Analyzing the phosphorus flow characteristics in the largest freshwater lake (Poyang Lake) watershed of China from 1950 to 2020 through a bottom-up approach of watershed-scale phosphorus substance flow model

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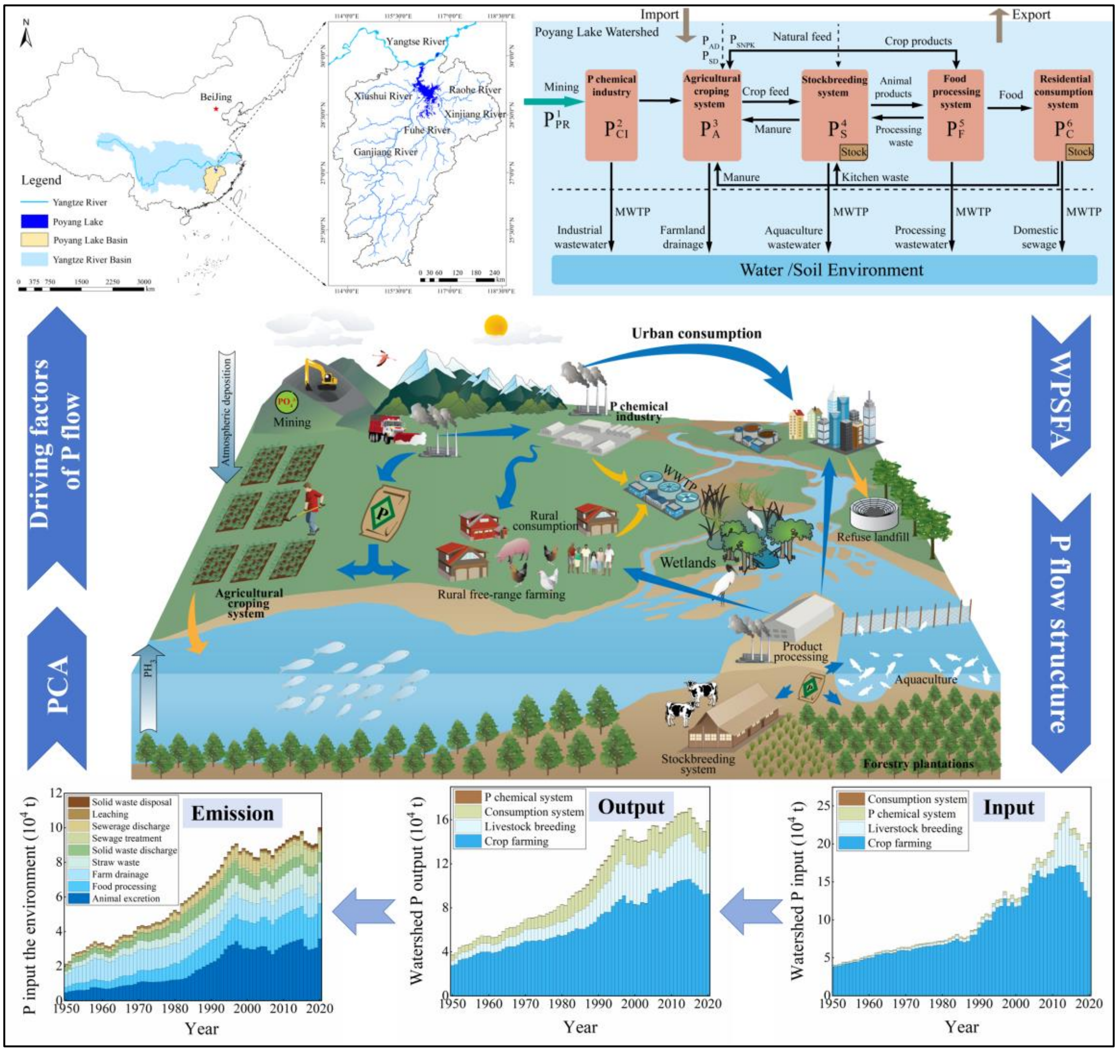
Abstract: Understanding the historical patterns of phosphorus (P) cycling is essential for sustainable P management and eutrophication mitigation in watersheds. Currently, there is a lack of long-term watershed-scale models that analyze the flow of P substances and quantify the socio-economic patterns of P flow. This study adopted a watershed perspective and incorporated crucial economic and social subsystems related to P production, consumption, and emissions throughout the entire life cycle. Based on this approach, a bottom-up watershed P flow analysis model was developed to quantify the P cycle for the first time in the Poyang Lake watershed from 1950 to 2020 and to explore the driving factors that influence its strength by analyzing multi-year P flow results. In general, the P cycle in the Poyang Lake watershed was no longer a naturally dominated cycle but significantly influenced by human activities during the flow dynamics between 1950 and 2015. Agricultural intensification and expansion of large-scale livestock farming continue to enhance the P flow in the study area. Fertilizer P inputs from cultivation account for approximately 60% of the total inputs to farming systems, but phosphate fertilizer utilization continues to decline. Feed P inputs have continued to increase since 2007. The expansion of large-scale farming and the demand for urbanization are the main factors leading to changes in feed P input patterns. The P utilization rate for livestock farming (PUEa) is progressively higher than international levels, with PUEa increasing from 0.64% (1950) to 9.7% (2020). Additionally, per capita food P consumption in the watershed increased from 0.67 kg to 0.80 kg between 1950 and 2020. The anthropogenic P emissions have increased from 1.67×10^4 t (1950) to 8.73×10^4 t (2020), with an average annual growth rate of 2.41%. Watershed-wide P pollution emissions have increased by more than five-fold. Population growth and agricultural development are important drivers of structural changes in P flows in the study area, and they induce changes in social conditions, including agricultural production, dietary structure, and consumption levels, further dominating the cyclic patterns of P use, discharge, and recycling. This study provides a broader and applicable P flow model to measure the characteristics of the P cycle throughout the watershed social system as well as provides methodological support and policy insights for large lakes in rapidly developing areas or countries to easily present P flow structures and sustainably manage P resources.

Keywords: Phosphorus flow; Substance flow analysis; Poyang Lake watershed; Phosphorus resource management



Graphical Abstract

1. Introduction

Anthropogenic phosphorus (P) is essential for human economic and social production, with more than 90% being used for food production (Gao et al., 2020; Huang et al., 2019a; Li et al., 2015; Metson et al., 2012a). As the foundation and key component of the food system, anthropogenic P encompasses the entire processes of phosphate mining, fertilizer production, crop cultivation, livestock and poultry farming, food manufacturing, and waste management. This profoundly influences the relationships between P and the water, soil, and ecosystems (Ma et al., 2014; Powers et al., 2016). With the continuous and rapid development of the global economy and society, the overall demand for P resources has significantly increased in nearly every country and region. It is projected that by 2050, the global demand for P resources will increase by 50 to 100% (Cordell et al., 2009; Haque et al., 2018). Excessive and inefficient use of P in the social system not only puts immense pressure on the sustainable supply of P resources (Wu et al., 2019), but also poses significant threats to water bodies, often resulting in eutrophication and other environmental issues (Bougarne et al., 2019; Liu et al., 2023a; Liu et al., 2023b). Essentially, resource shortage and water pollution caused by a disorder in the P flow systems result in ecological consequences (Gao et al., 2018; Huang et al., 2019b). Therefore, exploring the metabolic structure and flow pathways of P is of great practical importance to address resource scarcity and water pollution issues.

Substance Flow Analysis (SFA) model has been applied as an effective tool for analyzing element-specific pathways within the global system (Jiang and Yuan, 2015; Li et al., 2010; Liu et al., 2008; Theobald et al., 2016; Yuan et al., 2019; Yuan et al., 2011b; Yuan et al., 2011c; Yuan et al., 2014). Meanwhile, the P balance in a river watershed was determined using the P-SFA model (Drolc and Zagorc Koncan, 2002), and the SFA method was applied to quantify the flow of nutrients throughout a socioeconomic ecosystem (McDowell et al., 2002; Yuan et al., 2011a; Yuan et al., 2011c). Early field studies were characterized by subsystems and quantifications that considered only a portion of the natural or anthropogenic P flow in local systems or within a given year. These studies did not pay sufficient attention to the loss of P in the soil and surface water, which has contributed to our understanding of these systems (Jiang and Yuan, 2015). However, even in current research, the majority of studies focus on single subsystems, such as agriculture (Biswas Chowdhury and Zhang, 2021; Li et al., 2021), livestock (Vingerhoets et al., 2023), household consumption (Chen et al., 2021), and waste systems (Vujovic et al., 2020). It is well known that the large-scale socio-economic P cycles are interconnected and driven by multiple subsystems (Jiang and Yuan, 2015; Yuan et al., 2014).

Based on an extensive literature review, Chowdhury et al. (2014; 2016) synthesized a holistic view of all key sectors typically examined in P flow analyses at various geographical scales. Their findings revealed a lack of consistency in the key subsystems that contributed to the intensification of the P flow. We identified a substantial amount of research that examines "country," "regional," and "city" perspectives (Chowdhury et al., 2014; 2016). However, there is a relative scarcity of studies that approach the research from a "watershed" angle in the literature. In fact, quantifying historical patterns of P cycling at the catchment scale allows the estimation of current P loads and identification of drivers of P flows and sources of legacy P (Haygarth et al., 2014). Owing to the availability of official statistics, studies on P-SFA are typically conducted at the city or national level using a top-down approach (the method or process from totality to detail). Statistics at the watershed level are lacking because a watershed is typically defined by geography rather than politics (Liu et al., 2007; Yuan et al., 2014). To address these issues, an analysis of the entire pathway of nutrients in the socioeconomic ecosystem based on a bottom-up concept (the concept or way from details to totality) is essential (Nanda et al., 2020; Yuan et al., 2014). Based on this strategy, in recent years, P flow studies in large lake watersheds have received widespread attention, and whole-watershed P flow studies have been conducted in the Chaohu Lake (Jiang and Yuan, 2015; Yuan et al., 2014), Dianchi Lake (Yan et al., 2021), Erhai Lake (Fan et al., 2021), and Yangtze River (Liu et al., 2022) watersheds. Researchers have analyzed the P flow characteristics of major watersheds and made scientific recommendations for P flow optimization.

To date, no comprehensive P flow accounting has been conducted in the Poyang Lake (the largest freshwater lake in China and the largest lake connecting the Yangtze River) watershed, nor have the P flow characteristics of the socio-economic system and the drivers of the continuous enhancement of P flow been investigated. Fortunately, the high overlap between the boundaries of the Poyang Lake watershed and the national territory of Jiangxi Province provides a favorable basis for the establishment of model boundaries and the acquisition of activity data. Based on this, we can build a more comprehensive and detailed P flow model, which is sufficient for decision makers to obtain information on the characteristics of P flow, environmental P load, and the main drivers of P flow changes in the Poyang Lake watershed. Ultimately, we aimed to develop a universally applicable watershed P-SFA model that is not constrained by geographic features (Text S4). Specifically, we incorporated all social activities related to P production and consumption throughout the process, from mineral resources to the receiving environment. Several aspects distinguish our study from other large lake watershed P-SFA studies. One notable difference is the redivision of sub-systems: the significant differences between urban and rural social systems lead us to consider separate discussions on resident consumption systems, namely urban and rural consumption. Similarly, we divided livestock systems into industrial-scale and household free-range farming systems. We considered the P flow processes in wastewater and solid waste (including sludge). It is worth noting that we paid particular attention to the food processing system and treated it as an independent P flow transfer station rather than overlooking or hiding it within the livestock and consumption systems. The amount of P lost during product processing is astonishing and often overlooked because there are no explicit management guidelines on how to control P loss during food processing (Rothwell et al., 2022; Vingerhoets et al., 2023). This will help us choose appropriate strategies to optimize the P resource use structure and control P pollution.

Most studies on P flow have only analyzed data from a single year, with the exception of a few studies conducted at the national scale (Chowdhury et al., 2014; Mnthambala et al., 2021; Rothwell et al., 2022; Vujovic et al., 2020). Single-year studies on P flow are insufficient to capture long-term variations in P flux because important processes related to P flow at different geographic scales may take several years to manifest their impact (Bai et al., 2016; Chowdhury et al., 2014; Li et al., 2020a). For example, sudden natural disasters, major national policies, and long-term socio-economic, political, and technological factors can significantly influence the nature and magnitude of P flow. Ma et al. (2012) conducted a 25-year analysis (1984-2008) of P flow in China and found that socio-economic factors such as urbanization, improving living standards, and population growth were the main drivers behind the increase in P flow associated with mining, utilization, and waste generation, as well as the decrease in P recovery from waste. Gao et al. (2020) explored the spatiotemporal characteristics of P utilization efficiency and water load in China from 1995 to 2015. Li et al. (2020a) analyzed the P flow characteristics of China's consumption system from 1980 to 2015, providing insights into the changes in China's P consumption structure and the accumulation of legacy P. Long-term analyses are crucial for understanding the long-term fate and magnitude of P flow through systems. Continuous analysis over multiple years is particularly important for conducting future scenario analyses and making long-term P management decisions (Chowdhury et al., 2014; Li et al., 2020a). Jiang and Yuan (2015) simulated P flow dynamics in the Chaohu Lake watershed over a 35-year period and developed P management strategies for 2013 to 2050. However, there is limited research on multi-year P flow analysis at urban and regional scales. Chowdhury et al. (2014) also highlighted the knowledge gap regarding the long-term fate and magnitude of P flow at urban and regional scales. Therefore, it is crucial for current research on P-SFA to focus on a multi-year analysis of P flow and stocks at urban and regional scales.

In this study, we focused on the life cycle of P flows, including their extraction, production, consumption, waste management, water environment, and soil environment, from a watershed-wide social system perspective. We developed a large watershed P substance flow analysis (WPSFA) model with more comprehensive subsystems and finer P flows to quantify anthropogenic P flows, and selected the Poyang Lake watershed for a case study to conduct the longest known time series (1950-2020) analysis to answer (1) the P cycle patterns and drivers in the Poyang Lake watershed for the first time, including the characteristics of P flows in each subsystem; (2) the P pollution load in water bodies from social exogenous P flows; and (3) the coping strategies of large watersheds in the face of enhanced P flows.

2. Method and data source

2.1 System boundary

The study area is the Poyang Lake watershed, whose geographic boundary overlaps strongly with the administrative boundary of Jiangxi Province, China (Text S1, Fig. S1). This study covered the social metabolic subsystems in the entire watershed of Poyang Lake (Fig. 1), and an overview of the study area is described in detail in the supporting documents. At the horizontal scale, it mainly includes chemical, cropping, farming, processing, residential consumption, and P-receiving systems (arable land, non-arable land, and surface water) (Yuan et al., 2011c; Yuan et al., 2014). At the vertical scale, the upper boundary is defined as approximately 1 km above the earth's surface, focusing on atmospheric deposition but not atmospheric circulation, and the lower boundary is the lithosphere, including mineral resources, soil deposition, and rainfall-runoff, but not deep groundwater. The time scale was set from 1950 to 2020. Historical P stocks and flows (1950 to 2020) were calculated using a production-driven top-down approach.

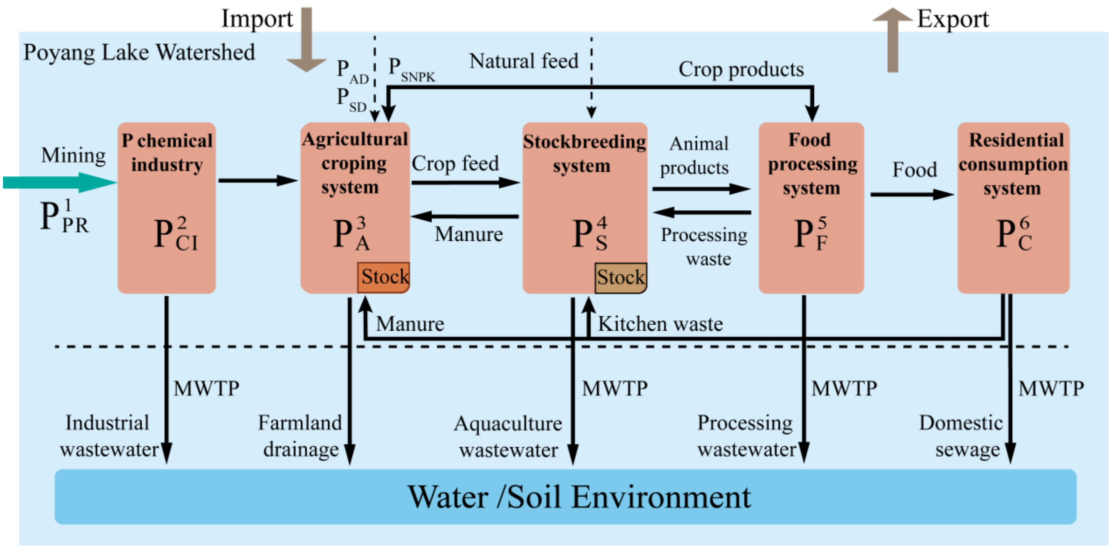


Fig. 1 Static analytical model boundary of anthropogenic P cycles in lake watersheds. P^1_{PR} : P phosphorite resources, P^2_{CI} : P chemical product production system, P^3_A : Agricultural croping system, P^4_S : Stockbreeding system, P^5_F : Food processing system, P^6_C : Residential consumption system, P_{AD} : The amount of P through atmospheric deposition, P_{SD} : The amount of P deposited in the soil, P_{SNPK} : The P content of returned straw, MWTP: Municipal wastewater treatment plant. Numbers represent the order of subsystems.

2.2 WPSFA Model

We developed a WPSFA model based on SFA and previous watershed P analysis methods to provide characteristics of interannual changes in watershed-wide social system P flows and inventories, including environmental stress and uncertainty analyses (Fig. 2) (the introduction and hierarchical structure of the WPSFA model refers to Texts S2 and S3). To provide a more comprehensive characterization of P flows and pollution in large watersheds, our improvements include, but are not limited to: (1) more P-related anthropogenic activities and natural processes are considered in the analytical framework, such as food processing, aquaculture, atmospheric deposition, soil deposition, and rainfall leaching; major subsystems are also refined, such as the farming system, which is divided into large-scale farming and household farming, where food processing is divided into primary and secondary processing, and consumption systems, which are divided into urban and rural consumption. The refinement of these social structures provides a favorable basis for tracking more specific P flows. (2) The system is divided into six subsystems based on the mutual services between these activities, including extraction (mining and P-chemical industries), production (crop farming, livestock, and food/feed industries), consumption, disposal (wastewater treatment and solid waste disposal), P-receiving environment (uncultivated land, surface water, and atmosphere), and exchange (import/export). (3) The calculation of the P flows between subsystems was improved, and independent calculations were performed as far as possible. For example, P flows associated with municipal and rural wastewater are calculated using independent methods, so they can be cross-checked by balancing the inputs and outputs. (4) The WPSFA model enables long-term sequential analysis of P flows in watersheds, allowing for the exploration of the driving factors influencing its strength and variability based on the dynamic changes in P flows over multiple years.

We classified all P flow calculation equations into three types (independent equation, dependent equations, and system balance equation). Among these three computational equations, the independent equation is given priority to reduce the interaction with other P-flows (refer to the support document (Text S5) for detailed P flow calculation).

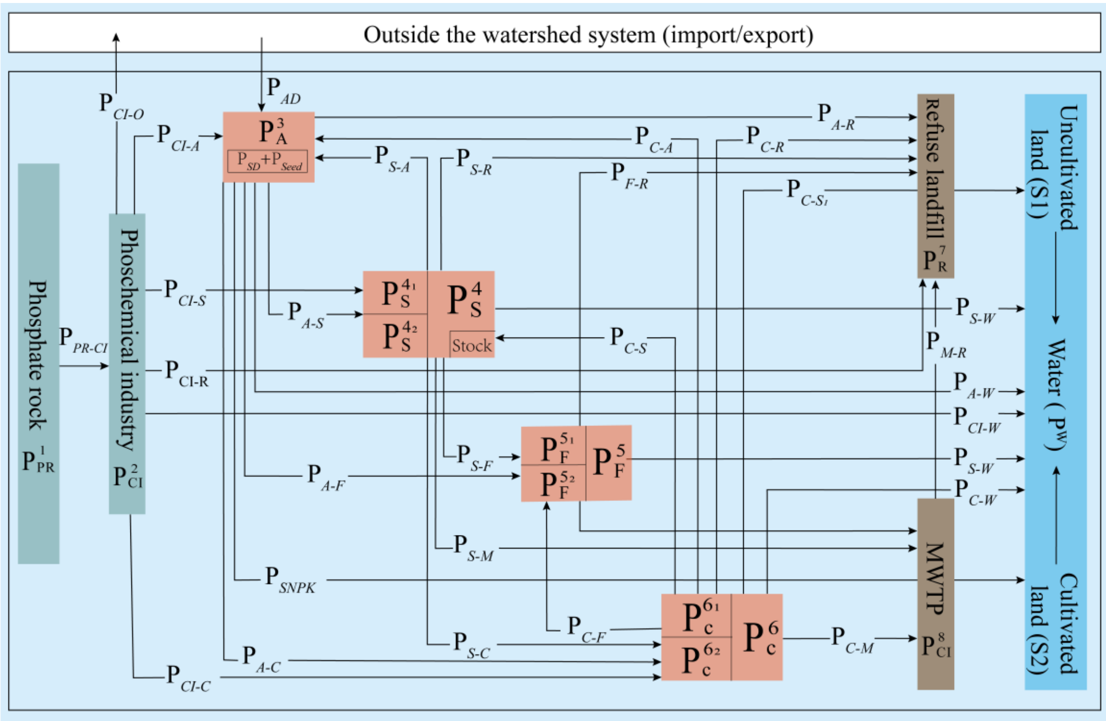


Fig. 2 The P flow paths of WPSFA model. The arrows describe the inputs and outputs of the P flow. For example, P_{CI-A} represents the flow from P_{CI} to P_A, symbol abbreviations are explained in Fig. 1 and Text S4, and the calculation process is described in Text S4. The boxes indicate individual processes or sub-processes in which

some human activities related to P flows occur. These sub-processes may change slightly in different ecosystems because of their different economic and consumption activities. Stock refers to the accumulation of P in these subsystems, which does not flow among different activities in a short time (at least one year).

