Phosphorus release from agriculture to surface waters: past, present and future in China

M. Chen and J. Chen

ABSTRACT

So far, there is no clear picture at national level regarding the severity, spatial distribution, trend and driving forces of phosphorus (P) release from agriculture to surface waters in China, which presents a major obstacle for surface water quality management and relevant policy-making. By applying a proposed Activity-Unit-Balance (AUB) methodology, this paper retrospects and prospects phosphorus release from agricultural activities to surface waters from1978 to 2050 in China. Modelling results reveal that P load from agriculture has increased 3.4 times during 1978–2005 and will increase by 1.8 times during 2005–2050. Although major contribution factors are mineral fertiliser application (MFA) and livestock feeding activities (LFAs), LFAs will be the single largest source of increased total P load in the next decades. Most importantly, agricultural pollution in China is spatially overlapped with industrial and domestic pollution, and regions in the southeast to "Heihe-Tengchong" line have to be confronted with an austere challenge to control and manage industrial and domestic pollution as well as pollution from agriculture at present and in future.

Key words | agriculture, elementary unit, Eubolism, nutrient balance, phosphorus

M. Chen
J. Chen
Department of Environmental Science & Engineering, Tsinghua University,
Beijing 100084,
China
E-mail: *cmp02@tsinghua.edu.cn*; *jchen1@tsinghua.edu.cn*

INTRODUCTION

Since 1980s, surface waters in China have experienced severe human-induced eutrophication (Jin 1995; Dokulil *et al.* 2000; Jin *et al.* 2005; Li *et al.* 2007; Wang *et al.* 2007). With effective control on pollution from industry and domestic activities in recent years, phosphorus (P) from agriculture is considered to be an increased factor in determining the trophic status of waters in China (e.g. Guo & Yan 1999; Li *et al.* 2000; Quan & Yan 2002; Liu 2004; Zhang *et al.* 2004; Cheng *et al.* 2005 etc.). On a broad review of existing researches, Chen (2007) summarised that 20–94% of Total Phosphorus (TP) discharging to important waters in China came from agriculture. Those important waters include not only the "three lakes" watersheds (Taihu Lake, Chaohu Lake and Dianchi Lake), which are the key national pollution control watersheds; but also Miyun Reservoir in Beijing, Yuqiao Reservoir in Tianjin and Erhai catchment in Yunnan province, which are vital drinking water sources for local residents.

doi: 10.2166/wst.2008.256

So far, however, there has been no clear picture of agricultural P pollution at a national level in China, since existing related studies are rather unsystematic and case specific, and mostly at micro level. They are unable to give a comprehensive interpretation or description at macro level of P agriculture pollution. Nowadays, ignorance of agricultural P pollution has presented a major obstacle for surface water quality management and relevant policy-making in China. Dealing with this issue is not straightforward, as pollution export from agriculture into water is typical non-point source pollution and varies widely as a complex function of soil type, climate, topography, hydrology, land use and agricultural practices. With no sufficient observed data and behaviours' information, China is unlikely to support a widespread yet expensive and data-demanding watershed-based survey for most rivers and lakes.

To resolve this problem, this paper provides a methodology framework and a modelling tool, i.e. *E*lementary-*U*nit-based nutrient *B*alance m*O*de*LL*ing in farm*I*ng-feeding *S*yste*M*s (Eubolism), to evaluate P pollution from China agriculture by using available data statistics. Furthermore, this paper performs a temporal analysis of agricultural P pollution during a long period of 1978–2050 and identifies its spatial pattern variance in that period. This is followed by a detailed discussion of the calculation before we conclude this paper.

METHODOLOGY

Generally, evaluation of agricultural pollution is conducted by relating agricultural activities directly to observed water quality by virtue of watershed-based simulations. Characterised by non-point source pollution, pollutants from agricultural activities enter surface water in a diffuse manner at intermittent intervals which is subject to stochastic climate and hydrologic events. Furthermore, pollution originating from agricultural practices, such as fertiliser application, livestock feeding activities, etc., is to a great extent determined by multiple individual farmers' behaviours and their diversity and complex interaction lead agricultural pollution source even more difficult to identify and control. As a result, watershed-based simulation, especially distributed physically-based modelling, of agricultural pollution is extremely relying on massive data, not only observed data of storm runoff and water quality monitoring, but also detailed survey and records on farmers' behaviours. These data, however, are either very sparse and unreliable or very impracticable and expensive to obtain in China.

