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Economics of social trade-off: Balancing wastewater treatment cost and ecosystem damage

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ABSTRACT

We have developed a social optimization model that integrates the financial and ecological costs associated with wastewater treatment and ecosystem damage. The social optimal abatement level of water pollution is determined by finding the trade-off between the cost of pollution control and its resulting ecosystem damage. The model is applied to data from the Lake Taihu region in China to demonstrate this trade-off. A wastewater treatment cost function is estimated with a sizable sample from China, and an ecological damage cost function is estimated following an ecosystem service valuation framework. Results show that the wastewater treatment cost function has economies of scale in facility capacity, and diseconomies in pollutant removal efficiency. Results also show that a low value of the ecosystem service will lead to serious ecological damage. One important policy implication is that the assimilative capacity of the lake should be enhanced by forbidding over extraction of water from the lake. It is also suggested that more work should be done to improve the accuracy of the economic valuation.

1. Introduction

Water quality standards are frequently used as the scientific basis for environmental water management policies. Environmental regulations in many countries are based on national quality standards. For example, the Safe Drinking Water Act (enacted in the United States in 1974, and amended in 1986 and 1996) was established to protect public health by regulating the nation's public drinking water supply. The Act also applies national standards set by the United States Environmental Protection Agency (US EPA) to control water sources in rivers, lakes, reservoirs, springs, and groundwater wells (Tiemann, 2010). Similarly, the water quality standards in China are nationally unified, including water quality, pollutant discharge, monitoring methods, and environmental sample standards, which were derived from, or based on, environmental quality standards of developed countries (Wu et al., 2010). This means that current water quality standards may not fit regional environmental conditions and demands. These standards may not fit into the eco-environmental character and economic situation in all regions and, thus, may over- or under-regulate the water quality in some bodies of water. A more location-specific approach that incorporates both the abatement cost and the ecological damage may perform better in meeting the specific social objectives of protecting both human health and ecosystem health. Furthermore, such an approach would provide a policy tool for evaluating the trade-off between ecosystem functions and economic activities.

Aquatic ecosystems (e.g., lake ecosystems) are able to store and absorb waste from human economic activities through dilution, assimilation, and chemical decomposition to a limited extent, acting as “free” water purification plants (De Groot et al., 2002). If the waste amount exceeds the aquatic ecosystem's purification capacity, the ecosystem will be damaged. On one hand, the over-exploitation of the ecosystem capability in attenuating pollution can compromise the long-term functionality of the aquatic ecosystem functionality. On the other hand, not fully using the receiving water system's assimilative capacities creates higher wastewater treatment costs than necessary. Wastewater treatment facilities are now the most commonly used abatement measures to resolve point-source water pollution. Many studies have focused on the analysis of wastewater treatment cost structures (Tsagarakis et al., 2003; Hernandez-Sancho et al., 2011). However, very few

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studies link the ecosystem response behavior with the level of wastewater treatment to allow the estimation of economic trade-off associated with setting optimal water quality levels. Some studies analyze the effects of wastewater discharge on lake ecosystems' functioning (Newcombe and MacDonald, 1991; Camargo and Alonso, 2006; Gücker et al., 2006; Machado and Imberger, 2012). However, the literature considers the issue from an ecological perspective only, with no reference to the economic value of ecosystem or water pollution control costs. Very few studies have managed to combine the pollution abatement cost with the economic value of ecosystems under different states of nature to provide information on the cost-effectiveness of different control policy options (Hein, 2006; Laukkanen and Huhtala, 2008). None of these studies provide information on the optimal water pollution control level, based on control costs and the valuation of ecosystem.

In this paper, we aim to fill this gap in the literature by applying a social optimization model, including wastewater treatment and ecological damage costs, to allow a socially optimal solution for pollutant control levels. Considering both wastewater treatment costs and valuation of ecosystem damage, this paper provides more options for decision-makers to choose from, based on their regional economic and environmental situations, in addition to existing rigid standards and regulations.

The paper proceeds as follows: The social optimization model is developed, and the relationship between key variables in the optimal solution are derived in section 2. Section 3 introduces the case of Lake Taihu in detail. In section 4, the wastewater treatment cost function and the ecological damage cost function are estimated, based on secondary data collected from existing publications. The theoretical model is empirically specified and applied in section 5 to the case of Lake Taihu, providing the empirical results. Section 6 concludes and discusses policy implications.

2. Social optimization model of wastewater treatment and discharge

The model is developed for a regional setup, in which several municipalities treat sewage and discharge it into a lake. The lake is used for recreation, benefitting the citizens of the municipalities. The dilemma of the region is to minimize the social cost of discharging wastewater by deciding on the quality of wastewater to be discharged into the lake. The trade-off is between the cost of treatment to reach high-quality discharged wastewater and the damage to the lake's ecosystem. Both of these are components in the social objective function of the region. There are differences in the level of economic development in various regions of China. People's valuation of ecosystem services also varies among regions, due to the level of economic development as well as environmental situations, local traditions, and institutions. Compared to the alternative option provided in our social optimization model, the cost incurred in meeting current unified water quality criteria does not reflect these local economic, traditional, institutional, and environmental situations.

Several simplifying assumptions were used, which took into consideration population levels, economic activity, as well as water volume and quality in the lake. The relationship between the water pollution level and the damage to the lake ecosystem was modeled using a steady-state approach (Hein, 2006; Bostian et al., 2015). This approach does not fully reflect the dynamic behavior of pollution. However, the objective of this study was to reflect the long-term steady state of the system so that scientific insights can be provided to the water quality regulator. This purpose was fully achieved by using the steady-state framework.

The model also assumed that water treatment was performed in one wastewater treatment facility, while in reality the lake water was used for irrigation and for drinking purposes. Since the interest of this study was in the trade-off between pollution control cost and ecological damage, it was assumed for simplicity and without loss of generality that the only use of the lake water was for recharge of the treated wastewater and for recreation. In this respect, our model is considered partial equilibrium. We also consider the lake as one homogeneous ecological ecosystem rather than a compartmental system. Finally, we assume that the only factor affecting social preferences was the total social cost − either as treatment expenses or as loss of benefits from recreation.

Based on recent literature (Hernandez-Sancho et al., 2011; Fraas and Munley, 1984; Goldar et al., 2001; Friedler and Pisanty, 2006), the wastewater treatment cost model in this paper incorporates both quantity and quality variables of wastewater treatment processes. The wastewater quality variable is the control variable of the social optimization model.

