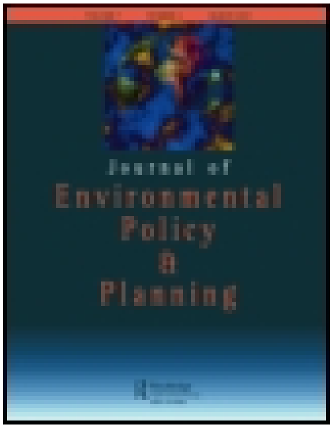

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Urban Sprawl and Ecosystem Services: A Half Century Perspective in the Montreal Area (Quebec, Canada)

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ABSTRACT *Urban sprawl is central to the issues surrounding sustainable urban development. It generally leads to multiple impacts on land-use change, including loss of sensitive natural areas, farmland and fragmentation of ecosystems, which negatively impact the production of a wide range of ecosystem services (ES). In this study, we evaluate the value of ES provided by forests, croplands, grasslands and wetlands. Four spatial analyses of the Montreal Metropolitan Region (Quebec, Canada) are used over a period of 45 years at 15 year intervals (1966, 1981, 1994 and 2010). We demonstrate that despite a variety of management strategies, urban sprawl continues to have negative impacts on ES economic value over time.*

KEY WORDS: urban sprawling, ecosystem services, economic valuation, North America

1. Introduction

The issue of urban sprawl is central to the challenges surrounding sustainable urban development. Although a universal definition for urban sprawl does not exist, most include factors related to: increased competition for land use, automobile transportation, growth at the periphery of city limits and the difficulty for policy-makers to establish common guidelines (Johnson, 2001). Since the Second World War, urban sprawl in North America can generally be defined as a process of suburbanization running with a continuous, but variable intensity and characterized by a high dependence on automobile use and low-density space occupation (Rothblatt, 1994). This leads to a strong demand on land use (Johnson, 2001) and eventual land-use conversion. Ecosystem services (ES), like food and fibre production, pollination, air purification and outdoor recreation (Kremen, 2005; Metzger, Rounsevell, Acosta-Michlikb, Leemans, & Schröter, 2006) are affected by these land-use pressures. The quality and quantity of these

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ES depends on the quality and availability of ecosystems and are sensitive to land-use changes (Foley et al., 2005; Mitchell, Bennett, & Gonzalez, 2013). Residents of cities and suburbs consume ES originating from different scales and proximities and their well-being is largely determined by the capacity of ecosystems in urban and peri-urban areas to generate these services (Folke, Jansson, Larsson, & Costanza, 1997; Gómez-Baggethun et al., 2013; Millenium Ecosystem Assessment, 2005). In a recent review on urban ES, Gómez-Baggethun et al. (2013) suggest that the use of this concept can play a critical role in understanding the links between the natural environment, and community well-being as well as resilience.

Studies have shown that changes in land use in urban areas often adversely affect the provisioning and regulation of supportive and cultural ES (Foley et al., 2005; Metzger et al., 2006; Millenium Ecosystem Assessment, 2005; Schroter et al., 2005) however, the links between urbanization, biodiversity, ecosystems and human well-being are still poorly understood and remain a challenge for planners and policy-makers (Crossman et al., 2013; Gómez-Baggethun et al., 2013; McDonald & Marcotullio, 2011).

One of the main problems that city planners are facing is that many of these ES do not refer to any existing economic market. Consequently, they are assigned a value of zero that leads to a lack of incentive for their preservation and contributes to the degradation of natural heritage (Bateman et al., 2013; TEEB, 2010). These economic distortions in land-use planning are particularly exacerbated in urban and peri-urban areas where trade-offs between land uses are apparent (Farber et al., 2006). One way to curb this problem is to demonstrate the real economic contribution of natural capital to the well-being of communities and to consider the cost of erosion of these amenities (TEEB, 2010; Troy & Wilson, 2006).

The economic valuation and mapping of these non-market natural assets, collectively known as ecosystem services value (ESV), provides an useful way to demonstrate how land-use and land-management decisions impact the quality of life and the economy of communities (Schägner, Brander, Maes, & Hartje, 2013; Troy & Wilson, 2006). The analysis of land conversion for ESV allows evaluators to specify the patterns of production of economic values, develop useful standards for benefit transfer and to assess the sustainability of the natural environment. Mapping and modelling ESV can also contribute to better integration into decision-making processes (Bockstael, 1996; Eade & Moran, 1996; Maes et al., 2012; Schägner et al., 2013; Troy & Wilson, 2006). For institutional users, these analytical maps are advantageous because they open the door to green accounting, assessment of land-use policies, resource allocation and aid in the design of new policies including payment for ES (Laurans, Rankovic, Mermet, Billé, & Pirard, 2013).

These scientific and political considerations led the spatial analysis of ESV to develop rapidly over the past 15 years (Schägner et al., 2013). Although analytical progress continues, the limitations in methods of biophysical quantification and economic valuation remain a challenge when identifying the impact of urban development patterns and loss of natural capital on the ESV (De Groot, Alkemade, Braat, Hein, & Willemen, 2010; Gómez-Baggethun et al., 2013). More research is needed in order to provide useful information on the relation between historical urban development and ESV.

To understand the effects of urban sprawl on ESV, we apply the value of non-market ES to four mapping analyses of the Montreal Metropolitan Region (MMR) over five decades. This enables us to show the economic effects of urban sprawl on a variety of ES of a typical metropolitan city in North America. Considering that past and present public development policies did not take into account the impact on ES of land-use conversion, this timeframe allows us to evaluate if different urban development strategies had an impact on the total ESV. Overall, this research has three objectives: (1) we characterize land-use changes and its dynamics in the Montreal area in the last half century, (2) the ESV related to the land-use changes is determined using a spatial analysis in a benefit transfer approach, (3) the relation between land-use changes and ESV in the light of socio-economic drivers and land-use planning and management policies is explained.