The AUB methodology

To deal with this problem, this paper is to relate agriculture activities to pollution emission load instead of water quality on the assumption that there is a positive relationship between agricultural activities and pollution. After that, it is most likely to establish the response relationship between water quality and emission load by analysing the correlation between total emission load of municipal pollution, industrial pollution and agricultural pollution, and routine monitoring data on water quality.

Therefore, this paper proposed an Activity-Unit-Balance-based (AUB) methodology, which can be a valuable tool to assess impact of agricultural activities on surface water in several aspects. Firstly, AUB is a 'top-down' methodology, which starts with identifying important agricultural activities types and land use types respectively as pollution components, and thus it takes statistical data and land use information as model inputs instead of observed data. These data and information in China are relatively more uniform, sufficient and easily accessible. Secondly, AUB is also a 'bottom-up' methodology since all computations are based on elementary unit of agricultural pollution, which is a computable, independent and homogenous minimal pollution discharge unit. By virtue of identification, definition and combination of different individual elementary units, AUB can be flexibly applied at micro (e.g. small catchments) as well as at macro (province, region and country) scale. Thirdly, AUB is a scientific, consistent and coherent system since it is in accordance with mass balance theory and assumes equilibrium between inflow and outflow.

The Eubolism model

Eubolism is the kernel of AUB, which is a quasi-physical modelling developed from nutrient balance budget and inventory analysis of agricultural pollution. In this model, agricultural activities are grouped into four major components, i.e. mineral fertiliser application (MFA), livestock feeding activities (LFAs), agricultural waste disposal (AWD) and rural life (RL). It comprises three modules, i.e. nutrient balance module (NBM), emission inventory module (EIM) and nutrient flow module (NF) (Figure 1). In NBM, seven inputs and four outputs are calculated and nutrients loss (indirect discharge) from agricultural soils of each land use type by denitrification, amination, storm runoff, and leaching into air, surface water and groundwater are all considered. In EIM, pollution discharge into three media is estimated by summarising direct discharge from agricultural activities and indirect discharge. Direct wastewater discharge from animal feedlots and gaseous loss in manure storage are regarded as direct P discharge. Finally, a nutrient flow chart can be drawn by applying mass balance theory

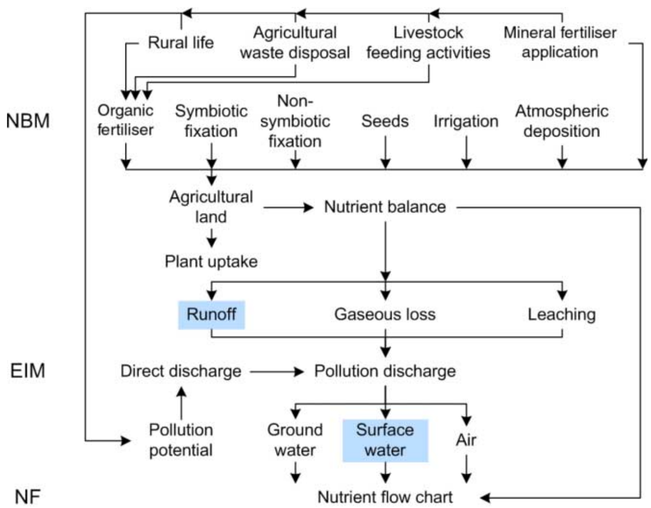

Figure 1 | Framework of Eubolism.

for each node in Eubolism. More details of AUB methodology and Eubolism can refer to Chen (2007).

In this paper, the Eubolism is spatially conducted in the provincial level, by which the entire China is divided into 33 parts excluding Hong Kong, Macao and Taiwan. Input data of Eubolism include production of major agricultural products (crops and animals), population, mineral fertiliser consumption, land use information and farmers' income etc., which are collected from a number of national databases including China Statistic Yearbook (CBYB), China Rural Statistical Yearbook (CRSYB), and China Agricultural Statistical Yearbook (CRSYB), and relevant coefficients are all listed in Chen *et al.* (2006*a*), Chen *et al.* (2006*b*), Chen (2007), Chen & Chen (2007) respectively.