The wastewater treatment cost − both investment cost and operation and maintenance (O&M cost) C is represented by $C = C(Q, F, E)$ expressed in million \$, where Q is the designed capacity of the plant expressed in m^3/day, F is the wastewater flow expressed in m^3/day, and E is the pollutant removal efficiency expressed in percentage. Q is used for investment cost function estimation, and F is used for O&M cost function estimation. E is defined as $(q_{in} - q_{out})/q_{in}$, where q_{in} represents pollutant influent concentration measured in mg/L, and q_{out} represents effluent concentration measured in mg/L. C is twice differentiable with $\partial C/\partial Q \geq 0$; $\partial C/\partial F \geq 0$; $\partial C/\partial E \geq 0$ and $\partial^2 C/\partial Q^2 \leq 0$; $\partial^2 C/\partial F^2 \leq 0$; $\partial^2 C/\partial E^2 \geq 0$. For simplicity, q_{in} and q_{out} are measured with one quality parameter E only.

The other aspect of the social optimization model is ecological damage cost. Several studies analyze a wide class of ecosystems' behaviors under human activities' stress (Holling, 1973; Carpenter and Pace, 1997; Ludwig et al., 1997; Scheffer et al., 2001). Scheffer et al. (2001) identified three main ecosystem response types (see Fig. 1). The first type (a) shows that the state of some ecosystems may respond in a continuous way to increasing stress. The second type (b) shows that the system state remains relatively stable over certain ranges of stress and then responds dramatically when the stress approaches a critical level. The third type, which is totally different (c) is not continuous. The response line is folded backward, which is known as a "catastrophe fold." Ecosystems respond to external stress following a curve that is folded backward, as shown in Fig. 1 (c). If the ecosystem state is on the upper line and close to point "A," small changes in the conditions may lead to a catastrophic switch to the lower line. To switch again to the upper line, the external conditions need to be reversed far enough to reach point "B" (Scheffer et al., 2001; Esteban and Dinar, 2016).

Fig. 1 illustrates the possible relationships between ecosystem state and human-induced stress. As indicated by Scheffer et al. (2000), much of the essence of ecosystem state can often be captured by a single key variable. That is because many aspects of the system's state tend to shift in concert with a few important key variables in a given type of ecosystem. For instance, possible key state variables can be total plant biomass (ecosystem population), or turbidity of the lake. The term "stress" is used to describe the effect of human use. Human use of the ecosystem can be through harvesting or destroying biomass, or stressing the system by affecting its abiotic conditions (Scheffer et al., 2000). The intensity of stress can be reflected by variables such as eutrophication level, groundwater reduction level, or water pollution level.

Keeler et al. (2012) introduced a comprehensive and generalizable framework for linking human-induced stress to values for water quality related ecosystem services. The framework is illustrated in Fig. 2.

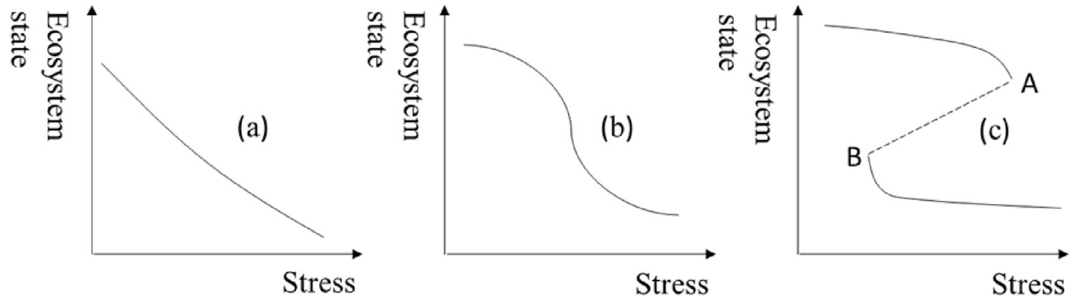

Fig. 1. Three main shift types of ecosystem states with increases of stress. Note: stress increases from left to right; ecosystem state worsens from top to bottom. Source: Adapted with modification from Scheffer et al. (2001).

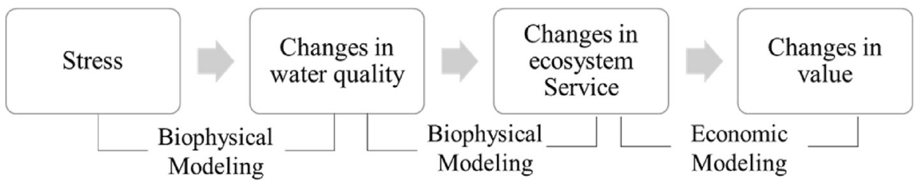

Fig. 2. Framework for linking human-induced stress to ecosystem service values. Note: Illustration is based on the framework proposed by (Keeler et al., 2012).

Linking human-induced ecosystem stress (e.g., water quality impairment) and change of ecosystem state can be achieved through biophysical models. The ecosystem state function is written as $S = S(F, E)$, F and E were defined earlier. The illustration of function $S(F, E)$ can be one of the three types in Fig. 1. From Fig. 1, we can infer that $\partial S/\partial E \geq 0$.

Economic value of ecosystem is usually obtained from valuating the service it can provide by eliciting citizens' willingness to pay for the service (Wilson and Carpenter, 1999; Loomis et al., 2000; Xie et al., 2008; Huang and Ma, 2013). The valuation not only depends on the ecosystem state, but also on socio-demographic factors, such as people's income, education, and so on. The ecosystem damage cost is difficult to measure directly. Instead it can be measured indirectly by using the citizens' utility loss from not being able to use the ecosystem service as a reflection. In other words, the ecosystem damage cost is the difference between economic value of a reference ecosystem state and the current ecosystem state. ξ is used to represent the unit economic value of the ecosystem service expressed in million \$/m. It is typically assumed that people are willing to pay more for one unit of improvement at a lower level of ecosystem state than at the higher level of state. Therefore, ξ may change along with the ecosystem state. S_0 is used to represent the reference ecosystem state. When the ecosystem state is worse than S_0, the ecosystem is damaged. D is used to represent the ecosystem damage cost expressed in million \$, $D(\xi, F, E) = \xi \cdot [S_0 - S(F, E)], \partial D/\partial E \leq 0$.

