Load estimation and source apportionment of nonpoint source nitrogen and phosphorus based on integrated application of SLURP model, ECM, and RUSLE: a case study in the Jinjiang River, China

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Abstract The nonpoint source (NPS) pollution is difficult to manage and control due to its complicated generation and formation. Load estimation and source apportionment are an important and necessary process for efficient NPS control. Here, an integrated application of semi-distributed land use-based runoff process (SLURP) model, export coefficients model (ECM), and revise universal soil loss equation (RUSLE) for the load estimation and source apportionment of nitrogen and phosphorus was proposed. The Jinjiang River (China) was chosen for the evaluation of the method proposed here. The chosen watershed was divided into 27 subbasins. After which, the SLURP model was used to calculate land use runoff and to estimate loads of dissolved nitrogen and phosphorus, and ECM was applied to estimate dissolved loads from livestock and rural domestic sewage. Next, the RUSLE was employed for load estimation of adsorbed nitrogen and phosphorus. The results showed that the 12,029.06 ta^{-1} pollution loads of total NPS nitrogen (TN) mainly originated from dissolved nitrogen (96.24 %). The major sources of TN were land use runoff, which accounted for 45.97 % of the total, followed by livestock (32.43 %) and rural domestic sewage (17.83 %). For total NPS phosphorous (TP),

its pollution loads were 570.82 ta^{-1} and made up of dissolved and adsorbed phosphorous with 66.29 and 33.71 % respectively. Soil erosion, land use runoff, rural domestic sewage, and livestock were the main sources of phosphorus with contribution ratios of 33.71, 45.73, 14.32, and 6.24 % respectively. Therefore, land use runoff, livestock, and soil erosion were identified as the main pollution sources to influence loads of NPS nitrogen and phosphorus in the Jinjiang River and should be controlled first. The method developed here provided a helpful guideline for conducting NPS pollution management in similar watershed.

Introduction

Nonpoint source (NPS) pollution has been increasingly recognized as a severe threat to aquatic environment in recent decades, and many studies have been conducted to evaluate NPS pollution and management (Daniel et al. 1998; Edwards and Withers 2008; Ongley et al. 2010). In North America, The role of NPS in water quality degradation became a subject of intensive research from the 1970s partly as a result of the eutrophication crisis of the North American Great Lakes. NPS inputs, especially from agricultural activity, have resulted in large amounts

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of nitrogen (N) and phosphorous (P) (Chang et al. 2004) being input into aquatic environments, which cause a wide range of problems such as toxic algal blooms, oxygen depletion, and loss of biodiversity. In addition, nutrient enrichment seriously degrades aquatic ecosystems and decreases the quality of water used for drinking, industry, agriculture, recreation, and other purposes (Farenga and Daniel 2007; Potter et al. 2004). Reducing nutrient is often considered the most effective strategy for controlling excessive algal growth. The U.S. Environmental Protection Agency has stated that water with nitrate–nitrogen content higher than 10 ppm is unsafe for use as a drinking water source (Ongley et al. 2010), which suggests that there is a need to reduce N and P losses to the environment.

Nutrient load estimation and source apportionment are an important and necessary process for efficient NPS control. This knowledge can contribute to improve our understanding of processes in the watershed and our design of measures for reducing surface water pollution. Currently, two major approaches can be used to estimate NPS loads: export coefficient modeling and deterministic modeling (Ongley et al. 2010).

The export coefficient model (ECM), which has evolved from the unit load approach that was developed in America in the early 1970s (Mcelroy et al. 1976), was used to predict eutrophication in stable water bodies. In this method, the nutrient load exported from a watershed is the sum of the losses from individual sources, such as land use, livestock, and rural living. A combination of field experiments and statistics are then used to elucidate the relationship between the characteristics of the sources and the pollutant concentrations in the surface water, which results in generation of a coefficient for each source (Bowes et al. 2008; Johnes 1996). Although the ECM disregards the complex processes involved in NPS pollution and requires less data input, it has acceptable precision, especially suitable for areas where little data are available, and can meet the needs of long-term NPS pollution load simulation. Indeed, the ECM has been used successfully in many studies (Bowes et al. 2008; Mattikalli and Richards 1996; Soranno et al. 1996; Worrall and Burt 1999; Ierodiaconou et al. 2005; Shrestha et al. 2008).

The deterministic models are also important for load estimation of NPS. Specifically, this method uses the continuous simulation of pollutants from transportation,

transformation, and output, to determine the pollution occurrence time and source. Many such models have been successfully developed for this purpose, including agricultural nonpoint source (AGNPS) (Cho et al. 2008; Polyakov et al. 2007; Young et al. 1989), ANSWERS (Beasley et al. 1980; Singh et al. 2006), HSPF (Bicknell et al. 2001; Ribarova et al. 2008), and soil and water assessment tool (SWAT) (Arnold et al. 1993; Easton et al. 2008).

The load estimation method selection depends on the local conditions and data availability (Shen et al. 2011). The ECM can achieve acceptable accuracy when adopted on an agricultural scale. The mechanistic watershed models can provide more accurate results on pollutant losses, but such models need a huge amount of input data, including topographic information in the form of a digital elevation map, a soil map, corresponding information on hydrological relevant parameters, a land use map, information on cultivation practice, data on precipitation, temperature, locations, and discharge characteristics of point sources, and management information on artificially controlled water bodies. These extensive input requirements make such a modeling task a highly demanding project, which hinders the application of these models to some extent (Singh et al. 2006). Thus, in a watershed with insufficient data, applying integrated model for load estimation and then identifying source distribution is a practical way of delineating and evaluating the influence of NPS, which will provide further support for NPS control.

