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 manage water quantity by storing water, regulating and recharging aquifers during wet seasons and thereby mitigate flooding in wet periods and support 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 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 can 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). Yet, a characteristic lacking from these studies is that they were not restricted to wetlands on agricultural landscapes.

Research has shown that agricultural production is an important driver for the loss and degradation of wetlands (Watmough and Schmoll 2007; Peimer et al. 2017). 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) are also motivating land managers to convert wetlands to agricultural lands (Dixon and Wood 2003). However, few studies have focused on estimating wetland values on agricultural landscapes. 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.

The main objectives of this study are to estimate wetland meta-regression value 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 value functions to guide the benefit transfer of wetland values in agricultural landscapes. Our study builds on the work of Brander et al. (2013) in two ways. First, we extend the three regulating ecosystem services studied in Brander et al. (2013) to include other wetland ecosystem regulating services (Morris & Camino, 2011)[[1]](#footnote-1). Second, we also conduct a meta-regression on provisioning wetland ecosystem service values (Morris & Camino, 2011)[[2]](#footnote-2). 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 the commodity inconsistency problem. 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 608 research articles published across ‘all years’ prior to May 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.

A second quantitative review was conducted using the same search keywords on Google Scholar to update the list of published studies to 2020 which resulted in 60 additional papers.

From these 608 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 acreage 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; described ecosystem services in terms of general trends without quantifying effect(s); did not provide specific examples, and/or failed to clearly describe overall methodology or effect (size). 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) acreage of wetlands under valuation or relevant information that could allow us to obtain wetland acreage information. (review and coding processes were conducted by EA and EA)

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 acreage, 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 in non-US dollar denominations to US dollars using the respective country’s exchange rate to the US$ at the time the study was conducted. All wetland values in US$/ha were converted to US$/ha/year by multiplying them by 365. 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. We are measuring possible benefits from carbon sequestration and acknowledge that these are maxima and would need to be offset by variable production of greenhouse gases. Converting wetlands to cropland may still be negative (i.e., produce even more GH gases) depending on the production system. Also, we did not include peatlands 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.

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 type was converted into monetary terms (2018 US$/ha/year). Provisioning wetland ecosystem service studies were conducted mainly in developing countries (i.e., Africa; see Figure 1a), while regulating service studies were conducted mainly in developed countries, particularly North America (Figure 1b). Table A1, in Appendix 1, lists the primary studies used in this study.

**Figure 1. Geographic Distribution of Study Sites for the (a) Regulating Wetland Ecosystem Services and (b) Provisioning Ecosystem Wetland 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 used a random intercept mixed effect model to 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 choose a mixed effect model to account for potential interclass correlation within clusters of multiple observations that were reported in some of the papers. However, if the null hypothesis that in equation 1 is rejected at the 10% significance level (using a likelihood ratio test), it will mean that the random intercept model is not appropriate for our study; in that case a fixed effect model will use a fixed effect ordinary least square model provided the key assumptions underlying that model are validated.

We used a log-log functional form, where we took the logarithms of the dependent variable and continuous explanatory variables to improve model fit and prevent heteroscedasticity (Brander et al. 2013). For 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). When the regressor is a binary variable, the effect is compared to its reference group. A general specification of the mixed effect model used to explain heterogeneity in wetland ecosystem service values (Yw) was given as:

where: is the dependent variable representing the logarithm of the value of wetland ecosystem service (US$/ha/year) for the ith study, is a vector of wetland policy variables, is a vector of human population and economic indicators for the ith study, is a vector of biodiversity richness indicators for the ith study, is a vector of study characteristics for the ith study, is the constant term parameter to be estimated, is a vector of parameters to be estimated that is associated with the policy variables, is a vector of parameters to be estimated that is associated with human population and economic indicators, is a vector of parameters to be estimated that is associated with biodiversity richness indicators, is a vector of parameters to be estimated that are associated with the study characteristics, stochastic error term for the ith study, which is assumed to be normally distributed with mean 0 and a constant variance ( which accounts for variation in wetland values due to differences between individual observations, stochastic error term for the ith study, which is assumed to be normally distributed with mean 0 and a variance ( which accounts for variation in wetland values due to differences between study observations, I is an N by N identity matrix, and N is the number of observations.

Two separate provisioning and regulating models with the same functional form as equation (1) were estimated using frequentist estimation procedure, with the “LM” and “LMER” R statistical software packages, for the random intercept model and fixed effect model, 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.