Taking 2004 as base year, this paper conducts a time series analysis from 1978 to 2050. It should be noted that input information of agricultural production and land use during 2010–2050 is extrapolated by historical data of population and per capita shares of agricultural product during 1978–2005 on the assumption that agricultural production in future is driven by growing demand of increasing population. Presently, Eubolism does not possess a population prediction module, and populations during 2010–2050 are medians of available studies from several leading organisations, such as Food and Agricultural Organization (FAO) of United Nations, United Nations Population Division, National Statistical Agency (NSA) in China, etc. As a trend extrapolation, per capita shares of major agricultural product, such as milk, meat, eggs, grains and vegetables, in China is to increase and amounts to world average or that in developed countries by 2050 (Chen 2007).

Most importantly, Eubolism assumes that there will be no management improvement or technology revolution regarding agricultural pollution, that is only business as usual (BAU) scenario is considered in this paper and critical coefficients (e.g. runoff coefficient, direct discharge coefficient, etc.) will keep constant. To make a simplification, Eubolism supposes that there is no import or export of agricultural products during 2010–2050 in China.

RESULTS AND DISCUSSION

TP release from agriculture into waters in 2004 in China

In 2004, the TP from Chinese agriculture into surface waters is 467.9 thousands of metric tons (kt). And TP removal from agricultural soils by agricultural runoff (indirect discharge), among other things, is the most dominant transfer route, which accounts for 69.3% of total emission load.

In terms of TP emission structure of four components, LFAs are the most leading pollution source, contribution of which to total emission load is 43.9% (Figure 2a). Nowadays, China is the top meat and eggs producer in the world, and in 2004 there are on average 157.4 million heads big animals (i.e. cattle, dairy, buffalo, horse, donkey, mules and camel), 481.9 million heads pigs and 356.4 million heads sheep (sheep and goats) in stock. At the same time, modern LFAs characterised by intensification and specialisation have broken up the link between conventional farming and feeding systems. Superfluous livestock wastes are generated in a confined bound, where no sufficient land can assimilate them, and it to a great extent restricts their utilisation. Without an effective pollution regulation in practice for LFAs recently, a part of P contained in those wastes and wastewater will be discharged directly into waters. It is estimated that direct P loss from livestock feedlots in China amounts to 119.3 kt in 2004.

Comparatively, MFAs are equally important, which contribute 41.2% to total emission load. P application of mineral fertiliser in 2004 in China is 5.9 millions of metric tons (Mt) (including 3.2 Mt came from P fertiliser and 2.7 Mt from

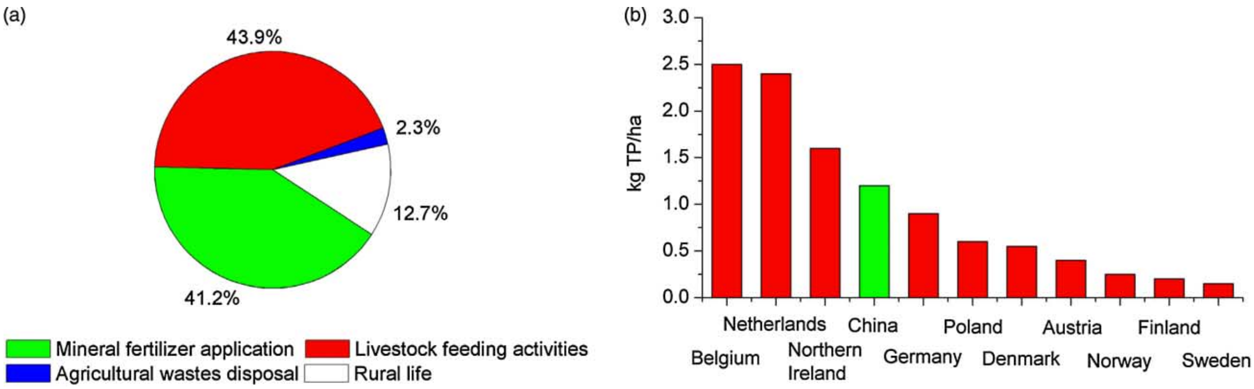

Figure 2 | Structure of TP emission load in 2004 and comparison of area-specific TP load by country. (Area-specific TP load in other countries see EEA 2005). (a) Structure of TP emission load in 2004. (b) Comparison of area-specific TP load.

compound fertiliser) and an overwhelming majority is applied to arable land. Since total P demand of crops in farming system is just 4.5 Mt, there is 10.9 kg/ha P surplus on an average. Most surplus P stores in agricultural soils as a nutrient source for future crop planting, but a part will be washed into waters by runoff or erosion. Generally, Chinese farmers have habits of dosing fertiliser before irrigation or rainfall, which aggravate P loss from soils into water.

