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Meta-analysis for the transfer of economic benefits of ecosystem services provided by wetlands within two watersheds in Quebec, Canada

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Abstract Climate change will cause major changes in ecosystems. Therefore, it is crucial that climate change policy consider the value of all services that are provided by watershed ecosystems. For this purpose, geospatial data and economic analysis are combined to determine a monetary value for wetland ecosystem goods and services (EGSs) in the watersheds of the Yamaska and Bécancour Rivers (Quebec, Canada). From published studies of wetland economic evaluations, we selected 51 relevant studies from 21 countries and performed a benefit transfer using meta-analysis. Our research emphasises the importance of considering multiple wetland characteristics when conducting a benefit transfer because of their complementary effects. We propose an approach that integrates spatial variables that have not been previously used, including type of wetland (complex or isolated) and land use (% of agricultural, urban, forest and water land cover), at a much finer geographical scale of 50 km². Simultaneous use of detailed spatial and economic characteristics in each wetland area allowed us to assign heterogeneous EGS values and map these values in sub-watersheds (50 km²) of the two rivers. Our results demonstrate that location and scale can affect wetland value. When wetland valuation was conducted based on mean values for geospatial characteristics, the EGS [2014 purchasing power parity (PPP) price] that is provided by wetlands was $5277 PPP/ha/year for the Yamaska River Watershed (YRW) and $3979 PPP/ha/year for the Bécancour River Watershed (BRW). When wetland valuation was performed at a more detailed sub-watershed scale to reflect variability within a watershed, EGS (2014 PPP price) that is provided by wetlands was valued higher at $9080/ha/year for YRW and $4702/ha/year (2014 PPP price) for BRW.

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Introduction

Human activity is a major factor that has led to widespread wetland losses throughout history. For example, the Millennium Ecosystem Assessment (MEA 2005) reported that between 80 and 98 % of the wetlands that are located in or adjacent to major urban centres have disappeared worldwide. Such a tendency has been exacerbated over the last 30 years. Since 1985, there has been a 26 % loss of wetland areas through intensive agricultural drainage. Of these wetland losses, 56–65 % have occurred in Europe and North America alone (Zedler and Kercher 2005). According to the MEA (2005), the major human activities that have contributed to wetland degradation are infrastructure development, changes in land use, drainage, eutrophication and pollution, overexploitation, and the introduction of invasive species.

Widespread conversion of wetlands has taken place because of shortages of land that is suitable for agriculture and urban settlement. This process was reinforced by failures to consider the value of ecosystem goods and services (EGSs) that wetlands provide to human society. Some EGS that are provided by wetlands are particularly important to humans, viz., regulation of water flows, sediment capture, habitat biodiversity, food and materials production (e.g., timber, peat), climate regulation, and recreation and education, among others. The values that are associated with these services to society are often greater than the economic benefits that would be obtained from converting wetlands to other uses (MEA 2005). However, as most EGS are non-market goods and services that take the form of positive externalities, they do not have a monetary value, which poses great difficulties for policymakers when they wish to conduct cost–benefit analyses.

Since the 1950s, various economic methods have been proposed to estimate non-market EGS values in monetary terms in response to society's growing demands to include the value of the environment and natural resources (Haluza-Delay et al. 2009). Positive and negative externalities, which generally are not considered explicitly in economic terms, subsequently

can be internalised and taken into consideration in an economic project.

We can distinguish five main categories of EGS valuation methods. First, methods that are based on market prices assess the direct use value of EGS in reference to their market value. Second, methods that are based on costs estimate the EGS value by the cost of avoided damage or the replacement cost of ecosystem losses. Third, revealed preference methods (e.g., hedonic price, travel costs) are based on consumer preferences that are revealed by their behaviour in an existing market and on the principle of weak complementarity between EGS and the goods exchanged in this market. An example of this method would consider the complementarity between air quality within an area and house prices (Malër 1974; Desaigues and Point 1993; Bateman et al. 2011). Fourth, methods that are based on stated preference (e.g., contingent valuation, discrete choice experiments) measure the value of EGS through simulated markets to identify survey respondents' trade-offs between the price to pay (or compensation to accept) and improvement (or degradation) of the environment. Lastly, benefit transfer methods involve estimating the value of EGS for a target site using existing valuation estimates from primary studies that explicitly use one of the four aforementioned methods for similar sites (Navrud and Ready 2007).

Each method has its advantages and disadvantages. The measure of cost/market price is relatively easy to use, but it only focuses on marketable characteristics of the ecosystem. These methods do not consider the preferences and social values of EGS. Revealed preference methods depend upon observable consumer behaviours in markets for complementary goods and, therefore, can only measure the direct- and indirect-use value of EGS. Stated preference methods provide a flexible approach for establishing a hypothetical market framework, which includes in its assessment both the use value and the non-use value of EGS.[1] However, stated preference methods face

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[1] Non-use values refer to values that have no connection with the use of the good in the present or in the future. We can distinguish three types of non-use values: existence values, which reflect the utility of knowing that certain services exist; bequest values, which reflect the utility preserved for future generations; and option values, which reflect the premium that a consumer is willing to pay to maintain the potential use of a good at a later date.

criticism that is related to their hypothetical nature and potential effects on respondents' reactions. For example, respondents may tend to exaggerate or understate their preferences through strategic responses; respondents may also only react partially or weakly to different levels of EGS improvement that have been proposed in the hypothetical scenarios, which can lead to systematic underestimation of EGS values. Also, there is empirical evidence suggesting that respondents tend to over-report their value for proposed EGS in a stated preferences valuation study because the hypothetical change on EGS will not materially affect the level of utility for respondents (List and Gallet 2001; Murphy et al. 2005; Harrison and Rutström 2008). Further, some researchers have observed divergence between the willingness to pay (WTP) and willingness to accept, because individuals tend to react more strongly to losses than to gains, even when the initial condition is the same (Hanemann 1991; Kahneman and Tversky 1991). Benefit transfer is a secondary method that extrapolates the results that are obtained by one or many primary studies applying methods such as revealed/stated preference or replacement costs. Generally, benefit transfer studies are less demanding than the revealed/stated preference methods from a time and financial perspective. Meta-analysis, a benefit transfer approach that relies upon multiple studies, provides some insight into how site-specific geospatial and environmental conditions affect EGS value. This is clearly not possible for a primary study focusing on a specific test area.

In this paper, we calculated the value of EGS flows that are provided by wetlands in southern Quebec at the resolution of 50-km^2 sub-watershed unit levels, based on a meta-analysis approach. In our meta-analysis, the wetland EGS value was regressed over detailed environmental and geospatial characteristics of the wetland sites that had been studied in the selected primary valuation studies. Some of the spatial variables are being used for the first time in a meta-regression model to value wetlands, including (1) wetland type (complex or isolated) and (2) surrounding land cover (% of agriculture, urban, forest and water in a 10 km radius around the wetland). In the value extrapolation step, the acquisition of both watershed and sub-watershed level data for environmental and geospatial characteristics allowed us not only to apply the meta-regression model at the catchment level, but also to report precise EGS values

at a sub-catchment level (50 km^2). Our results illustrate potential variation within watersheds. Considering that sub-watershed units are routinely used for water management purposes, our ability to assess wetland EGS value for these smaller spatial units will allow more accurate identification of the areas of intervention for the restoration and protection of wetlands.

