The impacts of urban sprawling on ecological patterns and processes in the Montreal Metropolitan Region (Quebec, Canada) between 1966 and 2010.

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Abstract: Urban sprawl is a widely recognized phenomenon in many major cities worldwide and is a significant land use planning and management issue. This process has many impacts on the ecological function and structure of the landscape. In this article, we analyze the effects of urban sprawl on the ecological patterns and processes in the Montreal Metropolitan Region (MMR) between 1966 and 2010. The dispersed sprawl of low-density urban areas within the territory during this period sharply increased the fragmentation of the territory, isolating the few remaining natural spaces and decreasing their ecological connectivity and, ultimately, biodiversity. The results obtained clearly show that land-use changes that occurred in the MMR have caused profound changes in landscape properties, both structurally and functionally, and especially from 1981 to 2010. In 1966, around 45% of the land had a high or very high level of connectivity, and almost 38% in 1981. By 2010 only 6.5% of the landscape was connected and 73% of the territory possessed no or low connectivity.

Keywords: urban sprawl, land-use change, fragmentation, ecological connectivity, Montreal.

1. Introduction

Urban sprawl is a widely recognized phenomenon in many major cities worldwide and is an significant land use planning and management issue (Newman and Kenworthy, 1991; Williams et al., 2000; Grazi et al., 2008). During the last 50 years, urban and transport networks have spread at the expense of former natural or agricultural spaces, frequently occupying the lands most suitable for agriculture (Breheny, 1992; Camagni et al., 2002). In North America, this urbanization of areas around cities for residential, industrial, commercial and infrastructure use has followed a model characterized by a low density of built structures with a strong dependence on the automobile, which has revealed itself as tremendously negative for natural habits (Fahrig, 2003; Turner, 2005; Doucet, 2007). On the other hand, in western societies agriculture has survived and has been able to counter urban pressure mainly by the means of intensification. This is very clear in Europe (Mazoyer and Roudart, 2006) and is also noticeable in North America (Anderson, 2008; Parcerisas and Ruiz, 2014). This strategy of agricultural intensification, however, contributes to environmental degradation (Mazoyer and Roudart, 2006; Krausmann et al., 2013). Indeed, the increased crop yields caused by agricultural intensification have been frequently associated with substantial ecological costs, such as fossil energy inputs, soil degradation, and biodiversity loss (Krausmann et al., 2013; Dupras et al., 2015a).

Although urban built-up areas cover only a small proportion of the land, their impact on ecosystems is significant. For example, in the United States, roads occupy only 1% of the territory, but they highly alter the structures and ecological functions of at least 20% of the territory (Forman, 2000). In Europe, urban areas and infrastructures accounted for a little less than 3% of the whole territory in 2006, while agricultural and forested areas represented almost 71% of the land (EEA, 2013). Despite what these figures may lead to us to think, there has been an increasing and progressive process of European and

North American landscape degradation over recent decades due to uncontrolled urban sprawl, especially in the vicinity of large urban and coastal areas (Foley et al., 2005; Gerard et al., 2010).

Ecological landscape theory has provided a set of quantitative tools (namely landscape metrics) needed to characterize landscape heterogeneity (Li, 2000) and to measure landscape change through time (Reed et al., 1996). It is widely accepted that a general association exists between landscape pattern and ecological processes (Turner, 2005). Because of this concepts and methods from landscape ecology also are useful for land planning and design (Corry and Nassauer, 2005). Landscape metrics might be a way to evaluate the consequences arising from a given plan to manage a landscape's structure (Opdam et al., 2001), or they could be used to evaluate outcomes arising from alternative plans for a particular landscape (Gustafson, 1998). In either case, they are evaluative tools for regional planning (Botequilha and Ahen, 2002).

Landscape connectivity is a highly significant landscape attribute for conservation biology, as it is generally accepted that it enhances population viability and species richness at local and regional scales (Gilbert-Norton et al., 2010). Setting up habitat corridors is a classic structural approach to landscape connectivity management (Hobbs, 1992) that has been advocated as a key conservation strategy in human-modified landscapes where urbanization, infrastructure development and other activities frequently sever natural connections (Pino and Marull, 2012).

Manning et al. (2004) highlighted the limitation of corridor networks for the understanding and management of ecological functionality at landscape scale. A more general approach focused on ecological connectivity, integrates the value of remaining land matrix which might provide habitats for many species and enhance patch connectivity by providing a positive ecological context for patches of natural habitat (Ricketts, 2001). Consequently, some research proposes a network view which augments corridors with stepping-stone like structures of habitat distributed throughout the landscape (Pino and Marull, 2012).

The important outcome of urban sprawl is the fragmentation of natural and semi-natural habitats, which is the isolation of the different parts of the territorial matrix and that, ultimately, can bring about long-term loss of biodiversity (EEA, 2011). Mitigating the effects of fragmentation requires re-establishing connectivity across the territory and treating it as a functioning whole (Loreau et al., 2003; Gonzalez et al., 2011). Emphasis on connectivity challenges the idea that protecting a number of isolated natural and semi-natural spaces will be sufficient to maintain the ecological integrity of the region (Pino and Marull, 2012). High levels of fragmentation resulting from urban sprawl can move a region past thresholds of connectivity that make restoration politically challenging and economically costly. For example, Marull and Mallarach (2005) showed that the artificial barriers that cover 18% of the Barcelona Metropolitan area have direct negative impacts on 57% of the ecological connectivity of the area.

The goal of this article is to analyze the impact of urban sprawl on the ecological patterns and processes of the Montreal Metropolitan Region (MMR) from 1966 to 2010. Several landscape metrics, like the Effective Mesh Size and Shannon indexes, and the Ecological Connectivity index were calculated from land cover maps of the area of 1966, 1981, and 2010. This last index (ECI) has been recently developed and has already been successfully applied in some European (Parcerisas et al., 2012; Marull et al., 2010, 2014) and North-American (Dazzini, 2007; Marull and Cunfer, in press) cases. After a brief presentation of the study area in Section 2, we present the methodology in Section 3 and results in Section 4. First, land-use changes during the period analysed are detailed, then a number of landscape properties are analyzed and, finally, the evolution of the ecological connectivity during the time frame is assessed. We discuss the results in section 5 before presenting our conclusion.

2. Study area

The MMR is located southwest of the Province of Quebec following the Saint-Lawrence River, comprising a total of 82 municipalities and covering an area of 4260 km_2 (Fig. 1). The core of the MMR is the City of Montreal, situated on the Island of Montreal, which is the most populated city in the province and second in Canada following Toronto, with a total population of 1,649,519 inhabitants in 2011 (Ville de Montréal, 2013).