Contribution of AWD and RL is just 2.3% and 12.7% respectively, from which we can conclude that agricultural pollution is not primarily driven by rural life but agricultural production.

As area-specific TP load is regarded as a good indicator to enable a comparison between different geographic boundaries, it is defined as ratio of TP emission load to agricultural land (sum of arable land and grassland) in this paper (EEA 2005). Figure 2b illustrates a comparison of P loads in European countries with China to their domestic aquatic environments. In 2004, area-specific P load to waters in China is 1.2 kg/ha, just behind Belgium, Netherlands and Northern Ireland, which calls for an urgency of policies and strategies for controlling P pollution from agriculture.

Temporal TP feature from agriculture during 1978–2050 in China

Since 1978, agriculture in China has seen a rapid development characterised by intensification, specialisation and regionalisation, which has effectively secured food demand of growing population (Figure 3a). As little abatement efforts have been implemented on China's agricultural pollution, however, P release from agricultural activities into aquatic environment is increasing steadily (Figure 3b).

In 1978 TP emission and area-specific load into surface environment were just 138.5 kt and 0.4 kg/ha respectively, no more than one third of that in 2004. Therefore, at

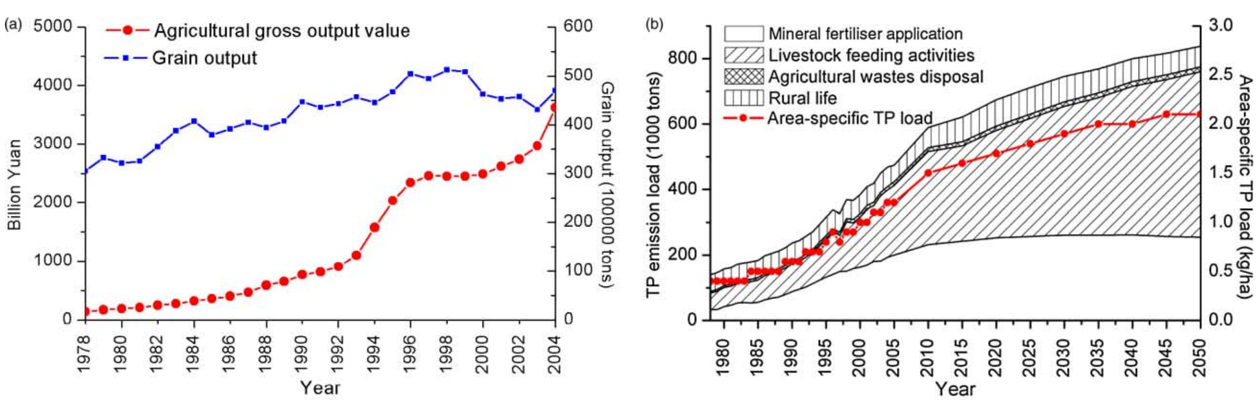

Figure 3 | Agricultural development and TP emission load in China. (a) Agricultural development during 1978–2004. (b) TP emission load and structure during 1978–2050.

that time agricultural pollution in China did not present an environment problem which should be paid attention to. By 2001, TP emission and area-specific load firstly exceeded 400 kt and 1.0 kg/ha respectively, and impact of agricultural production on waters stood out, which gripped public concerns.

In the next decades, ever-increasing population and growing food demands of residents, especially demands for meat, eggs and milk, will further agricultural development. Growing trend of P from Chinese agriculture into surface water is to continue. If no extra regulation or management is taken, by 2050 agricultural TP emission load will reach 838.1 kt and area-specific load will arrive at 2.1 kg/ha, which is 1.8 times as much as that in 2004. Extra 400 kt TP into waters will deteriorate eutrophication in rivers and lakes.

Although TP emission loads of four components all increased, their contribution to total P loss changed (Figure 3b). In 1978, RL was the most dominant factor, contribution of which is 36.6%, overrunning that of LFAs (35.8%) and MFA (23.8%). But after that, its contribution decreased and the trend is to continue. In 1994 its contribution fell to 18.4% and in 2004 it is just 12.7%. Although its contribution is to ascend and then descend during 2010–2030 due to a complex interaction between population increase and urbanization, in 2035 it will drop to 9.5%. In 2050, its contribution will be merely 7.5%.

