link to Dataset
Land-cover				
Quebec Ontario NY/VT/MA	North America Land Cover (Landsat and RapidEye)	30m	Commission for Environmental Cooperation (CEC).	http://www.cec.org/north-american-environmental-atlas/land-cover-30m-2015-landsat-and-rapideye/
Roads				
Quebec Ontario	DMTI 2018	Polyline	DMTI	Received from Concordia University Library - Concordia Licence
NY/VT/MA	TigerLine 2015	Polyline	United Staes Census Bureau	https://catalog.data.gov/dataset/2015-roads-national-geodatabase
MRC/County Boundaries				
Quebec	Civil Boundaries	Vector	Quebeec Government	Recived from the Nature Conservancy of Québec
Ontario	Civil Boundaries	Vector	Ontario Government	https://data.ontario.ca/
NY/VT/MA	TigerLine Shapefiles	Vector	United States Census Bureau	https://www.census.gov/geographies/mapping-files/time-series/geo/tiger-line-file.html
Proteted Areas				
Quebec	Protected areas (Canada)	Vector	Protected Planet	https://www.protectedplanet.net/country/CAN
	Areas protected by NCC (Québec)		Nature Conservancy of Québec	Recived from the Nature Conservancy of Québec
Ontario	Protected areas (Canada)		Protected Planet	https://www.protectedplanet.net/country/CAN
	Areas protected by NCC (Ontario)		Nature Conservancy of Québec	Recived from the Nature Conservancy of Québec
NY/VT/MA	Protected areas (USA)		Protected Planet	https://www.protectedplanet.net/country/USA
	Areas protected by TNC (USA)		The Nature Conservancy	TNC Lands

Table S4.6 Map layers used for this study and their sources.

MRC/County	SHPs 2000 m_{eff_CUT} (km^2)	SHPs 2015 m_{eff_CUT} (km^2)	Change in SHPs 2015-2000 m_{eff_CUT} (km^2)	Percent change in SHPs 2015-2000 m_{eff_CUT} (%)	SHPs 2000 m_{eff_CBC} (km^2)	SHPs 2015 m_{eff_CBC} (km^2)	Change in SHPs 2015-2000 m_{eff_CBC} (km^2)	Change in SHPs 2015-2000 m_{eff_CBC} (%)	SHPs 2000 m_{eff_CBC} - m_{eff_CUT} (km^2)	SHPs 2015 m_{eff_CBC} - m_{eff_CUT} (km^2)	Change in SHPs 2015-2000 m_{eff_CBC} - m_{eff_CUT} (km^2)
Antoine-Labelle	15932.0	11629.0	-4303.1	**-27.0**	73564.8	49794.6	-23770.3	**-32.3**	57632.8	38165.6	-19467.2
Matawinie	8958.9	6566.1	-2392.7	**-26.7**	68837.1	46690.5	-22146.6	**-32.2**	59878.2	40124.3	-19753.9
La Vallée-de-la-Gatineau	13320.4	8757.7	-4562.8	**-34.3**	72181.4	46370.0	-25811.4	**-35.8**	58861.0	37612.4	-21248.6
Papineau	2461.5	239.9	-2221.6	**-90.3**	65038.7	15742.3	-49296.3	**-75.8**	62577.2	15502.4	-47074.8
Hamilton	4695.2	4694.8	-0.4	0.0	14007.6	13979.8	-27.7	-0.2	9312.4	9285.1	-27.3
Les Laurentides	2344.3	79.1	-2265.1	**-96.6**	69504.6	10116.6	-59388.0	**-85.4**	67160.3	10037.5	-57122.8
Essex	2070.1	2055.1	-15.0	-0.7	9044.6	8994.4	-50.2	-0.6	6974.5	6939.3	-35.2
Herkimer	1221.8	1224.6	2.8	0.2	7952.2	7945.7	-6.4	-0.1	6730.3	6721.2	-9.2
