**Abstract**

Globally, the extent of inland wetlands has declined by approximately 70% since the start of the 20th century, resulting in the loss of important wetland-associated ecosystem services. To help inform wetland conservation and educate the public, there is a need for a global estimate of the value of wetland ecosystem services within agricultural landscapes. Here, we examine the contextual drivers of wetland values in agricultural landscapes to increase the effectiveness and reliability of benefit transfer tools to assign values to local wetland ecosystem services. We systematically reviewed 668 published studies that analyzed or documented wetland ecosystem services within agricultural environments globally and identified 45 studies across 22 countries that provided sufficient economic information to be included in a quantitative meta-analysis. We developed meta-regression models to identify main drivers of provisioning (e.g., food, building materials, firewood) and regulating (e.g., carbon sequestration, nutrient recycling, flood control) services. Provisioning wetland ecosystem values were best explained (direction of effects in parenthesis) by per capita income (proxied by a high-income binary variable, +), peer-reviewed journal publications (+), agricultural total factor productivity (-) and population density (+), while agricultural total factor productivity (-), income level (+) and wetland area (-) had significant effects on regulating wetland ecosystem values. Our models could help to estimate wetland values more reliably across similar regions and thereby help to inform wetland conservation actions and policies on agricultural landscapes.

**Keywords:** Agricultural landscapes, benefit transfer, provisioning ecosystem services, regulating ecosystem services, meta-regression, wetlands.

1. **Introduction**

The global extent of inland wetlands has declined almost 70% during the 20th century mainly due to land cover change for agricultural production (Davidson, 2014). This rate of wetland conversion has continued into the 21st century (Gardner et al. 2015) resulting in the loss of many important ecosystem services (Leemans and De Groot, 2003). Wetlands play an essential role in maintaining water quality by removing excess nutrients and pesticides, especially on agricultural landscapes, that can degrade downstream water quality (Vymazal, 2017). Wetlands also modify water quantity by storing water, regulating and recharging aquifers during wet seasons and thereby mitigating flooding in wet periods and supporting agricultural production during drier periods (Dixon and Wood, 2003). The role of water regulation is particularly crucial for conserving freshwater, a critical resource for community welfare and agricultural production. Other identified wetland ecosystem services include carbon sequestration, recreation, tourism, human and livestock foods, and habitat to support diverse biotic communities (e.g., Davies et al. 2008; Badiou et al. 2011; Gleason et al. 2011; De Groot et al. 2012).

Wetland ecosystem services have many of the characteristics of public goods and are often not traded in markets (e.g., habitat for biodiversity, water quality) and there is an incomplete understanding of the link between changes in ecosystem structure and function, and the goods and services that are produced for society (Mitsch and Goesselink 2000; Brander et al. 2006). As a result, it is often challenging to quantify a monetary value for many wetland ecosystem services that could be used in cost-benefit and tradeoff analyses, land-use planning, and wetland conservation policy development. To overcome this hurdle, a range of methods have been tested and adapted to estimate the monetary value of wetland ecosystem services, hereafter referred to as wetland values.

Due to time and budget constraints, it is not always possible or efficient to conduct site-specific studies to estimate wetland values. In these cases, benefit transfer methods may be used to supply information from comparable areas on ecosystem service values for policy decision-making. Richardson et al. (2015) identified three main benefit transfer methods: (1) a unit transfer function, (2) a benefit transfer function, and (3) a meta-analysis transfer function. A Richardson et al. (2015) suggested the meta-analysis method, which uses rigorous quantitative methods to analyze multiple empirical studies, may produce the most reliable benefit transfer values. Several studies have conducted meta-regression analysis on the value of wetland ecosystem services (Brouwer et al. 1999; Woodward and Wui, 2001; Brander et al. 2007; Ghermandi et al. 2010; Mitsch and Gosselink, 2000; Brander et al. 2006; Chaikumbung et al. 2019). These studies, however, did not focus on agricultural wetlands with the values of wetlands in agricultural landscapes often overlooked or misrepresented. Moreover, since wetlands are increasingly being converted to annual crop production in agroecosystems (Watmough and Schmoll 2007; Oliver et al. 2015; Peimer et al. 2017), we urgently need more comprehensive valuations on management alternatives to protect and restore wetlands in agricultural regions and other high valued resource areas (e.g., Turner et al. 2021).

The incentive to drain wetlands for agricultural production in developed countries has been driven by factors such as the higher cost of field obstructions with the increasing size of agricultural equipment, and the lower cost of wetland drainage with tools such as Global Positioning System (GPS) technology (Cortus et al. 2011; De Laporte 2014). In developing countries, increasing human population pressures and climate change are also motivating land managers to convert wetlands to agricultural lands (Dixon and Wood 2003). However, few studies have focused on estimating all wetland values on agricultural landscapes, and so the overall estimated values of natural wetlands in agricultural areas are currently underestimated and therefore potentially misunderstood by the public. A notable exception is Brander et al. (2013) who conducted a meta-analysis on ecosystem services provided by wetlands in agricultural landscapes with an emphasis on three regulating ecosystem services: flood control, water supply, and nutrient recycling. These authors estimated the values (at the means) for flood control, water supply and nutrient recycling to be 6,923 US$/ha/year, 3,389 US$/ha/year, and 5,788 US$/ha/year, respectively.

The main objectives of this study are to estimate wetland meta-regression functions for factors that drive the value of wetland regulating services and wetland provisioning services on agricultural landscapes, and to examine the potential for using these functions to guide the benefit transfer of wetland values in agricultural landscapes globally. Our study builds on the work of Brander et al. (2013) by including additional wetland regulating services and provisioning services, creating a more comprehensive analysis of all wetland values. Since the ecosystem services in the separate meta-regressions will be comparable in the way they regulate environmental processes or provide goods and services to society, and do not overlap, we are able to avoid commodity inconsistency problems (Vedogbeton and Johnson, 2020). Commodity inconsistency occurs when total ecosystem values in meta-regression analyses incorporate a broad range of wetland ecosystem services which often overlap and are difficult to compare due to their different impacts on society (Brander et al. 2013). Commodity inconsistency, which could cause biased meta-regression estimates and incorrect inferences or benefit transfers, has been identified as a problem in previous wetland ecosystem value studies (Brander et al. 2013; Vedogbeton and Johnston, 2020).

**2. Methodology**

2.1. Systematic Review

We completed a quantitative review of the results from published studies that analyzed or documented specific ecosystem services of wetlands within agricultural landscapes. A list of 668 research articles published prior to 2020 was generated using the keywords ‘ecosystem service OR economic’ AND ‘agricultural wetlands OR agriculture AND wetlands’ in the database of ISI Web of Science and with the Environmental Valuation Reference Inventory.

From these 668 papers, we examined each title and abstract to determine whether papers met the following criteria for inclusion in the meta-analysis: (i) measured quantifiable effects, (ii) provided the extent of wetland area change, (iii) listed a study location, and (iv) referred to wetlands in an agricultural context. This screening process identified 192 papers, which were reviewed in full to determine whether they contained relevant and usable data on agricultural freshwater wetlands. Papers were excluded if they measured coastal wetlands, peatlands or constructed artificial wetlands for waste management systems. From this subset, papers were excluded that did not provide (i) sufficient data to indicate overall trends/patterns, (ii) sufficient detail about wetland ecosystem service values or (iii) area of wetlands or information that enabled wetland area estimation.

