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Modelling the cost-effective solutions of nitrogen reduction in Jiulong River Watershed, China

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ABSTRACT

To inform the decision makers the value of taking a watershed approach to managing land-based sources of pollution, this paper presented a systematic framework and models to estimate the minimum cost solutions of nutrient reduction in watershed. The established models considered the spatial heterogeneity of emissions sources, emission impact on the receiving waters, and the abatement cost. Empirical estimates in Jiulong River Watershed of China indicated that a 30% reduction of the nitrogen load to the receiving waters can be achieved by abating about 29% of total nitrogen emission and at a cost of RMB 263 million per year. Reduced applications of fertilizers and livestock holdings were the main abatement measures, and the urban sub-basins the main abatement regions due to their high abatement capacity. It was necessary to specify a target water body located in the middle or upstream of the watershed to capture the local damage of excessive nitrogen emission although this will generates a high cost and a high reduction. Sensitivity analysis indicated that for the same absolute values of load reductions, the emission reductions are more robust against changes in the retention and leakage parameters than that of the total cost. Additionally, changing the parameter values for abatement functions will only affect the total cost.

1. Background

Eutrophication resulting from over enrichment of waters by nutrients threatens and degrades many aquatic ecosystems around the world (Mindy et al., 2008). Nutrient abatement has become one of the great challenges of many countries to improve the water quality and maintain the health of freshwater and coastal ecosystems. Nutrient reduction by different measures implemented in different regions have different impacts on the load to the receiving waters; this is due to the fact that the impact nutrients have on a receiving waters not only depends on the level of emissions but also on the buffering characteristics of the watershed in which they originate. Furthermore, the costs of different abatement measures located in different drainage basins are different. This implies that any kind of uniform policy or equal ratio abatement is inefficient, i.e. the environmental target could be reached at a lower cost. Due to nutrient reduction involving large amounts of investment, an approach considering the spatial heterogeneity of emissions sources, emission impact on the receiving waters, and the abatement cost is therefore necessary in order to achieve an ambient water quality target at least cost.

Similar studies have examined the cost effective solutions of nutrients reduction (Gren et al., 1997; Schleiniger, 1999; Turner et al., 1999; Rob, 2001; Ribaudo, 2001; Veeren and Lorenz, 2003; Elofsson, 2003, 2006; Nunneri et al., 2007; Fröschl, 2008; Gren, 2008). Most of these studies were carried out on a large spatial scale. Parameters used in estimating the impact of different sources were treated as an average for whole drainage basins covering large geographical areas, which might bias the results when dealing with medium and small scale watersheds. Moreover, to our surprise, there are few attempts to analyse and calculate such minimum cost solutions of nutrient reduction in China, where more than half of the lakes, reservoirs and coastal water are under the serious threat of eutrophication (MEP, 2012; SOA, 2012).

It is therefore the purpose of this study to develop a systematic framework and the respondent components of models to estimate the cost-effective solutions of nutrient reduction in the watershed. We try to obtain better estimates of demanded parameters in the models on the lowest possible geographical scale for the study region. For a small watershed, more precise measurements of the

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impact on as small scale as possible imply significant cost reductions from an optimal allocation of abatement measures. The remainder of this paper is organized as follows: section two presents the methods and models to calculate the minimum cost solution of nutrient reduction. The established models are employed in the Jiulong River Watershed, China to estimate the minimum cost solutions of nitrogen reduction in section three. Section four describes the limitations of the study. The conclusions are included in section five.

2. Methods and models

2.1. Cost-minimisation model

Cost-effective nutrient reductions are defined as minimum cost solutions to achieve the pre-specified environmental targets. The minimum cost solutions consist of an optimal mix of measures and an optimal localisation of the measures, so that a specified goal for reductions of loads to one or more receiving waters or target water bodies (TWBs) is obtained at the least abatement cost. It is necessary to divide the concerned watershed into sub-basins to reflect the fact that the emissions from each sub-basin contribute differently to the loads of the TWBs because of the spatial heterogeneity of retention, dilution and de-nitrification during transport to the receiving waters, and the abatement costs among measures located in different sub-basins. The minimization problem is constrained by the exogenously set limits on the potential emission reductions of the measures because of the fact that each measure has a limited feasibility range within each sub-basin. The cost minimization problem can be described by the following static nonlinear programming model:

$$\min_{x_{ik}} \sum_{i=1}^{I} \sum_{k=1}^{K} AC(x_{ik}) \tag{1}$$

s.t.

$$\sum_{i=1}^{I} \sum_{k=1}^{K} g^j(x_{ik}) = T^j \tag{2}$$

$$m(x_{ik}) \le M_i^k max \tag{3}$$

where i (=1, 2, 3, ..., I) is the sub-basin index; k (=1,2,3...K) is the abatement measure index; j (=1,2,3...J) is the TWBs index; x_{ik} stands for the nutrient emission reduction of measure k in sub-basin i; AC (x_{ik}) stands for the abatement cost function of measure k implemented in sub-basin i; g^j (x_{ik}) is the load response model describing the relation between reduced emissions from sub-basin i reaching TWBs j; m (x_{ik}) stands for the function describing the range of measure k implemented for reducing the emissions in sub-basin i; T^j is the environment target in terms of load reduction in TWBs j, and $M_i^k max$ is the maximum emissions reduction of measure k implemented in sub-basin i.

For a solution to above minimization problem, $x_{ik}{}^*$, the necessary conditions for optimality are the Kuhn-Tucker conditions. It is assumed that the total abatement costs are quasi-convex and all separate cost functions for abatement measures are convex. Because functions g^j (x_{ik}) and m (x_{ik}) are linear functions[1] and the objective function is quasi-convex, the Hessian matrix to the Lagrangian is positive semi-definite and a unique optimal solution exists.

The necessary Kuhn-Tucker conditions are given in (4)–(8).

$$\frac{\partial AC\left(x_{ik}^*\right)}{\partial x_{ik}} - \lambda_i \frac{\partial g^j\left(x_{ik}^*\right)}{\partial x_{ik}} - \beta_i \frac{\partial m\left(x_{ik}^*\right)}{\partial x_{ik}} = 0 \tag{4}$$

$$\sum_{i=1}^{I} \sum_{k=1}^{K} g^j\left(x_{ik}^*\right) = T^j \tag{5}$$

$$m\left(x_{ik}^*\right) \le M_i^k max \tag{6}$$

$$\beta_i m\left(x_{ik}^*\right) = 0 \tag{7}$$

$$\beta_i \ge 0\left(= 0 \;\; if \;\; m\left(x_{ik}^*\right) < M_i^k \max\right) \tag{8}$$

The condition in (4) ensures optimality; (5) and (6) are feasibility conditions; (7) is the complementary slackness condition; and (8) is a non-negativity condition. λ_i andβ_i are the Lagrangian multipliers, which represents shadow prices.

There are four components in the above models. (1) cost function of each measure implemented in a specific sub-basin; (2) load response functions of TWBs; (3) environmental target of TWBs in terms of load reduction; and (4) potential emission reductions of measures in each sub-basin, i.e. measure scale limit functions.

2.2. The cost functions

The abatement cost of each measure is a function the scale of the abatement measure in question, $AC = AC(M)$, i.e. the cost for that specific measure to abate nutrient. The emission reduction of a specific measure is a function of this measure's scale, i.e.$x = x(M)$. Then the cost function reflecting the relation between the cost and the reduced nutrient can be written as: $AC = AC(x(M))$.The cost function is assumed to be convex and increasing, implying that marginal abatement cost increases at an increasing rate. That is $AC'_x > 0$, and $AC''_x > 0$. In other words, it becomes successively more expensive for a measure to reduce one more unit of nutrient.