2. Material and Methods

2.1. *The Study Site*

The MMR is located in the southwest part of the province of Quebec (Figure 1). The City of Montreal is located on the island of Montreal, which is composed of the City of Montreal and 16 other municipalities. Today, the MMR is defined as a cluster of 82 municipalities on the islands of Montreal and Laval, and on the North and South Shores. In 2009, the MMR ranked 16th among the most populous metropolitan areas in North America with 3.9 million inhabitants (Communauté Métropolitaine de Montréal [CMM], 2010). Montreal's population is comparable to San Diego, Minneapolis, Seattle, San Francisco, Phoenix and Boston also located in North America (CMM, 2010). Table 1 shows the evolution of the region's population between the years 1966 and 2011. It presents a population increase from 2.6 to 3.8 million residents for the metropolitan area, while

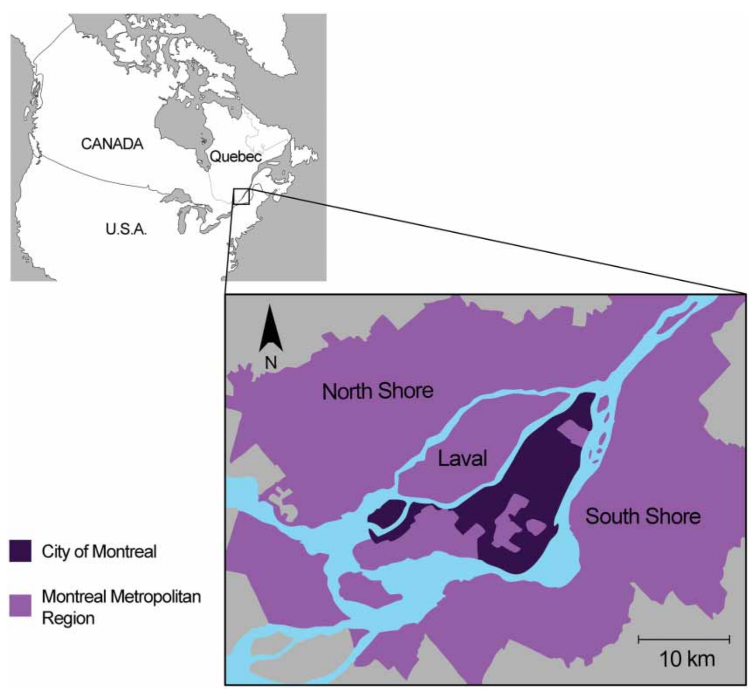

Figure 1. Localization of the MMR.

Table 1. Evolution of Montreal's population from 1966 to 2011

Year	City of Montreal	Island of Montreal	MMR	Part of the population living in Montreal island (%)
1966	1,293,992	1,923,171	2,570,985	74.8
1981	1,018,609	1,760,120	2,862,286	61.5
1996	1,016,376	1,775,788	3,326,447	53.4
2011	1,649,519	1,886,481	3,824,221	49.3

Source: Ville de Montréal (2013).

simultaneously showing a net decrease in the proportion of total metropolitan population living on the island of Montreal. This decrease in population of the city and the Island of Montreal in 1981 and 1996 was caused by a migration of urban residents to the suburbs (Sénécal, Hamel, Guerpillon, & Boivin, 2001).

Historically, the region was mainly composed of forests interspersed with lakes, rivers and a rich network of wetlands (Brisson & Bouchard, 2003). The development of human activity during the eighteenth and nineteenth centuries was primarily centred on agriculture and forestry and made the region one of the most cultivated areas in North America (Brisson & Bouchard, 2003). Until the late 1940s, the City of Montreal represented the most important industrial pole of the region with a densely populated centre, while outside the city, the island and shores were mostly made up of farms, forests and open spaces. The first effects of suburbanization are observed later when residential densities increased in the peri-urban fringe during a rapid urban and suburban population burst. Migration to the suburbs starting in the 1950s was caused by a population explosion and an increased need for single-family homes. This change doubled the need for roads and parking. Between 1971 and 2006, the residential density decreased by 32.8%, representing the highest loss in urban density for all metropolitan areas in Canada (Filion, Bunting, Pavlic, & Langlois, 2010). According to Sénécal et al. (2001), urban sprawl effects and management of the MMR have three overlapping phases in this study's spatial analysis.

From the 1950s to late 1970s, Montreal was subject to *functionalist* planning where a star-shaped development was favoured. The principle was that a strong centre structure would support satellite centres. This led to the establishment of an important highway network during the 1960s which greatly stimulated sprawl of urban functions to the extremities of the island and in the North and South Shores (Marois, Deslauriers, & Bryant, 1991; Sénécal et al., 2001). Suburbs of low-population density were established by converting forests and croplands into urban areas. Efforts to identify the management of peri-urban areas and proposals for conservation of large natural and agricultural areas were not found (Marois et al., 1991; Sénécal et al., 2001).

The second phase of urban sprawl, the *opposition* planning phase (1978–1993), where urban management opposes peoples migration to the suburbs, was instituted in order to mitigate sprawling and to protect agricultural areas (Sénécal et al., 2001). One of the major issues with suburban sprawl in MMR during the 1960s and 1970s is that it occurred on high-quality soils that were permanently lost to agriculture (Jobin, Latendresse, Grenier, Maisonneuve, & Sebbane, 2010; Marois et al., 1991). This issue was understood by local decision-makers during the 1970s and led the Quebec government in 1978 to establish a plan for the protection of agricultural land through land zoning. This planning was beneficial to agriculture; rapidly, the conversion of abandoned grassland to agricultural land occurred improving agricultural investment (Marois et al., 1991).

The *acceptation* planning (1994–2000s) differs very little from the previous period since the same observations are stated and the same remedies available (Sénécal et al., 2001). The difference lies in the participation of civil society in the planning and the decentralization of management. However, in the 1990s and 2000s, important urban pressure remained on agricultural land caused by a demand for urban development, municipal tax increases and the need for low cost housing (Dumoulin & Marois, 2003). The economical and demographical challenges that farmers faced combined with increasing demand for urban development led to a new phase of agriculture abandonment and speculation (Dumoulin & Marois, 2003).