2.3 Data source

Data sources included P-related activity data and coefficients. P-related activity data were obtained from the government statistical yearbooks and bulletins. These data include population and agricultural inputs, including crop acreage, fertilizers, and pesticides, and the production of P-related products, such as fertilizers, crops, livestock, and aquatic products. Calculated parameters (e.g., P content of products and rates of recovered by-products) were obtained from the published literature and statistical yearbooks. More details about the system definitions, model details, data sources, and information on the calculated parameters are provided in the Supplementary Information (Texts S1-S3 and Table S1). We assessed the computational accuracy of the model by calculating the deviations in the total P inputs and outputs of the subsystems as well as the variability of the parameters. In addition, we conducted a Monte Carlo simulation to quantitatively test the uncertainties (refer to Text S6 for a detailed uncertainty analysis) (Han et al., 2021; Jiang and Yuan, 2015; Wu et al., 2014).

3 Results

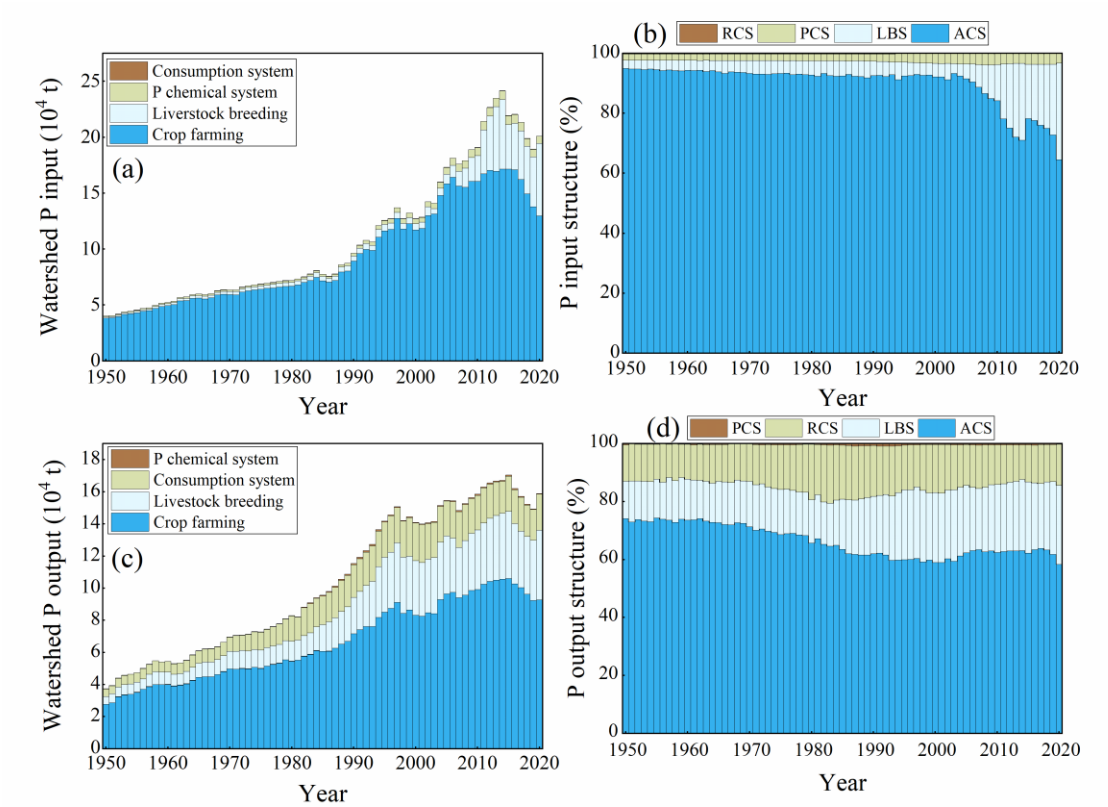
3.1 P flow pattern in Poyang Lake watershed

Figure 3 shows the general trend of the P flow in the social system of the Poyang Lake watershed over the past 70 years. The trend of P inputs generally rose and then fell in the Poyang Lake watershed from 1950 (4.04×10^4 t) to 2020 (2.01×10^5 t), reached a peak in 2014 (2.42×10^5 t), and then started to decline, with a reduction rate of approximately 1.04×10^4 t/a. Subsystem P inputs are shown in Figs. 3a and 3b, the contribution of P inputs from the agriculture crop system (ACS) was over 90% before 2006, then the contribution declined year by year to 64.58% (1.3×10^5 t) in 2020. The inputs of P from livestock breeding systems (LBS) have steadily increased since 2006, with inputs accounting for 32.18% (6.47×10^4 t) of the total system inputs in 2020. ACS and LBS accounted for over 90% of the whole system P inputs. In recent years, studies have shown that the P pollution of Poyang Lake mainly originates from continental input (90.8%), and agricultural pollution sources contribute 56.4% to the total pollution (Yang et al., 2020).

The interannual trend of Poyang Lake watershed P output was consistent with the input, with P emissions from the entire watershed peaking in 2015 (1.71×10^5 t) and then declining (Fig. 3c), with a whole watershed total output of 1.46×10^5 t in 2020 (including soil deposition). The highest P output from the ACS (58.37%) was reflected in the deposition of farmland soil, with 5.84×10^4 t in 2020. Except for the deposition of farmland soil, the structural share of other P output methods (solid straw waste, farmland drainage, and soil leaching) from the ACS steadily decreased, whereas the share of P output from livestock farming continued to increase (Fig. 3d). The P output from the LBS increased from 0.48×10^{4} t (1950) to 4.33×10^{4} t (2020). The P output from the LBS and ACS reached 80% of the entire watershed, whereas the P output from the residential consumption system (RCS) was relatively stable, although the trend was increasing, indicating that the P output drivers in the Poyang Lake watershed were mainly concentrated in agriculture and animal husbandry.

Figures 3e and S2 show the P flow in more detail. P chemical industrial system (CIS), which involves the industrial process of converting phosphate ore into phosphate fertilizer, organophosphorus (OP) pesticides, and synthetic detergents, is closely related to P flows from other subsystems and has the largest input among the nine subsystems. P inputs have increased significantly from 3.64×10^{4} t in 1950 to 1.92×10^{5} t in 2020. Because of low extraction, the Poyang Lake watershed has been dependent on the import of phosphate ore since 1980, accounting for more than 90% of the total P input at the peak. From 1950 to 2010, more than 90% of the P raw material was manufactured into fertilizer, and the fertilizer use increased from 3.55×10^{4} t to 1.55×10^{5} t, and only about 8% of the P input was used to produce pesticides and detergents (Jiang and Yuan, 2015). From 2010 onwards, the demands of urbanization have driven the continuous expansion of the livestock industry, and led to a change in feed P input patterns (from less than 10% to 34%) (Fig. 3a). This is closely linked to the rapid urbanization of the Poyang Lake watershed and the large changes in population proportions, with the watershed urbanization rate reaching 45% in 2011 (compared to 10.2% in 1950). To meet the characteristics of an urban diet, the production of meat products increased significantly, and the production of animal products exceeded 10,000 t of P from 2010, which was 130 t and 600 t, respectively, in the 1950s and 1980s; thus, the production of industrial feeds increased accordingly.

In general, the P cycle in the Poyang Lake watershed is no longer a naturally dominated cycle but significantly influenced by human activities during the flow dynamics between 1950 and 2015. The P flow structure of the Poyang Lake watershed mainly focuses on ACS, LBS, and RCS as the core, chemical systems as the main input sources, waste (wastewater) treatment systems, and soil and water environments as the subsystem P output sites. Therefore, an in-depth exploration of the P flow characteristics of the Poyang Lake watershed should focus on dissecting the more detailed P load and structural characteristics in the ACS, LBS, and RCS.



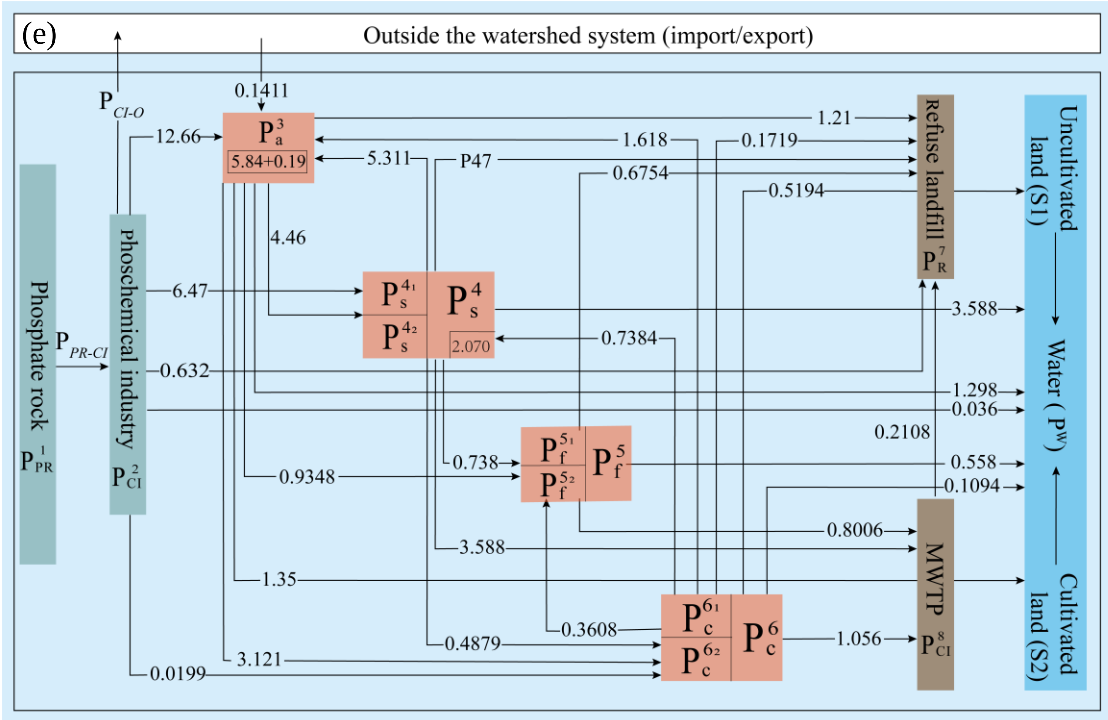


Fig. 3 P flow pattern in the Poyang Lake watershed. (a) Watershed input P, (b) Watershed P input structure; (c) Watershed output P; (d) Watershed P output structure; (e) Anthropogenic P flows for the entire watershed in 2020 (10^4 t). In this context, input P represents "new P" entering the system, encompassing industrial products such as fertilizers, feed, detergents, and other related compounds. RCS: Residential consumption system, LBS: Livestock breeding system, ACS: Agricultural cropping system, CIS: Chemical industrial system.

3.2 Subsystem P load in Poyang Lake watershed

3.2.1 Agricultural cropping system P load

The agricultural planting system P input in the Poyang Lake watershed was mainly crop seeds, chemical fertilizers and pesticides, human and livestock manure, and straw return to the field (Figs. 4a and 4b). The ACS total P input from crop farming increased from 5.84×10^4 t (1950) to 21.27×10^4 t (2020) in the Poyang Lake watershed. The P input to crop cultivation through chemical fertilizer increased nearly four times from 1950 (3.55×10^4 t) to 2020 (12.52×10^4 t) (Fig. 4a), accounting for 58.86% of farmland ecosystem P input. Furthermore, there is large-scale livestock and poultry farming in the watershed, and the livestock and poultry manure P input is second only to that of chemical fertilizers. In 2020, approximately 5.31×10^4 t P of animal manure was imported, accounting for 24.96% of the total farmland input. The input of P from returning organic fertilizer was nearly five times that in the early days of the People's Republic of China, reaching 8.28×10^4 t, accounting for 38.93% of the ACS total P input. In addition to fertilizers and manure, atmospheric deposition, seeds, and pesticides have the least impact on the water environment (4600 t in 2020 and 2900 t in 1950) because atmospheric deposition and seeds contain low P rates, and the current national production of OP pesticides and their P content are strictly limited.

The output of P from the ACS to the environment continued to increase, reaching 1.06×10^5 t in 2015 (2.77×10^4 t in 1950) with an average annual growth rate of 2.06%, the ACS P exported to soil (soil deposition, straw solid waste, and primary processing of agricultural products) and water (farm drainage and leaching) were 9.29×10^4 t and 1.31×10^4 t, respectively. After 2015, the ACS P output decreased to 9.28×10^4 t (2020) under the promotion of the national strategy of “Ecological Civilization Construction”. The P stored in cultivated soil was the largest part of the whole system output to the environment, accounting for 62.88% of the total output (2020), followed by straw waste (13.06%) and farmland drainage (12.77%). It is worth noting that the P flowing into soil/surface water from the ACS accounted for approximately 50% of the total system inputs, which means that crops use less than half of the ACS total P input (considering other hidden P flows that have not been calculated), and excess P input and loss pose a significant threat to the environment.

3.2.2 Livestock breeding system P load

Over the past 70 years, livestock farming P inputs have increased 6-fold, from 2.07×10^4 t (1950) to 12.78×10^4 t (2020) (Fig. 4c). Before 2011, the main sources of P in feed were residues from the initial processing of grain crops, straw, and feed grains (soybean cake, corn, and wheat), which account for 41–89% of new feed P inputs (Fig. 4d). In particular, the feed volume provided nearly half the LBS P input. Since 2011, with urbanization, changes in residents' dietary structure have increased the demand for animal food, and P production from live animals has increased 5-fold (1950–2020); thus, the production of industrial feeds has increased accordingly. Figure S4 shows a significant increase in P in the industrial feed from 2010 and a peak in 2014. Industrial feeds accounted for 50.64% of the "new" feed P in 2020 (6.47×10^4 t) compared with 5.12% in 1950 (0.1×10^4 t).

As shown in Fig. 4c, the LBS exported 5×10^4 t of P to the watershed environment in 2020, a tenfold increase from 1950, with an average annual growth rate of 3.23%. The largest output was animal manure, accounting for 69.61%–98% of the subsystem's output. In the last decade, owing to the policy of promoting organic fertilizer and strengthening waste (wastewater) management, the P output from animal manure has stabilized at approximately 3.0×10^4 t (70% of the subsystem's output). On the other hand, waste P from animal product processing continues to increase as animal food consumption increases, reaching 1.41×10^4 t in 2020 (160 t in 1950). This is because only approximately 20% of P in live animals is converted to animal-derived food (e.g., pork, beef, lamb, and poultry), most of which is stored as hydroxyapatite in the bones, blood, and hair of animals and discarded to landfills or non-cultivated land (Jiang and Yuan, 2015).

In the 21st century, the amount of animal P slaughtered and sold in the Poyang Lake watershed accounted for approximately 30% of the production and 8.4% of the total imported P for livestock and poultry breeding. The remaining P was maintained in the live animals (2.07×10^4 t in 2020). In 2020, the amount of P in animal products consumed within the watershed system was 0.49×10^4 t, accounting for approximately 45% of the slaughter volume, indicating that large livestock and poultry production has made Jiangxi province an important animal food P export province, with net exports accounting for 54%−79% of annual output in the past 40 years. In some years (1983, 1984, 1998-2001), the decline in animal product prices and the outbreak of animal diseases led to sudden declines in feed P consumption and animal food P production (Figs. 4c and 4d).

3.2.3 Residential consumption system P load

As shown in Fig. 4e, 4.22×10^4 t of P from chemical products, agricultural products, and animal products was consumed in 2020, which was nearly 4.5 times that in the early period of the founding of the country, 99% of which was food P. Agricultural products are the main source of RCS P input and have accounted for over 90% of the RCS total P input. Although the proportion of P from ACS to RCS has been decreasing since 2000, it still reached 86% in 2020, contributing 3.12×10^4 t of P. Similarly, from 1950 to 1999, the amount of P in food consumption increased from 1.06×10^4 t to 5.25×10^4 t and then decreased (3.63×10^4 t in 2020). It is worth noting that the proportion of animal food has increased (67 t in 1950 and 4900 t in 2020). It reached 13.52% in 2020, whereas only 0.63% in 1950. Additionally, per capita, food P consumption in the watershed rose from 0.67 kg to 0.80 kg between 1950 and 2020.

The export of P to the environment from the RCS occurs through solid waste and sewage, with an output of 2.22×10^4 t (0.37×10^4 t in 1950) in 2020, of which 0.17×10^4 t and 1.06×10^4 t of access to solid waste treatment plants and sewage treatment plants, respectively, account for 7.75% and 47.62% of the RCS total output. Correspondingly, the amounts of P discharged directly into the soil and surface water were 0.52×10^4 t and 0.11×10^4 t, respectively, accounting for 23.42% and 4.93% of the RCS total output, respectively. Thus, owing to the continuous improvement in the waste system, the direct discharge of waste P has been effectively controlled; however, the proportion is still high. The amount of P generated from the secondary processing of food in the RCS was 0.36×10^4 t (2020) (Fig. S3), mainly for soil/surface water and waste (wastewater) treatment systems, and this P loss was calculated separately owing to the limitations of urban and rural consumption data.

Urbanization has changed the pattern of P consumption in urban social systems. From 1950 to 2020, the population of the Poyang Lake watershed increased by 188.17% and the proportion of urban residents increased from 10.2% to 60.4% (Table. S1). Thus, urbanization, changes in agricultural production methods, increased consumption levels, and changes in dietary structure dominated the enhanced P flows and structural changes in the watershed (Liu et al., 2020b). Undoubtedly, this series of changes increased the P load to the natural environment of the watershed and exacerbated the problem of legacy P in the soil and water environment (Jiang and Yuan, 2015).

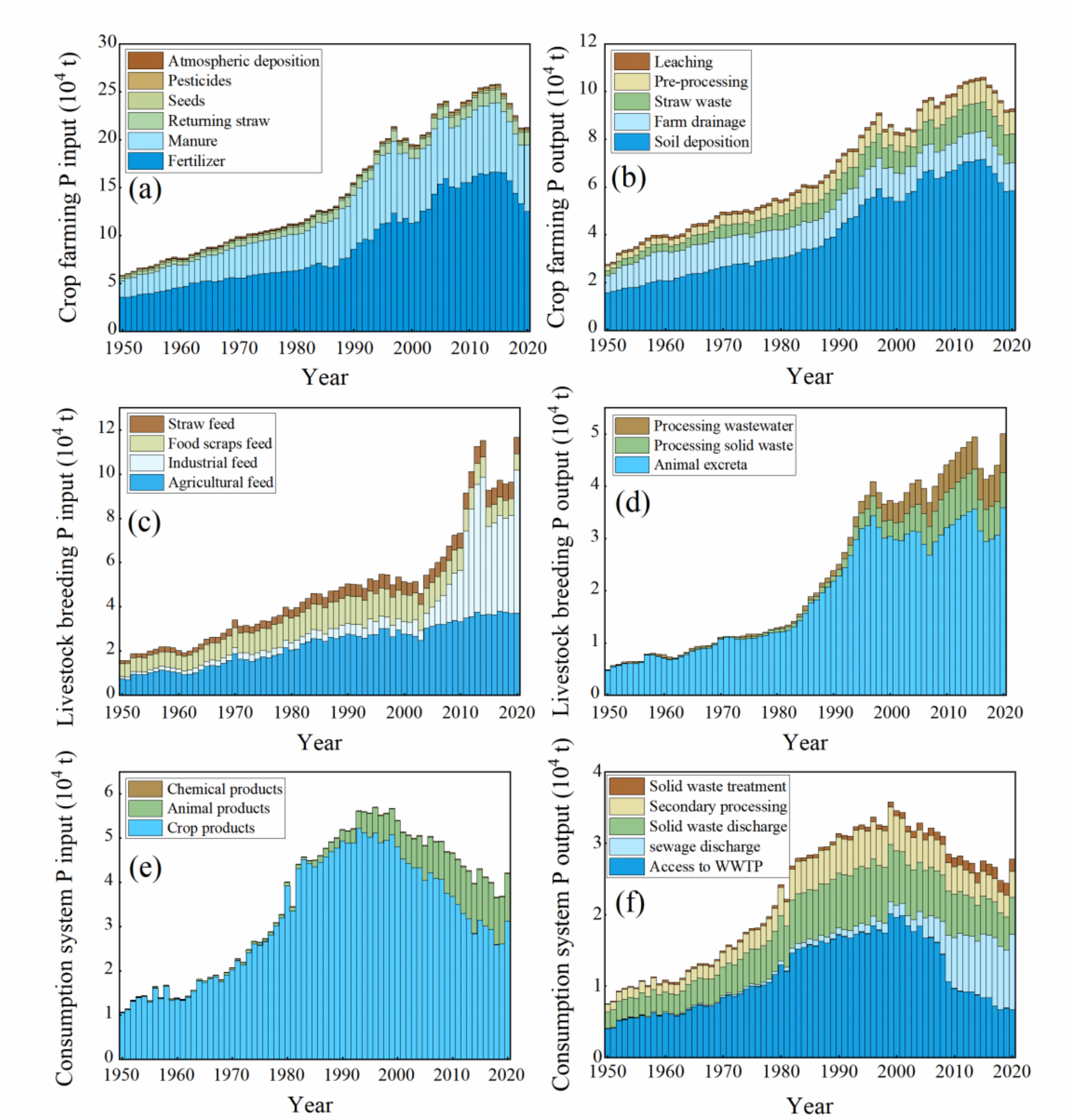


Fig. 4 Sub-watershed P input and output in the Poyang Lake watershed during 1950–2020. (a) P input into agricultural cropping system. (b) P output in the agricultural cropping system. (c) P input into the livestock breeding system. (d) P output in livestock breeding system. (e) P input into the consumption system. (f) P output in the consumption system. In a specific year, if production exceeds consumption, the excess implies that P exports. In contrast, a shortage indicates the P imports.