Abatement costs of each measure are represented by the change in welfare economic rent to society caused by implementing the measure. The change in economic rent includes three components. The first is the loss in agricultural economic rent from changing the initial resource use. The second component is the costs of establishing, operating and maintaining the new activity. The third is the possible secondary benefits, such as the reduction in climate gas emissions resulting from establishment of wetlands. Abatement costs might differ between sub-basins, due to, for example, differences in opportunity cost or abatement capacity.

2.3. Load response model

Two types of load response model must be established to estimate the cost effective solutions of nutrient reduction: (1) load response model of emission describing the relationship between nutriment emission from sources and sub-basins, and the load to the TWBs; and (2) load response model of reduction which describing the effect of reduced nutriment emitted from sources and sub-basins, on the reduced load to the TWBs.

1)Load response model of emission

[1] In the model, functions $g^j(x_{ik})$ and $m(x_{ik})$ are linear in x_{ik}; saying that there is a simple linear relationship between emissions from a sub-basin and the nutrient loads to a TWBs. Similarly there is a positive linear effect on nutrient emissions in a sub-basin of implementing a specific abatement measure.

Within a sub-basin, nutrient emitted from sources is up-taken by plants and soil, while a fraction discharges into the basin's stream network. Leached nutrient can thereafter reach a receiving waters in one or several ways of water transportation such as groundwater flow, surface water flow or direct deposition. During water transportation, nutrient is subject to plant assimilation, sedimentation and denitrification, all of which reduce the amount of nitrogen. The retained fraction of nitrogen that does not reach the receiving waters is referred to as retention. Leakage and retention therefore determine the load to TWBs. The load to a TWBs is a function of leakage, retention, and the emission of concerned sub-basins, which can be written by following equation.

$$L_i^j(e_{io}) = \left(1 - \eta_i^j\right) \sum_{o=1}^{O} \mu_o e_{io} \tag{9}$$

where o (=1, 2…O) is the emission source index; $L_i^j(e_{io})$ stands for the load to TWBs j from sub-basin i; μ_0 stands for the leakage rate of source o; η_i^j is the retention rate of sub-basin i during transportation into TWBs j; and e_{io} stands for the emission of source o in sub-basin i.

We can derive the equation to calculate the retention rate from the physical balance equation, $1 - \eta_i^j = c_i^j Q_i / c_{i0} Q_i = c_i^j / c_{i0}$ and the diffusion equation of pollutant concentration in river, $c_i^j = c_{i0} \exp(-Kd_i^j/u_i)$, which can be written as:

$$\eta_i^j = 1 - \exp\left(-Kd_i^j \middle/ u_i\right) \tag{10}$$

where Q_i stands for the average volume of flux in sub-basin i; c_{i0} and c_i^j stand for the initial and ending concentration of nutrient respectively; K is the comprehensive attenuation coefficient of pollutant; d_i^j stands for the distance from sub-basin i to TWBs j; and u_i stands for the average flow velocity in sub-basin i.

2)Load response model of reduction

The form of load response model of reduction is same as Eq. (9) besides the leakage. The leakage of each measure depends on whether the measure is used for reductions in nutrients at the source such as reductions in the fertilizers use, or reductions in leaching of nutrients into soil and water for given nutrient emission levels, such as creation of wetlands. The leakage of those measures used for reducing nutrient at the sources is equal to the leakage of the source that it aim to. The leakage of those measures reducing leaching nutrient depends on the deposition of nutrient on land, which in turn, depends on the emission level from the source, and the abatement efficiency of the measures. Then the load response model of reduced nutrient can be described by following function.

$$g_i^j(x_{ik}) = \left(1 - \eta_i^j\right) \sum_{k=1}^{k=m} \nu_k x_{ik} \tag{11}$$

where $g_i^j(x_{ik})$ stands for the reduced load of TWBs j due to emission reduction in drainage basin i; ν_k is the leakage rate of abatement measure k. The meanings of other variable and parameters are the same as with the above functions.

2.4. Environmental target

Environmental target in terms of load reduction to a specific TWBs can be specified according to its designated ambient environmental standard and the current load level. Designated ambient environmental standard of a specific TWBs determines its up-limit nutrient concentration and then the maximum load carrying capacity. The environmental target is the difference between the maximum load and the current load level which can be calculated employing Eq. (9).

The other approach to specify the environmental target is to designate directly an abatement ratio of load (γ^j) in a specific TWBs considering the factors such as environmental protection goals and fiscal ability, etc., which can be expressed as:

$$T^j = \gamma^j \sum_{i=1}^{I} L_i^j(e_{io}) \tag{12}$$

where $L_i^j(e_{io})$ can be calculated according to Eq. (9).

3. Empirical estimate

In this section the approach described above is applied to the Jiulong River Watershed (JRW) to estimate the minimum cost solutions of nitrogen reduction. First the characteristics of the region are described. Then sources of nitrogen are identified and the emissions are quantified. Thereafter the retention and leakage rates are accounted for. Using this information the final load of nitrogen to the receiving waters from the different sub-basins can be calculated. Costs and capacity of available abatement measures are thereafter described. Results and sensitivity analysis are presented in the last two parts.

3.1. The study area

As the second largest watershed in Fujian province, China, JRW covers an area of about14,000 km^2. It drains through 9 county level administrative units of Longyan, Zhangzhou and Xiamen Municipality. JRW consists of three major branches, Beixi River, Xixi River and Nanxi River, of which flows separately to the estuary and then empties to the adjacent coastal water, Xiamen Bay (Fig. 1). This watershed is the major contributor of nutrition to the Xiamen Bay which is under the serious threat of eutrophication (TIO and XMU, 2009; Chen et al., 2013). In addition, JRW is a vital drinking water source for more than 5 million people.

With the rapid economic development, urbanization and population growth in the watershed, environmental degradation and disruption of resource-base have seriously challenged the sustainable development of JRW. The Fujian provincial government initiated the Jiulong River Watershed Management Programme (JRWP) to address the key environmental and resources problems within the entire watershed since 1999 (FPG, 1999; FPG, 2001; FPG, 2006). After a decade implementation, JRWP successfully reduced the emission and concentration of COD and other industrial sources pollution in the watershed. However, the emission and concentration of nitrogen continues to increase, which results in the increase of the frequency of algal bloom and threatens safe drinking water supply (Peng et al., 2013). Finding the minimum cost solutions of nitrogen reduction is now of great concern of the watershed.

3.2. Nitrogen emission

This paper considers only the Beixi River Watershed due to its largest area in the watershed and the data availability. The study area is further divided into 15 sub-basins (B1 to B15 in Fig. 1). Among them, six sub-basins (B1, B2, B7, B13, B14, B15), where different downtowns are located, are named as urban sub-basins; while the rest 9 sub-basins are named as rural sub-basins The receiving waters of Beixi River, the Jiangdong Reservoir (JDR),

