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Valuing ecosystem services for agricultural wetlands: A systematic review and meta-analysis

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Valuing Ecosystem Services for Agricultural Wetlands: A systematic review and meta-analysis

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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 the 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 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. Richardson et al. (2015) suggested that 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), but these studies did not focus on agricultural wetlands and the values of wetlands in agricultural landscapes were 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 of 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 precision water management, GPS aided ditching, improved tile drainage equipment and other engineering tools (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

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 work by 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 average values 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) reported 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 detail about wetland ecosystem service values or (ii) area of wetlands or information that enabled wetland area estimation.

The final database consisted of 45 papers, of which 52% were peer reviewed publications. The non-peer reviewed publications were reports (e.g., Leschine et al. 1997; Schuijt 2002), a working paper (Meyerhoff and Dehnhardt 2004) and a technical report (Emerton 2005). 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 the 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 2018 consumer price index 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 CO₂/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.

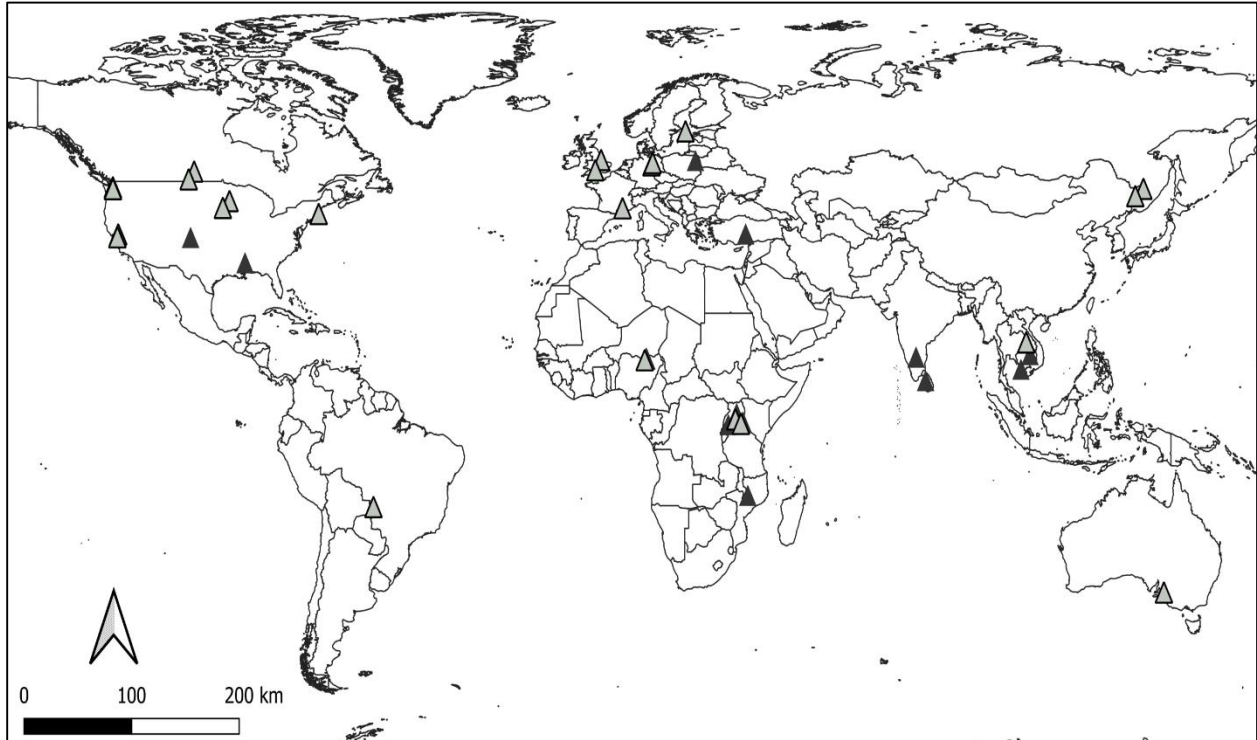
Wetland water storage was estimated by the storage capacity (m³) of water/ha. In some studies, the total surface area was provided and the volume was calculated from the area and average depth if reported in the paper. If a monetary value for water storage was not provided, then global averages reported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

Fig 1 Location of Study Sites for Provisioning and Regulating Wetland Ecosystem Services¹



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 reduce heteroscedasticity (Brander et al. 2013);

¹Dark and light grey triangles show study sites for regulating and provisioning ecosystem services, respectively.