2.2. *Land-Use Changes*

The available databases for the selected time points allowed us to analyse the majority of what is actually considered as the MMR (3850 out of 4260 km^2). The maps we used, whether for analysis or validation, were based on different systems of land-use taxonomy that were not compatible. This required reclassification to take into account certain specifications. To minimize bias of harmonization, we distinguished the land cover in the following broad categories: urban, croplands, forests, wetlands, grasslands, water and unproductive lands. The last category includes land used for rock, sand and mineral extraction, denuded surfaces and other types of areas. The details of this reclassification process were validated by three external geographic information systems (GIS) experts.

In order to minimize bias from the use of different classification systems, we compared coherence between each time point and each land cover. This resulted in the correction of several land classifications. After reclassification, correction and cleaning of the databases, the land-use maps were edited, calibrated and coded in ArcGIS software in order to perform a spatial analysis. Finally, by summarizing the cover of each land-use type for the four time points, we measured how each one changed over each of the 15-year time periods.

2.3. *Ecosystem Services Valuation*

To measure the economic consequences of land-use changes, we estimated an average value for each type of land-use cover classes based on the analysis of selected ES. To identify the ES applicable to the type of land-use cover found in the MMR, we conducted a literature review of studies linking land use and ESV. We first selected ES according to the work of Haines-Young and Potschin (2008), who identified a list of 11 ES related to urban areas that are in high need of conservation, restoration or improvement.

Since the MMR also includes a peri-urban zone with an important agricultural vocation as well as natural forests and wetlands, we added to this list, food production, pest management, erosion control and disturbance protection.

None of the studies placed value on noise buffering or spiritual ties for the Montreal area so these were not considered. Furthermore, since there is no timber harvest in the region, we did not include services provided by forests. Consequently, our analysis is based on a total of 13 ES. To identify the value of different services, we used both market-based (i.e. direct market prices and avoided costs) and benefit transfer methods.

The market price method is a relatively simple method used to estimate the economic value of ecosystem products or services that are exchanged in markets. For ES available in existing markets, it is possible to determine the consumer's willingness to pay for them at prices determined by the market. This method was used to identify values for food production, carbon sequestration, pollination and recreation.

The cost-based methods are based on the cost of damage due to lost services, the cost of replacing ES the cost or shortfall of ecosystems productivity loss or the cost of providing substitute services. They estimate the value of ES through payment for alternatives. This is based on the principal that economic agents incur costs to avoid damages caused by lost ES, thereby having to replace them. We used one of the cost-based methods, the avoided costs method, to estimate the value of water provisioning provided by forests.

The benefit transfer approach is a secondary method that uses values produced on previously studied sites to analyse a target site (Johnston & Rosenberger, 2010). We used it when the data to assess ESV from market prices or avoided costs were not available. Troy and Wilson (2006) identified three critical factors that must be taken into account when transferring results from a study site to a policy site: the biogeophysical similarity of both sites, the human population characteristics of source data and the differences in preferences weighed by income for the compared the populations. In order to address these issues, we selected studies performed in sites that are similar to the environmental characteristics of our study site.

The ecological filter that we used represents the comparability between services and ecosystems in the studies found in the literature and the site analysed in this project. Consequently, only studies produced on sites with resembling characteristics of southern Quebec (e.g. temperate forest, inland wetlands and similar crop varieties) were selected. In general, the ecosystems of Western Europe and North America had the most commonalities.

The socioeconomic filter refers to the living conditions of people in the countries where the studies were conducted, like, for example, their standard of living and education. This is particularly of interest when comparing individuals willingness to pay/households for ES since it tends to be highly dependent on socioeconomic characteristics (Johnston & Rosenberger, 2010). Thus, in this study, only studies from countries with high income, according to the Gross National Income per capita classification of countries by the World Bank, were considered. We used purchasing power parity (PPP) to minimize economic differences arising from non-Canadian studies and corrected values to inflation in order to present results in 2010 Canadian dollars.

We used this method to estimate values for air quality, water provisioning (for wetlands), disturbance protection, nutrient cycling, pest management, erosion control, biodiversity habitat and landscape aesthetics. We calculated the total value for each land-use cover by aggregating individual ES values per hectare and by multiplying it by the total area. To estimate ESV changes through time, the total estimate for a time point was subtracted with another time point. The following equations were used to assess total ESV:

$$\mathrm{ESV}_k = \sum_j A_k \times VC_{kj}, \tag{1}$$

$$\mathrm{ESV}_j = \sum_k A_k \times VC_{kj}, \tag{2}$$

$$\mathrm{ESV} = \sum_k \sum_j A_k \times VC_{kj}, \tag{3}$$

where ESV refers to the Ecosystem Service Value of land-use cover category k, and ES type j; A is the area and VC is the economic coefficient in \$/ha/year.

In order to test the relation between the total ESV and socioeconomic drivers, we identified relevant elements that could constitute temporal pressure factors on agriculture and natural ecosystems: total population, density of urban population, income per household and size of farms. Other drivers, such as gross domestic product, would have been interesting to test but specific historic information was not available for the study region. The degrees of freedom, based on the number of observations and the parameters to be estimated was very low so we could not perform linear nor multivariate regression to test the impact of these drivers on total ESV. We therefore performed a statistical analysis based on a non-linear regression model.

3. Results

3.1. *Land-Use Changes*

The land-use changes in the MMR during the five last decades are presented in Figure 2. During the 1966–2010 period, results show a significant decrease in croplands (20%) and forests (28%) that correlates with an increase of 59,700 ha of urban areas, representing an increase of 93% for the total area. Other land-use cover types show little variation, for example, a loss of 100 ha of wetlands (6%), 7800 ha of grasslands (30%) and no significant variation in water systems (less than 1%). The unproductive land-use cover class presents a challenge when comparing time points with different GIS data layers since the composition of this cover class is so variable.