By contrast, at all times MFA and LFAs are of great significance to TP emission from agriculture. From 1978–1994, contribution of MFA ascended from 23.8% to 42.1%, while that of LFAs increased from 35.8% to 37.4% during 1978–1986 and fluctuated during 1987–1994. Subsequently, MFA and LFAs became two factors of almost equal importance, and as one fell, the other rose. In 1997, the contribution of MFA reached its illusive peak of 46.1% and that of LFAs touched its illusive valley due to error resulted from national method adjustment of livestock statistic. However, since LFAs are extrapolated to grow at a faster speed than farming activities, in the next 40 years, contribution of LFAs is to increase and it will be the single largest source of increased TP load. Correspondingly, the contribution of MFA is to decrease. Obviously, LFAs should be given most priority when controlling agricultural pollution.

Spatial pattern variance of TP emission load during 1978–2050 in China

Although average area-specific P load from Chinese agriculture is not so high, some provinces are confronted with extremely serious P pollution from agriculture because of its extremely uneven distribution in space. In most polluted province (Fujian, 9.3 kg/ha), area-specific TP load is 300–400 times more than that in province with no distinct pollution (such as Tibet and Qinghai). According to statistical characteristics of area-specific TP load in 31 provinces in 2004, severity of TP pollution can be divided into five stages, with I stage being no distinct pollution, II stage very mild pollution, III stage mild pollution, IV stage severe pollution, and V stage very severe pollution (Figure 4b). In 2004, there are only two provinces within V stage (Beijing and Fujian), and five within IV stage (Guangdong, Hainan, Hunan, Jiangxi, Zhejiang and Tianjin). Except for two municipalities, all severely polluted provinces are located in southeastern area. Since P discharged from agriculture could be well

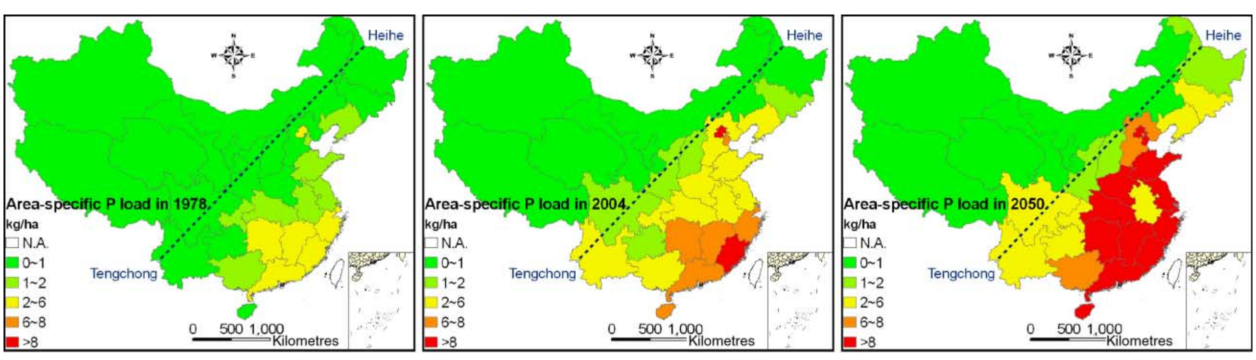

Figure 4 | Spatial pattern variance of TP from agriculture in China.

assimilated in arable land or grassland, it is not surprising that most severely polluted provinces are not always located in regions with most developed agriculture, such as Shandong, Henan and Sichuan, in possession of abundant arable lands.

By a comparison of spatial pattern between 1978, 2004 and 2050 (Figure 4), we can see that P pollutions in all 31 provinces in China have aggravated and the trend is to continue. In 1978, area-specific P load from agriculture of 31 assessed provinces are all not in excess of 6 kg/ha, and only seven provinces (Beijing, Fujian, Guangdong, Hunan, Jiangxi, Shanghai and Zhejiang) are mildly polluted. Interestingly, except Beijing, they are all located in Southeast China. By 2050, however, there are 13 provinces within V stage. Among them, six provinces facing most severe pollution are Fujian, Beijing, Hunan, Guangdong, Shanghai and Jiangxi, and area-specific P load of them will all surpass 10 kg/ha.