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:

$$Y_i = \mathbf{X}\boldsymbol{\beta} + u_j + \varepsilon_i \quad (1)$$

where:

i = subscript i represents the ith observation.

j = subscript j represents the jth study.

Y_i = dependent variable representing the logarithm of the value of wetland ecosystem service (US\$/ha/year).

\mathbf{X} = vector of independent variables (including wetland policy variables, human population and economic indicators, and biodiversity richness indicators) and a constant term.

$\boldsymbol{\beta}$ = vector corresponding parameters of \mathbf{X} to be estimated.

u_j = stochastic error term for the jth study, which is assumed to be normally distributed with mean 0 and a variance (σ_u^2).

ε_i = stochastic error term for the ith observation, which is assumed to be normally distributed with mean 0 and a variance ($\sigma^2 I$).

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 had 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 if 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).

To calculate an index, we compiled the global species richness of birds from species range maps ($\approx 28 \times 28$ km) by Birdlife International (<http://www.birdlife.org/>). The global species richness of amphibians ($\approx 1 \times 1$ km) was compiled by the International Union for the Conservation of Nature (IUCN) and the Columbia University Center for International Earth Science Information Network (CIESCN) (IUCN & CIESCN 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 is 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 in place 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 policies within the same country; for instance, some provinces in Canada have a no net loss policy whereas others do not. However, for this study we focused 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 Lanzaova (2017) showed that larger lakes are more valued than smaller lakes, because some ecosystem services 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.

318 Economic valuation methods are listed in Woodward et al. (2001) and Brander et al. (2006) and
319 include methods such as production function, replacement cost, and contingent valuation. Peer
320 reviewed studies is a binary variable which takes on a value 1 if study is peer reviewed and 0
321 otherwise. We expect peer review to have a positive effect on wetland values (Ghermandi and
322 Nunes 2013; Reynaud and Lanzasova 2017) because researchers may be more encouraged to
323 publish studies that produce more significant wetland values. The variable descriptions and their
324 expected effects on wetland values are summarized in Table 1.

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

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				
<i>Economic Variables</i>				
Pop Density	Human population density	Continuous	H/km ²	+
AgTFP	Agricultural total factor productivity	Continuous	2015 AgTFP = 100	+
High-Income	High-income country	Binary	1,0	+
<i>Biodiversity Variables</i>				
Birds	Bird Species Richness	Continuous	Counts/Ha	+
Amphibian	Amphibian Species Richness	Continuous	Counts/Ha	+
<i>National Wetland Policy</i>				
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	+
<i>Study Characteristics</i>				
Wetland Area	Wetland Area	Continuous	Ha	+/-
Economic Valuation Method	Economic Valuation Method	Binary	1,0	+/-
Peer Review	Peer Review Journal Publication Status	Binary	1,0	+

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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 average, were relatively larger 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 (1,003 humans/km²) and standard deviation (2,467 human population/km²) of population density were greater for the study regions in the regulating model than for wetlands in the provisioning model with a mean of 755 (standard deviation human of 2,223) population/km².

More jurisdictions in the regulating model had identified a wetland policy for conserving wetland 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.

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). There were also more bird species associated with wetlands in the provisioning model (283/ha) than wetlands in the regulating model (194/ha).

362 **Table 2 Summary statistics for each variable used in the modelling²**

Model Variables	Provisioning Model			Regulating Model		
	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						
<i>Economic Variables</i>						
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
<i>Biodiversity Variables</i>						
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
<i>Wetland Policies</i>						
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
<i>Study Characteristics</i>						
Wetland Area	8.7E05 (2.8E06)	1.64	1.38e07	2.73E06 (6.6E06)	0.7	2.7E07
Economic Valuation Method.	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

² More information on the variables and their scale in Table 2 is provided in Table 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 (using 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 compared to the 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 correlated with high-income ($r = 0.68$). We also 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 human 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 result in a \$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), bird species richness (p-value = 0.11), wetland area (p-value = 0.66), amphibian species richness (p-value = 0.56) were not significant. 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.