As seen in Table 1, between 1966 and 2011 the population of the Province of Quebec has increased 37% while population growth rate within the MMR has been of 49%, though in an irregular fashion within the territory over time. The population of the island of Montreal has followed a different path, showing a clear standstill, even a decrease until the 1990's, largely due to the migration of urban residents to the suburbs (Sénécal et al., 2001). Therefore, it may be established that the population boost in the Province of Quebec during the last decades mainly occurred in the MMR with a dispersion of population within the MMR, especially between 1996 and 2010, when growth rates were higher.

On the other hand, the urban area in the MMR has spread along the territory at a much higher rate than the population, more than doubling the surface area occupied in 1966. Despite the fact that the population increased by 49% during this time, urban spaces in the metropolitan area grew by around 119%, passing from 610 km2 to 1340 km2. The result has been the creation of low-density dispersed towns. The process of migration to the suburbs started in the 1950s, provoking the construction of infrastructure and transport networks. Despite the plans adopted by the local governments in 1978 to stop this process and protect agriculture and agroforested spaces, since the 1990s, urban pressure on agricultural land heightened, resulting in a new period of agricultural abandonment and speculation (Dumoulin and Marois, 2003; Dupras and Alam, 2015; Nazarnia et al., 2016).

3. Methodology

3.1.Conceptual approach

The land matrix – and the landscapes it contains – can be seen as a heterogeneous, dynamic and multi-scalar system organized in hierarchical levels of complexity depending on their scales of space and time. In order to understand the organization of this complexity, and its evolution, we believe it necessary to use a systems approach that takes into account the main factors that typify landscape patterns in a global and integrated manner. This approach, which is used in quantitative landscape ecology, allows us to turn current theories concerning the biophysical matrix into useful tools for sustainable metropolitan and regional land-use planning (Mallarach and Marull, 2006).

We used the landscape continuum model as a starting point (Fischer and Lindenmayer, 2006), where the land matrix has been defined as the outcome of the interrelationship between the biophysical matrix and the changes made by human activity. In the planner's perception it becomes the space that he or she intends to modify in order to generate the land matrix of the following transformation. To this end, a model of the ecological connectivity between elements of the landscape is considered (Marull and Mallarach, 2005). In this way, the land matrix is formally defined, and the parametric method operates by means of successive iterations in order to assess the effect that different land use plans or corrective measures may have on the underlying biophysical environment.

We consider the whole land matrix as a dynamic system:

$$T = F(X) = \{V \text{ is open of } T, \subset X\} \qquad (1)$$

where X is the land matrix, that is to say the total land area under study. T is taken to be a discreet topology: every subset (V)—e.g., a polygon- of X is open to flows in the system of T. The area defined in this way is continuous and quantifiable. Therefore the formal expression of X starts off from the gathering of all the points (pi) in a scope of a given study area:

$$X = \cup\, p_{i;}\, i \in I \qquad (2)$$

Our method relies on a series of topological analysis of land use and infrastructures maps, it has been entirely formalized mathematically, and it has been developed and implemented using GIS. Depending on the objectives of each analyses, we choose raster or vector formats from the ArcGIS program in all the stages of the project.

The following sections present the three key components of this methodology: the ecological functional areas, the Barrier Effect Index (BEI), and the Ecological Connectivity index (ECI), including three variations of this last index that can be used for different applications.

3.2. Ecological Functional Areas

The ECI uses a top-down, GIS-based approach to produce a cost distance map on the ecologically functional areas, which were considered the focal habitat patches to be connected. For the identification and mapping of these areas we used the following protocol: based on a land-cover map, we performed first a topological analysis grouping the original land-cover categories (Table A1), depending on its affinity, in simple ecological functional areas (Table A2). We defined these simple ecological functional areas by applying a minimum surface on each land use type (Marull and Mallarach, 2005). The areas that could not be considered simple ecologically functional areas were used to perform a second topological analysis, grouping them in agro-forest mosaics (Table A2). In this case, we applied the same criteria of minimum surface. The remaining areas were considered fragmented areas. We decided that ecologically functional areas should include mosaics, strong correlations exist between habitat heterogeneity and biodiversity (e.g., Heikkinen et al., 2004; Inger et al., 2014).

We then use these digital maps to calculate three landscape metrics: land cover richness index, Shannon index and effective mesh size index. The Land cover richness index, refers to the number of different patch types per cell. The more land use cover classes there are, the more diverse the study site is. Land cover richness is used to calculate the Shannon Index (Shannon, 1948), which quantifies the diversity of the landscape based on two components: the number of different patch types and the proportional area distribution among patch types. Commonly the two components are named richness and evenness. Richness is the compositional component and evenness, which refers to the area distribution of classes, is the structural component.

$$\text{Shannon} = \bullet\ (P_i \times \ln P_i) \qquad (3)$$

where Pi is the proportion of land matrix occupied by each type of cover.

The threat to biodiversity caused by landscape fragmentation can be assessed by the Effective Mesh Size (EMS) Index (Jaeger, 2000) of the network created by the fragmented elements present in the landscape. The EMS measures structural landscape connectivity, as it defines the probability that any two points chosen randomly in a region are connected; that is, not separated by barriers such as transport routes or built-up areas; the more barriers fragmenting the landscape, the lower the probability that the two points are connected, and the lower the effective mesh size. This index is a good indicator to assess fragmentation and suitable for comparing fragmentation among different regions with different characteristics (Jaeger, 2000; Girvetz et al., 2008).

$$\text{EMS} = (\bullet\ (P_i^2) / \bullet\ (P_i)) \times 1000 \qquad (4)$$

where Ai is the area of each polygon (every subset—V of X). What follows next is a presentation and discussion of two indices we defined to deal with ecological landscape connectivity: BEI and ECI.

3.3. Barrier Effect Index

Barriers include all artificial land uses that create obstacles to the flow of energy, information, or matter across the matrix, in other words, the landscape resistance. To calculate the effects of artificial barriers on ecological and landscape connectivity we defined a barrier effect index (BEI) as follows:

$$BEI = Y_i / Y_{max} \quad (5)$$

Where Y_i is the value of the barrier effect in a pixel and Y_{max} is the maximum value of the barrier effect calculated on a given area.