3.3 Watershed P cycle

The P load entering the social system in the Poyang Lake watershed increased 5.4 times, from 5.73×10^4 t (1950) to 3.9×10^5 t (2014). The P utilization of the whole watershed was 1.35×10^5 t in 2020, P utilization rate was 39.5%, which was 7.8% higher than that in 1950 (Fig. 5a). The P utilization rate of farming has been nearly 50% over the last five years. Although the P utilization rate of the farming industry has shown an increasing trend over the past 70 years, it remains still less than 50% in 2020. The P utilization rate for livestock breeding has continued to increase, from 0.63% (1950) to 9.69% (2019), but remains well below that of developed countries (12%–39%), indicating that the structure of P utilization in LBS in the study area needs to be optimized urgently to increase the amount of reused P and reduce P emissions.

Recycled P in the Poyang Lake watershed flows mainly to arable land and livestock farms. The P returned to land mainly comes from straw (1.35×10^5 t), animal manure (5.31×10^5 t), and human manure (1.62×10^5 t) in 2020, and P reused in livestock farming mainly comes from agricultural feed (4.46×10^5 t) and waste farming (7.4×10^4 t). In general, the P recycling rate in the Poyang Lake watershed showed a decreasing trend, reaching more than 40% before 2003 but 32%–40% in recent years. The returned organic fertilizer generated by the residential consumption system remained stable at approximately 2.0×10^4 t, accounting for 40% of the anthropogenic P export (Figs. 4e and 4f). P reused in the farming system accounted for approximately 20% of the anthropogenic P output (in the last five years), which was a decrease compared to the pre-2011 period. A possible reason for this is that the reuse of waste for farming occurs mainly in rural areas, while the rural population continues to decline due to rapid urbanization after 2010.

As shown in Fig. 5b, P emissions have increased more than fivefold, with emissions reaching 8.73×10^4 t, accounting for 25% of new P inputs across the watershed (2020) (Fig. 5a), which indicates that soil and surface water environments face great P pressure. The amount of P discharged from agricultural drainage, farming waste (wastewater), processing waste (wastewater), and anthropogenic emissions is getting higher, reaching 8.73×10^4 t in 2020, compared to only 1.83×10^4 t in 1950 (Fig. 5b). The contribution of P emissions from planting and livestock breeding is approximately 80%. Interestingly, the contribution rate of P emissions from cultivation declined and livestock increased each year, with a clear turnaround during 1981–1996, for complex reasons that may include (1) rapid development of industrial technology producing cheaper and more efficient fertilizers, (2) optimization of agricultural production technology and policies, (3) reduction of crop acreage due to reforestation, (4) urbanization leading to livestock flourishing, and (5) deficiency in P recovery or treatment technology for livestock farming.

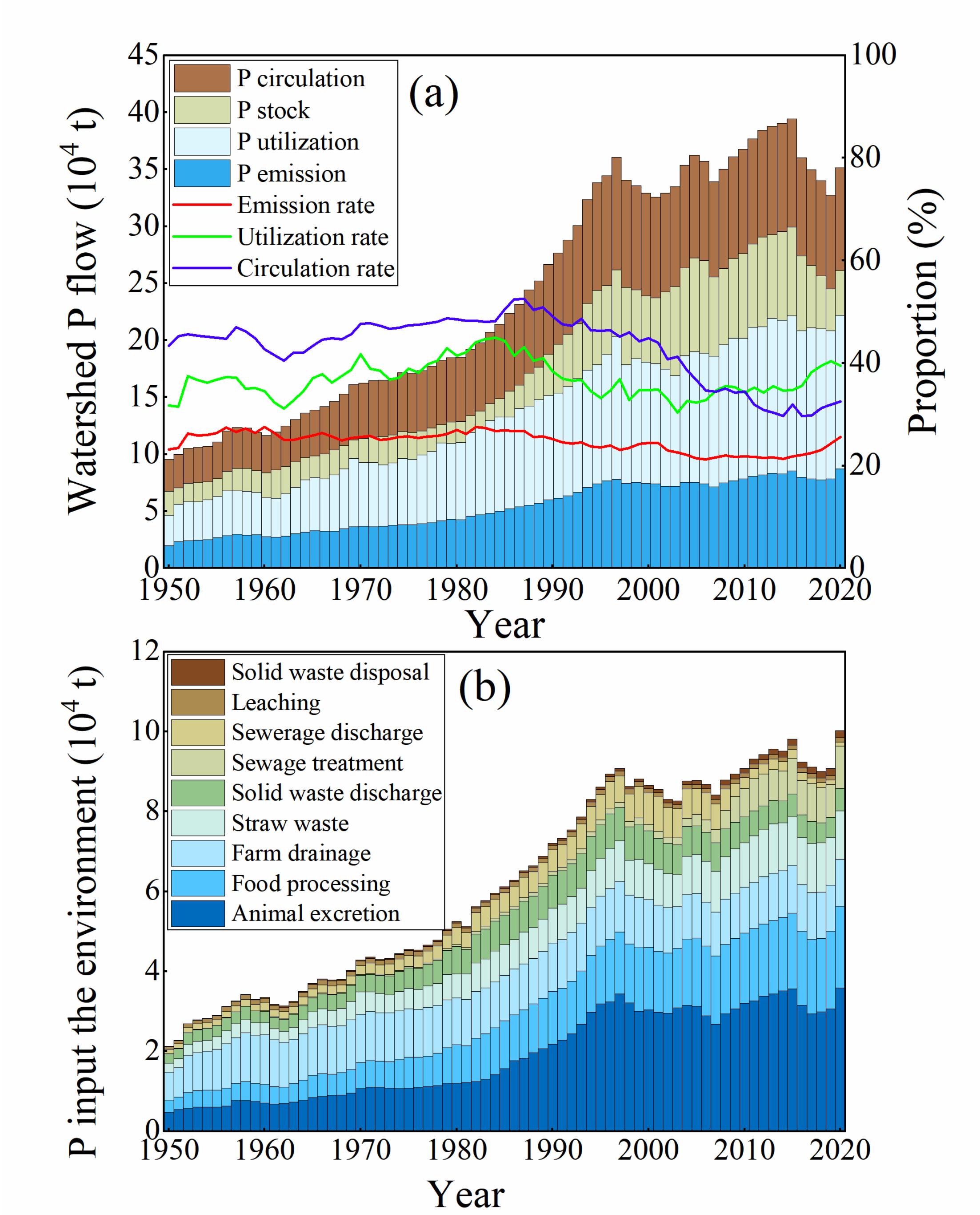


Fig. 5 P cycle characteristics (a) and sources of whole system P input (b) in the Poyang Lake watershed from 1950 to 2020.

3.4 Environmental P load

As shown in Fig. 6a, the soil and surface water environments in the study area increased with increasing P pressure. The total P flows to surface water, cultivated land, and non-cultivated land were 1.59×10^4 t, 1.85×10^4 t, and 0.45×10^4 t respectively in 1950, and 5.48×10^4 t, 7.18×10^4 t, and 1.73×10^4 t respectively in 2020. This change was mainly caused by increased fertilizer input, expansion of livestock farming, and inefficient use of P. In 2020, 2.51×10^4 t and 3.59×10^4 t of P were imported into the soil and water environment from farmland drainage and livestock wastewater, respectively, which contributed 78.27% of the total amount of P discharged. Due to the rapid expansion of large-scale farming and the lack of supporting waste (wastewater) treatment (recycling) technologies, LBS absolute P emissions increased nine-fold, from 0.48×10^4 t (1950) to 4.33×10^4 t (2020) (Fig. 5b). The same trend was observed for the amount of P entering landfills from residential consumption, which increased from 0.01×10^4 t (1950) to 0.17×10^4 t (2020).

The subsystem P emission to the environment causes a serious legacy P problem. The accumulation of legacy P accelerated, with annual growth rates of 2.37%, 2.97%, and 3.03% for cropland, wasteland, and surface water, respectively. The largest share of legacy P stocks was on cropland (varying between 36.76% and 60.83%), followed by surface water (varying between 26.86% and 46.64%) and uncultivated land (varying between 8.20% and 17.17%). In 2020, the amounts of P flowing to cropland, surface water, and uncultivated land were 7.70×10^4 t, 5.55×10^4 t, and 2.58×10^4 t, respectively. For uncultivated land, only 30% of legacy P is stored in landfills, whereas the rest is exposed to the environment, especially in rural areas (Jiang and Yuan, 2015).

In contrast, during the same period, the P emission in discarded domestic waste began to decline due to the gradual improvement of solid waste disposal systems in urban areas, but only a small proportion was reasonably disposed of in terms of percentage (11.58% in 2020) (Fig. 6b). The amount of P from the direct discharge of solid waste has gradually decreased since the 1990s, and the amount of P from direct waste discharge was 0.56×10^4 t in 2020, accounting for 6.4% of total P emissions. The total P loss from landfills increased from 0.34×10^4 t in 1950 to 2.90×10^4 t in 2020. Similarly, after 1997, owing to improvements in urban wastewater treatment facilities, the amount of P in the direct discharge of wastewater was controlled. In 2020, 0.11×10^4 t of P entered the environment, accounting for 1.3% of total P emissions. Compared to these sources, natural P losses (rainfall runoff, rainfall leaching, and atmospheric deposition) were relatively constant, contributing 0.81×10^4 t–15.5×10^4 t of P per year to the watershed.

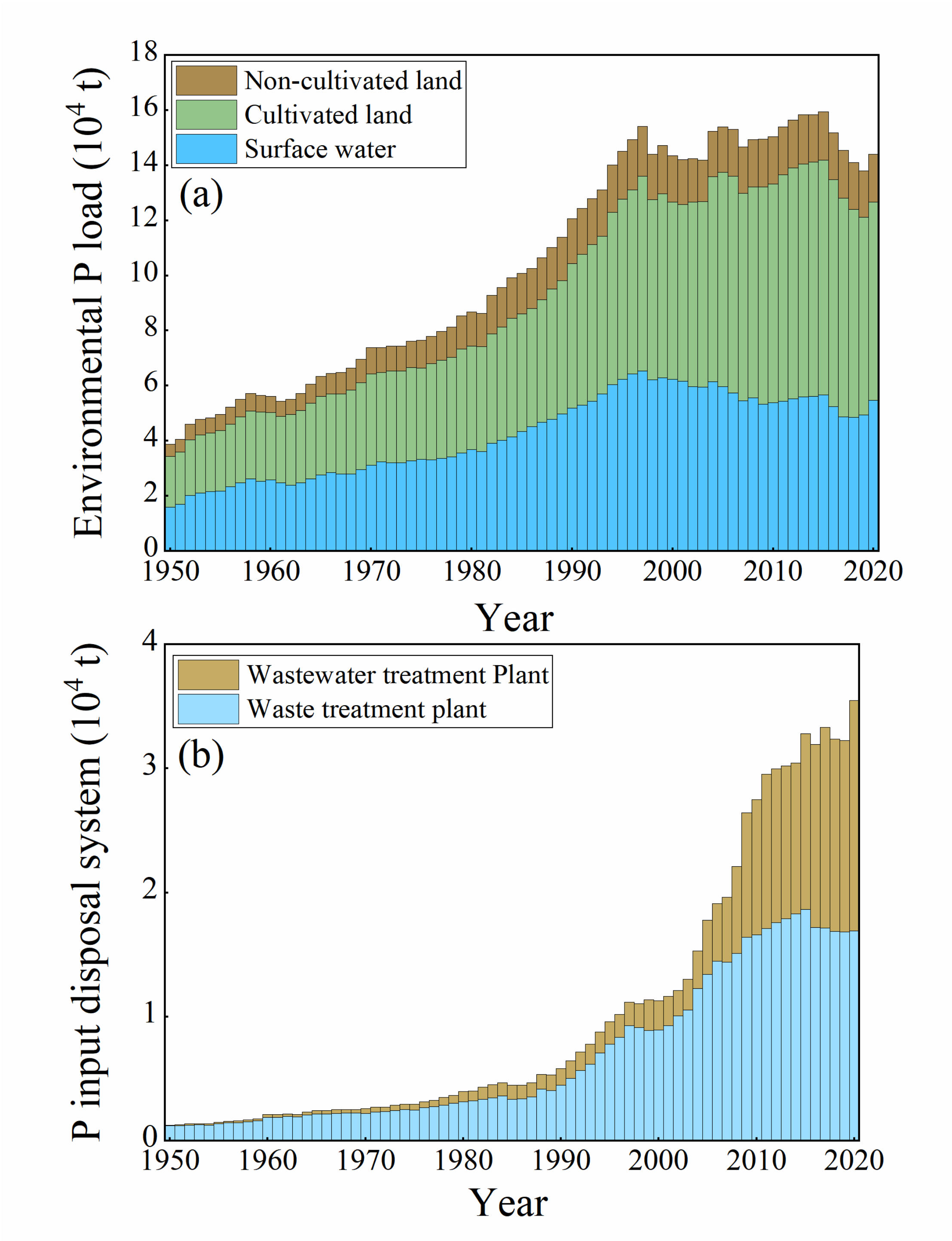


Fig. 6 P access to the environment (cultivated land, non-cultivated land, and surface water) (a) and waste (wastewater) disposal system (b) in the Poyang Lake watershed from 1950 to 2020.

4 Discussion

4.1 Driving forces of P-flow change.

Principal component analysis (PCA) was used to extract the main information and analyze the driving factors of the sample data (Liu et al., 2020a). Three factors with eigenvalues > 1 were extracted from the watershed P dataset using a correlation matrix (CM). The calculated factor loadings, cumulative percentages, and percentage variances explained by each factor are listed in Table S5. These factors (PCs) explain 95.06% of the total variance. The sample information of P flow in the Poyang Lake watershed was divided into five time zones by PCA; PC1 (64%) was the largest contributor to the total variance and therefore represented the dominant driving factors. PC1 was loaded with the sample information from 2001 to 2020. The loading values indicated the correlation between each indicator and the main components (Fig. 7b).

From 1950 to 2020, the enhancement in P flow was mainly due to changes in food production (CP 0.951 and AP 0.989), fertilizer use (0.983), population (0.888), and consumption levels (0.854). In addition, the urbanization rate (0.972), GDP per capita (0.868), and diet structure (0.908) also reflected strong loadings, and these indicators constituted the main part of PC1 (Fig. 7a, b, and Table S5). Population, urbanization rate, diet structure, P resources, and P emissions were all related to urbanization development in the Poyang Lake watershed (Fig. 7c).

The rapid growth of the population and living standards are key drivers of enhanced P flow (the population is still loaded in PC2 and PC3). From 1950 to 2020, the population of Jiangxi Province grew by 188.17%, whereas per capita GDP and consumption expenditure increased 800 and 400 times, respectively (Table S1). This has led to increased food demand and promoted a shift in the diets of animal proteins. The proportion of the urban population has increased from 9.5% to 60.44%, and the proportion of animal diets has increased from 0.63% to 13.52% in the last 70 years. As mentioned above, PCa is much higher than PCc; therefore, a higher intake of animal foods means that more P may be consumed to produce them, and the effect is more pronounced in urban areas. In contrast, P input from animal feces into the environment continues to increase, reaching 3.59×10^4 t in 2020, more than seven times the level in 1950.

This indicates that the Poyang Lake watershed has P flow characteristics that are similar to those in China (Liu et al., 2007; Ma et al., 2013). Population and economic levels led to an increase in grain yield, expansion of animal husbandry, and a substantial increase in consumption, which also resulted in maladjustment of fertilizer applications in the planting industry, unreasonable animal nutrition management, low manure recycling rate, and other problems, leading to a continuously low P utilization rate.

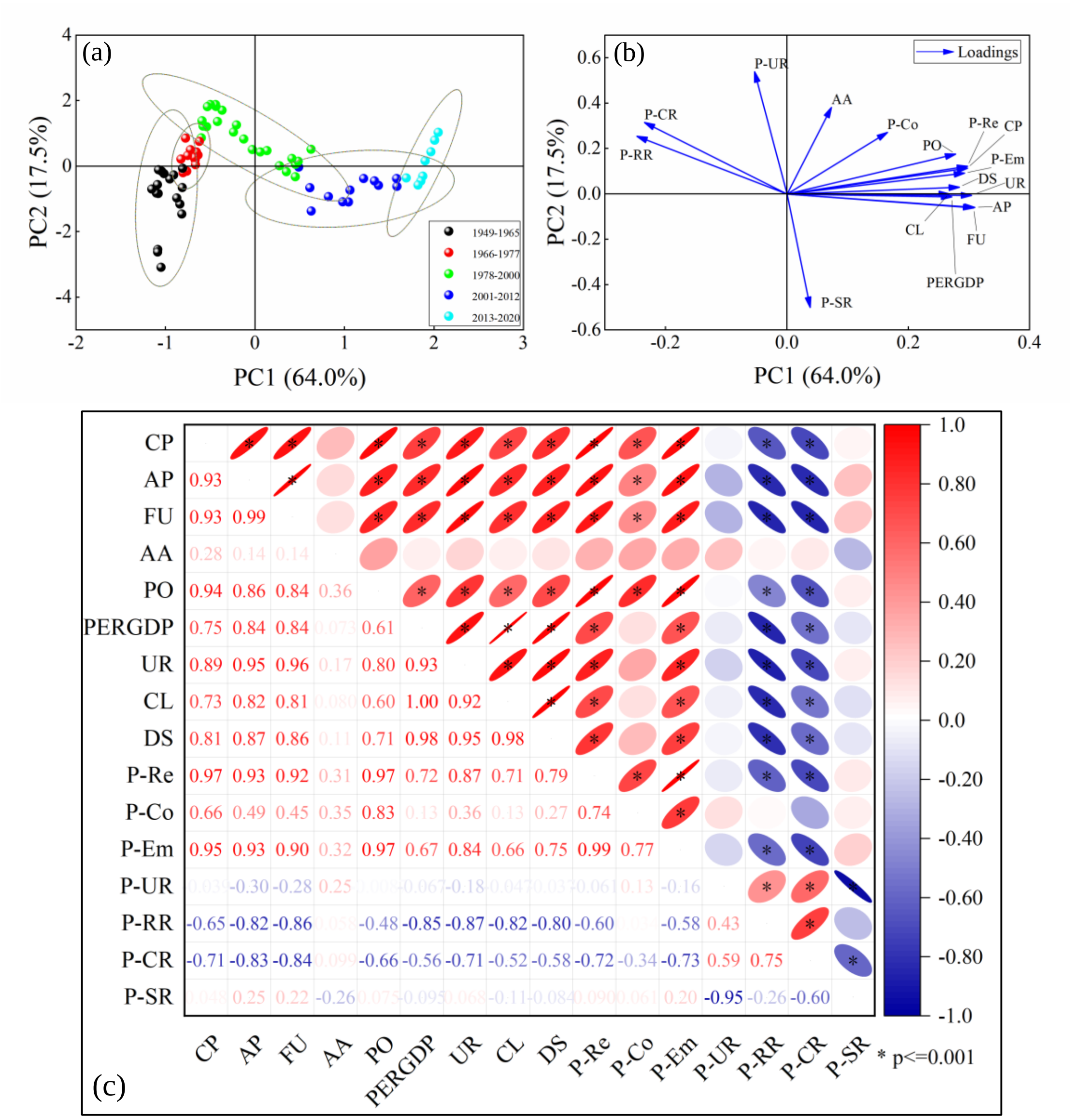


Fig. 7 PCA loading diagrams (a, b) and relatedness (c) of factors related to P flow.

CP: crop products; AP: animal products; FU: fertilizer usage; AA: arable area; PO: population; PERGDP: Per capita GDP; UR: urbanization rate; CL: consumption level; DS: dietary structure; P-Re: P resources; P-Co: P consumption; P-Em: P emissions; P-UR: P utilization rate; P-RR: P recovery rate; P-CR: P contamination rate; and P-SR: P stock rate.

4.2 Causes of the increased environmental P load

Our results indicate that agriculture is the largest contributor to surface water P input, followed by livestock, which is consistent with the results of studies conducted worldwide (Table 1). The P use efficiency (PUE) in crop production (PUEc) and animal husbandry (PUEa) are consistent with the results at the regional level but lie at the lower end of the national level and significantly below the international level. In China, PUEc declined from more than 90% before 1986 to 47% in 2014 and remained below 50% in Jiangxi Province (Fig. 6 and Table 1). The reasons for this lack of guidance on fertilizer application have led to widespread over-fertilization in China's agricultural intensification areas (Jiang and Yuan, 2015). It is estimated that Chinese producers overuse 51%, 27%, and 25% of phosphate fertilizers in maize, wheat, and rice cultivation, respectively (Jiang et al., 2019; Shi et al., 2016).

