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Key Points:
- The net ecosystem services value can reveal the real benefits of an ecosystem
- The net value is more practical than the payment for ecosystem services
- The net ecosystem services value provides decision-making for ecological conservation and environmental protection

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Net Value of Wetland Ecosystem Services in China

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Abstract Wetlands generate a wide range of ecosystem services that support human well-being and socioeconomic development. However, calculation of the value of these services generally fails to account for their costs. In order to understand the difference between the value and the net value of ecosystem services (VES and NES, respectively), we used government statistics and data from published papers about the values and costs of wetland services to calculate their NES in mainland China. After accounting for the opportunity costs, investment in wetland conservation and protection, and management to prevent natural disasters, the NES of China's wetlands totaled 828.1×10^9 RMB in 2014, which is 36.4% less than the VES of $1,301.9 \times 10^9$ RMB calculated using the traditional approach. From 1952 to 2014, the NES of wetlands across the country (adjusted for inflation) decreased from $2,215.9 \times 10^9$ RMB to 828.1×10^9 RMB. Accounting for the costs to determine the net value of ecosystem services will provide a better foundation to support land planning and utilization.

Plain Language Summary Calculating the value of ecosystem services is important to support planning and land utilization. However, the traditional valuation approach fails to account for the costs of these services, and the resulting overestimated values may lead to unwise decisions. In this paper, we propose a model to estimate the costs of wetland ecosystem services, which include the opportunity costs of water and land, investments in wetland conservation and protection, as well as management to prevent natural disasters, thereby providing a better (more holistic) estimate based on the net value. The net value provides stronger support for decision-making to support ecological conservation and environmental protection and will provide guidance for future land management.

1. Introduction

Wetlands are special and important ecosystems that form a bridge between land and water (Finlayson et al., 1999; Russi et al., 2013). Wetlands include a wide range of ecosystems, including marshes, bogs, fens, and swamps, but all are defined by their wet hydrology, the vegetation that has adapted to that hydrology, and the soils that evolve under the influence of that hydrology and vegetation (Schlesinger & Bernhardt, 2013). As a result of interactions among these factors, wetlands form distinctive landscapes that include floodplains, deltas, peatlands, estuaries, and coastal margins, each of which supports distinctive wetland types such as ox-bow lakes, wet woodlands, wet grasslands, and headwater depressions (Maltby & Acreman, 2011).

Wetlands occupy sites that are often valuable for human needs such as agriculture and urban expansion, and the conflicts between ecosystem and human needs have led to the disappearance of 50% of the world's original wetland area; these losses range from relatively minor in areas such as the boreal zone, where humans are not common, to extreme, with losses >90%, in heavily populated temperate or tropical countries (Clarkson et al., 2013; Ghermandi et al., 2010) Although the area (with estimates ranging from 5.3×10^6 to 12.8×10^6 km^2) is smaller than that of terrestrial ecosystems such as forests and grasslands, wetlands can have the most valuable resources per unit area (Costanza et al., 2014; Schlesinger & Bernhardt, 2013; Xie et al., 2015). This is because wetlands provide a wide range of high-value ecosystem services, such as protecting coastal regions from storms, carbon sequestration, nutrient transformation, and the provision of nursery habitats that support commercial fisheries and breeding habitats that support biodiversity (Clarkson et al., 2013; Kirwan & Megonigal, 2013; Rao et al., 2015).

Although calculating the value of ecosystem services reminds us of their importance, this value alone provides an incomplete basis for justifying ecosystem restoration or protection because it neglects the costs of these activities (Wegner & Pascual, 2011; Zhang et al., 2017). These costs are important because they inevitably involve trade-offs between meeting ecological and human needs, leading to opportunity costs that can reduce the net value of a wetland ecosystem by altering the flow of benefits it provides to humans (Grossmann, 2012). Cost-benefit analysis has been a traditional method of quantifying these tradeoffs but has not been widely applied to wetland management, leading to neglect of the costs of policies and projects that would otherwise seem to be highly beneficial but turn out to have lower economic efficiency than expected (Wegner & Pascual, 2011).