The Jinjiang River watershed, which covers a drainage area of approximately 5,629 km^2, is the third largest river with most sand in Fujian Province, Southeast China. The recent economic development in the region has resulted in increasing NPS pollution in the watershed. Due to its important geographical location, the control of NPS in this region is of great importance. This study was conducted to estimate the pollution loads of NPS nitrogen and phosphorous and then apportion source contribution. In this study, two forms of nitrogen and phosphorous were identified: dissolved and adsorbed. The ECM and semi-distributed land use-based runoff process (SLURP) model were applied to estimate losses of dissolved nitrogen and phosphorous. Furthermore, the revise universal soil loss equation (RUSLE) was implemented for the load estimation of adsorbed nitrogen and phosphorous. Last, the source contribution ratio

was calculated, which is more beneficial for watershed aquatic environment management.

Materials and methods

Site description and available data

The Jinjiang River, which is between eastern longitudes 117°25′ to 119°05′ and northern latitudes 24°30′ to 25°56′, is located in Quanzhou City, Southeast China. It is formed by the confluence of several rivers and drained a total area of 5,629 km² (Fig. 1). The

study area has a subtropical climate with mild winters and warm summers. The annual average temperature ranges from 19.5 to 21°C. The annual precipitation is from 1,010 to 1,756 mm. The topography of the upper reach of the study area is characterized by the Daiyun Mountains, while the middle reach is dominated by relatively flat landscapes. Red soil is the major soil present in the catchment (33.6 %), followed by paddy soil (24.6 %) and yellow soil (11.3 %). The main land uses in the watershed comprise forest (48.38 %), agricultural land (28.79 %), grassland (17.09 %), living land (5.74 %), etc. The agricultural land is classified into paddy field and dry land. Rice is grown in the
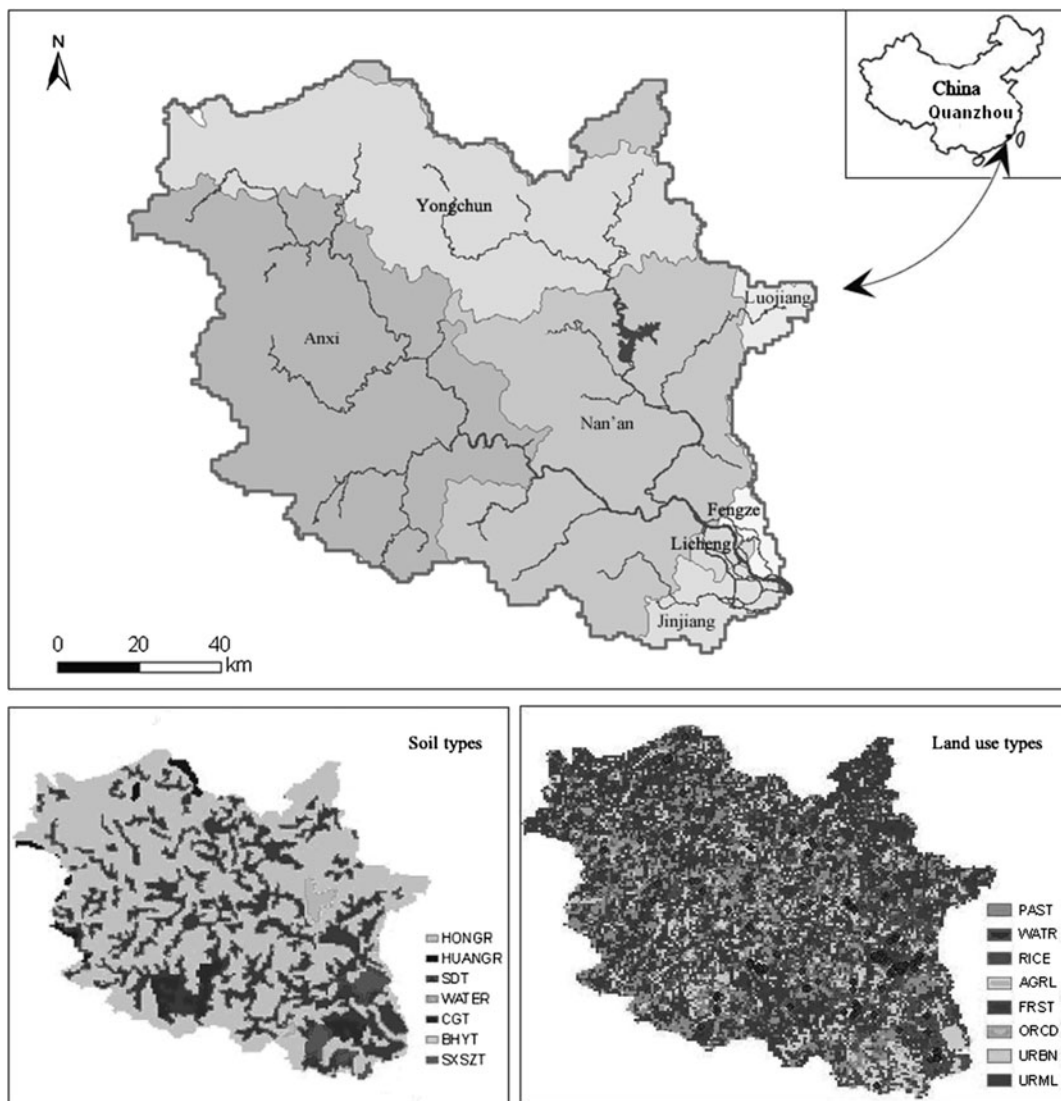
Fig. 1 Map of the Jinjiang River, soil types, and land use types

paddy field, whereas tea and oranges are used in dry land. Due to the high population density and intensive agricultural practices in the region, NPS pollution exhibits an increasingly severe trend. The data used in this study were listed in Table 1.