Materials and methods

Description of the two test areas

In Quebec, wetlands occupy 17 million ha, which is about 10 % of the land base of the province (Ministère du développement durable, de l'Environnement, de la Faune et des Parcs du Québec, MDDEFP 2013). Over time, a large number of wetlands have been converted to other uses, especially in southern regions of the province where most of the population is located. We chose to study wetlands of the Yamaska and Bécancour Rivers (Fig. 1) for four reasons. First, they are both representative of watersheds that are found in southern Quebec, but they do exhibit important differences. Both rivers flow northward and drain into the St. Lawrence River. The upstream portions of both watersheds are located in the mountainous Appalachian region and the downstream portions, which constitute about half of each watershed, are located in the physiographic region of the St. Lawrence lowlands, which is dominated by flat topography. The main geographic and demographic characteristics of these watersheds are summarised in Table 1. Second, both sites experienced substantial wetland losses between 1984 and 2011, i.e., 12.2 and 9.4 % for Yamaska and Bécancour, respectively. The pressure of future losses is still very intense (Fournier et al. 2013). Third, the two watersheds have wetlands with diverse geospatial and environmental characteristics, which are well documented in available hydrometric and climate-related databases. Last, these sites suffer from a wide range of natural and anthropic pressures, thereby providing a variety of interesting cases for benefit transfers.

The Yamaska River originates in Brome Lake and enters the St. Lawrence upstream of Lake Saint-Pierre. The northern lowland part of the Yamaska River Watershed (YRW) is dominated by agricultural land
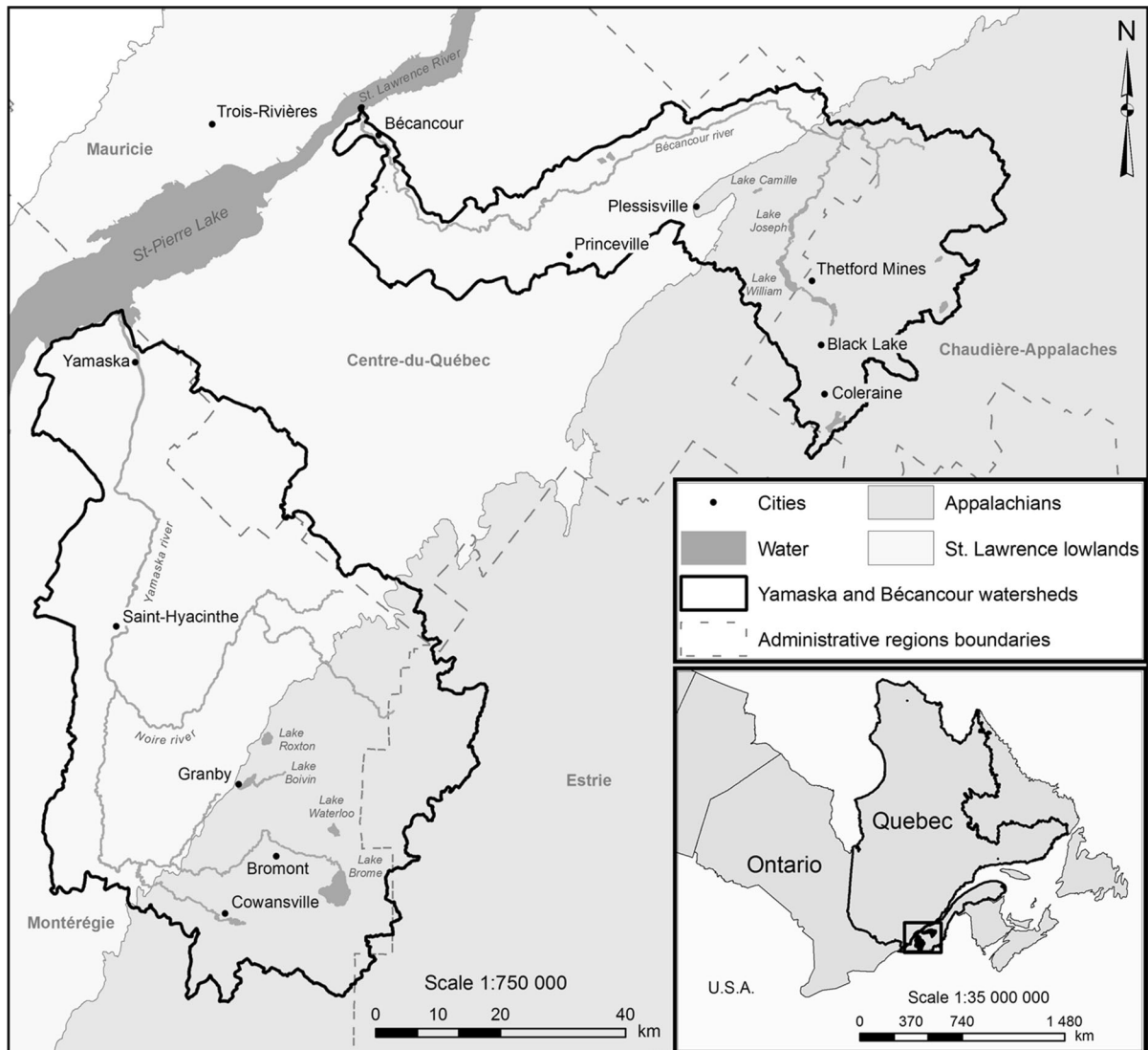
Fig. 1 Location of the Yamaska River and Bécancour River Watersheds (from Labbé et al. 2011)

used for the intensive production of maize (*Zea mays*) and soybean (*Glycine max*) that requires heavy use of fertilisers, and which exacerbates erosion and water pollution (MDDEFP 2013). The upstream portion of the watershed, which is located in the Appalachian region, is predominantly woodlands and fodder crops, given the rugged terrain (Comité de gestion du bassin versant de la rivière Yamaska, COGEBY 2010). Upstream agricultural activity has less of an impact on water quality than the more intensive cropping that is being conducted downstream (MDDEFP 2013). Almost half of the population living in the watershed

resides in the towns of Granby, Saint Hyacinthe and Cowansville. Consequently, both agriculture and urban development have greatly contributed to the loss of wetlands through drainage and filling, especially for wetlands that are located in the St. Lawrence lowlands (Canards Illimités Canada, CIC 2006; MDDEFP 2013). In 2011, wetlands covered only 4 % of the YRW (Fournier et al. 2013). Several indicators show a grossly inadequate proportion of wetlands, particularly considering the high levels of pollution that have been recorded for the Yamaska River, and the need for water flow regulation and

Table 1 Geographic and demographic characteristics of the Yamaska River and Bécancour River Watersheds

Characteristics	Yamaska River Watershed	Bécancour River Watershed
Geographic characteristics (Labbé et al. 2011; Varin 2013)		
Total area (km^2)	4784	2620
Agricultural area	2033 km^2 (43.0 %)	554 km^2 (21.0 %)
Urban area	338 km^2 (7.1 %)	89 km^2 (3.4 %)
Forest area	1690 km^2 (35.0 %)	1407 km^2 (54.0 %)
Wetland area	193.27 km^2 (4.0 %)	297 km^2 (11.0 %)
Number of wetlands	3168	4990
Demographic characteristics[a] (ISQ 2012a–c)		
Population	1,456,743 individuals	639,910 individuals
	600,220 households	271,157 households
Average persons/household	2.43	2.36
Average income[b] ($)	27,705	24,854
Proportion of men (%)	50	50
Average age	40.4	41.7

[a] Data for the regions of Montérégie, Centre-du-Québec and Chaudière-Appalaches, respectively
[b] 2014 US dollars (PPP)

extreme flood prevention (CIC 2006). Natural bogs, swamps and marshes are the most representative types of wetlands that are still in place (CIC 2006).

