Agricultural land use regulates the fate of soil phosphorus fractions following the reclamation of wetlands

Qin, Lei; Jiang, Ming; Freeman, Chris; Zou, Yuanchun; Gao, Chuanyu; Tian, Wei; Wang, Guodong

Science of the Total Environment

DOI:
10.1016/j.scitotenv.2022.160891

Published: 10/03/2023

Peer reviewed version

Dyfyniad o'r fersiwn a gyhoeddwyd / Citation for published version (APA):
Qin, L., Jiang, M., Freeman, C., Zou, Y., Gao, C., Tian, W., & Wang, G. (2023). Agricultural land use regulates the fate of soil phosphorus fractions following the reclamation of wetlands. *Science of the Total Environment*, *863*, Article 160891. Advance online publication. https://doi.org/10.1016/j.scitotenv.2022.160891

Agricultural land use regulates the fate of soil phosphorus fractions following the reclamation of wetlands

Lei Qin[1], Ming Jiang[1,3], Chris Freeman[2], Yuanchun Zou[1,3], Chuanyu Gao[1], Wei Tian[4], Guodong Wang[1*]

[1]Key Laboratory of Wetland Ecology and Environment, Heilongjiang Xingkai Lake Wetland Ecosystem National Observation and Research Station, Northeast Institute of Geography and Agroecology, Chinese Academy of Sciences, Changchun 130102, China

[2]School of Natural Sciences, Bangor University, Bangor LL57 2UW, UK.

[3]Joint Key Lab of Changbaishan Wetland and Ecology, Jilin Province, Changchun 130102, China

[4]Institute of Grassland Science, Northeast Normal University, Changchun, China

[*]Corresponding author: Guodong Wang (email: wanggd@iga.ac.cn)

Abstract:

Over half of the Earth's wetlands have been reclaimed for agriculture, leading to significant soil P destabilization and leaching risks. To evaluate the effects of agricultural land use on soil P stability, we used sequential P extraction to investigate the long-term effects of wetland cultivation for rice and soybean on soil P fractions, including labile and moderately labile inorganic/organic P (LPi, LPo, MPi, and MPo), and stable P in Northeast China. The results showed that soybean cultivation decreased the total P by 35.9%, whereas rice cultivation did not influence the total P content ($p < 0.05$). Both the soybean and rice cultivations significantly increased LPi ($p < 0.05$). Soybean cultivation significantly decreased the LPo and MPo compared to rice cultivation, and the latter increased MPi by 309.28% compared with the reference wetlands ($p < 0.05$). Redundancy analysis indicated that pH, poorly crystalline Fe (Feca), crystalline Fe (Fec), and soil organic carbon (TOC) explained similar variations in P fractions during soybean and rice cultivation (54.9% and 49.7%, respectively). Similarly, during soybean or rice cultivation, pH negatively influenced LPo and MPo, while Feca positively influenced MPi and LPi. Furthermore, TOC showed a positive role in LPo, and MPo, but a negative effect on LPi and MPi during rice cultivation. Hence, we concluded that the cultivation of soybean or rice create contrasting modifications to wetland soil P fractionation by altering TOC, Feca, Fec, and pH. Our study indicates that agricultural land use can regulate the fate of wetland soil P fractionation, with potential benefits to both i) P risk management in cultivated wetlands and ii) potential approaches for future wetland restoration.

Keywords: Soybean; rice; Sedge meadows; Poorly crystalline Fe; Crystalline Fe; Soil organic carbon

Introduction:

In recent decades, large-scale cropland expansion to meet rising food demand has severely compromised wetlands (Mogollón et al., 2021), causing approximately half of wetland losses globally (Zedler and Kercher, 2005). This conversion of wetlands not only causes a huge carbon loss (Leifeld and Menichetti, 2018; Säurich et al., 2019) but also leads to the eutrophication of aquatic systems due to phosphorus (P) leaching (Kleinman et al., 2011; Lou et al., 2015; Mogollón et al., 2021). Therefore, wetland restoration and cultivated wetland P leaching risk management have received widespread attention. Even if the effects of large P fertilizers inputs on wetland soil P fractions are disregarded, agricultural hydrological management (e.g. paddy or upland cultivation) can nevertheless alter wetland soil P stability and concentrations by influencing soil redox conditions (Lou et al., 2015; Pant and Reddy, 2001). It is thus essential to fully understand the effects of different agricultural land use practices on P stability if we are to design effective strategies for P management in cultivated wetlands and approaches for their future restoration.

Owing to the low oxidation decomposition rate under waterlogged conditions, only a small portion of P derived from plant and microbial necromass is recycled in wetlands, resulting in organic P sequestration (Menon and Holland, 2014). However, agricultural conversion of wetlands increases soil oxidation and the transformation of organic P to inorganic P form that leaches more easily than organic P (Holliday and Gartner, 2007). The consequent inorganic P losses could, however, be decreased by soil minerals through adsorption or coprecipitation which strongly decreases the exchange between P and soil solution (Miller et al., 2001). In addition, organic P includes nucleic acids, phospholipids, inositol phosphates, and sugar phosphates, have been found to have a strong affinity for sorption to the charged mineral surface and increase organic P stabilization (Berg and Joern, 2006). Thus, a comparison of the balance between P fractions such as labile, mineral-bound inorganic, and organic phosphorus pools, could provide a valuable indicator for the wider stability of P in soil (Sheklabadi et al., 2014; Wright et al., 2010). Evaluating the response of wetland soil P fractions to different agricultural land use regimes could greatly improve our understanding of P stability.

Hydrological management approaches differ between uplands and paddy agricultural systems. Generally, upland cultivation depends on atmospheric precipitation, in contrast, paddy cultivation from wetlands maintains flooded conditions under the rice growing season. Previous studies show a significant increase in total P, Olsen-P, and water-soluble P (Arruda Coelho et al., 2019; Liao et al., 2008; Takahashi and Anwar, 2007). Other studies only found that $NaHCO_3$-Pi and $NaHCO_3$-Po appeared to act as a sink for adding P fertilizer in paddy fields (Lan et al., 2012; LI et al., 2015; Shah et al., 2010). Where wetlands contain abundant organic P, the conversion of wetlands to uplands (e.g., wheat, soybean, and maize) leads to strong oxidative decomposition of organic P (Lou et al., 2015; Wang et al., 2012). In addition, upland cultivation could increase the formation of crystalline Fe oxides, which may decrease mineral-associated Pi (Emsens et al., 2017). In paddy cultivation from the wetlands, flooded conditions may suppress microbial and enzymatic activities and decrease organic P mineralization (Chen et al., 2021). However, the reduced conditions could also promote Fe(III) reduction, releasing Pi into soil solution (Lin et al., 2018) although related studies report that the formation of Fe(II) species creates a high surface area with abundant sorption sites that could adsorb Pi (Holford and Patrick, 1979; Lin et al., 2018). While the effects of agricultural land use on soil P fractions have been widely reported (Negassa and Leinweber, 2009), comparative studies of differing approaches to crop cultivation on wetland soil P fractions have remained limited.