The BEI is based on the weight that we assigned to each barrier type (Table A3), the affected land use class, and the distance from the barrier, according to a potential impact matrix (Table A4) and a logarithmic relationship with distance. Thus, it reflects an impedance surface, where ai corresponds to the maximum significantly affected distance for each type of barrier, and Ai corresponds to the potential impact value for each type. Since all these weighs are expert-based, we decided to assess its significance on the results of the ECI, by performing a sensitivity analysis for a random variation (+_0.3), to establish that the affect on the new ECI was negligible.

This model applies the cost–distance function using the grid module of the ArcGIS program and uses two databases: one origin surface (X_{Bs}; s = 1 - n) for each barrier type (Bs) (and one impedance surface (X_A) from the potential impact matrix (M_A). From this process, an adapted cost distance is obtained ($d_{0s} = b_s - d_s$; where $b_s - d_s > 0$; being d_s = cost distance). In this way, our model individually calculates the barrier effect Ys for each subclass type. Based on the literature (Marull and Mallarach, 2005) we assumed that the effect of a single barrier from a given point is logarithmic and decreasing as distance increases, according to the following expression.

$$Y_s = b_s - k_{s1} \ln(k_{s2}(b_s - d'_s) + 1) \quad (6)$$

where b_s is the weight of each barrier type, k_{s1} and k_{s2} are constants for logarithmic decreasing function, and d'_s is the adapted cost distance per barrier type.

Constants k_{s1} and k_{s2} are needed for adjusting the shape of the function to a specific logarithmic fall. An important procedural GIS aspect needs to be pointed out here. Since the barrier effect must have decreasing values, the calculated cost distance values need to be inverted, subtracting from b_s, and then one has to truncate the resulting values to 0, to avoid the appearance of negative values, which would be meaningless.

Thus, the entire barrier effect Y in the landscape is defined as the addition of the effects of all barrier types on a given area. The reason for this is that a combination of different barrier types, such as highways, railways and urban areas, has a potential effect much greater than the maximum impact of each individual type of barrier. In other words, it is a way of taking into consideration cumulative impacts. From this process, a barrier effect surface (X_Y) is obtained.

$$Y = \sum_{s=1}^{s=n} Y_s \tag{7}$$

BEI is a relative index, which means that for each given area where it is applied it has been designed to give values within an ordinal scale from 1 to 10. The reason for this is twofold. First, barrier impact cannot be easily measured in absolute terms, being always a function of the diverse natural systems or landscapes that are affected and the organisms that move within it. Second, grouping the high variability of continuous BEI potential values using a conventional discrete decimal scale helps in making interpretations and comparisons easier.

3.4.Ecological Connectivity Index

Ecological connectivity refers to the potential connection between the different elements of the landscape, from energy to information and matter; for instance, fluxes can include nutrients, pollen, and movements of flora or fauna within spatially structured populations and communities (Gonzalez et al., 2011). We defined an ecological connectivity index based on a cost–distance model that considers the different functional ecological areas (Table A2) and an impedance surface, which incorporates the barrier effect Y and a potential affinity matrix M_C for all the land use types (Table A5). This matrix includes the potential affinity range of values that we assumed are reasonable to expect among the different types of ecological functional areas in our study area (Marull and Mallarach, 2005).

Since all these weights are expert-based, we decided to assess its significance on the results of the ECI, by performing a sensitivity analysis for a chance variation of all MC weights of +_0.3, resulting in a negligible impact on the new ECI. In the calculation of ECI the barrier effect surface X_Y is very significant. However, among the areas where the barrier effect is minor, the surface of the affinity matrix X_C reveals interesting nuances that fully justify their elaboration.

The model applies the Cost-Distance function of the ArcGIS program, using two databases: one origin surface for each type of ecological functional area (XC'r; r = 1–n) and one impedance surface resulting from the application on the effect of the barriers and the potential affinity matrix ($X_I = X_C + X_Y$). In this way, we obtain an adapted cost distance by each type of ecological functional area (d'_r • 20,000 in order to avoid distorted results when several classes are combined). Finally, we calculate the total adapted cost distance value x for all the ecological functional area types, according to the following expression:

$$x = \sum_{r=1}^{r=n} d'_r \qquad (8)$$

To facilitate interpretation and comparisons, we decided to transform the continuous values of the cost distance to discrete values based on a decimal scale. Also, we decided to use a natural logarithm to emphasize high values, because low values are associated with more artificial surfaces, having less interest from the point of view of this index. Therefore, we define the ecological connectivity index as follows:

$$ECI = 10 – 9 \ln(1+(x_i - x_{min})) / \ln(1+(x_{max} - x_{min}))^3 \qquad (9)$$

where x_i is the adapted cost-distance value in a pixel, x_{max} are the maximum and x_{min} are the minimum adapted cost-distance values on a given area.

We consider that this index reflects a kind of general ecological connectivity, since its computation includes all the ecological functional areas C'. Thus, it is a generic approach which is not tied to specific indicator species. An interesting propriety of the ECI is that it has a relative distribution of values, always giving values between 0 and 10. However, the general ecological connectivity index cannot be used for comparing different geographical areas or different time periods in the same area.

However, in a particular case, when $x_{min} = 0$, $x_{max} = x_t$, we obtain a variation of the ECI, that we named basic ecological connectivity index (ECI_b). Therefore, we can define:

$$ECI_b = 10 - 9 \ln(1 + x_i) / \ln(1 + x_t)^3 \qquad (10)$$

where x_t is the maximum possible adapted cost–distance value. ECI_b is useful for calculating the ecological connectivity of different geographical areas, time periods, or ecological functional areas C'_r. Values of ECI_b vary between 1 and 10.

Finally, from this Basic Ecological Connectivity Index (ECI_b) we can derive another particular application, that we named the Absolute Ecological Connectivity index (ECI_a), which implies the addition of all ECI_b that had been calculated for the study area, according to the following expression:

$$ECI_a = \sum_{m=1}^{m=n} ECI_b / m \qquad (11)$$

where m is the number of ecological functional areas C'r considered. Values for ECI_a are more objective, in general, giving values lower than 10, and are very useful for comparing different territories or different temporal series, as well as for providing directions for regional and land use planning (future scenarios).

4. Results

4.1. Land-Use Changes between 1966 and 2010

Around the mid-twentieth century, the MMR presented quite a diversified landscape around the urban core representing the City of Montreal. Outside the city, a continuous ring of agriculture and natural spaces was found. The entire MMR held a high density of farms, more than 150 of farms per km^2, practising traditional crop- farming mixed with cow-farming (Parcerisas and Ruiz, 2014). The north side of the Saint Lawrence River was composed of less intensive agriculture than on the south side, where agriculture occupied around 60% of the total area, mainly dedicated to hay and oats. On the north side, a greater area was dedicated to forest uses even inside the farmlands (Parcerisas and Ruiz, 2014).

