**Abstract**

Globally, the extent of inland wetlands has declined by approximately 70% and coastal wetlands by over 60%, since the start of the 20th century, which has resulted 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 ecosystem services of wetlands within agricultural environments around the world and identified 45 studies across 22 countries that provided sufficient economic information to be included in a quantitative meta-analysis. We estimated provision and regulation meta-regression models and found that (direction of effects in parenthesis) per capita income (proxied by a high-income binary variable, +), peer-reviewed journal publications (+), agricultural total factor productivity (-) and population density (+) have significant effects on provision wetland ecosystem values (e.g., food, building materials, firewood, etc.), while agricultural total factor productivity (-), income level (+) and wetland area (-) have significant effects on regulation wetland ecosystem values (e.g., carbon sequestration, nutrient recycling, flood control, etc.). Our estimated models which have lower meta-regression transfer errors compared to mean value transfer error, could help estimate reliable wetland values across similar nations and thereby help to inform wetland conservation and policies on agricultural landscapes.

**Keywords:** Agricultural landscapes, benefit transfer, provision ecosystem services, regulation 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, regulation 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 across the globe. 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 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 ecosystems services that could be used in cost-benefit analysis, tradeoff analysis, 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 (Brander et al. 2016).

However, 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 on ecosystem service values for policy decision-making in comparable jurisdictions. 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 meta-analysis, which uses rigorous quantitative methods to analyze multiple empirical studies, is often considered to produce the most reliable benefit transfer values of the three (Richardson et al. 2015). 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 and so the values of wetlands in agricultural landscapes are often overlooked or misrepresented. Moreover, since wetlands are increasingly being converted to annual crop production in agrosystems (Watmough and Schmoll 2007; Peimer et al. 2017), we urgently need more comprehensive valuations on socioeconomic alternatives to protect and restore wetlands in agricultural regions and in other high valued resource areas (e.g., Turner et al. 2021).

The incentive to drain wetlands for agricultural production, mainly in developed countries, has been driven by factors such as the increased cost of field obstructions with the increasing size of agricultural equipment, and the decreased 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 (causing drier conditions and increasing the need for irrigation needs) 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 we argue that the overall estimated value of natural wetlands in agricultural areas are currently under calculated and therefore misperceived 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 regulation ecosystem services: flood control, water supply, and nutrient recycling. They estimated the values (at the means) for flood control, water supply and nutrient recycling to be 6923 US$/ha/year, 3389 US$/ha/year, and 5788 US$/ha/year, respectively.

The main objectives of this study are, therefore, to estimate wetland meta-regression functions for factors that drive the value of wetland regulation services and wetland provision 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 other wetland regulation services that they did not as well as provision services to create a more comprehensive analysis of all wetland values, especially those in developing countries. 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 problem (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 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 across ‘all years’ 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. 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 regulation and provision ecosystem services, following Morris and Camino (2011). We converted all wetland values (US$/ha/year) to US dollars using the respective country’s exchange rate to the US$ at the time the study was conducted. Finally, 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 maxima and would need to be offset by variable production of greenhouse gases. For instance, converting wetlands to cropland may still 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 rather than forests. Again, we also did not report emissions in each study location because we were solely extracting the sequestration data calculated by the original study.

Wetland water storage was estimated by the storage capacity of 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.

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

**Figure 1a. Distribution of Study Sites for Provision Wetland Ecosystem Services**

**Figure 1b. Distribution of Study Sites for Regulation Wetland Ecosystem Services**

**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 regress 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 shows 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 chose a mixed model to explain the variation in wetland values. A general specification of a mixed effect model 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 square with fixed parameters is estimated if the mixed model is rejected for this study. Two separate provision and regulation 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 model and fixed effect model, respectively. The dependent variable for the provision model was the logarithm of the total value of provision ecosystem services, while the dependent variable for the regulation model was the logarithm of the total value of regulation ecosystem services. The sample sizes for the provision and regulation models were 27 and 22, respectively. We tested for heteroscedasticity (using Breusch Pagan test) and multicollinearity (using the variance inflation factor) in our estimated models. 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 undermines their statistical significance; however, it does not affect the reliability of parameter estimates. An independent variable with a variable inflation factor less than 10 would mean it is not a source of multicollinearity.

Moreover, the best or final functional form was chosen if it produced 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 to ensure that they are robust and reliable. 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 regulation and provision 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 (CIESIN, 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 provision and regulation 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 measure 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 (provision and regulation).

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 (IUCN & CIESIN 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 effects on wetland values (provision and regulation).