Northeast China has the largest freshwater wetland area in China (1.06×10^6 km^2) (Yu et al., 2018). Large-scale development of that area for agriculture between 1949 and 1996 reduced wetland areas by 70%, with the main agricultural approaches adopted being paddy and upland cultivation (Wang et al., 2019). This area provides an important field platform to test the effects of different crop cultivation on wetland soil P factions, allowing us to distinguish the key factor for P stability. We selected rice and soybean cultivations to investigate the effects of well-drained and waterlogged agricultural land use on P stability, based on the variation in P fractions. We hypothesized that (1) soybean cultivation would severely decrease soil organic P fractions compared with paddy cultivation and (2) reactive Fe oxides would increase during paddy cultivation and increase mineral associated P.

Materials and methods:

2.1 Site description

Northeast China is characterized by a cold temperate continental monsoon climate with a short, warm, wet summer and a long, cold, dry winter. Many wetlands are found in this region. To meet the demand for food, over 70% of wetlands have been reclaimed as croplands across northeast China (Xing et al., 2015). Wetlands reclaimed for soybean cultivation mainly occur in Heilongjiang province, in 1954, the wetland area was approximately 5, 340, 000 ha, and large wetlands were originally reclaimed as soybean cropland. Here, we selected sites in the middle of the Naoli River in Heilongjiang Province (Figure 1), where the average mean temperature is 4.05 °C, and the annual precipitation is 503 mm. Most wetlands were cultivated for soybean during 1980-1995, and the P fertilizer was approximately at 70kg/ha/yr. While wetlands in Jilin

Province had an area of 12,000 ha in 1971, by 2007, more than 9000 ha had been reclaimed as cropland (Xing et al., 2015). Most of the wetlands were distributed in the Changbai Mountains region of Jilin Province, where the annual precipitation is 680 mm, and the average temperature is 2.8 °C; many wetlands were originally reclaimed to paddy during 1960 to 2000 (Zheng et al., 2017), and the average amount of P fertilizer is about 60kg/ha/yr (Shi et al., 2017).

2.2 Soil sampling and analysis

To reduce the heterogeneity of wetland type, space, cultivation age, rotation and type of fertilizer. We adopted the following criteria for selecting sampling sites: First, the cultivated wetlands all belong to sedge meadows. Second, according to a reference or control wetland concept that accounts for the variability of wetland conditions in space and time (Otte et al., 2021), each site must contain both natural sedge meadows and cultivated plots, and these two types must be adjacent to each other (< 500 m). Third, all cultivated fields experienced chemical fertilization and were reclaimed to soybean or rice for 20-30 years without rotation. According to consultation from local farmers, we selected eight study sites in total, four located in the middle of the Naoli River in Heilongjiang Province and four in the Changbai Mountain region in Jilin Province (Figure 1). Samples were taken along a transect at equidistant intervals (>50 m) in each wetland, soybean field or rice field. Three composed replicates (0–10 cm) were sampled with a five-point sampling method at the end of August 2020, for a total of 48 soil samples. All samples were transported to the laboratory in polyethylene bags on ice and subsequently freeze-dried for further processing.

Soil pH values were determined in a 1:5 soil: distilled water suspension using a digital pH meter (PHS-3E, Leici, China) (Bai et al., 2017). The soil organic carbon (TOC) content was determined by potassium dichromate sulfuric acid oxidation followed by titration with ferrous sulfate standard solution, according to the differences in oxidant mass before and after oxidation (Wang et al., 2019). To determine the total soil nitrogen, soil samples (0.4 g) were digested with catalysts (1.8 g, potassium sulfate: copper sulfate pentahydrate: selenium = 100:10:1) and 4 mL sulfuric acid (400 °C, 2–3 hours), and the digests were diluted with distilled water (100 mL) and measured using a continuous flow analyzer (SAN++, Skalar, Netherlands) (Wang et al., 2019). The soil samples were extracted by 0.25 M hydroxylamine hydrochloride in 0.25 M hydrochloric acid in a ratio of 1:30 mass to the solution (shaken for 4 h and centrifuged), which was measured for poorly crystalline Fe (Feca). The remaining soil pellets were further extracted with 10 ml 0.05 M sodium dithionate, shaken for 16 h, centrifuged, and filtered. The residues were washed with 0.05 M HCl for 1 h, centrifuged, filtered, and combined with the dithionite extract to measure crystalline Fe (Fec) (Wagai and Mayer, 2007). The iron in the extractions was measured using ICP–MS (NexION 350D, Perkin Elmer, USA).

Soil P fractions were estimated using a modified Hedley scheme (Lin et al., 2018) and divided into six P pools: soil total P, soil labile inorganic and organic P (LPi and LPo), moderately labile inorganic and organic P (MPi and MPo), and stable P. For total P, soil samples (0.5 g) were digested with nitric and perchloric acids (Carter and Gregorich, 2007). An additional 0.5 g soil sample was sequentially extracted using 0.5 M sodium bicarbonate ($NaCO_3$, pH = 8.5, 30 mL solution) and 0.1 M hydroxide solution (NaOH, 30 mL solution). $NaHCO_3$-extractable P is weakly adsorbed on soil particles, easily utilized by microorganisms and plants, and, therefore, defined as labile P. NaOH-extractable P is strongly bound to Fe and Al minerals, thought to represent a relatively more stable P fraction than $NaHCO_3$-extractable P, and defined as moderately labile P. The total P in both extracts ($NaHCO_3$-P_t and NaOH-P_t) was measured after digestion with nitric and perchloric acids. The inorganic P was measured in $NaHCO_3$ (LPi) and NaOH (MPi) extractions, and the organic P in $NaHCO_3$ (LPo) and NaOH (MPo) was estimated by subtracting the inorganic P (LPi and MPi) from the total P ($NaHCO_3$-P_t and NaOH-P_t). Stable P fractions refer to the primary mineral P, occluded inorganic P covered with sesquioxides, etc., which is calculated by subtracting the $NaHCO_3$- and NaOH-extractable Pt from the soil total P. The P fraction concentrations were measured using a continuous flow analyzer (SAN++, Skalar, Netherlands).

2.3 Statistical analysis

To assess the effects of wetland reclamation on the soil properties and quantity in the various P fractions, linear mixed effect models were employed to test the agricultural management approaches on soil properties and P fractions using the “lmer” function in the package “lme4” (Bates et al., 2015) in R ver. 4.1.2 (R Core Team, 2021). In the models, “agricultural management approaches” was included as a fixed effect (two levels: “reference” and “soybean or rice cultivation”) and “sites” was included as a random effect. Furthermore, to determine if the means between the two groups (“reference” and “soybean or rice cultivation”) were significantly different, the chi-squared test was used. The linear mixed model could reduce the site heterogeneity, and show the average estimated value for the variables in the reference wetlands (intercept) and the effects of crop cultivation on soil variables and P fractions in comparison to the reference wetlands (coefficient) across the study sites. Redundancy analysis (RDA) was used to determine the relationships between the various P fractions and the environmental soil variables. Owing to the high collinearity of TOC and TN, we selected TOC, pH, poorly crystalline, and crystalline Fe as environmental variables by using the "vegan" package in R. A regression model was used to test the environmental variables on different P fractions by using the "relaimpo" package in R.

3 Results

3.1 Changes in soil variables and P fractions during the different crop cultivation

Taking natural sedge meadows as a reference, we found that soybean cultivation significantly decreased the TOC and TN contents at four study sites (Tables 1&2); overall, the loss was stronger than that in paddy cultivation (Tables 1&2). Interestingly, poorly crystalline (Feca) and crystalline Fe (Fec) showed contrasting trends during soybean cultivation (Table 2); both the Feca and Fec contents increased during rice cultivation (Table 2).