This diversified landscape can still be appreciated in the land-use map of 1966 (Fig. 2), when croplands were the main land cover in the whole region, representing 40% of the total area (Table 2). In 1981, the process of urban sprawl had already begun and could be felt, representing a quarter of the total area. Within the farmed area, dairy-farming was losing importance and traditional crops such as hay and oats were the most important (Parcerisas and Ruiz, 2014). However, some farms, especially in the south-east, were part of the corn cropping boom found in the entire province of Quebec. This corn boom began in the 1970s and was associated with the pork industry (Parcerisas and Ruiz, 2014).

The landscape in 2010 had changed considerably. Developed land had grown at an average rate of 3% per year (1880 ha per year) from 1981, becoming practically the only land-use on the Island of Montreal and Laval. Moreover, it had invaded large parts of the former natural-space ring. Indeed, the cost of this urban-sprawl was the large reduction in natural spaces: grasslands have decreased by more than 50%, forests represent only 10% of the total area, and agriculture now represents 27% of the territory. In 2010, the urban area was larger than the sum of the total natural spaces and agro-forestry mosaics in the region. Agriculture, mainly in the south side, could only face urban sprawl by intensification. Although hay and oats are still the main crops in the region by surface area, corn and soya cropping have spread, destined to feed the increasing pork sector (Parcerisas and Ruiz, 2014)

4.2. Landscape properties

As expected, landscape metrics show a decrease in their values during the period studied (Table 3). In general, higher values, showing better ecological state of the land matrix, and are apparent as we move away from the core of the Island of Montreal (Fig. 3). However, the MMR natural and semi-natural areas have continued to deteriorate as a result of urban sprawl.

Land cover richness index has remained stable for the entire region. However, when looking at the map (Fig. 3), one can see how there has been an important decrease in certain areas of the ring around the Island of Montreal, especially on the north side. In 2010, cells in the map showing richness values of 6 and 7 are hard to find, contrary to 1981 or 1966. Shannon index is lower in 2010 than in the past, especially compared to 1981. This means there is less landscape diversity in 2010. Again, when looking at the map (Figs. 3 and 4), the high values existing in the peri-urban area of Montreal and Laval Island in 1981 are no longer found in 2010. Mesh size values have decreased through time showing that fragmentation occurred in the landscape matrix. Only some small parts of Saint Lawrence River in the southern region show high values as in 1966 and 1981.

Ecological Connectivity Indices (ECI) have sharply decreased between 1966 and 2010 due to the impact of the building of the transport network and, to the urban development of low-density scattered suburbs. However, the largest loss was produced from 1981 onwards (Figs. 4 and 5). It is remarkable to note that, despite the process of migration to the suburbs and the process of industrialization, the decline of agricultural activities had already begun decades earlier. In 1981, the MMR still possessed a high ecological connectivity, similar to 1966. The allocation of increasing population since 1981 to new urban developments scattered throughout the region, and the construction of new highways and roads to connect them, acted as new barriers fragmenting the territory to such a degree that the connectivity was practically erased. In 1966, around 45% of the land enjoyed a high or very high level of connectivity, and almost 38% in 1981. On the contrary, only 6.5% held this consideration in 2010 with up to 73% of the territory possessing no or low connectivity (Table 4). As we can see in Fig. 5, big negative changes in connectivity levels have occurred in the former natural-space ring around the Island of Montreal, both north and south. Today, acceptable levels of connectivity are maintained only in the small surviving agro-forestry areas of the peri-urban ring of the MMR.

ECI shows a significant decline between 1966 and 2010 due to the loss of forested lands, but also agricultural and agro-forestry mosaics (Fig. A1). Urban sprawl leading to loss of landscape diversity and ecotones has had very significant environmental impacts by isolating the remaining forested lands. Some critical areas have been detected for the preservation of the potential ecological connectivity between natural areas and the remaining agricultural mosaics in the MMR. The former have become increasingly isolated from the latter due to the spread of urban development, together with the barrier effect of linear infrastructures.

5. Discussion and conclusions

Our analysis clearly show 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. While in 1966, the MMR was mainly a mosaic of agriculture and remnant forest with an important urban and industrial center, today it is an urban region where urban spaces occupy most of the area and where agriculture and forest are now remnants.

Although migration from the Island of Montreal to the suburbs in the 1950s caused highway construction and industrialization of agriculture, the MMR's territory in 1981 still showed high levels of landscape heterogeneity and ecological connectivity, with even some improvement with respect to 1966. Disorganized urban sprawl still did not show its entire effects on the landscape, and the structuring of the territory and agricultural protection that began in 1978 must have helped this situation. However, since 1981, and despite governmental efforts to protect agricultural areas and prevent urban sprawl, there has been a large decrease in ecological connectivity in the entire MMR due to the high fragmentation of the territory produced by uncontrolled urban sprawl.

Lessons can be learned from this study. First, protecting spaces but allowing their isolation from the rest of the territory via the construction of barriers will decrease ecological connectivity, with likely impacts on remnant biodiversity. Our study shows this very clearly (Figs. 4, 5 and A1). Today, despite their protection, remaining forest and agriculture show very low levels of ecological connectivity because they have become isolated due to traditional rural abandonment, the increasing construction of transport networks and scattered urban villages. This loss of ecological connectivity through agro-forest mosaics also stresses the importance of natural corridors, like riparian zones, in maintaining the ecological processes within the landscape.

A recent review by Tscharntke et al. (2012) stresses that spatial heterogeneity creates ecological dissimilarity that determines landscape-wide biodiversity. Spatial spillover effects through the movement of organisms and resources across habitats can facilitate their persistence and adaptation in human-managed landscapes. The combination of landscape heterogeneity with ecological connectivity in metropolitan areas and urbanized regions enables spatial and temporal insurance, providing higher stability and resilience to ecological processes (Loreau et al., 2003; Gonzalez et al., 2009). Hence, a wildlife-friendly agro-ecological matrix can be seen as a useful strategy to enhance biodiversity.