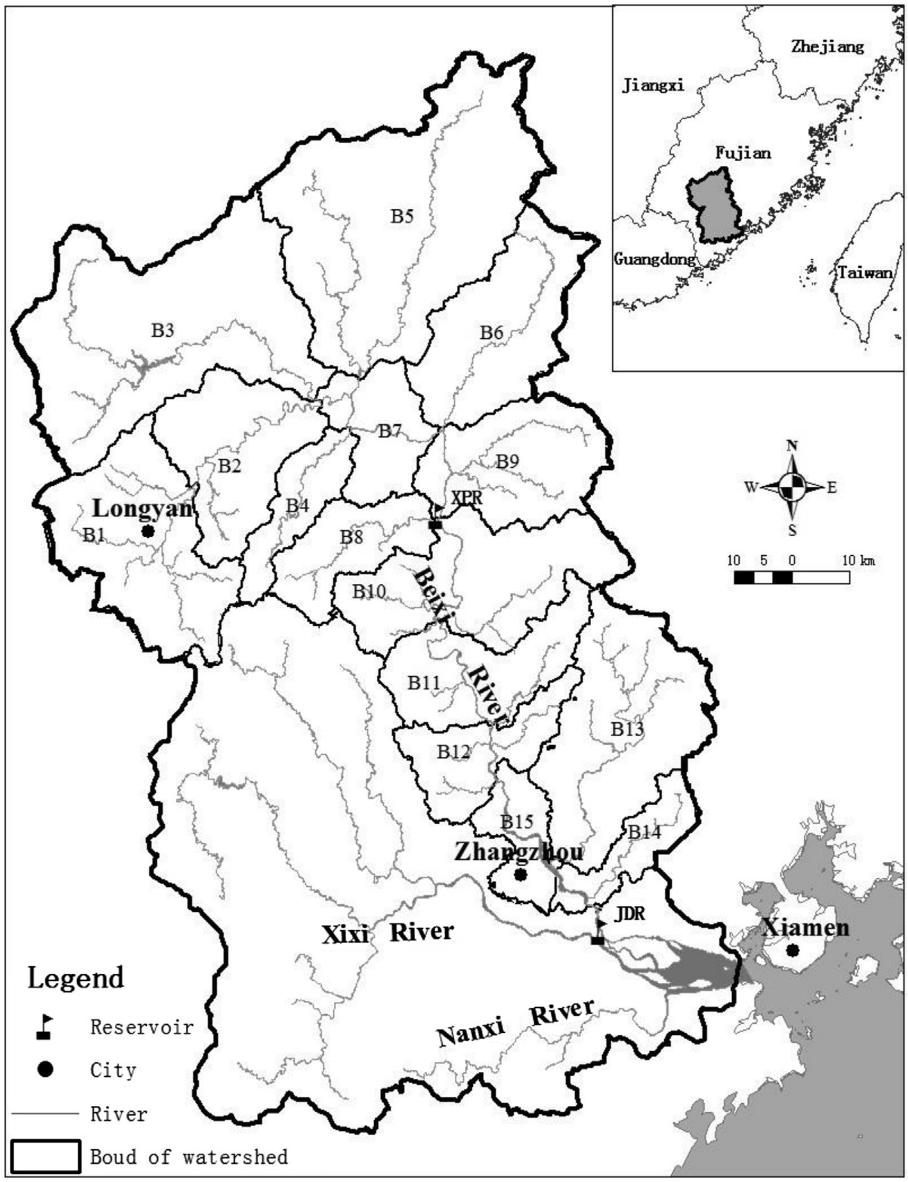


Fig. 1. Jiulong River Watershed.

which is the drinking water source and of concern, is defined as TWBs. There are many dams and reservoirs that have been built in the river for power generation and other purposes during past five decades. Many algal blooms in the watershed occurred first in the reservoirs located in the up or middle stream and then spread downstream (Chen et al., 2013). The Xipi reservoir (XPR), which is located in the middle of Beixi River, is also defined as a TWBs to capture the local damages of excessive emission of nitrogen. Nitrogen emitted from sub-basin B1 to B9 discharges into XPR.

The major part of the nitrogen emission in JRW comes from sources such as agriculture, livestock holdings, and domestic wastewater (COMI, 2013). Nitrogen emission from agricultural runoff are calculated according to the area and type of arable land (dry upland and paddy field), the volume of fertilizers application, the type of soil, the gradient, and the rainfall.[2] The emission from livestock are estimated based the output of livestock in different regions multiplied by lifecycle emission per head. Discharges of N from rural and urban domestic wastewater are calculated separately in order to designate the proper abatement measures, which are estimated based on data on annual emission per capita in different regions, and on connections of populations to sewage plants with different cleaning capacities. The air emissions from traffic and stationary combustion sources are not included in this paper due to the lack of data.

Above estimates are conducted at the township level, which is the lowest administrative level that the data for the emission estimate are available. The township level emissions are converted to each sub-basin based on the ratio of area of a specific sub-basin to the area of the towns and the information of estimated emission from different sources located in different towns.

The nitrogen emission by regions and sources is listed in Table 1. The total nitrogen emission in the watershed is 39,542 ton per year. Emission varies significantly across the sub-basins. The urban sub-basins account for almost 70% of the total emission, which indicates the high abatement capacity of these regions. Agricultural runoff

[2] For details see ESRC (2005).

Table 1
Emission, load and load response parameter.

Sub-basin	Emission (ton/yr)					Load (ton/yr)					Retention rate	
	Rural wastewater	Urban wastewater	Livestock holdings	Agricultural runoff	Total region	Rural wastewater	Urban wastewater	Livestock holdings	Agricultural runoff	Total region	JDR	XPR
B1	489	1005	3819	2139	7452	111	279	929	558	1877	0.65	0.43
B2	336	79	3545	1097	5057	86	25	975	323	1409	0.61	0.33
B3	125	23	1082	1350	2580	30	7	284	380	702	0.62	0.3
B4	49	4	318	506	877	13	1	88	150	252	0.6	0.24
B5	192	11	590	1519	2312	49	4	164	451	667	0.6	0.26
B6	161	14	208	897	1280	49	5	69	318	442	0.53	0.12
B7	151	359	1074	1441	3024	44	128	336	483	992	0.55	0.19
B8	107	2	122	278	510	43	1	53	130	227	0.38	0.03
B9	29	19	163	192	404	12	10	72	91	184	0.37	0.05
B10	223	116	320	998	1657	104	66	160	536	867	0.28	
B11	46	24	129	687	887	25	16	75	431	548	0.16	
B12	74	30	516	934	1554	42	21	317	614	994	0.12	
B13	529	130	1520	2794	4973	336	102	1039	2046	3523	0.02	
B14	303	178	1271	1063	2816	193	140	871	781	1985	0.02	
B15	666	441	2101	951	4159	406	331	1380	669	2786	0.06	
Total source	3481	2437	16,778	16,846	39,542	1544	1137	6813	7961	17,454		

and livestock account for the major part of the total emission, totalling 85%. The emission from rural domestic wastewater is larger than that of the urban wastewater. It must be noted that there are large amount of livestock and chemical fertilizers application in B1, B2, B7, B13, B14 and B15 although they are named urban sub-basins. All these indicates that JRW is dominated by agriculture.

3.3. Leakage and retention

Leakage rate varies among different sources. The numbers used for leakage rates of sources are based on the estimate by Hong et al. (2008). These estimates were conducted in several small sub-basins in JRW. The leakage rate of rural wastewater, urban wastewater, livestock holdings, and agricultural runoff are 0.65, 0.8, 0.7 and 0.75 respectively. We assume that the leakage rates of a same source located in a different sub-basin is same.

As discussed in Section 2.3, the leakage rates of measures depend on their targeted sources as well as their application toward reduction in sources or in leaching of nutrients. According to the main emission sources discussed in above, the measures considered in this paper include: (1) reduction of application of fertilizers use to reduce the nitrogen emission from agricultural runoff; (2) cultivation of catch crops to reduce the leaching of nitrogen from agricultural runoff; (3) reduction of livestock to reduce the emission from livestock holdings; (4) creation of wetland to increase retention of nitrogen emitted from rural and urban domestic wastewater; and (5) construction of new or enhancement of existing sewage plants to reduce the nitrogen emitted from urban wastewater.

Reducing application of fertilizers and livestock and sewage plants are measures used for reduction in sources. Their leakage rates are equal to that of the aimed sources. Wetland and catch crops are measures used for reduction in leaching of nutrients into soil and water. The wetland can remove averagely 60% of nitrogen emitted from domestic wastewater.[3] Then the leakage of wetlands is assumed to be 0.6. There is no report about the abatement efficiency of catch crops in JRW. Data from studies in other places show that the catch crops can reduce nitrogen leaching by an average of 209 kg per hectare (Vos and van der Putten, 1997; Ren, 2003; Ren et al., 2003). The average N emission of arable land in JRW is 324 kg/hm^2. Then the leakage rate of catch crops is assumed to be 0.64.