Compared with reference wetlands, the soil total P content significantly decreased in the four study sites after soybean cultivation (Figure 2 & Table 3). MPo exhibited the maximum loss, followed by LPo and StableP (Figure 2 & Table 3). Overall, LPi significantly increased during soybean cultivation (Table 3). However, the total P and stable P did not change during rice cultivation across the four sites (Figure 3; Table 3). Both MPi and LPi significantly increased at the four sites and LPo and MPo significantly decreased across the sites (Figure 3; Table 3).

3.2 The relationship between soil variables or P fractions

We found that the soil pH was negatively correlated with Feca and TN during soybean cultivation ($p < 0.01$, Figure 4a). Feca was positively correlated with Fec during rice cultivation ($p < 0.01$; Figure 4b). In both the rice and soybean cultivations, TN and TOC were negatively correlated with Fec ($p < 0.01$, Figure 4a&b), whereas TOC and TN exhibited a significantly positive relationship with Fec ($p < 0.001$, Figure 4b).

The P fraction correlation analysis showed that there were similar trends between soybean and rice cultivations, and there was a positive relationship between MPi and LPi, as well as MPo and LPo ($p < 0.01$, Figure 5). LPo, MPi, MPo and stable P had a positive relationship with TP ($p < 0.05$, Figure 5).

3.3 The key soil variables on wetland soil P fractions

Redundancy analysis was used to investigate the factors influencing soil P fraction variations during soybean and rice cultivation. The results showed that soil Feca, Fec, TOC, and pH explained 54.9% of the P fraction variations during soybean cultivation (Figure 6a); these factors explained 49.7% of the P fraction variation during rice cultivation (Figure 6b).

Specifically, the regression model was used to test the effects of each variable on P factions; during soybean cultivation, we found that Feca positively influenced MPi and LPi, and explained 77.67% and 53.99% of the variation, respectively; pH and Fec negatively influenced LPo and MPo, and explained 69.38% and 73.95% of the variation, respectively; TOC showed positive effects on StableP, and explained 38.86% of the variation (Figure 6c). During rice cultivation, TOC and Feca influenced MPi and LPi and explained 62.31% and 68.36% of the variation, respectively. TOC had strong negative effects, but Feca had positive effects. TOC and pH influenced LPo and MPo and explained 68.10% and 45.39% of the variation, respectively; interestingly, both TOC showed strong positive effects. Fec had a positive effect on StableP and explained 48.21% of the variation (Figure 6d).

4 Discussion

4.1 The effects of soybean and rice cultivation on P fractions

P fertilizer is often applied in agricultural ecosystems to maintain crop yields, and thus plays a dominant role in soil inorganic P (Pi) availability (Mahmood et al., 2020). For example, previous studies showed that soil P availability slightly increases with low P application in the short-term (<42 kg P/ha/yr), but substantially increased with high P (<95 kg P/ha/yr) (Oberson et al., 1993). In addition, long-term P fertilization (20 kg P/ha/yr) also increased soil Pi availability (Zheng et al., 2004). In fact, farmers often overuse P fertilizer to increase crop yields, from 1980 to 2007, the average amount of Pi increased from 7.4 to 24.7 mg/kg in Chinese agriculture (Li et al., 2011). Similarly, our results showed that both long-term rice and soybean cultivation increased the soil labile Pi at most sites (Table 3). In addition to the effects of P fertilizer on Pi, hydrological variation due to agricultural management changes the reductive conditions of wetlands to relatively oxidative conditions, which accelerate soil organic P mineralization (Sheklabadi et al., 2014). Furthermore, we also found that soybean cultivation showed stronger negative effects on labile organic P (Po) loss than rice cultivation (Table 3), implying that soybean cultivation would potentially increase soil Pi fractions by increasing organic mineralization.

Long-term cultivation could also alter moderate P fractions (Li et al., 2015; Mahmood et al., 2020). In our study, LPi showed a positive correlation with MPi during soybean or rice cultivation (Figure 5), but paddy cultivation significantly increased the MPi content, and soybean cultivation did not change the wetland soil MPi (Table 3). This is mainly due to the variation of MPi not only depends on the quantity of inorganic P input but also influences by soil minerals and surface runoff (Grenon et al., 2021; Lin et al., 2018). Furthermore, we found that paddy cultivation showed weaker effects on MPo than soybean cultivation (Figure 2&3;

Table 3), which supported our first hypothesis that reduced conditions inhibit organic P mineralization (Sheklabadi et al., 2014). A previous study showed that organic P also exhibits strong adsorption to mineral surfaces (Berg and Joern, 2006), but organic P stability was also determined by redox conditions, soil pH, and the recalcitrance of organic P (Spohn, 2020a).

Overall, our results showed that soybean cultivation significantly decreased wetland soil total P (Figure 2;Table 3), and as P cycling differs from N cycling in that it lacks atmospheric recharge (Zhang et al., 2003), the major pathway of P loss is through leaching into surface flow. We found a substantial loss of stable P from soybean-converted sites, but only weak changed under rice cultivation (Figures 2&3; Table 3), and as stable P is derived from primary or secondary mineral rock P that is insoluble and thus a stable P pool, this finding indicates that P leaching was greater following soybean cultivation than rice cultivation. While, we found that paddy cultivation did not influence soil total P (Figure 3; Table 3), we found noteworthy abundant mineral associated Pi formation and low organic P composition rate all contributed to total P sequestration.

4.2 Factors determining P fractionation during different crop cultivation

Soil properties play an important role in soil P fractions (Spohn, 2020a). Our results showed that soil pH, Feca, Fec, and TOC explained similar variations in P fractions during soybean and rice cultivation (54.9% and 49.7%, respectively), and the key factors determining P fractions were different (Figure 6a&b). Soil pH had a strong negative effect on LPo and MPo, while Feca showed positive effects on MPi and LPi (Figure 6c). The massive soil organic carbon loss and low redox conditions during soybean cultivation potentially raise soil pH during soybean cultivation (Sahrawat, 2005), as the mineralization of organic anions in croplands can increase proton consumption (Fujii et al., 2009). It is well known that low pH (<5.0) inhibits microbial activity and the extracellular activities of phosphatase enzymes (Eivazi and Tabatabai, 1977; Turner and Blackwell, 2013), hence it may also promote soil organic P decomposition when pH increases from 6 to 7 during soybean cultivation. Although Feca showed strong positive effects on inorganic P retention due to iron oxides having strong adsorption with Pi, soybean cultivation decreased Feca by 24.27% (Table 2), showing that the low Feca directly reduce the protection for Pi.

It was consistent with our second hypothesis that rice cultivation could increase the protection of Feca for inorganic P. Feca increased by 30.37% during paddy cultivation (Table 1&2). Although the previous study showed that reductive conditions would increase Fe reduction coupled release of mineral associated Pi (Emsens et al., 2017), ferrous Fe has a high basal area which also has strong adsorption for Pi (Lin et al., 2018), which also contributes to the accumulation of mineral associated Pi. In addition, we further found that TOC was the most important factor that determines P fractions, with positive effects on MPo and LPo, and negative effects on MPi and LPi. Our study showed that rice cultivation promoted less carbon loss than soybean cultivation (Tables 1 and 2). It should also be noted that soil organic carbon and the size of microbial biomass are positively correlated (Banu et al., 2004), and it is feasible that further Pi input would ultimately contribute to high soil microbial biomass P sequestration (Thanh Nguyen and Marschner, 2005). After microbes die, organic P in microbial necromass can remain persistent for decades (Spohn, 2020b), as the reductive conditions limit organic P decomposition, promoting organic P stability during rice cultivation.