Secondly, the policies applied by regional and provincial governments to protect agricultural land have not been successful. Indeed, in spite of these policies, agriculture has not been able to resist urban sprawl arising from the real-estate boom in recent decades. The result has been the abandonment of agricultural activities and urban sprawl into agricultural land, sometimes of high quality. The reason can be found in the progressive fall of agricultural land prices and income since the 1950s in western societies (Federico, 2008). Faced with the situation of progressive impoverishment despite the subsidies they receive, farmers, and especially their descendants, prefer to sell their land to urban developers than to keep farming, which cannot equal the rapid income offered by the latter. Land managers and decision makers should understand that they are losing agricultural landscapes that provide diverse ecosystem services. In addition to fiber and food, these landscapes maintain biodiversity, they recycle nutrients, regulate local microclimates, regulate local hydrological processes, affect the abundance of undesirable organisms and detoxify chemical harmful substances (Alam et al., 2014). The urban sprawl model followed over the last decades, based on low-density urbanization and a transportation network based on the use of the car has had a tremendously negative affect on the ecological quality of the land matrix, as well as affecting its long-term sustainability.

Recently several major cities in the world have adopted green infrastructures policies, including greenbelt and green networks, to manage the growth of urban areas and protect natural and agricultural areas (Taylor et al., 1995; Bengston et al., 2004). These green infrastructures are known to act positively on the ecological connectivity of metropolitan landscapes (Opdam et al., 2006). A green infrastructure project is currently being considered in the region of Montreal (Dupras et al., 2015b, 2015c) and if appropriately designed could mitigate the loss of connectivity we found in this study (Gonzalez et al., 2014). Adequate management for ecological connectivity will require a network of protected areas, but also the re-establishment of ecological connectivity via target restoration of high quality habitat and ecosystems.

The methodology employed in this study to assess land matrix fragmentation, landscape heterogeneity and ecological connectivity in metropolitan regions can reveal spatial processes, such as urban sprawl, and assess pressure on biodiversity from urbanization. Marull and Mallarach (2005) developed the Ecological Connectivity Index (ECI), which addresses landscape connectivity (sensu Lindenmayer and Fischer, 2007). The quantitative and cartographic approach adopted by ECI facilitates the communication of research to planners and policy makers. In addition, successive iterations can be used to check the impact of different alternative planning scenarios. We believe that any future policy for landscape protection and management in the MMR could make use of this approach to evaluate whether targets for restoration of ecological connectivity have been met.

It is expected that large increases in demographic growth (i.e., 530,000 additional people by 2031 according to CMM, 2011) and increased tourism in the coming years will lead to further pressures on the territory with the construction of new built-up areas, which will further fragment the land matrix. If we wish to preserve the remaining few natural agricultural spaces of the MMR and their biodiversity, there is an urgent need for policies that facilitate the protection and restoration of heterogeneous and ecologically connected landscapes.

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Table 1. Evolution of population and urban area, 1966-2011

year	Québec		MMR			Island of Montreal			MMR Urban area	
	inhab.	*1966 = 100*	*inhab.*	*1966 = 100*	*% Québec*	*inhab.*	*1966 = 100*	*% MMR*	*ha*	*1966 = 100*
1966	5.780.845	100	2.570.985	100	44,5	1.923.171	100	74,8	61.058	100
1981	6.438.403	111	2.862.286	111	44,5	1.760.120	92	61,5	77.529	127
1996	7.138.795	123	3.326.447	129	46,6	1.775.788	92	53,4	116.100	190
2011	7.903.001	137	3.824.221	149	48,4	1.886.481	98	49,3	133.926	219

Source: Ville de Montréal (2013), Statistics Canada (1971) and Dupras and Alam (2014).

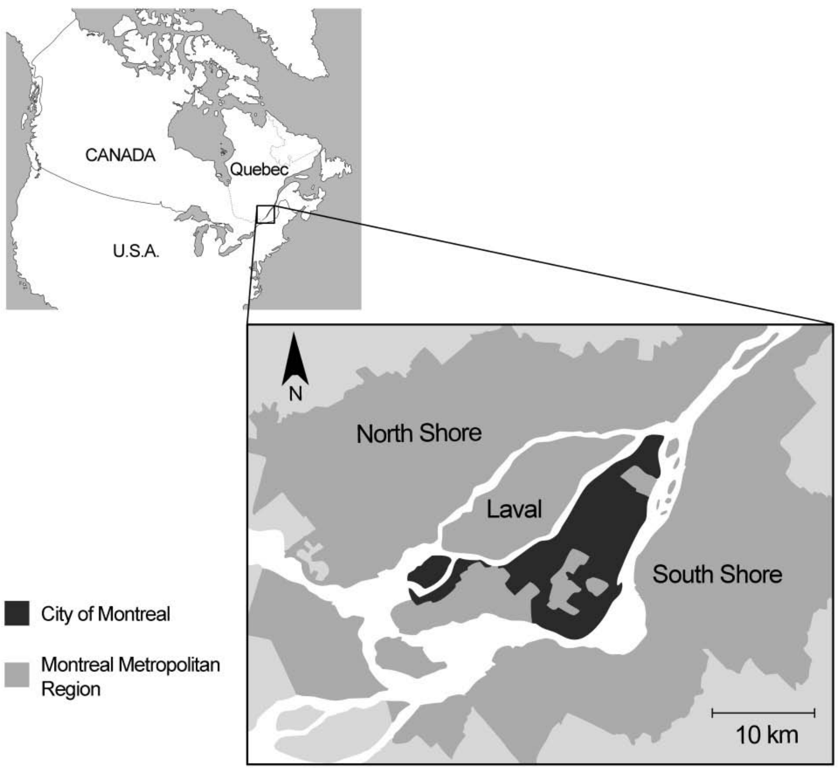
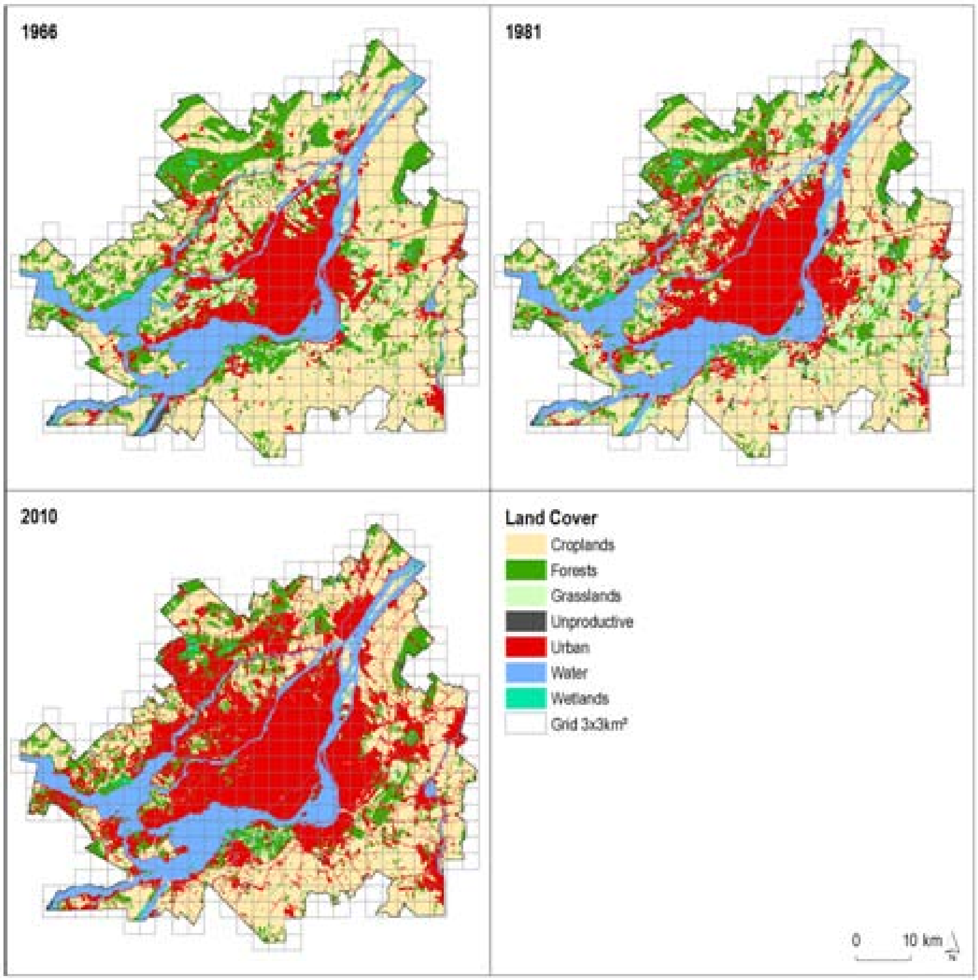


