Environmental Management (2013) 51:724–737
DOI 10.1007/s00267-012-9999-y

Modeling Nutrient Release in the Tai Lake Basin of China: Source Identification and Policy Implications

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Received: 23 September 2011 / Accepted: 11 December 2012 / Published online: 16 January 2013

Abstract Because nutrient enrichment has become increasingly severe in the Tai Lake Basin of China, identifying sources and loads is crucial for watershed nutrient management. This paper develops an empirical framework to estimate nutrient release from five major sectors, which requires fewer input parameters and produces acceptable accuracy. Sectors included are industrial manufacturing, livestock breeding (industrial and family scale), crop agriculture, household consumption (urban and rural), and atmospheric deposition. Results show that in the basin (only the five sectors above), total nutrient loads of nitrogen (N) and phosphorus (P) into aquatic systems in 2008 were 33043.2 tons N a^{-1} and 5254.4 tons P a^{-1}, and annual area-specific nutrient loads were 1.94 tons N km^{-2} and 0.31 tons P km^{-2}. Household consumption was the major sector having the greatest impact (46 % in N load, 47 % in P load), whereas atmospheric deposition (18 %) and crop agriculture (15 %) sectors represented other significant proportions of N load. The load estimates also indicate that 32 % of total P came from the livestock breeding sector, making it the second largest phosphorus contributor. According to the nutrient pollution sectors, six best management practices are selected for cost-effectiveness analysis, and feasible options are recommended. Overall, biogas digester construction on industrial-scale farms is proven the most cost-effective, whereas the building of rural decentralized facilities is the best alternative under extreme financial constraint. However, the reduction potential, average monetary cost, and other factors such as risk tolerance of policy makers should all be considered in the actual decision-making process.

Keywords Nutrient loads · Tai Lake Basin · Best management practice (BMP) · Cost-effectiveness analysis

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Introduction

Nutrient inputs have dramatically increased in recent decades, and have degraded water quality in many rivers, lakes, and oceans worldwide (Nixon 1995; Schindler and Vallentyne 2008; Vitousek and others 1997; Vollenweider 1982). Eutrophication caused by surplus nitrogen (N) and phosphorus (P) has thus become one of the most rapidly growing environmental crises of surface waters. China faces the same pollution challenges as many other parts of the world, and inland lakes are the most severely affected aquatic systems. The Tai Lake Basin in the Yangtze River Delta, which has witnessed dramatic eutrophication over the past three decades, is widely regarded as a mandatory target of water pollution control (Liu and Diamond 2005; Qin and others 2007; Yang 1996). Excessive anthropogenic nitrogen and phosphorus inputs have been implicated in the rapid rate of eutrophication of Tai Lake (Paerl and others 2011; Xu and others 2010). Nutrient discharges come from various point and nonpoint sources in most watersheds (Kaushal and others 2011). Distinct sources such as industrial or municipal wastewater discharge and runoff from farms or arable land have contributed to nutrient enrichment in the basin (Paerl and others 2011; Qin and others 2007, 2010; Xu and others 2010).

Since the early 1990s, water pollution control has been prioritized to protect the ecosystem of the Tai Lake Basin (Ellis and Wang 1997; Zhang and others 2008). During the 9th Five-Year Plan period (1995–2000), the central government and local authorities invested CNY 10 billion for environmental protection of the basin. Numerous measures and policies have continually increased pollution control of point sources, but concerning the mitigation threshold, the effects appear to have been limited (Chen and others 2003; JSEPD 2009). Water in the basin is still nutrient enriched, and the concentration of nitrogen alone has increased by more than 50 % in the past 10 years (Fig. 1). Owing to this fact, attention has gradually shifted to nonpoint pollution from agriculture and other rural activities. To improve water quality and relieve eutrophication in the entire watershed, detailed understanding of nutrient loads and sources is essential. Therefore, a systematic modeling tool capable of depicting the entire picture of basin nutrient release should be proposed (Kennedy and others 2007; Liu and Qiu 2007; Puckett 1994).

Excessive nutrient surplus in the Tai Lake Basin comes from various sources, among which industrial point source discharge can be easily quantified via monitoring or statistical sources, making nonpoint pollution estimation the primary challenge in assessing nutrient input. Various models and methods have been developed to tackle this mission. White box physical hydrologic models, such as ANSWERS (Beasley and others 1980), SWAT (Arnold and others 1998), AGNPS (Young and others 1989), INCA (Whitehead and others 1998) and HSPF (Donigian and others 1995) all provide reliable results after long-term continuous modeling. Unfortunately, strict input requirements significantly prevent their widespread application, especially for data scarce watersheds like the Tai Lake Basin (Parsons and others 2001; Zhao and others 2011).

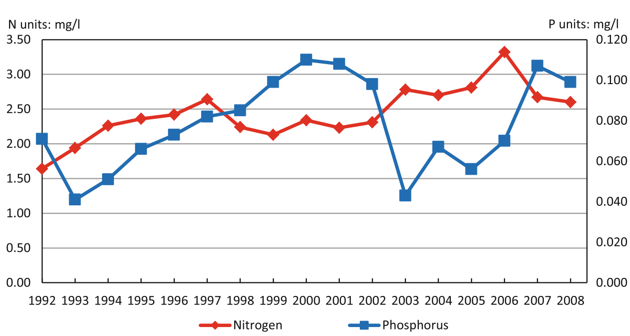

Fig. 1 Nutrients concentration in Tai Lake during 1992–2008 (based on monthly values). Adapted from Qin and others (2006), Zhang and others (2008)

Since extensive nutrient sampling and modeling expertise are often absent or unattainable, simple alternative methods like the export coefficient (Johnes 1996) or the hydrograph separation method (Sloto and Crouse 1996) remain attractive options. These methods describe most pollution processes and yield credible results (Ding and others 2010). In China, researchers have also developed applicable native models (Guo and others 2004; Hong and others 2010; Li and others 2009), most of which are constrained to specific areas and lack extensive testing (Shen and others 2011). For the Tai Lake Basin, there are also regional estimation models that show certain accuracy and potential (Lai and others 2006; Zhang and others 2010; Zhao and others 2011). However, because of data scarcity and difficulty of source identification, a more practical accounting method that requires fewer data and achieves acceptable accuracy is needed for watershed management.

In the context of this study, a nutrient load estimation framework for the Tai Lake Basin covering five major pollution sources was developed. This framework is presented in the following sections. Nitrogen and phosphorus release in 2008 was then calculated. In addition, considering mitigation potential for major polluters, selected best management practices (BMPs) were analyzed in a cost-effectiveness evaluation. The rest of the paper is organized as follows. The "Methodology" section describes the nutrient load estimation method and data sources, followed by empirical results and discussion in the "Results and Discussion" section. The "Conclusions" section offers major conclusions.

Methodology

Analytical Framework

Nutrient sources and flows in a lake basin are shown in Fig. 2. A portion of nutrient load flows from sources directly into surface waters, and the remaining sources discharge pollutants after treatment. In contrast to unit-area models like the nutrient load model of the PAMOLARE package (Zhang and Jørgensen 2005) or the CMSS system (Broad and Corkrey 2011), diffuse nutrient surplus estimates here assume that nutrient exports depend on pollution sectors, and each sector may have various sources, leading to nutrient generation and losses. However, the hydrology factor impact and nutrient release of fishponds are not explicitly addressed in this paper.