400 **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
R ²	0.732
Adjusted R ²	0.613
Breusch Pagan Test Statistic	5.26
F Statistic	6.114**
AIC	36.97
Transfer Error Summary Statistics	
MR- RMS	1.94
MV- RMS	2.65
MR- MAD	1.68
MV- MAD	2.34

401 N denotes number of observations; *** denotes significance at 1%; ** denotes significance at 5%; * denotes
 402 significance at 10%; MR- RMS denotes meta regression root mean square; MV- RMS denotes mean value root
 403 mean square; MR-MAD denotes meta regression mean absolute deviation; MV-MAD denotes mean value mean
 404 absolute deviation.

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 correlated with ecosystem service goal ($r = 0.69$) and economic valuation method ($r = -0.70$), 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 all explanatory variables were < 10 , indicating a lack of multicollinearity.

The model results showed that a 1% increase in wetland area resulted in a 0.31% decrease in the value of regulating wetland ecosystem services ($p = 0.012$). A 1% increase in agricultural factor productivity produced a 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. The results above are provided in Table 4.

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)
Economic Valuation Method	0.515 (1.141)
Longitude	-0.005 (0.007)
Log Amphibians	1.187 (0.990)
ESS Goal WP	-1.913 (1.282)
<hr/>	
Model Summary Statistics	
<i>N</i>	23
<i>R</i> ²	0.865
Adjusted <i>R</i> ²	0.772
Breusch Pagan Test Statistic	9.07
F Statistic	9.267***
AIC	36.80
<hr/>	
Transfer Error Summary Statistics	
MR-RMS	2.28
MV-RMS	5.27
MR- MAD	2.00
MV- MAD	3.85

N denotes number of observations; *** denotes significance at 1%; ** denotes significance at 5%; * denotes significance at 10%; MR- RMS denotes meta regression root mean square; MV- RMS denotes mean value root mean square; MR-MAD denotes meta regression mean absolute deviation; MV-MAD denotes mean value mean absolute deviation.

4.0. Discussion and Policy Implications

Wetlands are highly valued because they produce services that are useful and beneficial to humans (Mitsch and Gosselink 2000). Therefore, the positive effect of increasing human population density on wetland values (both provisioning and regulating ecosystem services) is expected and is consistent with previous studies (see Mitsch and Gosselink 2000; Brander et al. 2006; Brander et al. 2013). One explanation is that higher populations near wetland areas would mean a greater number of people could benefit from local wetland services with improved access to the wetland areas.

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 countries 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 (AgTFP) has been shown to have a positive impact on regulating wetland values and a negative effect on provisioning wetland values elsewhere (provide reference). 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). As AgTFP increases, there is perhaps less pressure for 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. It could be that relatively fewer 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 poor 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 regulating and provisioning ecosystem service values tend to be negatively related to wetland area (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). 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 Lanzasova 2017).

Our study showed that 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 been reported previously (Ghermandi and Nunes 2013; Reynaud and Lanzasova 2017). Our study shows that the presence of a national wetland policy could possibly 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 benefit transfer methods. Traditionally, benefit transfer approaches simply used mean values

from relatively comparable wetland study sites to represent values for the target site. Our study applies a meta-regression model to tailor those values from comparable wetland study sites to more effectively develop values that represent the biophysical, social and economic context of the wetlands. In a review of 38 meta-regression valuation studies, Rosenberger (2015) 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% to 684%. Our estimated mean meta-regression APE and mean value APE were 200% and 385%, respectively (for the provisioning meta-regression model) and 168% and 234%, respectively (for the regulating model) which are consistent with Schutt (2021). In contrast, our estimated benefit transfer errors are considerably higher compared to the average transfer errors in the literature (Rosenberger 2015) This may be due to the lack of sufficient data ($n = 23$ for the regulating model and $n = 27$ for the provisioning model) to allow us to efficiently estimate a global meta-regression value function to value wetlands on agricultural landscapes. 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 localized studies to value wetlands, our models could be used to relate the value of wetlands with our benefit transfer tool (compared to the mean value transfer approach) and aid in land-use planning and wetland conservation policy development.

5. Conclusion

Our study advances the knowledge around evaluating wetland ecosystem services globally by estimating a wetland regulating meta-regression model that extends the regulating services of wetlands beyond water quality, water supply, and nutrient recycling, to include groundwater recharge, climate regulation, erosion control, waste treatment and flood control as well as by deriving a provisioning meta-regression model.

