Values of natural and human-made wetlands: A meta-analysis

Andrea Ghermandi,[1,2] Jeroen C. J. M. van den Bergh,[3,4] Luke M. Brander,[5]
Henri L. F. de Groot,[6] and Paulo A. L. D. Nunes[1,2]

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[1] The values of goods and services provided by wetland ecosystems are examined through a meta-analysis of an expanded database of wetland value estimates and with a focus on human-made wetlands. This study extends and improves upon previous meta-analyses of the wetland valuation literature in terms of the number of observations, geographical coverage, wetland class and integrity, and the measurement of the effects of scarcity and anthropogenic pressure. We find that water quality improvement, nonconsumptive recreation, and provision of natural habitat and biodiversity are highly valued services. Substitution effects are observed through the negative correlation between values and abundance of other wetlands. Wetland values are found to increase with anthropogenic pressure. An extended metaregression model with cross effects shows that the valuation of specific services varies with the type of wetland producing them. Human-made wetlands are highly valued for biodiversity enhancement, water quality improvement, and flood control.

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1. Introduction

[2] The recognition of the wide range of ecological and economic benefits that natural wetland ecosystems provide to humans [*Turner*, 1991] has prompted increasing interest in the construction of human-made wetland ecosystems, which simulate the functions of natural wetlands in order to support human use [*Hammer and Bastian*, 1989]. Wetland ecosystems are generally constructed with the aim of replicating wetland processes such as water storage, flood retention, and water quality improvement for human benefit [*Kadlec and Knight*, 1996]. They may also be created with the broader aim of mimicking the foregone ecological functions of lost natural wetland ecosystems and compensating the destruction of natural habitats, such as mitigation wetlands constructed under the "no net loss of wetlands" policy in the USA.

[3] Purposefully planned, designed and operated human-made wetlands may provide a range of services well beyond the primary aim for their construction. Ancillary benefits of wastewater treatment wetlands may include, for instance, provision of habitat and wildlife diversity, support of recreational activities such as walking, bird and wildlife watching, water storage during periods of shortage and excess, and aesthetic value in urban environments [*Benyamine et al.*, 2004; *Knight*, 1997; *Knight et al.*, 2001]. Comparative studies investigating the ecological functions of both natural and human-made wetlands suggest that they fulfill similar ecological functions, even though the latter tend to resemble degraded natural wetlands rather than undisturbed reference ecosystems [*Campbell et al.*, 2002; *Brooks et al.*, 2005; *Confer and Niering*, 1992; *Balcombe et al.*, 2005].

[4] In this study, we use the technique of meta-analysis to investigate the provision of services of wetland ecosystems from an economic perspective and with a focus on the valuation of human-made wetlands. The paradigm adopted is an anthropocentric one, in which ecosystems are regarded as steering forces of human well-being insofar as they provide goods and services, and wetland values are determined by the consumption opportunities that they provide to humans [*Nunes and van den Bergh*, 2001]. In this approach, the values of wetland ecosystems are to be clearly distinguished from their ecological functions [*Woodward and Wui*, 2001; *Farber et al.*, 2002].

[5] Meta-analysis has been extensively used in environmental economics as a tool to synthesize the findings of primary valuation studies by means of a rigorous statistical analysis [*Bal and Nijkamp*, 2001]. Best practice guidelines for meta-analysis were developed [*Stanley*, 2001; *Nelson and Kennedy*, 2009] in order to deal with potential issues related to the heterogeneity of the environmental resources and economic instruments considered [*Smith and Pattanayak*, 2002], selection bias [*Hoehn*, 2006], heteroscedasticity, and correlation between observations [*Rosenberger and Loomis*, 2000a]. The potential of meta-analysis in identifying the sources of variation in empirical value estimates [*Johnston et al.*, 2003; *Scheierling et al.*, 2006] and as a tool for value transfer [*Bergstrom and Taylor*, 2006; *Rosenberger and Loomis*, 2000b] is generally acknowledged. Three previous meta-analyses of wetland values [*Brander et al.*,

[1]Fondazione Eni Enrico Mattei, Venice, Italy.
[2]Department of Economics, Cà Foscari University of Venice, Venice, Italy.
[3]Department of Economics and Economic History and Institute for Environmental Science and Technology, Universitat Autònoma de Barcelona, Bellaterra, Spain.
[4]ICREA, Barcelona, Spain.
[5]Institute for Environmental Studies, VU University Amsterdam, Amsterdam, Netherlands.
[6]Department of Spatial Economics, VU University Amsterdam, Amsterdam, Netherlands.

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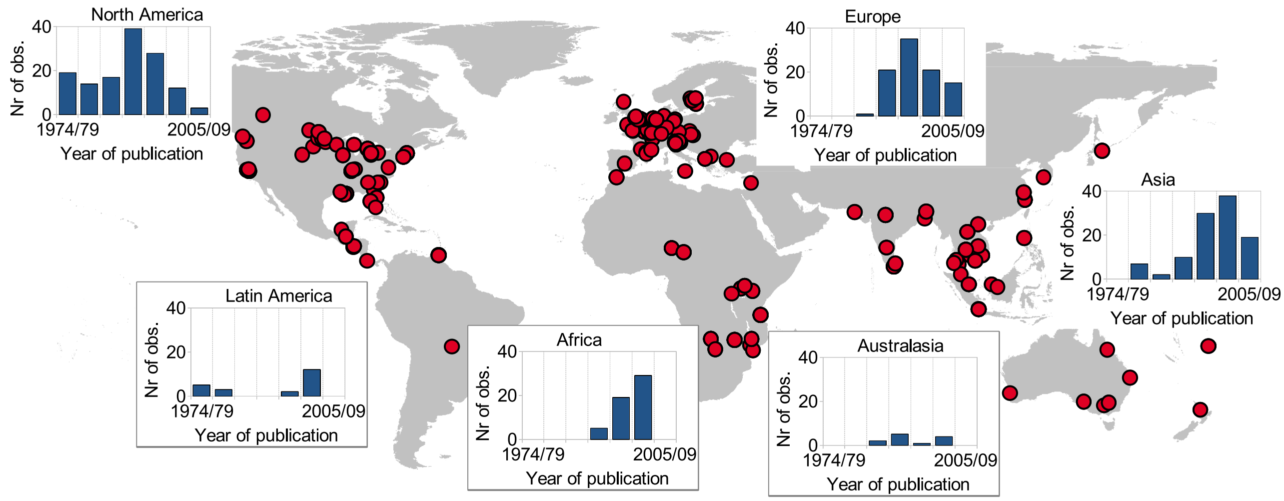

Figure 1. Number of observations of wetland values for 5 year intervals from 1974 to 2009 and for geographical locations of valued wetlands.

2006; *Brouwer et al.*, 1999; *Woodward and Wui*, 2001] provided a solid conceptual and empirical basis for the present investigation.

[6] The objectives of this study are twofold. First, we aim at improving the general understanding of both natural and human-made wetland values by conducting a meta-analysis that extends previous metaregression models with explanatory variables, such as the presence of substitute sites and the anthropogenic pressure exercised on the wetlands, which are chosen in order to get a better and more economically oriented explanation of observed differences in wetland valuations. Compared to previous studies, the meta-analysis relies on a much more comprehensive data set comprising 418 value observations derived from 170 valuation studies and 186 wetland sites worldwide. Second, we explore the variation in the valuation of human-made wetlands by means of a metaregression model that accounts for the interactions between wetland types and ecosystem services. We test formally whether human-made and natural wetlands are equally valued for flood protection, water quality improvement, and water storage and supply, i.e., the three main objectives for wetland construction (Hypothesis 1). We also assess whether ancillary benefits, such as support of recreational activities and biological diversity enhancement, provide a substantial contribution to the total economic value of human-made wetlands (Hypothesis 2).

[7] The organization of the remainder of this paper is as follows. In section 2 the characteristics of the data set used are described by means of descriptive statistics (section 2.1), an overview is given of the economic valuation studies of human-made wetlands (section 2.2), and the metaregression models are formulated (section 2.3). Section 3 presents the results of a metaregression estimation. Section 4 interprets the results and concludes.

2. Data and Conceptual Framework

2.1. Data Set of Wetland Valuation Studies

[8] The economic valuation of wetland services can proceed with different approaches depending on the type of service considered. Information on prices, costs, and how they affect the welfare of people whose income depends on wetlands can provide an estimate of the value of market activities, such as commercial fishing and hunting, harvesting of natural materials, and fuel wood collection. The cost of replacing services such as water quality improvement and flood control with engineered solutions may provide a measure of their benefits. Often, however, wetland services do not affect markets and market data are not available to value them. This holds for the welfare impact of recreational activities and aesthetic views, but also for passive benefits, such as the knowledge that a certain ecosystem exists or is protected for the benefit of future generations. In such cases, economic valuation may proceed by eliciting preferences from actual markets that are indirectly linked to the ecosystem service in question (as in the travel cost and hedonic pricing methods) or by simulating a market choice through a questionnaire administered to a sample of the affected population (as in the contingent valuation and in choice experiment methods).