Each 15 year period provides more details on the temporal dynamics of these land-use changes. This is especially true for the trade-offs between croplands and grasslands. While croplands show significant decrease during the 1966–1981 and 1994–2010 periods (24,700 and 18,400 ha), they increased by 9900 ha from 1981 to 1994. The opposite is true of grasslands, since they decreased by 42,000 ha from 1981 and 1994, but increased during the two other periods (18,200 and 16,000 ha, respectively). Forests show a constant decrease over the entire period, the biggest decrease during the first phase of urban sprawl (18%, 9% and 4%, respectively). While there is no significant variation in wetlands and water, urban areas constantly increased from 1966 to 2010 (26%, 43% and 7%, respectively). In the end, the cumulative loss of croplands, grasslands and forests (61,300 ha) is essentially equal to the increase in urban areas (59,700 ha). This

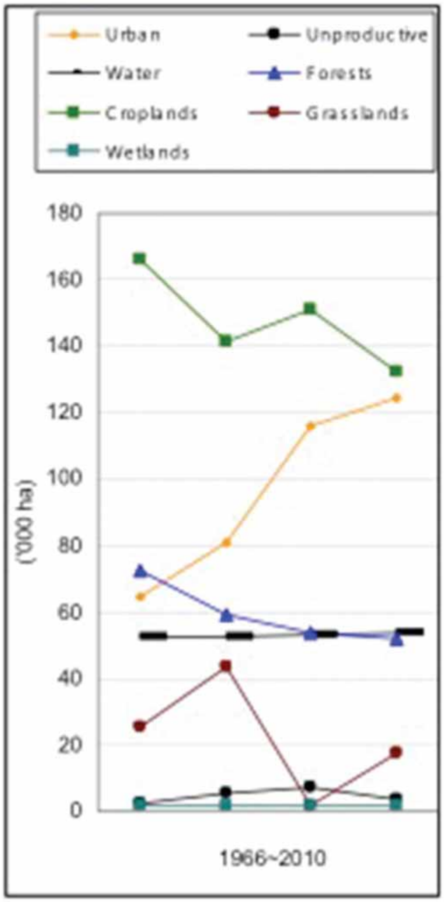

Figure 2. Land-use changes in the MMR from 1966 to 2010.

area broadly represents the land-use demand for human activities towards natural environment and agro-systems in the MMR over the last five decades.

The maps of the four time points presented in Figure 3 illustrate these changes. We can observe a development of the urban core through time. In 1966, urban areas were mostly concentrated on the island of Montreal, which is confirmed by the high proportion of the population of the MMR living in the city (50%) or on the island (75%) (Table 1). This expansion of the urban core is clearly visible on the 1994 and 2010 maps where urban areas represent a large proportion of the land-use cover in Laval (north of Montreal Island) and on the North and South Shores. This is consistent with results given in Table 1, where 43% of the total MMR population live in the city and only 49% on the island.

3.2. *Ecosystem Services Value*

3.2.1. *Food production*. There are approximately 2000 farms within the MMR (CMM, 2011), most of which are involved in cereal, hay, soya and maple syrup production. Fruit and vegetable type farms are the second most common, while farms dedicated to greenhouse, nursery and floriculture are the third. A value

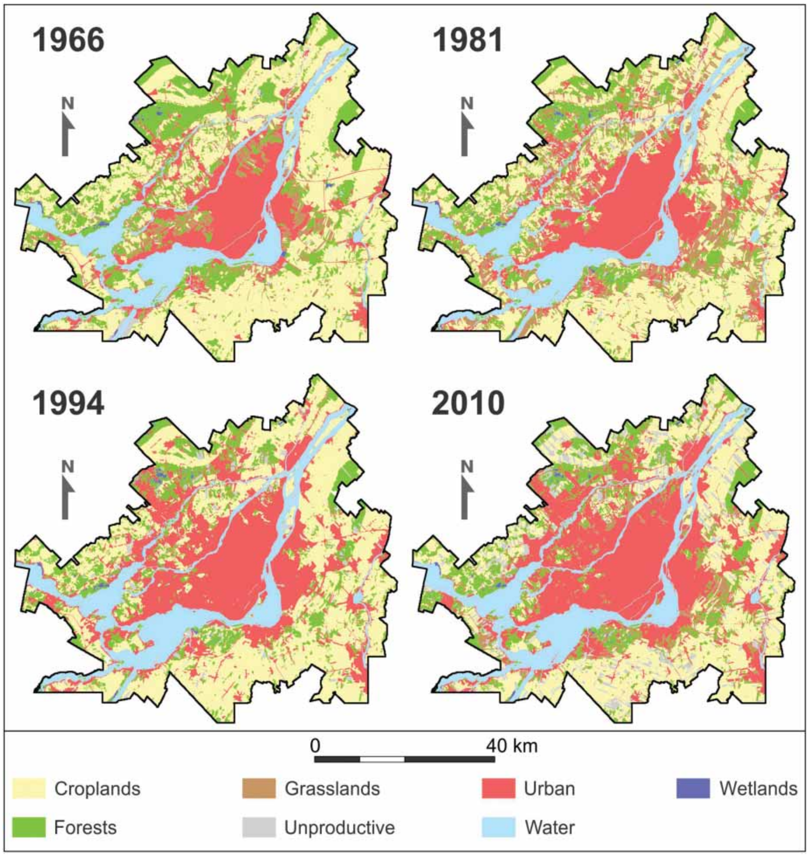

Figure 3. Land-use evolution of the MMR from 1966 to 2010.

of 3630 \$/ha/year was obtained by dividing the total regional agricultural income in 2010 by total area under agriculture (CMM, 2011).

3.2.2. *Climate regulation*. The quantity of sequestered carbon in forest environments corresponds to 1.93 tCO_2/ha/year, which is an average value of recorded rates between 1990 and 2009 (Environment Canada, 2011). The monetary value used to measure carbon sequestration corresponds to the social cost of carbon used in the evaluation of public policy by Environment Canada (25 \$/tCO_2e) (Environment Canada, 2010). When this value is applied to the region's forest areas, we obtain a value of 48 \$/ha/year for this service. Using the same method, the rate of carbon sequestration for wetlands in southern Quebec is estimated at 0.3 tC/ha/year (Ju & Chen, 2005), for a value of 28 \$/ha/year. For grasslands, the annual carbon sequestration is estimated to be between 2.17 tC/ha in Klumpp, Tallec, Guix, and Soussana (2011) for a value of 199 \$/ha/year.