Considering the complexity and multi-source characteristics of the nutrient cycle in an anthropogenic system, the developed model predicts total loads produced in the watershed by calculating nutrient loads generated from major pollution sectors and summing over all sectors. The

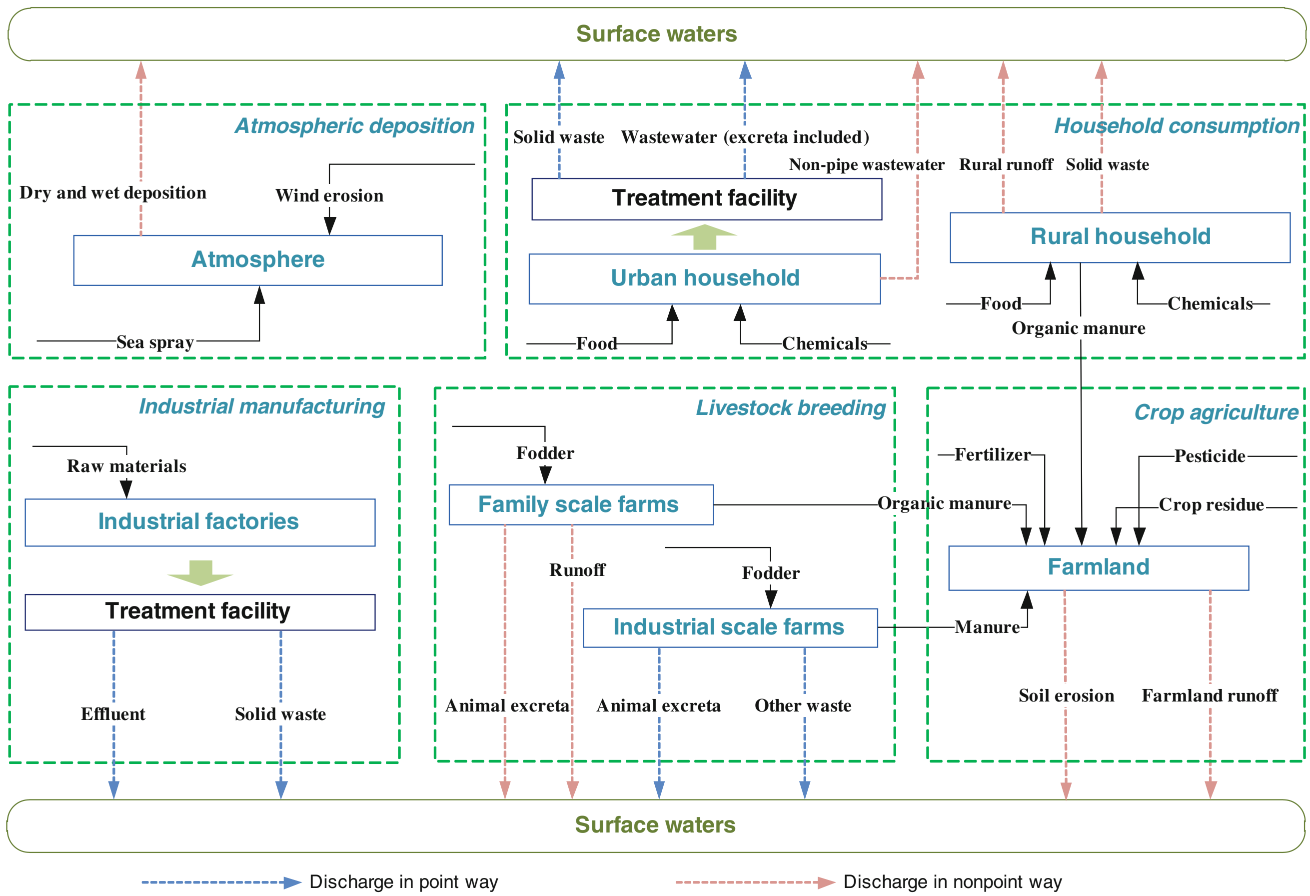

Fig. 2 Simplified nutrient sources and flows diagram for load release estimate of the Tai Lake Basin. In the above framework, the five pollution sectors are shown in *dashed green boxes*. The *solid blue boxes* refer to the nutrient sources in each sector, while the *dashed black arrows* indicate nutrient loads into surface waters. Adapted from Li and others (2010), Robertson and Vitousek (2009), and Smil (2000)

five major nutrient release sectors are industrial manufacturing, livestock breeding, crop agriculture, household consumption and atmospheric deposition. Classes of sectors and sources are presented in Table 1. Moreover, aiming to directly control nutrient loss contributors, applicable BMPs were selected for further discussion. The BMP referring to the atmospheric deposition sector is excluded, because it is difficult to directly reduce nutrient inputs from deposition using technical and managerial measures (Yang and others 2007). By way of a quantified cost-effectiveness analysis, the performance of identified BMPs were evaluated, the results of which may provide useful information to support policy making.

Because of differences in discharge pathway, the livestock breeding sector is divided into industrial and family-scale farms. The latter farms refer to husbandry work within the residential boundary of rural families, and solid waste and wastewater discharge are two different pathways of urine and manure loss from livestock farms. Additionally, household solid waste primarily refers to garbage disposed in urban and rural daily life, which might run off into surface waters without treatment from landfills in rural areas or even city outskirts. For existing or planned pollutant treatment facilities, removal efficiency was included in the accounting process. Considering different conditions of solid waste and sewage treatment infrastructures, urban and rural residents are discussed separately.

Study Area

The Tai Lake Basin of Jiangsu Province (colored regions in Fig. 3) includes three major cities (Suzhou, Wuxi and Changzhou) and covers approximately 19,399 km^2. The basin is downstream of the Yangtze River in Eastern China. It is one of the most industrialized areas in China, and the high residential density promotes social activity. The value added by agricultural production accounts for a moderate share of regional GDP. High nutrient runoff rates from cropland and farms are closely related to total nutrient loads. Atmospheric deposition has been shown to be a

Table 1 Detailed classification of pollution sectors and sources

Pollution sectors	Pollution sources
Industrial manufacturing	Effluent discharge
Livestock breeding	Solid waste from industrial scale farms
	Wastewater from industrial scale farms
	Solid waste from family scale farms
	Wastewater from family scale farms
Crop agriculture	Chemical fertilizer
	Organic manure
	Crop residue
Household consumption	Urban household solid waste
	Urban household wastewater (excreta included, pipeline network)
	Urban household wastewater (excreta included, non-pipeline area)
	Rural household solid waste
	Rural household wastewater (excreta included)
Atmospheric deposition	Atmospheric dry deposition
	Wet precipitation

significant nitrogen contributor, because of widespread coal-fired power plants and chemical industries. These characteristics fully represent the five major pollution sectors in the basin. Most importantly, the industrial manufacturing, livestock breeding, crop agriculture, and household consumption sectors are found in the above three cities. Atmospheric nutrient deposition is estimated based on the area of the basin and its 15 inflow rivers.

Accounting Approach

The nutrient load accounting methods are presented sequentially. In the industrial manufacturing sectors (NS_e), only effluent discharge is considered because the nutrient content of industrial solid waste and other types of waste tends to vary greatly in terms of products (Keller and others 1997). Nutrient release through effluent discharge is calculated based on data from a regional industrial census in 2008. This census covered all large enterprises within the study region, which account for almost 90 % of total regional industrial wastewater discharge (JSPSCO 2008;