The final database consisted of 45 papers, out of which at least 52% were peer reviewed publications. Five papers were split into multiple entries since they reported multiple study locations across 10 countries. Based on this set of 45 papers, we recorded geographic locations, study coordinates (if not reported, Google Earth was used to identify the coordinates), study year(s) (if study year was not reported, publication year was used), wetland area, the method used to value ecosystem services, the ecosystem services measured, and quantifiable effects of wetlands and their economic value when provided. Ecosystem services were classified into regulating and provisioning ecosystem services, following Morris and Camino (2011). We converted all wetland values to US dollars using the respective country’s exchange rate to the US$ at the time the study was conducted. Then, we multiplied the wetland values by the ratio of the consumer price index of 2018 to the consumer price index of the year the study was conducted to convert all values to US$2018/ha/year.

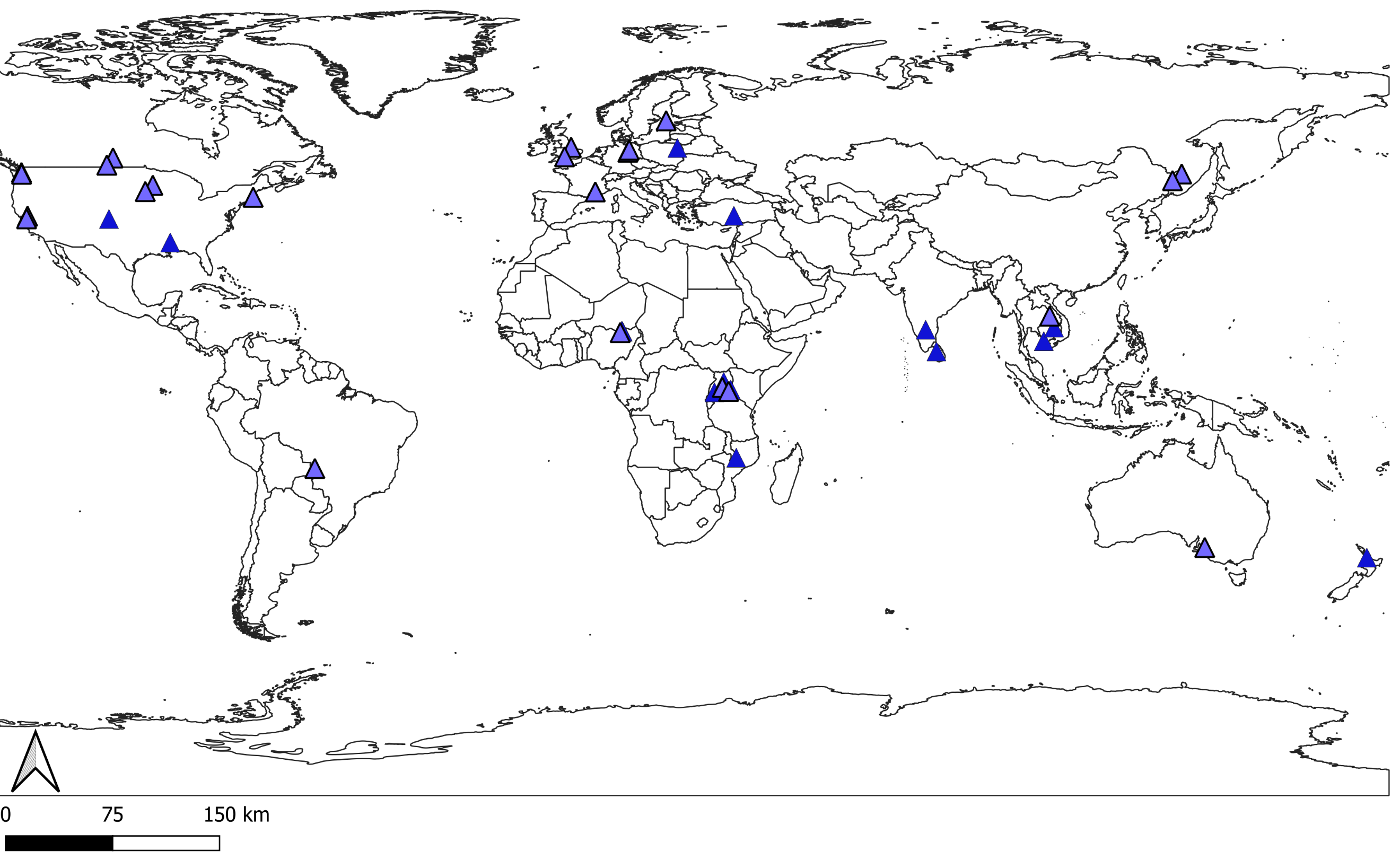
Carbon sequestration was estimated in tonnes C02/ha, and sequestration potential was then compared to values as determined by Canu et al. (2015). Local economic values (or geographically and economically similar ones) were used to determine the local monetary value of carbon sequestration. Since we measured possible benefits from carbon sequestration we acknowledge that these are upper bound values and would need to be offset by variable production of greenhouse gases. For instance, converting wetlands to cropland may produce even more greenhouse gases (depending on the production system). We also did not include peatlands in the study as we were focused on agricultural lands. Again, we also did not report emissions in each study location because we used sequestration data calculated by the original study.

Wetland water storage was estimated by the storage capacity (m3) of water/ha. In some studies, the total surface area was provided and the volume was calculated from the area and average depth if mentioned in the paper. If a monetary value for water storage was not provided, then global averages reported in De Groot et al. (2012) were used.

Nitrogen filtration was predominantly reported in North America, or regions with similar economic and environmental conditions. As such, the ability of wetlands to filter nitrogen was estimated in kg N/ha of wetland, and the monetary value was estimated by averaging the values that had been provided and applying them to papers where an effect was provided but with no accompanying monetary estimate.

Provisioning services were valued by amalgamating services that included food, building materials, crafting materials, or firewood. Because of the considerable variation in the types of provisioning goods and services, the overall value of each material was converted into monetary terms (2018 US$/ha/year). Provisioning wetland ecosystem service studies were conducted mainly in developing countries (e.g., Africa; see Figure 1), while regulating service studies were conducted mainly in developed countries, particularly North America (Figure 1). See Table A1, in Appendix 1, for a list of the primary studies used in this study.

**Figure 1. Distribution of Study Sites for Provisioning and Regulating Wetland Ecosystem Services[[1]](#footnote-1)**



**2.2. Empirical Model**

Meta-regression involves the application of regression analysis to a pool of comparable empirical estimates (Nelson and Kennedy 2009; Richardson et al. 2015). We regressed the wetland values (US$2018/ha/year) extracted from the systematic literature review on a vector of covariates representing national wetland policies, economic indicators, biodiversity richness indicators, and study characteristics.

We compared log-log and log-linear functional forms to estimate our meta-regression model. For the log-log, we took the logarithms of the dependent variable and continuous explanatory variables to improve model fit and prevent heteroscedasticity (Brander et al. 2013); we took only the logarithm of the dependent variable in the log-linear functional form. In the case of the log-log functional form, the coefficients of explanatory variables are interpreted as elasticities, which show that for continuous explanatory variables a 1% change in the variable will result in more than a 1% change in the dependent variable (for elastic effect) or less than a 1% change in the dependent variable (for inelastic effect); the coefficients in the case of log-linear function form represent a unit change in the dependent variable for a percentage change in the independent variables. When the regressor is a binary variable, the effect is compared to its reference group.