Fig 1. Geographic location of the Montreal Metropolitan Region.



Source: Dupras and Alam (2014).

Fig. 2. Land cover change in the MMR (1966-1981-2010).

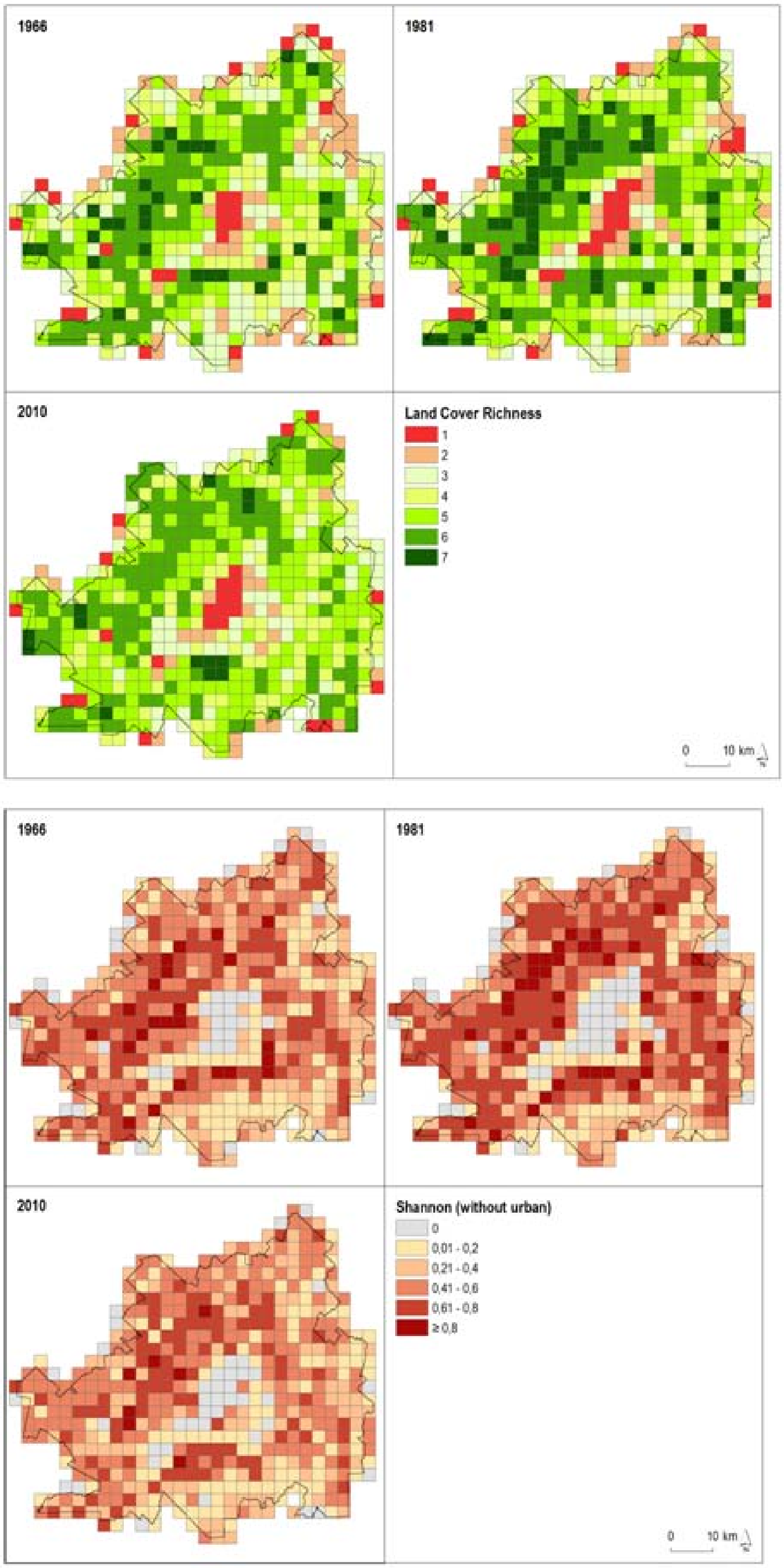
Table 2. Land uses in the MMR (1966-1981-2010).