The emissions from the agricultural sector accounted for 7% of total emissions of the total greenhouse gases in Quebec (MDDEP, 2009). However, the role of agricultural areas in carbon fluxes is ambiguous since they can act as both sources and sinks for CO_2, N_2O and CH_4 (VandenBygaart, Gregorich, & Angers, 2003). Consequently, we have not valuated this service for croplands.

3.2.3. *Pollination*. In Quebec, 350 different pollinator species have been reported (Chagnon, 2008). We measured the value of the pollination service following Chagnon (2008) methods relating the value of different cultures, their size and their respective rates of dependence on pollinators for fruit and vegetable production in Quebec. We applied the total value attributable to the action of pollinators for 2008 (\$166.1 $\times$ 10^6) to the entire agricultural area of the province (6.3 $\times$ 10^6 ha), resulting in a value of 26.4 \$/ha/year. After correcting for inflation, we obtained a value of 28 \$/ha/year. Considering that pollination service is factored into the food production service of croplands, we apply it to grasslands only to avoid double counting.

3.2.4. *Recreation*. To assess the contribution of the forest and wetland to recreational activities, we used the expenses incurred by residents of Montreal for tourism and activities related to nature and wildlife. Of the total recreational expenses in 2000 (\$448.7 $\times$ 10^6), 30% was spent inside the Montreal area (\$134.6 $\times$ 10^6) (Bouchard, 2003). Take this number and divide it by the total forest, wetland and water area and you get a value of 1525 \$/ha/year (with inflation correction). For tourism and recreational activities in agricultural lands, we used the tourism benefits associated with rural tourism in the region. We used the income from the agro-tourism of 66 agro-businesses in the region in 2005, an average of \$138,000, for a total of \$9.1 $\times$ 10^6 (MAPAQ, 2006). Distributed over the total agricultural land-use cover in the region and corrected due to inflation, we arrived at a value of 86 \$/ha/year for this service.

3.2.5. *Water provisioning*. In studying 30 water suppliers in the USA, Ernst, Gullick, and Nixon (2004) reported that operating costs of treating water decreased by 20% when the forest cover of the source increases by 10% over the entire watershed. This results in an economic value of between 0.006 and 0.003 \$/m^3, for an increase of 10% of the forest cover. In comparison, the cost of supplying treated drinking water in 2010 was 0.09 \$/m^3 for the Montreal area (CMM, 2010). Considering that urban and peri-urban forests cover 13.6% of the area, this will result, according to Ernst et al. (2004) in a reduction of 0.0252 \$/m^3 of treated water. If we consider only the City of Montreal, the volume of water treated is 1460 million of m^3 (CMM, 2010), resulting in an economic value of 36.8 $\times$ 10^6 \$/year or 701 \$/ha/year.

The references under the section 'References of the studies used in the benefit transfer' detail the 33 monetary estimates taken from the 19 studies that were used in the benefit transfer approach, as well as their valuation method and provenance. The mean results obtained for the 13 ES valued are described and synthesized in Table 2. With a value of 4593 \$/ha/year, wetlands are showing the highest ESV. The values of forests (3982 \$/ha/year) and croplands (3988 \$/ha/year) are very similar but croplands ESV is mainly explained by the food production service, while other non-market ES show lower values. Grasslands have the lowest value for land-use cover type at 2720 \$/ha/year, but their non-market ES value is still higher than those provided by croplands and is mainly explained by the habitat for biodiversity they provide.

Urban areas were excluded from the valuation process for two reasons: First, primary ESV studies in urban areas are scarce so there is a lack of information for benefit transfer at this scale. Second, our spatial analysis was not precise enough to

Table 2. Value of ES per type of land-use cover ($/ha/y)

ES	Forests	Wetlands	Croplands	Grasslands
Food production	–	–	3630 (MP)	–
Climate regulation	48 (MP)	28 (MP)	–	199 (MP)
Air quality	650 (BT, 1)	–	–	–
Water provisioning	701 (AC)	1130 (BT, 3)	–	–
Waste treatment	133 (BT, 1)	260 (BT, 1)	–	–
Erosion control	–	–	103 (BT, 3)	35 (BT, 1)
Pollination	–	–	–	28 (MP)
Disturbance protection	–	470 (BT, 1)	–	–
Biodiversity habitat	884 (BT, 9)	519 (BT, 1)	–	2261 (BT, 1)
Pest management	41 (BT, 1)	–	–	41 (BT, 1)
Nutrient cycling	–	–	169 (BT, 1)	–
Aesthetics	–	661 (BT, 3)	–	156 (BT, 5)
Recreation	1525 (MP)	1525 (MP)	86 (MP)	–
Total	3982	4593	3988	2720

Notes: In the parenthesis, the method is given as follows. AC,avoided costs; BT,benefit transfer; MP,market price. For benefit transfer, the number given represents the number of monetary estimations used.

distinguish pieces of land that could have an important ESV (e.g. urban parks or urban trees). We should keep in mind, however, that even in human-dominated systems, ES are produced (Haines-Young & Potschin, 2008; Li, Wenkai, & Zhenghan, 2010). For the same reason, unproductive lands were not evaluated. Considering that the total area of water bodies does not vary through time, we did not consider it for the ESV change analysis.