[9] To support the analysis in this study we constructed a data set of wetland valuations consisting of 418 value observations from 170 valuation studies and 186 wetland sites. Figure 1 illustrates the geographical distribution of wetland values in the data set. The largest number of observations is from North America (132), but significant numbers come from Asia (106), Europe (93) and Africa (53). South America (22) and Australasia (16) are somewhat underrepresented. Compared to the overall distribution of Ramsar wetlands of international importance, the geographical distribution of wetlands in the data set is skewed toward sites located at temperate Northern latitudes and in the equatorial region. In particular, wetlands at latitudes higher than 45°N are underrepresented. The geographic dispersion of wetland studies with respect to the distribution of wetland ecosystems suggests the existence of a research priority bias in the wetland valuation literature [*Hoehn*, 2006]. With respect to such geographical bias, the database used in this study represents nevertheless an improvement with respect to previous meta-analyses of wetland values, which were considerably more biased toward North American wetlands. Such shift in the geographical distribution of studies reflects changes in the practice of wetland valuation, which has been shifting away from North America toward Europe, Asia and Africa.

Table 1. Overview of Valuation Studies of Human-Made Wetland Ecosystems

Wetland Site	Size, ha	Value[a]	Valuation Method	Reference
Cheimaditida and Zazari lakes, Greece	11,400	12,490–39,140	CVM	*Ragkos et al.* [2006]
Cley marshes, United Kingdom	176	1,008–3,904[b]	CVM	*Klein and Bateman* [1998]
Constructed wetlands in Sweden	6,400	4,080	Repl. cost	*Byström* [2000]
Little River/Rooty Creek, Georgia, United States	134	9,352	CVM	*MacDonald et al.* [1998]
De Wieden, Netherlands	5,200	25–387	NFI, TCM	*Hein et al.* [2006]
Empuriabrava, Spain	7	78,321	TCM	*Seguì-Amórtegui* [2004]
Hula, Israel	24,000	163	CVM	*Baron et al.* [1997]
Lac du Der, France	4,800	687	CVM	*Scherrer* [2003]
Lake Kerkini, Greece	6,250	9,144	CVM	*Oglethorpe and Miliadou* [2000]
Oxelösund, Sweden	22	12,635	CVM	*Cravener* [1995]
River Ancholme washlands, United Kingdom	800	8,331	Repl. cost	*Posford Duvivier Environment* [1999]
River Elbe floodplains, Germany	55,000	114–2,066[b]	CVM, repl. cost	*Meyerhoff and Dehnhardt* [2007]
River Nar washlands, United Kingdom	150	8,201	Repl. cost	*Posford Duvivier Environment* [2000]
Upper and lower Bhoj wetlands, India	3,229	211–4,031[b]	Market prices, repl. cost, CVM	*Verma* [2001]
Waza Logone, Cameroon	20,000	1.7–101[b]	Market prices, prod. function, repl. cost	*Loth* [2004]
Kala Oya basin, Sri Lanka	285	1,908–13,269[b]	Market prices, prod. function, repl. cost	*Vidanage et al.* [2004]
Hangzhou Botanical Garden, China	0.06	151,810–8,013,754[b]	CVM, repl. cost	*Yang et al.* [2008]
Whangamarino, New Zealand	10,320	197–705	CVM	*Kirkland* [1988]

[a]The reported value is standardized to 2003 USD/ha/year using GDP deflators and PPP index as described in section 2.3.
[b]The estimated values vary according to the type of service provided.

[10] To identify candidate studies, we rely on a more stringent definition of wetlands than given by the Ramsar Convention, according to which any area of “marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six meters” is to be considered a wetland site. This definition potentially encompasses permanently inundated ecosystems such as all areas of coral reefs, sea-grass beds, most rivers and shallow lakes [*Scott and Jones*, 1995]. Such ecosystem types were excluded from this analysis not to overstretch the scope of the study and since their classification as wetland ecosystems is rather controversial. Furthermore, such ecosystems are of limited relevance for the analysis of human-made wetlands values.

[11] The data set developed by *Brander et al.* [2006] provided the starting point for our analysis. The original data set was substantially enlarged with new observations from recent studies. Studies were retrieved through searching online valuation databases, libraries, and contacting authors. The wide range of market-based and nonmarket valuation studies considered is described in section 2.3. Only primary valuation studies were included in the data set; that is, value transfer studies were not considered. The investigation also explored “grey literature”, including 86 reports for both public and private institutions, consultancy studies, and unpublished research results. Efforts to retrieve studies not published in the English language led to the inclusion of 16 more studies.

2.2. Literature on Values of Human-Made Wetlands

[12] The number of studies assessing the economic values of human-made wetlands is rather limited. Table 1 provides an overview of the available studies, implemented valuation methods, some basic characteristics of the valued sites and the estimated values standardized to Purchasing Power Parity (PPP) adjusted units.

[13] Several studies have assessed the benefits of constructed treatment wetlands, either in terms of their water quality improvement service or ancillary benefits. The value of abating nitrogen load from agricultural sources and upgrading secondary municipal effluent for water recycling were assessed for Swedish wetlands draining into the Baltic Sea [*Byström*, 2000] and for the Empuriabrava constructed wetlands in Catalonia, Spain, respectively [*Seguì-Amórtegui*, 2004]. The value estimated with the travel cost method in Empuriabrava (78,321 USD/ha/year) is substantially higher than the replacement costs for the Swedish sites (4080 USD/ha/year). The value of upgrading and reusing the eutrophic effluent of an ornamental fishpond in Hangzhou, China, by means of a 600 m^2 constructed wetland was estimated using the contingent valuation method (CVM) and the replacement cost method and ranged between 294,729 USD/ha/year and 8,013,754 USD/ha/year [*Yang et al.*, 2008]. The welfare impacts of other wastewater treatment wetlands services such as wildlife habitat enhancement and provision of passive values were assessed for constructed wetlands in the State of Georgia, USA [*MacDonald et al.*, 1998], Oxelösund, Sweden [*Cravener*, 1995], and Hangzhou, China [*Yang et al.*, 2008]. The estimates obtained with CVM ranged between 9352 USD/ha/year in Georgia and 151,830 USD/ha/year in Hangzhou.

[14] The value of artificial impoundments providing water storage was elicited in several studies in Europe and Asia. Estimates were highest for drinking and irrigation water supply, ranging between 4031 USD/ha/year in the Bhoj wetlands in India [*Verma*, 2001] and 13,269 USD/ha/year in Sri Lanka [*Vidanage et al.*, 2004]. Passive values and the value of supporting various types of recreational activities were elicited by means of CVM. The value of supporting recreational activities ranged between 687 USD/ha/year in Lac du Der, France [*Scherrer*, 2003] and 2048 USD/ha/year in India [*Verma*, 2001], while passive value estimates for Lake Kerkini in Greece were as high as 9144 USD/ha/year [*Oglethorpe and Miliadou*, 2000].

[15] Human-made wetlands created to provide flood protection and areas for flood storage in river floodplains were investigated in various locations in Europe. The value of flood protection along the Nar and Ancholme rivers in the UK was estimated to be 8201 USD/ha/year and 8331 USD/ha/year [*Posford Duvivier Environment*, 1999, 2000]. Other services provided by this type of wetland ecosystem include nutrient

removal and biodiversity enhancement. The value of such services was estimated in 114–2066 USD/ha/year for nutrient removal and 1942 USD/ha/year for biodiversity enhancement at various locations along the Elbe River in Germany [*Meyerhoff and Dehnhardt*, 2007].

[16] Another group of valuation studies concerns wetlands that are restored at the location of previously drained natural wetlands or that are not entirely artificial in origin but include constructed sections. Two studies eliciting the recreational value of a restored wetland in Israel [*Baron et al.*, 1997] and the benefits of provisioning services in a restored floodplain wetland in Cameroon [*Loth*, 2004] reported values of 163 USD/ha/year and in the range 2–101 USD/ha/year, respectively. Valuations of various services provided by wetlands that include human-made sections were conducted in various European countries and in New Zealand. For these sites, the conservation and enhancement of natural habitats was the most highly valued ecosystem service with monetary estimates ranging between 197 USD/ha/year in Whangamarino, New Zealand [*Kirkland*, 1988], and 27,678 USD/ha/year in Cheimaditida and Zazari, Greece [*Birol et al.*, 2006]. Cultural services supporting various types of recreational activities were also highly valued, ranging from 295 USD/ha/year in De Wieden, Netherlands [*Hein et al.*, 2006], and 3903 USD/ha/year in the Cley marshes, UK [*Klein and Bateman*, 1998].