Compared with PUEc, PUEa in Jiangxi Province increased significantly from 0.63% in 1950 to 9.69% in 2019, which is consistent with the findings from China (1.4%–6.2%) and other watersheds (Table 1). This is due to the rapid expansion of large-scale livestock farming. The rearing period in large-scale farming is shorter than that in family-free farming, and rearing technology has improved (Bai et al., 2014; Yuan et al., 2019). In addition, the development of modern animal husbandry has benefited from projects aimed at improving people's livelihoods and urbanization. The reform of China's rural economic system and the increase in the proportion of the urban population have created a favorable market environment and gained policy support (Zhou, 2010). This has facilitated a shift in dietary choices towards animal protein, with the diet structure (P share of animal foods) increasing from 0.63% (1950) to 13.52% (2020).

There are deficiencies in the manner and efficiency of agricultural P use in the Poyang Lake watershed, and more unused P flows into the environment, which is also proven by the fact that RAW is generally higher than that of developed countries. The low PUE leads to a large build-up of soil P (4.09×10^4 t P yr^{-1}; 1.9% annual average growth; 27.44% of ACS P inputs) and high P losses to the environment from animal production (2.34×10^4 t P yr^{-1}; 42.85% of LBS P inputs). From a catchment perspective, we should consider the overapplication of fertilizers (low PUEa, Table S3), soil P accumulation, and animal manure P output. Improving P use in the Poyang Lake watershed must focus on watershed-scale agriculture and environmental management strategies, increase the PUE in agriculture, improve animal nutrition, and adopt technologies and policies to reduce P discharge from the animal sector and recycle P as manure in agriculture. Furthermore, a series of measures for the planting industry must be implemented, such as fertilization guidance, soil testing formula, and organic fertilizer return to the field.

Table 1 Comparison of Results in Different Countries and Areas

		plant production			animal production	food chain	food consumption			references
country/area	year	PIN	DOC	PUEc	PUEa	PC_{c+a}	PCA	AFP	RAW	
U.S.	2007	68	70%	82%	22%	5.1	0.97	47%	-	(MacDonald et al., 2012)
Netherlands	2005	44	38%	61%	39%	2.3	1.13	61%	49%	(Smit et al., 2010)
Austria	2004−2008	35	35%	77%	25%	2.5	1	-	69%	(Egle et al., 2014)
UK	2009	44	27%	81%	17%	2.9	0.6	-	86%	(Cooper and Carliell-Marquet, 2013)
France	2002−2006	25	43%	68%	21%	5	1.24	80%	84%	(Senthilkumar et al., 2012)
Malaysia	2007	30	95%	34%	25%		1.31	-	-	(Ghani and Mahmood, 2011)
Turkey	2001	10	86%	80%	20%	2	0.7	16%	-	(Seyhan, 2009)
Busia District, Uganda	2010	39	2%	39%	12%	-	0.59	22%	-	(Lederer et al., 2015)
Harare District, Zimbabwe	2001	22	23%	15%	-	-	0.85	6%	-	(Gumbo et al., 2002)
Phoenix	2005−2010	62	37%	83%	16%	-	1.02	-	44%	(Metson et al., 2012b)
Thachin watershed, Thailand	2006	75	89%	32%	25%	-	0.44	-	88%	(Schaffner et al., 2009)
Dianchi watershed, China	2000	165	64%	29%	7%	-	-	-	41%	(Liu et al., 2007)
Beijing area China	1978	81	79%	30%	4%	5.1	0.7	4%	-	(Ma et al., 2014)
	2008	154	76%	24%	10%	4	0.7	27%	-	
Chaohu watershed China	1978	36	42%	54%	12%	2.4	1.1	1%	73%	(Jiang and Yuan, 2015)
	2012	142	62%	32%	22%	5.2	0.61	12%	83%	
China	1984	82	57%	64%	21%	2.8	0.9	5%	62%	(Ma et al., 2013)
	2008	102	72%	60%	33%	4.2	1.1	10%	85%	
Poyang Lake Watershed	1950	17.51	60.71%	31.33%	32.81%	2.28	0.67	0.63%	23.4	
	1978	28.67	57.33%	45.30%	28.68%	2.25	0.96	1.23%	37.3	
	2020	56.38	59.51%	48.47%	24.60%	4.64	0.8	13.52%	46.7	

Note: The ACS P input intensity (PIN, kg ha^{-1}) was calculated by dividing the total P input into crop farming by the cultivated land area. DOC is the percentage of chemical fertilizer in the total P input for crop farming. Per capita (PCA, kg P cap^{-1} yr^{-1}) was the annual P consumption per capita in the diet. PC_{c+a} (kg kg^{-1}) is the life-cycle P consumption required to deliver 1 kg of P into the food chain. AFP is the percentage of P in the diet derived from the animal products. RAW is the percentage of P from agriculture in the total P input into the surface water.

4.3 Phosphorus flow continues to increase

PUE is related to the structure and intensity of P flow in the entire watershed, and the driving factor analysis showed that PUE is affected by diet and consumption level. As population and regional economies continue to improve, human activities continue to promote P flow (Jiang et al., 2019; Li et al., 2020b).

We explored the relationship between these indicators and economic development (GDP per capita), and compared the results for Jiangxi Province with those for China and other countries (Fig. 8). The results showed that P utilization and diet structure were significantly associated with the level of economic development. The proportion of PUEa in animal food has increased significantly with economic development. When PUEa was at the same level, the GDP per capita in Jiangxi Province and China was much lower than that in developed countries, implying that China has a higher degree of intensive farming and industrial feed production technology has enhanced PUEa, which is clearly different from farm-based farming in Western countries. Correspondingly, most countries have a diet structure dominated by animal products (more than 40% meat), with a much higher percentage than that in Jiangxi Province (0.63%–13.52%). Due to rapid population growth, the per capita possession of animal products in Jiangxi Province (China) is much lower than that in developed countries but higher than that in developing countries such as India, an indicator that remains consistent with the economic level (GDP per capita).

These results are similar to the country-level P footprint study results of Jiang et al. (2019), with China (6.4 kg per capita) ranked between India (3.5 kg per capita) and the United States (11.1 kg). The PUEc decreased to $4,000 per capita in Jiangxi Province ($18,000 per capita in China) and then increased. The curves for both indicators are consistent with the environmental Kuznets curve hypothesis, which assumes an inverted U-shaped relationship between the pollution indicators and per capita income (Dinda, 2004). This means that an increase in agricultural P inputs is likely to be accompanied by a continued decline in utilization.

The above results suggest that the P utilization patterns of agricultural cultivation and large-scale farming and the food choices of the population in developing countries directly influence P cycling patterns based on the objective basis of rapid economic uplift. Comparative results from developed countries suggest that the demand for animal food (high-quality protein) will continue to rise in Jiangxi Province and China, and that livestock farming and industrial feed production will continue to expand (Figs. S3 and S4). Breaking through the P utilization rate of crop farming and livestock farming and improving the P recycling rate have become key issues in optimizing the P flow structure and reducing P emissions in the watershed.

Because of its low utilization, most of the P remains in the soil and water environments. Thus, the legacy P problem is serious and predictable. Cropland soils are currently the sink for a vast quantity of P input to the food chain in the Poyang Lake watershed (2.90×10^6 t during 1950-2020). The soil P stock is also a major source of non-point water pollution, primarily through soil erosion, a major problem in China, but also via runoff (8.19×10^5 t) and leaching (7.8×10^4 t) of dissolved P from over-fertilized soils. Efficient management of soil P stock will be a key aspect of P use efficiency for many years and should be a high priority for future research.

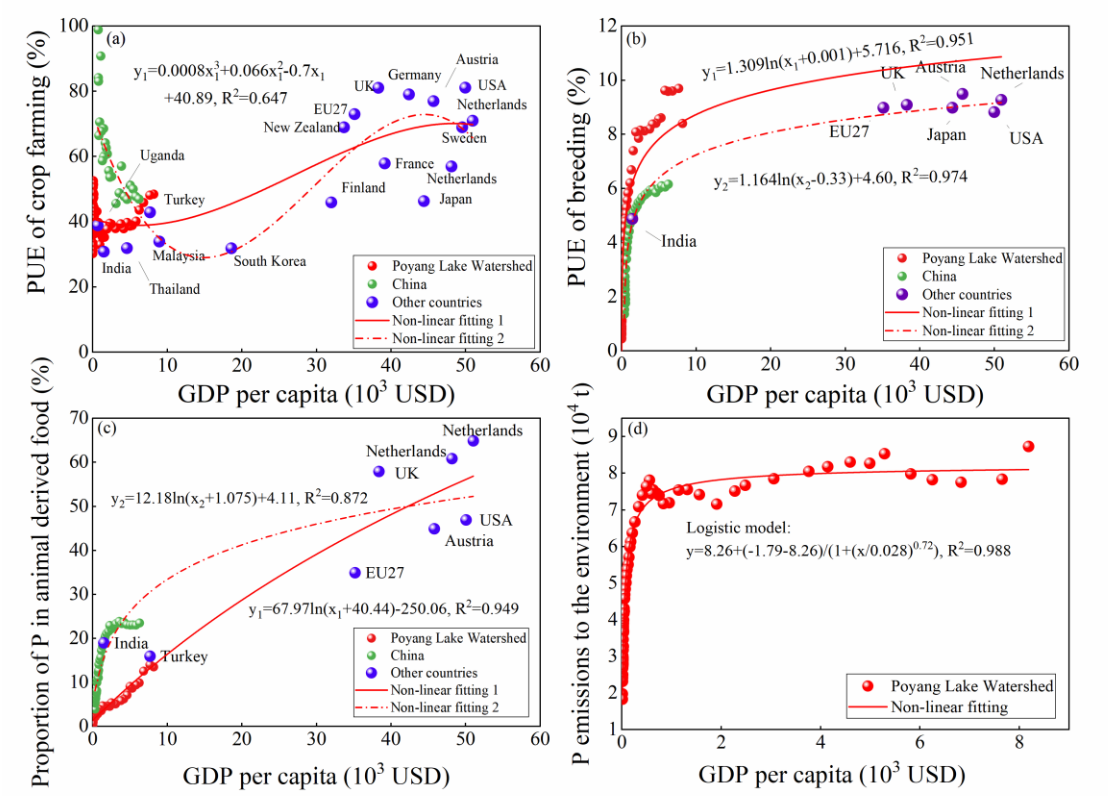


Fig. 8 Relationship between GDP per capita and the P-flow-related indicators, based on Jiang et al. (2015). We tested the relationship between these indicators and economic development (GDP per capita) and compared Jiangxi Province results with China and other countries, including Austria, EU27, Finland, France, Germany, India, Japan, Malaysia, Netherlands, New Zealand, South Korea, Sweden, Switzerland, Thailand, Turkey, Uganda, the United Kingdom and the United States (Antikainen et al., 2005; Cooper and Carliell-Marquet, 2013; Cordell et al., 2013; Jedelhauser and Binder, 2015; Jeong et al., 2009; Keil et al., 2018; Lederer et al., 2015; Li et al., 2015; Linderholm et al., 2012; Liu et al., 2016; MacDonald et al., 2012; Matsubae-Yokoyama et al., 2009; Matsubae et al., 2011; Prathumchai et al., 2016; Seyhan, 2009; Smit et al., 2015; Suh and Yee, 2011).

4.4 Solutions

As the population and regional economies continue to improve, human activities continue to promote the Poyang Lake watershed P flow. The continuous increase in the P input of ACS and LBS, low utilization efficiency, and declining cycling rate are the main barriers to P resource management and P pollution control in the Poyang Lake watershed. To minimize the release of P emissions, we recommend implementing strategies such as controlling inputs at the source, strengthening recycling process, and reducing end-point outputs. Furthermore, throughout the process of attenuating the P flow in the watershed and promoting P cycling, it is important to follow the principles of government-led active farmer participation and strict corporate cooperation.

Our assessment revealed that the primary sources of P in the Poyang Lake watershed are chemical P fertilizer, animal feed, and human food. Thus, future P management decisions should focus on reducing P input using these materials. Reducing fertilizer application is an effective way to reduce P loss during crop cultivation. Farmers' fertilizer application strategies are often based on empirical judgment, which leads to the misuse of fertilizers. Farmers may be more concerned about crop yields and investment costs in China, and government departments should improve farmer cooperation in P management. Therefore, we suggest that agricultural authorities should be more involved and guide farmers in implementing measures related to P management. For example, soil testing, precision fertilization, use of low-P fertilizers, legume crop rotation/intercropping, straw return, organic/green manure promotion, and reduction in mineral P input are worth considering.

ACS and LBS are key subsystems in P cycling within the watershed (new P inputs to the entire watershed). The most significant P cycle and utilization at the watershed level is the application of livestock manure to croplands, as the largest proportion of P emissions to the natural environment comes from livestock production systems. However, the proportion of livestock manure currently applied to croplands is < 30%. Similarly, the proportion of P recycled as feed P entering LBS was < 10%. Therefore, establishing a P-cycling link between CAS and LBS systems is crucial for reducing the overall P inputs and outputs within the watershed. Research has shown that developing a "closed-loop" system using livestock manure as a crop fertilizer and feeding crops or residues to animals can reduce the demand for new P by approximately 50% (Chowdhury et al., 2016; Metson et al., 2012a). Additionally, the straw produced in the ACS system can be 100% reused (for cropland application and animal feeding, currently with a comprehensive utilization rate of approximately 64%). Utilizing RCS human excreta in crop production can also increase P cycling because the biodegradable portion of kitchen waste, sludge, or urban solid waste can be composted and returned to farmlands.

The pressure to reduce the end-point outputs comes from each subsystem because there is a P flux connecting the water and soil within each subsystem. Strategies to reduce P outputs are evident. Apart from enhancing P cycling as mentioned above, it is essential to ensure centralized treatment of waste (e.g., wastewater). For example, controlling on-farm drainage through intermittent irrigation and providing guidance for the appropriate drainage timing are important for reducing P loss. To target legacy P in the soil, it is important to mobilize soil P so that plants can access it easily. This can help expedite the process and potentially minimize the risk of P loss to nearby water bodies (Jiang and Yuan, 2015). The P outputs from the LBS and RCS systems should be collected and disposed of as much as possible, either by meeting regulatory standards before discharge or by reusing it for irrigation purposes. Many small livestock farms in the study area have contributed to significant P loss from livestock farming. Integrating small-scale farms into intensive farming operations is beneficial for the unified collection and treatment of livestock waste, thereby reducing water and soil environment losses. The results of this study demonstrated that the urban waste management system in the Poyang Lake watershed is continuously improving; however, the solid waste collection rate remains low, while the treatment rate for rural domestic sewage is only 30%. Therefore, there is enormous potential to weaken the P flow between subsystems and the water-soil environment through the enhancement of waste (wastewater) treatment systems in the future.

As previously mentioned, the P flow in the Poyang Lake watershed began to weaken after the implementation of the ecological civilization in China (Fig. 3). This highlights the need for government agencies to play a leading role in weakening the P flow in the Poyang Lake watershed. The biggest challenge in achieving the goal attenuating P flow in the watershed is to activate those involved in P use and P recycling (e.g., residents, farmers, and waste treatment plant operators). This must be managed by the government. Chinese farmers consider government departments the most trusted source of agricultural information and rely on agribusiness. For example, farmers in

China may be more concerned about crop yields and investment costs. Therefore, government departments should focus on improving farmers' cooperation in P management. Therefore, agricultural authorities should be more involved in and guide farmers in implementing measures related to P management. Similarly, industrial and agricultural enterprises within the watershed must adhere strictly to P control policies under the constraints and guidance of regulatory authorities.

4.5 Uncertainty

Uncertainty exists in this study with respect to the flow quantification models and datasets. To reduce the uncertainty in the quantification models, alternative calculation methods for each P-flow were cross-checked to determine the most appropriate method. Efforts were also made to ensure the accuracy of the parameters by setting the series selection criteria (Text S4). In this study, uncertainties were used to provide information on the reliability of the data. The results indicated that uncertainties in the quantification of cropping, breeding, and consumption systems have decreased in recent years (Table S2). This indicates that the P flow structure of the Poyang Lake watershed stabilized to a certain extent. We also made great efforts to ensure the accuracy of parameter values in the context of various data sources by selection criteria of parameters (Text S1).

The uncertainty analysis showed that only a few parameters had coefficients of variation greater than 15% (Fig. S5). More specifically, in the last decade, the highest uncertainty came from the P stock in cultivation (35.36%), the amount of P provided by industrial feed (19.73%), and chemical products (31.7%), which may be related to the complex structure of P flows in agricultural systems. Monte Carlo simulations revealed that the comprehensive uncertainty of P input-output in the subsystem of the Poyang Lake watershed is estimated to be 10.44-14.83%. In 2020, the frequency distribution probabilities of CIS, ACS, LBS, and RCS are 0.06-13.02%, 0.08-14.62%, 0.01-14.83%, and 0.08-10.49% respectively (Fig. S6). Among all the significant P flows, the calculation of P from livestock manure application, animal products sales, and kitchen wastewater showed higher uncertainties (Fig. S7). This suggests a close interconnection of P flows within the farming system with complex parameters and processes involved in the calculations. Future research should focus on improving the understanding of the P flow processes and calculation methods within this subsystem. Overall, our model is relatively robust to this variation and the parameters and data were selected to suit the actual situation in the study area. In particular, when the model is used in other watersheds, efforts conducted to reduce variance in parameters mentioned above can largely reduce uncertainty in results.

5. Conclusion

In this study, a large-scale watershed P flow analysis model with a more comprehensive system and finer P flow processes was constructed, which can be utilized to analyze the nature and magnitude of multi-year P flow at the watershed scale. Both structurally and operationally, the WPSFA model takes into account primary relevant P flows and storage relating to key systems, subsystems, processes or components, and associated interactions of P flow to represent a typical P flow system at the watershed scale. The key advantage of this model over available P-SFA models is that it focuses on the characteristics of the watershed's socio-economic system. Furthermore, the model is capable of analyzing P flow over as many years as required at a time, and therefore, can indicate the trends or changes in P flow over many years. The construction framework of the WPSFA model is applicable to other large-scale watersheds.

The application of the WPSFA model in the case of the Poyang Lake watershed has indicated that the model produces useful information on the nature and magnitude of multi-year P flow at the watershed scale that can be utilized to formulate informed and effective policy towards achieving P sustainability. The study results indicated that the P utilization patterns of agricultural cultivation and large-scale farming and the food choices of the population in the watershed directly influenced the P cycling patterns based on the objective basis of rapid economic uplift. ACS and LBS accounted for > 90% of whole system P inputs. ACS is the primary contributor to the input of P in water and soil, followed by LBS. PUE is related to the structure and intensity of P flow in the entire watershed. The low PUE in these two subsystems is a key factor affecting P cycling and promoting increased P flow in watersheds.

The verification of the WPSFA model results relating to the case of the Poyang Lake watershed based on a reliability check performed in this study indicates that the model produces reliable results. Future research can concentrate on optimizing the WPSFA model further to improve its accuracy and applicability, thereby establishing it as a powerful tool for analyzing P flow at the watershed scale. Furthermore, attention should be given to conducting studies on the spatial distribution characteristics based on long-term P flow analysis at the watershed scale, as well as scenario-based predictions for P resource management. These efforts will serve as crucial research foundations for strengthening watershed P cycling and mitigating P flows.