2.2.1. 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 1 km resolution of gridded human population layer of the world that modeled the distribution of human populations 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 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. For this study, we used binary variables high-income and low-income to group countries into income groups. The high-income variable was given a value 1 if the gross national income (GNI) in current 2019 USD was greater than $12,535 and 0 otherwise (Serajuddin and Hamadeh, 2021). Similarly, the low-income variable was given a value 1 if the gross national income (GNI) in current 2019 USD was less than $1,036.

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 could mean a more profitable agricultural industry, which could pose a threat to wetlands (van Asselen et al. 2013) since land allocated to wetlands will impose a higher opportunity cost on land managers. Therefore, agricultural productivity is predicted to have a negative effect on wetland ecosystem values (provisioning and regulating) through the reduced quality transmission mechanism.

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 the economic valuation estimates are generally regarded as being limited and providing an incomplete perspective (Nunes et al. 2001).

Threatened biodiversity can be used as proxy to measure a loss in ecosystem values, including wetlands (Egoh et al. 2009; Mace et al. 2012). Therefore, we used gridded species distribution data for amphibians and birds as a standardized biodiversity index for all study sites. 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 and threatened species richness at each site using bilinear interpolation with ArcGIS 10.5. These indexes are expected to have positive impacts on wetland values (provisioning and regulating).

National Wetland Policy

           No net loss wetland policy seeks to maintain the total acreage 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 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).

Study Characteristics

There could be study-specific nuances or characteristics that drive the heterogeneity in wetland values (both regulating and provisioning ecosystem services). Study-specific variables are wetland acreage, year of publication, peer-review, valuation method, and geographic location (latitude and longitude). These variables are routinely added to meta-analyses (Brander et al. 2013). Wetland acreage, 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 effect on wetland values. Following Moeltner et al. (2019), for a specific study (or observation), the study year was defined as the difference between the year study was conducted minus the oldest year in the other studies (observations) plus 1. When the coefficient of this variable is positive, it will mean that there has been a growth in wetland values over time. 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** | **Variable Type** | **Variable Unit** | **Expected Effect on Wetland Values** |
| **Dependent Variables** |  |  |  |
| Provisioning Model: Total Value of Wetland  Regulating Ecosystem Services | Continuous | 2018US$/Ha/Year |  |
| Regulating Model: Total Value of Wetland  Regulating Ecosystem Services | Continuous | 2018US$/Ha/Year |  |
| **Explanatory Variables** |  |  |  |
| *Economic Variables* |  |  |  |
| Human Population Density | Continuous | H/km2 | + |
| Agricultural Total Factor Productivity (AgTFP) | Continuous | 2015 AgTFP = 100 | - |
| High-Income | Binary | 1,0 | + |
| Low-Income | Binary | 1,0 | - |
| *Biodiversity Variables* |  |  |  |
| Birds Species Richness | Continuous | Counts/Ha | + |
| Amphibian Species Richness | Continuous | Counts/Ha | + |
| *National Wetland Policy* |  |  |  |
| No Net Loss wetland policy | Binary | 1,0 | + |
| National Ecosystem Service Policy | Binary | 1,0 | + |
| Use Penalties to Conserve Wetlands | Binary | 1,0 | + |
| *Study Characteristics* |  |  |  |
| Wetland Acreages | Continuous | Ha | - |
| Year of Publication | Continuous | 1,0 | +/- |
| Economic Valuation Method | Binary | 1,0 | +/- |

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

**3. Summary Trends**

*3.1. Descriptive Results*

The mean statistic for wetland provisioning ecosystem service value was US$1645/ha/year (in 2018 US$) with a standard deviation of USD$3168/ha/year, indicating high variation in ecosystem value across studies. Also, the mean regulation ecosystem service value was US$8711/ha/year with a standard deviation of US$22375/ha/year. The mean (standard deviation) of the wetland acreage variable for the provisioning and regulation meta-regression models, were 870000 ha (2800000 ha) and 2730000ha (6600000 ha), respectively. This shows that wetlands in the regulation meta-regression model were relatively bigger 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 regulation model. However, more studies (65%) in the regulation model were published in peer-reviewed journals compared to studies in the provisioning 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-come countries in the regulation model 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 regulation model. Also, the mean population density was greater for wetlands in the regulation model (1003 humans/km2) and more heterogenous (2467 human population/km2) than for wetlands in the provisioning model with a mean and standard deviation of 755 human population/km2 and 2223 human population/km2, respectively.

Moreover, concerning the wetland policy variables, more countries in the regulation model had an ecosystem service goal (15% more), use of an incentive policy to conserve wetlands (11% more), use of penalties to conserve wetlands (33% more) and no net loss wetland policy (15% more) than countries in provisioning model. This shows that wetlands in the regulation model are expected to receive more protection than wetlands in the provisioning model.

