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N and P runoff losses in China's vegetable production systems: Loss characteristics, impact, and management practices

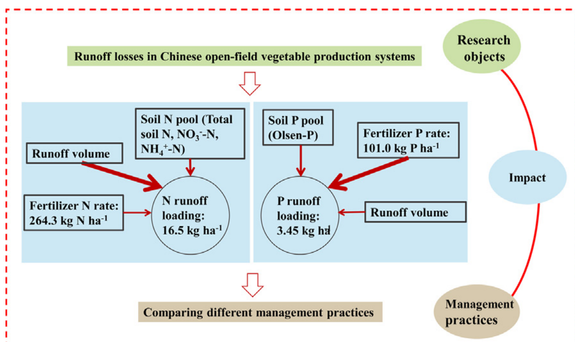
Rui Wang [a,b], Ju Min [b], Herbert J. Kronzucker [c], Yilin Li [b,⁎], Weiming Shi [b,⁎]

[a] University of Chinese Academy of Sciences, Beijing, 100049, China
[b] State Key Laboratory of Soil and Sustainable Agriculture, Institute of Soil Science, Chinese Academy of Sciences, Nanjing 210008, China
[c] School of Agriculture and Food, Faculty of Veterinary and Agricultural Sciences, The University of Melbourne, VIC 3010, Australia

HIGHLIGHTS

- Annual total N and P runoff in vegetable fields were 1.52 Tg and 0.33 Tg in China.
- Fruit vegetable fields produce more N and P runoff than other types of vegetables.
- Soil N pool dominates N runoff, and P fertilization affects mainly P runoff in field.

GRAPHICAL ABSTRACT



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ABSTRACT

Vegetable production systems involve the use of high rates of fertilizer application, leading to significant losses of nitrogen (N) and phosphorus (P) to the environment, resulting in water body eutrophication through surface runoff and leaching. We here quantify, at the national level, total N and P runoff losses and the key factors controlling runoff in open-field vegetable systems in China, by summarizing data from 151 publications pertaining to 13 Chinese provinces using literature dating back to 1990. Rank analysis was employed to clarify the impact of N and P runoff losses in vegetable systems, and different strategies for controlling N and P entering into water bodies are being compared. Vegetable production systems have higher fertilizer inputs (264.3 kg N ha^{-1}, 101.0 kg P ha^{-1}) compared with upland crop and rice cultivation. As a result, annual total nitrogen (TN) and total phosphorus (TP) losses via runoff from vegetable systems reached 16.5 kg ha^{-1} and 3.45 kg ha^{-1}, respectively, and the N and P loss ratio for fruit vegetable systems reached 13.1% and 3.95% of the total fertilizer input, respectively. In the summer-autumn growing season, soil nutrient losses were the highest, accounting for 44% to 89% of the whole year. Redundancy analysis revealed that the most critical factor determining runoff losses was runoff volume. N and P runoff losses were also largely dependent on total soil N (TSN) and Olsen-P, respectively. Therefore, quantitative data for the national N and P runoff losses in vegetable production systems provide a scientific basis for an effective optimization of fertilizer applications.

1. Introduction

The worldwide planted area and yield production of vegetables increased by 65.4% and 111%, respectively, between 1994 and 2013

⁎ Corresponding authors.
E-mail addresses: ylli@issas.ac.cn (Y. Li), wmshi@issas.ac.cn (W. Shi).

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(FAO, 2013; Du et al., 2011), and 45% of the world's total vegetable production areas are located in China (FAO, 2015). The area devoted to vegetable cultivation had increased to 22.3 million ha by 2016, accounting for 13.4% of the total dryland in China (NBSC, 2017). Substantial economic benefits and increasing demand for vegetables have prompted farmers to expand vegetable cultivation at the expense of wetlands (Wu et al., 2017; Wu et al., 2018). An examination of land-use data shows that the total vegetable growing area in China increased from 3.5 million ha to 17.9 million ha, while the area used for rice cultivation decreased from 33.3 million ha to 26.5 million ha between 1985 and 2003 (Sun et al., 2011). Accordingly, vegetable production around the world has experienced rapid growth. Total production in China amounted to 7.85 Tg (1 Tg $= 10^{12}$ g) in 2015, 205.4% higher than in 1995, and annual vegetable consumption now averages 600 kg per capita, placing China as the leading vegetable producer in the world (Chen and Chen, 2018).

Nevertheless, most vegetable production systems are an important source of nitrogen (N) and phosphorus (P) runoff, with losses of N and P to air and water being associated with a series of environmental, ecological, and health impacts (Neeteson and Carton, 2001). In the food chain, the largest output occurs through losses of N to the air and to, groundwater and surface water, estimated at 42.8 Tg in 2005, almost tripled the amount in 1980 (Ma et al., 2012). However, farmers still apply large amounts of N and P fertilizer to obtain high yields. According to survey data for 2013–2015, the average N and P_2O_5 input are, respectively, 2.7 and 5.9 times those recommended for open-field vegetable production in China (Huang et al., 2017). While this might be safe from an agronomic point of view, from the perspective of water quality, even very small amounts of N and P input can create adverse impacts on the environment (Hart et al., 2004; Neeteson and Carton, 2001). For example, Du et al. (2011) investigated groundwater pollution in the Beijing area, and found that the nitrate content of groundwater in vegetable fields was 13.8 mg L^{-1}, 2.8-fold higher than that in croplands. Zhang et al. (2009) surveyed surface water from open vegetable fields in the Taihu Lake region and found that total N (TN) and total P (TP) concentration were 13.45 mg L^{-1} and 1.51 mg L^{-1} on average, respectively, exceeding the threshold value (TN was 2 mg L^{-1}, and TP was 0.4 mg L^{-1}) for surface water of Grade V (GB3838–2002, China).