However, few studies have evaluated costs so that researchers can determine the net value of wetland ecosystem services. For example, Chen et al. (2009) estimated the net ecosystem services value of Beijing's wetlands by accounting for the construction costs, management costs, and virtual costs. In addition, Peh et al. (2014) estimated the net value of wetland ecosystem services in the UK using the TESSA model, but only accounted for the initial restoration costs and subsequent management costs. Although these studies adopted a cost-benefit perspective, they did not include all or even most costs; moreover, they did not account for variation in costs and benefits among the different regions of a country, leaving a gap in our knowledge of such variation. Furthermore, by focusing on direct costs such as management costs, they neglected indirect costs such as the relative values of land and water resources, the opportunity costs that result from using them for ecological rather than human needs, and the risk cost incurred to protect wetlands, thereby reducing the accuracy of their analyses of the net value (Naidoo et al., 2006; Zhang et al., 2017).

The value of China's wetlands is particularly poorly understood. During China's recent unprecedented economic development, the total area of its wetlands decreased from 568.7×10^3 km^2 in 1952 to 212.5×10^3 km^2 in 2014 (SFA, 1953–2014). Although wetlands in many areas remain threatened, China's membership in the 1992 Ramsar Wetlands Convention has led the central government to invest more than 0.8×10^9RMB annually to protect and restore wetlands (Yang et al., 2011). Recent estimates suggest that the annual VES per unit area of China's wetlands amounted to 61.3×10^3 RMB ha^{-1} yr^{-1} (Xie et al., 2015; Zhang et al., 2013), which is 12.68, 1.05, and 1.38 times the VES for China's grassland, farmland, and forests ecosystem (Xie et al., 2015). Because VES is typically estimated differently for different ecosystems, these ratios should be considered approximate and only a general indication of the relative magnitudes of the VES values. However, incomplete understanding of the costs and benefits of these wetlands has led to potentially inappropriate wetland conservation and construction projects in some areas, especially in the case of constructed wetlands that did not consider the true (net) value of their natural resources (Yang et al., 2011). Given the vast sums of money being invested in these ecosystems, it is necessary to improve the resulting efficiency of resource utilization and its role in the sustainable development of these resources. To accomplish this goal, it is necessary to more comprehensively evaluate both the benefits and the costs so that the net service value of wetlands can be determined.

In the present study, our aim was to calculate the net value of ecosystem services (NES) for China's wetlands, as well as the differences in NES among the different regions of this huge country. We calculate the direct costs, opportunity costs, and risk costs so as to allow a more holistic calculation of the NES of this vital ecosystem. Specifically, we used the changes in the area and NES of China's wetlands since 1952 to explore the significance of the wetland NES as well as the effectiveness of wetland protection and conservation. The results have important political and methodological implications for future decision-making and land-use planning, as well as for assessment of the value of natural resources elsewhere in the world, since NES provides a more holistic assessment of the benefits and costs of investments in ecological restoration, conservation, or even construction. Our results will also guide future research in other areas, such as payment for ecosystem services and determining the cost-effectiveness of ecological conservation and environmental protection projects.

2. Methods

To give our research results wide international significance and allow comparisons with other studies, we used the definition of wetlands in the Ramsar Convention: "areas 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 metres" (Maltby & Acreman, 2011). To calculate the costs and NES of China's wetlands, we obtained data from