National Wetland Policy

           No net loss wetland policy 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 incentives and use 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). It must be noted that 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

There could be study-specific nuances or characteristics that drive the heterogeneity in wetland values (both regulation and provision ecosystem services). Study-specific variables are wetland area, year of publication, peer-review journal publication, 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 this variable to have a negative or positive effect on wetland values. Loomis et al. (1993) have shown that 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. 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** |  |  |  |  |
| Provision Model: | Provision wetland values | Continuous | 2018US$/Ha/Year |  |
| Regulation Model: | Regulation wetland values | Continuous | 2018US$/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 | +/- |

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

**3. Summary Trends**

The mean statistic for wetland provision ecosystem service value was US$1,645/ha/year (in 2018 US$) with a standard deviation of USD$3,168/ha/year, indicating high variation in ecosystem value across studies (Table 2). Also, the mean regulation ecosystem service value was US$8,711/ha/year with a standard deviation of US$22,375/ha/year. The mean (standard deviation) of the wetland area variable for the provision and regulation meta-regression models, were 870,000 ha (2,800,000 ha) and 2,730,000 ha (6,600,000 ha), respectively. This shows that wetlands in the regulation meta-regression model were relatively bigger and more heterogenous in size than in the provision model. About 70% of the wetlands in the provision model were valued using an economic valuation method compared to 52% for the regulation model. However, more studies (65%) in the regulation model were published in peer-reviewed journals compared to studies in the provision model (52%).

In terms of the economic variables, the mean agricultural factor productivity variable in both models was the same at 114; however, the heterogeneity in the values was greater in the regulation model. Considerably more studies (70%) were conducted in high-income countries in the regulation model than in the provision model (37%). Conversely, about 22% of the studies in the provision model were conducted in low-income countries versus 9% in the regulation model. Also, the mean population density was greater for wetlands in the regulation model (1,003 humans/km2) and more heterogenous (2,467 human population/km2) than for wetlands in the provision 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 countries in the regulation model had an ecosystem service goal (15% more), use an incentive policy to conserve wetlands (11% more), use penalties to conserve wetlands (33% more) and have a no net loss wetland policy (15% more) than countries in the provision model. This shows that wetlands in the regulation model are expected to receive more protection than wetlands in the provision model.

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

**Table 2. Summary statistics for each variable used in the modelling.**

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
|  | Provision Model | | | Regulation Model | | |
| Model Variables | Mean (SD) | min | Max | Mean (SD) | min | max |
| **Dependent Variable** |  |  |  |  |  |  |
| Provision ESS | 1,644.79  (3,167.5) | 4.00E-04 | 12,341.87 |  |  |  |
| Regulation 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  (.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 |

More information on the variables is provided in Table 1; values in parentheses are standard deviation; min denotes minimum; max denotes maximum.

Table 2. Continued.

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
|  | Provision Model | | | Regulation Model | | |
| Model Variables | mean | min | Max | mean | min | max |

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| 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 |

More information on the variables are provided in Table 1; values in parentheses are standard errors; min denotes minimum; max denotes maximum.

*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 provision or regulation model structures, respectively. Therefore, the results of the fixed effect model, for both models, are reported in this section.

*3.2.1. Provision Meta-regression Model*

We chose restricted log-linear model because it produced the lowest meta-regression errors compared to the other models we tested (see Tables A4 and A5 in Appendix A). 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 economic valuation method because it was highly correlated with high-income (r = 0.68); also, we dropped no net wetland policy, use incentives wetland policy and use penalties wetland policy because they were consistently not significant across all the estimated models.

Overall, the model is significant at the 1% level (F statistic = 5.88, p-value = 0.0009) and explains 65.3% of the variation in the dependent variable (log of provision wetland ecosystem values). The model is homoscedastic, which means the variance of the error term is constant (Breusch Pagan statistic = 5.26, p-value = 0.87); multicollinearity is not a problem in the model since all the independent variables have variance inflation factors < 10.

Population density and high-income both have positive effects on the provision wetland values, which are significant at the 10% and 5% levels, respectively. The estimated coefficient of population density means that a 1% increase in the variable will have a $0.0004 increase in provision wetland value; similarly, a wetland located in a high-income country would have about $3.59 more provision value than those located in other income groups. Agricultural factor productivity has a negative effect on provision wetland values (significant at 10% level); specifically, a 1% increase in agricultural factor productivity would cause about $0.028 reduction in provision wetland values. The provision value of wetlands in peer-reviewed journal publications is about $3.45 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) are not significant. Moreover, the meta-regression benefit transfer errors for the provision 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. The provision model results are presented in Table 3.