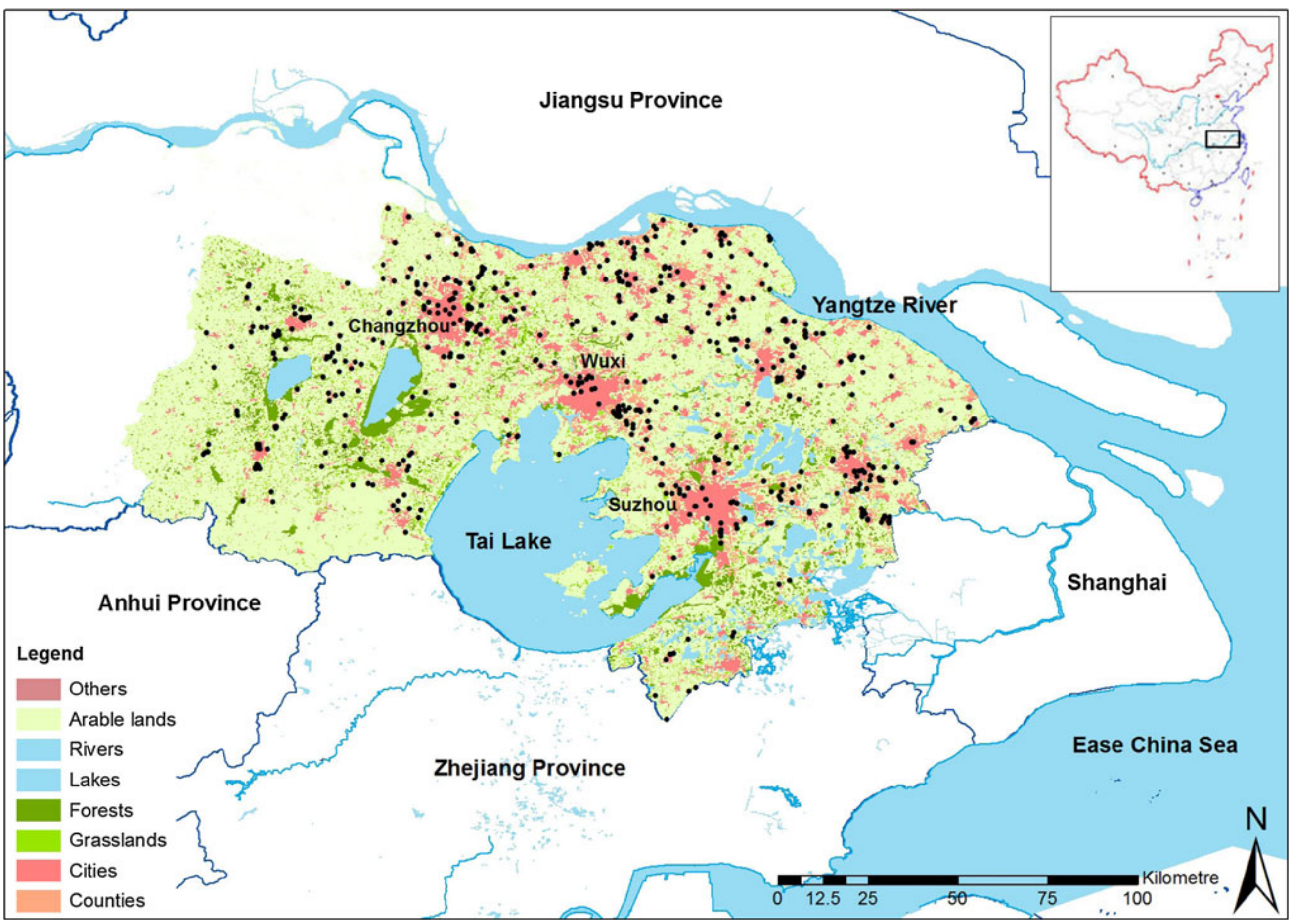

Fig. 3 Location and surroundings of the Tai Lake Basin in Jiangsu province, China. In the above figure, *black dots* represents main point sources (top 750 regarding nutrients discharge) within the study Tai Lake Basin boundary, which include industrial factories and WWTPs

JSSB 2010). Nutrient loads from industrial effluent can be described by the following equation:

$$NS_{e(N,P)} = L_{(N,P)} \times \left(1 - RE_{(N,P)}\right) \quad (1)$$

Here, $NS_{e(N,\ P)}$ is the nutrient load into surface waters from the industrial manufacturing sector, L is the total amount of nutrients in industrial effluent into wastewater treatment plants (WWTPs), and RE is the removal efficiency of nutrients. A modified anaerobic–anoxic–oxic (A^2/O) process was adopted for WWTPs in the study area. RE was 80.7 % for N and 91.0 % for P according to the pollution census (JSPSCO 2008) and field investigation of two plants within the basin.

Next, because pollutants from these sectors enter recipient water bodies in a diffuse manner and at intermittent intervals, nutrient release from livestock breeding (NS_l), crop agriculture (NS_a) and household consumption sectors (NS_h) are calculated using a similar unit-based method, which is adapted from the improved export coefficient method (Ding and others 2010; Johnes 1996). This method follows a bottom-up calculation process, and builds direct links between pollution sources and nutrient release through empirical data and coefficients, as given by the following equation:

$$NS_{(N,P)} = \sum_i \left[NC_{i(N,P)} \times \sum_j \left(EU_{ij(N,P)} \times EC_{ij(N,P)} \right) \right] \quad (2)$$

Here, nutrient load of a sector (NS_l, NS_a, and NS_h) is total release into surface waters of all relevant specific nutrient classes (NC). Specifically, for the *i*th nutrient class of the corresponding NS, the load is calculated based on quantity of the nutrient class (NC_i), nutrient amount in the corresponding emission unit (EU_{ij}), and its emission coefficient (EC_{ij}) into surface waters. For urban sewage discharge, areas with or without a wastewater pipeline network (see Table 1) are considered separately. In the crop agriculture sector, EU^a_{i2} (organic manure) is calculated by multiplying the nutrient amount in all possible organic manure (excreta) and the ratio of organic manure applied to farmland over total excreta produced. The excreta come from both household consumption (human excreta) and livestock breeding (livestock excreta) sectors. EU^a_{i3} (crop residue) is calculated by multiplying the amount of crop residues applied to farmland and the nutrient content of crop residues. Vegetables are excluded since their residue amount is too small to be considered (Li and others 2010; Smil 2000). In the household consumption sector, because the centralized treatment rate varies significantly by city, values of the three major cities are discussed separately. For urban areas not covered by a sewage pipeline network, nutrient release is included in “rural household consumption” instead of “urban household consumption.”

Atmospheric deposition into surface waters also contributes a significant share of watershed nutrient loads, especially in terms of nitrogen (Yang and others 2007; Zhai and others 2009). However, aquatic systems have a pollutant reduction effect, such as denitrification. Only deposition into Tai Lake and its 15 inflow rivers was considered; deposition onto other land use types (such as forest) was excluded. Nutrient loads can be calculated using the following equation:

$$NS_{d(N,P)} = SA \times AD_{(N,P)} \quad (3)$$

Here, nutrient input via atmospheric deposition (NS_d) is the total nitrogen or phosphorus load from both wet and dry deposition, which is calculated by multiplying total aquatic surface areas of Tai Lake and its 15 inflow rivers (SA) by the annual atmospheric nutrient deposition rate (AD) in the region.

Load accounting units, such as the multiple nutrient classes and emission units in the unit-based method or effluent volume in the industrial manufacturing sector, are presented in Table 2. Importantly, if treatment facilities exist or are planned (such as municipal wastewater treatment and manure recycling of industrial livestock farms), a centralized treatment rate and nutrient removal efficiency of adopted processes should also be considered.

Table 2 Accounting classes and units of the nutrient loads from the five major pollution sectors

Nutrient sectors/NS	Nutrient classes/NC	Emission units/EU
NS_e: Industrial manufacturing	–	Effluent nutrient load (L)
NS_l: Livestock breeding	Cattle (NC^l_1)	Animal excreta (EU^l_0)
	Pig (NC^l_2)	
	Sheep (NC^l_3)	
	Poultry (NC^l_4)	
NS_a: Crop agriculture	Paddy (NC^a_1)	Chemical fertilizer (EU^a_1)
	Wheat (NC^a_2)	
	Rape (NC^a_3)	Organic manure (EU^a_2)
	Vegetables (NC^a_4)	
		Crop residues (EU^a_3)
NS_h: Household consumption	Urban population (NC^h_1)	Household solid waste (EU^h_1)
	Rural population (NC^h_2)	Household wastewater (EU^h_2)
NS_d: Atmospheric deposition	Areas of water bodies (SA)	Atmospheric deposition (AD)

In areas where crop rotation is widely adopted, it is hard to separate out a single crop type as a nutrients class. In these cases, a suggested alternative is to use major rotation types, instead of the popular crop types, as the nutrient classes for load estimation. To be noted that, the emission coefficients (such as nutrients runoff rate from farmland) usually vary as the nutrient classes category changes

In summary, total nutrient loads can be calculated by summing the loads of all five sectors in the study region, as given by the following:

$$TNL_{(N,P)} = NS_{e(N,P)} + NS_{l(N,P)} + NS_{a(N,P)} + NS_{h(N,P)} + NS_{d(N,P)} \quad (4)$$

Here, total nutrient load into surface waters (TNL) is the sum of nitrogen or phosphorus loss from the industrial manufacturing (NS_e), livestock breeding (NS_l), crop agriculture (NS_a), household consumption (NS_h) and atmospheric deposition (NS_d) sectors.

For surplus nutrient reduction, managerial measures and engineering facilities, also known as BMPs (Cherry and others 2008; Robertson and Vitousek 2009), have been widely discussed. Using a quantified cost-effectiveness analysis, we evaluate the performance of BMPs targeting the pollution sectors. Typically, a BMP is designed to change nutrient discharge or runoff rates into surface waters. This is done by way of improving the removal efficiency of WWTPs, building retention facilities for runoff, or even modifying the deposition rate from the atmosphere. Specifically, the reduction potential [$RP_{(N,P)}$] of a potential BMP for each sector can be calculated by the following equations separately:

$$RP_{e(N,P)} = L_{(N,P)} \times |\Delta RE_{(N,P)}| \quad (5)$$

$$RP_{l(a,h)(N,P)} = \sum_i \left[NC_{i(N,P)} \times \sum_j \left(EU_{j(N,P)} \times |\Delta EC_{j(N,P)}| \right) \right] \quad (6)$$

$$RP_{d(N,P)} = SA \times |\Delta AD| \quad (7)$$

Here, nutrient reduction potential of a BMP [$RP_{e(N,P)}$, $RP_{l(N,P)}$, $RP_{a(N,P)}$, $RP_{h(N,P)}$, or $RP_{d(N,P)}$) refers to the maximum amount of reduced nutrient load, and Δ is the result of subtracting corresponding coefficients with and without a BMP. Since this might be positive or negative value in different situations, the absolute value is used during accounting processes.