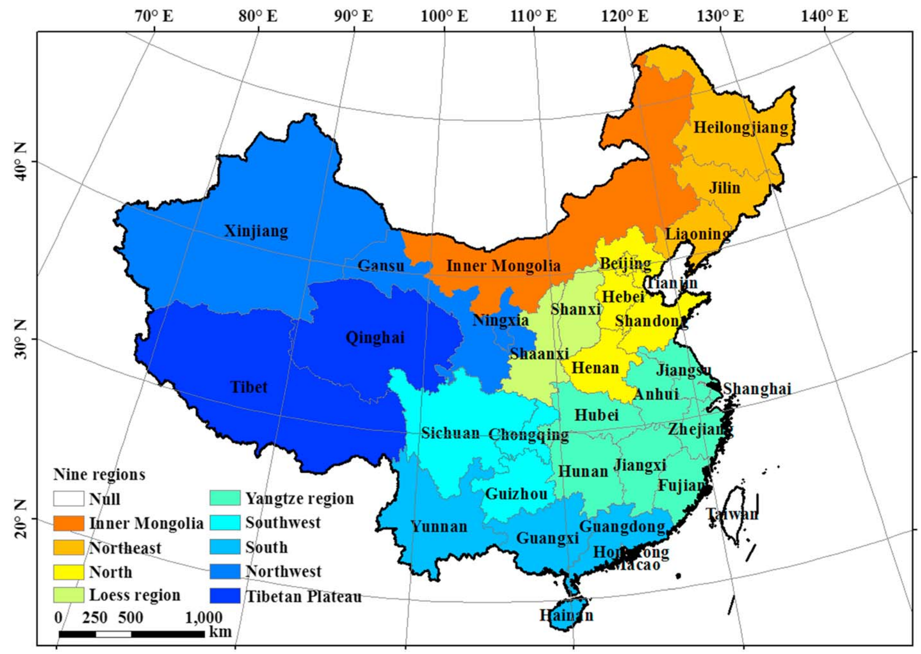

Figure 1. The nine regions of China and the provinces or autonomous regions that belong to them that were used to calculate the values of wetland ecosystem services in mainland China.

published papers on ecosystem services (Xie et al., 2015; Zhang et al., 2013), in which the authors evaluated the value of the following wetland ecosystem services: food production, raw materials, gas regulation, disturbance regulation, water regulation, water supply, waste treatment, provision of habitats or refuges, and recreational and cultural uses. In addition, we obtained national statistics on China's investments in wetland conservation since 1952 (NBS, 1953–2015) and data on the area of wetlands from government statistical data (SFA, 1953–2015). For a given cost parameter we used the same source, with data compiled at a provincial level by the same researchers or government agency; thus, the values for a given parameter should be directly comparable.

To analyze the different ecological characteristics of China's regions, we divided China into nine parts according to the climate conditions (including evaporation and precipitation) and location of wetlands (Figure 1). Inner Mongolia was treated as a separate region in the central northern part of China. The northeast region includes Liaoning, Jilin, and Heilongjiang provinces. The north region includes Beijing, Tianjin, Hebei, Shandong, and Henan provinces. The Loess region includes Shanxi and Shaanxi provinces. The Middle and Lower Yangtze regions include Shanghai, Jiangsu, Anhui, Hubei, Hunan, Zhejiang, Fujian, and Jiangxi provinces. The southwest region includes Chongqing, Sichuan, and Guizhou provinces. The south region includes Hainan, Guangdong, Guangxi, and Yunnan provinces. The northwest region includes Ningxia, Xinjiang, and Gansu provinces. The Tibetan Plateau includes Qinghai and Tibet provinces.

NES is defined as the net value (including income) after accounting for any costs:

$$\mathrm{NES} = \mathrm{VES} - C \tag{1}$$

C includes the direct costs of the investments in wetland conservation (C_d), the opportunity costs of utilizing the wetland's resources (C_o), and risk costs (C_r) such as the risk of failing to provide adequate ecosystem services or to protect against natural disasters:

$$C = C_\mathrm{d} + C_\mathrm{o} + C_\mathrm{r} \tag{2}$$

Data on investments in ecological restoration were obtained from government statistical reports, with values adjusted to 2014 values using published data on inflation rates (NBS, 1953–2015). To simplify the calculation, we defined the opportunity cost of wetland conservation projects (C_o) as the opportunity cost that results from using the land (C_l) and water (C_w) for other purposes:

$$C_o = C_l + C_w \tag{3}$$

Here the opportunity cost represents the value that is not obtained when the water and land are used to sustain wetlands rather than for other purposes such as agriculture. However, this is obviously a coarse estimation; among other things, there are many other costs that could be included in future analyses when reliable data become available. We hope that future researchers will refine our analysis using finer-grained data and the values for different combinations of land uses.