4.3 Implications for cultivated wetlands P management and recovery

Our studies emphasized the importance of organic carbon and Feca on soil organic P and Pi stability during wetland cultivation. In paddy fields, frequent redox conditions could promote the formation of reactive Fe, and abundant carbon also promotes the association of Fe and carbon (Lalonde et al., 2012; Riedel et al., 2013). Hence, the formation of Fe-C complexes not only also act as barriers to reduce Fe leaching but also show a strong adsorption on Pi (Yang et al., 2022). Despite surface runoff, we found that paddy management could maintain P stability. However, long-term cultivation would also accelerate organic carbon decomposition, decreasing P stability and increasing P leaching risk. Hence, restoring agricultural soil organic carbon could not only increase crop yields but also decrease the soil P leaching risk.

Hydrological variability including the timing, magnitude, frequency, and duration of inundation is a key determinant of wetland diversity and functions (Moreno-Mateos et al., 2012; Zedler, 2000). Many studies also report that soil nutrients influence seed germination and wetland plant distribution (Ardón et al., 2010; Duff et al., 2009). In our study, soybean cultivation significantly decreased soil Pi but paddy cultivation increased Pi, indicating that different crop cultivation approaches may have different effects on soil P pool. Furthermore, the legacy of soil P may differently influence wetland seed germination and plant communities (Emsens et al., 2017). Hence, we suggest that crop cultivation history and its effects on soil nutrient legacy should be considered in the development of approaches for wetland recovery.

4.4 Conclusions

Our study recognized differing effects of rice and soybean cultivation on soil P fractions in reclaimed wetlands. Soybean cultivation severely decreased soil total P, while rice cultivation did not influence total P. Both management approaches increased LPi and decreased LPo and

MPo, but rice cultivation also increased MPi content, which compensated for the loss of organic P. In addition to the common effects of pH (negative) and Feca (positive) on soil P fractions, we further found that TOC had strong positive effects on LPo and MPo during paddy cultivation. Finally, we propose that organic carbon restoration could increase soil P stability and reduce P leaching risk, while acknowledging that different crop cultivations create a P legacy, which should be considered when devising wetland recovery approaches.

Acknowledgments

This work was supported by the National Natural Science Foundation of China (42101114; 42230516; 41971136), the Youth Innovation Promotion Association CAS (2019234; 2020235), and the Professional Association of the Alliance of International Science Organizations (ANSO-PA-2020-14). We thanked two anonymous reviewers for their valuable suggestions.

References:

Ardón, M., Morse, J.L., Doyle, M.W. and Bernhardt, E.S., 2010. The water quality consequences of restoring wetland hydrology to a large agricultural watershed in the Southeastern Coastal Plain. Ecosystems, 13, 1060-1078.

Arruda Coelho, M.J., Ruiz Diaz, D., Hettiarachchi, G.M., Dubou Hansel, F. and Pavinato, P.S., 2019. Soil phosphorus fractions and legacy in a corn-soybean rotation on Mollisols in Kansas, USA. Geoderma Regional, 18, e00228.

Bai, J. et al., 2017. Phosphorus sorption-desorption and effects of temperature, pH and salinity on phosphorus sorption in marsh soils from coastal wetlands with different flooding conditions. Chemosphere, 188, 677-688.

Banu, N.A., Singh, B. and Copeland, L., 2004. Microbial biomass and microbial biodiversity in some soils from New South Wales, Australia. Soil Research, 42, 777.

Bates, D., Macgker, M., Bolker, B., Walker, S., 2015. Fitting linear mixed- effects models using lme4. Journal of Statistical Software 67, 1-48.

Berg, A.S. and Joern, B.C., 2006. Sorption Dynamics of Organic and Inorganic Phosphorus Compounds in Soil. Journal of Environmental Quality, 35, 1855-1862.

Carter, M.R., Gregorich, E.G., 2007. Soil sampling and methods of analysis. Canadian Society of Soil Science. Ann Arbor, USA: Lewis Publishers.

Chen, X. et al., 2021. Contrasting pathways of carbon sequestration in paddy and upland soils. Global Change Biology, 27, 2478-2490.

Duff, J.H. et al., 2009. Phosphorus and nitrogen legacy in a restoration wetland, upper Klamath Lake, Oregon. Wetlands, 29,735-746.

Eivazi, F. and Tabatabai, M.A., 1977. Phosphatases in soils. Soil Biology and Biochemistry, 9, 167-172.

Emsens, W.J., Aggenbach, C.J.S., Smolders, A.J.P., Zak, D. and van Diggelen, R., 2017. Restoration of endangered fen communities: the ambiguity of iron-phosphorus binding and phosphorus limitation. Journal of Applied Ecology, 54, 1755-1764.

Fujii, K., Funakawa, S., Hayakawa, C., Sukartiningsih and Kosaki, T., 2009. Quantification of proton budgets in soils of cropland and adjacent forest in Thailand and Indonesia. Plant and soil, 316, 241-255.

Grenon, G. et al., 2021. Phosphorus fate, transport and management on subsurface drained agricultural organic soils: a review. Environmental research letters, 16, 013004.

Holford, I.C.R. and Patrick, W.H., 1979. Effects of reduction and pH changes on phosphate sorption and mobility in an acid soil. Soil Science Society of America Journal, 43, 292-297.

Holliday, V.T. and Gartner, W.G., 2007. Methods of soil P analysis in archaeology. Journal of Archaeological Science, 34, 301-333.

Kleinman, P.J.A. et al., 2011. Managing agricultural phosphorus for water quality protection: principles for progress. Plant and Soil, 349, 169-182.

Lalonde, K., Mucci, A., Ouellet, A. and Gélinas, Y., 2012. Preservation of organic matter in sediments promoted by iron. Nature, 483, 198-200.

Lan, Z.M., Lin, X.J., Wang, F., Zhang, H. and Chen, C.R., 2012. Phosphorus availability and rice grain yield in a paddy soil in response to long-term fertilization. Biology and Fertility of Soils, 48, 579-588.

Leifeld, J. and Menichetti, L., 2018. The underappreciated potential of peatlands in global climate change mitigation strategies. Nature Communications, 9, 1071.

Li, H. et al., 2011. Integrated soil and plant phosphorus management for crop and environment in China. A review. Plant and Soil, 349, 157-167.

LI, Y. et al., 2015. Effects of long-term phosphorus fertilization and straw incorporation on phosphorus fractions in subtropical paddy soil. Journal of Integrative Agriculture, 14, 365-373.

Liao, W., Liu, J., Wang, X., Jia, K. and Meng, N., 2008. Effects of phosphate fertilizer and manure on Chinese cabbage yield and soil phosphorus accumulation. Frontiers of Agriculture in China, 2, 301-306.

Lin, Y. et al., 2018. Phosphorus fractionation responds to dynamic redox conditions in a humid tropical forest soil. Journal of Geophysical Research: Biogeosciences, 123, 3016-3027.