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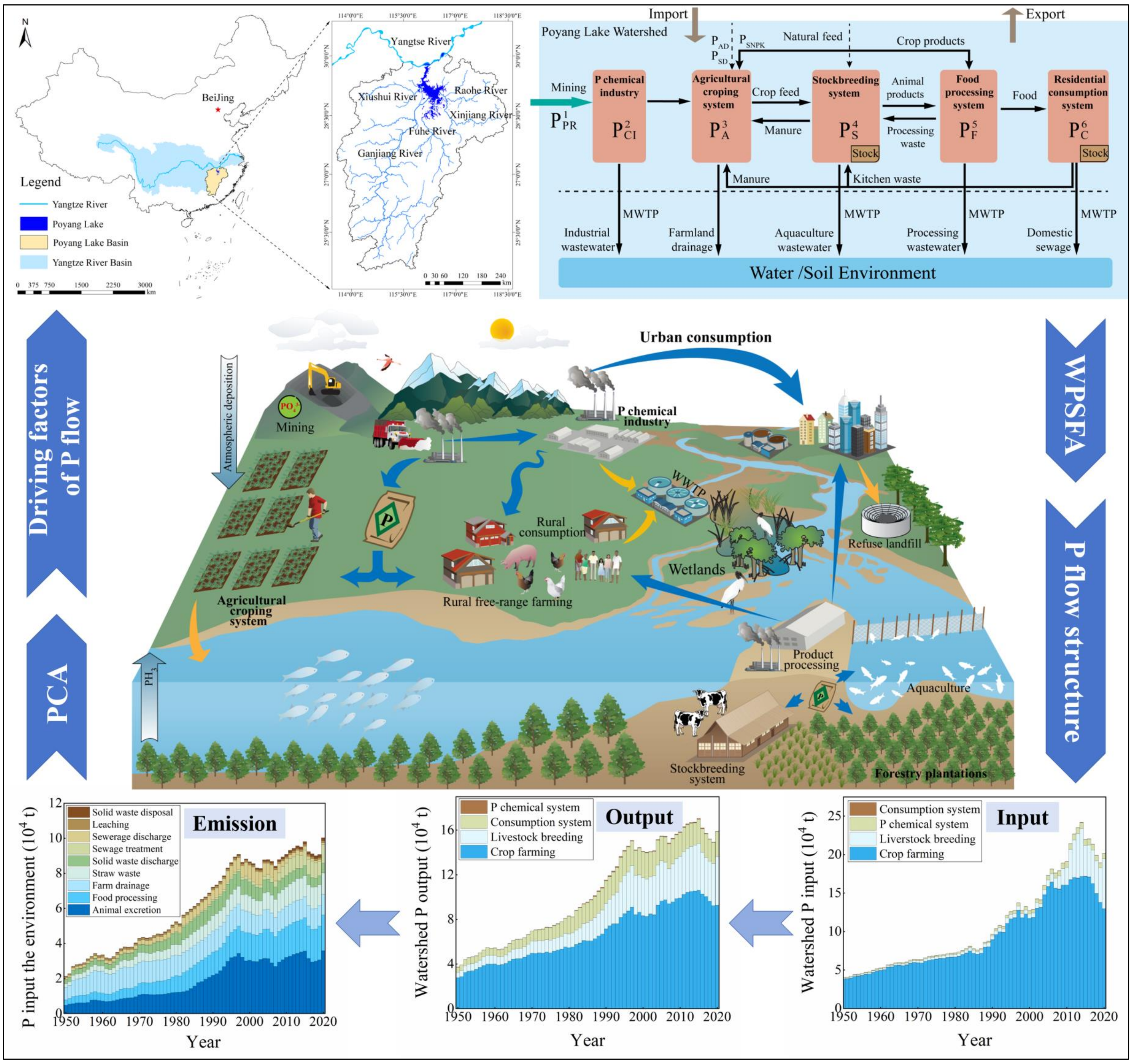

Graphical Abstract

Table 1 Comparison of Results in Different Countries and Areas

		plant production			animal production	food chain	food consumption			references
country/area	year	PIN	DOC	PUEc	PUEa	PC_{c+a}	PCA	AFP	RAW	
U.S.	2007	68	70%	82%	22%	5.1	0.97	47%	-	(MacDonald et al., 2012)
Netherlands	2005	44	38%	61%	39%	2.3	1.13	61%	49%	(Smit et al., 2010)
Austria	2004−2008	35	35%	77%	25%	2.5	1	-	69%	(Egle et al., 2014)
UK	2009	44	27%	81%	17%	2.9	0.6	-	86%	(Cooper and Carliell-Marquet, 2013)
France	2002−2006	25	43%	68%	21%	5	1.24	80%	84%	(Senthilkumar et al., 2012)
Malaysia	2007	30	95%	34%	25%		1.31	-	-	(Ghani and Mahmood, 2011)
Turkey	2001	10	86%	80%	20%	2	0.7	16%	-	(Seyhan, 2009)
Busia District, Uganda	2010	39	2%	39%	12%	-	0.59	22%	-	(Lederer et al., 2015)
Harare District, Zimbabwe	2001	22	23%	15%	-	-	0.85	6%	-	(Gumbo et al., 2002)
Phoenix	2005−2010	62	37%	83%	16%	-	1.02	-	44%	(Metson et al., 2012b)
Thachin watershed, Thailand	2006	75	89%	32%	25%	-	0.44	-	88%	(Schaffner et al., 2009)
Dianchi watershed, China	2000	165	64%	29%	7%	-	-	-	41%	(Liu et al., 2007)
Beijing area China	1978	81	79%	30%	4%	5.1	0.7	4%	-	(Ma et al., 2014)
	2008	154	76%	24%	10%	4	0.7	27%	-	
Chaohu watershed China	1978	36	42%	54%	12%	2.4	1.1	1%	73%	(Jiang and Yuan, 2015)
	2012	142	62%	32%	22%	5.2	0.61	12%	83%	
China	1984	82	57%	64%	21%	2.8	0.9	5%	62%	(Ma et al., 2013)
	2008	102	72%	60%	33%	4.2	1.1	10%	85%	
Poyang Lake Watershed	1950	17.51	60.71%	31.33%	32.81%	2.28	0.67	0.63%	23.4	
	1978	28.67	57.33%	45.30%	28.68%	2.25	0.96	1.23%	37.3	
	2020	56.38	59.51%	48.47%	24.60%	4.64	0.8	13.52%	46.7	

Note: The ACS P input intensity (PIN, kg ha^{-1}) was calculated by dividing the total P input into crop farming by the cultivated land area. DOC is the percentage of chemical fertilizer in the total P input for crop farming. Per capita (PCA, kg P cap^{-1} yr^{-1}) was the annual P consumption per capita in the diet. PC_{c+a} (kg kg^{-1}) is the life-cycle P consumption required to deliver 1 kg of P into the food chain. AFP is the percentage of P in the diet derived from the animal products. RAW is the percentage of P from agriculture in the total P input into the surface water.

Table S1 List of coefficients used in the WPSFA model.

$P_{arameter}$		Unit	Mean value	Distribution	Source	Quality
P chemical industry						
*f*1		%	42.36	(41.48, 42.36, 45.84) [a]	(CCIEP, 2003; Yan, 2008)	High
*f*2		%	12.01	(11.12, 12.01, 12.30) [a]	(CCIEP, 2003; Yan, 2008)	High
a		%	5.2	(4.7, 5.2, 5.4) [a]	(Xu, 2005; Yan, 2008)	Medium
P^{fodder}_{mean}		%	0.39	(0.38, 0.39, 0.40) [a]	(Liu, 2004; Xu, 2005)	Medium
P^{u}_{mean}		kg·per capita·yr^{-1}	3.264	(3, 3.264, 3.5) [a]	(Liu, 2004; Yan, 2008)	High
P^{r}_{mean}		kg·per capita·yr^{-1}	4.74	(4.6, 4.74, 4.85) [a]	(Liu, 2004; Yan, 2008)	Medium
K_1		%	0.11	(0.09, 0.11, 0.12) [a]	(CNLIC, 2006-2012; Gong and Mei, 2013; Liu, 2004)	High
k_2		%	80	(75, 80, 90) [a]	(Zhou et al., 2002; Smil, 2000; Xi, 1990)	High
K_3		%	100	100 [c]	(Zhou et al., 2002; Smil, 2000; Xi, 1990)	High
$P_{ardealite}$		%	1.22	(1.20, 1.22, 1.23) [a]		Medium
Agricultural croping system						
b		kg/(km^2·a)	25	(23, 25, 26) [a]	(Xu, 2005; Yan, 2008)	Medium
K_m	rice	kg/hm^2	0.53	(0.53, 0.14) [b]	(He et al., 1999; Jiang and Yuan, 2015; Wu et al., 2014; Xu, 2005; Yan, 2008)	High
	wheat		1.24	(1.24, 0.12) [b]		
	cotton		0.26	(0.26, 0.05) [b]		
	peanut		0.4	(0.4, 0.02) [b]		
	rapeseed		0.01	(0.01, 0.001) [b]		
	sesame		0.03	(0.03, 0.001) [b]		
	hemp		1.22	(1.22, 0.03) [b]		
	vegetables		0.01	0.01 [c]		
e_i	rice	%	1	(0.83, 0.90, 1.17) [a]	(He et al., 1999; Jiang and Yuan, 2015; Wu et al., 2014; Xu, 2005; Yan, 2008; Yuan et al., 2011b)	High
	wheat	%	1.1	(0.91, 1.1, 1.29) [a]		
	rapeseed	%	2.5	(1.5, 2.7, 3.0) [a]		
	peanut	%	1.5	(0.80, 1.5, 2.0) [a]		
	sesame	%	2.8	(2.4, 2.8, 3.3) [a]		
	cotton	%	3	(2.5, 3, 3.7) [a]		
	corn	%	1.2	(0.99, 1.2, 1.41) [a]		
	beans	%	1.6	(1.2, 1.6, 1.8) [a]		
	potatoes	%	0.5	(0.41, 0.50, 0.59) [a]		
r_i	rice	%	41.7	(36.6,41.7,46.8) [a]	(Cui et al., 2008; Gao et al., 2009; Jiang and Yuan, 2015; NATESC, 1999; Wu et al., 2014; Wu et al., 2010; Xu, 2005)	High
	wheat	%	40.2	(39.4,40.2,41.2) [a]		
	rapeseed	%	34.1	(33.1, 34.1, 35.6) [a]		
	peanut	%	26	(25, 26, 27) [a]		
	sesame	%	30	(28, 30, 31) [a]		
	cotton	%	16	(15, 16, 18) [a]		
	corn	%	32.2	(31.8, 32.2, 33) [a]		
	beans	%	16.8	(16.1, 16.8, 17.5) [a]		
	potatoes	%	100	100 [c]		
t_i	rice	%	0.13	(0.12, 0.13, 1.37) [a]	(Cui et al., 2008; Gao et al., 2009; Jiang and Yuan, 2015; NATESC, 1999; Wu et al., 2014; Wu et al., 2010; Xu, 2005)	High
	wheat	%	0.08	(0.072, 0.08, 0.087) [a]		
	rapeseed	%	0.14	(0.13, 0.14, 0.16) [a]		
	peanut	%	0.16	(0.08, 0.16, 2.0) [a]		

	sesame	%	0.15	(0.12 0.15, 0.18)		
	cotton	%	0.15	(0.13, 0.15, 0.17) [a]		
	corn	%	0.152	(0.138, 0.152, 0.166) [a]		
	beans	%	0.2	(0.171, 0.20, 0.281) [a]		
	potatoes	%	0.28	(0.22, 0.28, 0.32) [a]		
c		%	30	(23, 30, 40) [a]	(Busman et al., 2008; Daniels et al., 2001)	Medium
g_i	rice	%	0.36	(0.24, 0.36, 0.46) [a]	(CAAS, 2015; He et al., 1999; Jiang and Yuan, 2015; Lu et al., 1996; Smil, 2000; Wu et al., 2014; Xu, 2005; Yan, 2008)	High
	wheat	%	0.41	(0.33, 0.41, 0.67) [a]		
	rapeseed	%	0.68	(0.50, 0.68, 1.21) [a]		
	peanut	%	0.31	(0.25, 0.31, 0.67) [a]		
	sesame	%	0.59	(0.33, 0.59, 0.67) [a]		
	cotton	%	0.78	(0.51, 0.78, 1.05) [a]		
	vegetables	%	0.06	(0.026, 0.060, 0.064) [a]		
	fruits	%	0.06	(0.026, 0.060, 0.064) [a]		
	corn	%	0.27	(0.24, 0.27, 0.54) [a]		
	beans	%	0.48	(0.4, 0.48, 0.8) [a]		
	potatoes	%	0.16	(0.1, 0.16, 0.21) [a]		
	pork	%	0.13	(0.106, 0.13, 0.218) [a]		
	beef	%	0.168	(0.110, 0.168, 0.226) [a]		
	mutton	%	0.146	(0.095, 0.146, 0.197) [a]		
	Poultry meat	%	0.139	(0.122, 0.139, 0.156) [a]		
	Aquatic products	%	0.185	(0.185, 0.02) [b]		
	egg	%	0.13	(0.10, 0.13, 0.22) [a]		
	milk	%	0.073	(0.048, 0.073, 0.098) [a]		
	sugar	%	0.003	(0.002, 0.003, 0.004) [a]		
	honey	%	0.13	(0.10, 0.13, 0.22) [a]		
m		kg/(ha·a)	2.1	(1.8, 2.1, 2.3) [a]	(CAAS, 2015; Wang, 1997; Xu, 2005)	Medium
n		kg/(ha·a)	0.2	(0.18, 0.2, 0.26) [a]		
k_i	rice	%	32	(31, 32, 33) [a]	(Wu et al., 2014; Xing and Yan, 1999; Xu, 2005)	Low
	rice bran	%	27	(25.8, 27, 28.1) [a]		
	wheat bran	%	15	(14.6, 15, 15.8) [a]		
	corn	%	70	(69, 70, 71) [a]		
	wheat	%	15	(14.6, 15, 15.8) [a]		
	potato	%	20	(18.9, 20, 21) [a]		
Stockbreeding system						
K_n	pig	kg capita^{-1}	1.14	(1.14, 0.27) [b]	(He et al., 1999; Jiang and Yuan, 2015; CAU, 1997; Wu, 2005; Yang, 2002)	
	cow		5.1	(5.1, 1.77) [b]		
	sheep		1.6	(1.6, 0.27) [b]		
	poultry		0.18	(0.18, 0.01) [b]		
	rabbit		0.18	(0.18, 0.01) [b]		
X_f	pig	%	65	(63, 65, 77) [a]	(CAAS, 2015; Jiang and Yuan, 2015; Liu, 2004; Xu, 2005; Yan, 2008)	High
	cow	%	70	(63, 70, 77) [a]		
	sheep	%	40	(36, 40, 44) [a]		

	poultry	%	55	(49, 55, 61) [a]		
	rabbit	%	55	(49, 55, 61) [a]		
m_i	pork	%	0.13	(0.106, 0.13, 0.218) [a]		
	beef	%	0.168	(0.110, 0.168, 0.226) [a]		
	mutton	%	0.146	(0.095, 0.146, 0.197) [a]		
	Poultry meat	%	0.139	(0.122, 0.139, 0.156) [a]		
	Aquatic products	%	0.185	(0.185,0.02) [b]		
	egg	%	0.162	(0.10, 0.162, 0.22) [a]		
	milk	%	0.073	(0.048, 0.073, 0.098) [a]		
	sugar	%	0.003	(0.002, 0.003, 0.004) [a]		
	honey	%	0.162	(0.106, 0.162, 0.218) [a]		
n_i	cow	g capita^{-1}	2820	(2355, 2820, 4846) [a]	(CAAS, 2015; Liu, 2004; Wu et al., 2014; Xu, 2005)	Medium
	pig		460	(300,460, 620) [a]		
	sheep		280	(185, 280 375) [a]		
	rabbit		13	(12, 13, 15) [a]		
	poultry		13	(12, 13, 15) [a]		
Food processing system						
O_i	pig	g capita^{-1}	3.5	(3.3, 3.5, 3.7) [a]	(Du et al., 2021; Xu, 2005; Yuan et al., 2019; Zuo, 2017)	Medium
	cow		1.1	(0.9, 1.1, 1.3) [a]		
	sheep		7.35	(7.21, 7.35, 7.55) [a]		
	poultry		12.7	(12, 12.7, 13.2) [a]		
K_4		%	12	(11, 12, 13) [a]		
K_5		%	10	(9, 10, 12) [a]		
Consumption system						
K_6		%	56.8	(55, 56.8, 57.6) [a]	(Wang et al., 2009)	Low
K_7^u		%	27.10	(26.1, 27.1, 28.6) [a]	(Chen et al., 2008; Feng, 2006; Gu, 2015; He et al., 2010; He et al., 1999; Jiang and Yuan, 2015; Liu et al., 2011; SCPRC, 2008; Wang et al., 2009; Wu et al., 2014; Zhen et al., 2008)	High
K_8^u		%	5.60	(4.8, 5.6, 6.7) [a]		High
K_9^u		%	89.30	(86, 89.3, 91.6) [a]		High
K_7^r		%	95.70	(94, 95.7, 97.5) [a]		High
K_8^r		%	10	(8, 10, 12) [a]		High
P_{mean}^u		kg/per*yr	0.59	(0.48, 0.59, 0.63) [a]		High
P_{waste}^u		kg/per*yr	1.26	(1.12, 1.26, 1.38) [a]		High
P_{mean}^r		kg/per*yr	0.59	(0.45, 0.59, 0.71) [a]		High
P_{waste}^r		kg/per*yr	0.84	(0.64, 0.84, 0.91) [a]		High
K_9^r		%	70	(65, 70, 75) [a]		High
$\varphi_{domestic\ wate}$		%	0.00082	(0.00079, 0.00082, 0.00085) [a]		High
K_{10}		%	30.7	(29.3, 30.7, 31.5) [a]		Low
$Q_{household\ wa}^u$		kg/ren/d	0.525	(0.525, 0.01) [b]		Medium

$Q^{r}_{household\ wa}$		kg/ren/d	0.96	(0.96, 0.05) [b]		High
$\varphi_{kitchen\ refuse}$		%	0.05	(0.04, 0.05, 0.55) [a]		High
K^{u}_{11}		%	65.7	(64, 65.7, 66.5) [a]		High
K^{r}_{11}		%	50.6	(49.1, 50.6, 52) [a]		High
γ	1949-2020	%	6.11-97.5	6.11-97.5		Low
Other calculation equations						
$k_i^{oilcake}$	bean cake	%	20	(17, 20, 22) [a]	(Gao et al., 1997; Jiang and Yuan, 2015; Li, 2002; NRC, 1998; Tian, 2001; Wu et al., 2014; Xu, 2005)	High
	rapeseed	%	55	(50, 55, 62) [a]		
	peanut	%	45	(43, 45, 46) [a]		
	sesame	%	48	(47, 48, 50) [a]		
$f_i^{oilcake}$	bean cake	%	94	(93, 94, 95) [a]		High
	rapeseed	%	10	(8, 10, 11) [a]		
	peanut	%	100	100 [c]		
	sesame	%	100	100 [c]		
$s_i^{oilcake}$	bean cake	%	0.48	(0.46, 0.48, 0.49) [a]		High
	rapeseed	%	0.9	(0.85, 0.9, 0.96) [a]		
	peanut	%	0.5	(0.45, 0.5, 0.58) [a]		
	sesame	%	0.5	(0.45, 0.5, 0.61) [a]		

* The data in the table represents the relevant parameters and their sources for P flow calculations in the Poyang Lake Watershed. For parameters that can be traced to only one reference, uniform distribution is applied. For parameters that can be traced to more than one source, the probability distributions are assumed to be triangular (Jiang and Yuan, 2015). [a] represents the triangular distribution, [b] represents the normal distribution, [c] represents a unique value.

Table S2 Uncertainties in the quantification model

Year	1950			1978			2010			2020		
P flow (Gg yr^{-1})	**Q^{In}**	**Q^{Out}**	**RA**	**Q^{In}**	**Q^{Out}**	**RA**	**Q^{In}**	**Q^{Out}**	**RA**	**Q^{In}**	**Q^{Out}**	**RA**
Croping system	5.62	4.28	23.84%	10.96	0.8	7.30%	24.14	18.9	21.71%	21.27	1.87	8.79%
Animal husbandry	2.02	1.36	32.67%	4.54	3.63	20.04%	11.91	7.34	38.37%	12.78	11.67	8.69%
Consumption system	1.48	0.83	43.92%	3.64	3.09	15.11%	4.67	4.60	1.5%	4.22	4.22	0.043%

* RA_i = | (Q^{In} - Q^{Out})| /Q^{In}. The RA refers to the deviation of input and output in the subsystem. while Q^{In} and Q^{Out} represent the total phosphorus inputs and outputs of the subsystem, respectively

Table S3 P use efficiency and P recycling rate indicators

Category	Indicators	1950	1978	2000	2010	2020
	PUE_s (%)	31.72	40.44	34.72	34.21	39.51
P use efficiency	PUE_c (%)	31.33	45.30	39.77	37.91	48.47
	PUE_a (%)	0.63	1.35	5.33	8.12	8.41
P recycling rate	PRRs (%)	32.65	35.00	29.49	25.25	26.46

* PUEs is the P use efficiency of whole watershed, PUEc is the P use efficiency of croping system, PUEa is the P use efficiency of stockbreeding system.