Since multiple observations were reported for some of the studies, we initially developed a mixed effects model to explain variation in wetland values. A general specification of a mixed effects model (with study as a random effect) is given in equation 1:

where:

i = subscript i represents the ith observation.

j = subscript j represents the jth study.

= dependent variable representing the logarithm of the value of wetland ecosystem service

(US$/ha/year).

= vector of independent variables (including wetland policy variables, human population

and economic indicators, and biodiversity richness indicators) and a constant term.

**=** vector corresponding parameters of **X** to be estimated.

= stochastic error term for the jth study, which is assumed to be normally distributed with

mean 0 and a variance (.

= stochastic error term for the ith observation, which is assumed to be normally distributed

with mean 0 and a variance (.

We used a likelihood ratio statistic to test for the appropriateness of the mixed effect model (Dias and Belcher, 2015); an ordinary least squares model with fixed parameters is estimated if the mixed effects model is rejected. Two separate provisioning and regulating models with the same functional form as equation (1) are estimated using frequentist estimation procedure, with the “LMER” and “LM” R statistical software packages, for the mixed and fixed effects models, respectively. The dependent variable for the provisioning model was the logarithm of the total value of provisioning ecosystem services, while the dependent variable for the regulating model was the logarithm of the total value of regulating ecosystem services. The sample sizes for the provisioning and regulating models were 27 and 22, respectively, and we tested for heteroscedasticity using Breusch Pagan test and multicollinearity using the variance inflation factor. A heteroscedastic model means the variance of the observation level error term is non-constant which would cause inferences from our model to be unreliable. Multicollinearity would reduce the efficiency of parameter estimates and undermine their statistical significance; however, it does not affect the reliability of parameter estimates. A variable inflation factor < 10 signifies that an explanatory variable is not a source of multicollinearity.

The final functional model that will be chosen will have the lowest root mean square error (RMSE) and mean absolute error (MAE) prediction error metrics. We used a 10-fold cross validation procedure to estimate the prediction error metrics. For the 10-fold cross validation procedure, we 1) randomly divided the data into 10 equal groups or folds, 2) chose one of the folds as holdout test data, and estimated the model with the remaining 9 groups of dataset (k-1 folds); the prediction error metrics were estimated with the holdout test data, 3) repeated the process 10 times, using a different set of holdout test data each time, and finally 4) used the average of the estimated prediction error metrics (RMSE and MAE) from each iteration of the 10 fold cross validation procedure as the final statistic. The prediction errors from the estimated models are called meta-regression benefit function transfer errors. The meta-regression benefit transfer errors are compared with mean value errors to show their potential for benefit transfer applications where wetland values are predicted outside this study. For the mean value transfer error, we estimated the prediction metrics by comparing the predictions from the models with the mean of the dependent variable.

**2.3. Description of Variables and Effects on Wetland Ecosystem Services**

Economic Variables

Human population density is expected to have a positive impact on both the values of wetland regulating and provisioning ecosystem services (Brander et al. 2013). To calculate human population density, we used a global gridded human population layer (1 km resolution) that modeled the distribution of human population using counts and densities in 2015 (Center for International Earth Science Information Network, 2017), and extracted the relative population density for each study location using bilinear interpolation with ArcGIS 10.5. Six study locations provided no data because the coordinates overlapped ‘no data' cells. For these, we calculated human population density by extracting the nearest density available to that point.

The income level of a country is expected to have a positive effect on the value of both provisioning and regulating services (Brundtland 1987; Brander et al. 2006; De Groot et al. 2012; and Peimer et al. 2017) since higher levels of wealth are positively correlated with social willingness-to-pay. From the income level variable, we created a high-income binary variable which was given a value 1 if the gross national income (GNI) in current 2019 USD was greater than $12,535 and 0 it was less (Serajuddin and Hamadeh, 2021). The other income groups (lower-income and middle-income countries) served as the reference group.

Agricultural total factor productivity (AgTFP) is a measurement of the average productivity of all the inputs (land, labor, capital, and material resources) used in the production of crops and livestock (Economic Research Service, 2019). The reference period of the AgTFP is 2015 (AgTFP = 100) such that AgTFP value of 120 in 2016 shows that over the 1-year, AgTFP has increased by 20%. Higher values of AgTFP would mean a more efficient agricultural production system which might need less resources (including agricultural lands) to produce agricultural commodities compared to the status quo (International Food Policy Research Institute, 2018). Therefore, agricultural productivity is predicted to have a positive effect on wetland ecosystem values (provisioning and regulating).

Biodiversity Variables

While wetlands are important habitats for many plant and animal species (Davies et al. 2008), studies that reported a biodiversity metric did not provide information to enable a standardized link to a monetary value estimate. This is a common challenge in the empirical literature as biodiversity is generally viewed as having a positive cultural and social value, but not generally monetarized or often incomplete due to lack of data or knowledge (Nunes et al. 2001).

The global species richness of birds was compiled from species range maps (≈ 28 x 28 km) by Birdlife International (http://www.birdlife.org/). The global species richness of amphibians (≈ 1 x 1 km) was compiled by the International Union for the Conservation of Nature and the Columbia University Center for International Earth Science Information Network (International Union for Conservation of Nature & Center for International Earth Science Information Network 2015a, 2015b). Study locations were overlaid with global species richness grids to calculate the total species richness at each site using bilinear interpolation with ArcGIS 10.5. Species richness (birds and amphibians) is expected to have positive effects on wetland values (provisioning and regulating).

National Wetland Policy

           No net loss wetland policy, deployed in several jurisdictions, seeks to maintain the total area of wetlands via wetland reclamation, mitigation and restoration efforts when the original wetland has been drained. This policy is expected to help conserve wetlands, and hence increase their benefits to society. This binary variable was 1 if a country has this policy and 0 otherwise. Similarly, binary variables for national ecosystem policy, use of incentives and use of penalties to conserve wetlands, are expected to have positive impacts on wetland conservation, and therefore wetland values. Country-specific policy information was obtained from Peimer et al. (2017). There may be regional differences in wetland polices within the same country; for instance, some provinces in Canada have no net loss policy whereas others do not. However, for this study we focus on overall country-level wetland policies.