Warren	260.8	259.5	-1.3	-0.5	4595.2	4575.8	-19.5	-0.4	4334.4	4316.2	-18.2
Franklin	344.8	334.8	-10.0	-2.9	3926.9	3862.1	-64.8	-1.6	3582.1	3527.3	-54.8
Fulton	76.3	75.4	-0.9	-1.2	3284.4	3258.2	-26.3	-0.8	3208.1	3182.8	-25.3
Clinton	59.1	62.0	2.9	4.9	2004.0	2048.9	44.9	2.2	1944.9	1986.9	42.0
Lewis	58.0	58.6	0.7	1.2	1760.3	1748.3	-12.0	-0.7	1702.4	1689.7	-12.7
St Lawrence	59.0	56.8	-2.2	-3.7	1138.4	1135.7	-2.8	-0.2	1079.4	1078.9	-0.5
Les Collines-de-l'Outaouais	1451.5	0.3	-1451.1	**-100.0**	60407.9	737.3	-59670.6	**-98.8**	58956.5	737.0	-58219.5
Saratoga	6.2	5.9	-0.4	-6.2	746.1	721.1	-25.0	-3.3	739.8	715.2	-24.6
Oneida	1.5	1.5	-0.1	-3.9	303.6	297.0	-6.6	-2.2	302.1	295.6	-6.5
Lanark	1751.5	22.5	-1729.0	**-98.7**	5793.1	51.1	-5741.9	**-99.1**	4041.6	28.7	-4012.9
Washington	3.9	4.6	0.7	17.8	5.5	6.2	0.7	12.5	1.6	1.6	0.0
Oswego	0.7	0.8	0.1	7.1	2.6	3.0	0.3	12.8	1.9	2.2	0.3
Argenteuil	542.1	0.0	-542.1	**-100.0**	47292.2	1.1	-47291.1	**-100.0**	46750.1	1.1	-46749.0
Jefferson	0.0	0.0	0.0	29.7	0.1	0.1	0.0	24.1	0.1	0.1	0.0
Stormont/Dundas/Glengarry	94.3	0.0	-94.3	**-100.0**	94.3	0.0	-94.3	**-100.0**	0.0	0.0	0.0
Ottawa/Carleton	38.7	0.0	-38.7	**-100.0**	906.2	0.0	-906.2	**-100.0**	867.5	0.0	-867.5
Leeds/Grenville	638.2	0.0	-638.2	**-100.0**	3224.9	0.0	-3224.9	**-100.0**	2586.7	0.0	-2586.7
Prescott/Russel	28.7	0.0	-28.7	**-100.0**	31.4	0.0	-31.4	**-100.0**	2.8	0.0	-2.8
Montgomery	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Schenectady	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Les Pays-d'en-Haut	257.1	0.0	-257.1	**-100.0**	43868.1	0.0	-43868.1	**-100.0**	43611.0	0.0	-43611.0
Montcalm	27.0	0.0	-27.0	**-100.0**	14408.4	0.0	-14408.4	**-100.0**	14381.5	0.0	-14381.5
La Rivière-du-Nord	5.3	0.0	-5.3	**-100.0**	7934.4	0.0	-7934.4	**-100.0**	7929.1	0.0	-7929.1
Les Moulins	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Thérèse-De Blainville	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Mirabel	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Montréal	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Laval	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Deux-Montagnes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gatineau	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Vaudreuil-Soulanges	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Roussillon	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Beauharnois-Salaberry	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Les Jardins-de-Napierville	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Le Haut-Saint-Laurent	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Average	**1318.8**	**840.2**	**-478.6**	**-28.6**	**13531.6**	**5071.6**	**-8460.0**	**-28.3**	**12212.8**	**4231.4**	**-7981.4**