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

|  |  |
| --- | --- |
|  | **Provision Model - Restricted log-linear** |
|  | **Coefficient (Standard Error)** |
| Constant | 4.398\*\* (1.720) |
| High Income | 3.588\*\* (1.020) |
| Peer Review | 3.458\*\*\* (0.950) |
| AgTFP | -0.028\* (0.015) |
| Pop Density | 0.0004\* (0.0002) |
| ESS Goal WP | 0.701 (0.890) |
| Longitude | 0.084 (0.007) |
| Amphibians | 0.031 (0.053) |
| Latitude | 0.025 (0.024) |
| Birds | 0.007(0.004) |
| WL Area | -0.00000 (0.00000) |
| Model Summary Statistics  N | 27 |
| R2 | 0.79 |
| Adjusted R2 | 0.653 |
| Breusch Pagan Test Statistic | 5.26 |
| F Statistic | 5.88\*\*\* |
| 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. Regulation Meta-regression Model*

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

Overall, the model is significant at the 1% level (F statistic = 9.23, p-value = 0.0002) and explains about 78% of the variation in the regulation wetland value. The model is homoscedastic, which means the variance of the error term is constant (Breusch Pagan statistic = 9.07, p-value = 0.43). Multicollinearity is not a problem in the model because the variance inflation factors for the independent variables are all < 10.

The model results showed that a 1% increase in wetland area would cause about 0.31% decrease in regulation wetland values (p = 0.012; Table 4). A 1% increase in agricultural factor productivity will cause about 7.3% increase in regulation wetland values (p = 0.03). The regulation value of wetlands located in high-income economies are about 3.6% higher than similar wetlands located in other income group countries (p-value = 0.04). Latitude coordinate has a positive effect size of 0.054 and significant at the 10% level (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. The regulation model results are presented in Table 4.

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

|  |  |
| --- | --- |
|  | **Regulation 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 value because they 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 provision and regulation) is expected. This is consistent with observations in the literature (see for instance, Mitsch and Gosselink 2000; Brander et al. 2006; Branders et al. 2013). A possible explanation for the above observation is that higher population rates 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, rendering wetland values to be approximately zero (Mitsch and Gosselink 2000). In the above scenario, a national wetland policy could help conserve wetlands and enhance their values. Our study shows that the presence of a wetland policy in a country would have a positive impact on provision wetland values, but a negative impact on regulation wetland values (even though the variable is not significant even at the 10% level in both cases).

Moreover, this study has shown that wetlands in high-income countries have higher provision and regulation values compared to those in other income groups. The above observation is expected, because citizens in wealthy nations are mostly live above their subsistence levels (can satisfy their basic needs) and thus are more likely to be involved in protecting ecosystems, including wetlands Mitsch and Gosselink (2000); also, they are also more likely to have greater capacity and a higher willingness to pay for wetland services.

Also, agricultural total factor productivity has been shown to have a positive impact on regulation wetland values and a negative effect on provision wetland values. A positive change in agricultural total factor productivity (AgTFP) would mean a more efficient agricultural production system where less inputs (including agricultural land) will be needed to produce agricultural outputs than the pre-existing state of AGTFP (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 provision wetland values is contrary to expectation.

Again, our study has shown that wetland size is negatively related to its regulation and provision ecosystem values (even though the effect on provision service value is not significant at the 10% level). The negative relation between wetland area and regulation service value was also found in Brander et al (2013). A possible explanation for the above relation is that people may be willing to pay for 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 “most ecosystem services require a minimum threshold of wetland area (Brander et al. 2013; Reynaud and Lanzanova, 2017).

Further, our study has also shown that meta-studies that are published in peer reviewed journals have a positive effect on provision 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 to this study will be that we cannot generalize our results to all provision wetland values (Sutton et al. 2000). This observation has also been reported in other studies (Ghermandi and Nunes, 2013; Reynaud and Lanzanova, 2017).