The region is characterized by great variations in retention. Retention rates are estimated employing the Eq. (10). The distance from each sub-basin to each TWBs is measured by ArcGIS. The average flow velocity of each sub-basin is from Fujian Water Resource Bureau. There is no study about the comprehensive attenuation coefficient of nitrogen in JRW. Kou (2005) indicated that integrated attenuation coefficient (BOD) vary considerably among rivers in China, which ranging from 0.015 to 3.45. Wang (2007) simulated, employing the Q2K model, the attenuation coefficient of BOD, N—NH4 and organic phosphorus in this region, among which BOD's coefficient ranges from 0.2/d to 0.28/d. Considering that the attenuation coefficient of nitrogen is smaller than that of BOD, this paper designates that nitrogen's coefficient is 0.16/d. Load response parameters to JDR and XPR are calculated employing these data and Eq. (10). And the results are shown in column 12 and 13 of Table 1.

The leakage and retention are subject to a large degree of uncertainty because of the annual fluctuation of hydrological and metrological conditions, the inorganic nitrogen concentrations and above assumption of attenuation coefficient. This is addressed in a sensitivity analysis at the end of this section.

3.4. The nitrogen load

With the above information concerning emission of different sources located in different sub-basins, and the leakage and the retention rates, the final load of nitrogen to TWBs can be calculated employing Eq. (9). These loads and their distributions are presented in the column 7 to 11 of Table 1.[4]

Due to the up-taking of plant and soil and retention, only about 45 percent of the total discharge reaches the TWBs. The final load to the JDR is 17,454 tons per year. Our estimates are somewhat lower than the actual measurement, being 25,036 ton per year (CEE, 2013).[5] This can be explained by the fact that not all sources are

[3] Data source: Wuyishan wetland plant (personal Corresponding author. and Yu et al. (2011).

[4] Only the loads to JDR and their distributions are presented to simplify the analysis. The load to XPR follows the same pattern.

[5] This estimate was conducted based on the monthly concentration of nitrogen and volume of water flux in JDR.

included in this study (e.g. nitrogen deposition from atmosphere[6] and industrial sources), and the discharges from those areas located in the watershed but beyond the jurisdiction of Zhangzhou and Longyan municipalities[7] are not included in the estimates.

Each sub-basin's nitrogen load into TWBs are determined by the retention and emission. More than half of the total load to the TWBs comes from sources within their adjacent sub-basins. The high contribution of the most distant sub-basin (B1) can be explained by its large emission. Agricultural runoff is the largest source of the final load to the JDR (46.56%), indicating the application of fertilizer as a source of the eutrophication. The final load to the JDR from agricultural runoff exceeds the load from livestock sources about 7 percentage points even though the emission of these two sources are almost same (see Table 1), indicating the importance of concentrating on the final impact instead of the emission.

3.5. Costs and capacity of abatement measures

It is necessary to know the cost and capacity of available abatement measures in order to determine the optimal allocation of these measures. For the non-linear cost functions (resulting in linear or non-linear marginal abatement cost) the abatement cost curves may be continuous, which has the theoretically appealing feature that "corner" solutions are less frequent. In practice it is however difficult to estimate the non-linear cost functions due to the insufficiency of demanded extensive data. In this paper, those measures where data are available for estimation of non-linear cost functions, those cost functions are built into the model. For measures where establishment of non-linear cost functions is not feasible linear cost functions are applied. The needed economic data between the regions for the estimation of cost function are presented in Table 2. Following the cost functions of five abatement measures identified in Section 3.3 are established.

1)Reduced nitrogen fertilizers use

The cost of reducing fertilizers use is represented by the farmer's foregone profits when applying a lower quantity of fertilizer. This loss ($L\ (n_0,n_1)$) can be estimated, using information of the yield response function ($y\ (n)$), the price of crops (p^c) and N fertilizer (p^n), the initial and after abatement nitrogen fertilizer use level per hectare (n_0 and n_1), and the fixed costs per hectare. Assume that the yield response function is.$y(n) = an^2 + bn + t$. The reduced nitrogen (Δn) per hectare is the difference between n_0 and n_1. Then the lost profit function per hectare can be written as: $L\ (n_0, n_1) = -aP^c\Delta n^2 + (2an_0P^c + bP^c - P^n)\Delta n$. If the area of arable land in sub-basin i is s_i, the cost function of N fertilizer use reduction in sub-basin i ($AC_i^F(x_i)$) can be established as:

$$AC_i^F(\Delta n_i) = \left(A\Delta n_i^2 + B\Delta n_i\right)s_i \tag{13}$$

where$A = -ap^c, B = 2an_0p^c + bp^c - p^n$

The parameters of yield response function is estimated employing the historic data of crop output and N fertilizer use in JRW, which is,

$$\mathrm{y} = -0.002\mathrm{x}^2 + 3.608\mathrm{x} + 5221\left(\mathrm{R}^2 = 0.980\right)$$

The price of N fertilizer is 4.41 yuan/kg N (2010 price).[8] The average initial N fertilizer use level is 180 kg/hm^2 in JRW (PBFP, 2011), which is assumed to be same across all the sub-basins; the average prices of output in each sub-basin are listed in Column 10 of Table 2. The parameters in Eq. (13), A and B are calculated using these information (see Column 2 and 3 of Table 3).

The maximum N fertilizer reduction per hectare in each sub-basin is set at 35% of the initial fertilizer use.

2)Wetland construction

Construction of wetlands as an abatement measure could increase retention substantially due to various biogeochemical processes, such as denitrification, uptake in biomass and sedimentation. The costs of this measure consist of four components: (1) agricultural opportunity costs which occur when agricultural land is withdrawn from production and converted into wetlands; (2) construction costs; (3) administration costs including the operation and maintenance costs; and (4) secondary benefits in terms of reduced ammonia and climate gas emissions and benefits related to use and non-use values, which are not included in this paper because of the lack of data.

The first cost component is expressed as the value of the yearly economic rent from the current land use. In order to reflect the non-constant marginal abatement cost, the functional form of the opportunity cost curve for converting agricultural land into wetlands is assumed to be a quadratic. This assumption is reasonable because as more land is converted into wetlands higher yielding areas need to be chosen, thus leading to cost increase at the margin. It was justified by the empirical study (Byström, 1998).

Assume the ratio of construction costs (annual value) and administration costs per hectare to the annual net revenue of arable land (nr^L) are e and f respectively, then the cost function describing the relation between abatement cost and the area of wetland (s_{wet}) is $AC^{wet}(s_{wet}) = (cs_{wet}^2 + ds_{wet}) + (e+f)nr^Ls_{wet}$. Where the term $(cs_{wet}^2 + ds_{wet})$ is the opportunity cost; the $(e+f)nr^Ls_{wet}$ is the construction and administration cost.

If the capacity of wetland to remove nitrogen is w per hectare and assume that this abatement capacity is same for all sub-basins, then the cost function of wetland in sub-basin i reflecting the relation between abatement cost and reduced emission can be written as:

$$AC^{wet}(x_i) = \left(c\frac{x_i^2}{w^2} + d\frac{x_i}{w}\right) + (e+f)nr^L\frac{x_i}{w} = Cx_i^2 + Dx_i \tag{14}$$

where $C = c/w^2,\ \ D = [d + (e+f)nr^L]/w$

The estimate of c and d is done by a simple calculus (Schou et al., 2006). The known sub-basin specific factors are: economic rent on landed property for crop production and livestock, plus total size of cultivated land and total amount of livestock (see Column 2–7 of Table 2). Therefore the interval and the mean value of the sub-basin specific functions are known. The functions are then calculated by finding 3% of the total area and the corresponding percentage of income. It is easy then to estimate c and d on the specific sub-basins.3% is the maximum percentage of arable land converted into wetland in this paper.