Table S4.7 Changes in the Effective Mesh Size for suitable habitat patches (SHPs) between 2000 and 2015, for each MRC/county. MRC/County values ranked from highest to lowest 2015 $m_{\mathrm{eff\_CBC}}$ SHP value (i.e., from lowest to highest amount of fragmentation). Values in bold represent changes greater than 20%

MRC/County	OHPs 2000 m_{eff_CUT} (km^2)	OHPs 2015 m_{eff_CUT} (km^2)	Change in OHPs 2015-2000 m_{eff_CUT} (km^2)	Percent change in OHPs 2015-2000 m_{eff_CUT} (%)	OHPs 2000 m_{eff_CBC} (km^2)	OHPs 2015 m_{eff_CBC} (km^2)	Change in OHPs 2015-2000 m_{eff_CBC} (km^2)	Change in OHPs 2015-2000 m_{eff_CBC} (%)	OHPs 2000 m_{eff_CBC} - m_{eff_CUT} (km^2)	OHPs 2015 m_{eff_CBC} - m_{eff_CUT} (km^2)	Change in OHPs 2015-2000 m_{eff_CBC} - m_{eff_CUT} (km^2)
Matawinie	3801.7	1216.4	-2585.3	**-68.0**	11992.3	3088.8	-8903.4	**-74.2**	8190.6	1872.5	-6318.1
Antoine-Labelle	4884.6	1570.6	-3314.0	**-67.8**	10881.8	3017.5	-7864.3	**-72.3**	5997.2	1446.9	-4550.3
Les Laurentides	98.5	44.2	-54.3	**-55.1**	3501.4	1152.3	-2349.2	**-67.1**	3402.9	1108.1	-2294.9
Hamilton	419.0	389.5	-29.4	-7.0	821.0	705.4	-115.6	-14.1	402.0	315.9	-86.1
Herkimer	280.2	241.4	-38.8	-13.8	670.9	584.2	-86.7	-12.9	390.7	342.8	-47.9
La Vallée-de-la-Gatineau	341.0	365.5	24.6	7.2	2024.9	507.2	-1517.6	**-74.9**	1683.9	141.7	-1542.2
Essex	294.6	292.6	-2.0	-0.7	413.8	350.9	-62.9	-15.2	119.2	58.3	-60.9
Warren	60.6	46.5	-14.1	-23.2	124.7	87.9	-36.7	**-29.4**	64.0	41.4	-22.6
St Lawrence	20.7	22.5	1.7	8.4	83.7	85.1	1.4	1.7	63.0	62.7	-0.3
Franklin	37.0	24.9	-12.1	**-32.7**	135.7	83.2	-52.5	**-38.7**	98.8	58.3	-40.4
Fulton	14.2	14.2	0.0	0.0	68.6	69.2	0.5	0.8	54.5	55.0	0.5
Lewis	20.5	11.2	-9.3	**-45.4**	80.0	59.3	-20.7	**-25.9**	59.5	48.2	-11.4
Papineau	80.4	10.8	-69.6	**-86.6**	109.9	12.0	-97.9	**-89.1**	29.5	1.2	-28.3
Oneida	0.5	0.1	-0.4	**-80.4**	21.8	9.1	-12.8	**-58.5**	21.3	9.0	-12.4
Saratoga	5.4	2.8	-2.6	**-48.0**	9.8	5.5	-4.3	**-43.6**	4.4	2.7	-1.7
Clinton	3.5	3.6	0.1	1.7	4.1	4.6	0.5	12.5	0.6	1.1	0.5
Lanark	10.0	0.5	-9.5	**-94.9**	14.1	1.3	-12.8	**-90.7**	4.1	0.8	-3.3
Les Collines-de-l'Outaouais	28.5	0.1	-28.5	**-99.7**	45.3	0.5	-44.8	**-99.0**	16.8	0.4	-16.4
Washington	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Oswego	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Argenteuil	6.4	0.0	-6.4	**-100.0**	25.0	0.0	-25.0	**-100.0**	18.5	0.0	-18.5
Jefferson	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stormont/Dundas/Glengarry	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ottawa/Carleton	2.9	0.0	-2.9	**-100.0**	2.9	0.0	-2.9	**-100.0**	0.0	0.0	0.0
Leeds/Grenville	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Prescott/Russel	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Montgomery	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Schenectady	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Les Pays-d'en-Haut	4.0	0.0	-4.0	**-100.0**	7.0	0.0	-7.0	**-100.0**	3.0	0.0	-3.0
Montcalm	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
La Rivière-du-Nord	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Les Moulins	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Thérèse-De Blainville	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Mirabel	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Montréal	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Laval	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Deux-Montagnes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gatineau	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Vaudreuil-Soulanges	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Roussillon	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Beauharnois-Salaberry	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Les Jardins-de-Napierville	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Le Haut-Saint-Laurent	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Average	**242.2**	**99.0**	**-143.2**	**-23.4**	**721.8**	**228.5**	**-493.4**	**-25.4**	**479.6**	**129.5**	**-350.2**

Table S4.8 Changes in the Effective Mesh Size for optimal habitat patches (OHPs) between 2000 and 2015, for each MRC/county. MRC/County values ranked from highest to lowest 2015 $m_{\text{eff\_CBC}}$ SHP value (i.e., from lowest to highest amount of fragmentation). Values in bold represent changes greater than 20%

Suitable habitat patch	Area (km^2)	Primary & Secondary Roads (km/km^2)	Tertiary Roads (km/km^2)	Primary, Secondary & Tertiary Roads (km/km^2)
Québec mega-patch	58861+	0.01	0.33	0.33
Papineau	192	0.01	0.66	0.67
Lanark	611+	0.02	0.52	0.54
West of Lanark*	167	0.00	0.30	0.30
Adirondack mega-patch	14037	0.07	0.31	0.38
St. Lawrence	305	0.00	0.39	0.39
Lewis-Oswego	207	0.00	0.45	0.45
Warren-Washington	137	0.06	0.30	0.35
East of Washington*	676	0.11	0.55	0.66

Table S4.9 Suitable habitat patch sizes and their road densities in 2015. (+) signifies that the suitable habitat patch crosses the A2L boundary and therefore, its total area is larger than indicated. (*) signifies that the suitable habitat patch is not within the A2L boundaries.