As another crucial indicator of a BMP, the average mitigation cost (AC), should cover all expenditures on construction, operation, maintenance, and management. Here, we directly obtained the AC of BMPs from governmental documents, technical guides, interview data, or literature sources.

Data Collection

Statistical data were largely obtained from regional or local government documents, including statistical yearbooks, industrial statistics, census data, and technical guides. For example, regional effluent nutrient load into surface water was from the pollution source census (JSPSCO 2008). Centralized wastewater treatment rates in urban areas (88.9, 77.4 and 86.2 % for Suzhou, Wuxi and Changzhou, respectively), amounts of NC_s of each sector, and aquatic surface areas of Tai Lake and its inflow rivers are found in local statistical yearbooks (CZSB 2009; JSSB 2010; SZSB 2009; WXSB 2009).

As another category of key parameters, emission units have been set to correspond to the emission coefficient (EC_s) and efficiency of treatment facilities. Literature reviews, along with expert knowledge and field surveys, are the main sources for coefficients and parameters. For example, annual atmospheric nutrient deposition rates (summing those of dry and wet deposition) were 2,763 kg N km^{-2} and 70 kg P km^{-2}, adapted from Yang and others (2007) and Zhai and others (2009). Discharge rates of livestock farm excreta into surface waters were set to 3.6 % for family-scale farms and 2.8 % for industrial-scale farms (Chen and others 2008; Li and others 2010). Median values were assumed in cases of reported ranges or multiple sources. Detailed values and sources of data and parameters in load estimation are listed in Table 3.

Interview data was also used as a supplementary source of parameters. We conducted interviews with officials and technicians of administrative authorities. For example, according to local experienced agricultural technicians, the ratio of organic manure applied to farmland over total excreta produced by livestock excreta was set to 75 % (family-scale farms) and 50 % (industrial-scale farms), while that of human excreta was set to 100 %.

Results and Discussion

Annual Nutrient Release

Based on the framework described, nutrient loads of Tai Lake Basin can be systematically calculated. Figure 4 depicts nutrient loads into surface waters and their composition in 2008. Total nutrient loads into aquatic systems in 2008 were estimated at 33043.2 tons N a^{-1} and 5254.4 tons P a^{-1}. Annual area-specific nutrient loads, which are defined as loads per watershed area, were 1.94 tons N km^{-2} and 0.31 tons P km^{-2}, respectively. The values agree well with those of Lai and others (2006), Chen and others (2003), and Geng and others (2005), who adopted modeling or monitoring methods for load estimation. Furthermore, household consumption was found to be the major sector with greatest impact on surface waters, contributing 46 and 47 % to the nitrogen and phosphorus loads, respectively. Atmospheric deposition (6246.1 tons a^{-1} or 18 %) and crop agriculture (5045.6 tons a^{-1} or 15 %) sectors also represented significant proportions of the nitrogen load, whereas the livestock breeding sector was responsible for the second

Table 3 Values and sources of data and parameters in the nutrient load accounting process

Sector	Parameter	Description	Value	Source	Result (tons a^{-1})
NS_e	L	Amount of effluent nutrient load into WWTPs (tons a^{-1})	$L_N = 11272.0$ $L_P = 4494.4$	JSPSCO (2008), JSSB (2010)	$NS_{e(N)} = 2175.5$ $NS_{e(P)} = 404.5$
	RE^e	Treatment removal efficiency	$RE^e_N = 80.7\ \%$ $RE^e_P = 91.0\ \%$	JSPSCO (2008)	
NS_l	NC^l_i	Quantity of each livestock category (10^4 head a^{-1})[a]	$NC^l_{1F} = 4.7$ $NC^l_{1I} = 1.25$ $NC^l_{2F} = 71.5$ $NC^l_{2I} = 397.3$ $NC^l_{3F} = 18.7$ $NC^l_{3I} = 26.7$ $NC^l_{4F} = 2930.7$ $NC^l_{4I} = 8821.5$	CZSB (2009), SZSB (2009), WXSB (2009)	$NS_{l(N)F} = 1240.7$ $NS_{l(N)I} = 3267.0$ $NS_{l(N)} = 4507.7$ $NS_{l(P)F} = 468.3$ $NS_{l(P)I} = 1214.9$ $NS_{l(P)} = 1683.2$
	EU^l_{i0}	Nitrogen (N) and phosphorus (P) amount in excreta of each livestock category (kg head^{-1})	$EU^l_{10N} = 48.79$ $EU^l_{10P} = 9.84$ $EU^l_{20N} = 11.51$ $EU^l_{20P} = 3.20$ $EU^l_{30N} = 5.75$ $EU^l_{30P} = 1.06$ $EU^l_{40N} = 0.78$ $EU^l_{40P} = 0.34$	Wu (2005), Chen and others (2008), Li and others (2010)	
	EC^l	Discharge rate of excreta into surface waters	$EC^l_{F(N)} = 3.6\ \%$ $EC^l_{F(P)} = 3.6\ \%$ $EC^l_{I(N)} = 2.8\ \%$ $EC^l_{I(P)} = 2.8\ \%$	Chen and others (2008), Li and others (2010)	
NS_a	NC^a_i	Arable land area of each rotation type (10^3 ha)[b]	$NC^a_1 = 187.6$ $NC^a_2 = 52.5$ $NC^a_3 = 138.6$	CZSB (2009), SZSB (2009), WXSB (2009)	$NS_{a(N)} = 5045.6$ $NS_{a(P)} = 506.7$
	EU^a_{i1}	Nitrogen (N) and phosphorus (P) amount from chemical fertilizer application into farmland (kg ha^{-1} a^{-1})	$EU^a_{11N} = 589.9$ $EU^a_{11P} = 58.5$ $EU^a_{21N} = 522.9$ $EU^a_{21P} = 36.5$ $EU^a_{31N} = 1044.6$ $EU^a_{31P} = 356.1$	Xu (2005), JSPSCO (2008), Wang and others (2009a), Li and others (2010)	
	EU^a_{i2}	Nitrogen (N) and phosphorus (P) amount from chemical organic manure application into farmland (kg ha^{-1} a^{-1})	$EU^a_{i2N} = 325.9$ $EU^a_{i2P} = 104.7$	Xu (2005), Wang (2007), Chen and others (2008), Li and others (2010)	
	EU^a_{i3}	Nitrogen (N) and phosphorus (P) amount from crop residue application into farmland (kg ha^{-1} a^{-1})[c]	$EU^a_{13N} = 83.8$ $EU^a_{13P} = 15.8$ $EU^a_{23N} = 127.5$ $EU^a_{23P} = 19.4$	Zhuang and others (2002), Xu (2005), Li and others (2010)	
	EC^a_{ij}	Runoff rate of each additives applied to farmland (into surface waters)	$EC^a_{11N} = 0.88\ \%$ $EC^a_{11P} = 0.18\ \%$ $EC^a_{21N} = 1.12\ \%$ $EC^a_{21P} = 0.28\ \%$ $EC^a_{31N} = 1.46\ \%$ $EC^a_{31P} = 0.87\ \%$ $EC^a_{i2N} = 0.89\ \%$ $EC^a_{i2P} = 0.04\ \%$ $EC^a_{i3N} = 0.37\ \%$ $EC^a_{i3P} = 0.80\ \%$	Chen and others (2008), Li and others (2010)	