After translating the ecological damage into monetary terms, it can be well integrated into the social optimization model with the wastewater treatment cost. The social optimization model aims at minimizing the social cost, including both wastewater treatment cost and ecological damage cost. An increase of wastewater treatment cost leads to a decrease of ecological damage cost, and vice versa. The trade-off of reducing cost of either side is illustrated in Fig. 3. The illustration was inspired by the graphic theory in Scheffer et al. (2000), in which they used similar graphs to show how a theoretical society of "enjoyers" and "affectors" may obtain optimal social welfare from the use of an ecosystem. "Enjoyers" are users that benefit from the ecosystem but do not significantly affect the state of the ecosystem. "Affectors" are users that significantly affect the state of the ecosystem. The welfare of "enjoyers" increases with the ecosystem state, whereas the welfare of "affectors" increases with the level of stress imposed on the system by their activity (Scheffer et al., 2000). As indicated in Fig. 3, if there is no restriction, the minimum social cost can be obtained by a combination of the best ecosystem state and the highest level of stress (point A). However, the ecosystem state is a function of stress $S(F, E)$, and the ecosystem response will limit the possible combination of wastewater treatment and ecological damage costs to points on the stable equilibrium lines (e.g., the dash line in Fig. 3).

Mathematically, the social optimization problem is written as:

$$\underset{E}{Min}\, f = C(Q, F, E) + D(\xi, F, E) \quad (1)$$

subject to

$$0 \leq E \leq 1 \quad (2)$$

Write the Lagrange multiplier (Bertsekas, 1999) equation as

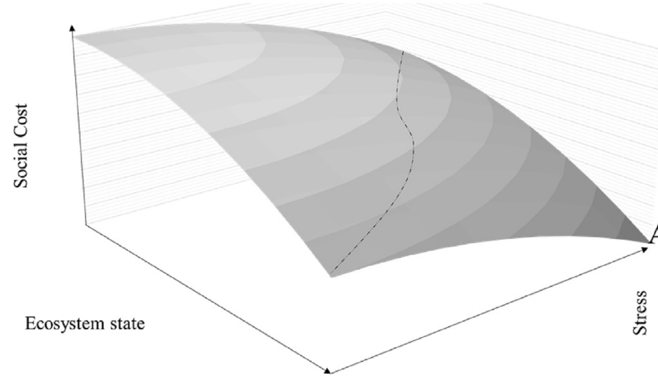

Fig. 3. Graphic illustration of social cost minimization model. Note: The figure is based on and inspired by the theory of Scheffer et al. (2000:458). The dashed line is an example of the equilibrium state, which corresponds to type "b" in Fig. 1. The equilibrium state can also be type "a" or type "c" in Fig. 1, depending on specific context.

$$L = C(Q, F, E) + D(\xi, F, E) - \alpha \cdot (-E) - \beta \cdot (E - 1) \quad (3)$$

The Karush-Kuhn-Tucker conditions (Boyd and Vandenberghe, 2004) are given as below:

$$\partial L/\partial E = \partial C/\partial E + \partial D/\partial E + \alpha - \beta = 0 \quad (4)$$

$$\alpha \cdot (-E) = 0 \quad (5)$$

$$\beta \cdot (E - 1) = 0 \quad (6)$$

$$0 \leq E \leq 1, \alpha \geq 0, \beta \geq 0 \quad (7)$$

Combine conditions (4)–(7), it can be derived that

$$\text{when } E = 1, -\partial D/\partial E \leq \partial C/\partial E \quad (8)$$

$$\text{when } E = 0, -\partial D/\partial E \geq \partial C/\partial E \quad (9)$$

$$\text{when } 0 < E < 1, -\partial D/\partial E = \partial C/\partial E \quad (10)$$

As indicated earlier, $\partial D/\partial E \leq 0, \partial C/\partial E \geq 0$, therefore the results under Karush-Kuhn-Tucker conditions can be interpreted as follows:

The equilibrium is reached when the absolute value of the avoided marginal ecosystem damage cost of decreasing the removal efficiency by one unit $(-\partial D/\partial E)$ is equal to the marginal treatment cost of increasing the removal efficiency by one unit $(\partial C/\partial E)$. When the removal efficiency (E) is at its minimal (0 percent) or maximum (100 percent), the system has a tendency to increase or decrease the removal efficiency to reach the equilibrium, respectively.

The theoretical model will be empirically applied in section 5, to the case of Lake Taihu in China. Prior to embarking on the empirical application, Lake Taihu and its economy will be introduced in the next section.

3. Basic facts on Lake Taihu

Lake Taihu, located in the Yangtze River Delta, is the third largest freshwater lake in China. It is located within the jurisdiction of Suzhou, Wuxi, and Changzhou municipalities in Jiangsu Province, which are among the most industrial and developed regions in China. Water pollution is very serious because of the industrial and agricultural development. Water pollution in Lake Taihu produces eutrophication and causes serious damage to the lake's ecosystem.

The area of Lake Taihu is 2338 km^2. The mean depth of the lake is 1.9 m, and maximum depth is 2.6 m corresponding to an elevation of 3.0 m above sea level. The shallow-water area with mean depth below 1.5 m is about 452 km^2, mostly in East Taihu, accounting for 19.3 percent of the total surface area. The deepest areas (over 2.5 m) are in the north and west, occupying 197 km^2 (8.4 percent of the total lake area) (Qin et al., 2007). The water volume is 44.33×10^8 m^3 (Hu et al., 2006).

Lake Taihu has a rich set of ecosystem services. It is an important source of drinking water supply in the basin area, a valuable tourism resource, and it supports fishery and extraction of water for irrigated agriculture. Furthermore, it is also a repository of waste from the urban, agricultural, and industrial sectors. Xu et al. (2010) estimated the economic value of the ecosystem services of Lake Taihu wetland at \$1.83 billion per year (see Table S1 in the supplementary materials for details). However, due to economic development in past decades, serious water pollution from industry, agricultural, and urban sectors has caused degradation of the lake's ecosystem and deterioration of its water quality and service. For example, algal bloom has degraded the economic potential of the region and damaged the tourism industry. The total economic loss incurred from the 1998 algal bloom in the catchment area was estimated at nearly \$6.5 billion (Le et al., 2010).[1] The algal bloom events that occurred during the summer of 2007 led to a shortage of water supplies for approximately two million residents in Wuxi city (Qin et al., 2010). According to Guo (2007), the root cause of severe eutrophication in Lake Taihu is an accumulation of nutrient-rich sewage and agricultural runoff in the shallow lake.

Municipal wastewater treatment plants in the Lake Taihu basin are required to follow the discharge standard of pollutants for municipal wastewater treatment plant (GB 18918-2002). The emission standard is categorized into four classes (see Table S2 in the supplementary materials). In addition to the current regulations implemented in Lake Taihu basin, some other measures are used to deal with the serious water pollution. For example, a water transfer project from the Yangtze River to Lake Taihu was initiated in 2002 to dilute polluted water and to accelerate flushing pollutants and algae out of the lake, and this transfer is still ongoing.