Load estimation of dissolved nitrogen and phosphorus from land use runoff based on SLURP model

By reference SWAT and AGNPS model, the load estimation of dissolved nutrient from land use runoff was outlined:

$$W_{\text{dis},l} = \lambda \times \sum_{u=1}^{U} \left(Q_u \times C_u \times 10^{-6} \right) \qquad (1)$$

where $W_{\text{dis},\ l}$ is the loss of nutrients from land use runoff, in tons, λ is the pollutant loss rate into the river, u is the land use type, Q_u is the land use runoff, in cubic meter, and C_u is the event mean concentration of rainfall runoff to land use u, in milligrams per liter.

The land use runoff can be calculated with the SLURP model, which is a continuous, spatially distributed watershed model that uses parameters associated with land cover or land use characteristics to simulate the hydrology of a watershed. It is able to simulate the behavior of a watershed continuously and avoids the data and calculation excesses of other distributed models (Kite 1997; Sanjay et al. 1998). SLURP carries out a vertical water balance for each element of the matrix represented by the number of subbasins and land uses occurring in each subbasin. The model describes processes such as actual evapotranspiration, infiltration, runoff, and groundwater

flow over each identified land use within each subbasin or aggregated simulation area (ASA) of the watershed. Weighted by the percentage of the ASA covered by a particular land use, the flow was converted to cubic meters per second and added to the total flow of the ASA (Romero et al. 2002; Susilo 2002). The determination of the ASAs can be undertaken by either a geographical information system (GIS) or by manual delineation.

Compared to other hydrology model, the SLURP model only requires three groups of data: physiographic data, parametric data, and time series data (Kite 1997). Physiographic data are the type of data that are related to the physical characteristics of the ASA in the watershed. These data can be obtained from topographic and land cover maps with GIS tools. Parametric data are constants that control the way the model transforms precipitation into runoff through processes such as evaporation, infiltration, and interception. These coefficients and parameters are derived from laboratory or field experiments. In this study, digital elevation model (DEM), land use, soil properties, etc. used by SLURP model are shown in Table 1. Model parameters of SLURP were calibrated using the shuffled complex evolution method, an automatic calibration technique incorporated into SLURP (Duan et al. 1994). Three years (2006 to 2008) of records were used for calibration. Data from 2009 were used to validate the model. SLURP performance was assessed by the Nash and Sutcliffe (1970) coefficient of performance. The goodness of fit of the predicted versus observed runoff during the calibration period (2006–2008) yielded a Nash/Sutcliffe criterion coefficient of 0.598, and the percent difference between total measured and predicted runoff

Table 1 Data types and descriptions

Data type	Format	Data item	Source
Digital elevation model	Raster	Elevation, overland, and channel slopes and lengths for 2004	1:250,000, Institute of Geographical and Natural Resources Research, Chinese Academy of Sciences; National Geometrics Center of China
Land use	Vector	Land use classifications for 2004	1:100,000, Institute of Geographical and Natural Resources Research, Chinese Academy of Sciences
Soil properties	Vector	Soil physical and chemical properties for 2004	1:1,000,000, Institute of Soil Science, Chinese Academy of Sciences
Weather	DBF	Precipitation from 1953 to 2007	China Meteorological Administration; local bureau of meteorology
Social economical data	DBF	Population, livestock rearing, fertilizer application	Statistics yearbook for 2009

was 16.6 %, indicating that total runoff was over-predicted during this period. Monthly measured and SLURP-predicted runoffs over the calibration period are shown in Fig. 2a. In the validation period (2009), the Nash/Sutcliffe criterion coefficient was 0.667, and the percent difference between total measured and predicted runoff was 13.4 %. Monthly measured and SLURP-predicted runoffs over the validation period are shown in Fig. 2b. Although SLURP over-predicted runoff, in general, on an annual basis, the model performed well in both years.

The event mean concentration of rainfall runoff, a weighted mean concentration of sample pollutants by flow in a whole rainfall, is used to indicate the mean concentration of some pollutants during a rainfall. It can be obtained by a biotope experiment or referring correlative research similar to the study area (USEPA 2001).

The pollutant loss rate into the river can be defined as $P_u \times R_u$ (USEPA 2001), where P_u is the ratio of storms producing runoff and R_u is the runoff coefficient for land use type u. The runoff coefficient for each land use type will be obtained with percent impervious equation with percent impervious which is extracted from the impervious terrain factor table with the equation: $R_u = 0.05 + (0.009 \times I_u)$. I_u is the percent imperviousness which is extracted from the impervious terrain factor table (USEPA 2001). It is easy to obtain with GIS tools based on watershed DEM. Other parameters have been obtained through literature review and shown in Table 2 (Xiong et al. 2007).