Table 3 continued

Sector	Parameter	Description	Value	Source	Result (tons a^{-1})
NS_h	NC_i^h	Urban and rural population (10^4 a^{-1})	$NC_1^h = 985.24$ $NC_2^h = 467.45$	CZSB (2009), SZSB (2009), WXSB (2009)	$NS_{h(N)U} = 9177.6^d$ $NS_{h(N)R} = 5890.7^d$ $NS_{h(N)} = 15068.3$ $NS_{h(P)U} = 1435.2^d$ $NS_{h(P)R} = 1066.0^d$ $NS_{h(P)} = 2501.2$
	EU_{ij}^h	Nitrogen (N) and phosphorus (P) amount in household solid waste and wastewater (kg per capita)	$EU_{11N}^h = 1.00$ $EU_{11P}^h = 0.40$ $EU_{12N}^h = 6.80$ $EU_{12P}^h = 1.18$ $EU_{21N}^h = 1.00$ $EU_{21P}^h = 0.30$ $EU_{22N}^h = 5.19$ $EU_{22P}^h = 1.04$	Chen and others (2008), JSPSCO (2008), Wang and others (2009a), Li and others (2010), Yi and others (2010)	
	β_{ij}^h	Centralized solid waste and wastewater treatment rate of residences	$\beta_{11}^h = 100, 100, 100\ \%$ $\beta_{12}^h = 88.9, 77.4, 86.2\ \%$ $\beta_{21}^h = 0, 0, 0$ $\beta_{22}^h = 0, 0, 0$	CZSB (2009), SZSB (2009), WXSB (2009)	
	RE_j^h	Treatment removal efficiency of landfills and WWTPs	$RE_{1N}^h = 100\ \%$ $RE_{1P}^h = 100\ \%$ $RE_{2N}^h = 83.7\ \%$ $RE_{2P}^h = 91.0\ \%$	Liu and others (2004), JSSB (2010)	
	EC_{ij}^h	Discharge coefficient into surface waters	$EC_{11N}^h = 0, 0^e$ $EC_{11P}^h = 0, 0^e$ $EC_{12N}^h = 100, 30.0\ \%^e$ $EC_{12P}^h = 100, 30.0\ \%^e$ $EC_{21N}^h = 19.0\ \%$ $EC_{21P}^h = 19.0\ \%$ $EC_{22N}^h = 16.2\ \%$ $EC_{22P}^h = 12.7\ \%$	Chen and others (2008), JSPSCO (2008), Wang and others (2009b), Li and others (2010)	
NS_d	SA	Aquatic surface areas of the Tai Lake and its inflow rivers (km^2)	SA = 2260.4	JSSB (2010)	$NS_{d(N)} = 6246.1$ $NS_{d(P)} = 158.8$
	AD	Annual atmospheric nutrient deposition rate (kg km^{-2} a^{-1})	$AD_N = 2763.0$ $AD_P = 70.0$	Yang and others (2007), Zhai and others (2009)	
TNL	–	–	–	–	$TNL_{(N)} = 33043.2$ $TNL_{(P)} = 5254.4$

[a] In livestock breeding sector, family scale and industrial scale farms are discussed separately. Therefore, in this table, we add letter "F" in subscripts to refer to values or results of family scale farms, while letter "I" refers to that of industrial scale farms

[b] During field survey, we found that rotation is widely adopted in the Tai Lake Basin. Considering this, crop type is substituted by rotation type as nutrient classes. We use NC_1, NC_2, and NC_3 to represent three major crop planting types, that is paddy–wheat rotation, paddy–rape rotation, and vegetable planting only, respectively

[c] Vegetable is excluded since its residue amount is too small to be considered (Smil 2000; Li and others 2010). Only farmland adopting paddy–wheat rotation and paddy–rape rotation is discussed here

[d] In the household consumption sector, urban and rural populations are discussed separately. Therefore, in this table, we add letter "U" in subscripts to refer to results of urban household consumption, while letter "R" refers to that of rural household consumption

[e] Even within urban areas according to administrative division, there is still some city outskirts that are not covered by sewage pipeline network. In this study, nutrient release of these areas is included in "rural household consumption". Here in Table 3, we list the values sequentially as urban areas with pipeline network and non-pipeline area, respectively

largest phosphorus load (1683.2 tons a^{-1} or 32 % in P load). With implementation of the Total Emission Control Policy, which required a 12 % average reduction of the major pollutants every five years after 1996, the industrial manufacturing sector gradually became a less dominant contributor, with less than 10 % proportions of both total N and P release.

For the livestock breeding and household consumption sectors, the load is directly linked with the quantity of emission units, such as livestock numbers or size of residential population. Although there were treatment facilities on most industrial-scale farms, the tremendous numbers of animals raised created much more pollution than that from

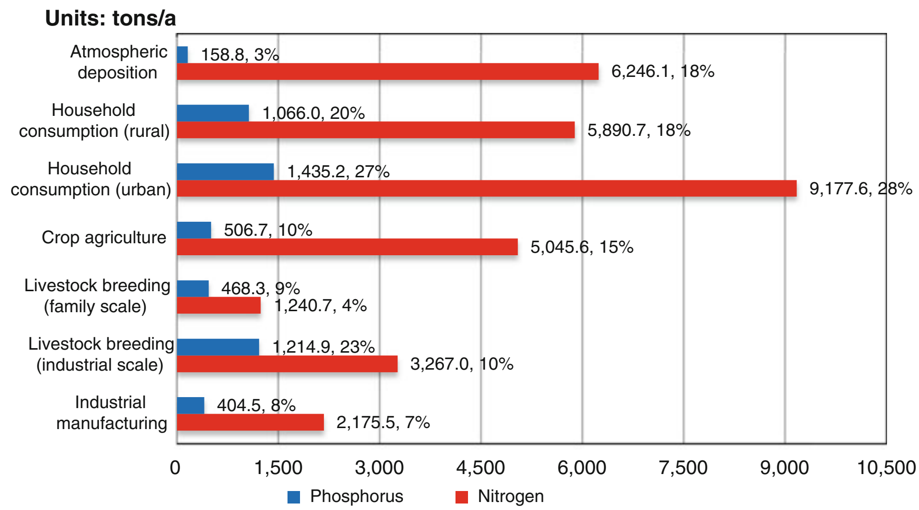

Fig. 4 Nitrogen and phosphorus loads and the distribution among various sectors (tons a^{-1})

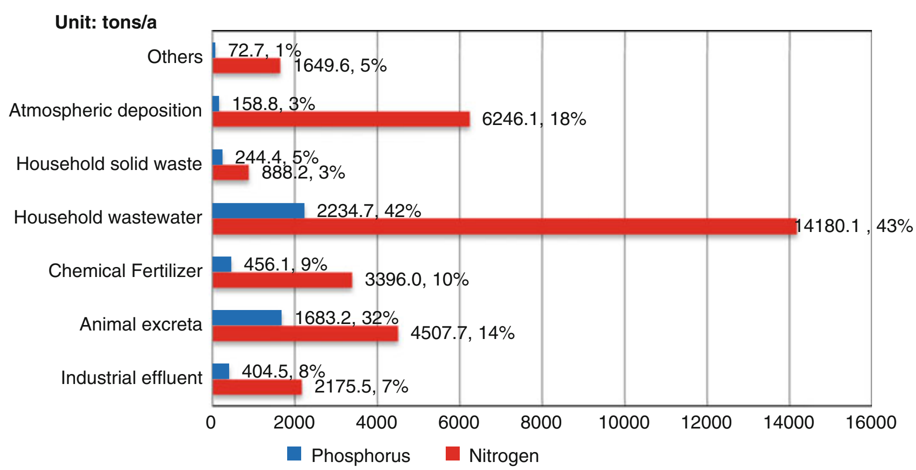

Fig. 5 Composition of nitrogen and phosphorus loads among various sources (tons a^{-1})

family-scale farms (10 vs. 4 % in N load, 23 vs. 9 % in P load). The population distribution between urban and rural areas has a similar effect. The urban population consists primarily of residents, and there is a higher level of infrastructure for waste treatment. As a result, urban and rural household consumption contributed nearly the same amount of nutrients.

Source Identification

A composition analysis of nutrient release further highlights the influence of various sources or activities in Tai Lake Basin. Figure 5 shows that nitrogen and phosphorus release generally shared a similar distribution trend among various sources. The results emphasize that household wastewater discharge (excreta included) was the most significant contributor of nutrient loads in the region (43 % in N load and 42 % in P load). Because human excreta were treated together with urban sewage, the large population size and resultant high nutrient content in the sewage unsurprisingly led to this result. However, because of a low rate of runoff into water bodies, solid waste disposal was not as important as sewage discharge, and accounted for only 4 % in N load and 5 % in P load.

Animal excreta were the second largest nutrient source, especially for phosphorus runoff into surface waters (1683.2 tons a^{-1} or 32 % in total P load). Industrial and family-scale farms had nearly equal contributions, although the former accounted for slightly more because of a greater number of livestock. In the crop agriculture sector, chemical fertilizer application was responsible for 10 and 9 % of total nitrogen and phosphorus release, respectively, exerting a much greater impact on surface waters than applied organic manure and other additives. Because of tough emission standards in the region, industrial effluent contributed approximately 8 % to total N and P load. However, dry and wet atmospheric deposition jointly contributed significant nutrient inputs, especially for nitrogen, which was 6246.1 tons a^{-1} or 18 % in total.