Table S4 Changes of various phosphorus indicators related to economic activity based on time series (1950-2020)

Year	Crop products	Animal products	Fertilizer usage	Arable area	Population	Per capita GDP	Urbanization rate	Consumption level	Dietary structure	P Resources	P consumption	P emissions	P Utilisation rate	P recovery rate	P contamination rate	P Stock Rate
unit	t	t	t	km^2	-	RMB	%	RMB	-	10^4 t	10^4 t	10^4 t	%	%	%	%
2020	41488466	6192953	647547	56444	45188635	56871	60.44	24089	13.52	13.45	3.61	8.73	39.51	32.48	25.63	14.19
2019	40790444	6239246	687849	55212	45159480	53164	59.07	23109	14.03	13.01	3.03	7.84	40.34	31.90	24.30	14.35
2018	40627496	6389313	713180	55559	45134968	47434	57.34	20477	12.58	13.28	2.96	7.76	39.54	31.23	23.09	17.27
2017	40440125	6320365	765535	55969	45114818	43424	55.70	17837	9.93	13.30	3.26	7.83	38.28	29.76	22.52	19.41
2016	38930535	6513878	794918	56021	44956495	40400	53.99	16204	9.43	12.86	3.33	7.98	35.57	29.64	22.07	22.60
2015	38448647	6907748	798302	56884	44845311	36724	52.30	14668	8.69	13.60	3.44	8.54	34.77	31.94	21.81	26.13
2014	37839409	6799598	791407	55705	44797265	34674	50.55	13293	9.12	13.47	3.12	8.27	34.68	29.69	21.29	24.40
2013	36897491	6570456	779427	56374	44755616	31930	49.04	11933	7.14	13.62	3.41	8.31	35.48	30.33	21.64	23.58
2012	36159960	6402684	772407	55978	44754934	28800	47.39	10426	6.22	13.01	3.51	8.18	34.21	30.89	21.50	25.99
2011	46907992	6058022	759596	55464	44739303	26150	45.75	9348	5.88	13.09	3.72	8.06	35.22	31.92	21.67	25.34
2010	41749182	5871879	716882	55050	44622489	21253	44.06	7846	5.11	12.33	3.88	7.85	34.21	34.42	21.77	28.20
2009	43541198	5688306	716882	54115	44321581	17277	43.18	6172	5.41	12.50	3.97	7.67	35.31	34.24	21.67	26.84
2008	43883747	5169174	680005	53541	44001038	15816	41.36	5692	4.57	12.10	4.25	7.51	35.47	35.02	22.02	26.89
2007	33201612	4948789	660307	52264	43684125	13270	39.80	4665	4.68	11.44	4.29	7.16	34.60	34.48	21.65	28.82
2006	31405484	4873222	642923	52556	43391287	10859	38.68	4052	4.49	11.44	4.41	7.42	32.76	34.60	21.24	32.49
2005	31753827	4685828	608659	53289	43112439	9172	37.10	3693	4.71	11.43	4.25	7.56	32.33	36.67	21.37	33.42
2004	30993020	4291798	559692	52581	42835667	7960	35.58	3277	3.57	11.11	4.49	7.54	32.57	38.66	22.10	32.91
2003	27040456	3957955	476871	49974	42542255	6636	34.02	2739	3.85	9.69	4.50	7.20	30.39	41.23	22.55	36.91
2002	31086724	3785234	466627	53551	42224273	5829	32.20	2651	3.28	10.31	4.58	7.17	33.01	40.76	22.96	31.81
2001	31102222	3653563	431915	55347	41857676	5221	30.41	2500	3.22	10.55	4.68	7.40	34.82	43.98	24.40	28.92
2000	30483060	3591084	422719	56508	41485447	4851	27.69	2396	2.90	10.61	4.95	7.46	34.72	44.89	24.41	29.03
1999	29214307	3599382	423515	58710	42311742	4402	26.79	2056	2.99	10.84	5.23	7.55	34.75	44.24	24.19	29.16
1998	28043083	3720968	397847	58040	41912074	4124	26.05	1973	2.77	10.40	5.09	7.45	32.83	45.98	23.50	33.31
1997	39819660	3844979	417000	60376	41503338	3890	25.32	1930	2.87	12.50	5.04	7.81	36.89	45.14	23.04	26.65
1996	27327321	3617137	375000	61053	41054635	3452	24.58	1857	2.67	11.07	5.26	7.65	34.62	46.45	23.90	29.53
1995	25627460	3407884	356000	59495	40625406	2896	23.85	1559	2.46	10.45	5.15	7.40	33.21	46.31	23.52	32.08
1994	26866736	3013922	341000	57534	40154459	2376	23.29	1182	2.20	10.30	5.24	7.09	34.52	46.42	23.75	29.81
1993	28037281	2521759	296000	57210	39660405	1835	22.55	887	2.24	9.98	5.34	6.68	36.65	48.67	24.50	26.94
1992	24744541	2077141	288000	58449	39130927	1472	21.82	770	2.13	9.57	4.99	6.37	36.59	47.24	24.34	26.16
1991	22312887	1803008	269000	58297	38646374	1249	21.08	706	2.06	9.23	4.99	6.14	36.98	47.58	24.57	25.41
1990	22363834	1621330	242000	57597	38106418	1134	20.35	666	1.91	9.15	5.03	5.99	38.44	49.08	25.16	22.86
1989	25969169	1505897	213000	55553	37462196	1013	20.22	580	1.96	9.10	4.87	5.71	40.98	50.89	25.68	18.92
1988	24642961	1412877	206000	53963	36838811	891	20.11	506	2.16	8.72	4.80	5.51	40.48	50.43	25.56	19.98
1987	24421759	1222691	187000	54827	36323111	729	20.00	427	2.11	8.66	4.68	5.37	43.12	52.52	26.74	15.04
1986	22401556	1014111	183877	54387	35757637	652	19.89	395	2.04	8.05	4.54	5.21	41.40	52.39	26.76	17.43
1985	24049417	941764	186085	54191	35097971	597	19.78	367	1.74	8.26	4.43	5.01	44.33	50.46	26.90	12.47
1984	24284803	382780	192219	54567	34578879	497	19.67	311	1.64	8.12	4.50	4.84	44.91	48.14	26.77	11.32
1983	23083332	332772	184860	54653	33945033	428	19.56	282	1.54	7.66	4.58	4.69	44.66	48.07	27.31	11.72
1982	20929726	656654	180058	55783	33483485	403	19.45	266	1.63	7.32	4.38	4.58	44.07	48.26	27.54	12.30
1981	20228695	617010	172883	55428	33039235	369	19.06	230	1.88	6.81	3.41	4.24	42.17	48.25	26.20	15.42
1980	19555906	556955	169518	55537	32701960	342	18.79	211	1.30	6.62	3.97	4.31	41.41	48.56	26.97	16.61
1979	19969176	476804	169716	56995	32289778	325	17.44	203	1.49	6.83	3.24	4.18	42.87	48.77	26.20	14.09
1978	19514956	409353	166541	57011	31828203	276	16.75	181	1.23	6.26	3.06	3.99	40.44	48.08	25.75	18.08
1977	18668657	362301	163176	57883	31180000	243	16.68	156	1.21	6.04	2.85	3.91	39.78	47.81	25.72	18.77
1976	17344489	367461	163174	56903	30480000	214	16.46	152	1.04	5.74	2.69	3.82	38.12	47.44	25.40	21.64
1975	17148505	393826	159437	57589	29685000	233	16.55	154	1.05	5.82	2.58	3.84	38.92	47.37	25.65	21.86
1974	15546036	378858	157000	58032	28883000	227	16.61	157	0.92	5.47	2.63	3.77	37.14	46.87	25.58	23.06
1973	16013674	394749	154220	58115	28105000	242	16.78	149	0.91	5.37	2.43	3.68	36.80	46.73	25.19	23.93
1972	16717889	345026	150600	57872	27230000	248	16.62	150	0.99	5.64	2.15	3.66	38.51	47.24	24.97	21.42
1971	16411551	248876	146366	59077	26523000	238	17.89	140	0.97	5.59	2.25	3.70	38.92	47.73	25.75	20.44
1970	14650849	277071	145256	57427	25845000	229	15.91	125	0.98	5.97	2.04	3.66	41.75	47.69	25.59	15.98
1969	14271907	318895	144619	59421	25047000	211	15.66	125	1.03	5.27	1.92	3.46	38.69	45.62	25.43	20.62
1968	13565461	281848	144096	55895	24182000	193	15.37	124	1.06	4.92	1.77	3.27	37.47	44.64	24.89	22.73
1967	12744597	210287	136930	56812	23544000	192	16.70	127	0.92	4.59	1.88	3.26	36.20	44.81	25.73	23.78
1966	14439871	184614	133310	58251	22836000	215	16.67	123	0.82	4.71	1.83	3.28	37.83	44.55	26.34	20.62
1965	13942523	247344	136388	56977	22095000	197	16.86	126	0.86	4.56	1.75	3.18	37.12	43.36	25.86	21.88
1964	12250001	204978	134304	55315	21436000	166	16.72	120	0.77	4.07	1.79	3.02	34.51	42.08	25.53	25.77
1963	11046875	188813	128460	53011	21010300	158	16.53	115	0.52	3.69	1.54	2.83	32.70	42.07	25.00	28.57
1962	9799368	132752	129088	53333	20399100	163	20.40	116	0.42	3.40	1.41	2.73	31.09	40.45	25.01	30.51
1961	10790526	134805	119697	55763	20226700	176	21.82	113	0.36	3.40	1.33	2.79	32.27	41.52	26.40	28.04
1960	10946562	93287	116194	59520	20098500	185	22.90	94	0.55	3.71	1.37	2.96	34.56	42.68	27.53	24.41

1959	10898672	107303	115632	57463	19759700	178	14.19	94	0.92	3.86	1.36	2.90	35.13	44.89	26.40	25.04
1958	9610857	142314	109768	58036	19128900	166	12.67	85	0.78	3.83	1.66	2.98	35.07	46.21	27.25	24.87
1957	10265196	153961	104636	54612	18514500	152	12.16	99	0.90	3.97	1.38	2.84	37.12	46.98	26.60	22.59
1956	9884575	158843	104143	53124	18000000	124	11.88	89	0.78	3.64	1.62	2.68	37.28	44.77	27.44	21.54
1955	9453857	146928	99541	48663	17634000	125	11.71	85	0.71	3.49	1.30	2.50	36.81	44.94	26.41	23.06
1954	10240685	165750	96608	46570	17297000	121	11.38	82	0.57	3.40	1.41	2.44	36.23	45.16	26.02	25.22
1953	9814468	118760	96591	45438	16953000	118	10.27	79	0.66	3.41	1.40	2.40	36.71	45.30	25.86	24.49
1952	9165397	118877	90673	44095	16557000	114	10.38	77	0.72	3.30	1.32	2.32	37.43	45.66	26.28	22.73
1951	7095500	100804	88266	35219	16439100	94	10.30	74	0.55	2.66	1.12	1.98	31.48	45.24	23.42	34.52
1950	6380756	94413	87365	33359	15681200	98	10.20	75	0.63	2.51	1.06	1.83	31.72	43.38	23.16	32.97

* The data in the table is used for analyzing the driving factors of long-term P flow variations in the Poyang Lake Watershed. The methods employed for this analysis are Pearson correlation analysis and Principal Component Analysis (PCA).

Table S5 Varimax rotated PCA of P flow-related indicators in the Poyang Lake watershed

	Principal components		
	PC1	PC2	PC3
CP	**0.951**	0.19	0.133
AP	**0.989**	-0.098	0.04
FU	**0.983**	-0.102	0.006
AA	0.233	0.638	0.215
PO	**0.888**	0.295	0.329
PERGDP	**0.868**	-0.021	-0.48
UR	**0.972**	-0.01	-0.189
CL	**0.854**	0.001	-0.49
DS	**0.908**	0.05	-0.37
P-Re	**0.951**	0.205	0.215
P-Co	0.529	0.454	0.665
P-Em	**0.935**	0.154	0.305
P-UR	-0.171	**0.901**	-0.276
P-RR	-0.792	0.426	0.355
P-CR	-0.752	0.528	-0.226
P-SR	0.121	-0.836	0.507
Eigenvalues	10.238	2.796	1.909
% of variance	63.985	17.476	11.931
Cumulative %	63.985	81.46	93.391

Signifcant values are in bold typeface.

Figure

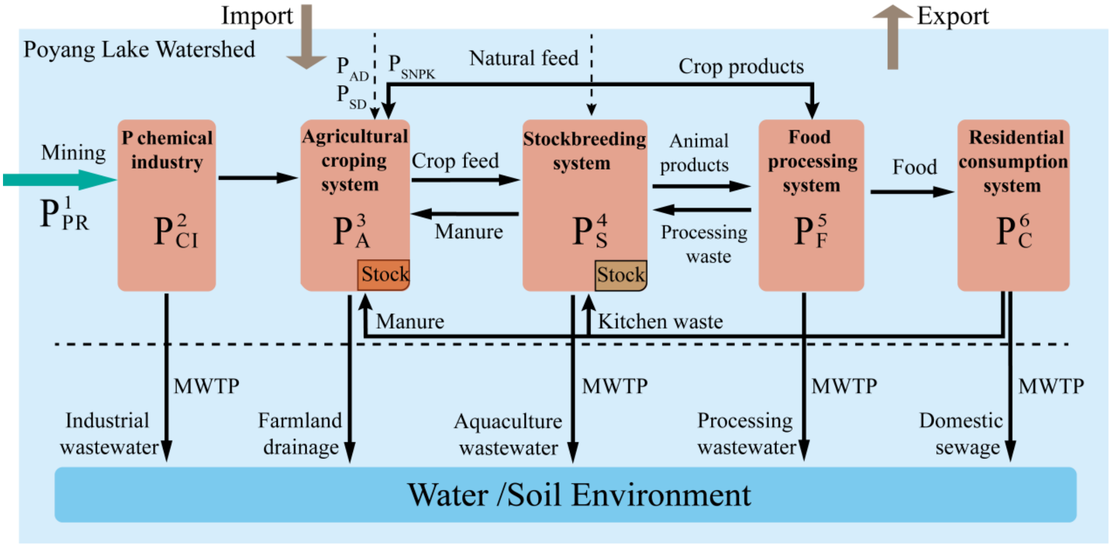

Fig. 1 Static analytical model boundary of anthropogenic P cycles in lake watersheds. P^1_{PR} : P phosphorite resources, P^2_{CI} : P chemical product production system, P^3_A : Agricultural croping system, P^4_S : Stockbreeding system, P^5_F : Food processing system, P^6_C : Residential consumption system, P_{AD} : The amount of P through atmospheric deposition, P_{SD} : The amount of P deposited in the soil, P_{SNPK} : The P content of returned straw, MWTP: Municipal wastewater treatment plant. Numbers represent the order of subsystems.

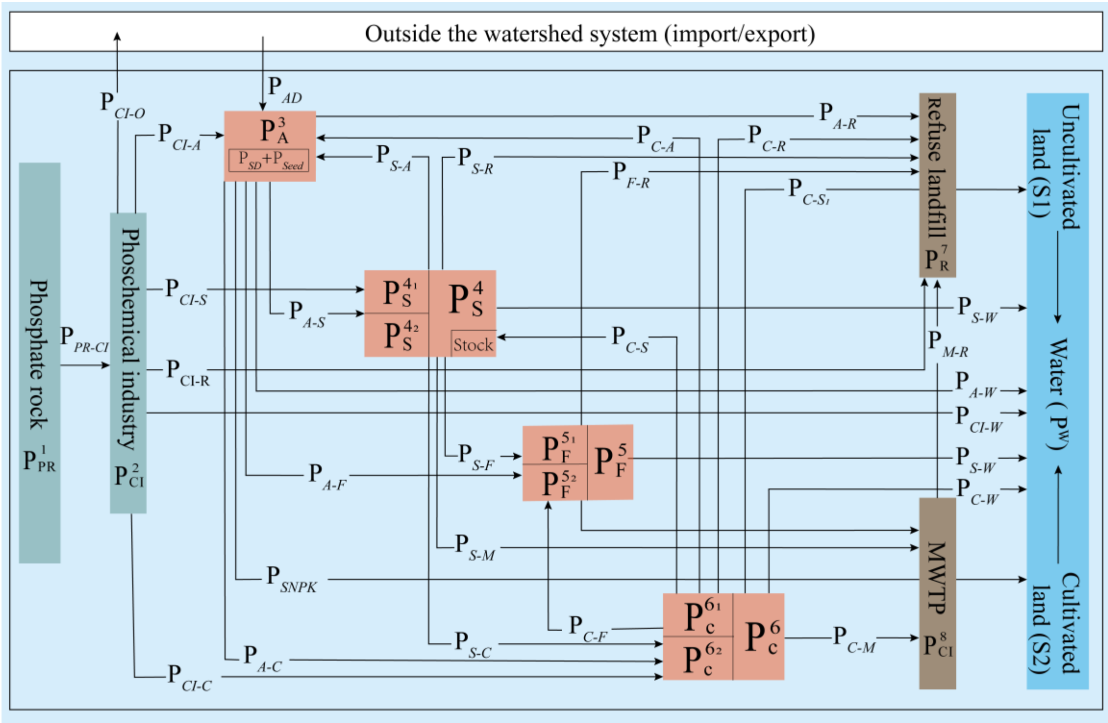

Fig. 2 The P flow paths of WPSFA model. The arrows describe the inputs and outputs of the P flow. For example, P_{CI-A} represents the flow from P_{CI} to P_A, symbol abbreviations are explained in Fig. 1 and Text S4, and the calculation process is described in Text S4. The boxes indicate individual processes or sub-processes in which some human activities related to P flows occur. These sub-processes may change slightly in different ecosystems because of their different economic and consumption activities. Stock refers to the accumulation of P in these subsystems, which does not flow among different activities in a short time (at least one year).

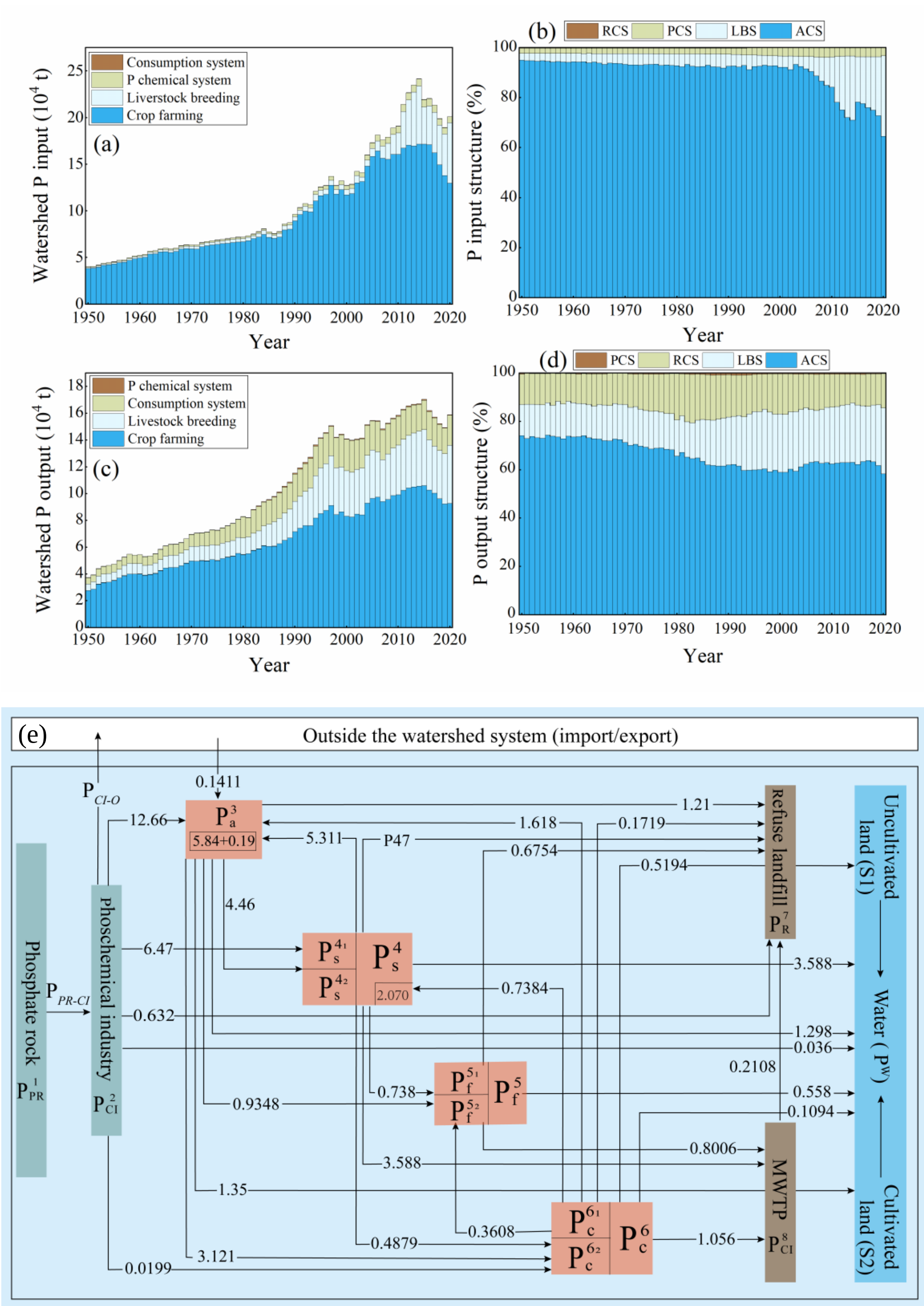

Fig. 3 P flow pattern in the Poyang Lake watershed. (a) Watershed input P, (b) Watershed P input structure; (c) Watershed output P; (d) Watershed P output structure; (e) Anthropogenic P flows for the entire watershed in 2020 (10^4 t). In this context, input P represents "new P" entering the system, encompassing industrial products such as

fertilizers, feed, detergents, and other related compounds. RCS: Residential consumption system, LBS: Livestock breeding system, ACS: Agricultural cropping system, CIS: Chemical industrial system.