Bécancour Lake is the source of the Bécancour River, which drains into the St. Lawrence River across from the city of Trois Rivières. Forest dominates the upstream portion of the Bécancour River Watershed (BRW), while the downstream portion is located in the lowlands and is more agricultural. Some municipalities have a high density of farm animal production. Forage crops dominate nearly two-thirds of crops that are grown in the watershed, which explains the reduced effects of agriculture on water quality compared to the YRW. A feature of the BRW is the production of cranberries (*Vaccinium macrocarpon*), which grow naturally in bogs and fens. Although cultivation is not exclusive to these environments, hundreds of hectares of these wetlands are used for the cultivation of this fruit (Morin and Boulanger 2005). The population density of the BRW is less than half of that of the YRW (see Table 1) and, unlike the latter, is declining over time. More than half of the population of BRW resides in the towns of Thetford Mines, Plessisville, and Princeville. Another difference distinguishing BRW from YRW is the presence of large areas of wetlands in the BRW, mainly in the lowlands. Over two-thirds of these areas are peat lands (Morin and Boulanger 2005). In 2011, wetlands covered about 10.6 % of the BRW (Fournier et al. 2013).

Between 1984 and 2011, wetland area decreased by 995 ha in the YRW and by 1044 ha in the BRW,

which corresponds to an annual decrease of 0.44 and 0.30 %, respectively; the rates of decline show no signs of slowing down (Fournier et al. 2013). This decrease in total area of wetlands has inevitably led to a loss of ecological functions that are provided by these ecosystems. Bogs and swamps within the study watersheds are known to provide many essential EGS, including flood control, sediment capture, and storage and sequestration of atmospheric CO_2, together with refuges and habitat provisions for biodiversity (Turner et al. 2000). Assessment of the economic value of these EGS is an important tool when communicating their importance to the public and to policy makers.

Benefit transfer method

A study of benefit transfer that was based on the approach of meta-analysis proceeded in three steps. During the first step, our study involved careful selection of primary studies to construct our meta-analysis database. This database was further expended to include the relevant economic, geospatial and environmental variables specific to the areas that had been evaluated in the selected primary studies. Second, using our database, we estimated the coefficients that were associated with each geospatial, environmental and socio-economic characteristic in the wetland EGS value determination function. Last, the estimated model was used with the specific characteristics of the test areas in the YRW and BRW to obtain the EGS value that was provided by their wetlands.

Data preparation

The selection of the primary studies was based on the quality of the publication (peer-reviewed journals) and the possibility that reported EGS values could be standardised. The database that was compiled from primary studies was further enriched by specific spatial variables from the test areas. To date, the inclusion of baseline site-specific spatial variables is a distinctive element that has been rarely used in meta-analysis. For example, Brouwer et al. (1999), Wood-ward and Wui (2001), Moeltner and Woodward (2009), and Brouwer (2009) did not consider spatial information for determining wetland EGS values. Our study also differed from previous studies that included spatial variables (e.g., Brander et al. 2006, 2012a, b; Ghermandi et al. 2010) by integrating spatial variables that were not used previously, such as wetland type (complex or isolated) and land cover type (% of agriculture, urban, forest and water) within a 10-km radius around the wetlands. Some of the spatial data were compiled using external sources (e.g., Google Earth, RAMSAR—http://www.ramsar.org/library).

EGS values/ha/year that had been reported in the selected studies were converted into US dollars (USDs) based on purchasing power parity (PPP) rates. We used data from the Penn World Table (Heston et al. 2012) to consider differences in purchasing power among countries and price deflators from the World Bank (2012) to standardise values to a constant price. For studies that provided only the value of the WTP per individual or household, we extrapolated their aggregate value or their total value depending upon the population in the study. When the total size of the population concerned was not directly given in the article, the relevant data were obtained from external sources. The aggregated wetland EGS value was further divided by the area of the baseline wetlands and converted to a per-hectare value. The final economic value can be a marginal, median or mean value, depending upon the methods used in the studies. In our primary studies, marginal values are associated with projects that increased the size of the wetland, while averages consider the value of its existing area. Integration of the marginal value into meta-analyses can also be found in Brander et al. (2006, 2012a), and Ghermandi et al. (2010). The marginal value often corresponds to stated preference approaches; for example, a contingent valuation study can identify

the WTP of individuals for a project that would increase the size of wetlands. In such cases, the per hectare-EGS wetlands value was obtained by dividing the total WTP of the population concerned by the proposed increase in the size of wetlands.

Meta-analysis

The basic model of the meta-regression that was used to estimate the value/ha/year for wetland EGS is specified as follows:

$$\ln Y_i = a + b_{SERV}X_{SERVi} + b_W X_{Wi} + b_{GEO}X_{GEOi} + b_{ECO}X_{ECOi} + b_{Type}X_{Typei} + u_i, \quad (1)$$

where the subscript i refers to each observation in our database that had been compiled from the selected studies, a is a constant and b are the coefficients to be estimated for the explanatory (independent) variables. The dependent variable Y_i is the value of EGS for observation i, measured in USD/ha/year. As is the case in most meta-analyses on the wetland EGS value, the economic values Y_i were transformed into natural logarithms.

Two reasons support ln-transformation of Y_i, the wetland EGS value/ha/year. First, Y_i exhibits wide variation (see Table 2), i.e., from tens of dollars to several hundred thousand dollars. Use of its ln-transformed value would facilitate regression on many explanatory variables, which are qualitatively dichotomous (1/0) or percentage points (0–100), except for income level and wetland area. Second, the use of ln-transformed Y_i in the estimation function implicitly suggests a multiplicative form for the contribution of different factors in the composition of the wetland EGS (c.f. Eq. 2). It means that the addition or deletion of a service or a characteristic also affects the contribution of other services or characteristics in the wetland EGS value.

The selection of explanatory variables was inspired by Brander et al. (2006) and Ghermandi et al. (2010), and by our intention to use geospatial characteristics of wetlands in the benefit transfer step. A description of each selected explanatory variable in Eq. 1 is given below:

- X_{SERVi} represented four EGS that were considered in our model, which were *flood control*, *sediment capture*, and *habitat biodiversity*, together with services related to *commercial activities*, e.g.,

Table 2 Mean values/ha, together with their standard deviations, which were attributed to wetlands by continent, climate and valuation method

	Number of observations	Value/ha/year in 2014 US$ PPA/ha/year	
		Mean	Standard deviation
Continents			
Africa	2	16.71	6.27
North America	29	158,958.34	355,809.09
Asia	21	59,674.46	212,347.09
Europe	34	36,452.59	57,769.84
Australia	20	7757.03	11,456.64
Biome			
Boreal	2	2452.76	2259.30
Subtropical	54	31,638.67	141,709.29
Temperate	34	127,806.98	289,434.10
Tropical	16	74,917.90	242,917.37
Methods			
Choice experiment	36	111,916.19	281,360.06
Contingent valuation	48	66,266.50	205,268.15
Market prices	5	614.88	970.38
Production function and net income	4	878.29	1441.49
Replacement cost	4	6607.56	6926.70
Travel cost	10	1473.70	2852.34

commercial fishing or hunting. Each of these four services was a binary variable (taking the value of 1/0) representing whether or not the wetlands provide the service. Except for the services related to commercial activities, the remaining three were EGS most frequently studied in the literature (Cedfeldt et al. 2000; Mitra et al. 2005). In principle, the existence of these services should increase the wetland EGS value and, therefore, we expected positive coefficients for them.