In future, agricultural P pollution severity in China is to concentrate in space. Totally, 17 provinces' TP load increments during 2005–2050 are quite lower than that during 1978–2004. But most pollution increases are confined in southeast provinces such as Fujian, Guangdong, Hunan, Jiangxi and Shanghai, so area-specific TP load at the national level is to increase (Figure 5). In northwest provinces, such as Gansu, Inner Mongolia, Ningxia, Qinghai, Shanxi, Shaanxi, Tibet and Xinjiang, P load increments during 1978–2050 are very limited, exerting no distinct influence on local pollution. As annual increments in 12 provinces in 72 years all exceed 0.1 kg/(ha·yr), they should be the prior control regions. Besides, some provinces located in most agriculture-developed regions, such as Henan, Shandong, Jiangsu and Heilongjiang, should also be attached great importance to.

From Figure 4 and Figure 5, we can conclude that pollution severity aggravated in the southeast to "Heihe-Tengchong" line, and these provinces are prone to be even more polluted in future. Heihe-Tengchong line is a well-known demographic and economic-graphical dividing line in China, which begins at Heihe in Heilongjiang province and ends at Tengchong in Yunnan province. In the southeast to it, there locate provinces with most dense population, most dynamic industrial economy as well as most intensified agriculture in China. But in the northwest, development in all aspects lags. In accordance with such socio-economic graphical feature, regions with most severe agricultural pollution are spatially overlapped with regions with most severe industrial and urban pollution, which have to be confronted with an austere challenge to control and manage industrial and domestic pollution as well as agricultural pollution.

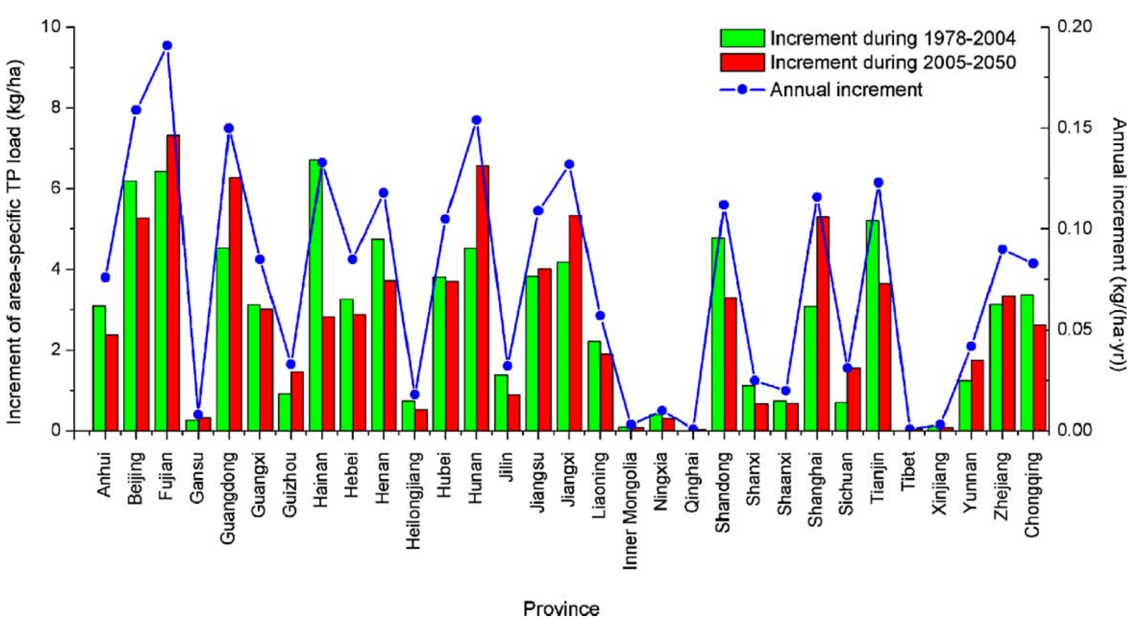

Figure 5 | P load increments by province and annual increment during 1978–2050.

CONCLUSIONS

A proposed AUB methodology, which relates agricultural activities to pollution load, is a sound alternative to expensive and data-demanding watershed-based simulation in China. This methodology takes advantages of sufficient statistical data and land use information, activities analysis and elementary-unit identification, and could enable Eubolism developed under its framework to be a very valuable evaluation tool for agricultural pollution in China.