To determine C_l, we used the average land rent for wetlands in every province from a website that provides prices for land transactions in China (http://www.tdzyw.com). In this analysis, we do not refer to the value of the terrestrial components of a wetland ecosystem, but rather to the value of the area occupied by the wetland if the wetland were drained and converted to a different use such as agriculture or residential construction.

Because many of the areas where wetland conservation or restoration have been conducted have an arid to semiarid climate, water is a precious resource in these areas. Therefore, to calculate the C_w, we used the potential evaporation to represent water consumption by wetlands; these data were obtained from published papers (Chen et al., 2014; Yang et al., 2003). We defined the cost of water as an opportunity cost because our goal in this study is to illustrate the impact of the tradeoffs between using water to sustain wetlands and using it for other (human) purposes. In this context, the source of the water (e.g., precipitation vs. municipal) is not important. We chose water costs to humans as the basis for this cost because the alternative (nonwetland) uses of the water are human uses. Because researchers vary widely in their estimation of the value of water, we simplified our analysis by linearly interpolating between the two data points that represented the lowest and highest water prices in China. This is based on the assumption that prices increase linearly with increasing scarcity of a resource, and therefore, the water price (RMB m^{-3}) should increase linearly with decreasing supply (i.e., increasing resource scarcity) in a given region. It would obviously be better to use actual water costs for each province in future analyses, but reliable data for all provinces are not currently available. It would also be better to confirm whether the relationship is linear or follows a different mathematical function, but that is beyond the scope of the present research. To estimate the water price, we used the following equation:

$$V_{it} = b – aP_{it} \tag{4}$$

where V_{it} is the value of the resource (here, water), P_{it} is the resource endowment per person in province i in year t, and b and a are curve-fitting parameters. To estimate these coefficients, we assumed that the water price in Beijing (the part of China with the lowest per capita water availability and thus the highest price) was 1.2 RMB m^{-3} in 2014 based on data from the South-to-North Water Diversion Project (Liu & Yang, 2012) and assumed that the water price in Tibet, which has the highest per capita water availability in China, was 0.17 RMB m^{-3}, which is the lowest price in China(Wang et al.,2013).

We also defined a risk cost (C_r) that must be paid for wetland management to compensate for the losses caused by natural disasters, such as floods or droughts. When natural disasters such as floods and droughts strike, wetlands regulate the water level; the value of this service is included in the value of ecosystem services for wetlands. The risk cost of wetlands then represents the management cost required to repair the wetlands after such disasters and to compensate for the loss of this water regulation service. We based our estimates of C_r on government statistical data on monetary losses caused by floods in China (NBS, 1953–2015).

3. Results

Table 1 summarizes the total NES calculations. The NES of China's wetlands totaled 828.13×10^9 RMB yr^{-1} in 2014, which is 36.4% less than the VES of 1301.9×10^9 RMB calculated using the traditional approach. The huge difference between the NES and the VES lies in the total cost of 473.8×10^9 RMB, which includes a

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Table 1
The Total Annual Values of Wetland Ecosystem Services (VES), Costs, and the Net Value of Wetland Ecosystem Services (NES = VES − Costs) for China as a Whole and for China's Nine Regions in 2014

Regions	Area (km^2)	VES ($\times 10^9$ RMB·yr^{-1})	Costs($\times 10^9$ RMB yr^{-1}) Water consumption (C_w)	Land rent (C_l)	Investments in ecological conservation (C_d)	Management to prevent natural disasters (C_r)	NES ($\times 10^9$ RMB·yr^{-1})
China	212,530.40	1,301.86	311.61	27.66	21.07	113.39	828.13
Northeast	29,980.29	139.57	43.57	4.42	2.54	19.90	69.14
Inner Mongolia	23,912.78	91.85	88.72	3.18	0.41	6.55	−7.01
North	14,525.27	81.21	28.95	2.19	0.86	11.77	37.44
Loess region	1,831.67	9.56	3.65	0.27	0.06	2.88	2.70
Yangtze area	36,683.16	253.12	58.14	5.80	2.12	30.56	156.50
Southwest	8,612.12	104.93	4.10	1.29	0.76	16.99	81.79
South	13,491.67	177.55	23.38	2.02	2.07	20.94	129.14
Northwest	23,271.06	140.73	51.16	3.02	4.89	3.40	78.26
Tibetan Plateau	58,373.99	303.31	9.93	5.46	7.36	0.40	280.16