The ratio of annual value construction costs and operational and maintenance costs to the annual net revenue of arable land are set to be 6% and 1% respectively in this paper.[9] The average abatement capacity of wetland is 1050 kg per hectare per year (Byström, 1998; Andersson et al., 2005; Xiong et al., 2005). The C and D in Eq. (14)

[6] Chen et al.'s study (2008) indicated that the nitrogen export from atmospheric deposition accounts for about 10% of the total load.

[7] This area accounts for about 12% of the watershed.

[8] Personal calculation according to Price Bureau of NDRC (2009, 2010, 2100).

[9] Data source: Wuyishan wetland treatment plant. Personal communication.

Table 2
Economic data for the estimation of cost functions.

Sub-basin	Arable land (hm^2)	Pig production (10,000 head)	Net revenue		Economic rent (million yuan/yr)			Emission coefficient (kgN/head)	Weighted average price of crops (yuan/kg)
			Arable land (crops) (yuan/hm^{2}yr)	Pig (yuan/head)	Crops production	Livestock production	Total		
B1	5339	60.96	35,280	218	188.37	132.89	321.26	5.58	5.14
B2	3304	41.28	21,330	189	70.48	78.02	148.50	6.23	4.03
B3	4058	15.26	19,875	189	80.65	28.83	109.49	6.23	4.46
B4	1536	3.15	22,905	189	35.19	5.94	41.13	6.23	4.07
B5	4668	6.38	22,875	189	106.79	12.07	118.86	6.23	4.12
B6	2770	2.06	16,320	189	45.21	3.90	49.11	6.23	4.07
B7	4350	14.91	27,885	208	121.31	31.01	152.32	5.90	4.94
B8	838	1.30	21,555	189	18.06	2.47	20.52	6.23	4.64
B9	581	2.22	23,400	189	13.59	4.20	17.79	6.23	4.75
B10	3120	3.80	24,975	189	77.92	7.18	85.10	6.23	4.60
B11	2135	1.39	20,250	189	43.24	2.63	45.87	6.23	3.99
B12	3198	5.72	18,900	189	60.45	10.81	71.26	6.23	4.01
B13	9454	15.33	25,275	189	238.94	28.98	267.92	5.90	5.22
B14	3200	19.15	27,375	218	87.59	41.75	129.34	5.58	5.66
B15	2899	34.26	28,785	218	83.44	74.70	158.14	5.58	5.90

Data source: COMI (2013).

can be calculated with these information (see Column 4 and 5 of Table 3).

3)Cultivation of catch crop

Catch crops, which are sown at the same time as the ordinary crop but continue to grow after the ordinary crop is harvested, can take up residual nitrogen in the soil, reduce the nitrogen leaching from the root zone. The cost of catch crops for the farmer consists of seed cost, sowing cost and profit loss due to decreased harvest, which is represented by the loss of net revenue of arable land due to the occupation of the crops planting time in this paper for simplified analysis. The cost function can be written as:

$$AC_i^{cat}(x_i) = \frac{T}{12}\frac{nr_i^L}{c^{cat}}x_i \tag{15}$$

Where T is the month(s) of catch crops occupied the land; c^{cat} is the capacity of catch crops per hectare to remove the nitrogen. The growing time of catch crops in JRW is 4 months; The abatement capacity of catch crops is 209 kg per hectare per year as discussed in Section 3.3; and the annual net revenue of arable land in each sub-basin are listed in Column 4 of Table 2. The value of parameters of abatement functions in each sub-basin can be calculated with these information (see Column 6 of Table 3).

Considering the intensive use of arable land in this region, the maximum feasible acreage with catch crops in each sub-basin are set at 3% of the agricultural area in rotation in this study.

4)Reducing the livestock production

With the information of net revenue (nr^{live}) and nitrogen emission (em_i) of livestock per head, the cost of reduced the livestock production in sub-basin i,$AC_i^{live}(x_i)$, can be established as:

$$AC_i^{live}(x_i) = \frac{nr_i^{Live}}{em_i}x_i \tag{16}$$

The net revenue and emission of pig are employed to estimate the cost function in this paper considering the pig is the main livestock production and main pollution source of livestock holdings in this region. The value of parameters of cost functions in each sub-basin are calculated with these information presented in column 5 and 9 of Table 2. And the results are shown in Column 7 of Table 3.

The maximum reduction in livestock production in sub-basin is set at 35% considering that livestock is one of the main income sources of many farmers in this region.

5) Construction of sewage plants

Sewage plant is one of the measures to reduce nitrogen emitted from domestic wastewater, which can be only used for those regions whose volume of discharged wastewater reach the scale that construction of sewage is efficient. Abatement cost, which is determined by reduction capacity, investment costs, and running costs, differs between different sewage plants. The unit costs are calculated by following formula:

$$C^{unit} = \left(\frac{(1+r)^t r}{(1+r)^t - 1}I + O\right) \Bigg/ \frac{Q\left(c^{in} - c^{out}\right)}{1000} \tag{17}$$

where C^{uint} is the unit cost; I is the investment cost; O is the running cost; Q is the annual handing capacity of sewage plant; c^{in} and c^{out} are influent and effluent concentration respectively. The term $(1+r)^t r/[(1+r)^t - 1]$ is the annuity formula converting the lump cost to annual cost.

Information of costs and influent and effluent concentration of wastewater of different scale sewage plants in Fujian Province are

Table 3
Parameters of cost functions.

Sub-basin	Reduced fertilizer use		Wetland		Catch crops	Reduced livestock	Sewage plant
	A	B	C	D			
B1	0.010	10.437	1.42E-04	35.80	42.20	39.07	33.44
B2	0.008	7.221	2.18E-04	21.51	25.51	30.34	
B3	0.009	8.461	5.34E-05	20.20	23.77	30.34	
B4	0.008	7.354	7.78E-05	23.26	27.40	30.34	
B5	0.008	7.498	1.69E-05	23.29	27.36	30.34	
B6	0.008	7.336	1.55E-05	16.61	19.52	30.34	
B7	0.010	9.863	4.99E-05	28.36	33.36	35.25	46.06
B8	0.009	9.000	1.11E-04	21.85	25.78	30.34	
B9	0.009	9.294	3.99E-04	23.43	27.99	30.34	
B10	0.009	8.870	2.26E-05	25.43	29.87	30.34	
B11	0.008	7.116	1.77E-05	20.62	24.22	30.34	
B12	0.008	7.160	3.23E-05	19.23	22.61	30.34	
B13	0.010	10.663	9.84E-06	25.75	30.23	32.03	46.06
B14	0.011	11.941	1.25E-04	27.77	32.75	39.07	46.06
B15	0.012	12.639	2.72E-04	29.05	34.43	39.07	46.06

collected to estimate the unit cost with discount rate 4%. Considering the volume of domestic wastewater in the study area, two scale of sewage plants, 20,000 to 50,000 ton per day and 100,000 ton per day, are designated in this study. The average unit cost of former sewage plant is 46.06 yuan/kg, and the latter is 33.44 yuan/kg. According to Table 1, the volume of urban wastewater in B1 is economically efficient for a large scale sewage plant. Small scale sewage plants are suitable for B7 and B13–15. The cost of each sub-basin by sewage plant are given in the last column of Table 3.

The maximum constraint of this measure in each sub-basin is set as 80% of the emission from urban domestic wastewater.

Table 3 indicates that the variation of abatement cost of measures located in different regions. The cost of reduced fertilizer use is the lowest cost option among all measures, and then the wetland. The sewage plant are the most expensive measures in most of regions except B1. In general, the abatement costs of measures located in urban sub-basins are higher than that of in the rural sub-basins.