4. Estimation of the wastewater treatment cost function and the ecological damage cost function

4.1. Wastewater treatment cost function

Following the literature (Tsagarakis et al., 2003; Fraas and Munley, 1984; Uluatam, 1991; Vanrolleghem et al., 1996) an exponential functional form is used to represent wastewater treatment cost function (either investment, or O&M cost function), as is presented below:

$$C_1 = \alpha \cdot Q^{\beta} \quad (11)$$

where C_1 is the wastewater treatment cost expressed in million \$, Q is the wastewater treatment plant capacity expressed in 10^4 m^3/day, α, β are coefficients.

Some other studies (Hernandez-Sancho et al., 2011; Goldar et al., 2001; Dasgupta et al., 2001) include quality variables, including effluent/influent concentration ratio and other variables, such as the input vector and character of the treatment plant (age, ownership, etc.) as shown in equation (12) below:

$$C_2 = \alpha' \cdot Q^{\beta'} \cdot P^{\gamma'} \cdot X^{\theta'} \quad (12)$$

where C_2 is the wastewater treatment cost expressed in million \$, Q is the designed capacity expressed in 10^4 m^3/day, P is the quality variable (i.e., effluent/influent concentration ratio), and X is the vector of input prices (labor, energy, and materials, etc.) in the location of the plants. α', β', γ', and θ' are the coefficients to be estimated.

Since the quality aspect of wastewater effluent is very important for our analysis, we also include a quality variable. Some adjustments are introduced. First, pollutant removal efficiency is used as the quality variable instead of effluent to influent ratio, which is used in literature (Goldar et al., 2001; Dasgupta et al., 2001). The higher the pollutant removal efficiency, the lower the effluent-to-influent ratio; second, it is assumed that the vector of input prices (labor, energy, and materials, etc.) is stable over space and is the

[1] The reason for the substantial cost of damage in the 1998 algal bloom is due to the estimation by Le et al. (2010), which includes indirect costs such as damaged investment environment of the region, and human health damage caused by water quality deterioration.

Table 1
Descriptive statistics of wastewater treatment plants data in China in our sample.

Treatment plant type	Observations	Capacity (Wastewater Flow) ($10^4 m^3/day$)	BOD influent (mg/L)	BOD effluent (mg/L)	COD influent (mg/L)	COD effluent (mg/L)	SS influent (mg/L)	SS effluent (mg/L)	Investment cost (million $)	O&M Cost (million $)
Primary	27 (18)	4.04 (4.37)	269.67 (230.11)	25.02 (23.27)	562.09 (472.13)	97.73 (94.60)	415.18 (366.13)	46.72 (36.02)	16.44 (18.66)	1.42 (1.33)
Secondary	184 (135)	7.23 (7.59)	164.83 (164.55)	13.46 (12.89)	371.00 (363.67)	49.8 (49.08)	221.17 (216.78)	16.43 (15.96)	26.53 (28.60)	1.89 (2.00)
Tertiary	13 (13)	9.80 (10.43)	94.5 (138.05)	6.28 (6.00)	290.86 (278.51)	28.09 (27.67)	213.25 (205.74)	7.13 (7.25)	45.78 (39.78)	1.95 (2.08)
Total	226 (166)	7.02 (7.46)	176.02 (169.58)	14.36 (13.47)	388.52 (368.77)	54.09 (52.34)	243.83 (232.11)	19.43 (17.46)	26.60 (28.39)	1.84 (1.94)

Note: All the numbers are mean values of the corresponding sample. In parentheses are the descriptive statistics (mean value) of O&M cost sample (N = 166).

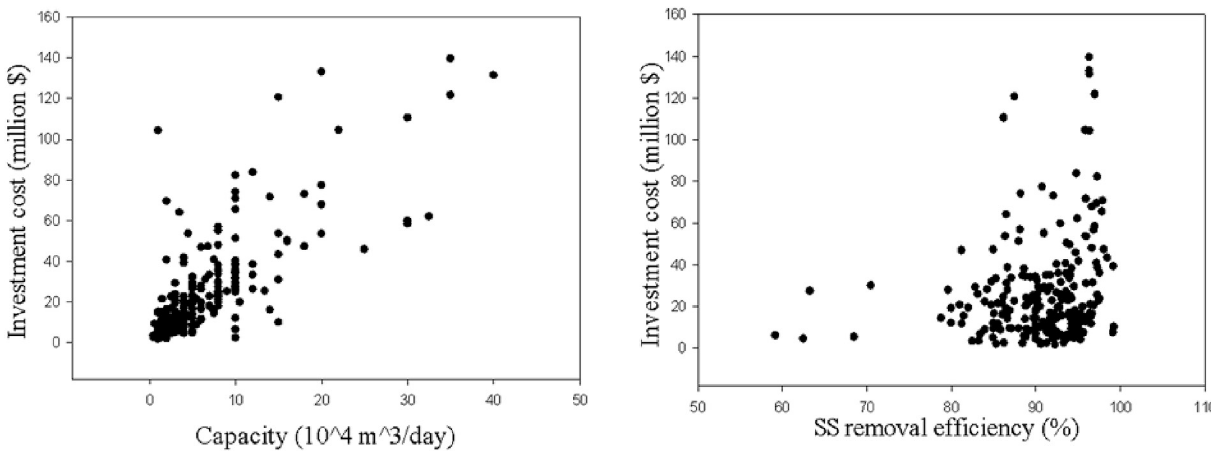

Fig. 4. Relationship between investment cost and wastewater treatment capacity (left panel), and SS removal efficiency (right panel).

same in different regions in China. Therefore, the vector of input prices variable in equation (12) is not included in this study; third, we use wastewater flow instead of treatment capacity to estimate the O&M cost function. That is because the O&M cost is determined by the actual wastewater quantity treated instead of the designed treatment capacity.

The investment cost function specification is presented below as:

$$INV = exp(a) \cdot Q^b \cdot E^c \quad (13)$$

where INV is the capital investment costs expressed in million \$, Q is the treatment capacity expressed in $10^4\ m^3/day$, and E is the pollutant removal efficiency expressed as percentage. a, b and c are parameters to be estimated.

The O&M cost function specification is presented below as equation (14):

$$VC = exp(a') \cdot F^{b'} \cdot E^{c'} \quad (14)$$

where VC is the O&M costs expressed in million \$, F is the wastewater flow expressed in $10^4\ m^3/day$, and E is the pollutant removal efficiency expressed as percentage. a', b' and c' are parameters to be estimated.