[17] The presented overview of the literature allows for the formulation of the hypotheses on the values of human-made wetlands that will be tested in section 3 by analyzing sign and significance of the coefficients of the respective variables in the metaregression model. First, the highest values are reported for the provision of the specific services for which the wetlands are constructed, e.g., wastewater treatment wetlands provide high values for water quality improvement. Second, a large number of valuation studies focus on cultural services such as support of recreational activities, and enhancement of natural habitat and biodiversity, suggesting that they might be important components of the total economic value of such ecosystems. Accordingly, in section 3 we will investigate whether (1) water quality improvement, water supply, and flood protection are the most highly valued services of human-made wetlands and whether (2) cultural services are highly valued as well in such ecosystems.

2.3. Specification of the Metaregression Model and Explanatory Variables

[18] The base meta-analytical regression model is specified as follows:

$$\ln(y_i) = a + b_S X_{Si} + b_W X_{Wi} + b_C X_{Ci} + u_i \qquad (1)$$

where the dependent variable $\ln(y_i)$ is the natural logarithm of the wetland value expressed in 2003 USD per hectare per year. The subscript i is an index for the 418 observations, a is a constant term, b_S, b_W and b_C are vectors containing the coefficients of the explanatory variables. X_{Si}, X_{Wi} and X_{Ci} are study-, wetland- and context-specific explanatory variables, respectively, and u is an error term that is assumed to be normally distributed and with a mean value of zero.

[19] To allow for a comparison between wetland values that have been calculated in different years and expressed in different currencies and metrics (e.g., WTP per household per year, capitalized values, and marginal value per acre) values were standardized to a common metric and currency. Standardizing wetland values to WTP per person as done by *Brouwer et al.* [1999] was not possible because several of the considered valuation methods do not produce WTP estimates. Following *Brander et al.* [2006] and *Woodward and Wui* [2001], all value estimates were standardized to the metrics of US$ per hectare per year. WTP per person or household were converted given information on the wetland area and the relevant population size. Primary studies that did not clearly report the size of the population on which WTP estimates should be aggregated or the areal extension of the ecosystem were excluded, avoiding potential discrepancies between the population of beneficiaries accounted for in the original valuation study and in the meta-analysis. The yearly flux of benefits from capitalized value estimates was calculated based on the discount rate and time period given in the primary studies. Values referring to different years were deflated using appropriate factors from the World Bank Millennium Development Indicators [*World Bank*, 2006], while differences in purchasing power among the countries were accounted for by the PPP index provided by the Penn World Table [*Heston et al.*, 2006].

[20] Table 2 provides an overview of the explanatory variables. They consist of three categories, namely characteristics of the (1) primary study X_S, (2) valued wetland X_W and (3) socioeconomic and geographical context X_C.

[21] The study characteristics accounted for in the model include the valuation method used, the year of publication and a dummy distinguishing between marginal and average values. The array of valuation methods used in the primary studies to assess wetland values include market-based methods, revealed preference methods, and stated preference methods. A series of dummy variables is included in the metaregression model to account for the heterogeneity of methods, since not all of them have a strong basis in welfare theory and produce estimates using different welfare measures. Also, to distinguish between marginal and average per hectare values, a dummy variable that equals one for marginal values is introduced [*Brander et al.*, 2006]. Such variable captures the distinction between average per hectare values, as calculated, for instance, dividing a total value by the size of the ecosystem, and marginal values which capture a value variation due to a small change in the extent of the ecosystem. Both can be expressed in annual per hectare values, but will in general not coincide due to, for instance, diminishing marginal returns to wetland size.

[22] Characteristics of the valued wetland site are the type and size of the wetland, the services provided, and the level of pressure exercised on it by human activities. The five basic wetlands systems of the Classification of Wetlands and Deepwater Habitats of the United States [*Cowardin et al.*, 1979] are used in combination with a sixth category, which identifies human-made ecosystems. The five basic wetland systems of the Cowardin classification are marine, estuarine, riverine, palustrine and lacustrine wetlands. Lacustrine systems include wetland and deepwater habitats. They are situated in a topographic depression or a dammed river channel, and lack trees and widespread persistent emerging vegetation. Palustrine systems include all nontidal wetlands dominated by trees, shrubs, and persistent emergent vegetation, and all such wetlands that occur in tidal areas where salinity due to ocean-derived salts is below 0.5 ‰. In classifying human-made wetlands as a separate category from natural wetlands

Table 2. Explanatory Variables Used in the Basic Metaregression Model[a]

Group	Variable	Units and Measurement	Mean (SD)	*N*
Study (X_S)	Contingent valuation method	Binary (range: 0 or 1)	0.21 (0.41)	89
	Hedonic pricing	Binary (range: 0 or 1)	0.01 (0.10)	4
	Travel cost method	Binary (range: 0 or 1)	0.11 (0.32)	48
	Replacement cost	Binary (range: 0 or 1)	0.17 (0.38)	71
	Net factor income	Binary (range: 0 or 1)	0.13 (0.34)	54
	Production function	Binary (range: 0 or 1)	0.07 (0.25)	28
	Market prices	Binary (range: 0 or 1)	0.32 (0.47)	133
	Opportunity cost	Binary (range: 0 or 1)	0.02 (0.15)	9
	Choice experiment	Binary (range: 0 or 1)	0.03 (0.17)	13
	Year of publication	Number of years since first valuation (1974)	21.77 (7.85)	418
	Average	Omitted category	-	366
	Marginal	Binary (range: 0 or 1)	0.12 (0.33)	52
Wetland (X_w)	Estuarine	Binary (range: 0 or 1)	0.31 (0.46)	129
	Marine	Binary (range: 0 or 1)	0.23 (0.42)	98
	Riverine	Binary (range: 0 or 1)	0.35 (0.48)	146
	Palustrine	Binary (range: 0 or 1)	0.31 (0.46)	131
	Lacustrine	Binary (range: 0 or 1)	0.25 (0.43)	105
	Human-made	Binary (range: 0 or 1)	0.09 (0.29)	39
	Wetland size	Natural log of hectares	9.26 (3.12)	418
	Flood control and storm buffering	Binary (range: 0 or 1)	0.13 (0.34)	54
	Surface and groundwater supply	Binary (range: 0 or 1)	0.12 (0.32)	49
	Water quality improvement	Binary (range: 0 or 1)	0.12 (0.33)	52
	Commercial fishing and hunting	Binary (range: 0 or 1)	0.24 (0.43)	99
	Recreational hunting	Binary (range: 0 or 1)	0.17 (0.38)	71
	Recreational fishing	Binary (range: 0 or 1)	0.17 (0.37)	69
	Harvesting of natural materials	Binary (range: 0 or 1)	0.17 (0.37)	70
	Fuel wood	Binary (range: 0 or 1)	0.07 (0.26)	31
	Nonconsumptive recreation	Binary (range: 0 or 1)	0.23 (0.42)	98
	Amenity and aesthetics	Binary (range: 0 or 1)	0.10 (0.30)	43
	Natural habitat and biodiversity	Binary (range: 0 or 1)	0.13 (0.33)	53
	Low pressure	Omitted category	-	150
	Medium-low pressure	Binary (range: 0 or 1)	0.42 (0.49)	175
	Medium-high pressure	Binary (range: 0 or 1)	0.16 (0.36)	65
	High pressure	Binary (range: 0 or 1)	0.07 (0.25)	28
Context (X_C)	GDP per capita[b]	Natural log of 2003 dollars (PPP)	9.32 (1.34)	418
	Population density[c]	Natural log of inhabitants in 50 km radius	12.79 (1.52)	418
	Wetland abundance[c]	Natural log of hectares in 50 km radius	9.47 (3.31)	418

[a]Note: The number of observations for the variables valuation method, wetland type, and service provided do not add up to 418. This is due to the fact that individual observations may pertain to two or more levels. *N* = number of observations for each variable or variable level; SD = standard deviation.
[b]At country level but for observations from USA (state) and EU countries (NUTS-2).
[c]Referring to year 2000.

we follow the Ramsar classification system of wetland types (available at http://www.ramsar.org/ris/key_ris_types.htm). Since wetland ecosystems may include areas with different characteristics, the same observation may refer to two or more wetland types. Similarly, wetlands that include both artificial and natural sections (i.e., Whangamarino, Cley marshes, Cheimaditida-Zazari and de Wieden) are classified as simultaneously belonging to the category of human-made wetlands and to one (or more) of the categories of natural wetlands. Such approach allows to account for ecosystems with characteristics that are intermediate between natural and human-made, such as those referred to as "heavily modified water bodies" in the European Water Framework Directive.