	1966		1981			2010		
	ha	%	ha	%	1966 = 100	ha	%	1966 = 100
Croplands	124.421,0	40,1	102.592,1	33,0	82,5	85.237,6	27,5	68,5
Forests	51.027,1	16,4	39.970,7	12,9	78,3	31.014,4	10,0	60,8
Grasslands	23.063,1	7,4	36.895,2	11,9	160,0	10.422,1	3,4	45,2
Water	46.738,8	15,1	47.006,4	15,1	100,6	47.665,8	15,4	102,0
Wetlands	2.384,7	0,8	1.909,6	0,6	80,1	1.885,6	0,6	79,1
Unproductive	1.807,5	0,6	4.596,7	1,5	254,3	348,7	0,1	19,3
Urban	61.057,8	19,7	77.529,1	25,0	127,0	133.925,7	43,1	219,3
Total	310.499,9	100,0	310.499,9	100,0		310.499,9	100,0	

Table 3. A comparison of Landscape metric values from 1966-2010.

Landscape metric	1966 (A)	1981 (B)	2010 (C)
Land Cover Richness (nº)	4.72	5.03 AC	4.70
Shannon Index	0.44 C	0.50 AC	0.39
Effective Mesh Size (km^2)	3.85 BC	2.96 C	2.23
Ecological Connectivity Index	4.19 BC	3.77 C	2.29

Note: The results are based on two-tailed tests assuming equal variances with a significance level of 0.05. For each significant pair, the key under the category (A-C) shows up beneath the category with a mayor average value. Using the Bonferroni adjustment, tests have been adjusted for all pair wise comparisons.



Source: Our own data.

Figure 3. Land Cover Richness and Shannon Index (without urban area). Territorial distribution expressed by 3x3km^2 cells.

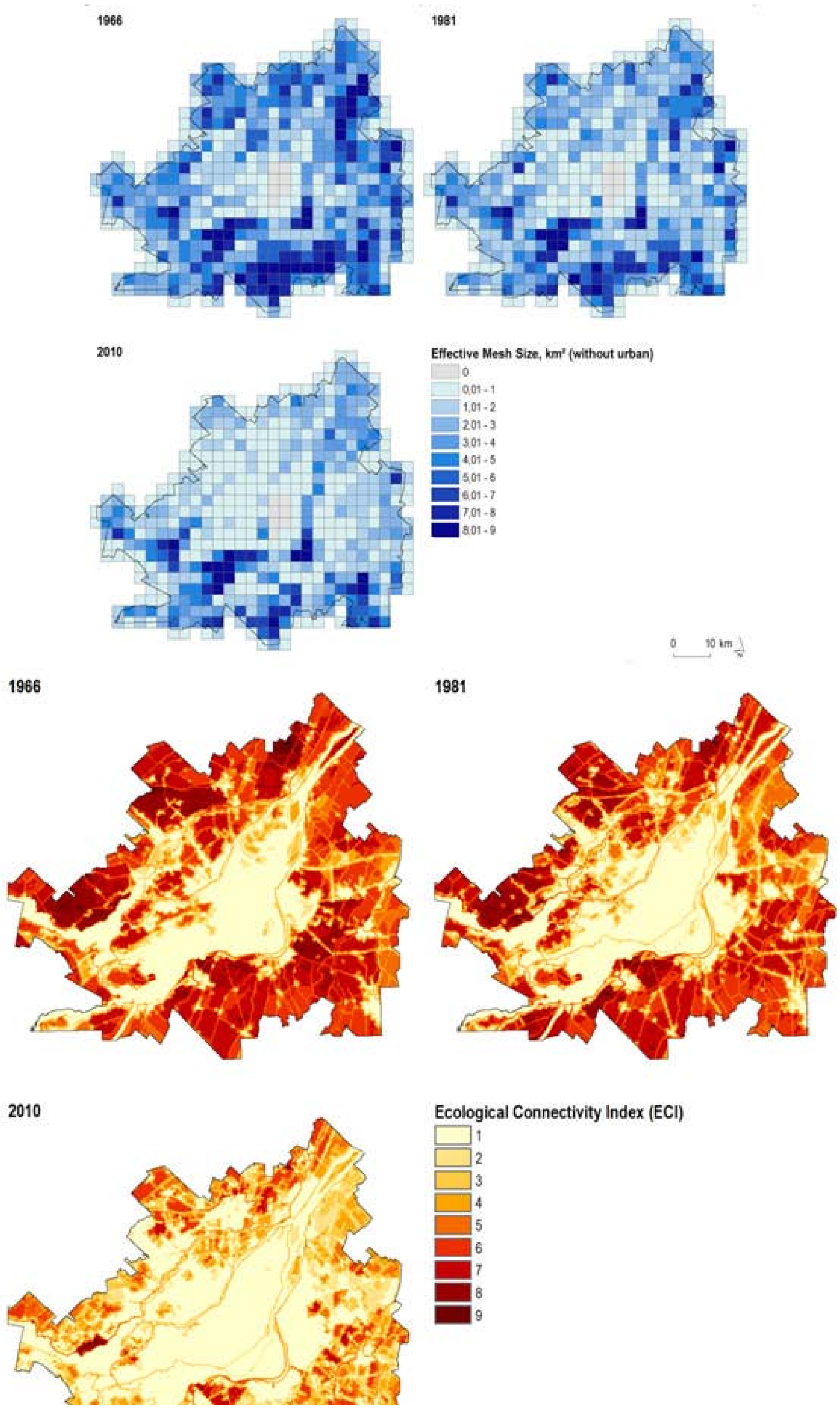


Figure 4. Effective Mesh Size Index (without urban area) and Absolute Ecological Connectivity Index. Territorial distribution expressed by 3x3km^2 cells.