- X_{Wi} represented variables concerning the wetland type. We grouped them into three dummy variables: *isolated*, *complex* and *manmade*. It is difficult to know a priori the sign of the coefficients that are associated with isolated or complex wetlands. Inclusion of the man-made wetland type was derived from Ghermandi et al. (2010); we used a binary response (1/0) to identify man-made wetlands. Ghermandi et al. (2010) concluded that man-made wetlands have the highest value. Their explanation relies upon the fact that man-made ecosystems are usually constructed with the

specific purpose of providing services for humans and, thus, their value is more easily recognised by the local populations. Following Ghermandi et al. (2010), we also expected a positive effect of this variable on the wetland EGS value because their construction is the consequence of human interests.

- X_{GEOi} included three selected geographical characteristics of wetlands: the percentage of the territory in urban and agricultural land cover (*urban* and *agricultural*) and the total size of the study area (*size*). For the variable that measures the percentage of urban land cover, we expected a positive coefficient. This result can be explained by two main factors. First, the scarcity of wetlands in the urban and peri-urban region increases their value. Second, the proximity of wetlands to urban or suburban areas generally allows more people to benefit from their ecological services. Therefore, a given wetland should have higher value when it is close to an urban area. Ghermandi et al. (2010) found that wetlands that were located in areas with

higher population density tended to have higher value. Following this reasoning, we would expect a negative sign for the coefficient of the variable measuring the percentage of agricultural land cover, since agricultural areas are often less densely populated. The variable *size* was a continuous variable measuring the total area (ha) of wetland under study. This variable also was employed in the respective studies of Brander et al. (2006, 2012a), and Ghermandi et al. (2010).

- X_{ECOi} represented economic variables that can affect wetland EGS values. Here, we include gross domestic product (GDP) per capita of the population that benefits from the EGS of the studied wetlands. The GDP per capita data was also transformed into USD based using the PPP ratio to measure the average income level of the concerned population. We anticipated a positive correlation of this variable with wetland value.
- X_{Typei} included three binary variables that were associated with the types of values in the primary studies and the types of valuation methods that were used, which were *marginal*, *median* and *stated preferences*.
- Finally, the error term U_i represented the residuals of the model.

To account for heteroskedasticity in the data, we used ordinary least squares regression with Huber–White Sandwich estimators (Huber 1967; White 1980). We also ensured that there was no collinearity in the model.

Benefit transfer applied to two Quebec watersheds

Based on estimates from the meta-regression model (1), we calculated the EGS values for the two target sites in Quebec. Such a benefit transfer consists of using the estimated coefficients of the meta-regression (1) and the corresponding specific characteristics of YRW and BRW in the calculation, which is described by the following equation:

$$
\begin{aligned}
\hat{Y}_j = &\exp(\hat{a}) * \exp\big(\hat{b}_{SERV} X_{SERVj}\big) * \exp\big(\hat{b}_{W} X_{Wj}\big) \\
& * \exp\big(\hat{b}_{GEO} X_{GEOj}\big) * \exp\big(\hat{b}_{ECO} X_{ECOj}\big) \\
& * \exp\big(\hat{b}_{Type} X_{Typej}\big),
\end{aligned} \tag{2}
$$

where j refers to each wetland that was assessed. For each wetland j, we used the characteristics described by X_{SERVj}, X_{Wj}, X_{GEOj}, X_{ECOj}, and X_{Typej} to calculate

EGS. Here, $\hat{a}$ was a constant and $\hat{b}_{var}$ were the coefficients that had been estimated by meta-analysis for the corresponding variables. As previously mentioned, taking the log of the wetland EGS value in the meta-analysis implicitly suggests that the addition or deletion of a service or a characteristic of wetlands also affects the contribution of other services/characteristics to the composition of the wetland EGS value, because the wetlands EGS value is a product of the contributions of different characteristics. This seemed an appropriate assumption because the co-occurrence of services can often increase wetland EGS values.

Results and their interpretation

Data

Our database included 106 observations from 51 studies that were chosen according to the quality of the publications and our ability to standardise the values. Table 2 shows the descriptive statistics of mean values in 2014 PPP USD/ha/year that were calculated for the 51 selected studies, by continent, climate and economic valuation method. Values vary greatly depending upon the continent or climate, with the highest values in North America and the Temperate Zone.

The methods of stated preferences (i.e., discrete choice experiment and contingent valuation) seem to generate higher values than do the other methods. This result can be explained by the fact that these methods incorporate non-use values that represent a significant portion of the total value. The non-use wetland EGS value is associated, for example, with the value of the habitat biodiversity, landscape aesthetics or existence value. Conversely, direct-use values are linked to the more limited use of wetlands for activities such as hunting or fishing. The portion of population that is positively affected by EGS greatly increases when considering non-use values. The total economic value thus obtained may increase substantially. Wattage and Mardle (2008) found that the proportion of aggregated preferences that were related to the use value was 55.3 % compared to 44.7 % for the non-use value to conserve a wetland ecosystem. Sander et al. (1990) calculated that within the total value of the 15 rivers in the state of Colorado, only 20 % was a direct use value (e.g., irrigation, swimming, fishing, tourism) and the difference was non-use value.

Table 3 Descriptions of predictor variables that were used meta-regression to predict EGS values

Categories	Variables	Mean (standard-deviation)	Summary description
Wetland ecosystem services	Habitat biodiversity	0.72 (0.45)	The function of wetland to contain a particular level of biodiversity and natural habitat
	Sediment capture	0.25 (0.43)	The function of wetland to filter water
	Flood control	0.15 (0.36)	The function of wetland to contribute to the management of floods and retention
	Commercial activities	0.32 (0.47)	The function of wetland to allow commercial activities that include commercial fishing, hunting or duck breeding
Type of wetland	Man-made	0.03 (0.17)	The wetland is not natural; it is constructed by humans
	Isolated	0.03 (0.17)	The wetland is isolated
	Complex	0.44 (0.50)	The wetland is situated within a complex of wetlands
Geographic characteristics	Agriculture[a]	45.30 (25.19)	The percentage of the territory that is in agricultural production, within a 10 km radius around the wetland
	Urban[a]	13.61 (11.40)	The percentage of the territory in an urban area, within a radius of 10 km around the wetland
	Ln-wetland size	8.97 (3.30)	Natural logarithm of wetland size in hectares
Socio-economic characteristics	Ln-per capita GDP[b]	10.09 (0.82)	Logarithm of per capita GDP in PPP 2014 USD
Type of value	Marginal	0.31 (0.47)	Economic value of wetland was determined for a marginal change
	Median[c]	0.03 (0.17)	The economic value of wetland reported in the primary study is a median
	Stated preferences	0.79 (0.41)	The study is based on either contingent valuation or choice experiment methods

[a] To ensure sufficient degrees-of-freedom for our estimates, we used the average of the variable when data were missing

[b] Determined on a national level, except for Canadian and US studies where the value is determined at the provincial or state level

[c] 50 % of respondents indicated a value lower and 50 % of respondents indicated a value higher than the median

The meta-analysis

The explanatory variables are detailed in Table 3 and the model results (Eq. 1) are shown in Table 4. Different combinations of explanatory variables were used in numerous estimations; here, we present the most consistent results according to the statistical criteria and the relevance to our objectives of transferring economic value to YRW and BRW. The use of Huber–White Sandwich estimators results in a normal distribution of the residuals of the model. The value of the adjusted R^2 value provides the explanatory power of the model, which is 53 %.