[23] The ecosystem services provided by wetlands are classified based on the classification proposed in the Millennium Ecosystem Assessment [*de Groot et al.*, 2006]. The largest number of observations in the data set relates to cultural services (309 observations) and provisioning services (257 observations), while relatively less information is available in the literature for regulating services (105 observations). No valuation could be included for provision of genetic materials, climate regulation, erosion protection, spiritual and educational values, and support of pollinators.

[24] The presence of pressure by human activities is accounted for in the metaregression model as it may affect the ecological status of a wetland and the level of provision of ecosystem goods and services. Since direct observations of the ecological status are lacking for most of the wetlands in the data set, an index was constructed that accounts for the degree of anthropogenic pressure exerted and may be interpreted as a broad, landscape assessment of a wetland's ecological conditions [*Fennessy et al.*, 2004]. The index takes into account three criteria: (1) the presence of alterations in the natural hydrologic regime of the wetland as induced, for instance, by the construction of dikes to regulate the water level in the wetland; (2) whether the wetland is located in an urban or rural setting; and (3) the site's protection status (viz. Ramsar site, national park, nature reserve or not protected). Each criterion is evaluated as a binary variable (controlled/natural hydrology, urban/rural, protected/not protected) and the index consists of a categorical predictor with four levels of pressure. The lowest level of pressure (i.e., all binary

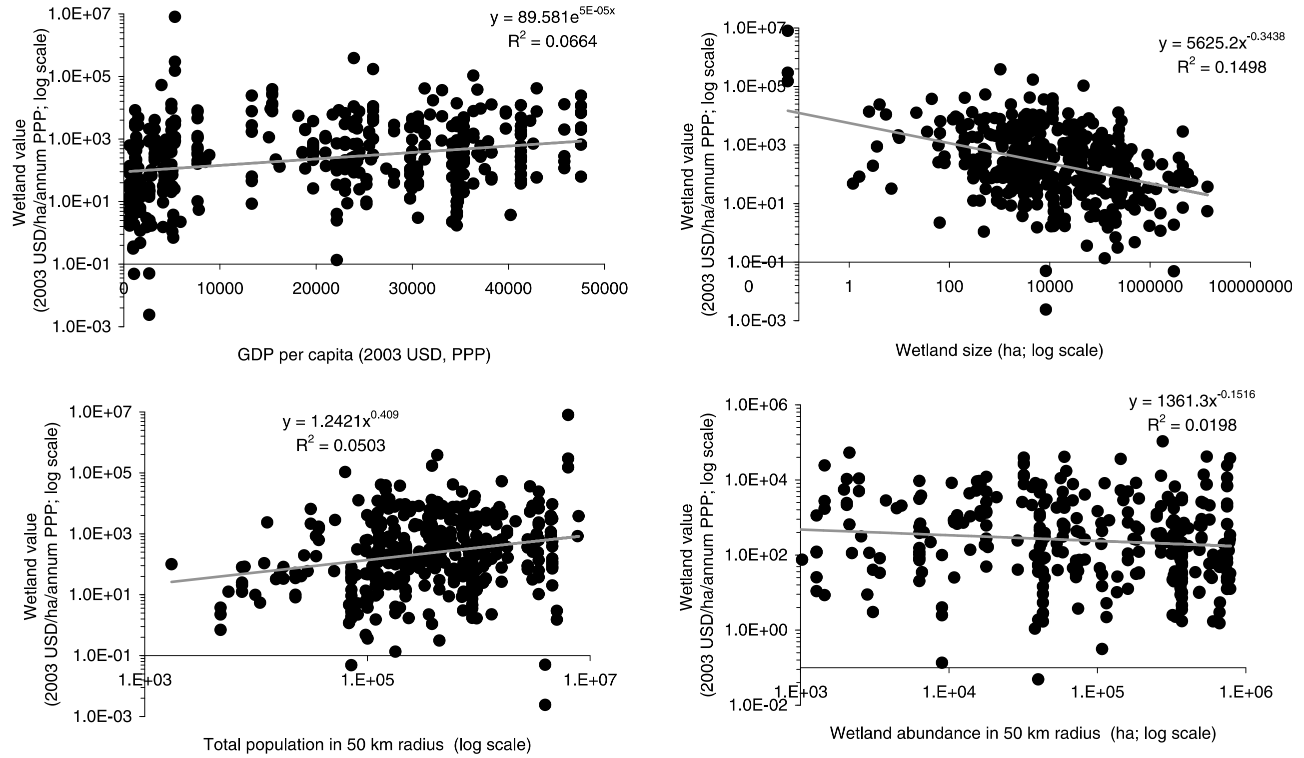

Figure 2. Standardized wetland value plotted against real per capita GDP (top left), wetland size (top right), total population (bottom left) and wetland abundance (bottom right) in a 50 km radius from the center of the valued wetland site.

variables equal to zero) identifies wetland sites with natural hydrology, located in a rural setting and protected. At the other end of the range, "high pressure" identifies unprotected wetlands with controlled hydrology that are located in an urban environment. The categories "medium low" and "medium high" pressure identify intermediate states.

[25] Three contextual variables are included in the metaregression model: real Gross Domestic Product (GDP) per capita; number of inhabitants surrounding the wetland; and total wetland area in proximity of the valued site. Contextual characteristics are expected to significantly influence the valuation estimates since environmental valuation studies carried out at different geographical sites and involving populations with different socioeconomic characteristics and consumer preferences typically produce different outcomes [*Brouwer*, 2000]. The values of real GDP per capita used in the metaregression model are measured in 2003 USD and calculated at the national level with the exception of observations from the US and EU countries, for which values are calculated for the individual states and NUTS2 regions, respectively. Due to the lack of distance decay testing in most primary studies, we relied on a constant radius of 50 km around the geographic center of the valued wetland to assess the total population and abundance of wetland ecosystems in its surroundings. The values of population and wetland abundance were estimated applying GIS techniques to the Global Demography Project map (available at http://sedac.ciesin.columbia.edu/gpw) and the Global Lakes and Wetlands Database map [*Lehner and Döll*, 2004].

[26] Figure 2 provides some descriptive statistics that summarize the variability of wetland values, expressed in 2003 USD, according to wetland size and context characteristics. A positive correlation with the wetland value is found for per capita GDP and total population living in a 50 km radius around the wetland centre, and a negative correlation for wetland size and wetland area within a 50 km radius. As indicated by the low values of goodness of fit, however, none of the variables alone explains a large proportion of the variation in the values.

[27] In addition to the base metaregression model (equation (1)), we also estimate an extended model that includes a series of cross-effect variables that capture the relationship between the provision of a specific wetland service and the type of wetland that provides it. In addition to the study and context characteristics discussed for the basic metaregression model, the extended model includes 66 dummy variables (11 wetland services multiplied by 6 wetland types). The use of cross products in meta-analysis is a simple and attractive way to statistically test for possible interactions between explanatory variables [*Nunes et al.*, 2009].

3. Econometric Results

[28] The results obtained for different specifications of the basic metaregression model described in equation (1) are presented in Table 3. Model A includes all explanatory variables in Table 2 and all observations. In model B, the dummy variables identifying the valuation method are dropped from the regression. Model C and model D address the issue of dependency among effect sizes by weighting observations (model C) and selecting only one observation per study (model D). In the estimated semilogarithmic model, the coefficients measure the constant proportional or relative change in the dependent variable for a given absolute change

Table 3. Results Obtained With the Basic Metaregression Model of Wetland Values[a]