Export coefficient model

The ECM, which was proposed by Omernik (1976), was advanced by Johnes (1996) with the idea that the

nutrient loads exported from a watershed are the sum of the losses from individual sources, such as livestock and rural living. Different export coefficients are adopted for different crops, as well as for different types of livestock, and the export coefficients for rural living are then determined based on the discharge and treatment efficiency of domestic sewage. The model allowed accurate estimation and analysis of nutrient pollutants (Johnes 1996). The ECM for rural pollution was outlined as:

$$W_{\text{dis},r} = 10^{-3} \times \sum_{i=1}^{n} E_i [A_i(I_i)] \qquad (2)$$

where $W_{\text{dis},r}$ is the loss of nutrients from rural pollution, in tons, E_i is the export coefficient for nutrient source i, in kilograms per unit per year, A_i is the number of livestock type i, or of people, and I_i is the input of nutrients to source i, in kilograms.

The number of rural human living and livestock can be collected from local yearbooks and social survey (Shen et al. 2011). Generally, the export coefficients of rural living and livestock can be determined from a literature review because these coefficients do not vary greatly among areas. The results of field experiments to determine the rate at which nutrients are lost from each identifiable source to the surface drainage network (Johnes 1996). In this study, the export coefficients of human and livestock were referenced from field surveys of pollution sources along the Jinjiang River and shown in Table 3.

RUSLE

Adsorbed pollution loads of nitrogen and phosphorus caused by soil erosion can be computed using
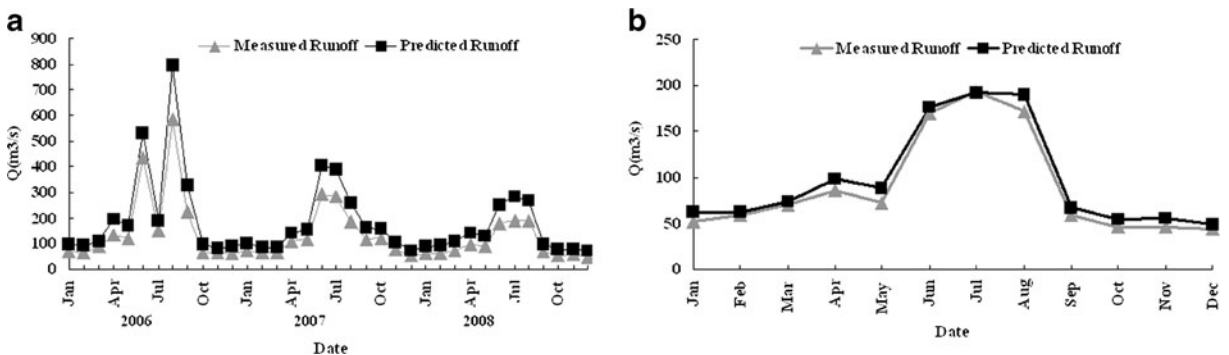

Fig. 2 Measured and SLURP-predicted runoff for 2006–2008 (**a**). Measured versus SLURP-predicted monthly runoff during validation for 2009 (**b**)

Table 2 Land use types, event mean concentration (EMC), and correction parameters for load estimation of nitrogen and phosphorus

Land use	I_U	R_u	EMC (mg L^{-1})		P_u	Background concentration (g kg^{-1})	
			DN	DP		AN	AP
Grassland	10	0.14	2.11	0.03	0.03	0.84	0.32
Water	100	0.95	1.49	0	0	0	0
Paddy field	75	0.725	2.45	0.18	0.18	0.91	0.42
Dry land	13	0.167	1.89	0.15	0.15	1.45	0.62
Forest	35	0.365	2.11	0.02	0.02	0.82	0.28
Orchard	20	0.23	2.11	0.04	0.04	1.06	0.34
Urban land	30	0.32	1.36	0.2	0.2	1.22	0.44
Rural land	40	0.41	13.28	0.1	0.1	1.40	0.49

the RUSLE, which is an empirical soil erosion model designed on the universal soil loss equation (Wischmeier and Smith 1978). It not only predicts erosion rates of ungauged watersheds using knowledge of the watershed characteristics and local hydroclimatic conditions (Angima et al. 2003), but also presents the spatial heterogeneity of soil erosion. Because of its convenience in application and compatibility with a GIS, the RUSLE has been the most frequently used empirical soil erosion model worldwide (Renard et al. 1997). With RUSLE, the adsorbed pollution loads of nitrogen and phosphorus can be calculated with follow equation:

$$W_{ads} = r_{ads} \times \left(\sum_{u=1}^{U} X_u \times A_u \times C_s \times t \right) \times 10^{-3} \quad (3)$$

$$X_u = R \times K_u \times LS \times S_u \times M_u \quad (4)$$

where W_{ads} is the adsorbed pollution load, in tons; r_{ads} is the transition ratio of soil; u is the land use type; X_u is the average soil loss caused by erosion,

Table 3 Export coefficients of nitrogen and phosphorus from rural domestic sewage and livestock

Nutrients	Export coefficient (kg unit^{-1} a^{-1})				
	Human	Cattle	Goat	Pig	Poultry
DN	1.554	6.0184	0.851	1.887	0.1628
DP	0.16	0.443	0.014	0.041	0.004

which can be calculated with Eq. (4), in tons per square kilometer; A_u is the area for land use u, in square kilometer; C_s is the background concentration of pollutants in soil, in grams per kilogram; t is the accumulation ratio of soil and sand; R is the rainfall erosivity factor; K_u is the soil erodibility factor; LS is the topographic factor; S_u is the support practice factor; and M_u is the cover and management factor.