Notes. The valuation of ecosystem services included all types of land production and the goods and services from ecological conservation; see section 2 for details. The costs of management include the cost of wetland conservation. Ecosystem service values were obtained from published references (Xie et al., 2015; Zhang et al., 2013). The potential evaporation values were also obtained from the literature (Chen et al., 2014; Yang et al., 2003). Calculations were based on the values per unit area in Table 2.

direct cost of 21.1×10^9 RMB, opportunity costs of 27.7×10^9 RMB and 311.6×10^9 RMB for using the land and for water consumption, respectively, and risk costs of 113.4×10^9 RMB.

The three regions with the highest NES were the Tibetan Plateau, Yangtze, and South regions, at 280.16×10^9, 156.50×10^9, and 129.14×10^9 RMB yr^{-1}, respectively. The two regions with the lowest NES were the Loess region and Inner Mongolia, at 2.70×10^9 and -7.01×10^9 RMB yr^{-1}, respectively. The area with the highest C_w was Inner Mongolia, at 88.72×10^9 RMB yr^{-1}, followed by the Yangtze, northwest, and northeast regions, at 58.14×10^9, 51.16×10^9, and 43.57×10^9 RMB yr^{-1}, respectively. The highest C_l was in the Yangtze Region, at 5.80×10^9 RMB yr^{-1}, followed by the Tibetan Plateau and northeast, at 5.46×10^9 and 4.42×10^9 RMB yr^{-1}, respectively. The highest C_d was in the Tibetan Plateau, at 7.36×10^9 RMB yr^{-1}, followed by northwestern China, at 4.89×10^9 RMB yr^{-1}. The Loess region had the lowest C_w, C_l, and C_d, at 3.65×10^9, 0.27×10^9, and 0.06×10^9 RMB yr^{-1}, respectively. The highest C_r was in the Yangtze region, at 30.56×10^9 RMB yr^{-1}, followed by the south, northeast, and southwest regions, at 20.94×10^9, 19.90×10^9, and 16.99×10^9 RMB yr^{-1}, respectively.

Table 2 summarizes the annual value and costs per unit area in 2014. The annual cost per unit area for wetlands averaged 22.29×10^3 RMB ha^{-1} yr^{-1}, which comprises a direct cost of 0.99×10^3 RMB ha^{-1} yr^{-1} for

Table 2
The Annual Values of Ecosystem Services (VES), Costs, and Resulting Net Ecosystem Services (NES = VES − Costs) Per Unit Area for China's Nine Regions in 2014

Regions	VES ($\times 10^3$ RMB ha^{-1} yr^{-1})	Costs($\times 10^3$ RMB·ha^{-1}·yr^{-1}) Water consumption (C_w)	Land rent (C_l)	Investments in ecological conservation (C_d)	Management to prevent natural disasters (C_r)	NES ($\times 10^3$ RMB ha^{-1} yr^{-1})
Average for China	61.26	14.66	1.30	0.99	5.34	38.97
Northeast	46.55	14.53	1.47	0.85	6.64	23.06
Inner Mongolia	38.41	37.10	1.33	0.17	2.74	−2.93
North	55.91	19.93	1.51	0.59	8.10	25.78
Loess region	52.19	19.93	1.47	0.33	15.72	14.74
Yangtze area	69.00	15.85	1.58	0.58	8.33	42.66
Southwest	121.84	4.76	1.50	0.88	19.73	94.97
South	131.60	17.33	1.50	1.53	15.52	95.72
Northwest	60.47	21.98	1.30	2.10	1.46	33.63
Tibetan Plateau	51.96	1.70	0.94	1.26	0.07	47.99