Study Characteristics

Study-specific nuances or characteristics may influence the heterogeneity in wetland values (both regulating and provisioning ecosystem services). Study-specific variables included wetland area, peer-review journal publication status, valuation method, and geographic location (latitude and longitude). These variables are routinely added to meta-analyses (Brander et al. 2013). Wetland area, a continuous variable, is the size of the wetland that is being evaluated in a specific study. We expect wetland size to have a negative effect on wetland values, because people may be willing to pay the same for a small subset of an environmental feature as for a large area (Loomis et al. 1993). However, Reynaud and Lanzanova (2017) showed that larger lakes are more valued than smaller lakes, because some ecosystem services many require a minimum threshold of wetland area (Brander et al. 2013). The valuation method is a dummy variable which equals 1 if the valuation methodology is an economic valuation method and 0 otherwise. Economic valuation methods are listed in Woodward et al. (2001) and Brander et al. (2006) and include methods such as production function, replacement cost, and contingent valuation. Peer reviewed studies is a binary variable which takes on a value 1 if study is peer reviewed and 0 otherwise. We expect peer review to a positive effect on wetland values (Ghermandi and Nunes, 2013; Reynaud and Lanzanova, 2017) because researchers may be more encouraged to publish studies that produces more significant wetland values. The variable descriptions and their expected effects on wetland values are summarized in Table 1.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Variable** | **Description** | **Variable Type** | **Variable Unit** | **Expected Effect on Wetland Values** |
| **Dependent Variables** |  |  |  |  |
| Provisioning Model: | Provisioning wetland values | Continuous | 2018US$/Ha/Year |  |
| Regulating Model: | Regulating wetland values | Continuous | 2018US$/Ha/Year |  |
| **Explanatory Variables** |  |  |  |  |
| *Economic Variables* |  |  |  |  |
| Pop Density | Human population density | Continuous | H/km2 | + |
| AgTFP | Agricultural total factor productivity | Continuous | 2015 AgTFP = 100 | + |
| High-Income | High-income country | Binary | 1,0 | + |
| *Biodiversity Variables* |  |  |  |  |
| Birds | Bird Species Richness | Continuous | Counts/Ha | + |
| Amphibian | Amphibian Species Richness | Continuous | Counts/Ha | + |
| *National Wetland Policy* |  |  |  |  |
| NNL WP | No Net Loss wetland policy | Binary | 1,0 | + |
| ESS Goal WP | National ecosystem policy | Binary | 1,0 | + |
| Penalties | Use penalties to conserve wetlands | Binary | 1,0 | + |
| Incentives | Use incentives to conserve wetlands | Binary | 1,0 | + |
| *Study Characteristics* |  |  |  |  |
| Wetland | Wetland Area | Continuous | Ha | +/- |
| Economic Valuation Method | Economic Valuation Method | Binary | 1,0 | +/- |
| Peer Review | Peer Review Journal Publication Status | Binary | 1,0 | + |

**Table 1. Variable Descriptions and Expected Effects on Wetland Values**

**3. Summary Trends**

The estimated mean value for wetland-based provisioning ecosystem services was US$1,645/ha/year (in 2018 US$) with a standard deviation of USD$3,168/ha/year, indicating high variation in ecosystem service values across studies (Table 2). Also, the estimated mean value for regulating ecosystem services was US$8,711/ha/year with a standard deviation of US$22,375/ha/year. The mean and standard deviation of wetland area for the provisioning meta-regression model were 870,000 ha and 2,800,000 ha, respectively. Similarly, the mean and standard deviation of wetland area for the regulating meta-regression model were 2,730,000 ha and 6,600,000 ha, respectively. This shows that wetlands in the regulating meta-regression model, on the average, were relatively larger and more heterogenous in size than in the provisioning model. About 70% of the wetlands in the provisioning model were valued using an economic valuation method compared to 52% for the regulating model. The other non-economic valuation method was mainly ecological modeling where the ecosystem end points of wetlands were identified, and their values estimated using corresponding monetary values reported in the literature (see Canu et al. 2015 and De Groot et al. 2012).

In terms of the economic variables, the mean agricultural factor productivity variable in both models was 114; however, the heterogeneity in the values was greater in the regulating model. Considerably more studies conducted in high-income countries were included in the regulating model (70%) than in the provisioning model (37%). Conversely, about 22% of the studies in the provisioning model were conducted in low-income countries versus 9% in the regulating model. Also, the mean population density was greater for the study regions in the regulating model (1,003 humans/km2) and more heterogenous (standard deviation of 2,467 human population/km2) than for wetlands in the provisioning model with a mean and standard deviation of 755 human population/km2 and 2,223 human population/km2, respectively.

Moreover, concerning the wetland policy variables, more jurisdictions in the regulating model had identified a goal of conservation of ecosystem services (15% more), used an incentive-based policy to conserve wetlands (11% more), used penalties to conserve wetlands (33% more) and had a no net loss wetland policy (15% more) than jurisdictions in the provisioning model. This suggests that wetlands in the regulating model are supported by a more comprehensive conservation policy framework.

Considering the biodiversity variables included in the models there were more amphibians associated on average with wetlands in the provisioning model, (16.2 species/ha) and more heterogeneity in the values of the variable (standard deviation of 10.6 species/ha) than in the regulating model with a mean (standard deviation) species/ha of 12.39 (9.3). Also, there were more bird species associated with wetlands in the provisioning model (283 /ha) than wetlands in the regulating model (194 /ha).

**Table 2. Summary statistics for each variable used in the modelling[[2]](#footnote-2).**

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
|  | **Provisioning Model** | | | **Regulating Model** | | |
| **Model Variables** | **Mean (Standard Deviation)** | **Minimum** | **Maximum** | **Mean (Standard Deviation)** | **Minimum** | **Maximum** |
| **Dependent Variable** |  |  |  |  |  |  |
| Provisioning ESS | 1,644.79 (3,167.5) | 4.00E-04 | 12,341.87 |  |  |  |
| Regulating ESS |  |  |  | 8,711.23 (22,375) | 6.00E-04 | 1.04E05 |
| **Explanatory Variables** |  |  |  |  |  |  |
| *Economic Variables* |  |  |  |  |  |  |
| Pop Density | 754.91 (2,223) | 0 | 10,164.5 | 1,003.06 (2,467) | 0 | 1.01E04 |
| AgTFP | 114.59 (29.35) | 64 | 181 | 114.41 (918.35 | 64 | 148 |
| High Income | 0.37 (0.49) | 0 | 1 | 0.70 (0.47) | 0 | 1 |
| Low Income | 0.22 (0.42) | 0 | 1 | 0.09 (0.29) | 0 | 1 |
| *Biodiversity Variables* |  |  |  |  |  |  |
| Amphibians | 16.19 (10.60) | 2 | 44 | 12.39 (9.30) | 0 | 40 |
| Birds | 283 (127.99) | 97 | 544 | 194 (0.80.45) | 8 | 408 |
| *Wetland Policies* |  |  |  |  |  |  |
| ESS Goal WP | 0.78 (0.42) | 0 | 1 | 0.91 (0.28) | 0 | 1 |
| Use Incentives | 0.85 (0.36) | 0 | 1 | 0.96 (0.21) | 0 | 1 |
| Use Penalties | 0.44 (0.51) | 0 | 1 | 0.70 (0.47) | 0 | 1 |
| No Net Loss WP | 0.63 (0.49) | 0 | 1 | 0.78 (0.42) | 0 | 1 |
| *Study Characteristics* |  |  |  |  |  |  |
| WL Area | 8.7E05 (2.8E06) | 1.64 | 1.38e07 | 2.73E06 (6.6E06) | 0.7 | 2.7E07 |
| Econ. Val. Mthd. | 0.70 (0.47) | 0 | 1 | 0.52 (0.51) | 0 | 1 |
| Peer Review | 0.52 (0.51) | 0 | 1 | 0.65 (0.49) | 0 | 1 |

*3.2. Meta-Regression Results*

The likelihood ratio test statistics of 0.52 (p-value = 0.47) and 0.12 (p-value = 0.73) indicated that a mixed model (study as a random term) was not supported for the provisioning or regulating model structures, respectively. Therefore, the results of the fixed effect models, for both categories of wetland ecosystem services, are reported in this section.

*3.2.1. Provisioning Meta-regression Model*

We chose a restricted log-linear (Table 3) model as it produced the lowest meta-regression errors log-log model. The log-linear model was restricted because we dropped the variables that were highly correlated (*r >* 0.6) with other variables or were consistently not significant even at the 10% level for all the estimated models (this was necessary because of our small dataset). For this model, we dropped the variable representing economic valuation method since it was highly correlated with high-income (r = 0.68). We also, we dropped no net loss wetland policy, use of incentives and use of penalties in wetland policy because they were consistently not significant across all the estimated models. Overall, the final model was significant (F statistic = 5.88, p-value = 0.0009) and explained 65.3% of the variation in the dependent variable (log of provisioning wetland ecosystem values). The model was homoscedastic, with the variance of the error term being constant (Breusch Pagan statistic = 5.26, p-value = 0.87); multicollinearity was not detected since all explanatory variables had variance inflation factors < 10.