4.2. Wastewater treatment data

The wastewater treatment data was taken from China's Urban Sewage Treatment Plant Assembly (Yang, 2006) and China Urban Wastewater Treatment Facilities List (Ministry of Environmental Protection of China, 2012). The detailed description of the data used for wastewater treatment cost function estimation is available at Jiang et al. (2013). Descriptive statistics of the observations are presented in Table 1. All the statistical analyses are conducted using IBM SPSS 20 software. All economic values in this paper are expressed in 2006 $US level.

4.2.1. Estimation of the capital investment cost function

We use only one quality parameter – suspended solids (SS) – removal efficiency as the quality variable, because we want the wastewater treatment cost function and the ecological damage function to use the same quality variable. SS is used in the literature (Newcombe and MacDonald, 1991) to represent stress level in ecosystems. The SS concentration alone shows relatively poor indication of stress. The combination of concentration and duration will be a better indicator of stress level. However, because the social optimization model in this paper is a static one, only SS concentration is used to represent the stress level.[2] The relationships between capital investment cost and designed capacity, and SS removal efficiency are presented in Fig. 4 (left panel and right panel, respectively).

As introduced in Jiang et al. (2013), dummy variables were used

[2] Dynamic relationships will be developed in the future work.

[3] We also estimated the investment cost function with designed capacity with the same sample (N = 166), which is used to estimate O&M costs (Results can be obtained from the authors upon request). We find that the coefficients, t-test, significance level, adjusted R square, F test and model significance level are all very close to the one we estimated with the larger sample (N = 226). Therefore, it is acceptable for us to use the regression results, which are estimated based on 226 samples (Table 4) to estimate the investment cost function and then use it in the social optimization model.

Table 2
Regression results of investment cost model (adjusted R square = 0.596, F = 111.632, p < .001***)[3].

Coefficients	Values	Standard error	t	p
a_1	2.647	0.145	18.261	< .001***
a_2	−0.782	0.097	−8.078	< .001***
b	0.484	0.049	9.804	< .001***
c	3.104	0.855	4.243	< .001***

Note: *** indicates significance at 0.1% level.

Table 3
Regression results of O&M costs model (adjusted R squared = 0.585, F = 78.423, p = .000***).

Coefficients	Values	Standard error	T	p
a'_1	0.119	0.138	0.863	.389
a'_2	−0.907	0.120	−7.577	< .001***
b'	0.413	0.050	8.211	< .001***
c'	2.575	0.758	3.396	< .001***

Note: *** indicate significance at 0.1% level.

to help with the imperfect performance of the data. The dummy variable used for capital investment cost function is defined as: $Dummy\_ESS = 1$, if $0.9 < E < 1$, and if $0 < INV < \$\ 20 million$; otherwise $Dummy\_ESS = 0$. Modifying Eq. (13), the capital investment cost function is written as

$$INV = exp(a_1 + a_2 \cdot Dummy\_ESS) \cdot Q^b \cdot E^c \tag{15}$$

where INV is capital investment cost in *million* \$/year, Q is designed capacity (10^4 m^3/day), E is the pollutant removal efficiency, $Dummy\_ESS$ is the dummy variable, a_1, a_2, b and c are coefficients to be estimated.

A general linear regression model (GLM) in natural logarithm is used to estimate the capital investment cost function. The regression result is shown in Table 2. Each parameter reaches significance at 0.1 percent level. The parameter of designed capacity is less than one (0.484), therefore, the capital investment cost function shows strong economies of scale.

4.2.2. Estimation of the O&M cost function

The relationships between O&M cost and wastewater quantity, and SS removal efficiency are presented in Fig. 5 (left and right panel, respectively).

The dummy variable used for O&M cost function is defined as: $Dummy\_ESS' = 1$ if $0.9 < E < 1$, and if $0 < VC < \$\ 1 million$; otherwise $Dummy\_ESS' = 0$. Modifying equation (14), the O&M cost function is written as:

$$VC = exp(a'_1 + a'_2 \cdot Dummy\_ESS') \cdot F^{b'} \cdot E^{c'} \tag{16}$$

where VC is O&M cost in *million* \$/year, F is wastewater flow (10^4 m^3/day), E is pollutant removal efficiency, $Dummy\_ESS'$ is the dummy variable for O&M cost function, a'_1, a'_2, b' and c' are coefficients to be estimated.

The regression result is presented in Table 3. The intercept is not significantly different from 0, which means that the O&M cost is zero when there is no wastewater flow.

4.2.3. Calculating annual total wastewater treatment cost function

To calculate the total annual costs, the capital investment cost is annuitized by using the capital recovery factor (CRF) (Tsagarakis et al., 2003). The CRF is calculated as:

$$\mathrm{CRF} = \left[r(1+r)^t\right] \Big/ \left[(1+r)^t - 1\right] \tag{17}$$

where r is the discount rate, and t is the designed lifetime of the treatment plant.

Considering the circumstances in China, a discount rate of 4 percent and lifetime of 20 years are set in this paper, based on Wang et al. (1992), Niu et al. (2011), and Yu et al. (2011). Thus, CRF is calculated as 0.0736.

The annual total wastewater cost function is therefore written as:

$$C = 0.0736 \cdot exp(2.647 - 0.782 Dummy\_ESS) \cdot Q^{0.484} \cdot E^{3.104} + exp(0.119 - 0.907 Dummy\_ESS') \cdot F^{0.413} \cdot E^{2.575} \tag{18}$$

where C is the annual wastewater treatment cost (*million* \$).

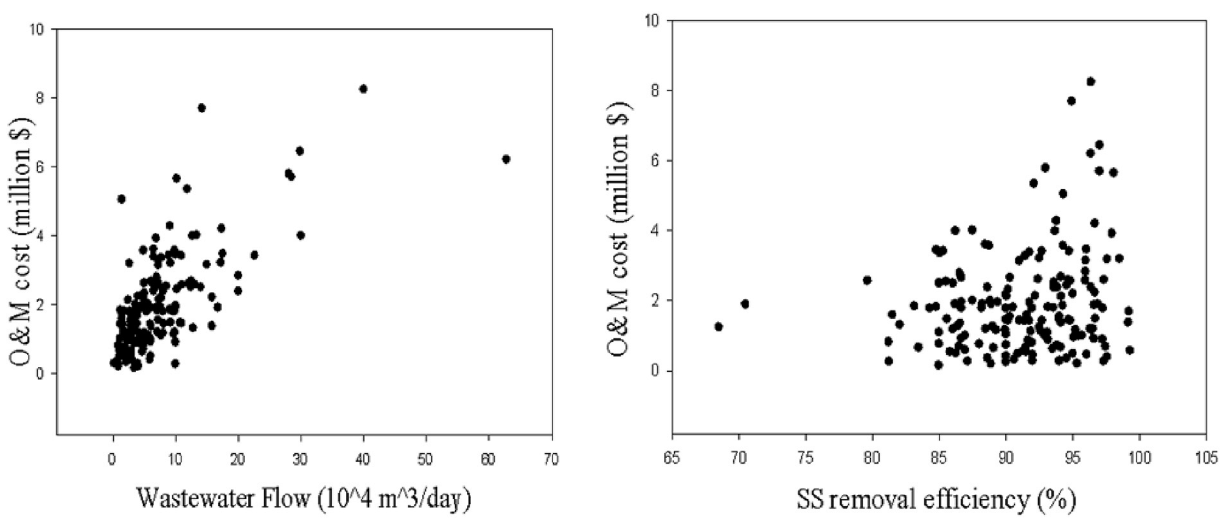

Fig. 5. Relationship between O&M costs and wastewater flow (left panel), and SS removal efficiency (right panel).