Variable	Model (A) Full Model Coeff.	Model (A) SE	Model (B) Reduced Model Coeff.	Model (B) SE	Model (C) Weighted Effects Coeff.	Model (C) SE	Model (D) Single Effect Coeff.	Model (D) SE
Contingent valuation method	0.043	0.531	-	-	-	-	-	-
Hedonic pricing	−1.342	1.209	-	-	-	-	-	-
Travel cost method	−0.633	0.530	-	-	-	-	-	-
Replacement cost	−0.472	0.527	-	-	-	-	-	-
Net factor income	−0.411	0.489	-	-	-	-	-	-
Production function	−0.902	0.560	-	-	-	-	-	-
Market prices	−0.632	0.461	-	-	-	-	-	-
Opportunity cost	−1.231	0.810	-	-	-	-	-	-
Choice experiment	1.188	0.812	-	-	-	-	-	-
Year of publication	−0.054***	0.018	−0.041**	0.016	−0.029	0.027	−0.045	0.028
Marginal	1.032***	0.375	0.713**	0.338	0.643	0.479	0.320	0.482
Estuarine	0.321	0.290	0.270	0.288	0.452	0.480	0.720	0.482
Marine	0.775***	0.282	0.754***	0.280	0.789*	0.462	0.971**	0.466
Riverine	0.360	0.259	0.380	0.257	0.434	0.422	0.490	0.425
Palustrine	−0.380	0.282	−0.480*	0.271	−0.280	0.452	−0.107	0.461
Lacustrine	0.268	0.277	0.332	0.268	0.364	0.430	0.528	0.429
Human-made	1.167***	0.411	1.023**	0.403	1.188*	0.627	1.018	0.628
Wetland size	−0.247***	0.042	−0.234***	0.040	−0.245***	0.063	−0.259***	0.064
Flood control, storm buffering	0.432	0.348	0.432	0.329	0.286	0.538	0.200	0.540
Surface and groundwater supply	−0.037	0.355	−0.099	0.334	−0.430	0.602	−0.715	0.586
Water quality improvement	0.677*	0.388	0.727**	0.332	0.720	0.566	1.224**	0.585
Commercial fishing and hunting	0.301	0.276	0.266	0.253	0.344	0.424	0.708	0.441
Recreational hunting	−0.905***	0.347	−1.007***	0.334	−0.743	0.557	−1.123**	0.556
Recreational fishing	0.033	0.355	−0.082	0.349	−0.060	0.565	−0.091	0.586
Harvesting of natural materials	−0.140	0.299	−0.202	0.286	−0.143	0.507	−0.499	0.513
Fuel wood	−1.031**	0.418	−0.968**	0.416	−0.842	0.709	−1.156	0.774
Nonconsumptive recreation	0.381	0.332	0.670**	0.303	0.287	0.466	0.187	0.480
Amenity and aesthetics	0.528	0.424	0.529	0.392	0.969*	0.544	0.963*	0.559
Natural habitat, biodiversity	0.580	0.375	1.143***	0.330	1.168**	0.464	1.189**	0.471
Medium-low human pressure	0.564**	0.258	0.572**	0.256	0.805*	0.426	0.998**	0.435
Medium-high human pressure	1.130***	0.359	1.243***	0.358	1.260**	0.575	1.430**	0.603
High human pressure	2.093***	0.505	1.992***	0.500	1.922**	0.871	2.233**	0.878
GDP per capita	0.295**	0.118	0.358***	0.110	0.237	0.199	−0.028	0.199
Population in 50 km radius	0.399***	0.075	0.399***	0.073	0.321***	0.123	0.260**	0.124
Wetland area in 50 km radius	−0.064*	0.036	−0.058	0.035	−0.076	0.058	−0.107*	0.057
Constant	0.854	1.856	−0.681	1.652	1.245	2.962	5.277*	2.987
Number of observations	416		416		169[b]		169	
R^2	0.49		0.47		0.46		0.47	
Adjusted R^2	0.45		0.44		0.36		0.37	

[a]Note: OLS results. SE, standard error; Coeff., coefficient; significance is indicated with ***, **, and * for 1%, 5%, and 10% statistical significance levels, respectively.

[b]Takes into account the different weights given to the 416 observations.

in the value of the explanatory variable. For the explanatory variables expressed as logarithms, the coefficients represent elasticities, that is, the percentage change in the dependent variable given a one percentage change in the explanatory variable.

[29] In the results obtained for model A, the low significance of the coefficients on valuation methods suggests that methodological heterogeneity in the primary studies does not influence the regression results in any substantial sense. This provides empirical support for pooling estimates from the various valuation methods after standardizing them to a common unit. Although one may expect on theoretical and empirical grounds that different methods will produce different outcomes, previous wetland meta-analyses fail to give clear indications on the relative sizes of value estimates. *Woodward and Wui* [2001] found positive and significant coefficients for replacement cost and hedonic pricing, while *Brander et al.* [2006] reported a positive coefficient for contingent valuation. The lack of significance of the methodological dummies in this study may be related to the moderate correlation with ecosystem services or the presence of observations which report results obtained combining different valuation methods. The results in Table 3 are obtained with the regression of 416 observations rather than 418, since two observations derived from a mangrove restoration study in Vietnam [*Hoang Tri et al.*, 1998] that were identified as influential outliers on the regression coefficients were dropped from the final regression. This did not affect sign and significance of the coefficient estimates, with the exception of the variable "opportunity cost" which becomes statistically insignificant. Coefficients and standard errors presented in Table 3 are obtained with OLS. Conducting the regressions with the Huber-White/sandwich estimators results in the coefficients of "production function" and "opportunity cost" becoming significant at the 10% level in model A. "Water quality improvement", on the other hand, becomes insignificant.

[30] Due to their limited statistical significance, the dummy variables identifying the valuation method are dropped from the regression in model B. The regression with robust estimators does not change the significance of any of the estimated coefficients. A series of diagnostic tests were per-

formed in order to investigate the robustness of the ordinary OLS regression results of model B. The analysis of residuals indicates that they are distributed between a maximum value of 3.167 and a minimum of −3.094 with mean −0.0005 ± 1.003. The Shapiro-Wilk test (p level = 0.860) does not reject the assumption of normal distribution of the residuals. Similarly, the null hypothesis of homogenous variance of the residuals cannot be rejected by means of White's test (p level = 0.143) and Breusch-Pagan test (Prob. > χ^2 = 0.764). The five largest leverages and the five most influential observations on estimated parameters were identified and the metaregression with basic model B was rerun excluding them from the analysis. All signs and significance of the coefficients were unchanged, with the exception of the coefficients of flood control and wetland abundance in 50 km radius, which become significant at the 10% and 5% level in the regression, respectively, without the most influential observations. The coefficient of palustrine wetlands becomes insignificant. Multicollinearity between the variables is not an issue of concern (the maximum Variance Inflation Factor is equal to 2.24) and visual inspection of the plots of standardized residuals against explanatory variables did not show substantial deviation from linear behavior. Regarding model specification, both the link test for model specification (p level = 0.571) and the regression specification error test (RESET) for omitted variables (p level = 0.908) do not suggest specification errors. Finally, since the main focus of the study is on human-made wetlands, we repeated the regression with model B excluding valuations of the constructed wetland in Hangzhou, China [*Yang et al.*, 2008], which are outliers in the estimated values (see Table 1). The significance of all coefficients is unchanged with exception of the coefficients of palustrine wetlands and wetland abundance in 50 km radius, which become insignificant and significant, respectively, at 5% level.

[31] In the metaregression models A and B we implicitly assume the 416 observations to be independent. Although dependency issues between observations derived from the same study are unlikely to play a key role in this study since the average and maximum number of observations per study, 2.4 and 12, respectively, are small compared to the size of the sample, a multilevel regression of an earlier version of the data set indicated the presence of significant authorship effects [*Ghermandi et al.*, 2008]. To investigate the potential influence of correlation between observations on the regression results we conducted two additional regressions using some of the techniques suggested in the literature to deal with dependency across observations [*Nelson and Kennedy*, 2009]. First, we conducted a regression weighting observations, in which each study in the data set receives equal weight, instead of each observation as in ordinary OLS (model C). This approach has the advantage of addressing dependency without discarding any observation from the complete set. Second, we performed a regression in which each study is represented by a single observation. Among the approaches suggested in the literature using the mean or median effect [*Matt and Cook*, 1994], the most theoretically relevant estimate [*Matt and Cook*, 1994], or a randomly selected observation [*Bijmolt and Pieters*, 2001], we implemented the latter and randomly selected for regression of model D one observation for each study. The results of the metaregression with models C and D are shown in Table 3. The R^2 value is the same or slightly lower than in model B and the adjusted R^2 decreases to 0.36 and 0.37, respectively, due to the smaller number of observations. Moreover, several coefficients lose statistical significance including that of the income variable, which is at odds with theoretical and empirical expectations. We conclude that model C and D do not constitute a substantial improvement in the metaregression with respect to model B.

[32] Most of the coefficients estimated for model B are statistically significant and with the expected sign. Wetland type appears to significantly affect the value. Palustrine wetlands produce low values compared to the other kinds of wetlands, whose coefficients are all positive. Human-made and marine wetlands are the most highly valued wetland types. Of the wetland functions, the coefficients on fuel wood and recreational hunting are negative, while the coefficient on water quality improvement, nonconsumptive recreation and provision of natural habitat and biodiversity are large and positive. The coefficient on wetland size is negative, indicating decreasing returns to scale, and marginal values are higher than average values. The coefficients on the environmental pressure variables are positive and increasing with pressure. Regarding context variables, wetland values are positively related both to GDP per capita (the coefficient comprised between 0 and 1 indicating an inelastic income effect) and to the population living in the area surrounding the valued wetland site. Further testing introducing a quadratic term on the GDP per capita variable did not provide evidence for the existence of a quasi Kuznets curve relationship. There is a negative relationship between wetland abundance and the value of the wetland, although the coefficient is not highly statistically significant (p level = 0.101).