3.6. Results

Information from the previous sections allows for solving the problem of a minimum cost solution of nitrogen load reduction to the TWBs. Since this study concentrates on the eutrophication of the XPR and JDR the objective is to reduce the final load to these two TWBs. These results were obtained by using GAMS modeling program (Brooke et al., 1992).

1) Total Cost and measures at different reduction level

The total cost for different load reduction targets are presented in Fig. 2a. It is, due to assumptions of measure's abatement capacity and parameter values, impossible to reduce the nitrogen load to the TWBs by more than 36.7%. Total costs increase rapidly for reductions above 20 percent, due to the limited capacity of low-cost measures. At higher reduction level the high-cost measures in distant sub-basins have to be implemented, which result in the fast increase of abatement cost. How different reduction targets affect costs is obvious from Fig. 2a. Small changes in targets might have a large impact on the cost at certain ranges. The final additional five percent reductions cause a 34.2% increase of total costs.

Fig. 2b and c shows that the reduction capacity vary significantly by measures and by regions respectively. Reduction by fertilizer as an abatement measure is always included for any given target due to the characteristics of its low cost. The wetland, catch crops, and reduction by livestock are added as abatement measures as the maximum capacity of the fertilizer reduction reached in 15% reduction. Sewage plants are not included until the reduction level reaches 30 percent. Reduction by livestock reaches it limited capacity at 35% reduction level (see Fig. 2b). The sub-basin B6, B8 and B9 reach their abatement capacity at 15% reductions, and then B4 and B5 at 20% reductions, which indicates the limited capacity of these regions. The capacity of urban sub-basins which have high abatement costs as discussed Section 3.5 is greater than that of the rural sub-basins, which can explain the fast increase of the total cost after 20% reductions level.

2) Abatements and costs for a 30 percent load reduction

30% load reduction to the TWBs can basically make the TWBs to reach the ambient water quality standards (COMI and EMS, 2012), which is designated as the environmental target in this paper. Results indicate that a 30 percent reduction of the nitrogen load to the TWBs can be achieved at a cost of RMB 263.12 million/year. The nitrogen load reduction to the TWBs, XPR and JDR will in this case be 3530 and 5236 tons N/year respectively according to Eq. (12).

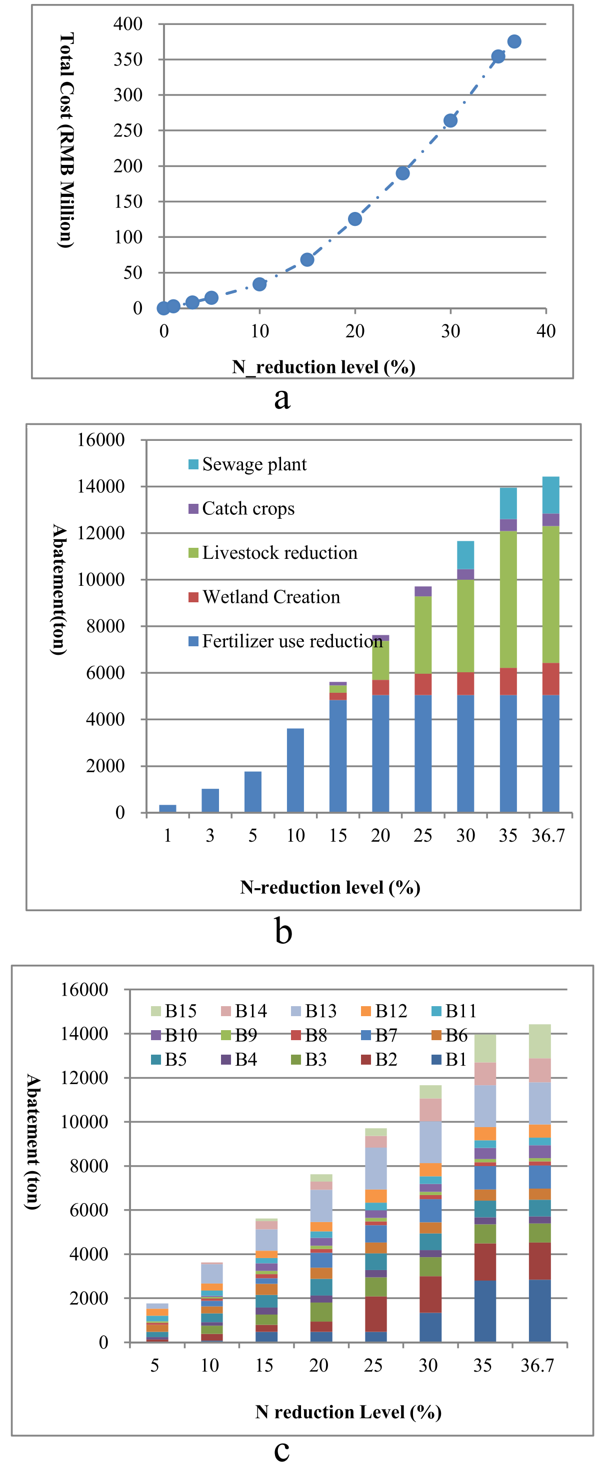
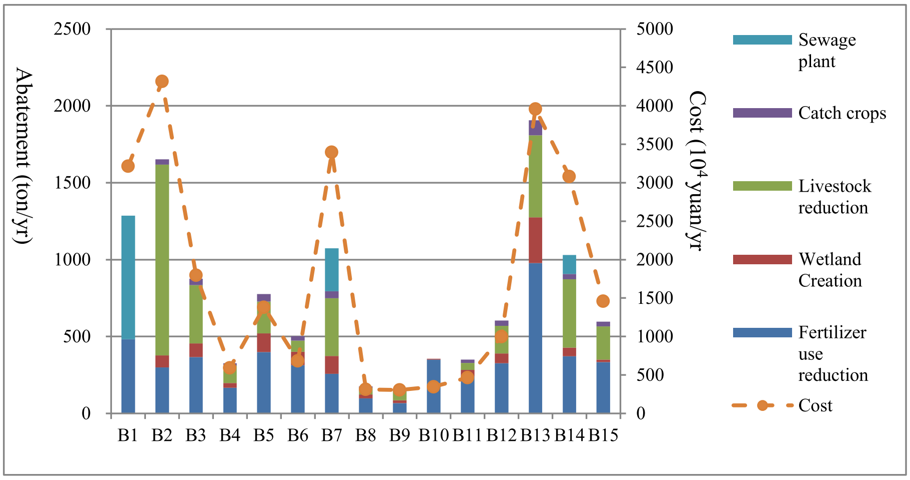
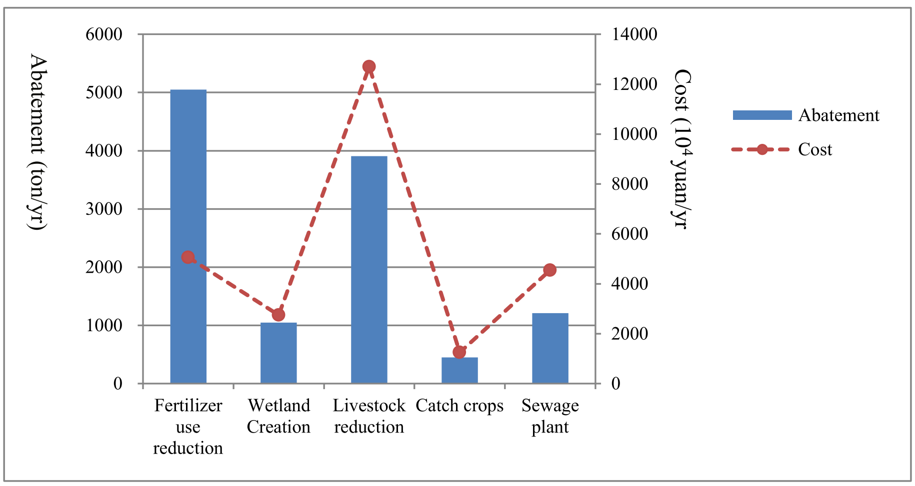


Fig. 2. Abatement cost, measures and regions at different reduction level (a) Total cost (b) Measures (c) Regions.