Most point-source pollution is relatively simple to measure and regulate and can often be controlled by treatment at the source, whereas the problem of non-point source pollution is more difficult to tackle and has attracted increased attention worldwide (Carpenter et al., 1998; Li and Li, 2014). Non-point source pollution has become the primary cause of water pollution. In the Netherlands, agriculture causes 60% of TN emissions and 40%–50% of TP emissions to the surface waters (Boers, 1996). In 270 Danish streams, diffuse sources accounted for 94% of riverine N loading and 52% of riverine P loading (Kronvang et al., 1996). Statistical findings suggest that rural non-point source pollution comprises about half of the total water pollution in China (Shi et al., 2011). Therefore, N and P management in farming system is of major global concern. Generally, there are two direct pathways of N and P transfer from agricultural soil into surrounding water systems, the first of which is surface runoff and the other leaching (Wang et al., 2015). According to data from the first National Survey of Pollution Sources Bulletin (MEP, 2010), the amount of TN loss from cropland is about 1.6 Tg, 20.03% and 12.98% of which are via surface runoff and leaching, respectively, and the remainder of N loss via atmospheric pathways; the amount of TP loss is 0.11 Tg. Although some studies estimate TN via runoff (9.7 Tg yr^{-1} in 2007 (Ti et al., 2012), 12 Tg yr^{-1} in 2010 (Cui et al., 2013)) in China to be higher than in the United States (4.8 Tg yr^{-1} (Gu et al., 2015)), the contribution from different crop systems remains poorly understood. Some studies (David and Correll, 1998; Hu et al., 2002) have shown that P loss is mainly due to runoff from dryland soil. Therefore, investigations of the factors influencing N and P losses in runoff are a key prerequisite for taking measures to control runoff losses. There have been many reports about N losses from agricultural soil, but most of those have focused on N surface runoff from paddy soils (Zheng and Zhou, 2007; Fan et al., 2015), and very few have quantified P losses. Although ocean bays and estuaries contribute to riverine N and P loading in China, with both oligotrophic and mesotrophic conditions being established, this problem has attracted little attention in runoff research. Regions where eutrophication is known to be serious, near Taihu Lake (e.g. Jiangsu Province, Zhejiang Province), Chaohu Lake (e.g. Anhui Province), and Dianchi (e.g. Yunan Province). Thus, the national total load of N and P runoff loss from open-field vegetable systems is still unknown. Therefore, a quantification of N and P surface runoff in different systems is important for understanding the phenomenon of water pollution. The present study was designed to (1) assess N and P fertilizer input under different cropping systems (vegetable, corn, and rice production systems), (2) compare and quantify N and P surface runoff losses in different cropping systems, (3) analyze the factors controlling N and P runoff losses, and (4) assess different management practices for reducing N and P losses.

2. Materials and methods

2.1. Data collection

To evaluate the extent of N and P runoff losses in Chinese vegetable production systems, we summarized literature published between 1990 and 2018 from 56 journal articles focusing on vegetables, 48 journal articles upland crops (maize, wheat, soybean and peanut), and 47 journal articles rice, in 13 Chinese provinces (Fig. 1), based on the Web of Science, Google Scholar, and the China Knowledge Resource Integrated (CNKI) databases. With the introduction of the “shopping basket program” in 1988 (Yu, 2003), the studies of vegetables increased. Thus, we chose 1990 as the starting point for our analysis. To minimize bias, we used the following criteria for the inclusion of studies: (1) N and P runoff loading were measured during the entire growing season for vegetables, upland, rice, and vegetable production systems were open-field (the total open-field vegetable plant area accounts for approximately 77.4% of the total vegetable plant area (CMA, 2014)). (2) Information on the input of N and P fertilizer (we focused on the result under current conventional fertilization regimes used by famers in annual crop systems to avoid other influence factors, i.e., timing and mode of application), growing period, TN and TP runoff flux, and the form of N and P losses could be extracted. N fertilizers farmers used were urea, 15-15-15 compound fertilizer, and organic fertilizer. To provide comparable data, we converted them into pure N. Measurement methods are summarized in Supplementary Information (SI). Using these criteria, a total of 309 data sets were collected (Table Supplementary 1). Growing periods of different types of vegetables vary significantly. Therefore, we divided into three main types of vegetables (leafy, fruit, and stem/root vegetables) and four cropping seasons (spring-summer, summer-autumn, autumn-winter, and winter-spring seasons) by growing period in our statistics. The time of spring-summer season ranges from January to June, summer-autumn season ranges from April to September, the autumn-winter season ranges from July to December, and the winter-spring season ranges from October to March.

2.2. Data analysis

2.2.1. N and P surface runoff loadings and their characteristics

Information on the input of N and P is based on applications of conventional fertilizer by farmers in annual crops, and the runoff loading per season in corresponding systems was determined. We used mean runoff loading as coefficient multiply cultivated area approach to estimate the national total N and P loading from three cropping systems in China. TN runoff loading per year was calculated as follows:

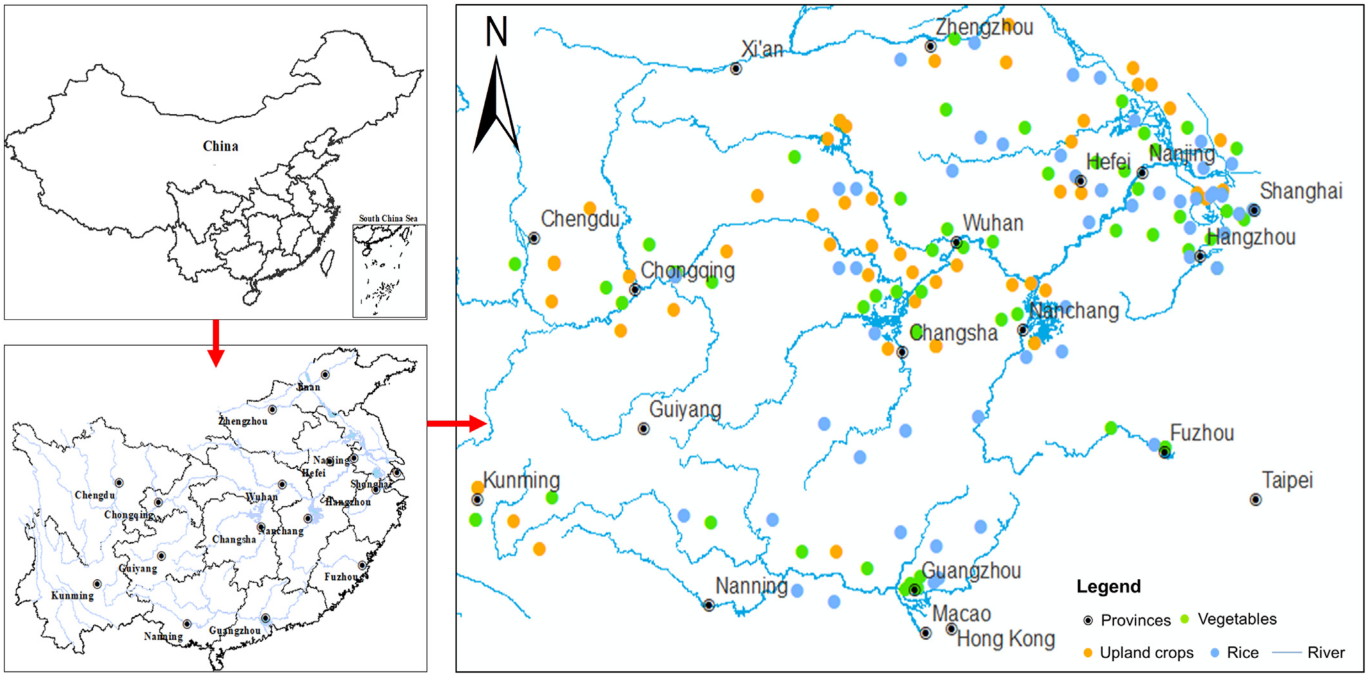

Fig. 1. Location of the study site on N and P losses via runoff. Data summarized from literature published in 1990–2018 from 151 journal articles conducted on vegetable fields, upland crops, and rice in 13 Chinese provinces, using Web of Science, Google Scholar, and the China Knowledge Resource Integrated (CNKI). 43 sites for vegetable fields are illustrated as solid green circles, whereas 55 sites for upland crops are indicated as solid yellow circles, and 68 sites for wetland are indicated as solid blue circles.

TN_Y runoff loading (Tg yr^{-1}) = TN runoff loss per season×multiple crop index × plant area. TN_Y stands for TN runoff loss per year.

$$\text{N runoff loss ratio } (\%) = \frac{NRL_{TN}}{A} \times 100\%,$$

$$\text{N net runoff loss ratio } (\%) = \frac{NRL_{TN(TP)} - NRL_{CK}}{A} \times 100\%,$$

NRL_{TN} is the TN runoff loss from a given N treatment (N fertilizer application is as conventional fertilizer) (kg ha^{-1}), NRL_{CK} is the TN runoff loss from the control treatment (no N fertilizer input) (kg ha^{-1}), A is the total amount of N applied to the plot (kg ha^{-1}). The identical procedure was used for P.

Multiple crop index (%) = $\frac{Plant\ area}{\text{Cultivated } area}$ ×100, vegetable plant area is 1.9×10^6 ha (CMA, 2011), and the area cultivated to vegetables is 3.548×10^6 ha (Wang et al., 2018). Therefore, the average multiple crop index for vegetable cultivation is 536%. The multiple crop index for upland crops and rice is 132%, from the research Li et al. (2016). The data in plant area is from China Statistical Yearbook (NBSC, 2017).

2.2.2. The influence factors determining N and P surface runoff losses

There are many factors influencing soil nutrient losses, such as rainfall intensity, soil physical properties, vegetative cover and agricultural management practice (Udawatta and Motavalli, 2006). Rank analysis was used to test the correlations between soil nutrient losses and the impact factors (Zhang et al., 2016b). Rank analysis included canonical correlation analysis (CCA) and redundancy analysis (RDA) to examine how much of the variation in one set of variables (X) explains the variation in another set of variables (Y) (Zuur et al., 2007). Method selection depended on the length of the gradient calculated by detrended correspondence analysis (DCA). If the length is smaller than 3.0, the RDA method is preferred; if the length is between 3.0 and 4.0, both RDA and CCA are suitable; otherwise, CCA is better (Lepš and Šmilauer, 2003). In our study, the gradient length calculated by DCA was smaller than 3.0, so we choose RDA to analyze the correlations between soil N and P runoff losses and the influencing factors. We chose the runoff volume, fertilizer inputs, soil physical properties, including pH, total soil N (TSN), total soil P (TSP), NO_3^--N, NH_4^+-N, Olsen-P, and soil organic matter (SOM) as the influencing factors. We quantified the relative contributions of impact factors and identified the significance of the various factors by making comparisons among all these contributions by rank analysis. The arrow indicates the influence factor. The longer the line is, the greater the contribution of the factor.

To identify and explore the correlation of runoff loss and the impact factors, network analysis was performed via the Molecular Ecological Network Analysis Pipeline based on random matrix theory (http://ieg4.rccc.ou.edu/mena/). The details of the approach are as described by Deng et al. (2012) and Zhou et al. (2010). Briefly, the network was constructed as follows: First, the dataset was prepared in the required format, instructed by the pipeline, and then submitted to the dataset. The network was then constructed with the default settings, and the threshold of similarity was chosen as 0.001. The topological properties of the network were calculated following the instruction. Then, node and edge files of the network, such as the R square of the power-law, node and edge number, average path distance, and average degree were generated on the pipeline. Finally, the files on node and edge attributes were modified to meet the requirement of Gephi, and the network plots were ultimately visualized with the interactive Gephi platform (Bastian et al., 2009). The data were analyzed using Microsoft Excel 2010. The significant correlations were analyzed by SigmaPlot (Version 12.5). RDA was programmed by R software (version 3.5.1).

3. Results and discussion

3.1. N runoff losses and their characteristics in vegetable production systems

Results summarized from 56 published studies in China show the input of N fertilizer into vegetable fields varied widely, from 38.3 to 728 kg ha^{-1} per season, with a mean value of 264.3 kg ha^{-1} (Table 1), and fruit vegetables received greater amounts of fertilizer (Table 2). This contrasts to N inputs of 210.2 kg ha^{-1} and 186.5 kg ha^{-1} used for upland crops and rice, respectively. Vegetable production systems

Table 1
N runoff flux and loss ratios of fertilizer N in different cropland systems.