[33] Table 4 presents the results for the extended metaregression model with cross effects. Although the extended model includes also the variables of model B in addition to the cross effects, the focus in Table 4 is on the cross-effect variables only since the signs and significance of the coefficient estimates for study and context variables remain unchanged as compared to the base metaregression model B. Table 4 also shows the number of observations available for each cross-effect variable. This provides a further indication of how the valuation studies are distributed across wetland types and services and helps to identify data gaps in the valuation literature.

[34] The Shapiro-Wilk test (p level = 0.049) indicates a certain deviation in the distribution of the residuals from the normal distribution. Since, however, the analysis of interquartile range does not identify any severe outlier in the sample and the kernel density plot does not reveal substantial deviation from normality, such deviation is considered of minor importance for the interpretation of the results. Further diagnostic testing does not provide indications of heteroscedasticity, multicollinearity (the maximum Variance Inflation Factor is equal to 7.45) or model misspecification (link test p level = 0.987, RESET p level = 0.668). Excluding the outlier observations from *Yang et al.* [2008] causes the coefficient on the cross-effect variable linking human-made wetlands and water quality improvement to become statistically insignificant.

[35] The coefficients of several cross-effect variables turn out to be statistically significant. The highest values for water quality improvement are provided by estuarine and human-made ecosystems. The results suggest that estuarine wetlands also provide high amenity and aesthetic values, although the coefficient is not statistically significant at the conventional

Table 4. Coefficients of the Cross Effects Variables in the Extended Model[a]

	Wetland Type					
Wetland Service	Estuarine Coeff.	Marine Coeff.	Riverine Coeff.	Palustrine Coeff.	Lacustrine Coeff.	Human-made Coeff.
Flood control and storm buffering	0.011 (11)	1.326 (11)*	0.666 (23)	−0.486 (15)	−1.225 (8)	2.845 (4)**
Surface and groundwater supply	−1.201 (5)	−0.036 (3)	−0.497 (18)	0.475 (21)	0.405 (20)	0.958 (5)
Water quality improvement	3.128 (5)***	0.305 (5)	0.033 (18)	0.717 (24)	−0.555 (9)	1.716 (9)**
Commercial fishing and hunting	0.410 (47)	0.652 (34)	1.177 (21)**	−2.475 (20)***	0.383 (16)	0.266 (7)
Recreational hunting	−0.294 (15)	−0.374 (14)	−0.509 (36)	−0.833 (45)	−0.426 (24)	−0.355 (1)
Recreational fishing	−0.019 (17)	−0.760 (21)	0.337 (19)	0.045 (31)	0.790 (22)	−3.990 (1)*
Harvesting of natural materials	−0.595 (28)	0.612 (22)	−0.288 (21)	0.218 (16)	−0.234 (14)	−0.565 (5)
Fuel wood	−0.901 (18)	0.136 (9)	0.622 (8)	−3.368 (2)**	−2.126 (4)	no obs.
Nonconsumptive recreation	0.072 (20)	−0.005 (27)	0.792 (28)	0.937 (33)	0.132 (28)	0.525 (12)
Amenity and aesthetics	1.873 (6)	0.158 (6)	−0.193 (17)	−0.177 (17)	−0.257 (15)	−0.109 (6)
Natural habitat and biodiversity	−1.189 (11)	1.831 (12)**	0.434 (21)	0.256 (20)	−0.011 (13)	2.261 (9)**

[a]Note: OLS results. In brackets, the number of observations for each type of cross effect. Only the results for cross-effect variables are presented since the coefficients of study and context variables remain unchanged as compared to the base metaregression model B. Coeff., coefficient; no obs., no observation available; R^2 = 0.55; adjusted R^2 = 0.46; significance is indicated with ***, **, and * for 1%, 5%, and 10% statistical significance levels, respectively.

10% level (p value = 0.128). Human-made and marine wetlands deliver high values for flood control and storm buffering and for the provision of natural habitat and biodiversity. The highest values for commercial fishing and hunting are provided by riverine ecosystems, while the values of palustrine ecosystems for such service are low. Fuel wood extraction is valued the lowest in palustrine ecosystems, but also lacustrine and estuarine wetlands provide low values for such service, although the coefficients for the two cross-effect variables are not statistically significant (p value = 0.107 and 0.135, respectively).

4. Discussion and Conclusion

[36] The present study provides an original contribution in terms of identification of the main determinants of the values of wetland ecosystems both in absolute and relative terms, using meta-analysis. Compared to previous meta-analyses of wetland values, we have substantially extended the number of primary studies on which the meta-analysis builds and have introduced a number of important additional explanatory variables, which we found to be statistically significant in variation in the valuation of wetlands.

[37] The data set developed for the present metaregression analysis is substantially more comprehensive than those gathered for previous meta-analyses and reflects a geographical distribution that is less biased toward temperate climate zone wetlands in the Northern hemisphere. The data set includes 418 observations from 186 wetland sites worldwide, which were derived from 170 primary valuation studies. *Brouwer et al.* [1999] used a sample size of 92 observations from 30 studies. The data set developed by *Woodward and Wui* [2001] included 65 valuations from 39 studies. Both studies limited the investigation to North American and European wetland ecosystems. *Brander et al.* [2006] assembled a data set of 215 value observations obtained from 80 studies. Their analysis adopts a broader geographical scope, including studies from both temperate and tropical regions, but North American sites still account for half of the total number of observations. Despite the improved geographical coverage of the valuation studies included in the present work, the geographical distribution of the valued sites is still skewed toward temperate Northern latitudes and equatorial regions, if compared with the natural occurrence of wetland ecosystems. This calls for new wetland valuation studies to target little studied wetlands such as those located at latitudes higher than 45°N.

[38] Some of the results of previous studies are confirmed by this meta-analysis. The coefficient on water quality improvement indicates high values for this service [*Brouwer et al.*, 1999; *Woodward and Wui*, 2001], while provision of fuel wood and recreational hunting are less valued [*Brander et al.*, 2006; *Woodward and Wui*, 2001]. In addition, in this study we found that nonconsumptive recreational activities and the provision of natural habitat and biodiversity are highly valued. The coefficients of the variables "wetland size" and "marginal" indicate decreasing returns to scale and that marginal values are higher than average values [*Brander et al.*, 2006]. Also, values are sensitive to income effects and increase with the population living in the surrounding of a wetland [*Brander et al.*, 2006]. With respect to previous studies, we chose not to include geographic region variables in the regression since these were found to be correlated with other variables such as GDP per capita and ecosystem services and did not significantly contribute in explaining the model residuals [*Ghermandi et al.*, 2008]. Although the explanatory powers of different meta-analyses are strictly speaking not directly comparable since they are based on different samples and underlying variation in the endogenous variable, for the sake of rough comparison it can be noted that the explanatory power of this metaregression (R^2 = 0.47; adjusted R^2 = 0.44 from model B) is higher than that given by *Brouwer et al.* [1999] (R^2 = 0.38) and slightly lower than given by *Woodward and Wui* [2001] (R^2 = 0.58 from metaregression model C) and *Brander et al.* [2006] (R^2 = 0.55; adjusted R^2 = 0.45).

[39] Some of the limitations of previous meta-analyses also apply to the present study. In standardizing value estimates to a common unit we followed the approach of *Woodward and Wui* [2001] and *Brander et al.* [2006] and included estimates from a range of market and nonmarket valuation techniques. We argued that the lack of statistical significance of the valuation method coefficients in the metaregression provides an empirical support for the comparability of estimates. *Smith and Pattanayak* [2002], however, caution against high heterogeneity in the primary data due to the risk of introducing inconsistencies both in the economic concepts and commodities being evaluated. Such limitation is of particular relevance for value transfer. Our standardization technique controls for size differences among wetlands and we included

a population variable to account for differences in market extension across wetlands. This approach, however, can only partially account for the extent of the populations affected by the provision of the various wetland services and we suggest that future meta-analyses will focus in particular on the issue of distance decay both for use and nonuse values.

[40] A further limitation of this study lies in the treatment of potential selection bias in the data set of wetland values. A selection bias arises for instance when ecosystems that are perceived more valuable *a priori* are more likely to be selected for valuation or when the probability of a study being published is correlated to the effect size measure [*Hoehn*, 2006; *Woodward and Wui*, 2001]. Such biases may have relevant consequences in particular when the results of a meta-analysis are used for value transfer [*Hoehn*, 2006; *Rosenberger and Johnston*, 2009]. Since value transfer is not the focus of the present study, however, we limited our treatment of potential selection biases to compiling the most complete data set as a necessary (but not sufficient) condition to deal with publication selection bias but we acknowledge that other types of selection bias may affect the wetland valuation literature and that this should be the object of further investigation in future studies.