Averagely about 29.49% of total nitrogen emission (11,662 ton/year) must be reduced to reach the environmental target. The distribution of emission reduction between the regions and measures are presented in Fig. 3. The five urban sub-basins (B1, 2, 7, 13 and 14) account for about 60% of the reduction which is explained by the high emission and so on the large abatement capacity of



a



b

Fig. 3. Distribution of abatement and cost by regions and measures at 30% reduction (a) By regions (b) By measures.

these regions (see Table 1) as well as the fact that measures in the other regions are subject to the capacity limit (see Fig. 2c).

The distribution of costs between regions follows the same pattern of the emission reductions. Comparing to the abatement, the ratio of costs in B1, B2 and B7 are higher. This can be explained by the fact that the high abatement measures such as livestock reduction and sewage plant are included in these regions (Fig. 3a). The distribution of costs between measures is different to that of the reduction. The reductions by fertilizer and livestock account for 43.30% and 33.49% of total reduction, while the costs of these two measures account for 19.22% and 48.27% (Fig. 3b). This can be explained by the fact that the cost of fertilizer use reduction is on average one third of that of the livestock reduction (see Table 3).

It must be addressed that the set of the TWBs in the middle stream to achieve the upstream water quality affects significantly the distribution of reduction and so on the cost between regions, and the total reduction and costs (see Fig. 4). In two TWBs scenario, the upstream regions undertake more reduction which alleviates the reduction distributed in downstream regions such as B15 whose emission is high and load response parameter is large. The upstream 9 sub-basins (B1–B9) account for 58.47% of the total reduction and cost 60.82% while specifying XPR and JDR as TWBs at the same time. These values decreased to 39.92% and 30.49% under one TWBs scenario. To achieve the 30% load reduction in JDR without specifying XPR as a TWBs, the emission reduction and the abatement cost are 86.79% and 83.33% of that with two TWBs respectively, while the load reduction to XPR in this case is only 60.83% of that with two TWBs.

3.7. Sensitivity analysis

There are large uncertainties related to the retention and leakage as mentioned in Section 3.3. Assumptions concerning these parameters are likely to have a significant effect on total cost and abatement. It is therefore of interest to determine the model's robustness against changes in these parameters. Since our result implies that the major part of reductions should be taken at fertilizer reduction it is also of interest to investigate the parameters of that specific measure. In this section, the reference case, being the 30% load reduction to the receiving waters, is compared to the five scenarios: (1) decreased leakage rate by 10%; (2) increased leakage rate by 10%; (3) decreased retention by 10%; (4) increased retention by 10%; and (5) increased costs of fertilizer reduction.

If the environmental target is set as the 30% load reduction to the receiving waters, results show that the change of leakage by 10% (increase or decrease) has no impact on the reduction and the cost and the distribution. The change of retention by 10% leads to less than 0.6% change of emission reduction and cost. This paper

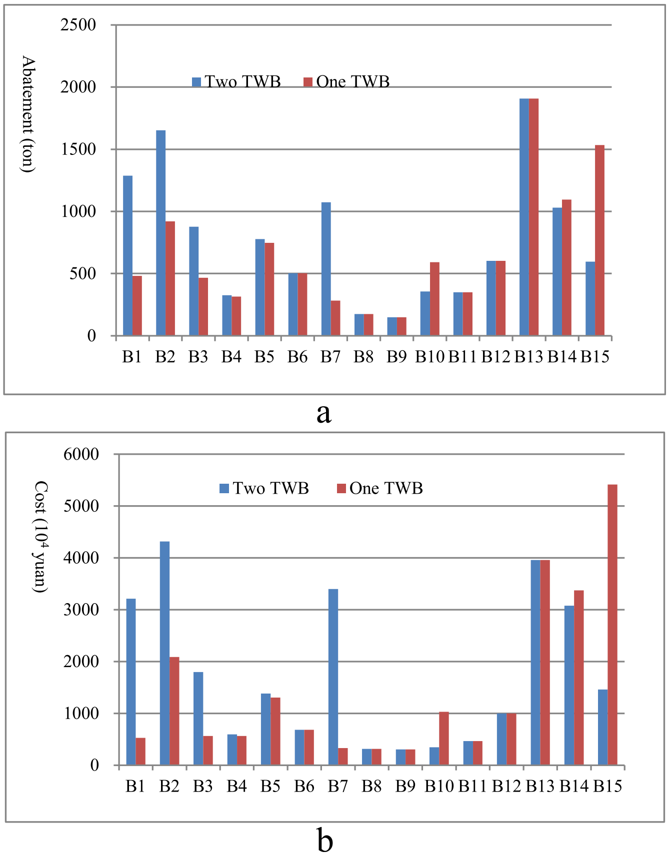


Fig. 4. Distribution of abatement and cost with one and two TWBs (a) Abatement (b) Cost.

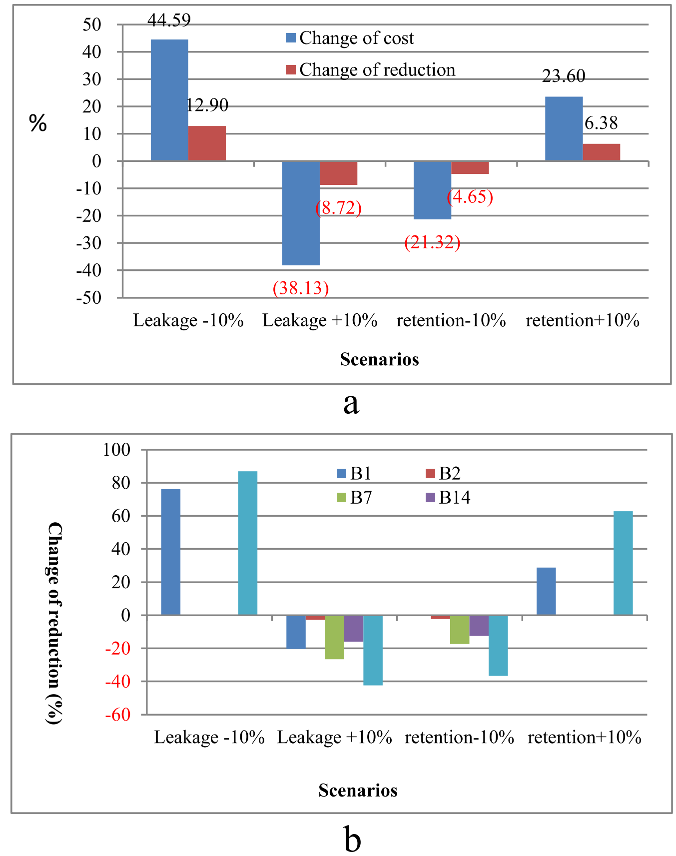


Fig. 5. The effect of Change leakage and retention by 10% (a) Change of cost and reduction (b) Change of distribution by region.

only presents the sensitivity analysis for same absolute numbers of load reduction to receiving waters as defined in the base case, i.e. 3530 ton in XPR and 5236 ton in JDR. A lower retention and a higher leakage will affect the total cost of the model in two ways. Firstly they generate a higher load to the receiving waters due to increased impact of land depositions, which will require a smaller emission reduction to achieve the target of the nitrogen load reduction, and thereby a lower cost. Secondly it decreases the marginal cost to the receiving waters by measures since their impact on the final load increases. The reverse effect occurs in the scenarios of increased retention and decreased leakage, increasing the emission reduction and total abatement costs.