The transition ratio of soil, accumulation ratio of soil and sand, and background concentration of pollutants in soil can be derived from literature review (Long et al. 2008). The following sections describe the computation of the R, K, LS, M, and S factors from precipitation data, soil surveys, a DEM, and land use maps (Xu et al. 2009).

The R factor quantifies the effect of rainfall impact and also reflects the amount and rate of runoff likely to be associated with precipitation events. It can be calculated by the following equation (Renard et al. 1997):

$$R = \sum_{i=1}^{12} 1.735 \times 10^{1.5 \times \lg \frac{p_i^2}{p_u} - 0.8188} \quad (5)$$

where p_u is the mean annual precipitation, and p_i is the mean monthly precipitation. The p_u values in each weather station were computed first and then the p_u values in the area were obtained through interpolation, extrapolation, and contour method in Arcview GIS (Shen et al. 2011).

The soil erodibility factor K value is the rate of soil loss per rainfall erosion index unit as measured on a standard plot and often determined using inherent soil

properties (Parysow et al. 2003). It can be computed as follows (Williams et al. 1983):

$$K = \{0.2 + 0.3 \exp[-0.0256 S_d (1 - \tfrac{S_i}{100})]\} \times [\tfrac{S_i}{C_i + S_i}]^{0.3} \times$$

$$\{1.0 - \tfrac{0.25 C_0}{C_0 + \exp(3.72 - 2.95 C_0)}\} \times [1.0 - 0.7(1 - \tfrac{S_d}{100})] /$$

$$\{1 - \tfrac{S_d}{100} + \exp[-5.51 + 22.9(1 - \tfrac{S_d}{100})]\} \quad (6)$$

where S_d is sand content (in percent), S_i is silt content (in percent), C_i is clay content (in percent), and C_0 is organic carbon content (in percent). The soil particle size data were obtained from the 1:1,000,000 soil databases in China.

Within the RUSLE, the LS factor reflects the effect of topography on erosion, the slope length factor (L) represents the effect of slope length on erosion, and the slope steepness factor (S) reflects the influence of slope gradient on erosion (Lu et al. 2004). It can be calculated by the following equation (Jabbar 2003):

$$LS = \left(\frac{\text{flow accumulation} \times \text{cell size}}{22.13}\right)^{0.4}$$

$$\times \left(\frac{\sin \text{ slope}}{0.0896}\right)^{1.3} \quad (7)$$

in which the flow accumulation, cell size, and slope can be extracted in a GIS map consisted of the spatial distribution (digital elevation model).

The M factor is used to reflect the effect of cropping and management practices on soil erosion rates in agricultural lands and the effects of vegetation canopy and ground covers on reducing soil erosion in forested regions (Renard et al. 1997), which varies with season and crop production system. It can be calculated by the following equation:

$$M = \begin{cases} 1 & m = 0 \\ 0.658 - 0.3436 \lg m & 0 < m < 78.3\% \\ 0 & m > 78.3\% \end{cases} \quad (8)$$

where, m is the yearly average cover ratio of vegetable. In this study, the tool of ArcSwat was employed to derive the data of m.

The S factor is the ratio of soil loss with a specific support practice to the corresponding loss with upslope and downslope tillage (Renard et al. 1997). Its value varies from 0 to 1 and can be obtained based on literature reviews (Statistics yearbook for 2009). In this study, the values of S factor for land use were shown in Table 4.

Results and discussion

Loads of dissolved nitrogen and phosphorus

Based on DEM, ArcSwat was used to define drainage range and divide Jinjiang River watershed into 27 subbasins according to the originating and jointing characteristics of the rivers. The land use types were extracted in Fig. 1, with the agricultural land accounting for about one fourth. Conventional tillage is the primary tillage pattern there. And downslope cultivation has been a common practice for years, which facilitates the downward movement of water and sediment during cultivation (Shen et al. 2011). Drainage runoff and pollution loads of dissolved nitrogen (DN) and phosphorus (DP) from land use runoff for 2009 in the Jinjiang River were shown in Table 5.

It can be found from Table 5 that the drainage runoff from all mixed land use types ranged from 0.98 to 19.62×10^9 m³ a⁻¹ with total of 53.02×10^9 m³ a⁻¹, which covers the range reported for the Jinjiang River by Wang (2008). With 19.62×10^9 m³ a⁻¹, most runoff was contributed by forest because of its largest area, followed by paddy field (9.68×10^9 m³ a⁻¹), grassland (8.70×10^9 m³ a⁻¹), and dry land (8.63×10^9 m³ a⁻¹). Meanwhile, it can be summarized from Table 5 that the DN loads from all mixed land use types ranged from 50.26 to 2,805.01 t a⁻¹, accounting to 5,529.87 t a⁻¹, and the DP loads ranged from 2.80 to 108.58 t a⁻¹ with the total of 261.01 t a⁻¹. For the load intensities from land use runoff, they were estimated to range from 0.372 to 1.735 t km⁻² a⁻¹ for DN and 0.021 to 0.124 t km⁻² a⁻¹ for DP, respectively.

With the ECM, the rural domestic sewage was estimated to be 2,145.02 t a⁻¹ for DN and 81.72 t a⁻¹ for DP using the export coefficients mentioned before (Table 3), and the livestock discharge was 3,901.54 t a⁻¹ for DN and 35.64 t a⁻¹ for DP. Then, the total dissolved nitrogen loads from land use runoff and rural pollution were 11,576.43 t a⁻¹, and the total dissolved phosphorus loads were 378.38 t a⁻¹. Their average load intensity was 2.1170 and 0.0692 t km⁻² a⁻¹, respectively.