All coefficients that were associated with wetland EGS variables were significant at the 10 % level, except for the variable *sediment capture* (significant at 17 %). These coefficients supported the hypothesis of the significant impact of these variables on the wetland EGS value. In addition, the values of the coefficients

also provided information on the relative importance of each EGS in the composition of the wetland EGS value.

The coefficient that was associated with the logarithm of per capita GDP is positive and significant at 5 %, which is consistent with most wetland valuation studies. EGS are thus found to be normal goods and a higher per capita income generates a greater WTP for EGS that are provided by wetlands. This result partly explains why an equivalent wetland is worth more in a "rich" country than in a "poor" one.

Wetland size was a negative and highly significant coefficient (1 %) in relation to economic value. This result is consistent with those found by Woodward and Wui (2001), Brander et al. (2006, 2012a, b), and Ghermandi et al. (2010). The negative relationship associated with wetland size can be explained by a decreasing marginal utility: the larger the wetland, the smaller the new value of an additional ha of wetland.

Table 4 Results of the econometric meta-regression model

Categories	Variables	Coefficients	Standard deviation	P value	Confidence interval (95 %)	
Wetland ecosystem services (X_{SERV})	Habitat biodiversity	1.584*	0.920	0.089	−0.243	3.411
	Sediment capture	0.893	0.649	0.173	−0.397	2.182
	Flood control	1.485*	0.873	0.092	−0.249	3.219
	Commercial activities	1.899***	0.696	0.008	0.516	3.281
Type of wetland (X_W)	Man-made	2.505**	1.069	0.021	0.382	4.628
	Isolated	−0.856	1.698	0.615	−4.228	2.516
	Complex	0.868	0.579	0.138	−0.283	2.019
Geographic characteristics (X_{GEO})	Agriculture	−0.019	0.012	0.118	−0.043	0.005
	Urban	0.007	0.025	0.788	−0.042	0.056
	Ln wetland size	−0.560***	0.070	0.000	−0.699	−0.420
Socio-economic characteristics (X_{ECO})	Ln-per capita GDP	1.291**	0.544	0.020	0.210	2.372
Type of value (X_{Type})	Marginal	1.484***	0.529	0.006	0.432	2.535
	Median	3.004**	1.252	0.018	0.518	5.490
	Stated preferences	1.087	0.845	0.202	−0.592	2.766
	Constant	−3.668	5.312	0.492	−14.219	6.884
Number of observations		106				
R^2 adj.		0.531				

* Significant at 10 %, ** significant at 5 %, *** significant at 1 %

Regarding the types of wetlands, man-made wetlands were associated with a positive coefficient significant at 5 %, signifying a higher value than natural wetlands, which is consistent with the findings of Ghermandi et al. (2010). Complex wetlands had a positive coefficient (significant at 15 %), while the coefficient associated with isolated wetlands was negative, but insignificant. This result might seem at odds with a biological perspective, where an isolated wetland has greater potential to play a key role in a region than a complex of wetlands within the same area. Our result may be explained by the sociological orientation of the primary studies selected for this meta-analysis, where wetland EGS values are defined by individuals who may not be aware of the biological aspects of wetlands and, which therefore would suggest a higher preference for complex wetlands. According to their perceptions, complex wetlands would offer more valuable EGS. However, we believe that this aspect needs to be studied more thoroughly in order to draw better conclusions, especially since the coefficient for the variable *isolate* is not statistically significant.

The coefficients of the variables that were related to geographical features provided expected but less statistically significant results. For example, the percentage of agricultural land cover had a negative coefficient (only significant at 15 %) and the coefficient associated with urban land cover was positive, but not significant.

Finally, for coefficients associated with the three dichotomous control variables that identified the type of values and the type of methods, two of the three coefficients (*median* and *marginal*) were positive and significant. First, the *marginal* variable coefficient was positive and significant at 1 %, which meant that primary studies evaluating marginal value associated with the increase of the area of wetlands generally attributed a higher value to wetland EGS. This same conclusion was reached in the studies undertaken by Brander et al. (2006, 2012a), and Ghermandi et al. (2010). Brander et al. (2012a) noted that the positive coefficient of the marginal values (for a small change in wetland area) was less constrained by household income than total assessments (e.g., the total loss of a wetland), from which we estimated average values.

The coefficient of the binary variable *median* was included in the equation to capture the adjustments of the wetland EGS value for studies that reported a *median* instead of a *mean* value. The difference between median and mean values depends upon the distribution of the data in the primary studies: this variable had a positive coefficient sign if the *median* was greater than the average value, a negative sign in the opposite case, and was not significant if the *median* value was close to the average. In our study, a positive coefficient that was significant at 5 % for this variable demonstrate in fact a distribution of per hectare wetland values with few strong outliers. Lastly, the positive coefficient of the *stated preferences* binary variable, although not significant, illustrated another general trend that has been observed in previous meta-analyses, including those conducted by Brander et al. (2006, 2012a, b), and Ghermandi et al. (2010). These authors have reported higher economic wetland values assessed by stated preference methods, which are often due to the previously mentioned fact that these methods include both use and non-use values.

Benefit transfer applied to the two watersheds

We first conducted benefit transfers based on the general/average situations for the two study watersheds (Table 5). In the first column of Table 5, for comparison, we have reported the average per hectare value calculated directly from our meta-analysis database, while subsequent columns report the respective per hectare values of wetlands in the YRW and BRW. Here, we used mean values of geospatial characteristics of YRW and BRW, which were percentages of agricultural land cover (43.3 and 21.4 %, respectively), urban areas (7.3 and 3.5 %),

the percentage of isolated wetlands (65.6 and 78.8 %), and the percentage of complex wetlands (34.4 and 21.2 %). We also used average per capita GDP of the two watersheds: $25,485 for YRW and $21,542 for BRW in 2014 international USDs at constant price. In general, we found that the wetland EGS value was lower in YRW and BRW relative to the average value for test areas of the 51 primary studies that had been selected for our meta-analysis. This result could be explained by the absence of commercial activities associated with the target wetlands or of man-made wetlands; these are two features that would substantially increase the value. In addition, both watersheds had a greater percentage of isolated wetlands than the wetlands that were used for our meta-analysis. Finally, the size of wetlands for the two target watersheds was larger compared to the average size obtained from the database, another factor that drove down the per hectare value. When we compared per hectare values that were extrapolated for both watersheds, we can see that the value for YRW is higher than that for BRW. Two factors explain this situation. First, per capita GDP is lower in BRW and, second, the total size of the BRW wetlands is greater than that of YRW.