Further, we have estimated meta-regression value functions that produce less prediction errors than the traditional mean value transfer method. Specifically, we showed that (for root mean squared error) our provision meta-regression model produced 57% less prediction error than would have been the case with a mean value transfer, and 48% less in the case of mean absolute deviation error. For the regulation model, the mean meta-regression transfer error is about 28% less than mean unit value transfer error (for both root mean squared error and mean absolute deviation error). Rosenberger and Loomis (2017) report a 11% error difference between mean meta-regression transfer error (36%) and mean unit value transfer error (45%). This implies that 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 builds on the work of Brander et al. (2013) by estimating a regulation meta-regression model that extends the regulation services of wetlands beyond water quality, water supply, and nutrient recycling, as well as a provision meta-regression model. We showed that the prediction errors from our models compared to those from unit value transfers were lower than similar estimates reported in the literature. This means our study could be used to reliably value wetlands using a benefit transfer approach. The ability to reliably estimate wetland values on agricultural landscapes could significantly improve on the implementation 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 regulation model; income level of the country, peer- reviewed journal articles, agricultural total factor productivity, latitude, and population density for the provision model. Also, the less important variables are population density, economic valuation method, no net loss wetland policy, amphibian richness, and longitude for the regulation model, and no net loss wetland policy, latitude, longitude, amphibian and bird richness, and wetland area for the provision model.

However, there is a need for more detailed studies on wetland values in developing regions. A restricted western focus provides a narrow and likely biased interpretation of wetland values and wetland conservation at the global level. Similarly, the data for estimating the provision model come mostly from lower income nations (6 out of 27 studies (22%) for the provision model, and 2 out of 23 studies (9%) for the regulation model). More studies recording provision ecosystem services in developed nations could help present a comprehensive representation of this service.

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**Appendix 1**

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

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Paper** | **Study Year** | **Nation** | **Provision**  **(2018US$/ha/year)** | **Regulation**  **(2018US$/ha/year)** |
| Acharya | 2000 | Nigeria | 63.70 | 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 | 12341.87 | 263.11 |
| Gleason et al. | 2011 | United States | 266.95 | 51.36 |
| Golterman | 1995 | France | NA | 5272.56 |
| Grygoruk et al. | 2013 | Poland | 741.83 | NA |
| Hao et al. | 2012 | China | 5112.50 | 336.67 |
| Jansson et al. | 1999 | Baltic Sea nations | NA | 0.10 |
| Jones | 2011 | United States | 21.61 | NA |
| Kakuru et al. | 2013 | Uganda | 31.12 | NA |
| Karpuzcu and Stringfellow | 2012 | United States | NA | 4234.84 |
| Kipkemboi et al. | 2007 | Kenya | 2560.31 | NA |
| L. Emerton (ed) | 2005 | Cambodia | 0.09 | NA |
| L. Emerton (ed) | 2005 | Cambodia | 0.09 | NA |
| L. Emerton (ed) | 2005 | Sri Lanka | 8.14 | NA |
| L. Emerton (ed) | 2013 | Uganda | 29.99 | NA |
| L. Emerton (ed) | 2004 | Brazil | NA | NA |
| Lant et al. | 2005 | United States | NA | NA |
| Leschine et al. | 1997 | United States | NA | 29149.92 |
| Leschine et al. | 1997 | United States | NA | 104966.00 |
| McCartney et al. | 2011 | South Africa | NA | NA |
| Meyerhoff | 2004 | Germany | NA | 425.00 |
| Meyerhoff | 2004 | Germany | NA | 7735.00 |
| Prato and Hey | 2006 | United States | NA | NA |

NA denotes not applicable, which means that the study did not specify a type of ecosystem service

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Paper** | **Study Year** | **Nation** | **Provision**  **(2018US$/ha/year)** | **Regulation**  **(2018US$/ha/year)** |
| Ramchandra et al. | 2005 | India | 15.03 | NA |
| Ribaudo et al. | 2001 | United States | NA | 974.21 |
| Roley et al. | 2016 | United States | NA | 2237.50 |
| Rouquette et al. | 2011 | United Kingdoms | 5800.00 | 6864.88 |
| Rouquette et al. | 2011 | United Kingdoms | 5800.00 | 19123.59 |
| Schuijt | 2002 | Nigeria | NA | NA |
| Schuijt | 2013 | Uganda | 47.78 | 0.37 |
| Schuijt | 2002 | Nigeria | 81.41 | 6.52 |
| Schuijt | 2002 | Malawi | 87.70 | NA |
| Schuijt and Brander | 2004 | United States | 8871.00 | 18703.00 |
| Schuijt and Brander | 2004 | New Zealand | 58.24 |  |
| Simonit et al. | 2013 | Kenya | NA | 0.17 |
| Smith et al. | 2011 | United States | 1155.00 | NA |
| Verhoeven and Setter | 2010 | South East Asia | NA | 0.26 |
| Wang et al. | 2015 | China | 60.93 | 1.54 |