Lou, H. et al., 2015. Phosphorus risk in an intensive agricultural area in a mid-high latitude region of China. Catena, 127, 46-55.

Mahmood, M. et al., 2020. Changes in phosphorus fractions and its availability status in relation to long term P dertilization in Loess Plateau of China. Agronomy, 10, 1818.

Menon, R. and Holland, M.M., 2014. Phosphorus release due to decomposition of wetland plants. Wetlands, 34, 1191-1196.

Miller, A.J., Schuur, E.A.G. and Chadwick, O.A., 2001. Redox control of phosphorus pools in Hawaiian montane forest soils. Geoderma, 102, 219-237.

Mogollón, J.M. et al., 2021. More efficient phosphorus use can avoid cropland expansion. Nature Food, 2, 509-518.

Moreno-Mateos, D., Power, M.E., Comín, F.A. and Yockteng, R., 2012. Structural and functional loss in restored wetland ecosystems. PLoS Biology, 10, e1001247.

Negassa, W. and Leinweber, P., 2009. How does the Hedley sequential phosphorus fractionation reflect impacts of land use and management on soil phosphorus: A review. Journal of Plant Nutrition and Soil Science, 172, 305-325.

Oberson, A., Fardeau, J.C., Besson, J.M. and Sticher, H., 1993. Soil phosphorus dynamics in cropping systems managed according to conventional and biological agricultural methods. Biology and Fertility of Soils, 16, 111-117.

Otte, M.L., Fang, W. and Jiang, M., 2021. A framework for identifying reference wetland conditions in highly altered landscapes. Wetlands, 41, 1-12.

Pant, H.K. and Reddy, K.R., 2001. Hydrologic influence on stability of organic phosphorus in wetland detritus. Journal of Environmental Quality, 30, 668-674.

Riedel, T., Zak, D., Biester, H. and Dittmar, T., 2013. Iron traps terrestrially derived dissolved organic matter at redox interfaces. Proceedings of the National Academy of Sciences, 110, 10101-10105.

R Core Team. 2021. R: A language and environment for statistical computing. R Foundation for Statistical Computing.

Sahrawat, K.L., 2005. Fertility and organic matter in submerged rice soils. Current Science, 88, 735-739.

Säurich, A. et al., 2019. Drained organic soils under agriculture-The more degraded the soil the higher the specific basal respiration. Geoderma, 355, 113911.

Shah, A.L., Biswas, J.C., Solaiman, A. and Panaullah, G.M., 2010. Phosphorus fertilization in rice (*Oryza sativa* L) cultivation changes Soil P-Fractions. The Agriculturists, 5, 20-29.

Sheklabadi, M., Mahmoudzadeh, H., Mahboubi, A.A., Gharabaghi, B. and Ahrens, B., 2014. Land use effects on phosphorus sequestration in soil aggregates in western Iran. Environmental Monitoring and Assessment, 186, 6493-6503.

Shi, Y. et al., 2017. Using 13C isotopes to explore denitrification-dependent anaerobic methane oxidation in a paddy-peatland. Scientific Reports, 7, 1-7.

Spohn, M., 2020a. Increasing the organic carbon stocks in mineral soils sequesters large amounts of phosphorus. Global Change Biology, 26, 4169-4177.

Spohn, M., 2020b. Phosphorus and carbon in soil particle size fractions: A synthesis. Biogeochemistry, 147, 225-242.

Takahashi, S. and Anwar, M.R., 2007. Wheat grain yield, phosphorus uptake and soil phosphorus fraction after 23 years of annual fertilizer application to an Andosol. Field Crops Research, 101, 160-171.

Thanh Nguyen, B. and Marschner, P., 2005. Effect of drying and rewetting on phosphorus transformations in red brown soils with different soil organic matter content. Soil Biology and Biochemistry, 37, 1573-1576.

Turner, B.L. and Blackwell, M.S.A., 2013. Isolating the influence of pH on the amounts and forms of soil organic phosphorus. European Journal of Soil Science, 64, 249-259.

Wagai, R. and Mayer, L.M., 2007. Sorptive stabilization of organic matter in soils by hydrous iron oxides. Geochimica et Cosmochimica Acta, 71, 25-35.

Wang, G. et al., 2019. Does the element composition of soils of restored wetlands resemble natural wetlands? Geoderma, 351, 174-179.

Wang, Y., Liu, J., Wang, J. and Sun, C., 2012. Effects of wetland reclamation on soil nutrient losses and reserves in Sanjiang Plain, Northeast China. Journal of Integrative Agriculture, 11, 512-520.

Wright, D.M. et al., 2010. Do leaves of plants on phosphorus-impoverished soils contain high concentrations of phenolic defence compounds? Functional Ecology, 24, 52-61.

Xing, Y., Zhang, H.W., Jiang, Q.G., Li, W.Q., 2015. Properties of wetland succession in three northeast provinces based on RS. Wetland Science & Management, 11, 50-54 (In Chinese).

Yang, W. et al., 2022. Phosphorus sorption capacity of various iron-organic matter associations in peat soils. Environmental Science and Pollution Research, 29, 77580-77592.

Yu, X. et al., 2018. Review of rapid transformation of floodplain wetlands in Northeast China: roles of human development and global environmental change. Chinese Geographical Science, 28, 654-664.

Zedler, J.B., 2000. Progress in wetland restoration ecology. Trends in Ecological and Evolution, 15, 402-407.

Zedler, J.B. and Kercher, S., 2005. Wetland resources: status, trends, ecosystem services, and restorability. Annual Review of Environment and Resources, 30, 39-74.

Zhang, G. et al., 2003. Pedogenic evolution of paddy soils in different soil landscapes. Geoderma, 115, 15-29.

Zheng, X.J. et al., 2017. Landscape dynamics and driving forces of wetlands in the Tumen River Basin of China over the past 50 years. Landscape and Ecological Engineering, 13, 237-250.

Zheng, Z., MacLeod, J.A., Sanderson, J.B. and Lafond, J., 2004. Soil phosphorus dynamics after ten annual applications of mineral fertilizers and liquid dairy manure: fractionation and path analyses. Soil Science, 169, 449-456.

Table Captions

Table 1 The changes in soil variables between cultivated (C) (s = soybeans, and r = rice) and reference (R) wetlands. Rice cultivation: Cr; the reference wetland for rice cultivation: Rr; soybean cultivation: Cs; the reference wetland for soybean cultivation: Rs. TOC: total organic carbon; TN: total nitrogen; Feca, poorly crystalline Fe; Fec, crystalline Fe.

Table 2 Linear mixed model to test the effects of cultivation on soil properties. The research site was a random factor; the intercept is the baseline mean, which is also the average estimating value for the variables in the reference wetlands; the fixed model estimates the value of the changes during cultivation in comparison to the reference wetlands (coefficient). Rm^2, the variance explained by fixed effects; Rc^2, the variance explained by both fixed and random effects; TOC: total organic carbon; TN: total nitrogen; Feca, poorly crystalline Fe; Fec, crystalline Fe; values in parentheses are standard errors; significant P values, ***<0.001; **<0.01; *<0.05.