The simulated spatial distribution of DN and DP loads were shown in Fig. 3. Among the 27 subbasins within the catchment, numbers 7, 19, 11, and 2 were identified as areas with high DN and DP. These are several possible reasons for this: These subbasins are located high upstream, have steep slopes, are

Table 4 Values of *M* factor for land use in the Jinjiang River

Land use	Grassland	Water	Paddy field	Dry land	Forest	Orchard	Urban land	Rural land
M factor	0.8	0	0.3	0.36	1	0.6	0.35	0.32

subjected to tillage, and experience many major rainfall events, while downstream areas are mostly flat and experience fewer major rainfall events, resulting in less runoff, although these areas are intensively cultivated. This is in good agreement with the distribution varied much between the western part and eastern part of the watershed.

From the viewpoint of contribution ratio, livestock was the main contributor (33.70 %) for DN, followed by paddy field (24.23 %), rural domestic sewage (18.53) and forest (14.36 %). For DP, paddy field, forest, rural domestic sewage, and livestock were the main contributors with 28.70, 22.70, 21.60, and 9.42 %, respectively.

Loads of adsorbed nitrogen and phosphorus

The RUSLE model was run by means of a GIS platform (ESRI ArcGIS 9.3). The computations of the *R*, *K*, LS, and *M* factors were used with precipitation data, soil properties, a digital elevation model, and land use maps based on equation mentioned before and Table 1. The estimation loads of absorbed

nitrogen (AN) and phosphorus (AP) were shown in Table 6.

It can be found from Table 6 that the soil erosion from all mixed land use types ranged from 6.01 to 64.89×10^4 t a^{-1}. The soil erosion modulus ranged from 28.1 to 898.4 tkm^{-2} a^{-1}. Dry land and residential med/low-density field were at the high peak of soil erosion modulus. Compared to those of other land use excluding dry land and rural land, paddy fields have relative higher soil erosions for the tillage pattern. In the Jinjiang River watershed, conventional tillage is the primary tillage pattern. Moreover, downslope cultivation has been a common practice for years, which facilitates the downward movement of water during cultivation. Meanwhile, it can be summarized from Table 6 that the AN loads from all mixed land use types ranged from 13.44 to 194.84 ta^{-1}, and the AP loads ranged from 5.12 to 85.11 ta^{-1}. For the erosion modulus from land use, they were estimated to range from 0.0058 to 0.0318 tkm^{-2} a^{-1} for AN and from 0.002 to 0.139 tkm^{-2} a^{-1} for AP, respectively. Compared to the dissolved, absorbed nitrogen and phosphorus showed lower erosion modulus.

Table 5 Land use runoff and loads of dissolved nitrogen and phosphorus in the Jinjiang River (2009)

Sources		Area (km²)	Ratio (%)	Runoff (billion m³ a⁻¹)	DN Modulus (kg km⁻² a⁻¹)	DN Loads (t a⁻¹)	DP Modulus (kg km⁻² a⁻¹)	DP Loads (t a⁻¹)
Land use	Grassland	934.60	17.09	8.70	371.59	347.29	21.09	19.71
	Water	62.33	1.14	0.98	–	–	–	–
	Paddy field	869.89	15.91	9.68	3,224.56	2,805.01	124.82	108.58
	Dry land	612.40	11.2	8.63	634.95	388.84	47.93	29.35
	Forest	2,645.43	48.38	19.62	628.41	1,662.42	32.47	85.89
	Orchard	92.30	1.69	1.45	544.52	50.26	30.34	2.8
	Urban land	135.22	2.47	2.13	550.96	74.50	25.22	3.41
	Rural land	116.14	2.12	1.83	1,735.34	201.55	97.03	11.27
Rural pollution	Human	–	–	–	–	2,145.02	–	81.72
	Livestock	–	–	–	–	3,901.54	–	35.64
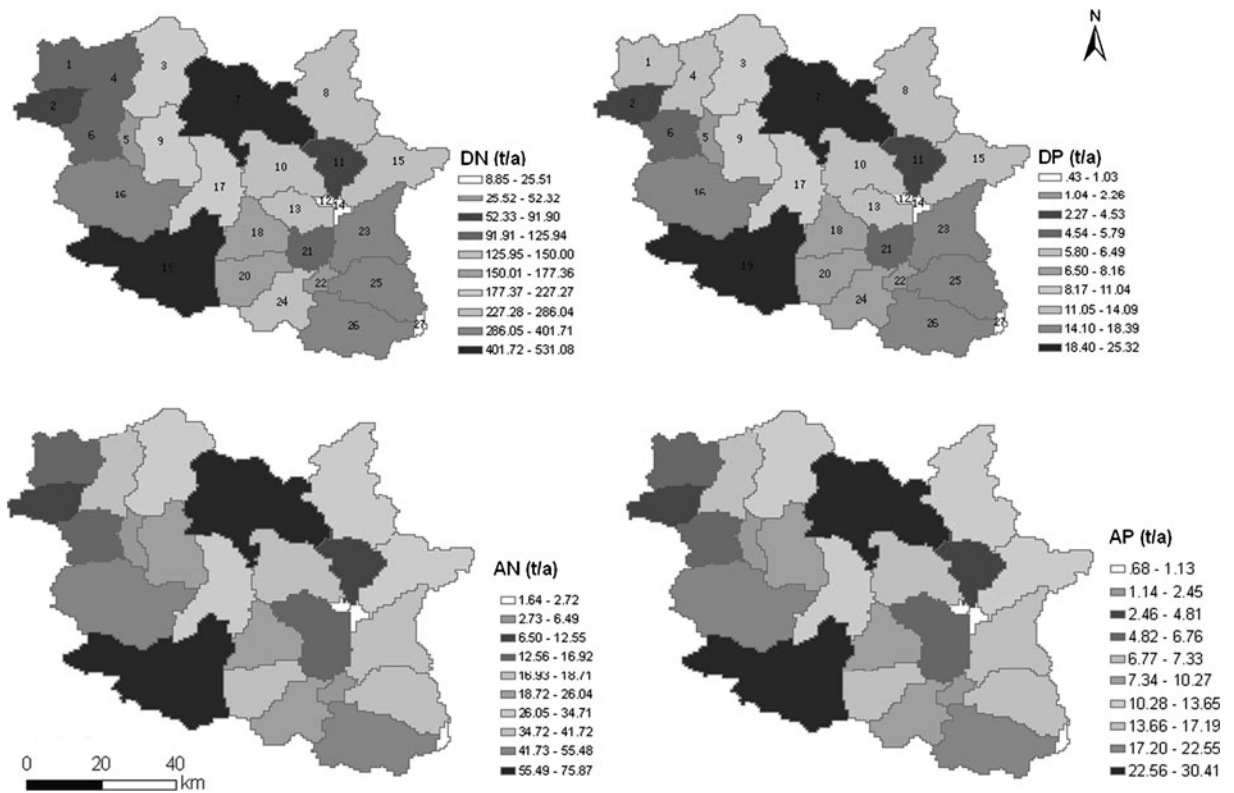
Fig. 3 Spatial distributions of nitrogen and phosphorus loads in the Jinjiang River