We find that the prediction errors from our models, compared to those from mean value unit transfers, were lower than similar estimates reported in the literature. This suggests our study results could be used to support the development of more reliable and representative wetland values using a benefit transfer approach compared to those values estimated through a mean value transfer method, especially in the absence of original valuation studies. For instance, the results can be used to help calculate the total value of wetlands in areas where localized studies are not available. This would enable planners to implement better informed wetland conservation policies and can assist in the estimation of the tradeoffs of wetland conversion or conservation on agricultural lands. Wetland managers generally consider the regional context of target wetlands, and we would recommend using the estimated meta-regression value functions to help develop estimates of local wetland values by selecting landscape appropriate levels of key independent variables in an analysis.

Our study indicated key variables to inform the effective implementation of a benefit transfer procedure to value wetlands are agricultural total factor productivity, the income level of a country, and the wetland area under study for the regulating model, and income level of a country, peer-reviewed journal articles, agricultural total factor productivity, latitude, and population density for the provisioning model.

528 While the studies used for this analysis were mostly based on study areas located in
529 developed countries, this is still useful in the context where there are significant pressures to
530 convert wetlands to the production of agricultural commodities. Future studies in developing
531 countries would enable a more accurate benefit transfer of wetland ecosystem service valuations.
532 In the meantime, our benefit transfer study can be useful to support their own localized
533 calculations to make policies that consider the total ecosystem services that wetlands provide,
534 and integrate them into land use planning.
535

Declarations

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Conflicts of interest/Competing interests

Apart from the funding information above, the authors have no conflicts of interest to declare that are relevant to the content of this article.

Availability of data and material

The datasets used and/or analyzed during the current study are available from the corresponding author on reasonable request.

Code availability

The codes used during the current study are available from the corresponding author on reasonable request.

Authors' contributions

All authors contributed to the study conception and design. Material preparation, data collection and analysis were performed by Asare Eric, Mantyka-Pringle Chrystal, Anderson Eric, Belcher Kenneth, and Clark Robert. The first draft of the manuscript was written by Anderson Eric and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Compliance with Ethical Standards

559 *Conflict of Interest*

560 The authors declare no conflicts of interest

561 *Research involving Human Participants and/or Animals*

562 The authors declare that this study did not involve human participants and/or animals

563 *Informed consent*

564 Not applicable

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Appendix 1

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

738

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

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

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

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[Click here to view linked References](#)

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Editor in-Chief
Wetlands Ecology and Management

October 2, 2021

Dear Sir/Madam,

I am happy to submit an original research article titled “Valuing Ecosystem Services for Agricultural Wetlands: A systematic review and meta-analysis” for publication in the Wetlands Ecology and Management Journal. The authors of this manuscript, Asare Eric, Mantyka-Pringle Chrystal, Anderson Erik, Belcher Kenneth and Clark Robert, have approved the manuscript and agree with its submission for publication.

Our manuscript evaluates, using a meta-analysis, a suite of biophysical and socioeconomic factors that potentially influence the value of ecosystem services provided by wetlands (provisioning and regulating) on agricultural landscapes. The results of this analysis suggested that per capita income, peer-reviewed journal publications, agricultural total factor productivity and population density had significant effect on provisioning values from wetlands, while agricultural total factor productivity, income level and wetland area had significant effects on regulating wetland ecosystem values. Also, we showed that meta-regression transfers errors were significantly lower than mean-value transfer errors from our estimated models.

The world has lost more than 50 percent of its original wetland area at a greater rate than many of the ecosystems in the world. The international wetland research community and policy makers need to understand the drivers of wetland values, on agricultural landscapes in different regions. Also, policy makers could use our estimated wetland regression value functions to inform

approaches to estimate wetland values on agricultural landscapes, which could provide the foundation for benefit cost analyses of land use policies and other initiatives to conserve wetlands and wetland function.

We confirm that this manuscript has not been published elsewhere and is not under consideration by another journal. We have no conflict of interest to disclose.

Thank you for your consideration!

Sincerely,

Eric Asare, Ph.D.
Postdoctoral Fellow,
University of Saskatchewan.