Table 3 Linear mixed model to test the effects of cultivation on P fractions; the research site was a random factor; the intercept is the baseline mean, which is also the average estimating value for the variables in the reference wetlands; the fixed model estimates the value of the changes during cultivation in comparison to the reference wetlands (coefficient). Rm^2, the variance explained by fixed effects; Rc^2, the variance explained by both fixed and random effects. TP: total phosphorus; StableP: residual phosphorus; MPo, moderate labile organic phosphorus; MPi, moderate labile inorganic phosphorus; LPo, labile organic phosphorus; LPi, labile inorganic phosphorus; the values in parentheses are standard errors; significant P values, ***<0.001; **<0.01; *<0.05.

Figure Captions

Figure 1 Location of sampling sites in Northeast China. The brown color represents the soybean cultivation sites in Heilongjiang Province, including Shengli (SL), Hongwei (HW), Daxing (DX), and Baoping (BP). The green color represents the rice cultivation sites in Jilin province, including Yushugou (YSJ), Hunchun (HC), Jichuan (JC), and Sipeng (SP). Each site contains both natural sedge meadows and cultivated plots, where these two types were present adjacent to each other (< 500 m).

Figure 2 Changes in soil phosphorus fractions following soybean cultivation. (a) Hongwei (HW); (b) Shengli (SL); (c) Baoping (BP); (d) Daxing (DX). The yellow bars indicate soybean cultivation in wetlands and the blue bars indicate the reference wetlands. The asterisk indicates a significant difference between the soybean and rice cultivation and their reference wetlands. TP: total phosphorus; StableP: residual phosphorus; MPo, moderate labile organic phosphorus; MPi, moderate labile inorganic phosphorus; LPo, labile organic phosphorus; LPi, labile inorganic phosphorus.

Figure 3 Changes in soil phosphorus fractions following rice cultivation. (a) Hongwei (HW); (b) Shengli (SL); (c) Baoping (BP); (d) Daxing (DX). The yellow bars indicate rice cultivation in wetlands and the blue bars indicate the reference wetlands. The asterisk indicates a significant difference between the soybean and rice cultivations and their reference wetlands. TP: total phosphorus; StableP: residual phosphorus; MPo, moderate labile organic phosphorus; MPi, moderate labile inorganic phosphorus; LPo, labile organic phosphorus; LPi, labile inorganic phosphorus.

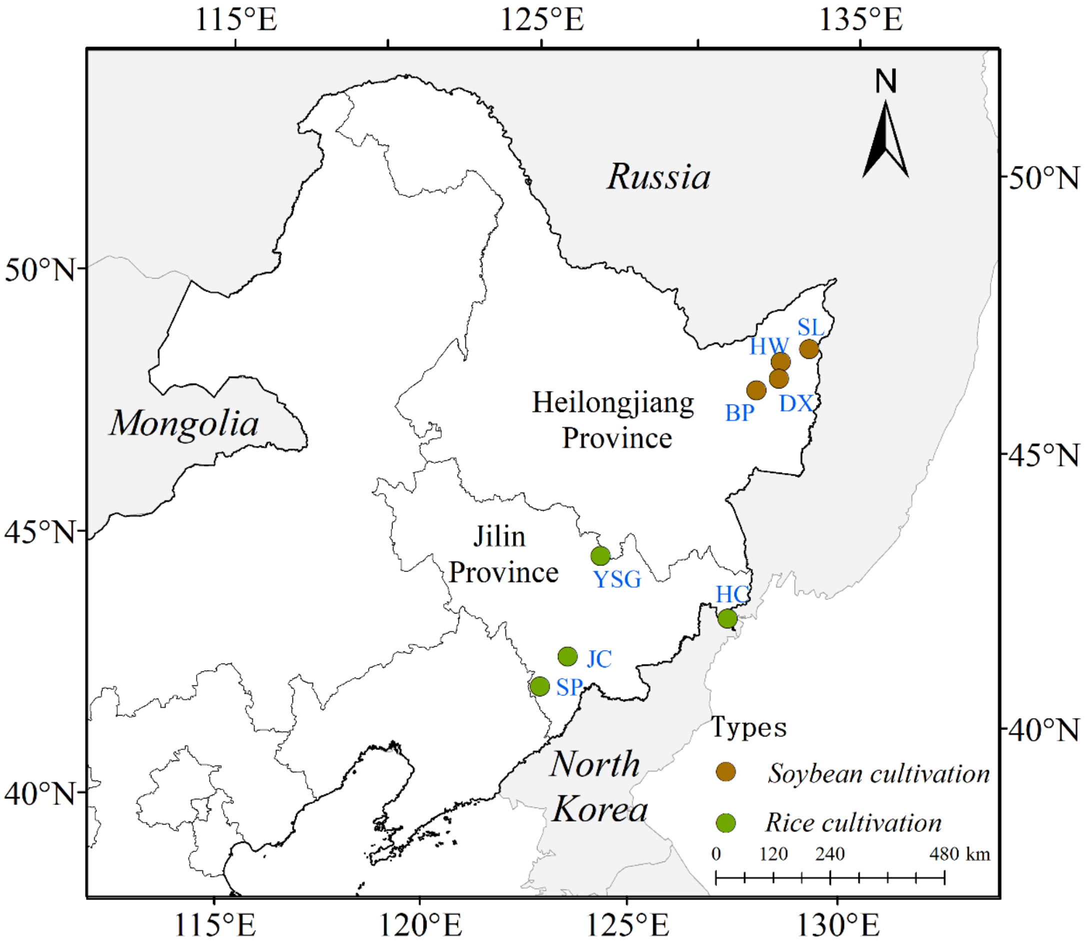
Figure 4 Correlations among soil properties; (a) soybean cultivation; (b) rice cultivation; significant P values, ***<0.001; **<0.01; *<0.05. TOC: total organic carbon; TN: total nitrogen; Feca, poorly crystalline Fe; Fec, crystalline Fe; both circle size and value on bottom line represent Spearman correlations, and red and blue color represent negative and positive correlations, respectively.

Figure 5 Correlations among soil P fractions; (a) soybean cultivation; (b) rice cultivation; significant

P values, $^{***}<0.001$; $^{**}<0.01$; $^{*}<0.05$. StableP: residual phosphorus; MPo, moderate labile organic phosphorus; MPi, moderate labile inorganic phosphorus; LPo, labile organic phosphorus; LPi, labile inorganic phosphorus. Both circle size and value on bottom line represent Spearman correlations, and red and blue color represent negative and positive correlations, respectively.