4.3. Estimation of the ecological damage cost function

The ecological damage function in Lake Taihu is estimated following the framework introduced earlier (see Fig. 2), proposed by Keeler et al. (2012).

Step 1: Link human activities (wastewater discharge) and changes in water quality

The estimated association is based on the assumption that our model is a static one, and the suspended solids (SS) from the wastewater treatment facility is diluted by the water in the lake after being discharged. The SS concentration in the lake after wastewater discharge is:

$$SS = (q_{out} \cdot F)/(F + Q_L\ ) + S_o \tag{19}$$

where SS is the final suspended solids concentration in Lake Taihu (mg/L), S_o is the original concentration in the lake before wastewater discharge mg/L, and Q_L is the water quantity in Lake Taihu (m^3).

Step 2: Link changes in water quality to changes in ecosystem services

As indicated in the beginning of this paper, the recreation and waste disposal are considered to be the only functions of the lake. In our study, waste disposal means the discharge of the (treated) wastewater into the lake. Waste disposal service is not associated with water quality, while the recreation service (e.g., swimming, angling, and viewing) depends on water quality, especially water clarity. Thus, changes in water clarity will lead to changes in the recreation service. Water clarity is usually measured in Secchi Transparency. Although Secchi Transparency is often used as a water quality indicator, it also can be used as an ecosystem state indicator, due to its close association with algal biomass. For example, in the ecosystem health assessment study of Xu et al. (1999), Secchi Transparency was used as one of the ecological indicators of ecosystem state. As indicated in section 2, Scheffer et al. (2000) argued that much of the essence of the state of the ecosystem can often be captured by a single key variable. Therefore, we adopted the Secchi Transparency as the indicator of the state of Lake Taihu's ecosystem.

The empirical relationship between Secchi Transparency and suspended solids in Lake Taihu was adopted from the study of Qin et al. (2007), as shown in Equation (20):

$$ST = exp\left(1.39 - 1.17 \cdot SS^{0.25}\right) \tag{20}$$

where ST is the Secchi Transparency (m), and SS is the suspended solids concentration (mg/L) in Lake Taihu.

Equation (20) is plotted in Fig. 6, which is similar to type (a) in Fig. 1. It shows that the ecosystem state (indicated by Secchi Transparency) worsens dramatically at the beginning when the human-induced stress (indicated by concentration of suspended solids) increases. After reaching a certain level of stress, the deterioration rate slows down.

Step 3: Link changes in ecosystem services to changes in values

Zhang (2011) measured the economic value of water quality improvement in Lake Taihu by eliciting local citizens' willingness to pay for a hypothetical water quality improvement project. The target of the hypothetical project was to improve the water quality

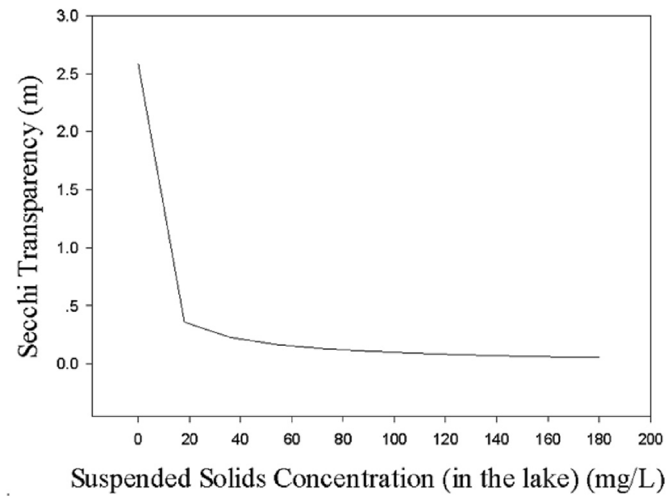

Fig. 6. Empirical relationship between Secchi Transparency and SS concentration in Lake Taihu.

to Grade IV and partly to Grade III, based on national surface water quality standard (GB 3838-2002). As presented in Zhang (2011), water quality in most areas of the survey cities (69.1 percent) surrounding Lake Taihu is worse than Grade V. Therefore, it is acceptable to equal the economic value measured by Zhang (2011) to the economic value of improving the water clarity from values worse than Grade V to values within Grade IV.

The standard for Secchi Transparency was deleted in the recent version of the water quality standard (GB 3838-2002). The difference between GB 3838-2002 and GB 3838-88 is mainly about adding or deleting a few indicator items. The values of each indicator under different grades and the description for each grade are exactly the same. In the previous version of the national surface water quality standard (GB 3838-88), the values of Secchi Transparency of Grade IV, Grade V, and worse than Grade V, are 2.5 m, 1.5 m, and 0.5 m, respectively. Therefore, improving the water clarity from values worse than Grade V to values within Grade IV is also assumed to be equal to increase the water clarity from 0.5 m to 2.5. Since the willingness to pay for water quality improvement measured by Zhang (2011) may include other aspects of water quality improvement in addition to water clarity, the economic value we take from Zhang (2011) may exceed the actual economic value of increasing water clarity by 2 m. There is a very high uncertainty level associated with this economic value, therefore, a sensitivity analyses will be applied.

As defined earlier, ξ represents the unit economic value of ecosystem service. It was assumed in the theoretical part of section 2 that citizens are willing to pay more for one unit of improvement at lower levels of the ecosystem state than at the higher levels of the ecosystem state, which means ξ will change as the ecosystem state changes. However, the data in the literature does not allow us to differentiate ξ. In the numerical application, we can only assume that the marginal recreation utility of increasing Secchi Transparency by 1 m is the same under various water clarity levels. Ideally, in the future when the data is more complete, we can relax this assumption. The economic value from Zhang (2011) is \$27.8 $million$/meter. Therefore, $\xi = \$27.8\ million$/meter.