Table 3 and Figure 4 show the progression of ESV for each of the land-use cover types through time. We observed a general decrease in total economic value with a notable average loss of 236×10^6 $/year between 1966 and 2010. This decrease represents 23% of the total ESV in 1966 and the most significant losses for the entire period are attributed to the loss of forest and cropland areas. We found that the total loss in ESV is essentially constant during the two first time periods but decreases in the third. A total ESV of 101×10^6 $/year is lost on average between 1966 and 1981 and 96×10^6 $/year between 1981 and 1994 while, 39×10^6 $/year is lost between 1994 and 2010. Although the forests ESV decreased in each period, the wetlands' ESV remained constant. The main trade-offs were found between croplands and grasslands. While croplands ESV losses are significant between 1966 and 1981 (99×10^6 $/year) and between 1994 and 2010 (74×10^6 $/year), there is actually a gain between

Table 3. Total ESV in the MMR from 1966 to 2010

	Total economic value ($\times 10^6$ $/y)							
Land-use cover	1966	$\Delta_{1966-1981}$	1981	$\Delta_{1981-1994}$	1994	$\Delta_{1994-2010}$	2010	$\Delta_{1966-2010}$
Forests	288.1	−51.8	236.4	−21.4	215.0	−7.8	207.2	−80.9
Wetlands	7.9	−0.5	7.4	0.2	7.6	−0.5	7.2	−0.7
Croplands	661.3	−98.6	562.7	39.5	602.1	−73.5	528.7	−132.6
Grasslands	69.6	49.5	119.0	−114.0	5.0	43.3	48.3	−21.3
Total	1026.9	−101.4	925.5	−95.7	829.7	−38.5	791.4	−235.6

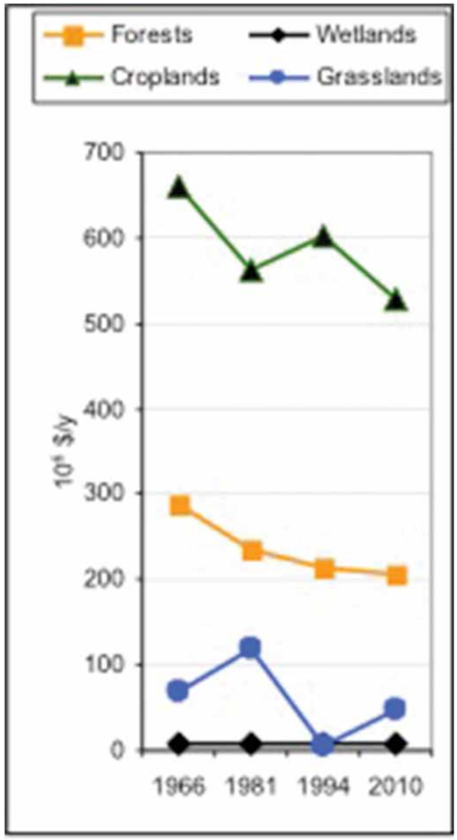

Figure 4. Changes in ESV per land-use class.

Table 4. Values and trends of ES from 1966 to 2010

	1966		1981		1994		2010		
ES	10^6 \$/y	%	10^6 \$/y	%	10^6 \$/y	%	10^6 \$/y	%	Trend
Food production	601.8	58.6	512.2	55.3	548.5	66.1	481.3	60.8	↑
Climate regulation	8.5	0.8	11.6	1.3	2.9	0.3	6.0	0.8	–
Air quality	47.1	4.6	38.6	4.2	35.1	4.2	33.8	4.3	↓
Water provisioning	52.7	5.1	43.4	4.7	39.8	4.8	38.3	4.8	↓
Waste treatment	10.1	1.0	8.3	0.9	7.6	0.9	7.3	0.9	–
Erosion control	18.0	1.8	16.1	1.7	15.6	1.9	14.3	1.8	–
Pollination	0.7	0.1	1.2	0.1	0.1	0	0.5	0.1	–
Disturbance protection	0.8	0.1	0.8	0.1	0.8	0.1	0.8	0.1	–
Biodiversity habitat	122.8	12.0	152.3	16.5	53.3	6.4	87.1	11.0	↓
Pest management	4.0	0.4	4.2	0.5	2.3	0.3	2.9	0.4	–
Nutrient cycling	28.0	2.7	23.9	2.6	25.5	3.1	22.4	2.8	–
Aesthetics	4.1	0.4	7.9	0.8	1.4	0.2	3.9	0.5	–
Recreation	127.3	12.4	105.1	11.4	97.9	11.8	93.1	11.8	↓
Total	1026.9	100	925.5	100	829.7	100	791.4	100	

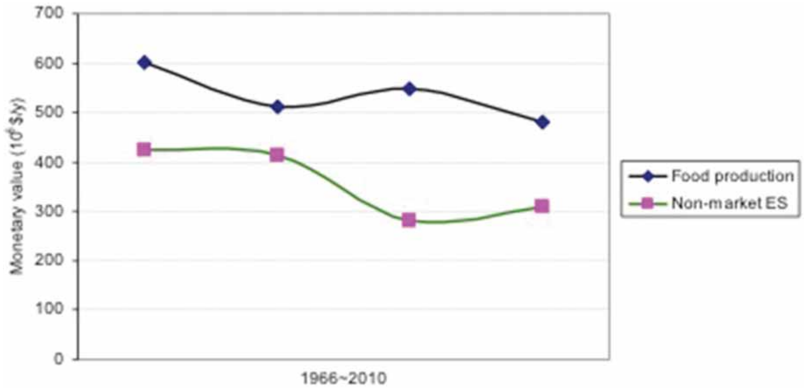

Figure 5. Evolution and trade-offs between market and non-market ES.

1981 and 1994 (40×10^6 \$/year). Conversely, grasslands show the opposite trend. When combined, the general trend of croplands and grasslands ESV over time is generating total losses of 49×10^6\$, 75×10^6\$ and 30×10^6 \$/year, respectively.

Table 4 shows the variation of the values for the 13 ES evaluated. When compared to the total ESV of each time period, all regulating and cultural services are exhibiting a stable or declining trend. Food production is the only one showing an increase. A decrease is observed for air quality, water provisioning, biodiversity habitat and recreation. Figure 5 is showing a clear trade-off between food