In the view of land use, the soil erosion in the Jinjiang River was mainly derived from dry land and paddy field. Obviously, soil loss has a close relationship with area, slopes, and topography of land use. For AN, the dry land and paddy field have a high erosion ratio of 43.05 and 34.89 %, respectively. Whereas, its erosion ratio was 44.22 and 37.49 % for AP to these land use types.

According to Fig. 3, it can be seen that the distribution of AN varied much between the western part and eastern part of the Jinjiang River. Owing to the higher altitude, the pollutants would inevitably accumulate in the eastern part (Ding et al. 2010). It also can be seen that there were two subbasins of relatively higher pollutant loads. One was located in the little blue brook in Anxi County where AN loads were

Table 6 Soil erosion and loads of absorbed nitrogen and phosphorus in Jinjiang River (2009)

Land use	Soil erosion modulus (t km^{-2} a^{-1})	Soil erosion (10^4 t a^{-1})	AN		AP	
			Erosion modulus (kg km^{-2} a^{-1})	Loads (t a^{-1})	Erosion modulus (kg km^{-2} a^{-1})	Loads (t a^{-1})
Grassland	68.2	6.37	14.38	13.44	5.48	5.12
Paddy field	746.0	64.89	181.49	157.88	82.94	72.15
Dry land	898.4	55.02	318.16	194.84	138.98	85.11
Forest	28.1	7.44	5.81	15.37	2.00	5.3
Orchard	651.1	6.01	174.65	16.12	56.45	5.21
Urban land	568.0	7.68	176.23	23.83	63.01	8.52
Rural land	775.8	9.01	268.21	31.15	94.97	11.03

Table 7 Loads and source contribution ratios of nitrogen and phosphorus in Jinjiang River (2009)

Nonpoint source types	TN		TP	
	Load (t)	Ratio (%)	Load (t)	Ratio (%)
Dissolved	1,1576.43	96.24	378.38	66.29
Land use runoff	5,529.87	45.97	261.02	45.73
Agricultural field runoff	3,244.11	26.97	140.73	24.65
Other field runoff	2,285.76	19.00	120.28	21.07
Rural domestic sewage	2,145.02	17.83	81.72	14.32
Livestock discharge	3,901.54	32.43	35.64	6.24
Adsorbed (soil erosion)	452.63	3.76	192.44	33.71

61.23 ta^{-1}. The other lies in the Tao brook in Yong-chun County where AN loads were 52.64 ta^{-1}. Agriculture is the mainstay of these areas. Crop of tea and oranges are the main products of these areas. Thus, the intensive agricultural practices raised the severe contamination. The spatial distribution of AP loads (Fig. 3) exhibited almost the same pattern as AN. This similarity demonstrated that the NPS-N and NPS-P pollution were consistent in the Jinjiang River watershed to some extent. Therefore, appropriate strategies should be devised to protect these critical areas where soil erosion is most serious.

Source apportionment for NPS nitrogen and phosphorus

Using the total simulation results, it was possible to identify significant source. Table 7 showed the total nitrogen and phosphorus loads. It can be summarized from Table 7 that the TN loads from land use runoff, rural pollution, and soil erosion were 12,029.06 ta^{-1}. In them, DN loads were 11,576.43 ta^{-1} with contribution

ratio of 96.24 %, and the AN loads were 452.63 ta^{-1} with 3.76 %. For TP, its total loads were 570.82 ta^{-1}; DP and AP contributed 378.38 and 192.44 ta^{-1} with 66.29 and 33.71 %, respectively. Obviously, TN mainly originated from DN including land use runoff and livestock discharge.