System	n	N fertilizer input[b] (kg N ha^{-1})	TN_S Runoff loading (kg ha^{-1})	Percentage (%)		R_{NRL}[d] (%)	R_{NNRL}[d] (%)	Multiple crop index (%)	Plant area (10^3 ha)	TN_Y Runoff loading (Tg N yr^{-1})
				NO_3^--N	NH_4^+-N					
Vegetable	76	264.3 (38.3–728)[c]	16.5(0.11–90.1)	49.2	4.30	6.27	3.79	536	17,281	1.52
Upland crops[a]	66	210.2(45–610.6)	10.8(0.18–49.5)	50.8	6.44	5.32	4.14	132	82,856	1.18
Rice	65	186.5(69–300)	15.6(0.43–88.2)	36.4	31.7	8.79	4.61	132	30,178	0.62

[a] Upland crops include wheat, maize, soybean, and peanut.
[b] Input of N fertilizer application is as conventional fertilizer by farmers in annual crops, and the TN runoff flux is per season.
[c] Values in parentheses are reported ranges.
[d] R_{NRL} is the N runoff loss ratio (%), and R_{NNRL} is the N net runoff loss ratio (%).

have high fertilizer N inputs largely because vegetables have a higher inherent N demand (Lei et al., 2010; Shi et al., 2009) and lower root density (Yu et al., 2013) than cereal crops. Meanwhile, the variable input quantities of fertilizer within each cropping system are a universal phenomenon in China (Ju et al., 2007) and this might be attributed to the difference in fertilization habits, soil types, and planting system etc. Of particular note, the amount of TN runoff loss under conventional fertilizer application from vegetable systems at its highest reached to 16.5 kg ha^{-1} per season, and the TN runoff loss was 1.52 Tg yr^{-1}, 1.29 and 2.45 times those from upland crops and rice systems, respectively (Table 1). Our estimate of TN runoff (3.32 Tg yr^{-1}) at the national level in China is higher than that reported in Gu et al. (2015) (2.4 Tg yr^{-1}). One reason for this might be that runoff loading for vegetable systems was underestimated in that study as the same parameter was used for upland as for vegetable land. Another reason could be that we include new data reflecting eight additional years covering the period of rapid expansion of vegetable production in China. The estimate of TN runoff loss from rice is consistent with that of Hou et al. (2017), there are 0.67 ± 0.05 Tg N yr^{-1} for paddy fields. The N runoff loading for fruit vegetables reached 37.8 kg ha^{-1}, 2.60 and 6.31 times than that for leafy and stem/root vegetables, respectively (Table 2). This might be attributable to the growing period of fruit vegetables being mostly in the summer (Fig. 2(a)) where strong rainfalls prevail. In the summer-autumn season, the N runoff loading reached its highest value, accounting for 44% of the total across all seasons (Fig. 2(c)). However, the net loss ratio (the difference between conventional fertilizer treatment to no fertilizer treatment divided by the total amount of fertilizer input) of N from vegetable fields was the lowest, for two reasons: First, the relatively poor growth of plants without fertilizer increases the risk of N loss in surface runoff (Yi et al., 2018). Second, plant coverage by upland crops and rice is higher than for vegetables and can slow down the runoff velocity and thereby reduce surface runoff.

The percentages of TN represented by NO_3^--N runoff losses were 49.2%, 50.8%, and 36.4% from vegetable, upland crop, and rice systems, respectively. By contrast, the percentages of TN in runoff represented by NH_4^+-N runoff losses were 4.30% in vegetable systems, 6.44% for upland crops, and 31.7% for rice (Table 1). These results indicate that NO_3^--N was the main form of N in the runoff. The form of N via surface runoff can be divided into particulate N (PN) and total dissolved N (TDN), and TDN is mainly composed of NO_3^--N, NH_4^+-N, and dissolved organic N (DON) (Zheng et al., 2014). In the present study, the N in the runoff consisted predominantly of soluble inorganic N, and the main form was NO_3^--N. The results are consistent with those reported in other studies. Jiao et al. (2010) reported that the DON and NO_3^--N were the main forms of dissolved N and inorganic N, respectively, from dryland systems under natural rainfall. NO_3^--N loading loss was highly significantly correlated with the amount of TN (Fig. S1). Consequently, NO_3^--N runoff can be used to estimate TN runoff (Hou et al., 2017). Conversely, the percentages of TN represented by NH_4^+-N runoff losses were lower than the percentages represented by NO_3^--N. The reason is that NH_4^+-N is more easily adsorbed by soil colloids and soil particles than NO_3^--N, but NH_4^+-N can also be converted into NO_3^--N by nitrification (Si et al., 2000). However, Li et al. (2006) reported that PN was the dominant fraction in the runoff from vegetable fields, comprising 64% of total TN runoff losses, and NH_4^+-N was the major form of soluble N, accounting for 50% of soluble N. These results can be attributed to the preferential loss of easily mobile NH_4^+-N during successive rainfall events. Rainfall is the primary factor reducing ammonia volatilization and nitrification rate, therefore increasing NH_4^+-N runoff (Wang et al., 2011). Meanwhile, organic fertilizers have been reported to increase the runoff loss of NH_4^+-N, due to the fact that organic fertilizers contain small quantities of NH_4^+ and can also be readily mineralized following fertilizer application and generate additional quantities of NH_4^+ (Shan et al., 2015).

3.2. P runoff losses and their characteristics in vegetable production systems

Similar to the above scenarios regarding N, a high extent of P fertilizer application, and variability therein, was also found in the different systems. P fertilizer input was highest in vegetable fields (101.0 kg P ha^{-1}) (Table 3), especially for fruit vegetables (126.0 kg P ha^{-1}) (Table 2). TP loading in runoff ranged between 0.01 and 45.6 kg ha^{-1} in the different systems, and the runoff from vegetable systems reached the highest value (on average 3.45 kg ha^{-1}) during the growing period, and the TP runoff loss was 0.33 Tg yr^{-1}, accounting for 66% of total farmland P runoff losses. TP losses from vegetable systems equaled 4.08% of the P fertilizer input, and the loss ratio for fruit vegetables reached 13.1% (Table 2), whereas only 1.91% and 1.49% of applied P was lost from upland crop and rice systems, respectively. R_{PNRL} was negative for vegetable systems (Table 1), i.e., P runoff loading without fertilizer was higher than that with fertilizer. Poor growth of vegetables might have increased the risk of P losses in surface runoff, as vegetation cover can

Table 2
N and P runoff loading and loss ratios of different vegetable types.