The use of an average value in the benefit transfer method that was applied to the entire watersheds masks variation in the data for the sub-watersheds. To perform a finer-scale benefit transfer at the level of the geographic units that are usually considered by watershed-level management, we divided watersheds into sub-watershed areas of about 50 km^2, based on hydrological boundaries. Variation in characteristics of the sub-watersheds is illustrated by the standard deviations (SDs) of each geospatial and environmental variable, the values of which are sometimes higher than the mean value of the corresponding variable, this

Table 5 Comparison of values/ha/year based on averages data for meta-analysis of the Yamaska River and Bécancour River Watersheds

	2014 USD		
	Meta-analysis	Extrapolations	
		Yamaska	Bécancour
Value/ha/year when all three services co-exist	20,057	5277	3979

The three services are sediment capture, flood control and biodiversity habitat. We impose an EGS value of 1 regarding the three services, which means that in our study, we consider the potential value of the YRW and BRW wetland EGS where these characteristics are present. To our knowledge, the watersheds have no commercial activities that would fit our description of such a service; consequently, the dummy variable representing commercial activities is set to 0

Statistics for 2011		Yamaska	Bécancour
Number of wetlands		3168	4990
Number of sub-watersheds		105	56
Wetland size (ha)	Total	18,785	27,864
	Mean (SD)	5.93 (20.16)	5.58 (22.90)
% Complex[a]	Mean (SD)	39.47 (23.82)	22.28 (9.41)
% Isolated[a]	Mean (SD)	57.68 (24.88)	77.72 (9.41)
% Agriculture	Mean (SD)	42.14 (25.08)	22.04 (13.10)
% Urban	Mean (SD)	7.16 (6.26)	3.56 (4.78)
Economic value/ha (2014 USD)	Total	$170,573,066	$131,019,418
	Mean (SD)	$9080 ($5326)	$4702 ($1620)

Table 6 Descriptive statistics and economic values that were calculated for the sub-watersheds

[a] The sum does not equal 100 because three sub-watersheds do not have wetlands

illustrate strong variation of the related variables among sub-watershed areas. Using these sub-watershed level spatial variables, we re-conducted a benefit transfer for each sub-watershed to better reflect the potential variability of the wetland EGS value among sub-watersheds. It has therefore been possible to estimate a new total value of $170,573,066 for the entire YRW and a total value of $131,019,418 for the BRW, which are the sum of the extrapolated wetland EGS values of the 50-km^2 sub-watersheds belonging to the same target watershed. The mean per hectare values that were thus calculated were respectively $9080 and $4702/ha/year.

Divergence of the mean per hectare values of wetland between Tables 5 and 6 actually reveals the potential sensitivity of benefit transfer methods with respect to the scale of the extrapolation. Using average characteristics of watershed cancels out the variability of wetland values among sub-watersheds. For this reason, we believe there are advantages to using more disaggregated data to obtain more precise wetland EGS value estimates.

Figures 2 and 3 illustrate the detailed variability of per hectare EGS values among the 50-km^2 sub-watersheds for the YRW and BRW separately. To simplify the figure, per hectare values of specific
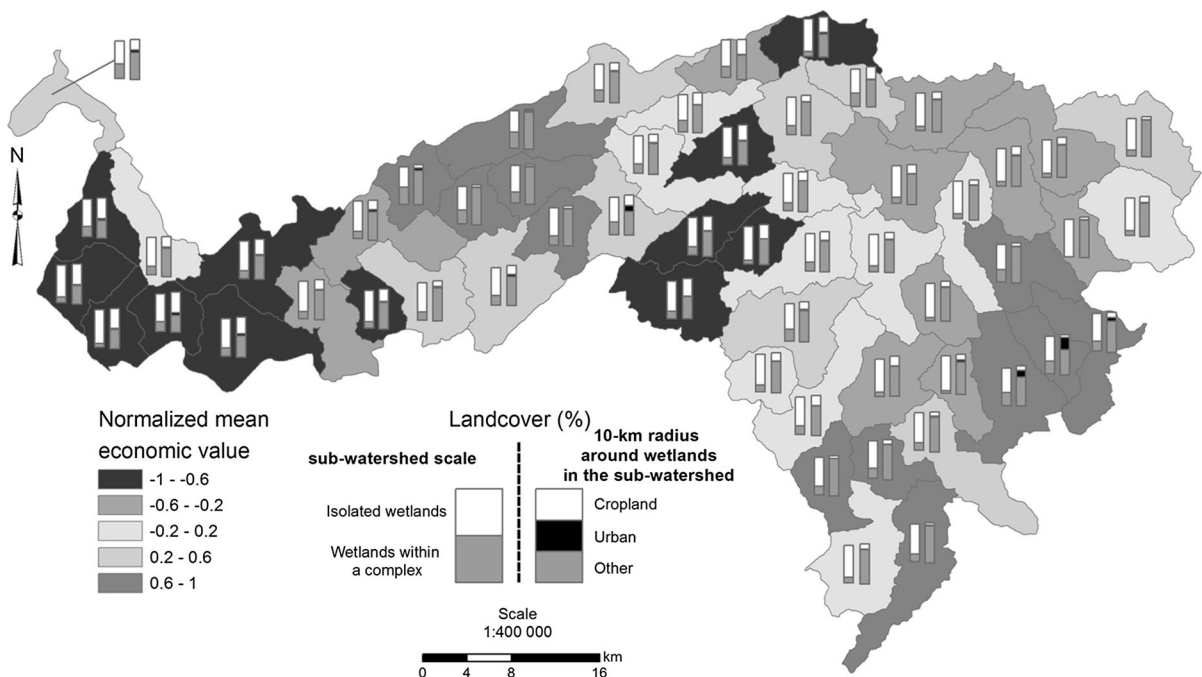
Fig. 2 Per hectare values and wetland characteristics for the Bécancour River Sub-watersheds (approximately 50 km^2)

wetlands within the sub-watersheds were normalised to the interval [−1, 1] and divided into quintiles: [−1, −0.6], [−0.6, −0.2], [−0.2, 0.2], [0.2, 0.6] and [0.6, 1]. In this way, sub-watersheds within the same watershed have per hectare values that differ greatly from one another according to their geographic, spatial and economic characteristics.