As introduced in section 2, the ecological damage cost is defined as the difference between economic values of the current and reference ecosystem states. We select the highest Secchi Transparency as the reference ecosystem state. Based on the fact that Lake Taihu is a shallow lake with a maximum depth of 2.6 m, the

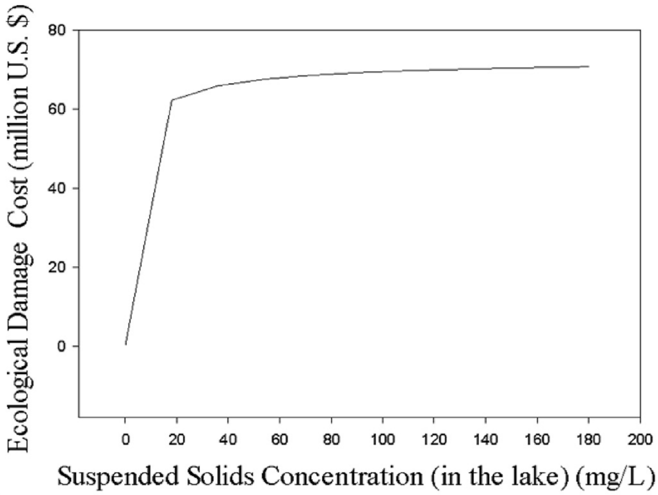

Fig. 7. Ecological damage cost function plot. Note: The range of this horizontal axis is supported by Qin et al. (2007) (Fig. 4 in their paper).

reference ecosystem state without any damage is defined as 2.6 m of water clarity as measured by Secchi Transparency. The corresponding suspended solid concentration to the maximum transparency is 0.02 mg/L. This implies that ecological damage will occur when SS concentration in the lake is higher than 0.02 mg/L.

$$Dummy\_ESS = 1,\ if\ 0 < exp(2.647 - 0.782Dummy\_ESS)\cdot Q^{0.484}\cdot E^{3.104} < 20\ and\ 0.9 < E < 1;\ \ otherwise,\ Dummy\_ESS = 0 \tag{29}$$

$$Dummy\_ESS' = 1,\ if\ 0 < exp(0.119 - 0.907Dummy\_ESS')\cdot F^{0.413}\cdot E^{2.575} < 1\ and\ 0.9 < E < 1;\ \ otherwise, Dummy\_ESS' = 0 \tag{30}$$

The change in the state of Lake Taihu's ecosystem is indicated by the change in water clarity, measured by Secchi Transparency. The reference ecosystem state is $S_0 = 2.6\,m$, as introduced earlier. Combined with equation (20), the change of water clarity is measured by equation (21):

$$S_0 - ST = 2.6 - exp\left(1.39 - 1.17\cdot SS^{0.25}\right) \tag{21}$$

where S_0 is the reference ecosystem state measured by Secchi Transparency (m).

Hence, the ecological damage cost is the product of unit economic value of ecosystem service (ξ) and change of ecosystem state (equation (21)),

$$\begin{aligned} D &= \xi\cdot\left[2.6 - \exp\left(1.39 - 1.17\cdot SS^{0.25}\right)\right] \\ &= 72.28 - 27.8\cdot exp\left(1.39 - 1.17\cdot SS^{0.25}\right) \end{aligned} \tag{22}$$

where D is the ecological damage cost ($million$ \$)

Equation (22) is plotted in Fig. 7.

5. Numerical application of social optimization model to Lake Taihu in China

Based on the results in section 4, the numerical social optimization model is written as equation (23):

$$\begin{aligned} \underset{E}{Min} &= C(Q,E,F) + \xi\cdot D(S) \\ &= 0.0736\cdot exp(2.647 - 0.782Dummy\_ESS)\cdot Q^{0.484}\cdot E^{3.104} \\ &\quad + exp(0.119 - 0.907Dummy\_ESS')\cdot F^{0.413}\cdot E^{2.575} \\ &\quad + 72.28 - 27.8\cdot exp\left(1.39 - 1.17\cdot SS^{0.25}\right) \end{aligned} \tag{23}$$

The empirical constraints of this model are as below:

$$E\cdot q_{in} = q_{in} - q_{out} \tag{24}$$

$$SS = (q_{out}\cdot F)/(F + Q_L) + 0.02 \tag{25}$$

$$0 \le E \le 1 \tag{26}$$

$$SS \ge 0.02 \tag{27}$$

The total wastewater discharged to Lake Taihu is $2.92\times10^9\ m^3$ in 2006 (Taihu Basin Authority, 2006), hence the wastewater flow per day is $8\times10^6\ m^3/day$. The discharged wastewater quantity is taken as exogenous.

$$F = Q = 8\times10^6 m^3\Big/day \tag{28}$$

As mentioned in section 3, the water volume of Lake Taihu is $44.33\times10^8\ m^3$. We applied the average value of SS (see Table 1) as the influent concentration in the model, which is 243.83 mg/L. All the coefficients of parameters used for the numerical application are listed in Table 4. The non-linear programming optimization is conducted in LINGO 14.0.

The optimal solution of the model is based on the coefficients in Table 4 and presented in Table 5.

Because the coefficients (Table 4) used for the base run are assumed, high uncertainty is associated with their values. Sensitivity analyses are therefore applied by changing the coefficients of Lake Taihu's water quantity and the economic value of its ecosystem service. The sensitivity analyses results are shown in Table S3 in the supplementary materials.

The sensitivity analyses results presented in the following paragraphs explore how the water quantity in Lake Taihu and the economic value of its ecosystem service can affect the treatment level and social cost in the optimal solutions.

Fig. 8 shows that treatment level and social optimal costs are sensitive to the change of the water quantity coefficient values. When the water volume in Lake Taihu increases, the optimal wastewater treatment level decreases, and so does the corresponding social optimal cost. It is reasonable because the assimilative capacity of the lake is enhanced when the water volume increases. An example was already mentioned in section 3: water transfer from the Yangtze River to Lake Taihu was initiated in 2002

Table 4
Coefficients of parameters used for the social optimization model of Lake Taihu.