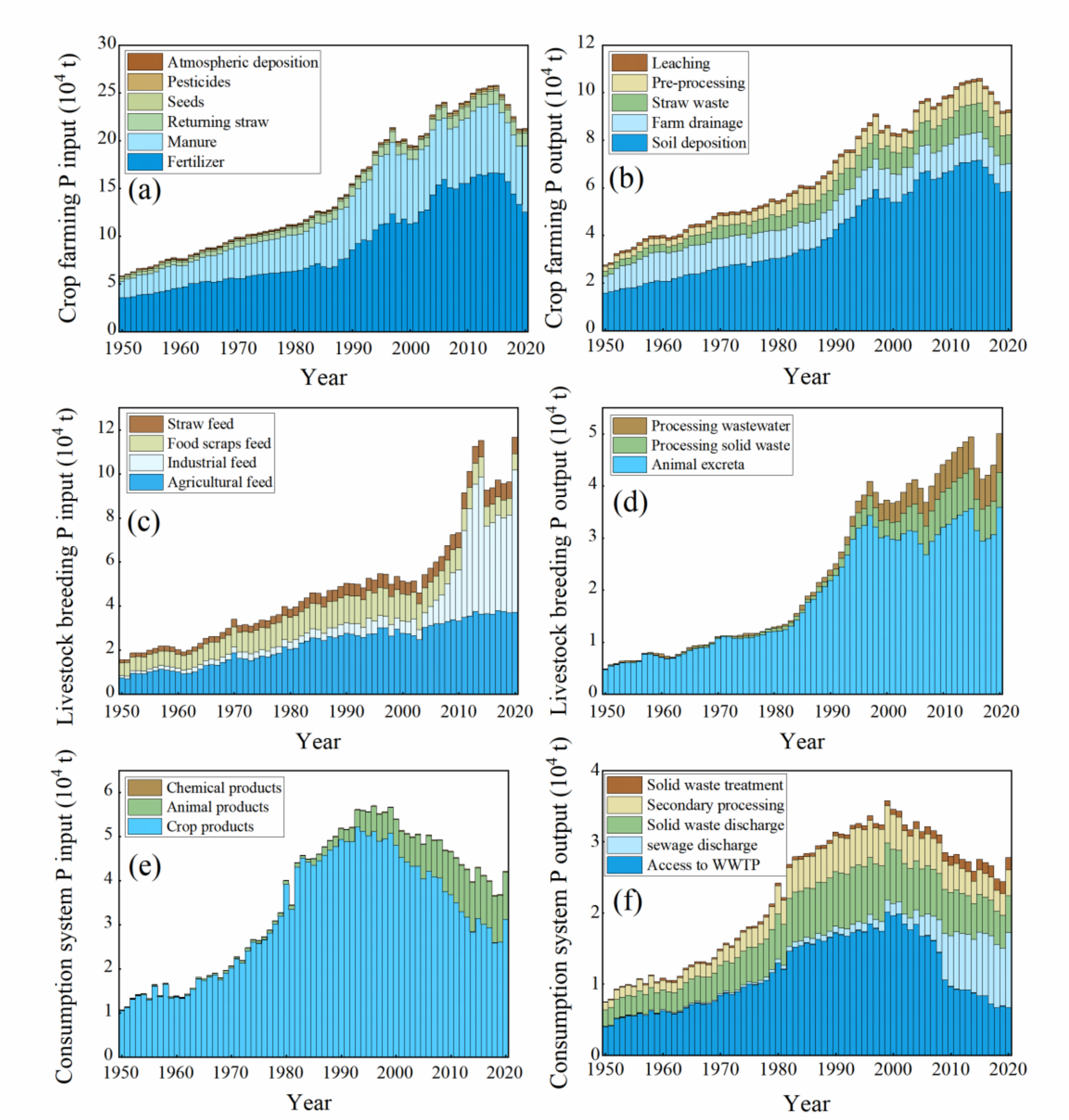

Fig. 4 Sub-watershed P input and output in the Poyang Lake watershed during 1950–2020. (a) P input into agricultural cropping system. (b) P output in the agricultural cropping system. (c) P input into the livestock breeding system. (d) P output in livestock breeding system. (e) P input into the consumption system. (f) P output in the consumption system. In a specific year, if production exceeds consumption, the excess implies that P exports. In contrast, a shortage indicates the P imports.

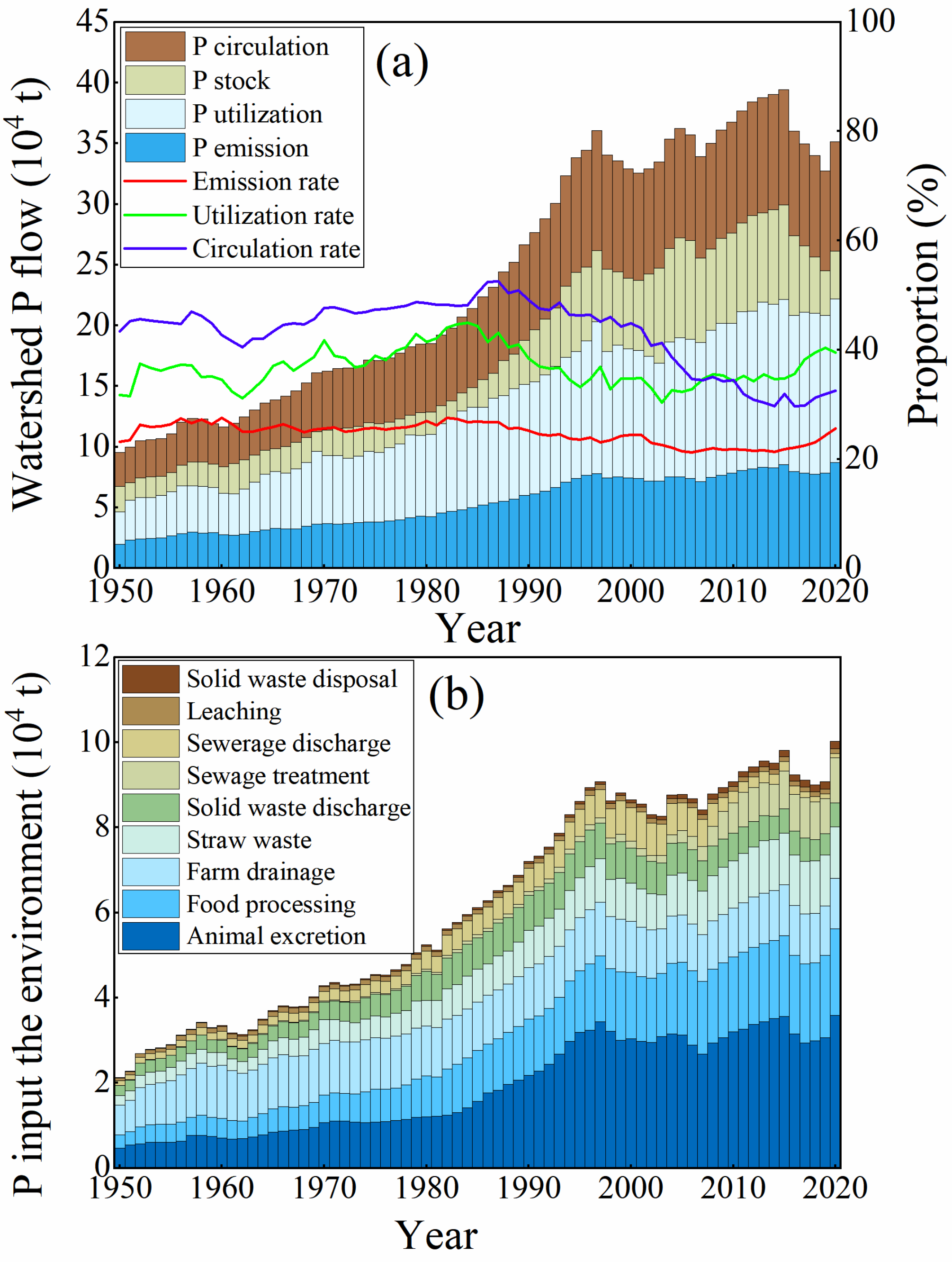

Fig. 5 P cycle characteristics (a) and sources of whole system P input (b) in the Poyang Lake watershed from 1950 to 2020.

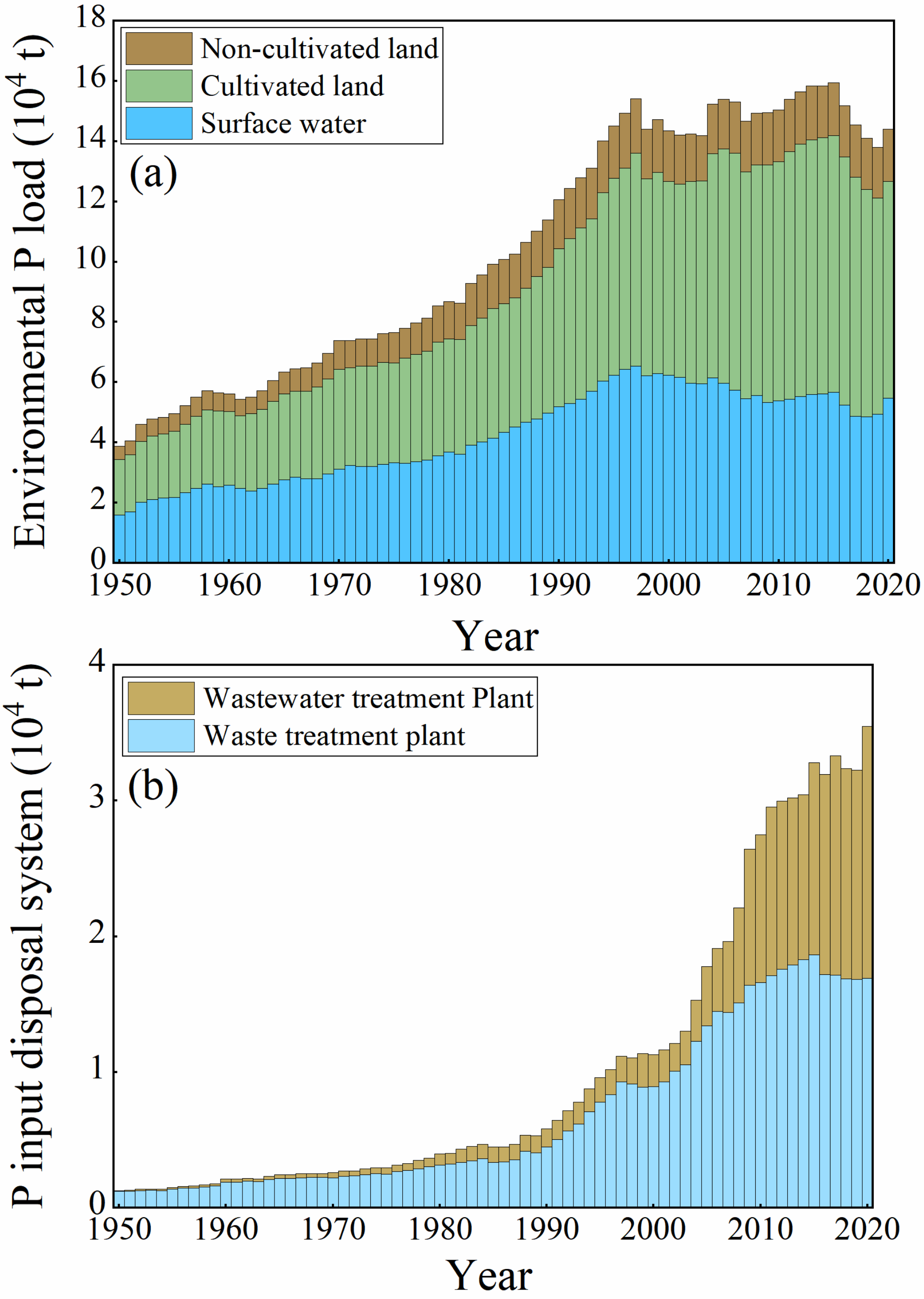

Fig. 6 P access to the environment (cultivated land, non-cultivated land, and surface water) (a) and waste (wastewater) disposal system (b) in the Poyang Lake watershed from 1950 to 2020.

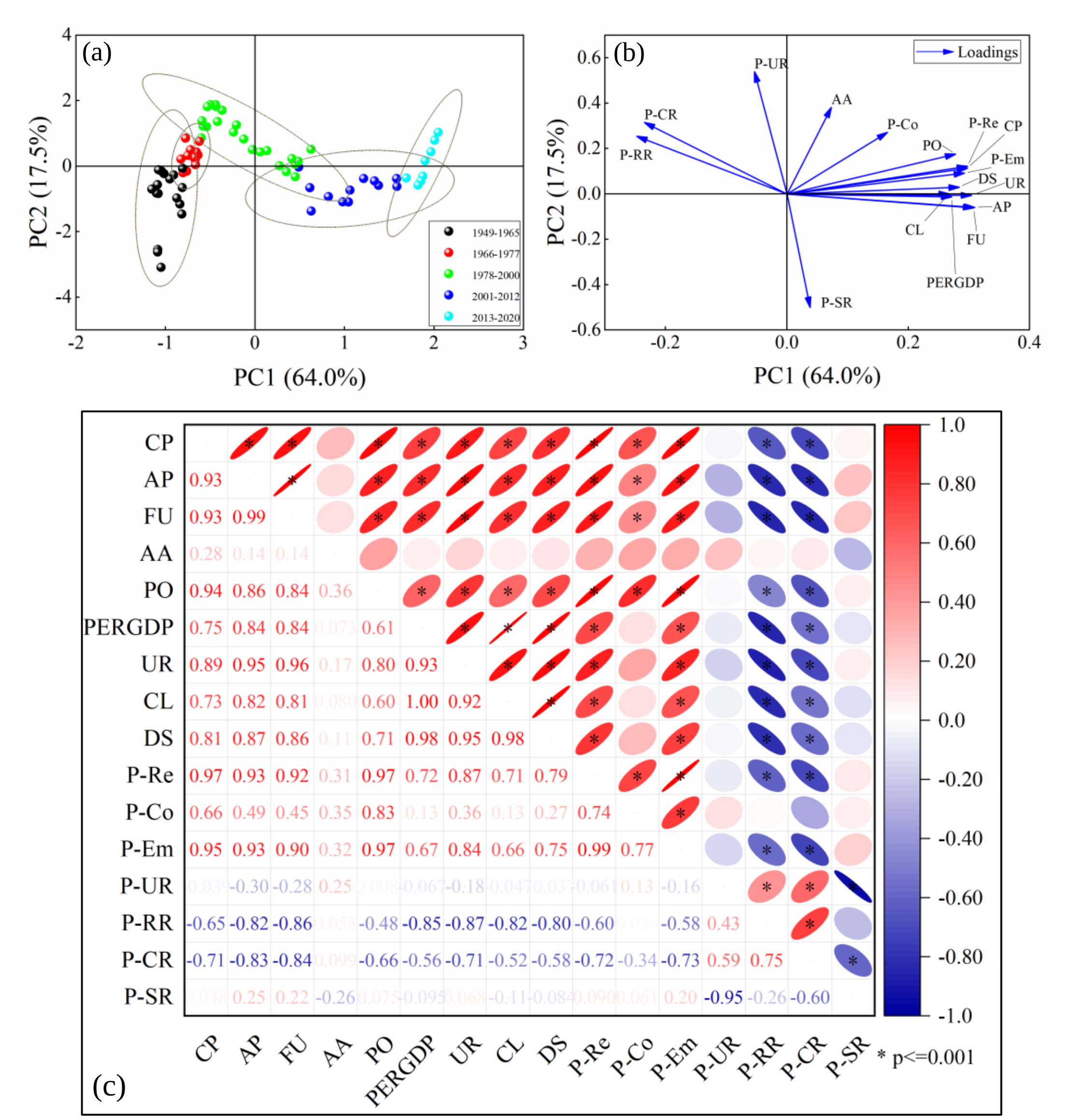

Fig. 7 PCA loading diagrams (a, b) and relatedness (c) of factors related to P flow. CP: crop products; AP: animal products; FU: fertilizer usage; AA: arable area; PO: population; PERGDP: Per capita GDP; UR: urbanization rate; CL: consumption level; DS: dietary structure; P-Re: P resources; P-Co: P consumption; P-Em: P emissions; P-UR: P utilization rate; P-RR: P recovery rate; P-CR: P contamination rate; and P-SR: P stock rate.

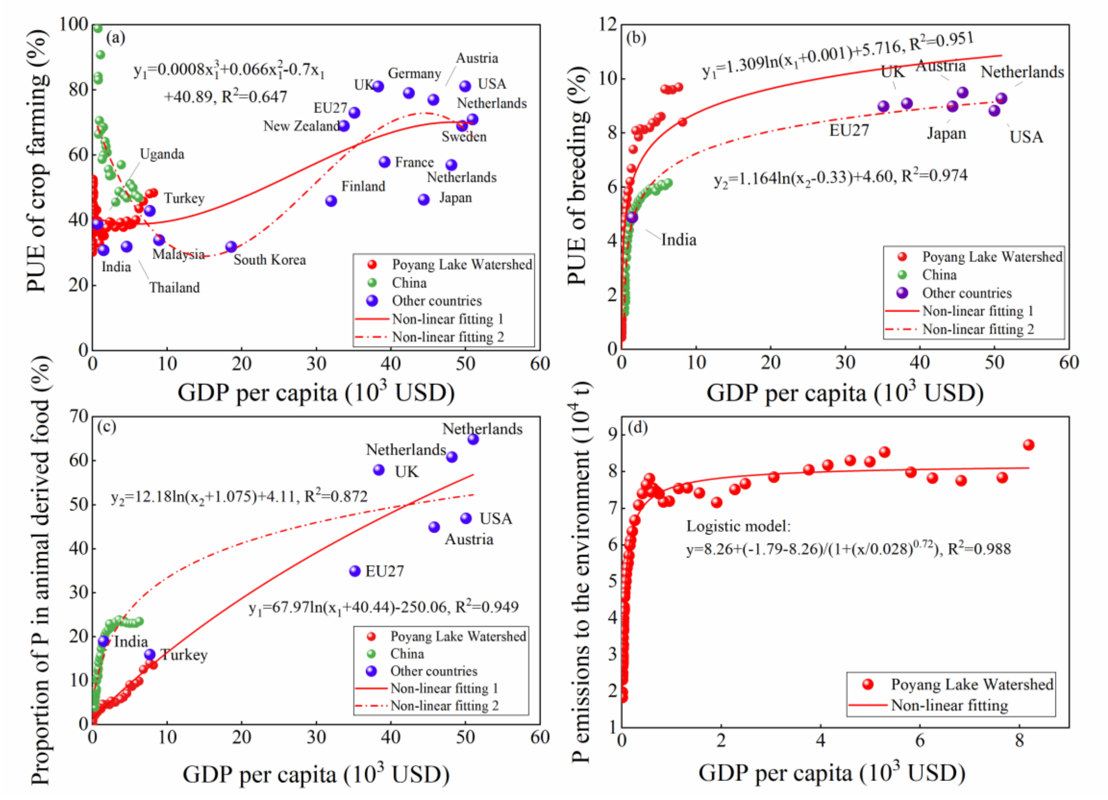

Fig. 8 Relationship between GDP per capita and the P-flow-related indicators, based on Jiang et al. (2015). We tested the relationship between these indicators and economic development (GDP per capita) and compared Jiangxi Province results with China and other countries, including Austria, EU27, Finland, France, Germany, India, Japan, Malaysia, Netherlands, New Zealand, South Korea, Sweden, Switzerland, Thailand, Turkey, Uganda, the United Kingdom and the United States (Antikainen et al., 2005; Cooper and Carliell-Marquet, 2013; Cordell et al., 2013; Jedelhauser and Binder, 2015; Jeong et al., 2009; Keil et al., 2018; Lederer et al., 2015; Li et al., 2015; Linderholm et al., 2012; Liu et al., 2016; MacDonald et al., 2012; Matsubae-Yokoyama et al., 2009; Matsubae et al., 2011; Prathumchai et al., 2016; Seyhan, 2009; Smit et al., 2015; Suh and Yee, 2011).

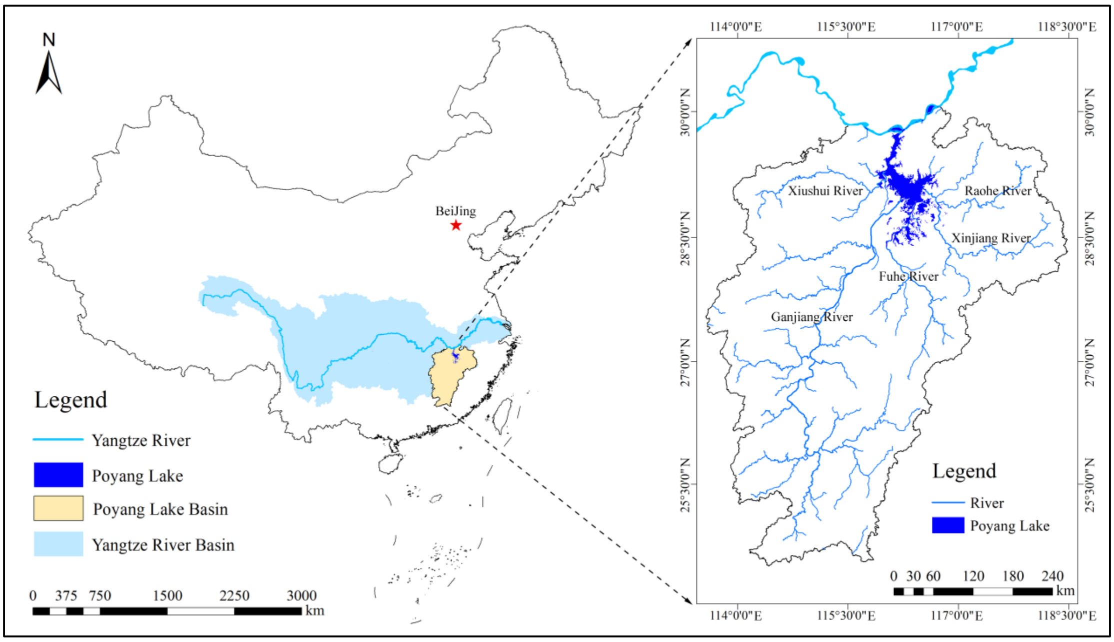

Fig. S1 Location of the Poyang Lake watershed.