Population density and high-income both had positive effects on the provisioning wetland values, which were significant at the 10% and 5% levels, respectively (Table 3). The estimated coefficient of population density means that a 1% increase in density will result in a $0.0004 increase in provisioning wetland value; similarly, wetlands located in a high-income country would have about $2.324 more provisioning value than those located in other income groups. Agricultural factor productivity had a negative effect on provisioning wetland values (significant at 10% level); specifically, a 1% increase in agricultural factor productivity would cause about $0.028 reduction in the value of provisioning wetland ecosystem services. The provisioning value of wetlands in peer-reviewed journal publications was about $3.22 more than values in other studies (significant at 1% level). Ecosystem service goal (p-value = 0.57), longitude (p-value = 0.26), latitude (p-value = 0.31), birds (p-value = 0.11), wetland area (p-value = 0.66), amphibian (p-value = 0.56) were not significant at the 10% level. Moreover, the meta-regression benefit transfer errors for the provisioning model are about 0.71 and 0.70 lower (for root mean square and mean absolute error statistics, respectively) than the mean value transfer errors.

**Table 3. Provisioning Meta-Regression Model Results.**

|  |  |
| --- | --- |
|  | **Provisioning Model - Restricted log-linear** |
|  | **Coefficient (Standard Error)** |
| Constant | 4.398\*\* (1.720) |
| High Income | 2.342\*\* (0.839) |
| Peer Review | 3.225\*\*\* (0.950) |
| AgTFP | -0.028\* (0.015) |
| Pop Density | 0.0004\* (0.0002) |
| ESS Goal WP | 0.701 (0.890) |
| Longitude | 0.003 (0.070) |
| Amphibians | 0.080 (0.053) |
| Latitude | 0.008 (0.024) |
| Birds | 0.009 (0.005) |
| WL Area | -0.00000 (0.00000) |
| Model Summary Statistics  *N* | 27 |
| R2 | 0.732 |
| Adjusted R2 | 0.613 |
| Breusch Pagan Test Statistic | 5.26 |
| F Statistic | 6.114\*\* |
| AIC | 36.97 |
| Transfer Error Summary Statistics |  |
| RMSE (MRTE) | 1.94 |
| RMSE (MVTE) | 2.65 |
| MAD (MRTE) | 1.68 |
| MAD (MVTE) | 2.34 |

N denotes number of observations; \*\*\* denotes significance at 1%; \*\* denotes significance at 5%; \* denotes significance at 10%; RMSE denotes root mean square error; MAD denotes mean absolute deviation; MRTE denotes meta-regression transfer error; MVTE denotes mean value transfer error; values in parenthesis are standard errors.

*3.2.2. Regulating Meta-regression Model*

We choose the restricted log-log model because it produced the lowest meta-regression errors. Like the restricted provisioning model, we dropped variables that had high correlation coefficients (r > |0.6|) with other explanatory variables. For instance, we dropped use of incentives, and peer-review because they were highly correlated with ecosystem service goal (r = 0.69) and economic valuation method (r = -0.7), respectively. Several variables (no net loss wetland policy, use incentives wetland policy, use penalties wetland policy, peer-reviewed journal publications and bird richness) were dropped from the final model because they were not significant across all the estimated models.

Overall, the model was significant (F statistic = 9.23, p-value = 0.0002) and explained about 78% of the variation in the regulating wetland value. The model was homoscedastic, which means the variance of the error term was constant (Breusch Pagan statistic = 9.07, p-value = 0.43). Variance inflation factors for the explanatory variables were all < 10, indicating a lack of multicollinearity.

The model results showed that a 1% increase in wetland area caused about 0.31% decrease in the value of regulating wetland ecosystem services (p = 0.012). A 1% increase in agricultural factor productivity caused about 7.3% increase in regulating wetland values (p = 0.03). The regulating values of wetlands located in high-income economies were approximately 3.6% higher than similar wetlands located in jurisdictions with lower income (p-value = 0.04). Latitude coordinate had a positive effect with a magnitude of 0.054 (p-value = 0.06). All other variables (population density, economic valuation method, longitude, amphibians, ecosystem service goal) were not significant even at the 10% level. The meta-regression benefit transfer errors are about 3.00 and 1.85 lower (for root mean square and mean absolute error statistics, respectively) than the mean value transfer errors.

**Table 4. Regulating Meta-Regression Model Results.**

|  |  |
| --- | --- |
|  | **Regulating Model - Restricted log-log** |
|  | **Coefficient (Standard Error)** |
| Constant | -31.793\*\* (12.254) |
| Latitude | 0.053\*\* (0.026) |
| Log AgTFP | 7.266\*\* (2.892) |
| High Income | 3.611\*\* (1.559) |
| Log WL Area | -0.418\*\* (0.143) |
| Log Pop Density | 0.204 (0.210) |
| Econ. Val. Mthd. | 0.515 (1.141) |
| Longitude | -0.005 (0.007) |
| Log Amphibians | 1.187 (0.990) |
| ESS Goal WP | -1.913 (1.282) |
| Model Summary Statistics  N | 23 |
| R2 | 0.865 |
| Adjusted R2 | 0.772 |
| Breusch Pagan Test Statistic | 9.07 |
| F Statistic | 9.267\*\*\* |
| AIC | 36.80 |
| Transfer Error Summary Statistics |  |
| RMSE (MRTE) | 2.28 |
| RMSE (MVTE) | 5.27 |
| MAD (MRTE) | 2.00 |
| MAD (MVTE) | 3.85 |

N denotes number of observations; \*\*\* denotes significance at 1%; \*\* denotes significance at 5%; \* denotes significance at 10%; RMSE denotes root mean square error; MAD denotes mean absolute deviation; MRTE denotes meta-regression transfer error; MVTE denotes mean value transfer error; values in parenthesis are standard errors.

**4.0. Discussion and Policy Implications**

Wetlands have tangible values because they have been shown to produce services that are useful or beneficial to humans (Mitsch and Gosselink, 2000). Therefore, the positive effect of human population density on wetland values (both provisioning and regulating ecosystem services) is expected which is consistent with some previous work (see for instance, Mitsch and Gosselink 2000; Brander et al. 2006; Branders et al. 2013). A possible explanation for this observation is that higher populations near wetland areas would mean a greater number of people could benefit from wetland services. However, in a scenario where the population density in an area is significantly higher, the functional ability of wetlands to produce ecosystem services could be greatly impaired thereby significantly decreasing wetland values (Mitsch and Gosselink 2000).

Moreover, we found that wetlands in high-income countries have higher provisioning and regulating ecosystem service values compared to those in other income groups. Most citizens in wealthy nations live above their subsistence levels (can satisfy their basic needs) and thus are more likely to and have a higher capacity and willingness to pay for the support or be involved in protecting ecosystems, including wetlands (Mitsch and Gosselink (2000).