[41] The principal original contributions of this study are in the analysis of substitution effects, value variation with anthropogenic pressure, and valuation of human-made wetlands. In this study we used the presence of all available similar ecosystems within a distance from a valued site as criteria to determine the availability of substitute sites. The assessed negative relationship between wetland abundance and the value of the wetland indicates the presence of substitution effects for at least some of the wetland services. The abundance of wetland ecosystems in a certain region reflects the uniqueness of a wetland environment and may influence people's perceptions and preferences due to the presence of other sites that can act as a substitute for some of the services provided. We suggest that future research should focus more closely on testing for substitution effects in meta-analysis, i.e., on determining how a change in one ecosystem characteristic affects the demand for goods and services provided by another ecosystem.

[42] The coefficients for the environmental pressure variables are all positive and increase with pressure indicating that a high pressure of human activities on the wetland produces high values. Possible explanations for this are that human activities contribute to translate potential uses into values or that human interventions in a wetland often improve the level of provision of specific wetland services, such as water quality improvement in the case of constructed treatment wetlands. Furthermore, wetlands surrounded by densely populated areas and with unrestricted access, thus with high environmental pressure according to the index proposed in this study, are likely to be relatively easily accessible for the enjoyment of their recreational functions. High anthropogenic pressure on a wetland, however, raises questions about the sustainability of values. Unfortunately, this issue cannot be addressed with the snapshots of values inferred from the valuation studies.

[43] Among wetland types, human-made wetlands have the highest values followed by marine wetlands. A possible explanation for the high value of human-made wetlands is that man-made ecosystems are usually constructed with the specific purpose of providing services for human use and thus their value is more easily realized and recognized by the local populations. The analysis of the results of the extended model allows us to identify that flood control, storm buffering, and water quality improvement are highly valued in human-made wetland ecosystems. The first hypothesis formulated in section 1 which states that human-made and natural wetlands provide the same level of values for such services (Hypothesis 1) is thus rejected. In our analysis, human-made wetlands are substitutes of marine and estuarine wetlands, respectively, in terms of the provision of flood control and water quality improvement services. Remarkably, the coefficient of provision of natural habitat and biodiversity in human-made wetlands is positive and highly statistically significant, despite the fact that such service is generally not a primary goal in the creation of such ecosystems. The large size of the coefficient, compared to other cross effects, supports the hypothesis that this ancillary cultural benefit of human-made wetlands represents an important component of their total economic value (Hypothesis 2). This valuation result signals the potential value of these ancillary ecosystem services and thus the importance of taking them into account in the design and evaluation of alternative policy scenarios and cost benefit exercise. We suggest that future research on human-made wetlands should account for the fact that the (re)construction of wetlands and their habitat may have a significant role in terms of local biodiversity enhancement. This suggests a high potential for landscape and waterscape architecture projects to bring substantial welfare gains to communities with limited access to natural ecosystems such as those in urbanized areas.

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References

Bal, F., and P. Nijkamp (2001), In search of valid results in a complex economic environment: The potential of meta-analysis and value transfer, *Eur. J. Oper. Res.*, *128*(2), 364–384, doi:10.1016/S0377-2217(00)00078-3.

Balcombe, C. K., J. T. Anderson, R. H. Fortney, and W. S. Kordek (2005), Wildlife use of mitigation and reference wetlands in West Virginia, *Ecol. Eng.*, *25*(1), 85–99, doi:10.1016/j.ecoleng.2005.03.003.

Baron, M. G., N. Zaitsev, and M. Schechter (1997), *Expected Recreational Benefits of the Hula Project: Economic Analysis. Final Report*, Hula Proj. Auth., Haifa, Israel.

Benyamine, M., M. Bäckström, and P. Sanden (2004), Multi-objective environmental management in constructed wetlands, *Environ. Monit. Assess.*, *90*(1–3), 171–185, doi:10.1023/B:EMAS.0000003577.22824.8e.

Bergstrom, J. C., and L. O. Taylor (2006), Using meta-analysis for benefits transfer: Theory and practice, *Ecol. Econ.*, *60*(2), 351–360, doi:10.1016/j.ecolecon.2006.06.015.

Bijmolt, T. H. A., and R. G. M. Pieters (2001), Meta-analysis in marketing when studies contain multiple measurements, *Mark. Lett.*, *12*(2), 157–169, doi:10.1023/A:1011117103381.

Birol, E., K. Karousakis, and P. Koundouri (2006), Using a choice experiment to account for preference heterogeneity in wetland attributes: The case of Cheimaditida wetland in Greece, *Ecol. Econ.*, *60*(1), 145–156, doi:10.1016/j.ecolecon.2006.06.002.

Brander, L. M., R. J. G. M. Florax, and J. E. Vermaat (2006), The empirics of wetland valuation: A comprehensive summary and a meta-analysis of the literature, *Environ. Resour. Econ.*, *33*(2), 223–250, doi:10.1007/s10640-005-3104-4.

Brooks, R. P., D. H. Wardrop, C. A. Cole, and D. A. Campbell (2005), Are we purveyors of wetland homogeneity? A model of degradation and res-