wetland conservation, opportunity costs of 1.30 × 10^3 RMB ha^{-1} yr^{-1} for land rent for agricultural production and 14.66 × 10^3 RMB ha^{-1} yr^{-1} for water consumption, and risk costs of 5.34 × 10^3 RMB ha^{-1} yr^{-1} to compensate for losses due to natural disasters. Given all of these costs, the NES of China's wetlands totaled 38.97 × 10^3 RMB ha^{-1} yr^{-1}, which is only 63.6% of the total VES of 61.26 × 10^3 RMB ha^{-1} yr^{-1} (Table 2). The three regions with the highest ratios of NES to VES were the Tibetan Plateau and southwest and south regions, at 92.4, 77.9, and 72.7%, respectively, followed by the Yangtze, northwest, northeast, and north regions, at 61.8, 55.6, 49.5, and 46.1%, respectively. The regions with the lowest ratio were the Loess region and Inner Mongolia, at 28.2 and −7.6%, respectively.

The two regions with the highest NES per unit area were south and southwest, at 95.72 × 10^3 and 94.97 × 10^3 RMB ha^{-1} yr^{-1}, respectively, followed by the Tibetan Plateau and the Yangtze Region, at 47.99 × 10^3 and 42.66 × 10^3 RMB ha^{-1} yr^{-1}, respectively. The lowest NES per unit area was Inner Mongolia, at −2.93 × 10^3 RMB ha^{-1} yr^{-1}. The four regions with the highest water costs per unit area were the Inner Mongolia, northwest, north, and Loess regions, at 37.10 × 10^3, 21.98 × 10^3, 19.93 × 10^3, and 19.93 × 10^3, respectively. The Tibetan Plateau and southwest regions had the two lowest values of C_w, at 1.70 × 10^3 and 4.76 × 10^3 RMB ha^{-1} yr^{-1}, respectively. There was little difference in C_l between the regions, with the highest values in the Yangtze and north regions, at 1.58 × 10^3 and 1.51 × 10^3 RMB ha^{-1} yr^{-1}, respectively, and the lowest in the Tibetan Plateau, at 0.94 × 10^9 RMB ha^{-1} yr^{-1}. The three regions with the highest C_d were the northwest, south, and Tibetan Plateau regions, at 2.10 × 10^3, 1.53 × 10^3, and 1.26 × 10^3 RMB ha^{-1} yr^{-1}, respectively. The Inner Mongolia region had the lowest C_d, at 0.17 × 10^9. The three regions with the highest C_r were the southwest, Loess, and south regions, at 19.73 × 10^3, 15.72 × 10^3, and 15.52 × 10^3 RMB ha^{-1} yr^{-1}, respectively. The regions with the lowest C_r were the Inner Mongolia, northwest, and Tibetan Plateau regions, at 2.74 × 10^3, 1.46 × 10^3, and 0.07 × 10^3 RMB ha^{-1} yr^{-1}, respectively.

We adjusted the historical values to 2014 values to account for the effects of inflation and provide comparable total values of VES, NES, and the costs in each year since 1952 (Figure 2). The NES of wetlands decreased from 2,215.9 × 10^9 RMB in 1952 to 828.1 × 10^9 RMB in 2014, which was 36.4% less than the value in 1952. However, as a result of a national wetland conservation strategy, which was implemented in 2000, the wetland area, VES, and NES had stopped decreasing by around 2008. Except for Inner Mongolia, where NES increased steadily throughout the study period, all regions showed the same recovery after 2008 because of the implementation of the national policy.

4. Discussion

Despite the fact that wetland ecosystems have the highest ecosystem service value per unit area around the world (Costanza et al., 1997, 2014; Xie et al., 2015), there is a high cost associated with providing these ecosystem services, especially when we face different trade-offs among services to meet ecological and human needs (Hansson et al., 2005; Maltby & Acreman, 2011). For example, people have different needs during different periods of economic development, creating differences in the pressure to drain wetlands to provide agricultural or urban land (Ghermandi et al., 2010; Maltby & Acreman, 2011). In addition, there are trade-offs between wetland ecosystem services even when wetlands are retained. For example, Hansson et al. (2005) found that a small, deep wetland is likely to be more efficient than a larger and shallower wetland for phosphorus retention, but less valuable in terms of biodiversity. Our results showed that the total NES of China's wetlands in 2014 was only 63.6% of the total VES (Table 1). In addition, due to differences in resource endowments, there were also large differences in the net value among the nine regions.