With the biodiversity variables, there were more amphibians associated with wetlands in the provisioning model, on the average, (16.2 count/ha) and more heterogeneity in the values of the variable (10.6 counts/ha) than in the regulation model (mean (standard deviation) count/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 regulation model (194 /ha). The summary statistic results are provided in Table 2 below.

**Table 2. Summary Statistic Results**

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
|  |  | Provisioning Model | | | Regulation Model | | |
| Model Variables | Variable Description | mean | min | max | mean | min | max |
| **Dependent Variable** |  |  |  |  |  |  |  |
| Provisioning ESS | Provisioning wetland values ((2018USD/ha/year) | 1644.79  (3167.5) | 4.00E-04 | 12341.87 |  |  |  |
| Regulation ESS (2018USD/ha/year) | Regulation wetland values ((2018USD/ha/year) |  |  |  | 8711.23  (22375) | 6.00E-04 | 1.04E05 |
| **Explanatory Variables** |  |  |  |  |  |  |  |
| *Economic Variables* |  |  |  |  |  |  |  |
| Pop Density | Population density (human population/km2) | 754.91  (2223) | 0 | 10164.5 | 1003.06  (2467) | 0 | 1.01E04 |
| AgTFP | Agricultural factor productivity | 114.59  (29.35) | 64 | 181 | 114.41  (918.35 | 64 | 148 |
| High Income | High-income country (binary) | 0.37  (0.49) | 0 | 1 | 0.70  (0.47) | 0 | 1 |
| Low Income | Low-income country (binary) | 0.22  (0.42) | 0 | 1 | 0.09  (0.29) | 0 | 1 |
| *Biodiversity Variables* |  |  |  |  |  |  |  |
| Amphibians | The number of amphibian species associated with wetland (count/ha) | 16.19  (10.60) | 2 | 44 | 12.39  (9.30) | 0 | 40 |
| Birds | The number of bird species associated with wetland (count/ha) | 283  (127.99) | 97 | 544 | 194  (.80.45.) | 8 | 408 |
| *Wetland Policies* |  |  |  |  |  |  |  |
| ESS Goal WP | Ecosystem service wetland policy (binary) | 0.78  (0.42) | 0 | 1 | 0.91  (0.28) | 0 | 1 |
| Use Incentives | Use incentives to conserve wetlands (binary) | 0.85  (0.36) | 0 | 1 | 0.96  (0.21) | 0 | 1 |

Table 2. Continued.

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
|  |  | Provisioning Model | | | Regulation Model | | |
| Model Variables | Variable Description | mean | min | max | mean | min | max |

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
| Use Penalties | Use penalties to conserve wetlands (binary) | 0.44  (0.51) | 0 | 1 | 0.70  (0.47) | 0 | 1 |
| No Net Loss WP | No Net Loss wetland policy (binary) | 0.63  (0.49) | 0 | 1 | 0.78  (0.42) | 0 | 1 |
| *Study Characteristics* |  |  |  |  |  |  |  |
| WL Acreage | Acreage of wetland under valuation (ha) | 8.7E05  (2.8E06) | 1.64 | 1.38e07 | 2.73E06  (6.6E06) | 0.7 | 2.7E07 |
| Econ. Val. Mthd. | Economic valuation method (binary) | 0.70  (0.47) | 0 | 1 | 0.52  (0.51) | 0 | 1 |
| Peer Review | Peer reviewed paper (binary) | 0.52  (0.51) | 0 | 1 | 0.65  (0.49) | 0 | 1 |
| Year |  |  | 1993 | 2000 |  | 1995 | 2016 |

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 random coefficient models, provisioning and regulation, did not fit the data well. The intra correlation coefficient for both models were 0.28 and 0.07, for the provisioning and regulation models, respectively. The above intra correlation coefficients show that the within correlation of observations with study clusters were considerably very low to warrant a mixed effect model. Likelihood ratio tests for both models, provisioning (likelihood ratio value = -55, p-value = 0.64) and regulation (likelihood ratio value = -44.6, p-value = 0.92), validated the observation of the intra correlation coefficient results that random coefficient mixed models were not appropriate for this study. Therefore, the results of the fixed effect model, for both models, are reported next, starting with the provisioning model.

Overall, the provisioning model is significant at the 1% level (F statistic of 3.94). Also, the model explained about 64.4% (adjusted R-square) of the variation in dependent variable (log of provisioning wetland ecosystem values). The model is homoscedastic, which means the variance of the error term is constant (Breusch Pagan statistic = 18.55, p-value = 0.29). The in-sample prediction error (where the values of the original dependent variable to compared to corresponding predicted variable) from the estimated provisioning meta-regression value function is 15.38% which is significantly lower the in-sample benefit transfer (values of the original dependent variable is compared to the mean) prediction error (40%). We reported the median prediction error because of outliers (4 values) which can bias the mean value upwards. The observation level prediction errors are provided in Table A2 in appendix.