Agricultural total factor productivity (AgTEF) has been shown to have a positive impact on regulating wetland values and a negative effect on provisioning wetland values. A positive change in AgTFP implies a more efficient agricultural production system where relatively less inputs (including agricultural land) are required to produce equivalent agricultural outputs than in the pre-existing AgTFP state (International Food Policy Research Institute, 2018). Therefore, as AgTFP increases, there is less likely to be pressure on agricultural land expansion (including wetland conversion) to produce agricultural commodities; in this case wetland functions would have more time to evolve to produce ecosystem services to benefit society. However, the negative effect of AgTFP on provisioning wetland values is contrary to expectation. A plausible explanation could be that relatively less countries (37%) in the provisioning model are in the high-income status compared to 70% for the regulating model; people in developing nations are relatively poorer so might see the need to convert wetlands to croplands (to satisfy their subsistence needs) even in the face of increasing agricultural total factor productivity.

Our study has shown that wetland size is negatively related to its regulating and provisioning ecosystem service values (even though the effect on provisioning service value is not significant at the 10% level). The negative relation between wetland area and regulating service value was also reported by Brander et al. (2013). A possible explanation is that people may be willing to pay for a representative wetland in a given landscape, but they do not express proportionally larger values for larger areas of wetland, that is a small subset of an environmental feature but not for a large area (Loomis et al. 1993). However, a potential positive relationship may exist between wetland area and ecosystem values because ecosystem services may require a minimum threshold of wetland area (Brander et al. 2013; Reynaud and Lanzanova, 2017).

Further, our study showed that meta-studies that are published in peer reviewed journals have a positive effect on provisioning wetland values, suggesting a potential publication bias (i.e., studies with significant results on wetland values are more likely to be published than those with less encouraging results). An implication of publication bias is that caution is needed when generalizing results to all provisioning wetland values (Sutton et al. 2000). This observation has also been reported in other studies (Ghermandi and Nunes, 2013; Reynaud and Lanzanova, 2017). Our study shows that the presence of a national wetland policy would have a positive impact on provisioning wetland values, but a negative impact on regulating wetland values (even though the variable is not significant even at the 10% level in both cases).

Our meta-regression value functions generate lower prediction errors than do traditional mean value transfer methods. In a review of 38 meta-regression valuation studies,  Rosenberger ([2015](https://link.springer.com/article/10.1007/s10640-021-00536-2#ref-CR56)) reports that the average absolute percentage error (APE) for meta-regression and mean value transfers are 65% and 140%, respectively. Also, in a meta-analysis study to estimate the effect of waste sites on residential property values, Schutt (2021) reports a mean APE meta-regression error ranging from 133% and 684%. Our estimated mean meta-regression APE and mean value APE of 200% and 385%%, respectively, for the provisioning meta-regression model, and 168% and 234%, respectively, for the regulating model are consistent with Schutt (2021). Even though, our estimated benefit transfer errors are considerably higher compared to the average errors in the literature (Rosenberger, 2015). A plausible reason is the lack of sufficient data (23 observations for the regulating model and 27 for the provisioning model) to allow us to efficiently estimate a global meta-regression value function to value wetlands on agricultural landscapes compared. However, our general observation that meta-regression transfer errors are significantly lower than mean transfer errors is consistent with the literature on benefit transfer errors. In the absence of original studies to value wetlands, our models could be used as an effective benefit transfer tool (compared to the mean value transfer approach) to better estimate the value of wetlands and aid in land-use planning and wetland conservation policy development.

**5. Conclusion**

Our study advances previous work (Brander et al. 2013) by estimating a wetland regulating ecosystem service meta-regression model that extends the regulating services of wetlands beyond water quality, water supply, and nutrient recycling, as well as by deriving a provisioning meta-regression model. We find that the prediction errors from our models, compared to those from unit value transfers, were lower than similar estimates reported in the literature. This suggests our study results could be used to support the development of reliable wetland values using a benefit transfer approach than a mean value transfer method, especially in the absence of original valuation studies. For instance, the results from our models can be used to help tailor existing wetland ecosystem service valuation results to better represent the specific characteristics of the study area. The ability to reliably estimate wetland values on agricultural landscapes could guide the implementation or modification of policies to conserve wetlands on agricultural landscapes globally.

We identified important variables which could help in the effective implementation of a benefit transfer procedure to value wetlands. The important variables are agricultural total factor productivity, the income level of the country and the wetland area under study for the regulating model; income level of the country, peer-reviewed journal articles, agricultural total factor productivity, latitude, and population density for the provisioning model. Also, the analysis highlights those variables that may be less important in determining ecosystem service values including population density, economic valuation method, no net loss wetland policy, amphibian species richness, and longitude for the regulating model, and no net loss wetland policy, latitude, longitude, amphibian and bird species richness, and wetland area for the provisioning model; there were not enough information in the data to effectively capture those effects.

The insights provided by the models developed in this research will help inform wetland ecosystem service valuation exercises and therefore assist in the development of appropriate wetland conservation policy, the studies used where overwhelmingly based on study areas located in developed countries. While this is useful in developed country agricultural landscapes where there are significant pressures to convert wetlands to the production of agricultural commodities, the literature provides less information to develop appropriate models enabling benefit transfer in developing country context. Future studies are encouraged to conduct more wetland ecosystem service valuations in developing countries.

**Acknowledgements:**

We thank Mark Balman from BirdLife International and IUCN for access to data. Funding for this project was provided by the Global Institute for Water Security, Environment and Climate Change Canada and a Prairie Water Grant.

**Declarations of interest:** none

**References**

Badiou, P., McDougal, R., Pennock, D. and Clark, B., 2011. Greenhouse gas emissions and

carbon sequestration potential in restored wetlands of the Canadian prairie pothole region. Wetlands Ecology and Management, 19(3), pp.237-256.

Barbier, E.B., 1993. Sustainable use of wetlands valuing tropical wetland benefits: economic

methodologies and applications. Geographical Journal, pp.22-32.doi:10.2307/3451486

Brander, L., Brouwer, R., & Wagtendonk, A. 2013. Economic valuation of regulation services

provided by wetlands in agricultural landscapes: A meta-analysis. Ecological Engineering, 56, 89-96.

Brander, L.M., Florax, R.J. and Vermaat, J.E., 2006. The empirics of wetland valuation: a

comprehensive summary and a meta-analysis of the literature. Environmental and Resource Economics, 33(2), pp.223-250.

Brander, L.M., Van Beukering, P. and Cesar, H.S., 2007. The recreational value of coral reefs: a

meta-analysis. Ecological Economics, 63(1), pp.209-218.

Brouwer, R., Langford, I. H., Bateman, I. J., & Turner, R. K. (1999). A meta-analysis of wetland

contingent valuation studies. Regional Environmental Change, 1(1), 47-57.

Brundtland, G.H., 1987. Our Common Future: Report of the World Commission on Environment

and Development. United Nations Commission (Vol. 4).

Canu, D.M., Ghermandi, A., Nunes, P.A., Lazzari, P., Cossarini, G. and Solidoro, C., 2015.

Estimating the value of carbon sequestration ecosystem services in the Mediterranean Sea: An ecological economics approach. Global Environmental Change, 32, pp.87-95. doi:10.1016/j.gloenvcha.2 015.02.008.

Center for International Earth Science Information Network - CIESIN - Columbia University,

2017. Gridded Population of the World, Version 4 (GPWv4): Population Density, Revision 10. Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). <https://doi.org/10.7927/H4DZ068D>. Accessed November 14th 2018.

Chaikumbung, M., Doucouliagos, H. and Scarborough, H., 2019. Institutions, Culture, and

Wetland Values. Ecological Economics, 157, pp.195-204.