Because of the huge evaporative demand in northern China (a semiarid to arid region with a potential evaporation of more than 2,000 mm yr^{-1} vs. mean regional precipitation of 232 mm yr^{-1}), the water opportunity cost is high, especially in Inner Mongolia (Table 2), since the price of this resource is defined by its scarcity (He et al., 2013). Land rent showed the opposite pattern because land was more available in the north due to the lower population density (Table 1). As a result, the land rent increased from northwest to southeast and was highest in China's southeastern coastal areas, which have the highest population density. VES was highest in the south and south-central areas due to the warmer climate and more abundant precipitation, and lowest in the cold and relatively arid north (Xie et al., 2015; Zhang et al., 2013). Given the huge difference between NES and VES and its effect on the outcome of management and land use decisions, it will be necessary to

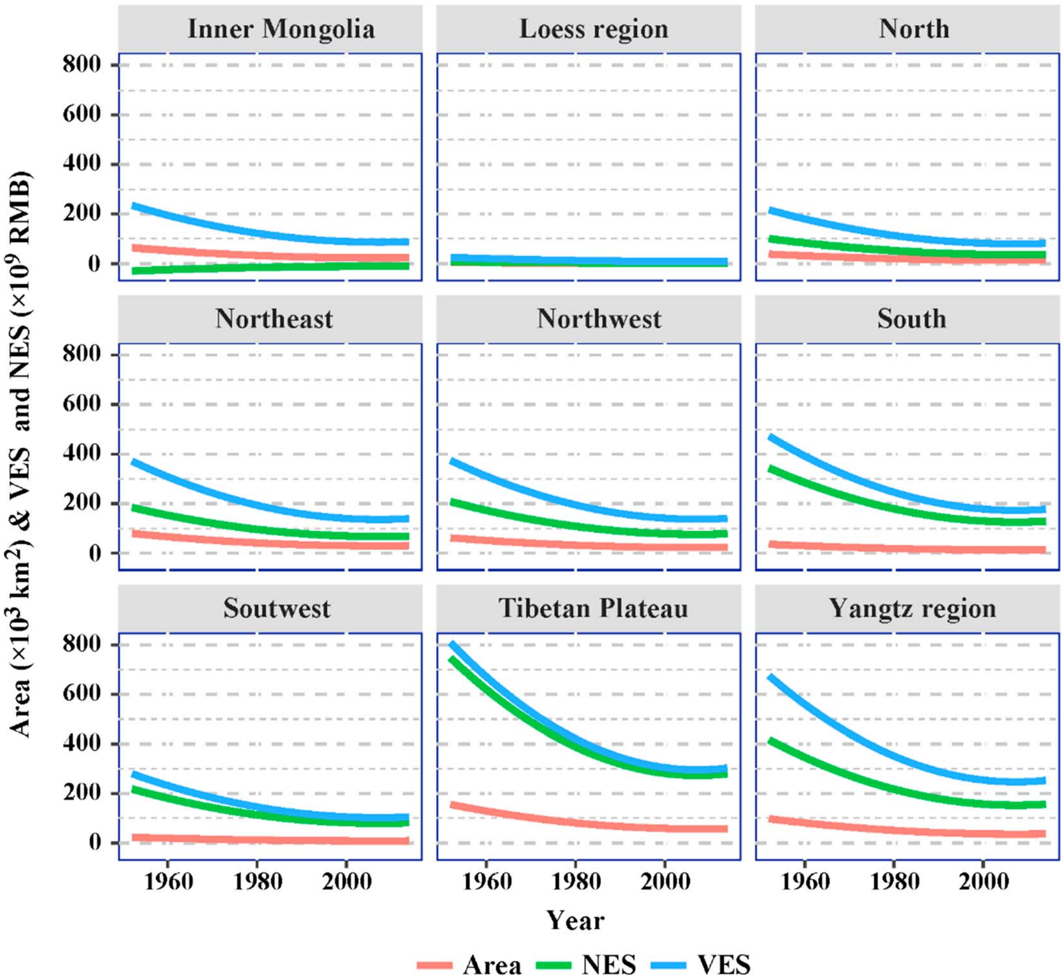

Figure 2. Changes in the wetland area, value of wetland ecosystem services (VES), and net value of wetland ecosystem services (NES = VES minus the costs shown in Table 1) for each of the nine regions of China from 1952 to 2014.

maximize the effectiveness of ecological conservation by basing investment and management decisions upon NES rather than VES. This is the only way to ensure that planning accounts for the natural characteristics of each region.

Our current research results show that the NES for Inner Mongolia was negative. One important reason for this result is a limitation of the method of ecosystem service value evaluation. Because the current ecosystem service value is mainly assessed by means of the market value method and willingness-to-pay method, the results are influenced by differences in the wealth of different populations; the poorest citizens may have the greatest desire to pay for a scarce resource such as water, but the least ability to pay (Costanza et al., 2014; Farley, 2012). The impact of the wealth gap means that cost estimates often fail to reflect the true value of ecosystems or their services, leading to ecosystem service values that are often lower than the true value. In the present study, we also conducted our analysis using Chinese national data based on the market value method. Although this means that we used a consistent approach for all areas, differences in the ability of each region to pay for the ecosystem services means that the NES may not be perfectly comparable between different regions.

Many scholars have emphasized the importance of determining the impact of ecosystem changes on human welfare through cost-benefit analyses of ecological services from the perspective of marginal changes (Farley, 2012; Turner et al., 2010). However, assessing these changes requires a large amount of data that is often unavailable, including the existence and position of thresholds for changes in ecosystem structure and function, and changes in people's needs, all of which can lead to changes in the estimated value of ecosystem services