Habitat type	Study Area	Québec	Ontario	New York
SHP area (km^2)	49633	34679	269	14686
SHP area protected (km^2)	14605	3410	0	11195
Proportion of SHP area protected (%)	**19**	**10**	**0**	**76**
OHP area (km^2)	35434	24682	40	10712
OHP area protected (km^2)	11700	2641	0	9059
Proportion of OHP area protected (%)	**22**	**11**	**0**	**85**
LCC area (km^2)	21607	2293	5850	10334
LCC area protected (km^2)	2012	315	101	1596
Proportion of LCC area protected (%)	**9**	**14**	**2**	**15**

Table S4.10 Amount of suitable habitat patch (SHP) area, optimum habitat patch (OHP) area, and least-cost corridor (LCC) area, protected by Canadian/U.S. government agencies and Nature Conservancy of Canada/The Nature Conservancy in 2015, at the scale of the study area and each provincial/state portion.

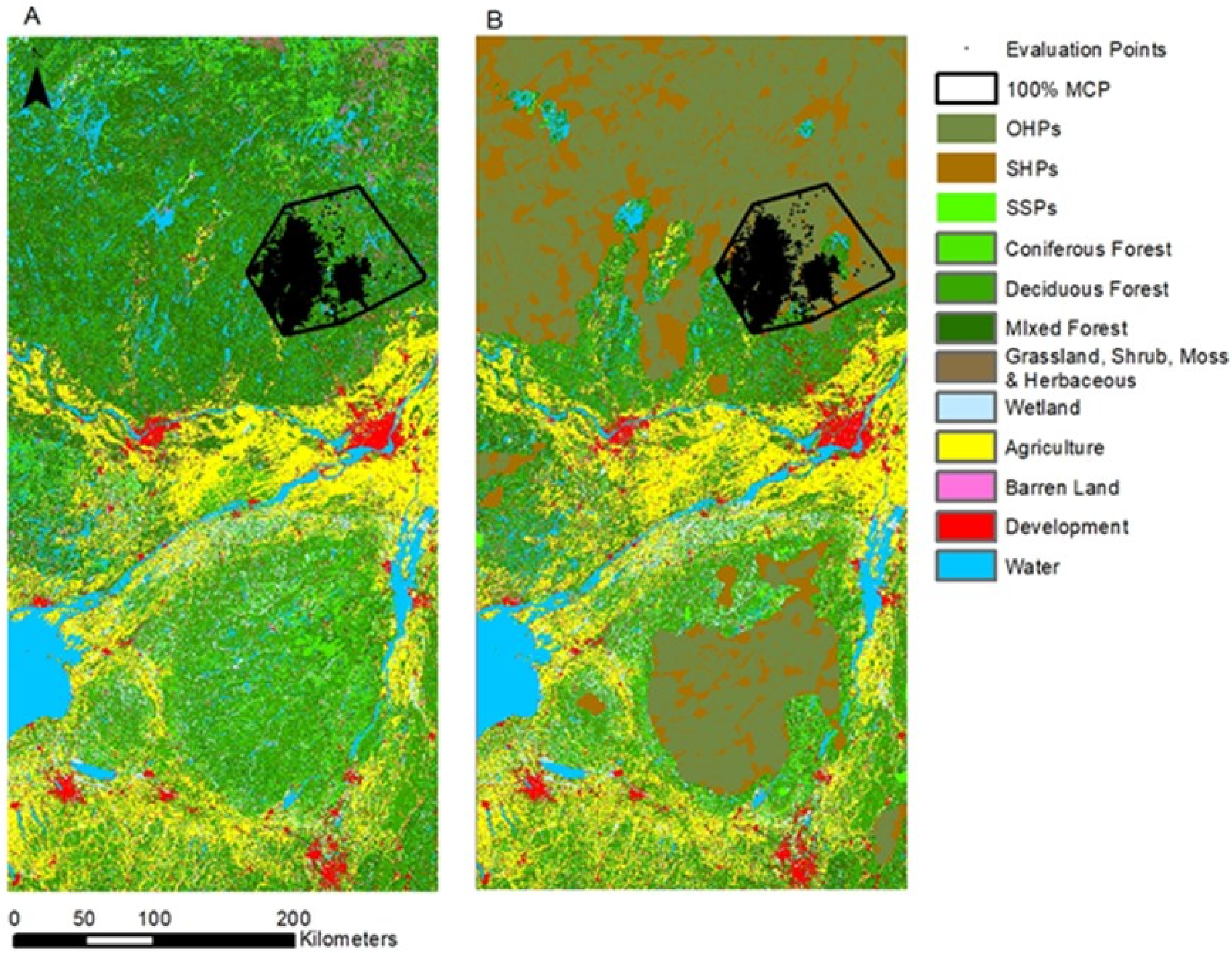

Figure S4.1 A) Validation points within the 100% minimum convex polygon (MCP) relative to suitable habitat in 2015, B) Validation points within MCP relative to habitat patches in 2015.