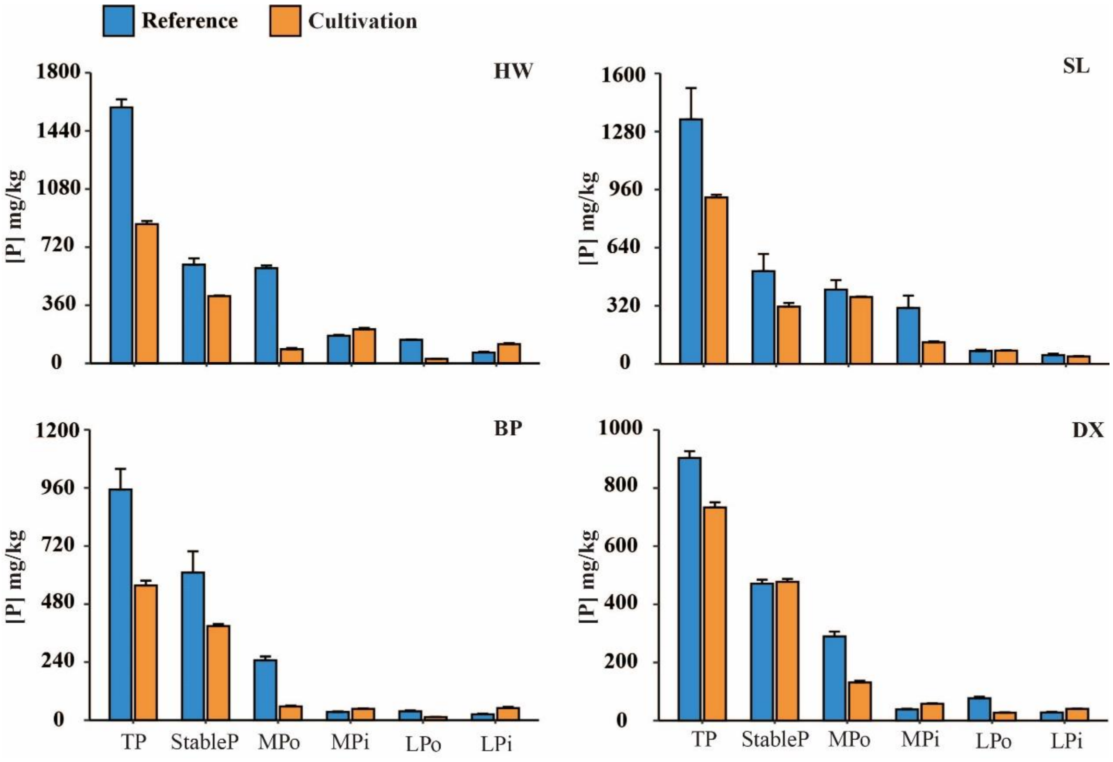
Figure 6 The control for soil P fractions; (a) redundancy analysis assessing the relationship between P fractions and soil factors, (a) soybean cultivation (b) rice cultivation; the contribution of soil variables to each P fraction based on correlation and best multiple regression model, (c) soybean cultivation; (d) rice cultivation. RDA1 and RDA2 represent the proportion of explained variability for P fractions during crop cultivation. Circle size represents the variable importance (that is, proportion of explained variability calculated via multiple regression modeling and variance decomposition analysis). Colors represent Spearman correlations. TOC: total organic carbon; Feca, poorly crystalline Fe; Fec, crystalline Fe; StableP: residual phosphorus; MPo, moderate labile organic phosphorus; MPi, moderate labile inorganic phosphorus; LPo, labile organic phosphorus; LPi, labile inorganic phosphorus.

Figure 1



115°E
125°E
135°E
50°N
45°N
40°N
50°N
45°N
40°N
115°E
120°E
125°E
130°E
N
Russia
Mongolia
Heilongjiang Province
Jilin Province
North Korea
SL
HW
DX
BP
YSG
HC
JC
SP
Types
Soybean cultivation
Rice cultivation
0
120
240
480 km

531 Figure 2



532

533

Figure 3

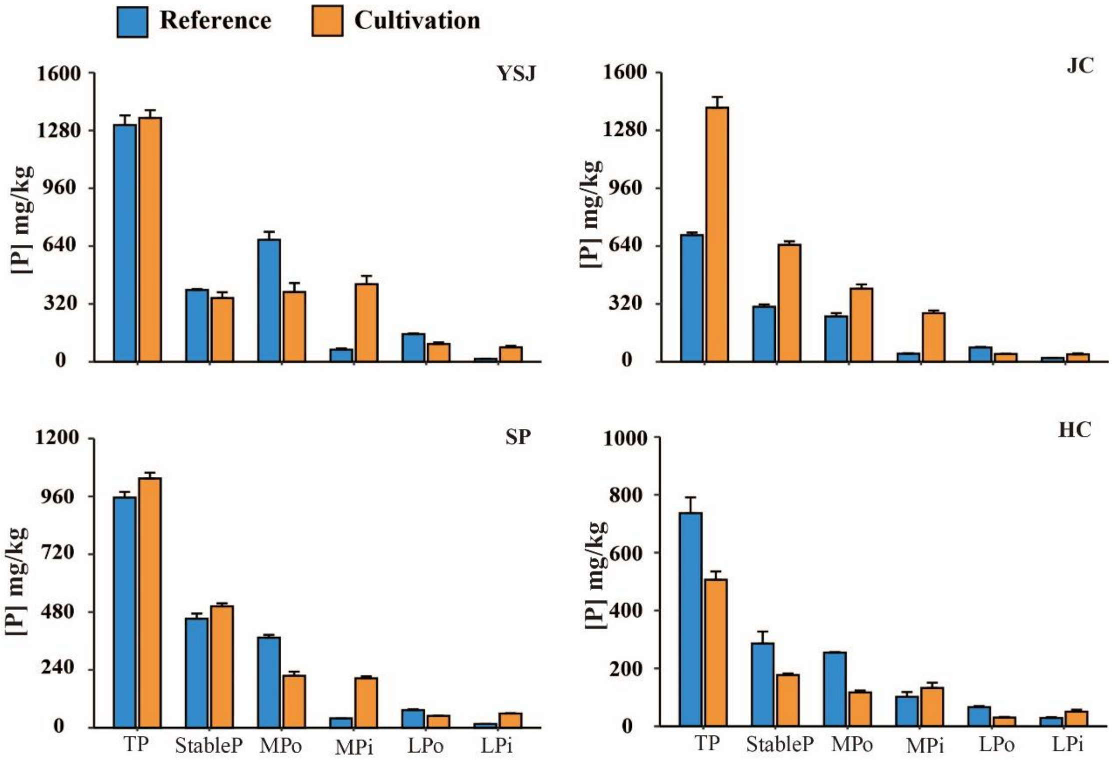

Reference
Cultivation
YSJ
JC
SP
HC
[P] mg/kg
1600
1280
960
640
320
0
1200
960
720
480
240
0
1000
800
600
400
200
0
TP
StableP
MPo
MPi
LPo
LPi