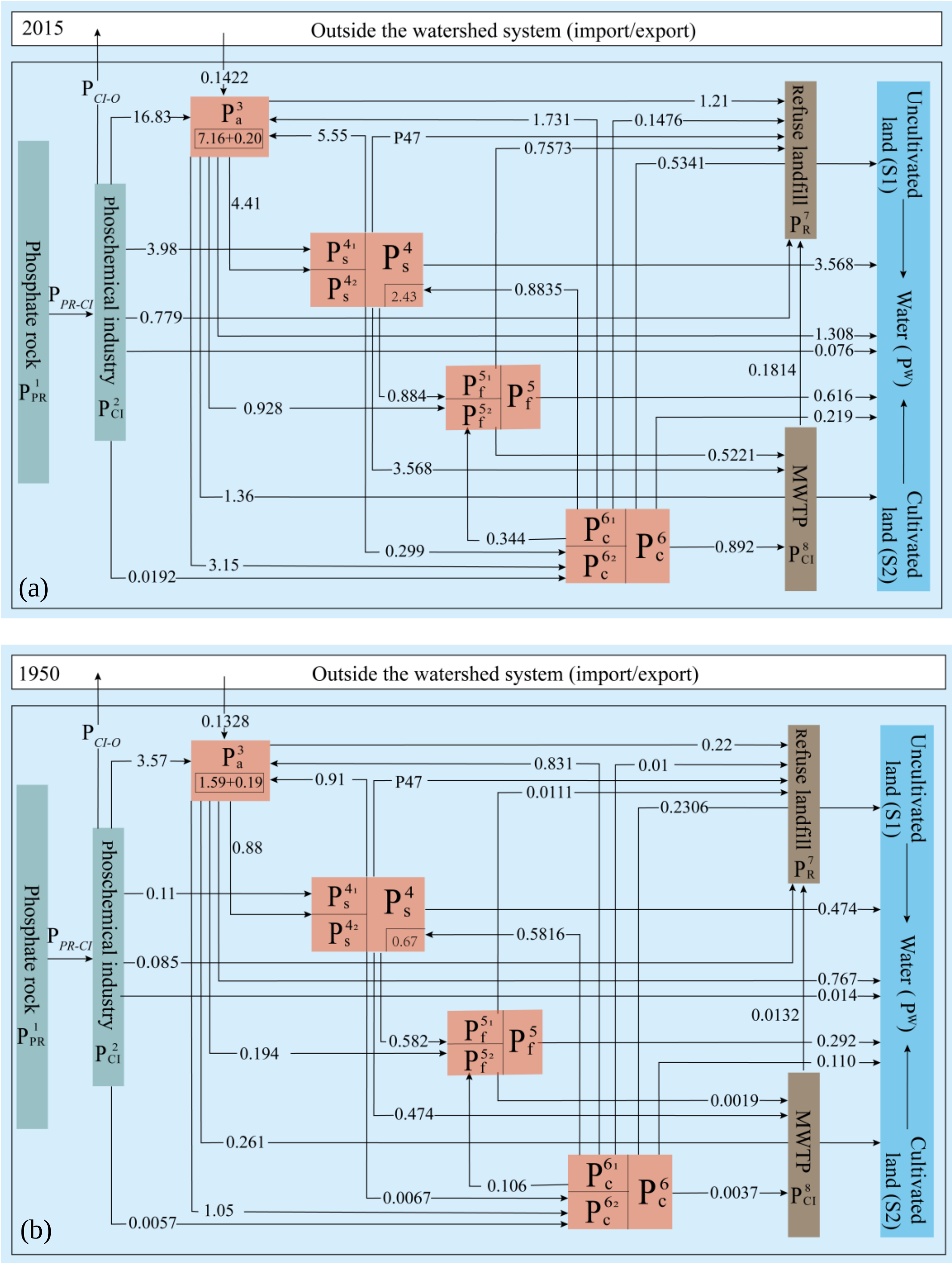

Fig. S2 Anthropogenic P flows for the entire watershed in (a) 2015 and (b) 1950 (10^4 t). In this context, input P represents "new P" entering the system, encompassing industrial products such as fertilizers, feed, detergents, and other related compounds. P_{PR}^{1} : P phosphorite resources, P_{CI}^{2} : P chemical product production system, P_{A}^{3} :

Agricultural croping system, P_S^4 : Stockbreeding system, P_F^5 : Food processing system, P_C^6 : Consumption system.

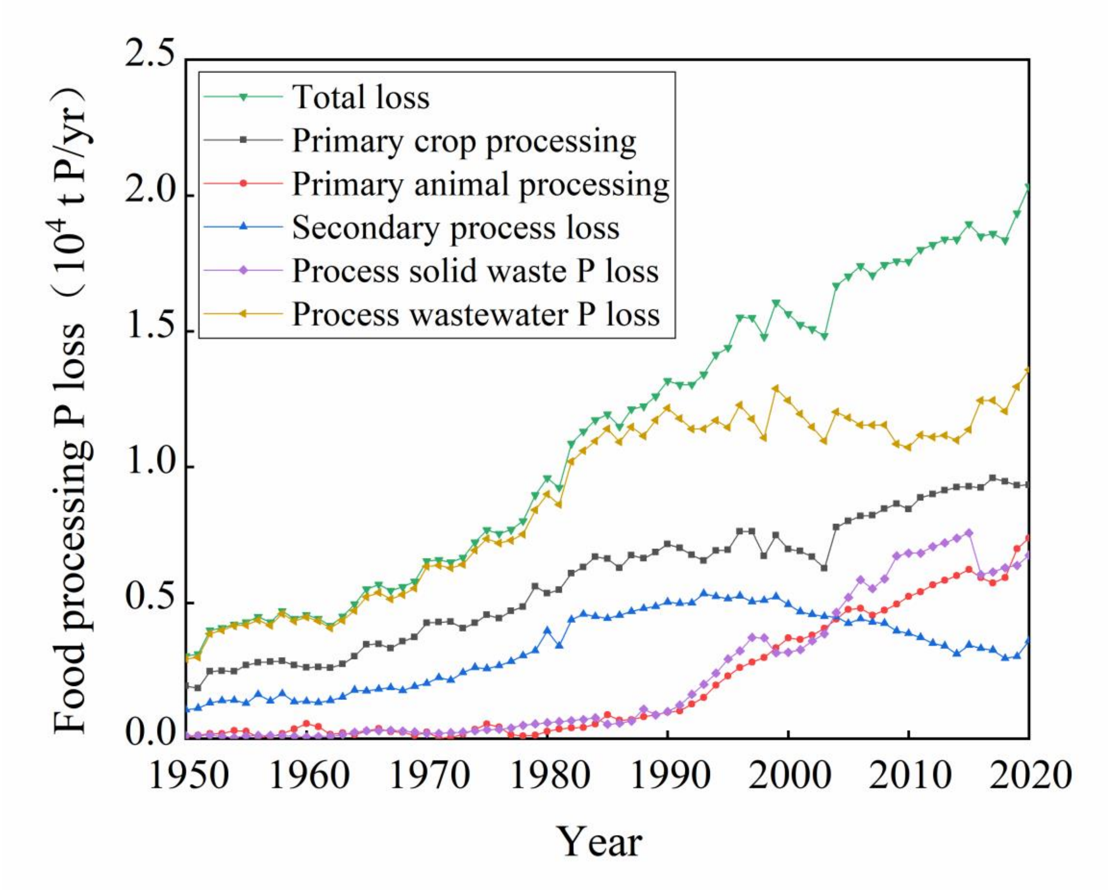

Fig. S3. P Loss in food Processing in the Poyang Lake watershed from 1950 to 2020.

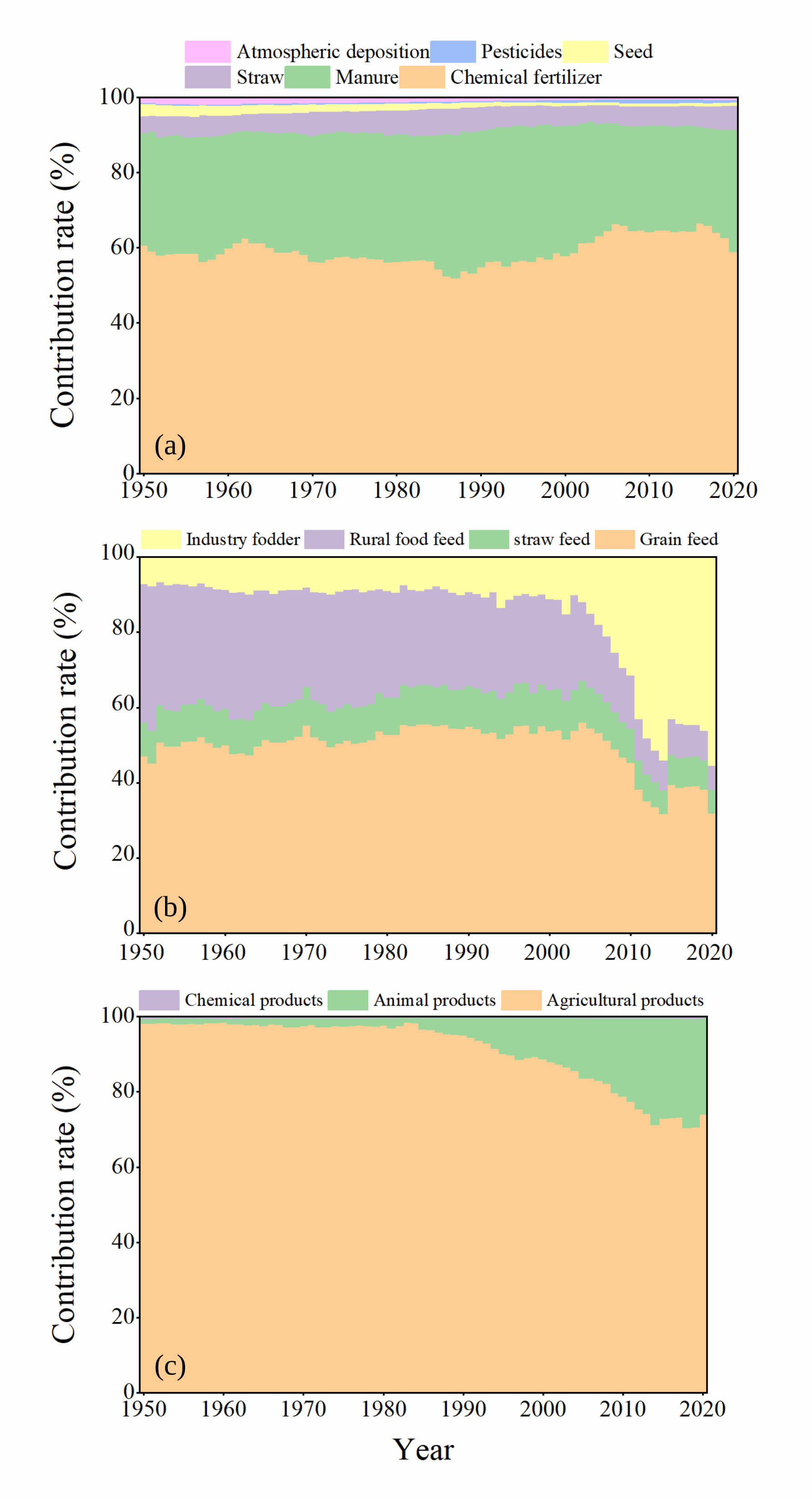

Fig. S4. Subsystem P input structure in the Poyang Lake watershed from 1950 to 2020. (a) Agriculture crop system (ACS), (b) Livestock breeding systems, (c) Residential consumption system.

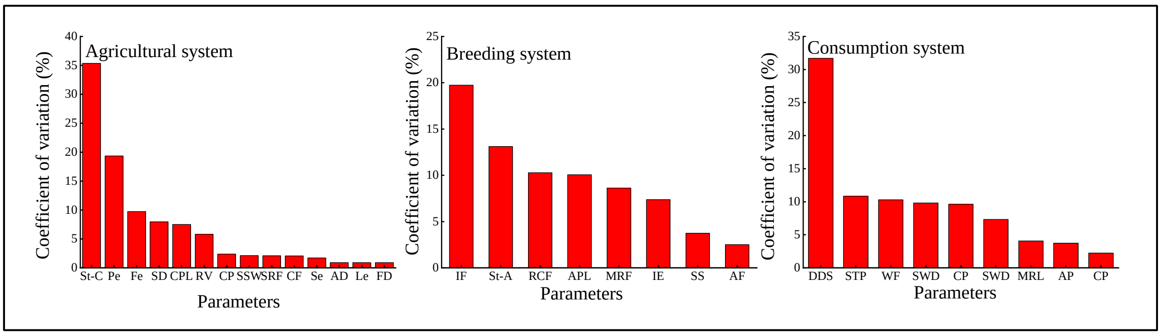

Fig. S5. Variation coefficients of parameters on the variation of P flow in agriculture crop system (a), livestock breeding systems (b) and residential consumption system (c) in 2020. St-C is stock. Pe is pesticides. Fe is fertilizers. SD is soil deposition. CPL is crop product processing losses. RV is return volume. CP is crop products. SSW is straw solid waste. SRF is straw returned to the field. CF is Crop feed. Se is seeds. AD is atmospheric deposition. Le is leaching. FD is farm drainage. IF is industrial feeds. St-A is stocking. RCF is rural consumption into farming. APL is animal product processing losses. MRF is manure and urine returned to the farm. IE is input to the environment. SS is slaughtered and sold. AF is agricultural feed. DDS is direct discharge of sewage. STP is into sewage treatment plants. WF is waste farming. SWD is solid waste disposal. CP is crop products. DDSW is direct discharge of solid waste. MRL is manure return to land. AP is animal products. CP is chemical products.

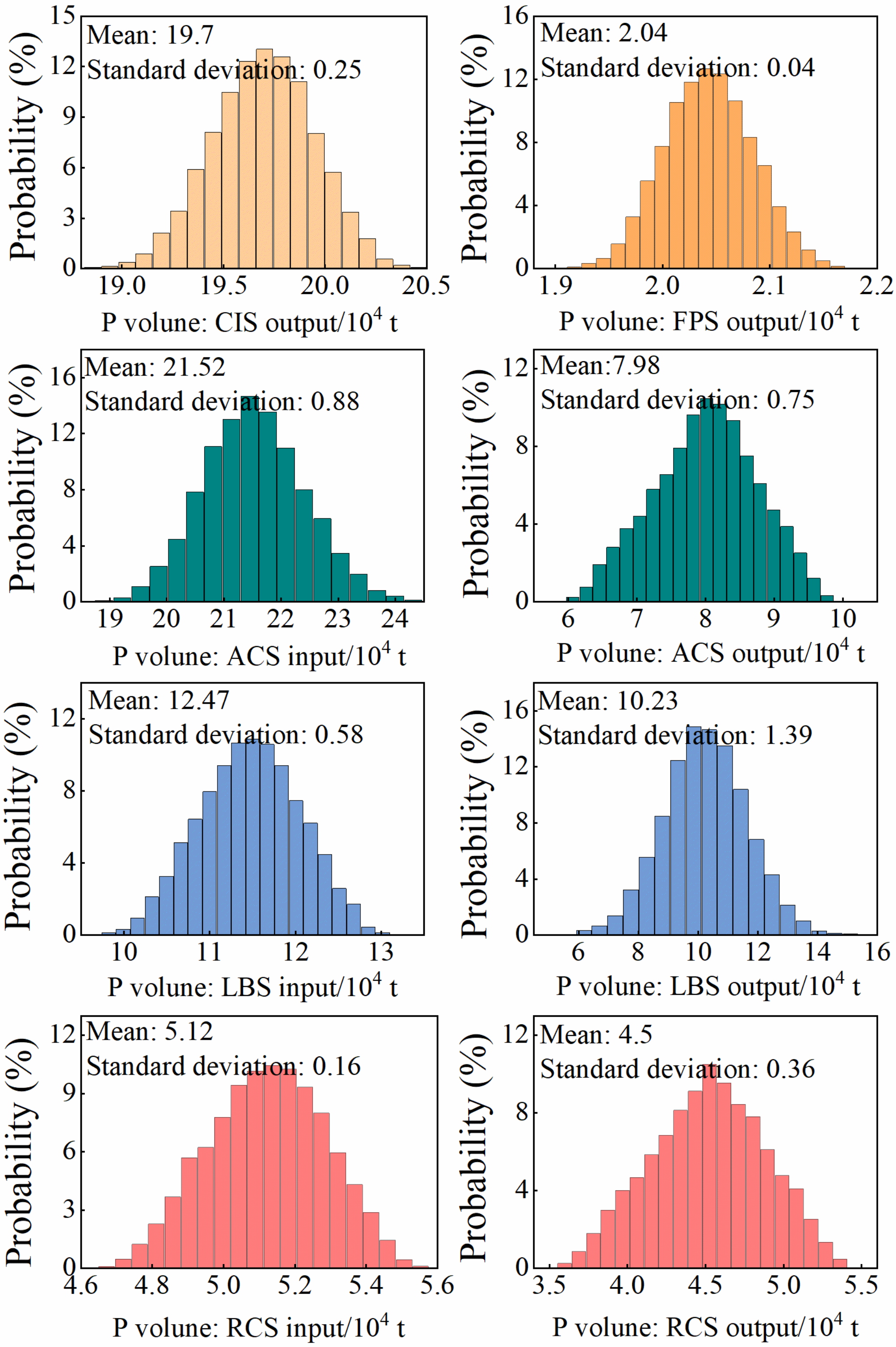

Fig. S6. The aggregated uncertainties of P input-output subsystem in the Poyang Lake watershed. CIS: Chemical industrial system; FPS: Food processing system, ACS: Agriculture crop system, LBS: Livestock breeding systems, RCS: Residential consumption system.

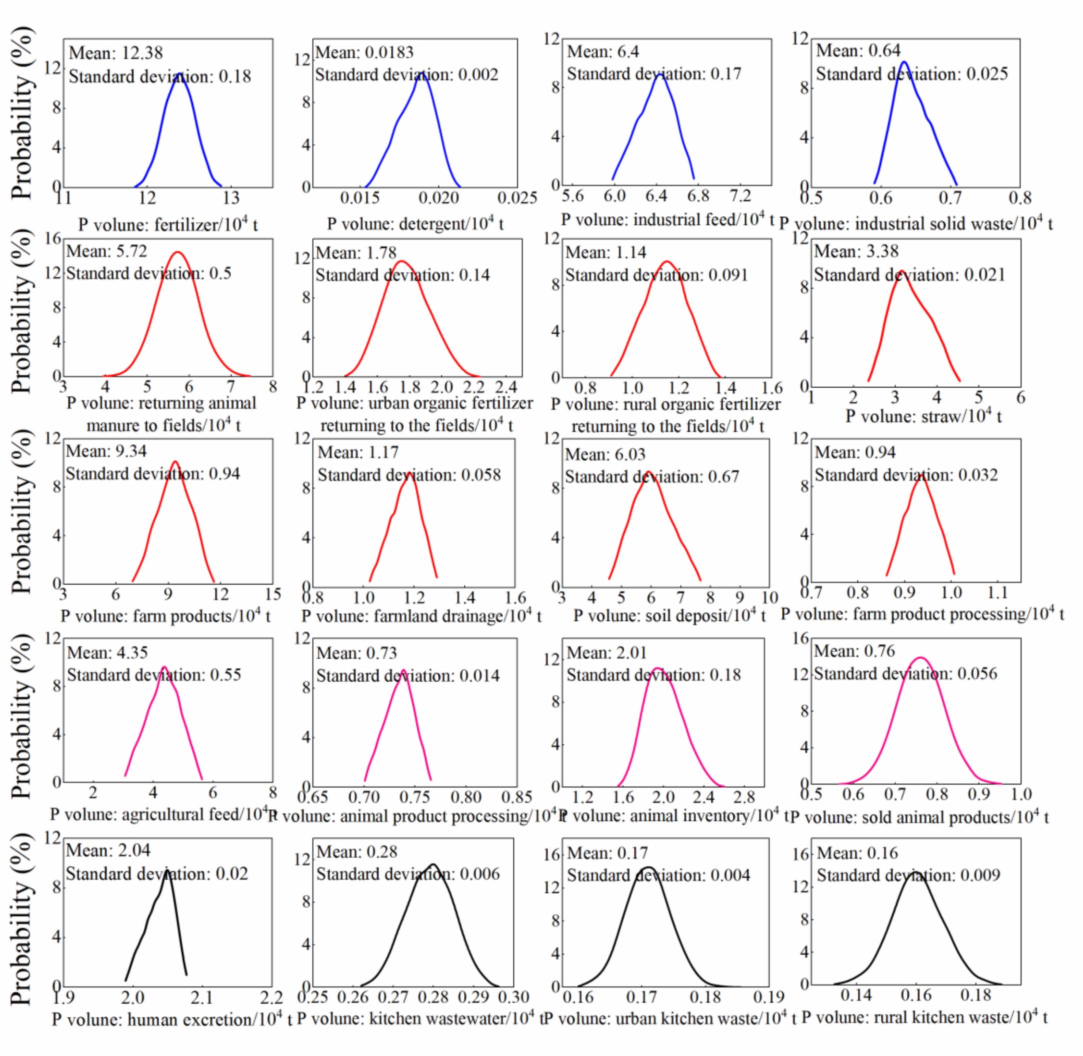

Fig. S7. The uncertainty of major P flows in the Poyang Lake watershed. Blue represents the major P flows associated with the CIS, red represents the major P flows associated with the ACS, pink represents the major P flows associated with the LBS, and red represents the major P flows associated with the RCS.