Table 4 Comparison of accounting results with reported previous studies (annual values)

Comparison item	Study area	Nitrogen	Phosphorus	Sources
Contribution of nonpoint sources	Tai Lake	55 %	42 %	This study
		Approximately 56 %	Approximately 85 %	Qin and others (2007)
	Dianchi Lake (in China)	–	Approximately 64 %	Liu and others (2004)
	Egirdir Lake (in Turkey)	71 %	72 %	Gunes (2008)
	The U.S. average	82 %	84 %	Carpenter and others (1998)
Nutrient load per watershed area	Tai Lake	1.94 t/km^2	0.31 t/km^2	This study
		1.08 t/km^2	0.54 t/km^2	Lai and others (2006)
	Egirdir Lake (in Turkey)	3.24 t/km^2	0.32 t/km^2	Gunes (2008)
Household consumption release per capita	Tai Lake	1.04 kg	0.17 kg	This study
		1.11 kg	–	Lai and others (2006)
	Egirdir Lake (in Turkey)	1.07 kg	0.16 kg	Gunes (2008)
Crop agriculture release per farmland area	Tai Lake	13.32 kg/ha	1.34 kg/ha	This study
		34.10 kg/ha	1.75 kg/ha	Guo and others (2004)
	China average	–	14.7 kg/ha	Chen and others (2008)

Comparison of Results

The findings agree with other studies of Tai Lake Basin (Liu and others 2004; Qin and others 2007), as well as studies in Europe and the United States (Carpenter and others 1998; Gunes 2008). Urban household consumption and industrial-scale livestock breeding sectors were treated as point sources, whereas only industrial discharge was considered in other studies. As shown in Table 4, accounting results agree with this classification since the share of nonpoint sources appeared smaller, for phosphorus in particular (42 % in the total). For nutrient load intensities, defined as loads per watershed area, the results conformed to the estimate of Lai and others (2006) for the entire Tai Lake Basin. Compared with the 3.24 t km^{-2} of Egirdir Lake Basin in Turkey (Gunes 2008), the estimated nitrogen intensity of 1.94 t km^{-2} was dramatically lower. These inconsistences might be explained by large temporal span, climate variation, and differences of hydrographic conditions, landscape composition, and leading industries.

Urban and rural households jointly contributed 46 % in N load and 47 % in P load, which approach values from previous studies of the same basin (Qin and others 2007; Zhang and others 2008). Phosphorus release per capita was slightly higher than that observed in other studies. This was most likely because we considered not only household sewage (excreta included) but also solid waste emissions in the accounting process.

Meanwhile, the crop agriculture sector accounted for 15 % of N loss and 10 % of P loss, lower than the average value of 30 % in other basins (Drolc and Koncan 2002; Gunes 2008; Jaworski and others 1992). The relatively small contribution of agricultural production (both farming and breeding) in the regional economy most likely explains this difference (CZSB 2009; SZSB 2009; WXSB 2009), and might also be responsible for a lower area-specific nutrient release from farmland than the China average. While the livestock breeding sector contributed only 14 % of total nitrogen load, the same as that of the crop agriculture sector, this was also much lower than the results of previous studies (Chen and others 2010; Zhao and others 2011).

Performance of BMPs

All six BMPs are targeted on a nutrient contributor and should be directly effective. Detailed BMP information and changed values of parameters and results of RP and AC are listed in Table 5. As shown in Fig. 6, there was no BMP that satisfactorily balanced cost and reduction effect. Overall, biogas digester construction for industrial-scale farms reduced the greatest amount of N (1633.5 tons a^{-1} or 4.9 %) and P (607.5 tons a^{-1} or 11.6 %) at a relatively acceptable average cost (1.11 $\times$ 10^4 CNY/ton N a^{-1} and 5.91 $\times$ 10^4 CNY/ton P a^{-1}), thus making it a very satisfying alternative for the basin. On the other hand, though construction of rural wastewater treatment networks has the largest nitrogen reduction potential, the high average cost prevented its widespread application.

Building decentralized rural wastewater treatment facilities had certain mitigation effects (particularly for N), and the low financial requirement favors this when cost is the constraining factor. When facing strict budget pressures, this approach might be preferred owing to the moderate average cost (0.72 $\times$ 10^4 CNY/ton N a^{-1} and 4.80 $\times$ 10^4 CNY/ton P a^{-1}). Nonetheless, comparatively weak phosphorus mitigation performance may minimize

Table 5 Description, coefficient values and performance of BMPs

BMPs	Targeting sector	Description	Changed values	RP (tons a^{-1})	AC (10^4 CNY/tons a^{-1})[a]
Increasing removal efficiency of industrial effluent nutrients	Industrial manufacturing	The current process of industrial effluent treatment in WWTPs is a modified anaerobic–anoxic–oxic (A^2/O). If the A^2/O is replaced by a OCO process, the removal efficiency of nutrients will increase, according to He and Yang (2003). Cost mainly comes from construction and maintenance process	$RE^e_N = 83.0\ \%$ $RE^e_P = 93.0\ \%$	N 259.3 (0.8 %) P 89.9 (1.7 %)	0.53 5.77
Improvement of wastewater centralized rate in urban areas	Household consumption	Improve the wastewater treatment network in urban outskirts area, including WWTPs and pipelines, which will lead to a 100 % wastewater centralized rate in urban areas. Cost mainly comes from construction and maintenance process	$\beta^h_{12} = 100, 100, 100\ \%$	N 1237.9 (3.7 %) P 241.5 (4.6 %)	1.23 7.02
Rural wastewater treatment network construction	Household consumption	Construct a wastewater treatment network in rural areas, including WWTPs and pipelines. Centralized rate, removal efficiency, and discharge coefficient of household wastewater in rural areas will change. Cost mainly comes from construction and maintenance process	$\beta^k_{22} = 44.5, 38.7, 43.1\ \%$ $RE^h_{2N} = 83.7\ \%$ $RE^h_{2N} = 91.0\ \%$ $EC^h_{22N} = 100\ \%$ $EC^h_{22P} = 100\ \%$	N 1772.4 (5.4 %) P 281.5 (5.4 %)	1.23[b] 7.02[b]
Rural decentralized wastewater treatment facility construction	Household consumption	Wetland system is one typical decentralized wastewater treatment facility, which gains pilot constructions in Jiangsu province. According to field interview, this facilities will account for maximized half the rural residences under status quo. Discharge coefficient of rural runoffs into surface waters will change. Cost mainly comes from construction and maintenance process	$EC^h_{22N} = 6.7\ \%$[c] $EC^h_{22P} = 4.4\ \%$[c]	N 1411.7 (4.3 %) P 286.5 (5.5 %)	0.72 4.80
Industrial farms excreta biogas digesters construction	Livestock breeding	Livestock biogas digesters for industrial scale farms could store and digest livestock excreta into biogas, which leads to a 100 % nutrients reduction (Liu and Chen 2005). According to literature review, only maximized half the industrial scale farms would construct digesters under status quo. Cost mainly comes from construction and maintenance process	$EC^l_{I(N)} = 0$ $EC^l_{I(P)} = 0$	N 1633.5 (4.9 %) P 607.5 (11.6 %)	1.11 5.91
Limited use of chemical fertilizer into farmland	Crop agriculture	Regulations to limit the use of chemical fertilizers have been widely promulgated in the Tai Lake Basin. The crop nutrient requirements would be supplied by organic manure application. Runoff rate of additives will change (Chen and others 2008; Li and others 2010). Cost mainly comes from subsidies to stimulate peasant to use organic manure	$EC^a_{iIN} = 0.89\ \%$ $EC^a_{iIP} = 0.04\ \%$	N 878.3 (2.7 %) P 431.2 (8.2 %)	0.85 5.19

[a] Values of BMPs average mitigation costs (AC) are mainly directly obtained from governmental documents, technical guides, interview data, as well as papers of Chen (2010), Zhou (2001), Liu and Chen (2005) and Zhang (2008). All monetary cost has been adjusted to the prices of calendar year 2008

[b] Same as the situation in urban outskirts without pipeline network, the BMP named "Rural wastewater networks construction" also requires construction of both pipeline and treatment plans. Therefore, these two BMPs share the same average nutrients reduction cost

[c] On average, 58.5 % of nitrogen (Kivaisi 2001; Schulz and Peall 2001) and 65.0 % (Yan and others 1998; Kivaisi 2001) of phosphorus can be removed from runoffs, and this leads to the change of discharge coefficient of rural runoffs

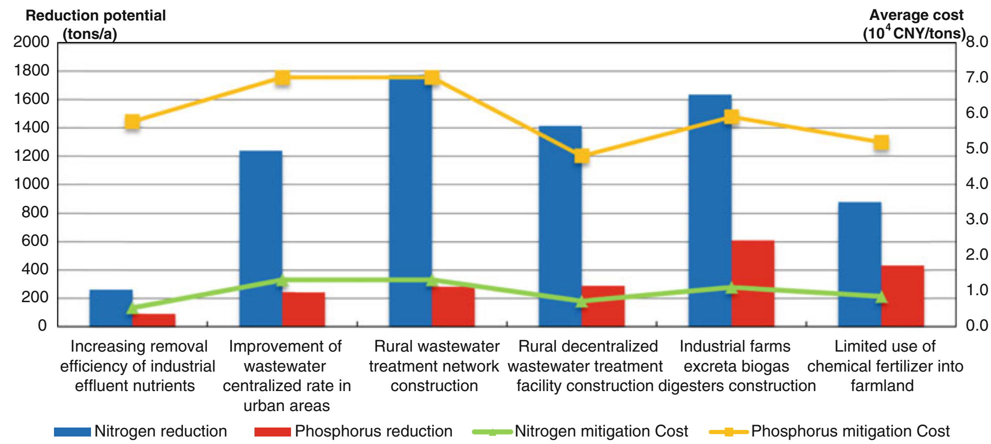

Fig. 6 Comparison of nutrients reduction performance of BMPs (annual values)

the total effect, particularly when nitrogen is not the limiting nutrient. Moreover, as a widely implemented BMP, limiting the use of chemical fertilizer was overshadowed because of low mitigation capacity, especially for N (878.3 tons a^{-1}).