Clearly, most "low value" wetlands were in the western part of the BRW and the north-western part of the YRW, where dense farming practices were widespread (c.f. the bars associated to sub-watersheds that indicate the land cover % within a 10 km radius of the wetland) and where the percentage of isolated wetlands is very high. Sub-watersheds exhibiting the highest values included towns. For example, the towns of Princeville, Plessisville and Laurierville in the centre of the BRW, while the towns of Thetford Mines, Black Lake and Coleraine are located in the southeast. The towns of Granby and Bromont are located in the southern part of the YRW. The proportions of agricultural and urban areas within the sub-watersheds actually play an important role in reducing/increasing the average EGS values that were calculated for the sub-watershed levels compared to that was calculated for the whole watershed level.
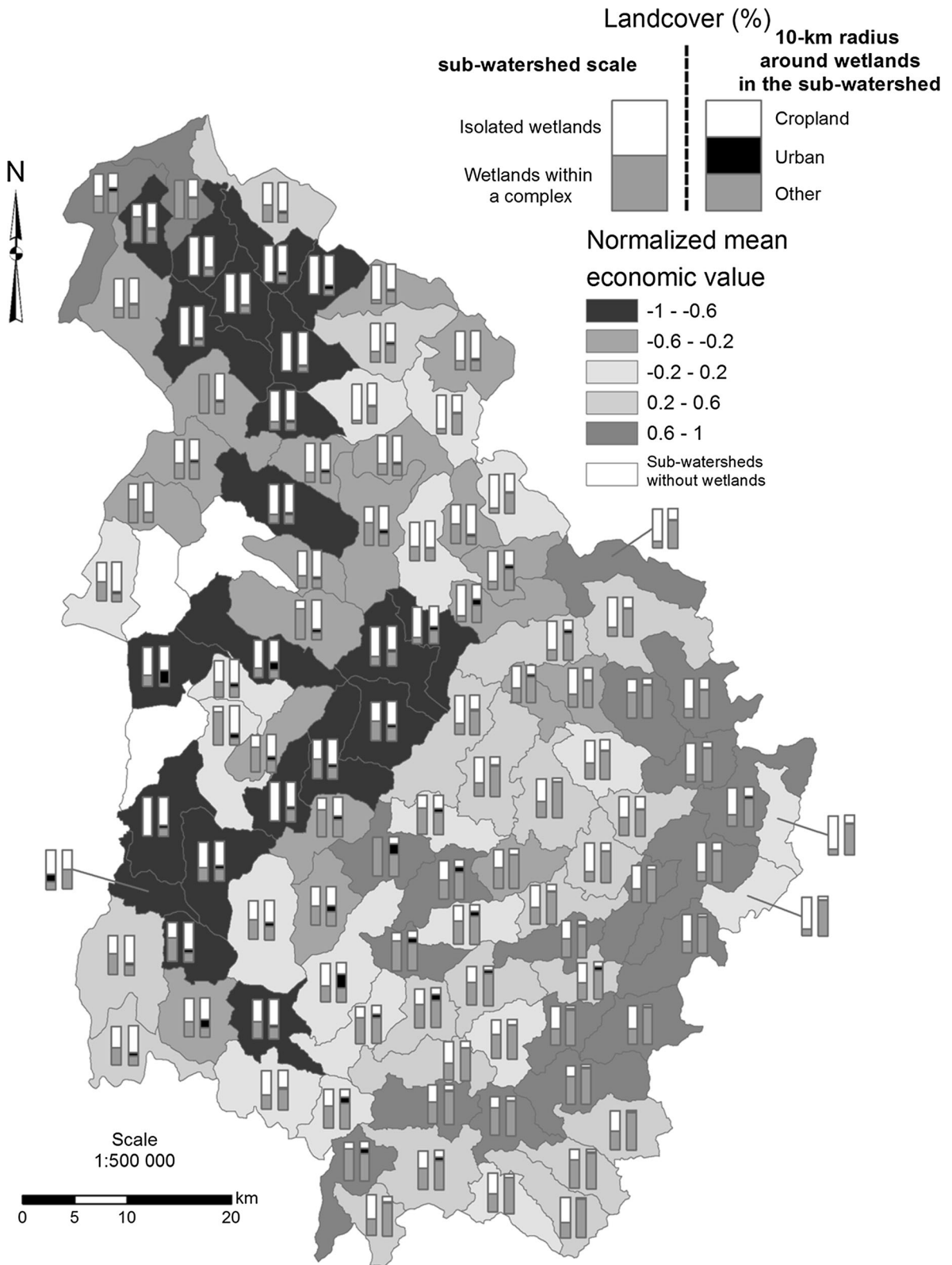
Since the choice of 'Type of value' can greatly influence the results, we checked their sensitivity according to the valuation method. To do this, we modified the values of two dichotomous variables: (1) stated preferences and (2) the marginal variable. From the modified variables, we constructed three different cases (Table 7). The first case represents the average wetland EGS value only from studies that use methods other than stated preferences and, therefore, contain only the use value. The second case represents the average of the related per-hectare EGS value when all valuation methods were combined together. The third case represents the EGS value extrapolated only from stated preference studies. Our extrapolated values for wetlands for YRW and BRW, according to studies using stated preferences methods, yielded values that were 10 times higher than those obtained with other methods. This result is very consistent with the findings of several previous studies, including Ghermandi et al. (2010) and Brander et al. (2012a, b).

A further comparison was conducted between the benefit transfer that was based on meta-analysis and the other two benefit transfer methods, i.e., simple benefit transfer and the transfer of function. Simple

benefit transfer directly uses the wetland EGS value (per household/ha) that is reported in one primary baseline study as the value for the target site wetlands, frequently after income-scale adjustment. The transfer of function directly bases the benefit transfer on the valuation determination function estimated by the previously identified single primary baseline study (Genty 2005). Both methods are based upon one single baseline study, which differs from the meta-analysis that depends upon a series of selected primary studies (Rosenberger and Loomis 2001; Genty 2005; Johnston and Rosenberger 2010). To conduct simple benefit transfer and the transfer of function, we chose Pattison et al. (2011) as the baseline study. This study was also part of the database used in the meta-analysis. Pattison et al. (2011) used the contingent valuation method to measure the value of five services (water quality, flood control, soil erosion, wildlife habitat, carbon capture and storage) in wetlands that were located in the Prairie Pothole Region of Manitoba (Canada). The choice of this study was motivated by its quality, geographic proximity (in Canada), and recent history. As yet, there are no primary evaluation studies assessing the wetland EGS value in Quebec. Since Quebec's wetlands do not share all the attributes of those in the Prairies, we should expect reduced the relevance for the benefit transfer based either on the direct value transfer or function estimate.

Comparisons between the different methods of benefit transfer are reported in Table 8. To extrapolate WTP per household/year, the simple benefit transfer was based on income scale adjustment between Manitoba and Quebec; the transfer function used coefficients from the econometric model of Pattison et al. (2011). WTP/household/year that was extrapolated from the two benefit transfer methods was then used to calculate the EGS values of wetlands in the two watersheds/ha/year, which are reported in Table 8. It should be noted that economic values obtained from the different methods of benefit transfer converged (Table 8), which may suggest that these were plausible values. Per-hectare values range from $16,874 to $17,899/ha/year for YRW, and from $10,005 to $13,722/ha/year for BRW. These different values corresponded reasonably well to those that are shown in the last column of Table 7, which were obtained by meta-analysis from the combined results of 42 studies using only stated preference methods.

Landcover (%)

sub-watershed scale

10-km radius
around wetlands
in the sub-watershed

Isolated wetlands

Wetlands within
a complex

Cropland

Urban

Other

Normalized mean
economic value

-1 - -0.6

-0.6 - -0.2

-0.2 - 0.2

0.2 - 0.6

0.6 - 1

Sub-watersheds
without wetlands

N

Scale
1:500 000

km

0   5   10   20

◀ **Fig. 3** Per hectare value and wetland characteristics for the Yamaska River Sub-watersheds (approximately 50 km²)

Discussion

The extended data at the sub-watershed level allowed us to capture the effects of geospatial and environmental characteristics of wetlands on the wetland EGS values. While the ability to link biophysical and EGS data is a relatively recent phenomenon (Kreuter et al. 2001; Troy and Wilson 2006), it provides an interesting perspective for decision makers who would have access to a spatially explicit analysis of the regions and sub-regions. Brander et al. (2012a, b) proposed a methodology for calculating changes in well-being due to variation in geospatial features of wetlands on a large geographic scale. Our approach differs from these studies by implementing a method that allowed us to integrate spatial variables that have not been previously used, such as the type of wetland (complex or isolated) or land use (% of agricultural, urban, forest and water land cover) within a much finer

geographical scale of 50-km radius. This information was particularly useful in the application of benefit transfer because it allowed us to obtain greater accuracy in when extrapolating wetland EGS values.