Vegetable type[a]	n	N fertilizer input (kg N ha^{-1})	TN runoff loading (kg N ha^{-1})	R_{NRL} (%)	n	P fertilizer input (kg P ha^{-1})	TP runoff loading (kg P ha^{-1})	R_{PRL} (%)
Leafy vegetables	29	226.9(38.3–531.2)	10.5(0.63–43.1)	4.63	14	78.9(0–155.8)	2.03(0.07–27.5)	2.57
Fruit vegetables	17	287.7(53.6–630)	37.8(1.74–90.1)	13.1	10	126.0(27.5–282)	4.97(0.09–7.32)	3.95
Stem/Root vegetables	7	279.8(116.10–341)	5.17(0.36–16.13)	1.85	7	83.9(33.8–137.4)	0.06(0.01–0.18)	0.07

[a] Leafy vegetables include Chinese cabbage (*Brassica chinensis* L.), cabbage (*Brassica oleracea* L.), amaranth (*Amaranthus tricolor* L.), mater convolvulus (*Ipomoea aquatica* Forsk), leaf mustard (*Brassica juncea* L.) and broccoli (*Brassica oleracea* var. *italic* Plenck). Fruit vegetables included eggplant (*Solanum melongena* L.), pepper (*Capsicum annuum* L.), squash (*Cucurbita pepo* L), wax gourd (Benineasa hispida Cogn.), and bitter melon (*Momordica charantia* L.). Stem/Root vegetables include tuber mustard (*Brassica juncea* var. tumida) and daikon (*Raphanus sativus* L).

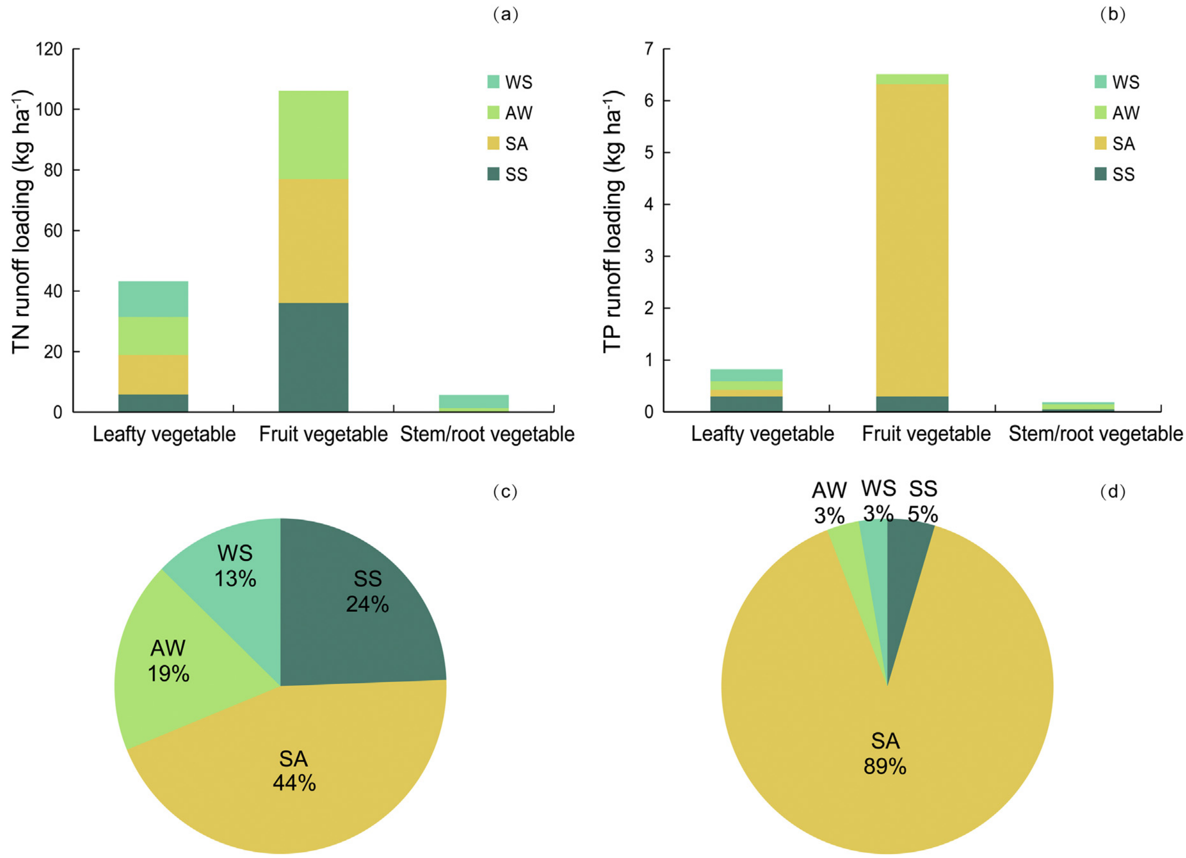

Fig. 2. The characteristics of vegetables' TN/TP runoff losses in four cropping seasons. (a) TN runoff loading of different vegetable types in different growing seasons. We divided vegetables into four cropping seasons, including spring-summer, summer-autumn, autumn-winter, and winter-spring seasons. SS ($n = 15$) represents the spring-summer season, SA ($n = 12$) represents the summer-autumn season, AW ($n = 13$) represents the autumn-winter season, WS ($n = 14$) represents the winter-spring season. (b) TP runoff loading of different vegetable types in different growing seasons. SS ($n = 5$) represents the spring-summer season, SA ($n = 10$) represents the summer-autumn season, AW ($n = 8$) represents the autumn-winter season, WS ($n = 9$) represents the winter-spring season. (c) TN runoff loading in different growing seasons. In the summer-autumn season, the N runoff loading reached its peak, accounting for 44% of the full season. (d) TP runoff loading in different growing seasons. In the summer-autumn season, the P runoff loading reached its peak, accounting for 89% of the full season.