Change of retention and leakage parameters by 10% the effects on emission reduction and costs can be observed in Fig. 5a. For example, by decreasing the retention parameter by 10 percent emission reduction and total cost are 4.65% and 21.32% lower than that of in the base case. By increasing the leakage rate by 10 percent emission reduction and total cost are 8.72% and 38.13% lower than that of in the base case. The total costs are more sensitive to the parameters change than that of the emission reduction, which can be explained by the fact that the total cost are under double impact with same direction: emission reduction and marginal abatement measures. The impact of retention and leakage parameters change on the distribution of emission reduction are presented in Fig. 5b. The mainly affected regions are those urban sub-basins with higher emission for reduction, especially B1 and B15. The effect of decreased leakage and increased retention are greater than that of increased leakage and decreased retention.

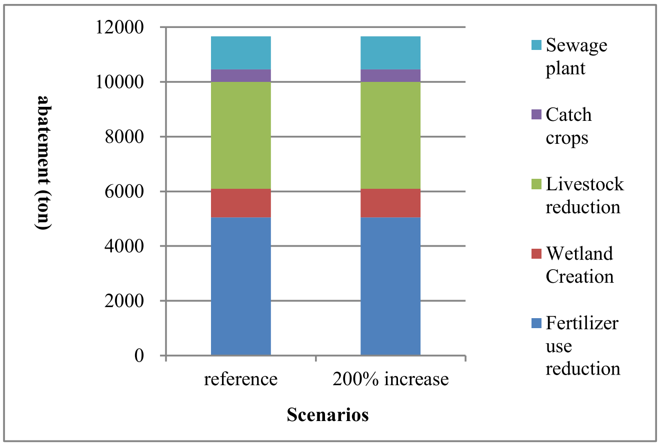
A doubling of the cost of fertilizers use reduction generates a 38.3% increase in total cost, but no changes concerning the allocation between measures and regions (Fig. 6). The allocation of measures and regions is therefore very robust against changes in this parameter.

The conclusion from the sensitivity analysis is that changes in cost parameters for reduction of fertilizers use mainly affect total costs while allocation between measures and regions is to a larger extent affected by the assumptions concerning actual leakage and retention. The emission reduction are more robust against changes in the retention and leakage parameters than that of the total cost. The high sensitivity of total cost to the changes in the retention and leakage parameters can be explained by double effect of the changed parameters, the emission reduction and the marginal abatement cost.

4. Discussions

This study, like most, has several limitations. These shortcomings have to be accounted for, in order to prevent the results being interpreted (and maybe used) in the wrong way. One of the shortcomings comes from the fact that emission from areas located in the watershed but beyond the jurisdiction of Zhangzhou and Longyan municipalities, and from atmospheric deposition are not considered due to the lack of data. This might cause an underestimation of required reduction since our nitrogen load reduction is expressed as a percentage. Targeted nitrogen load should therefore be expressed in the exact amount of nitrogen load necessary to reduce in order to reach the required concentration level. If these sources were included costs of reaching the different targets would be higher and probably require other, more expensive, abatement measures to be included.

The cost estimates of the abatement measures are based on assumptions of biological, physical and economic character, such as



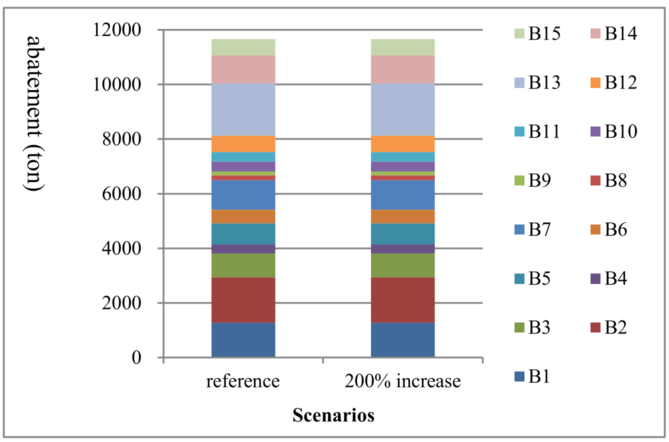


Fig. 6. Reduction allocation effect by increasing cost of fertilizers use reduction (a) By measures (b) By regions.

retention, abatement capacity limits and homogeneity of abatement costs at the source. The degree of uncertainties, with regard to these parameters, has to be emphasized. The actual values of estimated results should therefore be considered with caution. And the side benefits from measures such as provision of biodiversity of wetlands are neglected in the study which implies an overestimation of total costs.

Our study confirms the need of a watershed perspective when dealing with water pollution issues, in order to minimize the abatement cost achieving the pre-specified environmental targets. The success of such management will depend on the availability of data and reliable models of pollutant transportation. Any improvements in the understanding of pollutant transportation will increase the benefits generated by the approach described above. Limitations in this study may be viewed as topics for future studies.

5. Conclusion

This paper develops a systematic framework to estimate the minimum cost solutions of nitrogen reduction in watershed. The cost functions of abatement measures, the load response model of emission and reduced emission, and the abatement capacity functions have been assembled in the established framework.

The established models are employed in the JRW, an agriculture dominated watershed whose nitrogen emitted from agricultural runoff and livestock holding account for more than 85% of the total emission, and contribute almost the same percentage of load to the receiving waters. In total, five measures, reduced application of fertilizer, reduced livestock, sewage plants, creation of wetland and cultivation of catch crops, have been included for nitrogen reductions. The former three measures are classified as emission oriented measures reducing nutrient discharges at the sources, and the latter two as the leaching and retention oriented measures.

Calculations reveals that the costs of different measures located in different sub-basins, which are determined by the economic rents of crops and livestock of each sub-basin, the abatement capacity of measures vary significantly. Reduced fertilizer use is the lowest cost options, and other less expensive measures for nitrogen reductions are wetland creation. The costs of measures located in urban sub-basins is higher than that of rural sub-basins.

Calculated minimum cost solutions for different level of nitrogen load reduction to the receiving waters indicates large variation in total costs for the same percentage load reduction. The distributions of reductions between regions and measures, which are determined by abatement cost, the leakage and retention in the sub-basin, and most importantly the abatement capacity of regions and measures, are different at different load reduction level.

The specification of TWBs will affect the solutions. A 30 percent reduction of the nitrogen load to the two TWBs (XPR and JDR) can be achieved at a cost of RMB 263.12 million/year. 29.49% of total nitrogen emission in this case must be reduced. In one TWBs scenario, i.e., to achieve the 30% load reduction in JDR, the emission reduction and the abatement cost are respectively 86.79% and 83.33% of that of in two TWBs scenario. But the load reduction to XPR in this case is only 60.83% of that with two TWBs, which indicates the excessive nitrogen emission to the XPR and the necessity of inclusion of XPR as a TWBs to capture the local damage.

Costs of different measures, and thereby estimated minimum cost solutions are sensitive to parameter values on leaching, retention, and abatement cost and capacity. Sensitivity analyses were therefore carried out which showed that minimum cost solutions of nitrogen reductions are mainly affected by changes in the assumptions concerning actual leakage and retention. Changes in cost parameters for fertilizers use reduction, the main abatement measure, mainly affect total costs.

It must be addressed that the estimate should not be interpreted as an indication of which administration should eventually bear the costs. This is important to note when passing the results to policy makers, as the model prescribes how the effort should be mixed in order to reach the least cost solution but not how this solution is reached in a political economic context. As an example a scenario may indicate that a huge effort should be put in B1 and B2 located in Xinluo district, but how this effort is financed, i.e. which county actually bears the costs, is a political economic question, which cannot be addressed by this model.

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