On the other hand, some already well-managed nutrient sectors or sources might fall short in nutrient reduction potential. Although enhancing industrial effluent treatment had a low monetary requirement for process innovation, deficiency of its reduction potential made this option not very favorable (259.3 tons a^{-1} or 0.8 % for N and 89.9 tons a^{-1} or 1.7 % for P) for watershed nutrient control. The same situation also applies to the BMP of improving the centralized urban wastewater treatment rate in city outskirts. This approach reduced nutrients less effectively than rural treatment network construction, but retained a monetary requirement equal to the average mitigation cost (1.23×10^4 CNY/ton N a^{-1} and 7.02×10^4 CNY/ton P a^{-1}).

Overall, BMPs with high reduction potential most likely require large monetary investment. Construction of rural treatment networks is an example. However, limited expenditure cannot usually ensure sufficient mitigation effect. However, in cases where phosphorus pressure was minor, building rural decentralized wastewater treatment facilities could be a good choice. Since BMPs mitigate nitrogen and phosphorus with different removal efficiencies, it is important to choose the more pressing indicator of the two eutrophication factors. Considering that point sources in Tai Lake Basin have been somewhat well controlled in recent years, typical diffuse sources appear to have more reduction potential than industrial effluents. This should attract more attention in watershed management.

Conclusions

We proposed an empirical accounting framework to estimate nitrogen and phosphorus loads from five major sectors in Tai Lake Basin. Total nutrient loads were estimated at 33043.2 tons N a^{-1} and 5254.4 tons P a^{-1} in 2008, and annual area-specific nutrient loads were 1.94 tons N km^{-2} and 0.31 tons P km^{-2}. Among the five major sources addressed, the household consumption sector was found to be the major contributor with greatest impact on surface water (46 % in N load and 47 % in P load), whereas household wastewater discharge was the major emission source. Atmospheric deposition and animal excreta loss from livestock farms also contributed a significant share of nitrogen and phosphorus, respectively. Our accounting method uses easily accessible data largely obtained from statistical databases and open publications, and provides a complete picture of nutrient pollution at the watershed level. It may be used to support policy making, which promotes its wide application to nutrient load estimation worldwide.

Considering reduction potential and average monetary cost, six BMPs implemented or under design in Jiangsu Province were selected for evaluation, and targeting proposals were discussed. Overall, biogas digester construction on industrial-scale farms was proven the best alternative for the Tai Lake Basin, whereas the building of rural decentralized wastewater treatment facilities would be a good choice under tight budget restrictions. Compromise is inevitable when facing realistic problems. During the decision-making process, reduction potential, average monetary costs, prior eutrophication indicators and other factors, such as the risk tolerance of policy makers, should all be considered.

It is possible to obtain a reasonable load estimate on a microscale (e.g., onsite soil, field, plot and farm). However, this cannot realistically be achieved on a macroscale such as in a lake basin (Chen and others 2008), which would be subject to estimation uncertainties. One of the causes of these uncertainties is exclusion of minor nutrient sources such as fishponds (or other aquaculture) or natural soil storage (Smil 2000), and the use of statistically median values may also have produced uncertainty in the nutrient load estimation. Moreover, since this work mainly addressed load estimation and source identification, it does not explicitly account for the effects of hydrology, and average BMP costs were mostly obtained directly instead of calculated. These are important issues to tackle in future work.

Acknowledgment This research was supported by the National Science Foundation of China (Grant No. 70903030).

References

Arnold JG, Srinivasan R, Muttiah RS, Williams JR (1998) Large area hydrologic modeling and assessment, part I: model development. JAWRA 34(1):73–88

Beasley DB, Huggins LF, Monke EJ (1980) ANSWERS: a model for watershed planning. Trans ASAE 23:938–944

Broad ST, Corkrey R (2011) Estimating annual generation rates of total P and total N for different land uses in Tasmania, Australia. J Environ Manag 92(6):1609–1617

Carpenter SR, Caraco NF, Correll DL, Howarth RW, Sharpley AN, Smith VH (1998) Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecol Appl 8(3):559–568

Chen M, Chen J, Sun F (2008) Agricultural phosphorus flow and its environmental impacts in China. Sci Total Environ 405(1–3):140–152

Chen M, Chen J, Sun F (2010) Estimating nutrient releases from agriculture in China: an extended substance flow analysis framework and a modeling tool. Sci Total Environ 408(21):5123–5136

Chen Y (2010) Cost benefit analysis of nutrients pollution control in household consumption sector: a case of Changzhou, China (China). Nanjing University, Nanjing

Chen Y, Fan C, Teubner K, Dokulil M (2003) Changes of nutrients and phytoplankton chlorophyll-*a* in a large shallow lake, Taihu, China: an 8-year investigation. Hydrobiologia 506–509(1):273–279

Cherry KA, Shepherd M, Withers PJA, Mooney SJ (2008) Assessing the effectiveness of actions to mitigate nutrient loss from agriculture: a review of methods. Sci Total Environ 406(1–2):1–23

CZSB (2009) Changzhou municipal statistical yearbook (China). China Statistics Press, Changzhou

Ding XW, Shen ZY, Hong Q, Yang ZF, Wu X, Liu RM (2010) Development and test of the export coefficient model in the upper reach of the Yangtze River. J Hydrol 383(3–4):233–244

Donigian AS, Bicknell BR, Imhoff JC (1995) Hydrological simulation program—Fortran (HSPF). Computer models of watershed hydrology. WRP, Highlands Ranch

Drolc A, Koncan JZ (2002) Estimation of sources of total phosphorus in a river basin and assessment of alternatives for river pollution reduction. Environ Int 28(5):393–400

Ellis EC, Wang SM (1997) Sustainable traditional agriculture in the Tai Lake region of China. Agric Ecosyst Environ 61(2–3):177–193

Geng J, Niu X, Jin X, Wang X, Gu X, Edwards M, Glindemann D (2005) Simultaneous monitoring of phosphine and of phosphorus species in Taihu Lake sediments and phosphine emission from lake sediments. Biogeochemistry 76(2):283–298

Gunes K (2008) Point and nonpoint sources of nutrients to lakes—ecotechnological measures and mitigation methodologies—case study. Ecol Eng 34(2):116–126

Guo H, Zhu J, Wang X, Wu Z, Zhang Z (2004) Case study on nitrogen and phosphorus emissions from paddy field in Taihu region. Environ Geochem Health 26(2):209–219

He W, Yang H (2003) Analysis on processes of nitrogen and phosphorus removal for municipal sewage. J Huazhong Univ Sci Technol (Nat Sci Ed) 20(1):85–87

Hong B, Swaney DP, Howarth RW (2010) A toolbox for calculating net anthropogenic nitrogen inputs (NANI). Environ Model Softw 26(5):623–633

Jaworski NA, Groffman PM, Keller AA, Prager JC (1992) A watershed nitrogen and phosphorus balance—the upper Potomac River basin. Estuaries 15(1):83–95

Johnes PJ (1996) Evaluation and management of the impact of land use change on the nitrogen and phosphorus load delivered to surface waters: the export coefficient modelling approach. J Hydrol 183(3–4):323–349

JSEPD (2009) Environment quality bulletin of Jiangsu province in 2009 (China). http://www.jshb.gov.cn/jshbw/hbzl/ndhjzkgb/201006/t20100622_156746.html. Accessed 5 Dec 2010