effectively reduce runoff and increase the adsorption of P in soil (Zeng et al., 2012). Li et al. (2010b) found that TP runoff losses correlated inversely with size of cropping area. In any of systems, a well-calibrated fertilizer regime is obviously critical to reducing P runoff losses. In terms of variability across the growing season, P loss loading reached its peak value in the summer-autumn season, accounting for 89% of the total season (Fig. 2(d)). PP was the main form of P in the runoff, and PP runoff losses exceeded half of TP, which ranged from 66.7% to 79.3% (Table 3). P runoff loading for fruit vegetable systems was 4.97 kg ha^{-1}, which was 2 and 83 times that of leafy and stem/root vegetables, respectively (Table 2). This might be attributed to the growing period for fruit vegetables, which is mostly restricted to the summer season (Fig. 2(b)). R_{PRL} for stem/root vegetables was only 0.07%, for two reasons: (1) the growing seasons for stem/root vegetables are not in the rainy season (Fig. 2(b)), and (2) small sample size might influence the results.

The transport of P in runoff occurs in particulate and dissolved forms (Schindler, 1977). PP includes P associated with soil particles and organic material during runoff, and might constitute a variable but long-term source of potentially bioavailable P in downstream water bodies (Sharpley et al., 1992). In our analysis, over half of the P losses via runoff were in PP, while TDP was a minor contributor. The results are consisted with those reported in other studies. Wang et al. (2015) reported that PP was the dominant form of P in runoff, accounting for 60%–75% of TP from a dryland system in the Chaohu Lake region. Although the amount of P lost in dissolved forms was relatively small (TDP only accounted for 25.4%–34.0%), TDP could be available for biological uptake (Daverede et al., 2004). The PP and TP losses tended to follow similar

Table 3
P runoff flux and loss ratios of fertilizer P in different cropland systems.

System	n	P fertilizer input[b] (kg P ha^{-1})	TP_S	Percentage (%)		R_{PRL}[d] (%)	R_{PNRL}[d] (%)	Multiple crop index (%)	Plant area (10^3 ha)	TP_Y
			Runoff loading (kg ha^{-1})	TDP	PP					Runoff loading (Tg P yr^{-1})
Vegetable	50	101.0(0–555)[c]	3.45(0.01–45.6)	21.6	78.4	4.08	−0.45	536	17,281	0.33
Upland crops[a]	76	54.6(0–160)	1.05(0.01–16.4)	21.7	78.3	1.91	1.36	132	82,856	0.12
Rice	93	77.8(0–300)	1.26(0.01–12.6)	22.2	66.7	1.49	1.11	132	30,178	0.05

[a] Upland crops include wheat, maize, soybean, and peanut.
[b] Input of P fertilizer application is as conventional fertilizer by farmers in annual crops, and the TP runoff flux is per season.
[c] Values in parentheses are reported ranges.
[d] R_{PRL} is the P runoff loss ratio (%), and R_{PNRL} is the P net runoff loss ratio (%).

trends, and correlation analysis indicated that PP losses were significantly correlated with TP via runoff (Fig. S2). In our study, the accurate estimation of TP runoff loading at the national level might be helpful for our improved understanding of P cycling in various types of agroecosystems and the development of best practice in the management of global P reserves.

3.3. Factors impacting N and P surface runoff losses

We used redundancy analysis to test the correlations between soil N and P runoff losses and the impact factors, and to quantify the contributions. The nutrient loss via runoff is mainly due to high fertilizer input and runoff volume, while background soil (TSN, NO_3^--N, NH_4^+-N, Olsen-P) properties also are important contributors (Fig. 3). In the present study, larger TN and TP runoff losses were found for vegetables compared to upland crops and rice, which can be attributed to several factors: First, high input of fertilizer N and P would result in high fertilizer-induced runoff losses; our results show the amounts of fertilizer application for vegetables to be 0.26–0.85 times greater than in other crop systems (Tables 2, 3). Second, background N and P runoff losses were greater from vegetables than upland crops and rice in this study; according to our data, high background N and P runoff losses, at 16.5 kg N ha^{-1} and 9.56 kg P ha^{-1}, respectively, in vegetable fields exceed those in other cropping systems (for upland crops, these are: 8.24 kg N ha^{-1} and 0.95 kg P ha^{-1}; for rice: 11.7 kg N ha^{-1} and 1.26 kg P ha^{-1}) (Table S2). High background runoff losses were likely related to background soil (TSN, NO_3^--N, NH_4^+-N, Olsen-P) properties, related to a long history of high N and P fertilizer input (Zhou et al., 2016; Yan et al., 2013). Redundancy analysis also indicated that soil N pool contributed to N runoff loss more than fertilizer input but that fertilizer P input contributed to P runoff loss more than the soil P pool (Fig. 3(a), (b)). We speculated that some of the reasons might be as follows: (1) Excessive, continuous, and high-frequency N fertilizer applications to the intensive vegetable systems result in a large amounts of N accumulated in the vegetable soil, consequently create a large soil N pool, N fertilizer is applied generally in the form of NH_4^+-N form, which is easily adsorbed or fixed by soil clay (Du et al., 2010), thus fertilizer N should have lower mobility than that of the endogenous soil N pool. (2) In the case of P, most P in the soil pool could be fixed by iron and manganese oxides or be locked up in other insoluble forms (McBeath et al., 2005) and precipitation and adsorption render it especially ineffective in calcareous soils. However, fertilizer P can also be transformed into insoluble forms quickly, and the availability and mobility of newly formed insoluble P in soil is higher than for the resident soil P pool (Yu et al., 2013), for which reason P fertilizer may contribute more to runoff losses. Therefore, recommended management practices for reducing N and P runoff losses must be different. For N, these include applying organic manure and planting legume crops to increase the capacity of the N pool. For P, reducing P fertilizer application is imperative, such as applying biochar and root growth stimulator and selection of high P utilization efficient vegetable varieties.