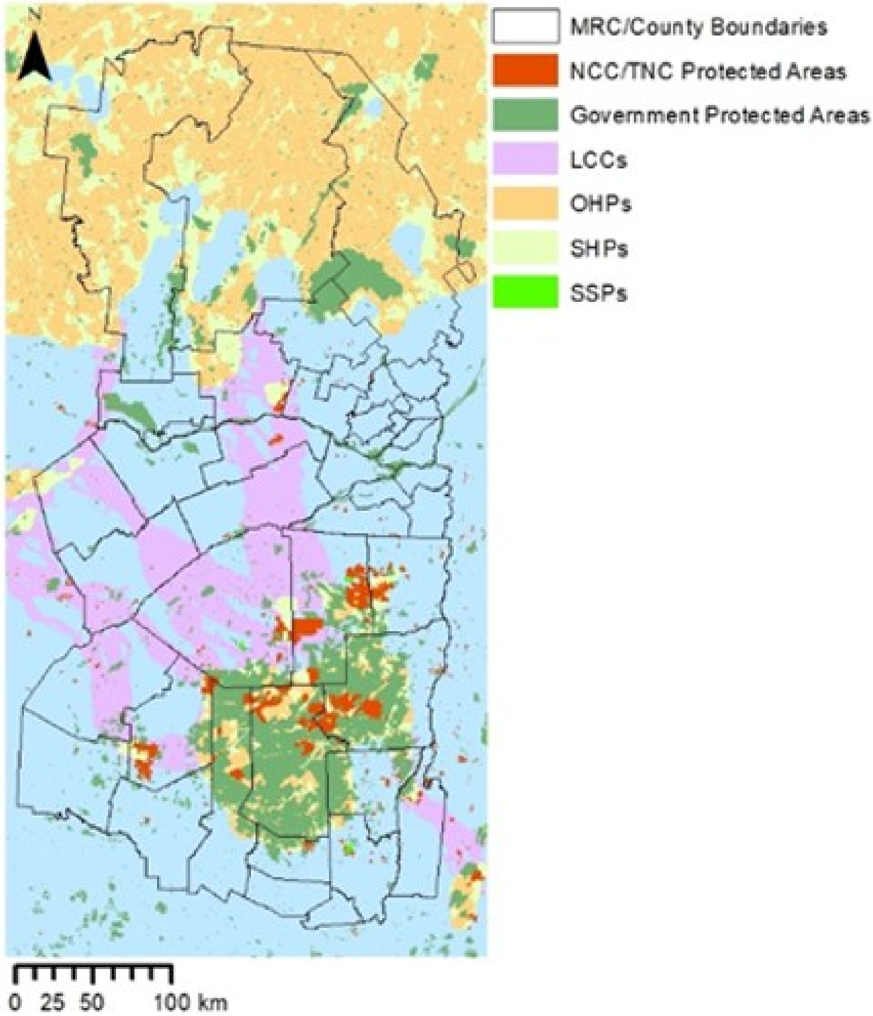

Figure S4.2 Protected areas within the Adirondack-to-Laurentians (A2L) study area in 2015. LCCs = Least-cost corridors, SHPs = Suitable habitat patches, OHPs = Optimal habitat patches, SSPs = Stepping-stone patches.

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5. Synthesis

This thesis quantified the impacts of anthropogenic land transformation on habitat amount, fragmentation, and connectivity in the Adirondack-to-Laurentians (A2L) transboundary wildlife linkage between 1992 and 2018 and identified priority areas for conservation and restoration.

Statement of Originality

In Chapter 2, I presented the first analysis of landscape structure on the swath of land between the Laurentian Mountains in Québec, Canada, and the Adirondack Mountains in New York, USA. I also gave the region its name - the Adirondack-to-Laurentians (A2L) transboundary wildlife linkage. This is also the first study of the A2L region using multiple points in time. Although there have been several "static" studies of the extent of landscape connectivity within the larger region, such as the Algonquin-to-Adirondacks (A2A) region (Quinby et al., 1999), Southeastern Canada/Northeastern USA (Carroll, 2003), and Montréal and the Saint Lawrence Lowlands (Mitchell et al., 2015; Albert et al., 2017; Rayfield et al., 2019; Gonzalez et al., 2019), this was the first "temporal" study of changes in landscape structure within the A2L. This was the first study to use both the effective mesh "CUT" and "CBC" procedures together to determine (1) the degree of habitat loss and fragmentation strictly within each MRC/county, and (2) the degree of habitat loss and fragmentation within each MRC/county that included the area of habitat patches that crossed over MRC/county boundaries. This produced a much better interpretation of the extent of fragmentation throughout the study area. This was also the first study to calculate the difference between $m_{\text{eff\_CUT}}$ and $m_{\text{eff\_CBC}}$ values to determine the degree of patch sharing between adjacent MRCs/counties. I identified shared patches in the hope that collaborative conservation strategies will be created between MRCs/counties to ensure that these patches continue to serve as vital habitats, connectivity corridors, and stepping-stones for a wide range of species within the A2L. Finally, the data and methods I presented in chapter 2 will provide valuable information for land-use, transportation, and conservation planning, and could be used to monitor and evaluate the impacts of future land-use development scenarios.