Regarding the economic variables, log population density, and high-income are significant (at 10% and 1%, respectively) and have positive effects on log of provisioning wetland values. This means that a 1% increase in the level of population density will cause about 0.48% increase in the value of provisioning wetland values. Provisioning ecosystem values of wetlands in high-income countries are expected to be about 8.2% greater than wetlands in other countries, including medium-income countries. The direction of effects of these variables on provisioning wetland values are expected. Also, a 1% increase in the agricultural factor productivity is expected to cause about 3.9% decrease in provisioning wetland values; this effect is not significant at the 10% level. Concerning study variables, model results show that a 1% increase in the acreage of wetland is expected to increase the provisioning value of the wetland by about 0.54%, at a 5% significance level. Also, the provisioning wetland values in this study have increase about 1.2% since 1993 (the year of the oldest study), at 10% significance level. The provisioning value of wetlands in peer-reviewed studies is about 4.1% more than values in other studies, at 1% significance level. The provisioning value of wetlands in that were evaluated with economic valuation method is about 2.4% more than values evaluated with other methods; however, this effect is not significant at 10% level. With the wetland policy variables, provisioning wetland values in countries that have a no net loss wetland policy or an ecosystem service goal or use penalties to enforce wetland conservation policies are about 0.31%, 5.5%, 0.45%, respectively, more than wetlands in other countries. The effect of the ecosystem service goal is significant at 5%, but effects of the other wetland policy variables are not significant at the 10% level. However, the provisioning wetland value in countries that use incentives to enforce their wetland conservation policies is about 6.84% lower than countries that do not use incentives. Regarding the biodiversity variables, 1% increases in species richness/ha of amphibians and birds are expected to increase the provisioning wetland values by about 0.2% (significant at 10%) and 0.002% (not significant at 10%), respectively. The location variables, latitude and longitude are significant at 5% and 10% significant, respectively; this show that these variables maybe capturing the effect of location specific variables that were omitted from the model.

Similar to the provisioning model, the regulation model is, overall, statistically significant at the 1% level (F statistic of 7.83). Also, the model explained about 80% of the variation in the log of regulation wetland value (the dependent variable). The model is homoscedastic, which means the variance of the error term is constant (Breusch Pagan statistic = 6.40, p-value = 0.89). The in-sample median prediction error (where the values of the original dependent variable to compared to corresponding predicted variable) from the estimated regulation meta-regression value function is 15.81% which is significantly lower the in-sample benefit transfer (values of the original dependent variable is compared to the mean) prediction error (47.48%). We reported the median prediction error because of outliers (2 values) which can bias the mean value upwards. The observation level predictions errors are provided in Table A3 in appendix.

None of the economic variables are significant at the 10% level. However, the model results show that the regulation values of wetlands in high-income countries are about 1.845 more than the values of wetlands in other countries. Also, a 1% increase in the value of agricultural factor productivity and population density are expected to increase the regulation value of wetlands by about 3.52% and 0.34%, respectively,. With the study variables, a 1% increase in wetland acreage will decrease regulation wetland values by about 0.46%, which is significant at 5%. Also, the growth in the regulation values in wetlands have decreased by about 1.06% since 1995 (the study year of the oldest paper). The regulation wetland values that were estimated in peer-reviewed studies or evaluated with economic methods are about 0.20% and 0.34%, respectively, more than in other studies; however, the effects are not significant at the 10% level. The regulation values of wetlands in countries that have a no net loss wetland policy are about 2% less than in countries that do not have that policy; the effect is not significant at the 10% level. A unit increase in bird species/ha causes ~0.02% increase in the regulation value of wetlands. The coefficient of one of the location variables, latitude is significant at 10%.