In the present study, the most critical factor was the runoff volume, in addition to fertilizer use rate (Fig. 3(a), (b)). In general, one mechanism for surface runoff is "infiltration-excess", and it occurs when rainfall intensity exceeds the infiltration capacity of a soil (Horton, 1933). Hence, rainfall density, surface slope, and vegetation coverage have been considered the most important factors determining soil nutrient runoff losses (Franklin et al., 2007). Therefore, N and P runoff losses reached their peaks in the summer-autumn season, accounting for 44% and 89%, respectively, of the whole year's balance (Fig. 2(c), (d)).

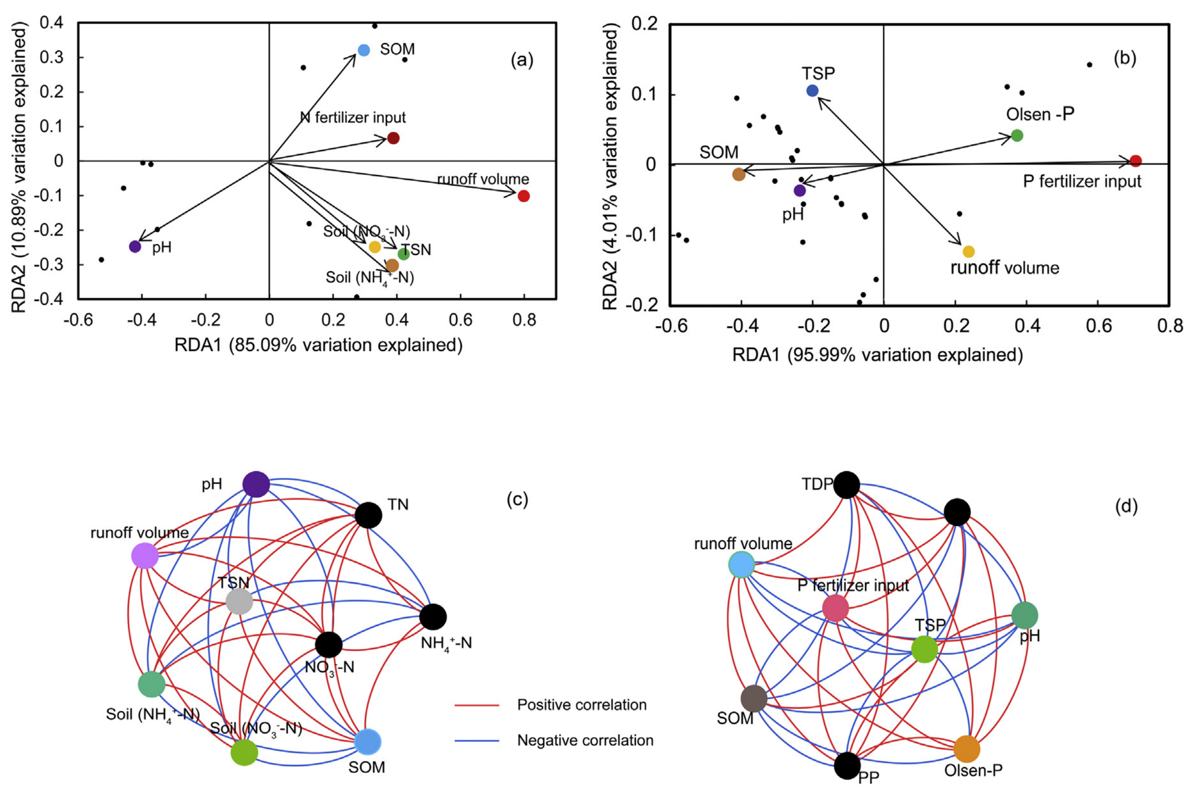

Fig. 3. The impact factors determining N/P runoff losses by redundancy analysis (a)/(b) and the network graph by N (c)/P (d). (a) We used redundancy analysis to test the correlations between soil nutrient runoff losses and the impact factors, and to identify the contributions of impact factors. The longer the line is, the greater the contribution of the factor is. The black and small node is measured data. Each colorful node signifies the influencing factor. The most critical factor was the runoff volume, which is related to rainfall. For N runoff losses, the impact of total soil N was higher than the influence of N fertilizer input. (b) For P runoff losses, the most critical factor was the P fertilizer application. Olsen-P in the soil also was an important factor. (c) Different colors of the nodes indicate different factors including SOM, TSN, runoff volume, pH, the application of N/P fertilizer, NO_3^--N, and NH_4^+-N, of soil. A blue edge indicates a positive interaction between two individual nodes, while a red edge indicates a negative interaction. (d) Different colors of the nodes indicate different factors including SOM, TSP, runoff volume, pH, the application of P fertilizer, Olsen-P of soil.

3.4. Control measures to reduce nitrogen and phosphorus runoff losses

Techniques used for reducing and controlling N and P runoff losses can be classified into two types: in-field management practices and edge-of-field management practices (Fig. 4). In-field management practices are useful to reduce the potential pollution loads and pollutant production at the source, i.e., optimized cropping structure and conservation tillage. Conservation tillage and optimized cropping structure could reduce surface runoff velocity and prolong the interaction between runoff and topsoil (Ali et al., 2007). On the other hand, conservation tillage significantly reduces the amount of erosion which also reduces the amount of nutrient loss via runoff (Zhang et al., 2004). Optimized cropping structure can improve the utilization efficiency of water and fertilizer resources and change the distribution characteristics of roots (Zhan et al., 2012), which is especially important for vegetables to maximize uptake of P. Edge-of-field management practices rely on controlling the migration process to reduce losses through improvements in ecological engineering systems, including, vegetated filter strips, ecological ditches, and constructed wetlands (Wu et al., 2013; Zhang et al., 2016a). Vegetated filter strips reduce the loss of more nutrients mainly via physical processes such as slower runoff velocities, settling of sediment-bound nutrients and the facilitation of plant uptake (Wu et al., 2013). Results show that the load reduction rate for N and P pollutants in surface runoff through vegetated filter strips achieved 69.9% and 85.1%, respectively (Li et al., 2010a), and the effect was stronger for P than N (Ni et al., 2002), the possible reason for which may be that P is easily taken up by plants and intercepted by substrate adsorption (Li et al., 2011). In other countries, such as the USA, Canada, and France, vegetated filter strips have been included under best management practices (BMPs) to prevent non-point pollution from rivers and other water bodies (Abu-Zreig et al., 2003; Lacas et al., 2005; Mayer et al., 2007). Ecological ditches are effective for trapping and removing N and P (Fu et al., 2014). N and P removal mechanisms are considered to occur via three main pathways: adsorption by sediment, uptake by plants, and enhanced sedimentation due to a slow-down in water flow velocity (Wang et al., 2010). For N, the reduction mechanism also includes the denitrification process, by virtue of which N is transformed into N_2, NH_3, and N_2O gas (Hu et al., 2010). Different treatment techniques are suitable for different locations. For example, vegetated filter strips with relatively large space requirements are suited to areas with a high potential pollutant load; ecological ditches, which are easy to construct, might lead to water loss (leakage and evaporation) and must be avoided in drought-affected areas; constructed wetlands with their large space requirements are suited to remote rural areas (Wu et al., 2013). Constructed wetlands have been proposed as among the most attractive options available to growers in California to reduce pollutant discharge (O'Geen et al., 2010). By contrast, in China, some studies found that combined measures were superior to singular measures. Min et al. (2015) found that a combined ecological ditch-paddy field wetland system resulted in a hydrologically closed system, and 79.3% of the TN was eliminated, superior to the application of only an ecological ditch method (49.9%). This represents a new, but promising, approach for the alleviation of non-point-source pollutants.