JSPSCO (2008) Pollution sources census of Jiangsu province (China). China Statistics Press, Nanjing

JSSB (2010) Statistical yearbook of Jiangsu province (China). China Statistics Press, Nanjing

Kaushal SS, Groffman PM, Band LE, Elliott EM, Shields CA, Kendall C (2011) Tracking nonpoint source nitrogen pollution in human-impacted watersheds. Environ Sci Technol 45:8225–8232

Keller J, Subramaniam K, Gösswein J, Greenfield PF (1997) Nutrient removal from industrial wastewater using single tank sequencing batch reactors. Water Sci Technol 35(6):137–144

Kennedy C, Cuddihy J, Engel-Yan J (2007) The changing metabolism of cities. J Ind Ecol 11(2):43–59

Kivaisi AK (2001) The potential for constructed wetlands for wastewater treatment and reuse in developing countries: a review. Ecol Eng 16(4):545–560

Lai G, Yu G, Gui F (2006) Preliminary study on assessment of nutrient transport in the Taihu basin based on SWAT modeling. Sci China Ser D Earth Sci 49(supp 1):135–145

Li S, Yuan Z, Bi J, Wu H (2010) Anthropogenic phosphorus flow analysis of Hefei city, China. Sci Total Environ 408(23):5715–5722

Li Z, Yang G, Li H (2009) Estimated nutrient export loads based on improved export coefficient model in Xitiaoxi Watershed (China). Environ Sci 30(3):668–672

Liu J, Chen Y (2005) Biogas technology for control and prevention of multiple nonpoint source pollution in agriculture (China). China Biogas 23(4):40–42

Liu J, Diamond J (2005) China's environment in a globalizing world: how China and the rest of the world affect each other. Nature 435(7046):1179–1186

Liu W, Qiu R (2007) Water eutrophication in China and the combating strategies. J Chem Technol Biotechnol 82(9):781–786

Liu Y, Chen JN, Mol APJ (2004) Evaluation of phosphorus flows in the Dianchi watershed, Southwest of China. Popul Environ 25(6):637–656

Nixon SW (1995) Coastal marine eutrophication: a definition, social causes, and future concerns. Ophelia 41(1):199–219

Paerl HW, Xu H, McCarthy MJ, Zhu G, Qin B, Li Y, Gardner WS (2011) Controlling harmful cyanobacterial blooms in a hyper-eutrophic lake (Lake Taihu, China): the need for a dual nutrient (N & P) management strategy. Water Res 45(5):1973–1983

Parsons JE, Thomas DL, Huffman RL (2001) Agricultural non-point source water quality models: their use and application. Southern Cooperative Series Bulletin #398, SAAESD

Puckett LJ (1994) Nonpoint and point sources of nitrogen in major watersheds of the United States. Water-resources investigations report 94-4001. U.S. Geological Survey, Reston, p 9

Qin B, Xu P, Wu Q, Luo L, Zhang Y (2007) Environmental issues of Lake Taihu, China. Hydrobiologia 581:12–14

Qin B, Zhu G, Gao G, Zhang Y, Li W, Paerl HW, Carmichael WW (2010) A drinking water crisis in Lake Taihu, China: linkage to climatic variability and lake management. Environ Manag 45:105–112

Qin B, Zhu G, Zhang L, Luo L, Gao G, Gu B (2006) Estimation of internal nutrient release in large shallow Lake Taihu, China. Sci China Ser D Earth Sci 49(0):38–50

Robertson GP, Vitousek PM (2009) Nitrogen in agriculture: balancing the cost of an essential resource. Annu Rev Environ Resour 34(1):97–125

Schindler DW, Vallentyne JR (2008) The Algal Bowl: overfertilization of the world's freshwaters and estuaries. University of Alberta Press, Edmonton

Schulz R, Peall SKC (2001) Effectiveness of a constructed wetland for retention of nonpoint-source pesticide pollution in the Lourens River catchment, South Africa. Environ Sci Technol 35(2):422–426

Shen Z, Liao Q, Hong Q, Gong Y (2011) An overview of research on agricultural non-point source pollution modelling in China. Sep Purif Technol 84:104–111

Sloto RA, Crouse MY (1996) HYSEP: a computer program for stream flow hydrograph separation and analysis. Water-Resources Investigations Report 96-4040. U.S. Geological Survey, Lemoyne, p 53

Smil V (2000) Phosphorus in the environment: natural flows and human interferences. Annu Rev Energy Environ 25(1):53–88

SZSB (2009) Suzhou municipal statistics yearbook (China). China Statistics Press, Suzhou

Vitousek PM, Mooney HA, Lubchenco J, Melillo JM (1997) Human domination of earth's ecosystems. Science 277(5325):494–499

Vollenweider RA (1982) Eutrophication of waters: monitoring, assessment and control. OECD, Paris

Wang H, Xi Y, Chen R, Xu X, Wei Q, Li J (2009a) Research of fertilizer and pesticides excessive application in Taihu Lake region (China). Agro Environ Dev 2009(26):3

Wang J, Xu Z, Peng X, Chen Z, Feng L, Li M (2009b) Experimental investigation of pollutants in human excreta (China). Res Environ Sci 22:1098–1102

Wang T (2007) Study on nitrogen and phosphorus losses from field used manure in Dianchi watershed by simulated rainfall method (China). The Graduate School of Chinese Academy of Agricultural Sciences, Beijing

Whitehead PG, Wilson EJ, Butterfield D (1998) A semi-distributed integrated nitrogen model for multiple source assessment in catchments (INCA): part I—model structure and process equations. Sci Total Environ 210(211):547–558

Wu S (2005) The spatial and temporal change of nitrogen and phosphorus produced by livestock and poultry & their effects on agricultural non-point pollution in China (China). Agricultural Resources and Regional Planning Institute, The Graduate School of Chinese Academy of Agricultural Sciences, Beijing

WXSB (2009) Wuxi municipal statistical yearbook (China). China Statistics Press, Wuxi

Xu H, Paerl HW, Qin B, Zhu G, Gao G (2010) Nitrogen and phosphorus inputs control phytoplankton growth in eutrophic Lake Taihu, China. Limnol Oceanogr 55(1):420–432

Xu J (2005) Phosphorus cycling and balance in "agriculture–animal husbandry–nutrition–environment" system of China (China). Agricultural University of Hebei, Baoding

Yan W, Yin C, Tang H (1998) Nutrient retention by multipond systems: mechanisms for the control of nonpoint source pollution. J Environ Qual 27(5):1009–1017

Yang L, Qin B, Hu W, Luo L, Song Y (2007) The atmospheric deposition of nitrogen and phosphorus nutrients in Taihu Lake (China). Oceanologia et Limnologia Sinica 38(2):104–110

Yang Q (1996) Algal bloom in Taihu Lake and its control (China). J Lake Sci 8(1):67–74

Yi W, Wang X, Wang A, Zhao H (2010) Discharge index of pollutants from village sewage in Taihu region—a case study in Kunshan (China). J Agro Environ Sci 29(7):1369–1373

Young RA, Onstad CA, Bosch DD, Anderson WP (1989) Agricultural non-point source pollution model for evaluating agricultural watersheds. J Soil Water Conserv 44(2):168–173

Zhai S, Yang L, Hu W (2009) Observations of atmospheric nitrogen and phosphorus deposition during the period of algal bloom formation in northern Lake Taihu, China. Environ Manag 44(3):542–551

Zhang J, Jørgensen SE (2005) Modelling of point and non-point nutrient loadings from a watershed. Environ Model Softw 20(5):561–574

Zhang L, Xia M, Zhang L, Wang C, Lu J (2008) Eutrophication status and control strategy of Taihu Lake. Front Environ Sci Eng China 2(3):280–290

Zhang Q, Chen Y, Jilani G, Shamsi IH, Yu Q (2010) Model AVSWAT apropos of simulating non-point source pollution in Taihu Lake basin. J Hazard Mater 174(1–3):824–830

Zhang Y (2008) Environmental economic assessment on best management practices in Taishitun town of Miyun county, Beijing (China). Capital Normal University, Beijing

Zhao GJ, Hormann G, Fohrer N, Li HP, Gao JF, Tian K (2011) Development and application of a nitrogen simulation model in a data scarce catchment in South China. Agric Water Manag 98(4):619–631

Zhou B (2001) Analysis on the operating cost of urban wastewater treatment plant in east China region (China). China Water Wastewater 17(8):29–30

Zhuang H, Cao W, Lu J (2002) Simulation of nitrogen release from decomposition of straw manure (China). Acta Ecologica Sinica 22(8):1358–1361