In Chapter 3, I presented the first assessment of species-specific habitat amount, habitat fragmentation, and functional connectivity in the A2L. This was also the first study to identify, evaluate, and quantify suitable and optimal habitat patches. This study is, to my knowledge, the only one to incorporate avoidance behavior distances in a suitable-habitat assessment; and it is the first study to determine the amount of habitat protection for each of the focal species in the A2L transboundary wildlife linkage. Finally, this was the first study to use Euclidean distance, least-cost path distance, cost-weighted distance and effective resistance to measure changes in functional connectivity for a species across a network of habitat patches.

In Chapter 4, I presented the first study identifying potential wolf habitat amount within the A2L. This was also the first study to use multiple points in time to assess changes in habitat amount, habitat fragmentation, and functional connectivity for the wolf within in the A2L. This study was the first assessment of functional connectivity for the eastern wolf in the region since *circa* 2000 (Harrison & Chapin, 1998; Mladenoff & Sickley, 1998; Paquet et al., 1999; Carroll, 2003); and it was the first study to determine the amount of suitable wolf habitat protected in the A2L transboundary wildlife linkage.

Main Findings

These studies revealed that extensive changes in landcover occurred within the A2L where forest areas declined by 1363 km^2 and wetland areas declined by 1365 km^2 (69%). This was most pronounced in the Québec portion of the A2L where wetland areas declined by 872 km^2 (88.5%) (Chapter 2). These landcover changes had drastic effects on species-specific habitat amount. Between 2000 and 2015, suitable and optimal habitat patch area declined for all species examined (i.e., moose, white-tailed deer, black bear, and fisher). Moose and black bear habitat patches experienced the greatest habitat loss, with reductions of 16,842 km^2 (26%) and 8894 km^2 (11%), respectively. Most of these losses took place in the Québec portion of the study area where suitable habitat patch area declined by 13,382 km^2 (28%) for moose, and 6891 km^2 (14%) for black bear (Chapter 3). In addition, wolf suitable habitat patch area decreased by 18,245 km^2 (27%), with losses of 28% in the Québec portion, 95% in the Ontario portion, but only 0.3% in the New York portion (Chapter 4). These studies revealed that species that exhibit avoidance behaviour experienced considerably greater loss of habitat patch area, due to indirect habitat loss. Indirect habitat loss (i.e., functional loss of habitat) differs from direct habitat loss because some suitable habitat remains but is now avoided due to human disturbance (Dyer et al., 2001; Polfus et al., 2011; Barré et al., 2018; Plante et al., 2018; Heinemeyer et al., 2019; Ellis et al., 2023). For species that exhibit avoidance behaviour, even a small amount of anthropogenic land transformation can have drastic effects on the remaining habitat amount.

Habitat fragmentation also increased throughout the study area, with a $m_{\mathrm{eff\_CUT}}$ size decrease of 60% in the Québec portion, 12% in the Ontario portion, and 4% in the New York portion. Forest areas in the Québec portion experienced the greatest amount of fragmentation with a $m_{\mathrm{eff\_CUT}}$ decline of 3262.5 km^2 (58.5%). This study revealed that although 76% (128,813 km^2) of the study area was made up of natural landcover (i.e., forests, wetlands, grasslands, etc.) it was highly fragmented by anthropogenic elements (i.e., roads, development, etc.). In 2015, natural landcover was made up of 67,790 habitat patches of which, 59,589 were less than 1 km^2, and the average patch size was 1.42 km^2 (Chapter 2). Landscape fragmentation had severe effects on species-specific habitat fragmentation. Suitable and optimal habitat patch fragmentation increased for all species within the study area between 2000 and 2015. Moose experienced the highest level of suitable habitat patch fragmentation with an 8674 km^2 (46%) reduction in $m_{\text{eff\_CUT}}$ size. The majority of habitat fragmentation took place in the Ontario and Québec portions of the study area where moose also experienced the highest level of suitable habitat patch fragmentation with an 18,672 km^2 (49%) reduction in $m_{\text{eff\_CUT}}$ size, and in the Ontario portion, moose had a 298 km^2 (99%) reduction in $m_{\text{eff\_CUT}}$ size (Chapter 3). In addition, wolf suitable habitat patch fragmentation also increased with an 8937 km^2 (45%) reduction in $m_{\text{eff\_CUT}}$ size. This was most pronounced in the Québec and Ontario portions with $m_{\text{eff\_CUT}}$ size reductions of 19,012 km^2 (48%) and 1195 km^2 (99%) respectively, whereas the New York portion experienced a reduction of 15 km^2 (0.4%) (Chapter 4).