Parameters	Description	Units	Value	Source
q_{in}	SS influent concentration	mg/L	243.83	Yang (2006)
Q_L	Water volume of the lake before discharging the wastewater	m^3	44.33×10^8	Hu et al. (2006)
ξ	Unit economic value of ecosystem	$million\ \$/m$	27.8	Zhang (2011)

Table 5
The solutions of the social optimization model based on coefficients in Table 4 (base run).

q_{in} (mg/L)	Q_L m^3	ξ $(million\ \$/m)$	E	Ecosystem damage (m)	q_{out} (mg/L)	Optimal Solution $(million\ \$)$
243.83	44.33×10^8	27.8	1	0	0	44.21

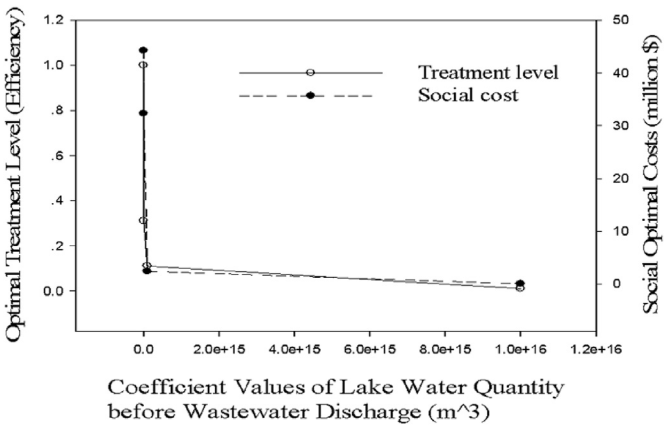

Fig. 8. Sensitivity analysis by changing the coefficients of water quantity value in Lake Taihu.

to dilute polluted water and to accelerate flushing pollutants and algae out of Lake Taihu. This process is still ongoing.

Fig. 9 suggests that the optimal treatment level and social optimal costs are sensitive to the changes of economic value of ecosystem service. As presented in Table 4 and Table S4 in the supplementary materials, when the economic value of the ecosystem service is \$27.8 *million*/meter, measured by Zhang (2011), the optimal treatment already reaches the highest level and no ecosystem damage occurs. This implies that in the context of using the lake ecosystem for recreation services only, the \$27.8 *million*/meter valuation level is high enough to convince society to prioritize ecosystem health rather than damaging it. A higher valuation level of ecosystem service will not change both the optimal treatment level and overall social optimal cost (see Table S4 in the supplementary materials). When the valuation level is lower than the base case, as shown in Fig. 9, the optimal treatment level and social optimal cost decrease, which implies that the decision-maker may decrease the treatment level when the valuation of the ecosystem is low in order to achieve the minimal social cost. However, as presented in Table S4 in the supplementary materials, the corresponding ecosystem damage increases to a very high level. This shows that a low perceived value attached to the ecosystem service will lead to serious ecological damage. An accurate valuation, which is close to the “real” value of the ecosystem service is very important in the decision-making process with respect to ecosystem protection.

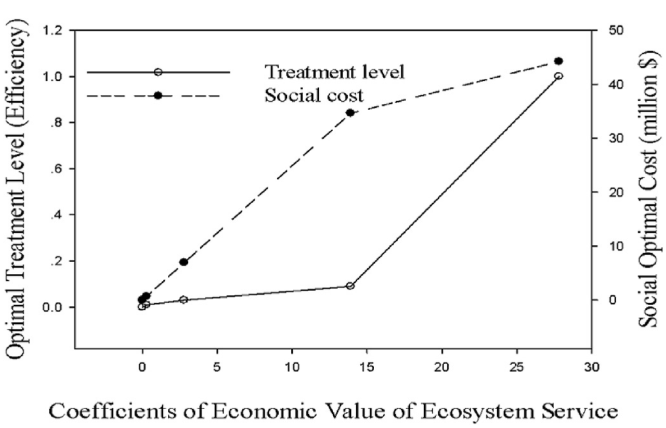

Fig. 9. Sensitivity analysis by changing the coefficients of economic value of ecosystem service in Lake Taihu.

6. Conclusions and policy implications

In this paper, a social optimization model – specifically, a social cost minimization model – was developed in order to economically assess the trade-off between human-based pollution abatement and the damage to the ecosystem of the receiving water body. The empirical wastewater treatment cost function estimated for China shows strong economies of scale in terms of capacity, and diseconomies in terms of pollutant removal efficiency (treatment level). The ecosystem damage cost function was estimated by integrating the existing biophysical model and the economic valuation model in the literature. By integrating the treatment cost decision and the ecological damage in one model, policy-makers are better equipped to identify the trade-off for socially optimal solutions under various conditions, such as initial volume of water in the receiving body and the value of the services that can be obtained from the lake's ecosystem.

A theoretical model was applied to the well-known case of Lake Taihu in China, using existing secondary data from the literature, and additional simplifications to demonstrate the concept of endogenous considerations of the ecological damage. Sensitivity analyses were conducted to key variables (water volume in the lake and economic value of ecosystem service) to assess the robustness of the model results.

The base run of our model suggests that a full treatment of pollution is required in the region to minimize the social cost. Results also show that the greater the lake water volume, the lower the requirement for the treatment level, and the lower the social optimal cost. This can be explained by the lake's assimilative capacity and dilution ability. Furthermore, we found that a low economic valuation of the ecosystem will lead to serious ecosystem damage. When the valuation reaches a certain level, it will not affect treatment level and social optimal cost because the treatment level is high enough to prevent the ecological damage.

One important policy implication is that the assimilative capacity of the lake should be enhanced by forbidding over-extraction of water from the lake. It is proven that the lake's water helps dilute the pollution, and the requirement for treatment will be reduced as well as the social cost. Another policy implication is that more work needs to be done on the economic valuation of the ecosystem service in order to guarantee reliable information for decision-making involving the lake's ecosystem protection.

Given the data limitations that exist at present in various countries, and their implications for policy, planning, and decision-making, we suggest that regulatory agencies adjust the list of variables that are collected from wastewater treatment plants to the possible mediums to which such water are disposed off. The same holds for the water quality of the water medium used for wastewater disposal. For example, in our study we had only information on pollutant variables COD, BOD and SS, which are not good measures for ecosystem stress. With a relatively low cost of data collection, information on nitrogen and phosphorous concentrations (inflow and outflow) in wastewater treatment plants and in receiving bodies, such as lakes, would be much more useful.

The work presented here is useful for environmental practitioners concerned with managing water pollution issues. The social optimization model introduced in this paper may equip environmental practitioners with a tool to help identify a socially optimal pollution control level that can reflect the local population's valuation of aquatic ecosystem services.

In a future study, the partial equilibrium model will be upgraded to a general equilibrium model by including more sectors that benefit from the lake's ecosystem. The ecological damage cost function will be further developed to include more complex ecosystem health indicators. A dynamic model will also be applied. Furthermore, Lake Taihu can be divided into several sub-zones in order to take spatial heterogeneity of vulnerability into consideration.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.jenvman.2018.01.047.

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