The average load intensity for TN was 2.225 and 0.106 tkm^{-2} a^{-1} for TP. Compared to the results in other basins in China, Jinjiang River watershed showed relatively lower pollution load intensity. For the Yangtze River, which is the largest river in China, the average TN and TP load intensities were estimated to be 3.38 and 0.24 tkm^{-2} a^{-1} (Shen et al. 2011), respectively, which were higher than that in the present study. These differences demonstrated that land use type plays an important role in NPS pollution since the dominant land use type is agricultural field (38.3 %) in the studied watershed in Shen et al. (2011), whereas in the present study, the major land use type is forest (48.38 %).

The result of source apportionment is shown in Fig. 4. The major sources of TN were land use runoff,

Fig. 4 Overall contributions of NPS nitrogen and phosphorus in the Jinjiang River
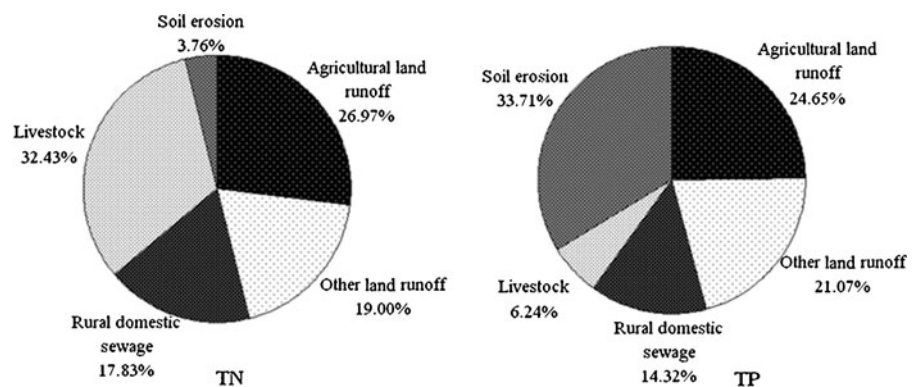
which accounted for 45.97 % of the total, followed by livestock (32.43 %) and rural domestic sewage (17.83 %). Among those eight land use types, agricultural field runoff, which included paddy field, dry land, and orchard, contributed the most (26.97 %). Rice, tea, and oranges were the dominant crop in agricultural land. For the economic yield from this crop, the use of fertilizers may have raised environmental contamination. Furthermore, crop residue, which is not systematically managed, is an N-rich pollution source. It has been documented that regardless of the residual level, conventional tillage is less effective at reducing N in water and sediments than no-till systems. Conventional tillage is just the primary tillage pattern in the Jinjiang River watershed. Moreover, in the vast southeast area of China, downslope cultivation has been a common practice for years. Although downslope cultivation is considered to be easier for plowing fields, it also facilitates the downward movement of water during cultivation. Additionally, downslope cultivation promotes soil erosion in response to every rainfall event during the growing season (Ding et al. 2010).

The major sources of TP ranked differently from those of TN: with land use runoff first (45.73 %), followed by soil erosion (33.71 %), rural domestic sewage (14.32 %), and livestock (6.24 %). Paddy field runoff was still identified as the dominant contributor among those eight land use types. And the contribution of rural domestic sewage showed little difference between these two nutrients. However, soil erosion contributed a much higher amount of TP (33.71 %) than TN (3.76 %), which may have been counterbalanced by the smaller portion from livestock discharge P deposition (6.24 %) than livestock discharge N deposition (32.43 %).

Taking account of synthetic factors for loads to TN and TP, therefore agricultural field runoff, livestock discharge, and soil erosion became the main pollution sources to influence loads of nonpoint source nitrogen and phosphorus in the Jinjiang River watershed. Obviously, these sources should be controlled first.

Conclusions

This study has estimated the loads of NPS nitrogen and phosphorus in the Jinjiang River and further identified the pollution sources. To accomplish this, an integrated pollution load model, which is established with parameters based on DEM, subbasins, land use, soil type obtained with the GIS tool, and data from study and experiment, was used here to calculate the loads of nitrogen and phosphorus and study pollution sources. The pollution loads of dissolved nitrogen and phosphorus from land use runoff, rural domestic sewage, and livestock were estimated based on the SLURP model and ECM. And loads of adsorbed nitrogen and phosphorus from soil erosion were been calculated with RUSLE.

A case study in Jinjiang River watershed showed the 12,029.06 ta^{-1} pollution loads of NPS-N were mainly derived from dissolved nitrogen with ratio of 96.24 % and interrelated with livestock, agricultural field runoff, and rural domestic sewage, whose contribution ratios were respectively 32.43, 26.97, and 17.83 %. For NPS-P, its total pollution loads were 570.82 ta^{-1} and made up of dissolved and adsorbed with 66.29 and 33.71 %, respectively. Soil erosion, agricultural field runoff, and rural domestic sewage were the main sources for phosphorus with contribution ratios of 33.71, 24.65, and 14.32 %, respectively. Taking account of the synthetic factor, therefore agricultural field runoff, livestock, and soil erosion became the main pollution sources to influence loads of NPS nitrogen and phosphorus in the Jinjiang River and should be controlled first. In conclusion, it is expected that the use of the integrated tools of SLURP model, ECM, and RUSLE for load calculation and source identification provide a useful guideline for supporting the management of NPS pollution.

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