Functional connectivity decreased for all species, with moose experiencing the greatest declines in all connectivity measurements (i.e., Euclidean distance, cost-weighted distance, least-cost path length, and effective resistance) (Chapter 3). Functional connectivity also significantly decreased for the wolf, with mean distances and the cost of travelling these distances more than doubling (Chapter 4). Recent studies by Pither et al. (2023) and van den Bosch et al. (2023) also found weak connectivity between the Adirondack Mountains and the Laurentian Mountains (A2L) and much stronger connectivity in the A2A corridor that runs between Lake Ontario and Ottawa and from Algonquin Park to the Adirondack Mountains (A2A).

Finally, these studies revealed that habitat loss and fragmentation decreased with the amount of area protected; with New York having the greatest protection and the least amount of habitat loss and fragmentation and Ontario having the least protection and the greatest amount habitat loss and fragmentation (Chapters 3 & 4). Most protected areas within the A2L are also too small to support viable populations of large-ranging species. For example, in 2015, the average government protected area size was 13.3 km^2, and the average Nature Conservancy of Canada/The Nature Conservancy protected area size was 5.6 km^2, which are considerably below the average home range size of fisher (19 km^2), moose (42 km^2), black bear (101 km^2), and wolf (182 km^2) (Chapters 3 & 4).

Limitations

I used least-cost corridor and circuit theory approaches to map species-specific functional connectivity routes between suitable habitat patches within the A2L. Both methods used a raster resistance surface where each cell was attributed a value reflecting the energetic cost/difficulty of moving across that cell (McRae et al., 2008). The least-cost approach identifies potential routes of connectivity based on minimum resistance to movement between locations, but assumes the disperser has prior knowledge of the landscape (Pinto & Kiett, 2009; Fletcher & Fortin, 2018); whereas the circuit theory approach uses random walkers to model an overall resistance to movement and can account for path redundancies and non-optimal movement (McRae, 2006; McRae et al., 2008). To work within the confines of the available time and computer memory, I ran the Circuitscape software within the boundaries of the least-cost corridors (Shah & McRae, 2008; McRae, 2012). This allowed for the identification of pinch-points within the least-cost corridors that were important for maintaining connectivity, as well as measure the overall effective resistance of the corridor (McRae, 2012). However, doing so creates artificial boundaries (i.e., boundaries that exist on the resistance map but not on the ground). These artificial boundaries act as barriers reducing the number of paths linking the two habitat patches which can artificially increase the effective resistance (Koen et al., 2010). Use of least-cost corridor boundaries could also lead to pinch-points that could be exaggerated in locations where the corridors are narrow (Koen et al. 2010). One way around this issue is with the addition of a buffer (i.e., comprised of habitat data) around the artificial boundary giving random walkers more “room to walk” providing a more accurate measure of effective resistance (Koen et al., 2010).

Another limitation with our method was that I could not determine if changes in connectivity were due to changes in Euclidean distance (i.e., decrease in patch size or the complete loss of patches) or landcover change (i.e., decrease in habitat value and an increase in resistance value) between habitat patches. An alternative method that would have ruled out changes in connectivity due to changes in Euclidean distance would have been through the use of fixed nodes across the resistance surfaces of each time point. However, this method still wouldn’t aid in the distinction between changes in connectivity due to habitat loss and changes due to landcover change, and would have required computer memory beyond the limits of what was available.

Even with these subtle limitations I believe our results are entirely accurate. I did not rely on one measurement to determine changes in connectivity, but four different measurements (i.e. Euclidean distance, cost-weighted distance, least-cost path length and effective resistance) that all resulted in significant changes in functional connectivity within the A2L.

Conclusion

The results of this thesis suggest that extensive ecological restoration is necessary in the southern Québec and Ontario portions of the A2L to reclaim the habitat that has been lost in these regions.