Figure 4

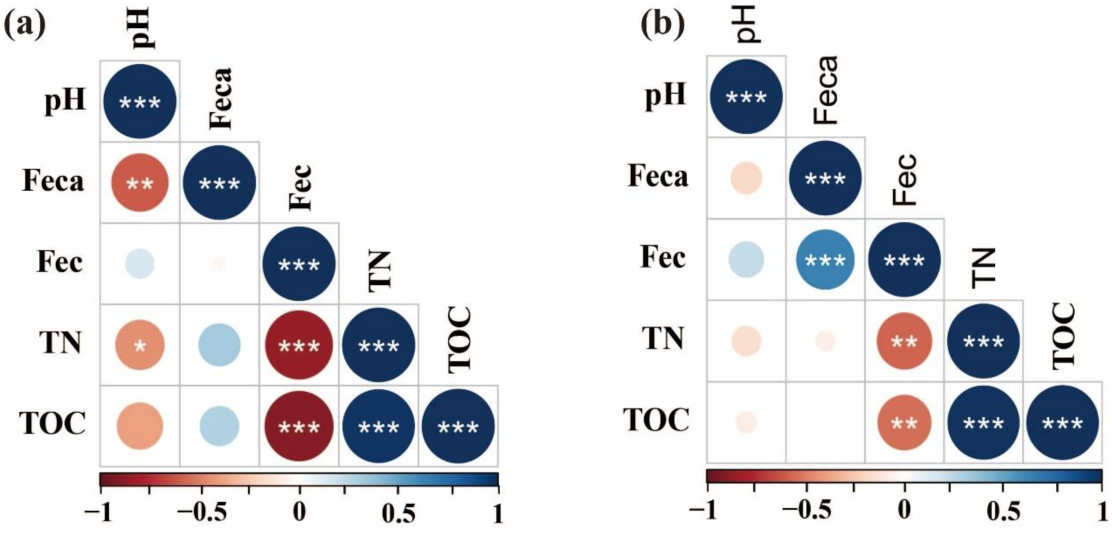


Figure 5

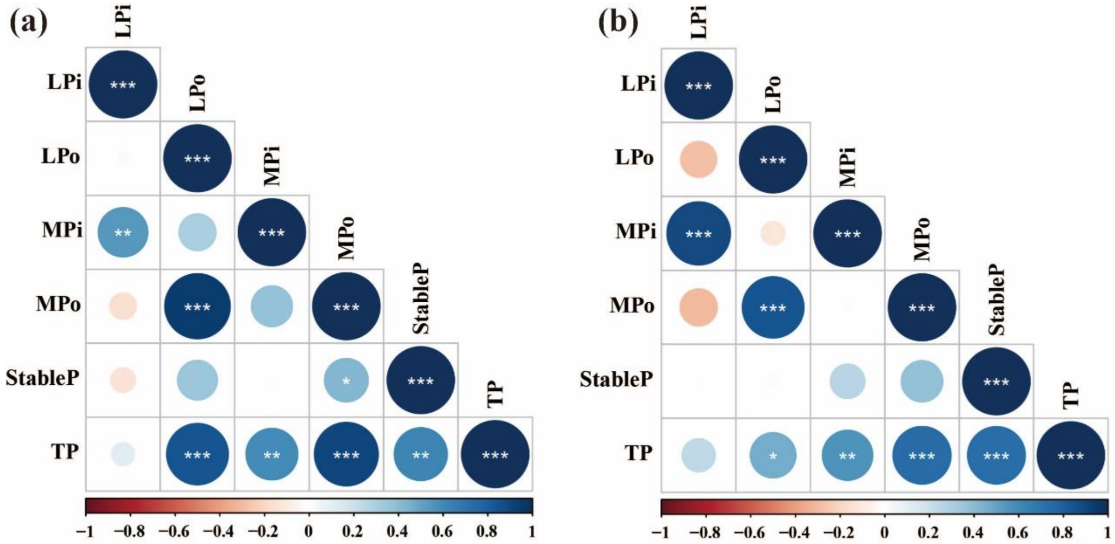
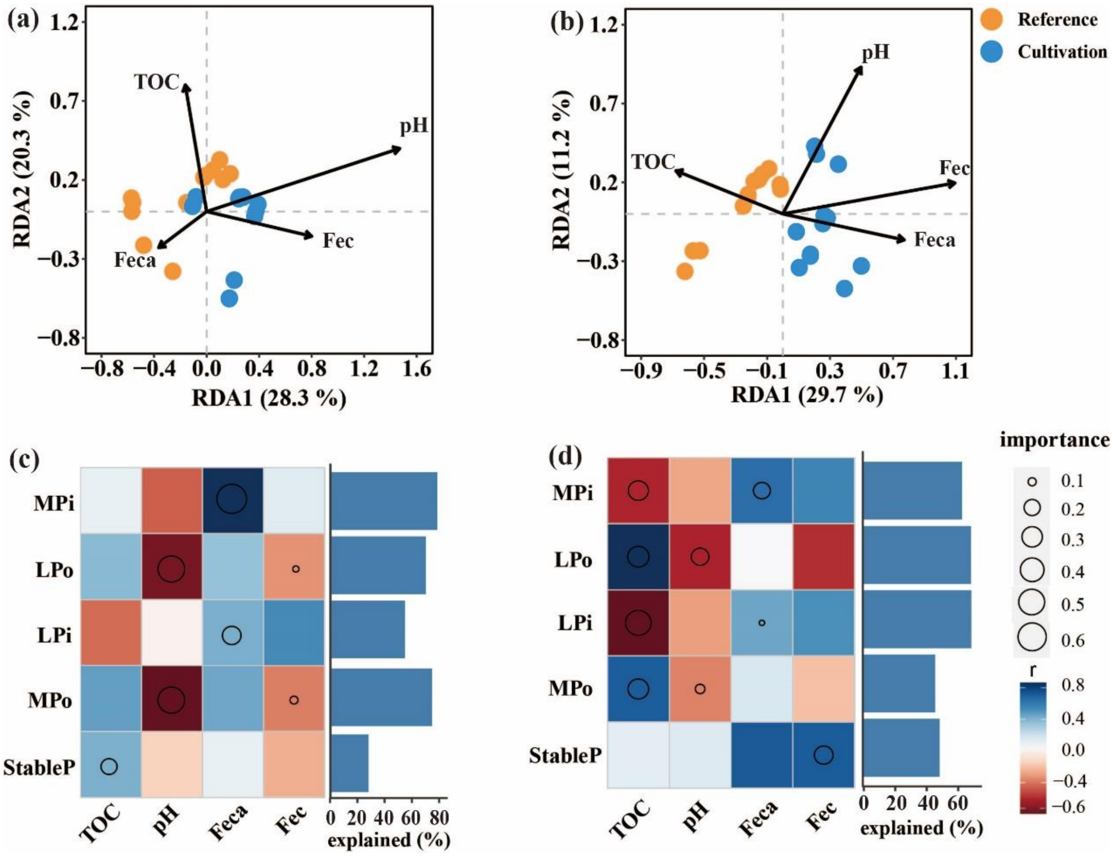

(a)
LPi
LPo
MPi
MPo
StableP
TP
***
***
** ***
*** ***
* ***
*** ** *** ** ***
−1 −0.8 −0.6 −0.4 −0.2 0 0.2 0.4 0.6 0.8 1
(b)
LPi
LPo
MPi
MPo
StableP
TP
***
***
*** ***
*** ***
***
* ** *** *** ***
−1 −0.8 −0.6 −0.4 −0.2 0 0.2 0.4 0.6 0.8 1

Figure 6


(a)
TOC
pH
Fec
Feca
RDA1 (28.3 %)
RDA2 (20.3 %)
1.2
0.7
0.2
−0.3
−0.8
−0.8 −0.4 0.0 0.4 0.8 1.2 1.6
(b)
pH
TOC
Fec
Feca
RDA1 (29.7 %)
RDA2 (11.2 %)
1.2
0.7
0.2
−0.3
−0.8
−0.9 −0.5 −0.1 0.3 0.7 1.1
Reference
Cultivation
(c)
MPi
LPo
LPi
MPo
StableP
TOC
pH
Feca
Fec
0 20 40 60 80
explained (%)
(d)
MPi
LPo
LPi
MPo
StableP
TOC
pH
Feca
Fec
0 20 40 60
explained (%)
importance
0.1
0.2
0.3
0.4
0.5
0.6
r
0.8
0.4
0.0
−0.4
−0.6