Cortus, B.G., Jeffrey, S.R., Unterschultz, J.R. and Boxall, P.C., 2011. The economics of wetland

drainage and retention in Saskatchewan. Canadian Journal of Agricultural Economics/Revue Canadienne d'agroeconomie, 59(1), pp.109-126.

Colloff, M.J., Lavorel, S., Wise, R.M., Dunlop, M., Overton, I.C. and Williams, K.J., 2016.

Adaptation services of floodplains and wetlands under transformational climate change. Ecological Applications, 26(4), pp.1003-1017. doi:10.1890/15-0848

Dadaser-Celik, F., Coggins, J.S., Brezonik, P.L. and Stefan, H.G., 2009. The projected costs and

benefits of water diversion from and to the Sultan Marshes (Turkey). Ecological Economics, 68(5), pp.1496-1506. doi:10.1016/j.ecolecon.2008.10.012

Davies, B., Biggs, J., Williams, P., Whitfield, M., Nicolet, P., Sear, D., Bray, S. and Maund, S.,

2008. Comparative biodiversity of aquatic habitats in the European agricultural landscape. Agriculture, Ecosystems & Environment, 125(1-4), pp.1-8.

Davidson, N., 2014. How much wetland has the world lost? Long-term and recent trends in

global wetland area. Marine and Freshwater Research. 65. Pp. 936-941.

DeGregorio, B.A., Willson, J.D., Dorcas, M.E. and Gibbons, J.W., 2014. Commercial value of

Amphibians produced from an isolated wetland. The American Midland Naturalist, 172(1), pp.200-205.

De Groot, R., Brander, L., Van Der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Christie, M.,   
 Crossman, N., Ghermandi, A., Hein, L. and Hussain, S., 2012. Global estimates of the

value of ecosystems and their services in monetary units. Ecosystem Services, 1(1), pp.50-61. doi:10.1016/j.ecoser.2012.07.005.

De Laporte, A., 2014. Effects of crop prices, nuisance costs, and wetland regulation on

Saskatchewan NAWMP implementation goals. Canadian Journal of Agricultural Economics/Revue Canadienne d'agroeconomie, 62(1), pp.47-67.

Dias, V. and Belcher, K., 2015. Value and provision of ecosystem services from prairie

wetlands: A choice experiment approach. Ecosystem Services, 15, pp.35-44.

Dixon, A.B. and Wood, A.P., 2003, May. Wetland cultivation and hydrological management in

eastern Africa: Matching community and hydrological needs through sustainable wetland use. In Natural resources forum (Vol. 27, No. 2, pp. 117-129). Oxford, UK: Blackwell Publishing Ltd.

Duffy, W.G. and Kahara, S.N., 2011. Wetland ecosystem services in California's Central Valley

and implications for the Wetland Reserve Program. Ecological Applications, 21(sp1), pp.S128-S134.

Economic Research Service, United States Department of Agriculture (2019). International

agricultural productivity. https://www.ers.usda.gov/data-products/international-agricultural-productivity/. Accessed on 13 April 2020.

Gardner, R.C., Barchiesi, S., Beltrame, C., Finlayson, C., Galewski, T., Harrison, I., Paganini,

M., Perennou, C., Pritchard, D., Rosenqvist, A. and Walpole, M., 2015. State of the world's wetlands and their services to people: a compilation of recent analyses.

Ghermandi, A., & Nunes, P. A. (2013). A global map of coastal recreation values: Results from a

spatially explicit meta-analysis. Ecological economics, 86, 1-15.

Ghermandi, A., Van Den Bergh, J.C., Brander, L.M., de Groot, H.L. and Nunes, P.A., 2010.

Values of natural and human‐made wetlands: A meta‐analysis. Water Resources Research, 46(12).

Gleason, R.A., Euliss Jr, N.H., Tangen, B.A., Laubhan, M.K. and Browne, B.A., 2011. USDA

conservation program and practice effects on wetland ecosystem services in the Prairie Pothole Region. Ecological Applications, 21(sp1), pp. S65-S81.

Golterman, H.L., 1995. The labyrinth of nutrient cycles and buffers in wetlands: results based on

research in the Camargue (southern France). Hydrobiologia, 315(1), pp.39-58.

Grygoruk, M., Mirosław-Świątek, D., Chrzanowska, W. and Ignar, S., 2013. How much for

water? Economic assessment and mapping of floodplain water storage as a catchment-scale ecosystem service of wetlands. Water, 5(4), pp.1760-1779.

Hao, F., Lai, X., Ouyang, W., Xu, Y., Wei, X. and Song, K., 2012. Effects of land use changes

on the ecosystem service values of a reclamation farm in Northeast China. Environmental Management, 50(5), pp.888-899.

International Food Policy Research Institute (IFPRI), 2018. Agricultural Total Factor

Productivity (TFP), 1991-2014: 2018 Global Food Policy Report Annex Table 5, <https://doi.org/10.7910/DVN/IDOCML>, Harvard Dataverse, V1.

International Union for Conservation of Nature - IUCN, and Center for International Earth

Science Information Network - CIESIN - Columbia University, 2015a. Gridded species distribution: global amphibian richness grids, 2015 Release. Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). <https://doi.org/10.7927/H4RR1W66>. Accessed November 28th, 2018

International Union for Conservation of Nature - IUCN, and Center for International Earth

Science Information Network - CIESIN - Columbia University, 2015b. Gridded species distribution: global mammal richness grids, 2015 Release. Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). [https://doi.org/10.7927/H4N014G5. Accessed November 28th 2018](https://doi.org/10.7927/H4N014G5.%20Accessed%20November%2028th%202018).

Leemans, R., & De Groot, R. S. 2003. Millennium Ecosystem Assessment: Ecosystems and

human well-being: a framework for assessment. A report of the Conceptual Framework Working Group of the millennium Ecosystem Assessment. Island Press.

Loomis, J., Lockwood, M., & DeLacy, T.,1993. Some empirical evidence on embedding effects

in contingent valuation of forest protection. Journal of Environmental Economics and Management, 25(1), 45-55.

Mitsch, W.J. and Gosselink, J.G., 2000. The value of wetlands: importance of scale and

landscape setting. Ecological Economics, 35(1), pp.25-33.

Morris, J. and Camino, M., 2011. UK national ecosystem assessment. Cambridge, UK.

Morreale, S.J. and Sullivan, K.L., 2010. Community-level enhancements of biodiversity and

ecosystem services. Frontiers of Earth Science in China, 4(1), pp.14-21.

Nelson, J.P. and Kennedy, P.E., 2009. The use (and abuse) of meta-analysis in environmental

and natural resource economics: an assessment. Environmental and Resource Economics, 42(3), pp.345-377.

Nunes, P.A. and van den Bergh, J.C., 2001. Economic valuation of biodiversity: sense or

nonsense?. Ecological Economics, 39(2), pp.203-222. doi:10.1016/S0921-8009(01)00233-6.

Oliver, T.H., Heard, M.S., Isaac, N.J., Roy, D.B., Procter, D., Eigenbrod, F., Freckleton, R.,

Hector, A., Orme, C.D.L., Petchey, O.L. and Proença, V., 2015. Biodiversity and resilience of ecosystem functions. Trends in ecology & evolution, 30(11), pp.673-684.

Peimer, A.W., Krzywicka, A.E., Cohen, D.B., Van den Bosch, K., Buxton, V.L., Stevenson,

N.A. and Matthews, J.W., 2017. National-level wetland policy specificity and goals vary according to political and economic indicators. Environmental Management, 59(1), pp.141-153.