**Table 3. Meta-Regression Model Results. Regression coefficients (SE)?**

|  |
| --- |
|  |
|  | Provisioning Model | | | | Regulation Model | | |
|  | OLS | RIM | | | OLS | RIM | |
|  | (1) | (2) | | | (3) | (4) | |
|  | | | |
| *Economic Variables* |  |  |  | | | |  | |
| Log Pop Density | 0.475\* (0.231) | 0.332 (0.256) | 0.336 (0.192) | | | | 0.398\*\* (0.188) | |
| Log AgTFP. | -3.861 (2.516) | -3.850 (3.156) | 3.523 (6.046) | | | | 6.525 (5.667) | |
| High Income | 8.171\*\*\* (2.120) | 5.891\*\*\* (2.175) | 1.843 (1.555) | | | |  | |
| Low Income | 0.791 (1.645) | 1.946 (1.767) | -4.101 (3.259) | | | | -4.311 (3.353) | |
| *Biodiversity Variables* |  |  |  | | | |  | |
| Log Amphibians | 0.161\* (0.073) | 0.096 (0.079) |  | | | |  | |
| Log Birds | 0.002 (0.005) | 0.002 (0.006) | 0.020 (0.015) | | | | 0.014 (0.014) | |
| *Wetland Policy* |  |  |  | | | |  | |
| No Net Loss WP | 0.308 (1.312) | -0.656 (1.494) | -1.991 (1.493) | | | | -2.415 (1.509) | |
| ESS Goal WP | 5.462\*\* (2.313) | 2.511 (2.249) |  | | | |  | |
| Use Penalties | 0.452 (1.536) | 0.226 (1.868) |  | | | |  | |
| Use Incentives | -6.847\* (3.140) |  |  | | | |  | |
| *Study Characteristics* |  |  |  | | | |  | |
| Log WL Acre | 0.541\*\* (0.242) | 0.394 (0.282) | -0.457\*\* (0.144) | | | | -0.497\*\*\* (0.145) | |
| Log Year | 1.214\* (0.663) | 1.073 (0.784) | -1.055\* (0.568) | | | | -1.214\*\* (0.572) | |
| Peer Review | 4.117\*\*\* (1.128) | 3.169\*\* (1.290) | 0.197 (1.458) | | | | -0.276 (1.450) | |
| Econ. Val. Mthd. | 2.366 (1.608) | 2.032 (1.924) | 0.342 (1.496) | | | | -0.594 (1.310) | |
| Latitude | 0.060\*\* (0.026) | 0.051\* (0.030) | 0.076\* (0.039) | | | | 0.068\* (0.040) | |
| Longitude | 0.023\* (0.011) | 0.011 (0.010) | -0.003 (0.008) | | | | -0.009 (0.007) | |
| Constant | 1.306 (10.789) | 3.784 (13.317) | -11.940 (23.879) | | | | -21.818 (23.038) | |

**Table 3. Continued.**

|  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- |
|  | | Provisioning Model | | | | Regulation Model | | |
|  | | OLS | | RIM | | OLS | | RIM |
|  | | (1) | | (2) | | (3) | | (4) |
| N | 27 | | 27 | | 22 | | 22 | |
| R2 | 0.863 | |  | | 0.913 | |  | |
| Adjusted R2 | 0.644 | |  | | 0.796 | |  | |
| ICC (Adjusted) |  | | 0.28 | |  | | 0.07 | |
| ICC (Conditional) |  | | 0.11 | |  | | 0.01 | |
| Log Likelihood |  | | -54.839 | |  | | -44.599 | |
| Akaike Inf. Crit. |  | | 145.679 | |  | | 117.199 | |
| Bayesian Inf. Crit. |  | | 169.004 | |  | | 132.474 | |
| Residual Std. Error | 1.736 | |  | | 1.787 | |  | |
| F Statistic | 3.936\*\* | |  | | 7.825\*\*\* | |  | |
| Breusch-Pagan Test | 18.55 | |  | | 6.40 | |  | |
| Meta-regression TE | 15.3% | |  | | 15.81% | |  | |
| Benefit TE | 40% | |  | | 47.48% | |  | |

N denotes number of observations; \*\*\* denotes significance at 1%; \*\* denotes significance at 5%; \* denotes significance at 10%. OLS denotes ordinary least squares; RIM denotes random intercept mixed model; standard errors are in parenthesis; Log Amphibians; ESS Goal WP; Use Penalties; and Use incentives were excluded from the regulation model because of multicollinearity problem; TE denotes transfer error

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**Appendix 1**

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

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Paper** | **Study Year** | **Nation** | **Provisioning**  **(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 the specific type of ecosystem service

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Paper** | **Study Year** | **Nation** | **Provisioning**  **(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 |

1. Nutrient retention; water treatment/purification; groundwater recharge; climate regulation; disturbance regulation; erosion control; nutrient recycling; waste treatment; flood control; pollution reduction and nitrogen mitigation. [↑](#footnote-ref-1)
2. Crop production; livestock grazing/pasture; irrigation; fodder gathering; fuel/firewood; construction materials; food gathering; potash; open water/drinking; herbs; pollination; commercial fishing and hunting. [↑](#footnote-ref-2)