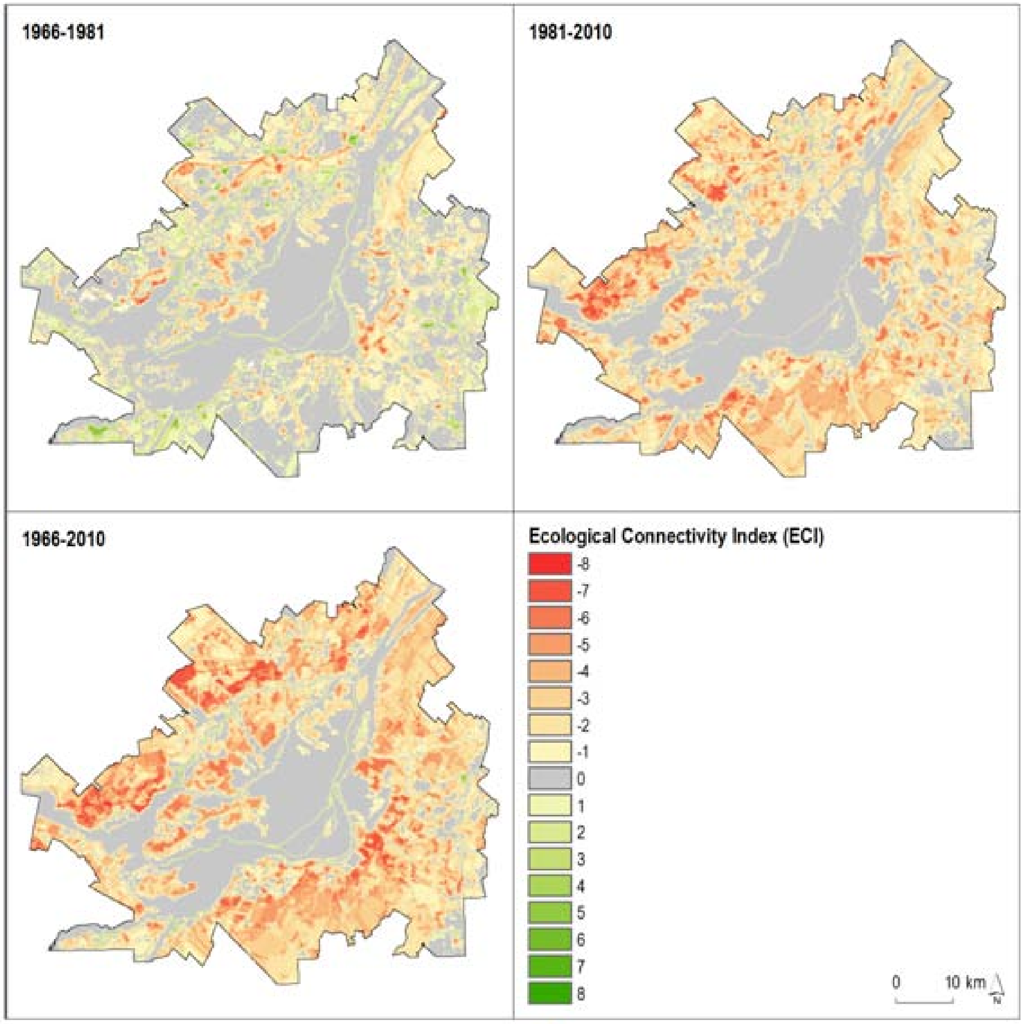


Figure 5. Change in the Absolute Ecological Connectivity Index during three distinct three periods over the last 50 years.

Table 4. Results of the application of the Absolute Ecological Connectivity Index (ECIa) (1966, 1981, and 2010).

ECI_a	Category	1966		1981		2010	
		ha	%[a]	ha	%[a]	ha	%[a]
1	No connectivity	94,133	24.41	106,259	27.55	168,843	43.78
2	Low connectivity	24,601	6.38	28,716	7.44	54,139	14.04
3		24,220	6.28	29,326	7.60	57,722	14.97
4	Medium connectivity	27,456	7.12	32,006	8.30	49,181	12.75
5		40,862	10.59	44,153	11.45	30,447	7.89
6	High connectivity	72,883	18.90	59,721	15.48	19,401	5.03
7		69,873	18.12	63,622	16.49	4,849	1.26

8	Very high connectivity	28,757	7.46	20,885	5.41	959	0.25
9		2,926	0.76	1,034	0.27	164	0.04
10		0	0	0	0	0	0
	Total	385,711	100	385,721	100	385,706	100

Note: [a] ECI level percentage for each time point.

Appendix 1. Ecological Connectivity Index calculation tables (1966-1981-2010)

Table A1 Land Cover categories

Land Cover	
C_1	Forest
C_2	Wetland
C_3	Cropland
C_4	Grassland
N_1	Unproductive
B_1	Urban
B_3	Water
R_1	Corridor[a]

[a] Updated by rivers cartography.

Table A2 Ecological Functional Areas (1966-1981-2010)

Code	Land Cover	Sr (ha)	1966		1981		2010	
			ha	%	ha	%	ha	%
C'_1	Forest	50	70.664	83,8	46.854	80,2	30.809	67,9
C'_2	Wetland	50	3.020	45,7	838	36,5	835	37,0
C'_3	Cropland	50	164.523	98,1	133.277	95,3	112.141	91,5
C'_4	Grassland	50	25.136	58,3	30.840	71,6	2.721	22,8
M'_1	Agro-forest mosaic	50	29.261	31,6	9.248	26,9	10.341	27,2

Table A3 Basic Barrier types (B_s)

Code	Type	Weight (b_s)	ks_1[a]	ks_2[a]
B_1	Urban areas	$b_1 = 80$	$k1_1 = 44.420$	$k1_2 = 0.063$
B_2[c]	Main communications	$B_2 = 40$	$k2_1 = 22.210$	$k2_2 = 0.126$
B_3	Water	$b_3 = 60$	- b	- b
[a] Constants for a logarithmic fall of 30% (• = 0.3)			• = $Y_s (b_s /2)/b_s$	
[b] For s = 3 there is not surrounding spatial affectation			$Y_3 = b_3$	
[c] Used as barrier (intersected by rivers)				

Table A4 Impact Matrix (M_A) for the calculation of the Barrier Effect Index

Code	Type	Classes included [a]	Affectation coefficient (a_l) [b]	Affectation value (A_1)
V_1	Neutral	N_1	a_1 =1000 m	A• $_1$ = 0,10
V_2	Agriculture	C_3, C_4	a_2 =750 m	A• $_2$ = 0,13
V_3	"Natural"	C_1, C_2	a_3 =500 m	A• $_3$ = 0,20
V_4	Barrier	B_1, B_3	a_4 =250 m	A• $_4$ = 0,40
V_5	Corridor	R	a_5 = $_1$ m	A• $_5$ = 100
[a] Class description in table A1 [b] A_1 defines the maximum significantly affected distance by each type				($A_n = b_5 / a_n$)

Table A5 Affinity Matrix (M_C)

Code	Land Cover	C'$_1$	C'$_2$	C'$_3$	C'$_4$	M'$_1$
C_1	Forest	0	0,1	0,6	0,4	0,4
C_2	Wetland	0,1	0	0,5	0,3	0,4
C_3	Cropland	0,6	0,5	0	0,2	0,4
C_4	Grassland	0,4	0,3	0,2	0	0,4
B_1	Urban areas	1	1	1	1	1
B_3	Water	1	1	1	1	1
R	Corridors	0.1	0.1	0.1	0.1	0.1
N	Unproductive	0.9	0.9	0.9	0.9	0.9

Fig. A1. Comparison of Basic Ecological Connectivity Index (ECIb) for each ecological functional area (1966–1981–2010).