Reynaud, A., & Lanzanova, D. (2017). A global meta-analysis of the value of ecosystem services

provided by lakes. Ecological Economics, 137, 184-194.

Richardson, L., Loomis, J., Kroeger, T. and Casey, F., 2015. The role of benefit transfer in

ecosystem service valuation. Ecological Economics, 115, pp.51-58.

Rosenberger R.S. 2015. Benefit transfer validity and reliability. In: Johnston RJ, Rolfe J,

Rosenberger RS, Brouwer R (eds) Benefit transfer of environmental and resource values: a guide for researchers and practitioners. Springer, Netherlands, Dordrecht, pp 307–326.

Rouquette, J.R., Posthumus, H., Morris, J., Hess, T.M., Dawson, Q.L. and Gowing, D.J.G., 2011.

Synergies and trade-offs in the management of lowland rural floodplains: an ecosystem services approach. Hydrological Sciences Journal, 56(8), pp.1566-1581. doi:10.1080/02626667.2011.629785

Schütt, M., 2021. Systematic variation in waste site effects on residential property values: a

meta-regression analysis and benefit transfer. Environmental and Resource Economics, 78(3), pp.381-416.

Serajuddin, U. & Hamadeh, N. 2021. New World Bank country classifications by income level:

2020-2021. https://blogs.worldbank.org/opendata/new-world-bank-country-classifications-income-level-2020-2021. Accessed on August 30, 2020.

Sutton, A.J., Song, F., Gilbody, S.M. and Abrams, K.R., 2000. Modelling publication bias in

meta-analysis: a review. Statistical methods in medical research, 9(5), pp.421-445.

Turner, A.C., Young, M.A., Moran, M.D. and McClung, M.R., 2021. Comprehensive valuation

of the ecosystem services of the Arctic National Wildlife Refuge. Natural Areas Journal, 41(2), pp.125-137.

Vedogbeton, H. and Johnston, R.J., 2020. Commodity consistent meta-analysis of wetland

values: An illustration for coastal marsh habitat. Environmental and Resource Economics, 75(4).

Vymazal, J., 2017. The use of constructed wetlands for nitrogen removal from agricultural

drainage: A review. Scientia Agriculturae Bohemica, 48(2), pp.82-91.

Watmough, M. and Schmoll, M.J., 2007. Environment Canada's Prairie & Northern Region

Habitat Monitoring Program phase II: recent habitat trends in the Prairie Habitat Joint Venture. Canadian Wildlife Service, Prairie and Northern Region.

Watson, K.B., Ricketts, T., Galford, G., Polasky, S. and O'Niel-Dunne, J., 2016. Quantifying

flood mitigation services: The economic value of Otter Creek wetlands and floodplains to Middlebury, VT. Ecological Economics, 130, pp.16-24.

Woodward, R. T., & Wui, Y. S. (2001). The economic value of wetland services: a meta-

analysis. Ecological economics, 37(2), 257-270.

Xiao-Hui, L., Xian-Guo, L., Ming, J. and Xi-Gang, W., 2011. Loss–gain estimation of

marshland carbon sequestration and paddy productivity in Fuyuan County, Heilongjiang Province, China. Outlook on Agriculture, 40(2), pp.165-170. doi:10.5367/oa.2011.0044

**Appendix 1**

**Table A1. The list of Studies used in this Meta-analysis**

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Paper** | **Study Year** | **Country** | **Provisioning**  **(2018US$/ha/year)** | **Regulating**  **(2018US$/ha/year)** |
| Acharya | 2000 | Nigeria | 64.70 | NA |
| Barbier | 1993 | Nigeria | 322.10 | NA |
| Beas and Smith | 2014 | United States | NA | NA |
| Bo et al. | 2013 | China | NA | NA |
| Bortolotti | 2016 | Canada | NA | 11.03 |
| Colloff et al. | 2016 | Australia | 911.87 | 0.80 |
| Dadaser-Celik et al. | 2009 | Turkey | 20.17 | NA |
| Degregorio et al. | 2014 | United States | NA | NA |
| Duffy and Kahara | 2011 | United States | 12,341.87 | 263.11 |
| Gleason et al. | 2011 | United States | 267.95 | 51.36 |
| Golterman | 1995 | France | NA | 5,272.56 |
| Grygoruk et al. | 2013 | Poland | 741.83 | NA |
| Hao et al. | 2012 | China | 5,113.50 | 336.67 |
| Jansson et al. | 1999 | Baltic Sea nations | NA | 0.10 |
| Jones | 2011 | United States | 22.61 | NA |
| Kakuru et al. | 2013 | Uganda | 32.12 | NA |
| Karpuzcu and Stringfellow | 2012 | United States | NA | 4,234.84 |
| Kipkemboi et al. | 2007 | Kenya | 2,561.31 | NA |
| L. Emerton (ed) | 2005 | Cambodia | 1.09 | NA |
| L. Emerton (ed) | 2005 | Cambodia | 1.09 | NA |
| L. Emerton (ed) | 2005 | Sri Lanka | 8.14 | NA |
| L. Emerton (ed) | 2013 | Uganda | 30.99 | NA |
| L. Emerton (ed) | 2004 | Brazil | 1.00 | 1.00 |
| Lant et al. | 2005 | United States | NA | NA |
| Leschine et al. | 1997 | United States | NA | 29,149.92 |
| Leschine et al. | 1997 | United States | NA | 104,966.00 |
| McCartney et al. | 2011 | South Africa | NA | NA |
| Meyerhoff | 2004 | Germany | NA | 425.00 |
| Meyerhoff | 2004 | Germany | NA | 7,735.00 |
| Prato and Hey | 2006 | United States | NA | NA |

NA denotes not applicable, which means that the study did value the specified ecosystem service

**Table A1 continued. The list of Studies used in this Meta-analysis**

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Paper** | **Study Year** | **Nation** | **Provisioning**  **(2018US$/ha/year)** | **Regulating**  **(2018US$/ha/year)** |
| Ramchandra et al. | 2005 | India | 16.03 | NA |
| Ribaudo et al. | 2001 | United States | NA | 974.21 |
| Roley et al. | 2016 | United States | NA | 2,237.50 |
| Rouquette et al. | 2011 | United Kingdoms | 5,801.00 | 6,864.88 |
| Rouquette et al. | 2011 | United Kingdoms | 5,801.00 | 19,123.59 |
| Schuijt | 2002 | Nigeria | 1.00 | NA |
| Schuijt | 2013 | Uganda | 48.78 | 0.37 |
| Schuijt | 2002 | Nigeria | 81.41 | 6.52 |
| Schuijt | 2002 | Malawi | 88.70 | NA |
| Schuijt and Brander | 2004 | United States | 8,872.00 | 18,703.00 |
| Schuijt and Brander | 2004 | New Zealand | 59.24 |  |
| Simonit et al. | 2013 | Kenya | NA | 0.17 |
| Smith et al. | 2011 | United States | 1,156.00 | NA |
| Verhoeven and Setter | 2010 | South East Asia | NA | 0.26 |
| Wang et al. | 2015 | China | 61.93 | 1.54 |

NA denotes not applicable, which means that the study did value the specified ecosystem service

1. Deep and light blue triangles show study sites for provisioning and regulating ecosystem services, respectively. [↑](#footnote-ref-1)
2. More information on the variable in Table 2 is provided in Table 1. [↑](#footnote-ref-2)