Agronomic practices are a direct consequence of farmers' decisions concerning management strategies. Therefore, policies may have to set limits on the extent of non-point pollution. Various policies have been developed to reduce point-source water pollution in different nations. In the state of Iowa, USA, insurance policies help farmers adopt a best N management plan (BNMP) that reduces non-point pollution in agriculture, benefiting both farmers and insurance companies (Huang et al., 2001). The Danish government has implemented an N fertilizer application tax for non-point source pollution caused by chemical fertilizers and manure, and imposes variable tax rates (Berntsen et al., 2003). In China, in addition to learning from foreign experience, several methods to reduce runoff losses could also be readily developed. Government could encourage farmers to construct vegetated filter strips by paying a targeted subsidy for land occupation.

4. Conclusions

N and P in vegetable production systems are more prone to losses from cropping soils compared to upland crops and rice systems, especially those based on fruit vegetable production. Our estimates of the TN and TP emissions via runoff from farmland were, respectively, 3.32

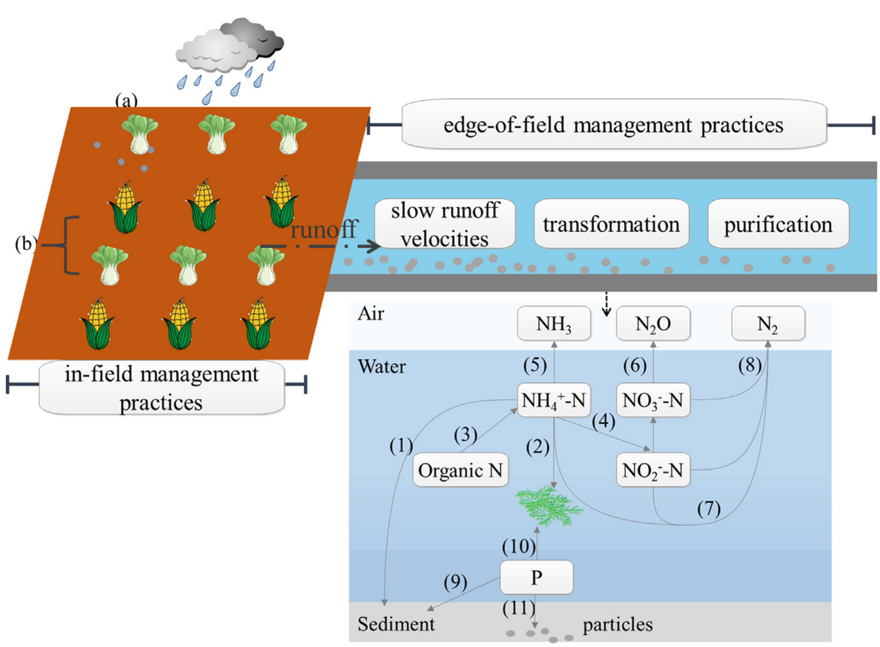

Fig. 4. Schemes for reducing N and P runoff losses. Techniques used for reducing and controlling N and P runoff losses can be classified into two types: in-field management practices and edge-of-field management practices. In-field management practices are useful to reduce potential pollution loads and pollutants production at the source. In-field management practices include (a) reduced fertilizer application; (b) adjusted planting structure. Edge-of-field management practices include (1) adsorption by sediment; (2) uptake by plants; (3) ammonification; (4) nitrification; (5) volatilization; (6) denitrification; (7) anaerobic ammonia oxidation (ANAMMOX); (8) volatilization; (9) adsorption by sediment; (10) uptake by plants; and (11) precipitation.

Tg N yr^{-1} and 0.5 Tg P yr^{-1} in China, and the emissions from vegetables account for 46% and 66%, respectively. N runoff losses from fruit vegetables were 2.60 and 2.31 times greater than those from systems for leafy vegetables and stem/root vegetables, respectively, and 0.59 and 81.8 times greater for P runoff losses. Therefore, controlling non-point-source pollution from vegetable fields should be given priority, especially so for fruit vegetable production. N and P runoff losses were highest in the summer-autumn season, because of a higher intensity of rainfall. Redundancy analysis indicates that the soil N pool contributes to N runoff losses more than fertilizer input, whereas fertilizer P input contributes to P runoff losses more than the soil P pool. Management of N and P, and especially so P, in vegetable systems are of a major concern globally. We hope that the current analysis will be helpful not only for our improved understanding of N and P cycling in agroecosystems but also for the development of more sustainable vegetable production systems.

Abbreviations

N	nitrogen
P	phosphorus
TN	total nitrogen
TP	total phosphorus
NO_3^--N	nitrate N
NH_4^+-N	ammonium N
TDP	total dissolved phosphorus
PP	particulate phosphorus

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2019.01.368.

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