In a context where it is increasingly common to use spatial data, our study showed that the flexibility and adaptability of the value determination model, which had been obtained from a meta-analysis, proved to be a potentially interesting tool for providing useful information that would guide decision-making in natural resource protection. Decision makers can use other spatial and socio-economic data and insert those values into the model to investigate heterogeneity in wetlands ecosystem values. This information can also be combined with the expertise of biologists when land-use decisions involving wetlands have to be taken. In addition, our methodology provided different values depending upon the wetland geospatial features; this approach respects the heterogeneity of the landscape and suggests interesting innovations for EGS valuation mapping and modelling (Troy and Wilson 2006; Eigenbrod et al. 2010; Schägner et al. 2013). For example, we can consider the effect of

Table 7 Influence of the type of valuation method on the per hectare value according to data that were collected for the Yamaska River and Bécancour River Watersheds and the averages that were calculated from the database (values/ha/year in 2014 USD)

Cases[a]	Case 1	Case 2	Case 3
Type of valuation method	Other than stated preferences	All methods (as in database)	Stated preferences
Yamaska	1291	5277	16,874
Bécancour	974	3979	12,722
Meta-analysis	4906	20,057	64,134

[a] Here, we assume that the three EGS (biodiversity habitat, sediment capture and flood control) are all present

Table 8 Summary of economic values obtained using different methods of benefit transfer

| Methods | Wetland EGS value (2014 USD/ha/year) | |
	Yamaska River Watershed	Bécancour River Watershed
(1) Simple benefit transfer adjusted according to the total area of wetlands[a]	16,878	13,272
(2) Transfer of function[a]	17,899	10,005
(3) Meta-analysis[b]	16,874	12,722

The area of wetlands that would be lost within 12 years in each watershed has been calculated according to a status quo scenario and are 995 and 1044 ha for YRW and BRW, respectively

[a] Benefit transfers from Pattison et al. (2011)

[b] Benefit transfer from this study

scarcity that is associated with urban or peri-urban wetlands, which increases the per hectare value of wetlands, or the significantly lower EGS value for the wetlands that are located in close proximity to intensive agricultural activities. Our methodology therefore has the advantage of not being based upon a single average value/ha, but based upon several values that depend upon the geospatial and environmental context.

Our final database contained a total of 106 observations from 51 studies. Although this number of observations was similar to that of Brander et al. (2012b), who used 130 observations from 41 studies, we agree that the relatively small number of observations and studies in our database was limiting when comparing it to Ghermandi et al. (2010), who used 170 studies. One principal reason is that our study gave priority to studies that were published in peer-reviewed journals; given that only three non-published studies were selected for inclusion in our database, we believe that this more restrictive selection strategy regarding primary studies allowed us to avoid potentially greater biases from the primary studies, which could be increased by the transfer method (Wilson and Hoehn 2006). Therefore, this approach would increase the quality of the benefit transfers that were proposed in our study.

The priority given to peer-reviewed journal publications in the selection of primary studies is not without sacrifices. One related weakness is the necessity for including a small number of valuation studies on coastal mangroves and wetlands, together with studies from very dissimilar continents, e.g., Africa. Some peculiarities of coastal mangrove and Africa wetlands, which are not shared by temperate wetlands in Canada, could lead to lower transferability efficiency. Ideally, the meta-analysis should take a subset of wetlands studies that would be more comparable with the two watersheds that we aimed to value. The co-application of the two selection criteria, which would be the quality of the study and the similarity of wetlands, however, would drastically reduce the number of usable studies and the necessary variability required of the spatial/environmental variables. Therefore, meta-regression with richer set of predictor variables similar to what we have proposed in this paper would become unfeasible. For this reason, the wetland value determination function that we obtained in the paper should be regarded as an average based on

international experience. However, the close per hectare values of Quebec wetlands that were estimated when doing a benefit transfer with our meta-analysis and with other benefit transfer methods (c.f. Tables 7, 8) would seem to imply that the potential bias caused by inclusion of this international average experience is relatively small.

Another limitation of our meta-analysis and the benefit transfer was the use of binary variables in estimating the value function. Continuous explanatory variables would probably explain greater variability in the wetland value if they were available (Brouwer 2000). This is particularly true for the variables that describe the wetland EGS, which indicate only the existence of services with little information on their quality. The inclusion of qualitative dichotomous variables could potentially reduce the explanatory power of our econometric model and, therefore, the accuracy of our study. However, the wetlands selected in our meta-analysis database did not include quantitative quality indicators for the EGS provided by the studied wetlands. The integration of EGS quality into wetland valuations is an area for future research.

The use of the dummy variable "man-made" wetland in our meta-analysis study as explanatory variable for wetland value may also create potential conflicts within the context of climate change policies, where the impact of a program should be considered on larger geographical scales. Although a man-made wetland can be beneficial for its local population, its potential unintended consequences should not be ignored. These consequences include disruptions to flow, or more severe flooding events in other areas, for example. Following such considerations, we should consider the wetland value that was extrapolated by our study as an upper limit when it concerns a man-made wetland. However, for the two Quebec watersheds studied in our paper, our extrapolation was not affected, give that there were no man-made wetlands.

From a broader perspective, we also note that meta-analysis studies depend heavily upon the availability of data in each of the primary studies, which very often contain some specific characteristics of particular environments, such as the presence of endangered species or the cultural value of an environment. As a result, the choice of primary studies in the meta-analysis may result in a systematic transfer bias, which is an error analogous to the measurement errors in the primary studies (Brouwer 2000). This bias is a

consequence of non-compliance with implicit assumptions in sample choices, which are (1) that the selected primary studies respect the characteristics of a non-biased random sample of the existing valuation literature, and (2) the empirical estimates provide non-biased representative EGS values. In reality, researchers often tend to study wetlands that have a higher value due to political or scientific interests. This may result in a non-representative set of EGS wetland economic valuation studies. Hoehn (2006) indicated that this selection bias could result in an upward bias up to four times higher for the wetland EGS value estimation.

Conclusions

Climate change will cause major changes in ecosystems, which are likely to be further complicated by economic and political initiatives, such as the development of the Utica Shale natural gas deposits in Quebec. Given that such developments are drawing more and more public attention, it is crucial that management decisions for these ecosystems internalise the economic value of the services that they provide. Our study has combined geospatial and environmental data for a benefit transfer calculation based on meta-analysis. It aimed to determine the values for the EGSs provided by wetlands in Quebec. Overall, our results report the respective average values/ha/year for EGS that were provided by the wetlands of the YRW and BRW were $9080 and $4702 (with SDs of $5326 and $1621).

By integrating the details of the geospatial and environmental characteristics at the level of sub-watersheds of 50 km^2, the benefit transfer conducted in our study is also able to report detailed EGS values for each of the 105 sub-watersheds in the YRW and the 56 sub-watersheds in the BRW. By increasing the accuracy of value extrapolation and identifying the wetland units that need urgent protection and restoration interventions, our method provides another useful way for ensuring more effective wetland management.

Finally, the inclusion of the sub-watershed level geospatial and environmental conditions in extrapolations of the values of wetland EGS emphasised the sensitivity in the mean per hectare values of wetland when the scale of the extrapolation was altered. Using average characteristics of watershed cancels out the variability in wetland values between sub-watersheds and, for this reason, we believe it is more advantageous to use detailed sub-watershed data when these are available to obtain more precise wetland EGS value estimates.

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