toration to improve wetland mitigation performance, *Ecol. Eng.*, *24*(4), 331–340, doi:10.1016/j.ecoleng.2004.07.009.
Brouwer, R. (2000), Environmental value transfer: State of the art and future prospects, *Ecol. Econ.*, *32*(1), 137–152, doi:10.1016/S0921-8009(99)00070-1.
Brouwer, R., I. H. Langford, I. J. Bateman, and R. K. Turner (1999), A meta-analysis of wetland contingent valuation studies, *Reg. Environ. Change*, *1*(1), 47–57, doi:10.1007/s101130050007.
Byström, O. (2000), The replacement value of wetlands in Sweden, *Environ. Resour. Econ.*, *16*(4), 347–362, doi:10.1023/A:1008316619355.
Campbell, D. A., C. A. Cole, and R. P. Brooks (2002), A comparison of created and natural wetlands in Pennsylvania, USA, *Wetlands Ecol. Manage.*, *10*(1), 41–49, doi:10.1023/A:1014335618914.
Confer, S. R., and W. A. Niering (1992), Comparison of created and natural freshwater emergent wetlands in Connecticut (USA), *Wetlands Ecol. Manage.*, *2*(3), 143–156, doi:10.1007/BF00215321.
Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe (1979), *Classification of Wetlands and Deepwater Habitats of the U.S.*, U.S. Dept. of the Interior, Fish and Wildlife Serv., Washington, D. C.
Cravener, M. (1995), Samhällsekonomisk värdering av den anlagda våtmarken i Oxelösund, en tillämpning av Contingent Valuation metoden, Master thesis, Dept. of Econ., Stockholm Univ., Stockholm, Sweden.
de Groot, R. S., M. A. M. Stuip, C. M. Finlayson, and N. Davidson (2006), *Valuing Wetlands: Guidance for Valuing the Benefits Derived From Wetland Ecosystem Services. Ramsar Tech. Rep./CBD Tech. Ser. 3/27*, Gland, Switzerland.
Farber, S. C., R. Costanza, and M. A. Wilson (2002), Economic and ecological concepts for valuing ecosystem services, *Ecol. Econ.*, *41*(3), 375–392, doi:10.1016/S0921-8009(02)00088-5.
Fennessy, M. S., A. D. Jacobs, and M. E. Kentula (2004), *Review of Rapid Methods for Assessing Wetland Condition*, *Rep. EPA/620/R-04/009*, U.S. Environ. Prot. Agency, Washington, D. C.
Ghermandi, A., J. C. J. M. van den Bergh, L. M. Brander, H. L. F. de Groot, and P. A. L. D. Nunes (2008), The economic value of wetland conservation and creation: A meta-analysis, *Fondazione Eni Enrico Mattei Working Pap.*, *238*, 20 pp.
Hammer, D. A., and R. K. Bastian (1989), Wetlands ecosystems: Natural water purifiers, in *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural*, edited by D. A. Hammer, pp. 6–20, Lewis Publ., Chelsea, Mich.
Hein, L., K. Van Koppen, R. S. de Groot, and E. Van Ierland (2006), Spatial scales, stakeholders and the valuation of ecosystem services, *Ecol. Econ.*, *57*(2), 209–228, doi:10.1016/j.ecolecon.2005.04.005.
Heston, A., R. Summers, and B. Aten (2006), *Penn World Table*, version 6.2, Cent. for Int. Comparisons of Prod., Income and Prices at the Univ. of Penn., Philadelphia.
Hoang Tri, N., W. N. Adger, and P. M. Kelly (1998), Natural resource management in mitigating climate impacts: the example of mangrove restoration in Vietnam, *Global Environ. Change*, *8*(1), 49–61, doi:10.1016/S0959-3780(97)00023-X.
Hoehn, J. P. (2006), Methods to address selection effects in the meta regression and transfer of ecosystem values, *Ecol. Econ.*, *60*(2), 389–398, doi:10.1016/j.ecolecon.2006.05.021.
Johnston, R. J., E. Y. Besedin, and R. F. Wardwell (2003), Modeling relationships between use and nonuse values for surface water quality: A meta-analysis, *Water Resour. Res.*, *39*(12), 1363, doi:10.1029/2003WR002649.
Kadlec, R. H., and R. L. Knight (1996), *Treatment Wetlands: Theory and Implementation*, Lewis Publ., Boca Raton, Fla.
Kirkland, W. T. (1988), Preserving the Whangamarino Wetland: An application of the contingent valuation method, Master thesis, Agric. Sci., Massey Univ., Dumsday, New Zealand.
Klein, R. J. T., and I. J. Bateman (1998), The recreational value of Cley marshes nature reserve: An argument against managed retreat?, *J. Chart. Inst. Water Environ. Manage.*, *12*(4), 280–285, doi:10.1111/j.1747-6593.1998.tb00186.x.
Knight, R. L. (1997), Wildlife habitat and public use benefits of treatment wetlands, *Water Sci. Technol.*, *35*(5), 35–43, doi:10.1016/S0273-1223(97)00050-4.
Knight, R. L., R. A. Clarke Jr., and R. K. Bastian (2001), Surface flow(SF) treatment wetlands as a habitat for wildlife and humans, *Water Sci. Technol.*, *44*(11), 27–37.
Lehner, B., and P. Döll (2004), Development and validation of a global database of lakes, reservoirs and wetlands, *J. Hydrol.*, *296*(1-4), 1–22, doi:10.1016/j.jhydrol.2004.03.028.
Loth, P. E. (Ed.) (2004), *The Return of the Water: Restoring the Waza Logone Floodplain in Cameroon*, World Conserv. Union, IUCN, Gland, Switzerland.
MacDonald, H. F., J. C. Bergstrom, and J. E. Houston (1998), A proposed methodology for measuring incremental environmental benefits from using constructed wetlands to control agricultural non-point-source pollution, *J. Environ. Manage.*, *54*(4), 259–267, doi:10.1006/jema.1998.0240.
Matt, G. E., and T. D. Cook (1994), Threats to the validity of research syntheses, in *The Handbook of Research Synthesis*, edited by H. Cooper and L. V. Hedges, pp. 503–520, Russell Sage Found., New York.
Meyerhoff, J., and A. Dehnhardt (2007), The European Water Framework Directive and economic valuation of wetlands: The restoration of floodplains along the River Elbe, *Eur. Environ.*, *17*(1), 18–36, doi:10.1002/eet.439.
Nelson, J. P., and P. E. Kennedy (2009), The use (and abuse) of meta-analysis in environmental and natural resource economics: An assessment, *Environ. Resour. Econ.*, *42*(3), 345–377, doi:10.1007/s10640-008-9253-5.
Nunes, P. A. L. D., and J. C. J. M. van den Bergh (2001), Economic valuation of biodiversity: Sense or nonsense? *Ecol. Econ.*, *39*(2), 203–222, doi:10.1016/S0921-8009(01)00233-6.
Nunes, P. A. L. D., E. Ojea, and M. L. Loureiro (2009), Mapping of forest biodiversity values: A plural perspective. *Fondazione Eni Enrico Mattei Working Pap.*, *264*, 28 pp.
Oglethorpe, D. R., and D. Miliadou (2000), Economic valuation of the non-use attributes of a wetland: A case-study for Lake Kerkini, *J. Environ. Plann. Manage.*, *43*(6), 755–767, doi:10.1080/09640560020001665.
Posford Duvivier Environment (1999), River Ancholme flood storage area progression, *Rep. E3475/01/001*, Environ. Agency, Peterborough, U. K.
Posford Duvivier Environment (2000), River Nar improvement scheme, Engineers report, Environ. Agency, U. K.
Ragkos, A., A. Psychoudakis, A. Christofi, and A. Theodoridis (2006), Using a functional approach to wetland valuation: the case of Zazari–Cheimaditida, *Reg. Environ. Change*, *6*(4), 193–200, doi:10.1007/s10113-006-0019-8.
Rosenberger, R. S., and R. J. Johnston (2009), Selection effects in meta-analysis and benefit transfer: avoiding unintended consequences, *Land Econ.*, *85*(3), 410–428.
Rosenberger, R. S., and J. B. Loomis (2000a), Panel stratification in meta-analysis of economic studies: An investigation of its effects in the recreation valuation literature, *J. Agric. Appl. Econ.*, *32*(3), 459–470.
Rosenberger, R. S., and J. B. Loomis (2000b), Using meta-analysis for benefit transfer: In-sample convergent validity tests of an outdoor recreation database, *Water Resour. Res.*, *36*(4), 1097–1107, doi:10.1029/2000WR900006.
Scheierling, S. M., J. B. Loomis, and R. A. Young (2006), Irrigation water demand: A meta-analysis of price elasticities, *Water Resour. Res.*, *42*, W01411, doi:10.1029/2005WR004009.
Scherrer, S. (2003), *Evaluation économique des aménités récréatives d'une zone humide intérieure: Le cas du lac du Der*, *IFOP Rep. 03-E05*, Min. de l'Ecol. et du Dév. Durable, Paris.
Scott, D. A., and T. A. Jones (1995), Classification and inventory of wetlands: A global overview, *Plant Ecol.*, *118*(1–2), 3–16, doi:10.1007/BF00045186.
Seguì-Amórtegui, L. A. (2004), Sistemas de regeneración y reutilización de aguas residuales. Metodología para el análisis técnico-económico y casos, Doctoral thesis, Depart. de Ing. Agroal. y Biotecnol., Univ. Politéc. de Cataluña, Barcelona, Spain.
Smith, V. K., and S. K. Pattanayak (2002), Is meta-analysis a Noah's ark for non-market valuation?, *Environ. Resour. Econ.*, *22*(1–2), 271–296, doi:10.1023/A:1015567316109.
Stanley, T. D. (2001), Wheat from chaff: Meta-analysis as quantitative literature review, *J. Econ. Perspect.*, *15*(3), 131–150, doi:10.1257/jep.15.3.131.
Turner, R. K. (1991), Economics and wetland management, *Ambio*, *20*(2), 59–63.
Verma, M. (2001), Economic valuation of Bhoj wetlands for sustainable use, report for World Bank assistance to government of India, Indian Inst. of For. Manage., Bhopal, India.
Vidanage, S., S. Perera, and M. Kallesoe (2004), *Kala Oya River Basin, Sri Lanka: Integrating Wetland Economic Values Into River Basin Management*, Environ. Econ. Programme, IUCN Sri Lanka Country Off., Colombo.
Woodward, R. T., and Y. S. Wui (2001), The economic value of wetland services: A meta-analysis, *Ecol. Econ.*, *37*(2), 257–270, doi:10.1016/S0921-8009(00)00276-7.

World Bank (2006), *World Development Indicators*, Int. Bank for Reconstruct. and Dev., World Bank, Washington, D. C. (Available at http://devdata.worldbank.org/wdi2006/contents/index2.htm)

Yang, W., J. Chang, B. Xu, C. Peng, and Y. Ge (2008), Ecosystem service value assessment for constructed wetlands: A case study in Hangzhou, China, *Ecol. Econ.*, *68*(1–2), 116–125, doi:10.1016/j.ecolecon.2008.02.008.

L. M. Brander, Institute for Environmental Studies, VU University Amsterdam, De Boelelaan 1087, NL-1081 HV Amsterdam, Netherlands.

H. L. F. de Groot, Department of Spatial Economics, VU University Amsterdam, De Boelelaan 1105, NL-1081 HV Amsterdam, Netherlands.

A. Ghermandi and P. A. L. D. Nunes, Fondazione Eni Enrico Mattei and Department of Economics, Cà Foscari University of Venice, Isola di San Giorgio Maggiore, I-30124, Venice, Italy. (andrea.ghermandi@feem.it)

J. C. J. M. van den Bergh, Department of Economics and Economic History and Institute for Environmental Science and Technology, Universitat Autònoma de Barcelona, Edifici Cn-Campus UAB, E-08193 Bellaterra (Cerdanyola), Spain.