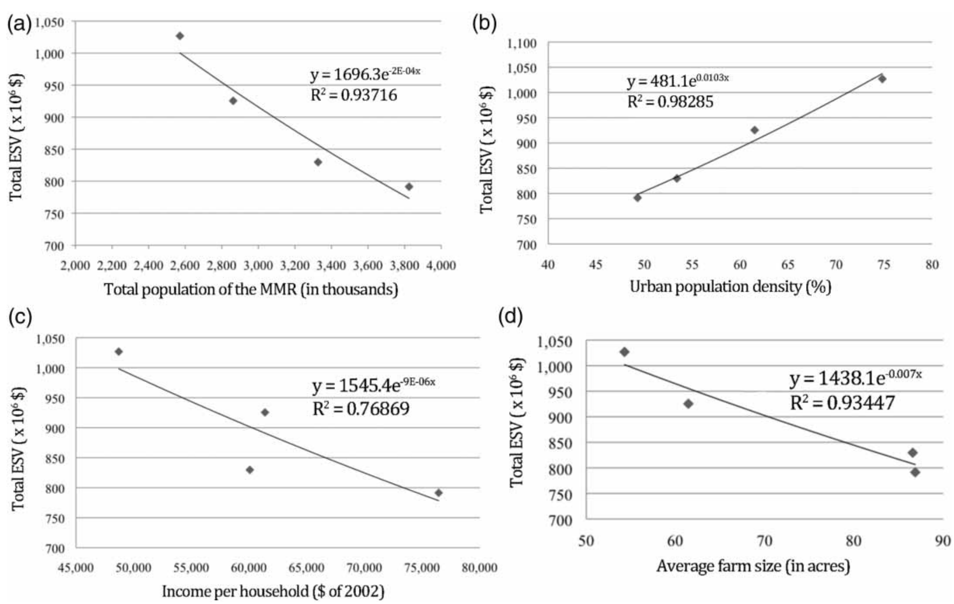

Figure 6. The relationships between (a) total population and total ESV, (b) urban population density and total ESV, (c) income per household and total ESV and (d) farm size and total ESV. *Source*: (a) and (b) Ville de Montréal (2013); (c) Statistics Canada, Census of Canada (1971, 1981, 1996), Statistics Canada, National household survey (2011) and (d) Statistics Canada, Census of Agriculture (1971, 1981, 1996, 2011).

production (provisioning services) and all other non-market services. For all time periods, we see that as market services increase, non-market services decrease and vice versa, i.e. as the food production decreased non-market services increased.

3.3. *ESV and Socioeconomic Drivers*

Figure 6(a) and 6(b) show a nonlinear relationship between total ESV and population and total ESV and urban population density, respectively. The redefinition of City limits in 2002 influenced this study's approach whereby, the population on the island of Montreal and the MMR are used instead of the official city population. The MMR population increased by 49% between 1966 and 2010, while the percentage of people living on the island of Montreal compared to the population of the MMR declined from 75% to 49% (Table 1). This demographic trend negatively impacted the total ESV. Figure 6(c) indicates the negative relation between total ESV and income per household. The income per household varies from 10,325 \$/hh in 1966, to 27,191 \$/hh in 1981, to 52,795 \$/hh in 1994 and to 87,736 \$/hh in 2010. This demonstrates that economic growth in the last decades had a negative impact on the provision of ES, especially non-market ES. Figure 6(d) shows the decline of the total ESV in relation to average farm sizes. Since it was not possible to find values for the average farm surface area for the MMR, we used Montreal Island and Laval's historic average farm size (based on the relation of croplands and the number of farms), which varies from 54 ha in 1966, to 61 ha in 1981, to 87 ha in 1994 and 2010.

4. Discussion

The results of this study are coherent with others conducted across the globe measuring the economic effects of land-use changes on ESV through time in urban or peri-urban areas (Table 5). All of the studies listed show a decrease in the total ESV per year. The number of ES categories evaluated remaining constant in the majority of these studies is explained by the use of ESV coefficients from Costanza et al. (1997) for global ecosystems and Xie, Lu, Leng, Zheng, and Li (2003) for ES in China. The trends in ESV, given in Table 3 and Figure 5, reveal a loss of regulating and cultural services towards provisioning services. This is in accordance with the findings of the Millennium Ecosystem Assessment (2005). Similarly, Raudsepp-Hearne, Peterson, and Bennett (2010) showed significant trade-offs between providing services and regulating cultural amenities in areas of intensive agricultural production on the South Shore of Montreal. This highlights the focus that previous public policies put on market products of ecosystems in opposition to non-market ones (Bateman et al., 2013).

4.1. *Urban Sprawl Phases and ESV Losses*

The variations in croplands, grasslands and forests that we identified are closely related to the different urban sprawl phases that have taken place in Montreal. *Functionalist* planning resulted in ESV losses of 101×10^6\$/year and is linked to cropland and forest losses during that period. If they are slightly compensated through a gain in grasslands, this increase in grassland areas was partly caused by the abandonment of agriculture and these grasslands are projected to become urban areas (Marois et al., 1991).

Table 5. Review of studies measuring the land-use changes effects on ESV in urban or peri-urban areas

Authors	Location	Time period	Number ES	ESV variation/ year
Kreuter, Harris, Matlock, and Lacey (2001)	San Antonio, USA	1976–1994	17	-6.0×10^6 USD
Zhao et al. (2004)	Chongming Island, China	1990–2000	17	-2.0×10^8 USD
Li et al. (2007)	Pingbian County, China	1973–2004	–	-2.4×10^8 USD
Li et al. (2010)	Shenzen, China	1996–2004	9	-2.3×10^8 Yuan
Estoque and Murayama (2012)	Baguio, Philippines	1988–2008	17	-3.2×10^6 USD
Liu, Lia, and Zhang (2012)	Taiyuan, China	1990–2005	9	-2.0×10^7 Yuan
Su, Xiao, Jiang, and Zhang (2012)	Hang-Jia-Hu region, China	1994–2003	9	-8.5×10^9 Yuan
Mendoza-González, Martínez, Lithgow, Pérez-Maqueo, and Simonin (2012)	Boca del Rio, Mexico	1995–2006	9	-1.4×10^3 USD
	Chachalacas, Mexico	1995–2006	9	-7.0×10^5 USD
	Costa Esmeralda, Mexico	1995–2006	9	-1.0×10^3 USD
Hu, Wu, Hong, Qiu, and Qi (2013)	Fuzhou City, China	1986–2006	9	-1.2×10^9 Yuan
Wu, Ye, Qi, and Zhang (2013)	Hangzhou, China	1978–2008	9	-7.6×10^8 Yuan

Emerging from concerns about agricultural erosion, the *opposition* planning phase led to the protection of agricultural land. During the 1980s, peri-urban agriculture developed within this legislative framework and a reconversion of grasslands into croplands explains in part the trade-offs between the two land-use classes given in Table 3. This legislation somehow contained the urban development of agriculture land but still presents an important ESV decline of 96 × 106 $/year, which is mainly explained by the loss of grasslands. Even if short-term positive effects occurred in agriculture, urban pressure was still present for low-density residential, commercial, industrial and infrastructure purposes (Marois et al., 1991) and led to a loss of grasslands with a 112 × 106 $/year impact on total ESV.