Currie et al. (2023) came to similar conclusions in a nation-wide study. They suggested that transformed lands that would benefit the most from ecosystem restoration were clustered in southern Canada, with the greatest spatial extent of disturbance in croplands of the prairies, and in southern Ontario and Québec. The ecological restoration of degraded landscapes is so imperative to the conservation of biodiversity and human well-being that the United Nations General Assembly proclaimed 2021–2030 the United Nations Decade on Ecosystem Restoration (UN, 2020); and Target 2 of the Kunming-Montreal Global Biodiversity Framework requires that "at least 30 percent of areas of degraded terrestrial, inland water, and marine and coastal ecosystems are under effective restoration, in order to enhance biodiversity and ecosystem functions and services, ecological integrity and connectivity" by 2030 (CBD, 2024). Therefore, the most effective way to halt biodiversity loss within the A2L, and help mitigate the impacts of climate change, is to both significantly reduce the rate of anthropogenic land transformation and deploy large-scale habitat restoration projects (Banks-Leite et al., 2020). Degraded regions of the A2L provide an opportunity to implement nature-based solutions to restore ecosystems and re-establish large wilderness areas that are ecologically intact; and thereby enhance biodiversity conservation and climate-change adaptation and mitigation.

Conservation strategies to reduce fragmentation are also necessary. Roads represent a major driver of fragmentation of natural habitat for wildlife species (Bennet, 2017). Roads reduce habitat area and quality, increase wildlife mortality due to collisions with vehicles, prevent accessibility to resources, and subdivide wildlife populations (Jaeger et al., 2005), all of which can have wide-ranging implications for regional population dynamics, species diversity, and ecosystem function (Fahrig & Rytwinski, 2009). Therefore, mitigating the harmful effects of roads within the A2L constitutes a significant conservation priority. Mitigation measures such as road closures, prevention of new road development, limiting speed limits, wildlife crossing structures, fencing which can prevent wildlife from crossing a road and funnel them towards wildlife crossing structures (Rytwinski et al., 2016), and upgrading and widening of existing highways over construction of new highways at additional locations (Jaeger et al., 2011), would greatly reduce the harmful effects of roads on wildlife populations in the A2L.

Maintaining and/or restoring ecology connectivity, especially in the Ontario portion of the study area, will be vital to ensuring the functionality of the A2L transboundary wildlife linkage. Connectivity ensures access to additional resources when remnant habitat patches are too small for a single patch to sustain a species, and is essential when patches are larger, as movement between them decreases population extinction risk, facilitates re-colonization, and enables range shifts in response to climate change (Riva et al., 2024). Connecting protected areas within the A2L is also part of Canada's commitment to Target 3 of the KMGBF, which asserts that protected areas be "well connected" by 2030 (CBD, 2024).

Finally, protection of natural habitats in the A2L must be increased significantly. While protection is quite high in the New York portion, more protection is needed in the Québec portion and in the Ontario portion where protection is almost non-existent. The A2L is made up of many different ecosystem types. Some ecosystem types, such as wetlands, have been reduced considerably. Protection of native habitats across all ecosystem types is a prerequisite for effective biodiversity conservation (Riva et al., 2024). Protecting as much native habitat as possible is also key to safeguarding biodiversity (Valente et al., 2023). In 2015, the A2L consisted of 67,790 natural habitat patches, of which 59,589 were less than 1 km^2, and only 100 patches were over 100 km^2 (Chapter 2). Thus, under such circumstances the proverbial questions arise. What is better for conservation, land-sharing or land-sparing, or, is it better to protect some large or serval small (SLOSS) habitat patches? The answer in clear – both. Protecting both the remaining large habitat patches and the many small habitat patches within the working landscapes (Kremen & Merenlender, 2018; Riva et al., 2024) will be essential for protecting as much native habitat as possible in the A2L. While it is true that large habitat patches are vitally important for ecological integrity and must be protected (Haddad et al., 2015; Bateman & Balmford, 2023), it is also true that protecting multiple small habitat patches is critical for global conservation, particularly in human-dominated landscapes (Arroyo-Rodríguez et al., 2020; Riva & Fahrig, 2022). However, small patches are often considered less valuable than large habitat patches in less modified regions, which is inadvertently leading to widespread cumulative loss of habitat from millions of small patches (i.e., patches smaller than 100 km^2) across the globe (Riva et al., 2022). Similarly, focusing habitat protection solely on large habitat patches risks neglecting extensive areas of the planet with unique flora and fauna persisting in many small habitat patches surrounded by anthropogenic land uses (Haddad et al., 2015). Finally, failing to protect small habitat patches reduces landscape connectivity among larger patches due to the loss of stepping stone patches that can facilitate movement between larger patches (Riva et al., 2024).

The results of this thesis highlight the degree to which anthropogenic land transformation has impacted habitat amount, fragmentation, and connectivity in the A2L transboundary wildlife linkage between 1992 and 2018. These results suggest that to achieve long-term functionality of the A2L, collaborative and coordinated conservation actions must be initiated to preserve the integrity of the Québec mega-patch, restore extensive habitat in southeastern Ontario, and reestablish or maintain connectivity throughout the linkage. Left unaddressed, continued anthropogenic land transformation is likely to have detrimental effects on the ability of the A2L to function as a transboundary wildlife linkage.

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