(Ricketts & Lonsdorf, 2013; Rieb et al., 2017). Although the assessment of marginal changes is of great importance, assessing the static value of ecosystem services at a given time (i.e., a snapshot), in a relatively stable ecosystem, can provide guidance immediately, before an analysis of dynamic changes becomes possible, especially for large areas affected by a wide range of factors that differ among regions (e.g., at a national scale). Nonetheless, future research should try to account for the importance of marginal costs.

The decrease in wetland area caused a proportional decrease in NES during the study period (Figure 2). Therefore, protection of the remaining wetlands will be the key to planning future environmental protection and ecological conservation projects. To sustain the health of China's ecosystems and begin increasing both VES and NES, China's government should pay more attention to wetland conservation. Unfortunately, conservation of any land in China is an intractable problem because of the nation's huge population and the need to fully exploit local resources to sustain this population.

5. Conclusion

The present study provides an original contribution because it improves on previous cost-benefit accounting by providing a more holistic evaluation of the NES of China's wetlands, and by quantifying the variation in this value among the regions of China. Using NES instead of VES provides better support for decision-making because it forces planners to consider the costs of their plans more completely. Research on these costs will have great significance for the protection of wetlands and for land use planning, as it will support the development of more effective and sustainable environmental restoration policies. Because our analysis considered only a small subset of the full range of potential benefits and costs, the actual balance between costs and benefits contains significant uncertainty. An important challenge will be to persuade China's government to record better and more complete data on a wider range of benefits and costs so that future researchers can derive progressively more accurate estimates of NES. In addition, because management plans and national policies are implemented at local levels, finer-resolution data are required to support regional and local planning; the aggregated provincial-scale and regional data that we used in this study are sufficient to demonstrate the importance of accounting for costs but are inadequate to support local-scale policy implementation.

Additional Information

The authors declare no competing financial interests.

Author Contribution statement

S.C. designed the research, J.Z. and W.S. performed most of the data analysis, and S.C and J.Z. wrote the main manuscript text. All authors have reviewed the manuscript and approved it for submission.

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