By applying AUB methodology and Eubolism, this paper estimates TP emission load and area-specific load in a very long time frame, i.e. from 1978 to 2050. With the development of Chinese agriculture, TP emission load into aquatic environment from Chinese agriculture has increased tremendously and the trend is to continue in the next 40–50 years, which calls for effective environmental management. Because of their notable contribution to P emission presently and in future, mineral fertiliser application as well as livestock feeding activities should be the priority of agro-environmental management.

Area-specific P load in China is distributed unevenly in space, and southeast provinces confronted with the most serious industrial and urban pollution are facing the most severe pollution from agriculture. They certainly should be the prior regions to control agricultural pollution, but overlapping of three pollutions has made agricultural pollution management even more difficult to implement.

REFERENCES

Chen, M., Chen, J. & Du, P. 2006*a* An inventory analysis of rural pollution loads in China. *Water Sci. Technol.* **54**(11/12), 65–74.

Chen, M., Chen, J. & Lai, S. 2006*b* Inventory analysis and spatial distribution of Chinese agricultural and rural pollution. *China Environ. Sci.* **26**(6), 751–755. (in Chinese).

Chen M. 2007 *Nutrient balance modelling and policy evaluation in China farming-feeding system*. PhD Thesis, Department of Environmental Science and Engineering, Tsinghua University, Beijing, People's Republic of China.

Chen, M. & Chen, J. 2007 Inventory of regional surface nutrient balance and policy recommendations in China. *Environmental Science* **28**(6), 1305–1310. (in Chinese).

Cheng, B., Zhang, Z., Chen, L., Yuan, Z. H. & Sun, X. R. 2005 Eutrophication of Taihu Lake and Pollution from agricultural non-point sources in Lake Taihu Basin. *J. Agro-Environ. Sci.* **24**(s), 118–124. (in Chinese).

Dokulil, M., Chen, W. & Cai, Q. 2000 Anthropogenic impacts to large lakes in China: the Tai Hu example. *Aquat. Ecosys. Health Manage.* **3**, 81–94.

European Environmental Agency (EEA) 2005 *Source apportionment of nitrogen and phosphorus inputs into the aquatic environment (No.7/2005)*. EEA, Copenhagen, Denmark.

Guo, H. G. & Yan, Z. S. 1999 Eutrophication and diffuse pollution control in Dianchi lake. *Res. Environ. Sci.* **12**(5), 43–44. (in Chinese).

Jin, X. C. 1995 *Lakes in China-Research of their environment (Vols I and II)*. China Ocean Press, Beijing (in Chinese).

Jin, X. C., Xu, Q. J. & Huang, C. Z. 2005 Current status and future tendency of lake eutrophication in China. *Sci. China Seri. C: Life Sci.* **48**(s), 948–954.

Li, R. G., Xia, Y. L., Wu, A. Z. & Qian, Y. S. 2000 Pollutants sources and their discharging amount in Taihu lake area of Jiangsu province. *J. Lake Sci.* **12**(2), 147–153. (in Chinese).

Li, R. Q., Dong, M., Zhao, Y., Zhang, L. L., Cui, Q. G. & He, W. M. 2007 Assessment of water quality and identification of pollution sources of Plateau Lakes I Yunnan (China). *J. Environ. Quality* **36**, 291–297.

Liu, Y. 2004 *Study on phosphorus societal metabolism and eutrophication control policy in China*. PhD thesis, Department of Environmental Science and Engineering, Tsinghua University, Beijing, People's Republic of China.

Quan, W. M. & Yan, L. J. 2002 Effects of agricultural nonpoint source pollution on eutrophication of water body and its control measure. *Acta Ecol. Sini.* **22**(3), 291–299. (in Chinese).

Wang, X. L., Lu, Y. L., Han, J. Y., He, G. Z. & Wang, T. Y. 2007 Identification of anthropogenic influence on water quality of rivers in Taihu watershed. *J. Environ. Sci.* **19**, 475–481.

Zhang, W. L., Wu, S. X., Ji, H. J. & Kolbe, H. 2004 Estimation of agricultural nonpoint source pollution in China and the alleviating strategies (1): estimation of agricultural nonpoint source pollution in China in early 21st century. *Sci. Agricult. Sini.* **37**(7), 1008–1017. (in Chinese).