During the *acceptation* planning period, several years after the adoption of the Agricultural Act, the economic and demographic challenges faced by farmers, combined with continued pressure for urban development led to a new phase of agriculture abandonment and speculation (Dumoulin & Marois, 2003). Although abandonment was not as high during the *functionalist* phase, there was still an increase in total grassland area and a decrease in croplands. The low number of protected forest areas in the Montreal region weakened their conservation capacity and allowed public and private developers to convert the majority of 2000 ha of forest lost between 1966 and 2010 to urban spaces.

The low variation in wetland area and ESV can be explained by the fact that most of the changes occurred prior to our study period. These losses are mainly attributed to draining for agricultural development and filling of open water and wetlands in favour of houses, roads and agriculture as well as the construction of facilities for water treatment, including dams and the development of the St. Lawrence Seaway observed before the 1970s (Jean & Létourneau, 2011). Since the 1970s, wetland area has remained constant, but there have been changes in the type and location of wetlands (Jean & Létourneau, 2011; Jobin et al., 2010). Although wetlands continue to be threatened by urbanization, restoration efforts and lower water levels have resulted in net gains of marshes and swamps. Consequently, in order to get a clear picture of wetland ESV variation, a longer time period of study would be necessary.

In the end, we can conclude that the three management strategies implemented as part of Montreal's urbanization plan over the last five decades have all had negative effects on ESV. The socioeconomic drivers that characterize resulting urban sprawl and agriculture intensification, include a rising population, a decline in the percentage of people living in the central city, increase in income per household and increase in farm size.

The latest demographic projections for the MMR, predict an increase of 425,000 households between 2006 and 2031 (André, Fleury-Payeur, & Lachance, 2009). In order to accommodate these new households, 13,000 ha of vacant land must be reserved for future residential development (CMM, 2011). Based on an average residential density of 18.1 houses per hectare from 1999 to 2004 in the MMR (CMM, 2011), the residential capacity of the region is estimated at 315,000 new households. At the projected rate of growth, the region could reach its full capacity in residential development by 2023 without eroding agriculture and natural spaces. Welcoming 425,000 new households in the region by 2031 will prove to be a challenge, especially in terms of optimizing the available space and preserving ESV.

4.2. *Mapping and Valuing ES: Methodological Caveats*

Since Costanza et al.'s land mark study in 1997, spatially explicit benefit transfers and ES mapping studies are a burgeoning field of research (Schägner et al., 2013; Troy & Wilson, 2006). This can mainly be attributed to the development of GIS technologies in the past 15 years and the rise of interest towards ES. However, in this study, we used both land-use cover and monetary indicators as proxies, which can be seen as a probative limitation to this approach.

Using the land use as a proxy for ES measurements assumes homogeneity in their production and distribution. It is a strong assumption to postulate that every unit area of land produces the same amount of services, moreover, that they are constant in time. This consideration of ES as uniform, unmoving and site-bound elements leads to ignoring the importance of biotic and abiotic movements, surrounding environment and landscape connectivity (Eigenbrod et al., 2010; Mitchell et al., 2013). The supply of many ES, especially the provisioning and regulating services, largely depend on the landscape composition and configuration, and their functional connectivity (Kremen, 2005; Mitchell et al., 2013).

Another important spatial limitation of this study is linked to the evaluation of land-use change. The comparison of land-use cover over time is weakened by the different methodologies used for mapping over the four time points. Even though special care was paid to minimize incoherence, it is obvious that harmonizing land-use classes can generate classification errors and does not take into account the changes within the classes.

Moreover, scaling can have significant effects on ES measurement, both through land use and valuation analysis (Hein, van Koppen, de Groot, & van Ierland, 2006; Konarska, Sutton, & Castellon, 2002). From this perspective, the scale at which ES are measured can strongly influence their valuation. In this study, the land-use data used as a proxy for ES provisioning are at scales of 1:20,000 and 1:50,000, while the ES valuation coefficient used was not necessarily produced at that scale. This is particularly true for benefit transfers based on stated preference studies.

In this study, the majority of valuation proxies were based on transferred values. In order to minimize transfer bias, we selected values produced for similar ecosystems within the study area. When these values were transferred from other locations, we used PPP to adjust the original values. However, other sources of bias, such as temporal effects, generalization errors or double counting may have impacted the net ESV (Johnston & Rosenberg, 2010). Moreover, a gap analysis of the studies we used for benefit transfer (Table 2) tell us that a number of ES have not been evaluated for all ecosystems. A more exhaustive analysis of the contribution of ecosystems to economics and to community well-being would certainly tend towards a higher total ESV and would increase ESV losses generated by urban sprawling.

5. Conclusion

Our results share similar findings to those obtained for many other cities and regions: urban sprawl generates significant losses in ESV. In the MMR, the difference in land-use cover between 1966 and 2010 led to a yearly loss in ESV of $\$236 \times 10^6$. Mapping ESV provides a tool that can be used to inform decision-makers on the effects of land-use conversion by highlighting the amenities from which communities can benefit from ecosystems.

Urban sprawl is one of the most common drivers for land-use change generating a variety of impacts on natural and agro-systems. Public decisions are slowly beginning to integrate the cost of land-use change into their land-use management. Many decisions, however, are still based only on the market economy and neglect the contribution of ES to the economy and community well-being. Planners should consider different ways to maximize the provisioning of ES, the value of which has been demonstrated herein. Conservation of pristine ecosystems and key natural areas is certainly the core of biodiversity preservation, but strategic planning calls for connections between natural areas in order to maximize ES and a densification of built-up areas in order to reduce land-use demand.

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