Xie *et al. Chem. Biol. Technol. Agric.* (2023) 10:59
https://doi.org/10.1186/s40538-023-00420-9

Chemical and Biological Technologies in Agriculture

RESEARCH

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Phosphorus leaching risk from black soil increased due to conversion of arid agricultural land to paddy land in northeast China

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Abstract

Background Land-use change from arid agricultural land to paddy land may increase soil phosphorus (P) leaching in the black soil region. However, little information is available for soil P leaching risk assessment from soil profiles due to the land-use conversion in the black soil region of northeast China.

Results This study explored the effect of land-use change from arid agricultural land to paddy land on soil P leaching change point, P leaching risk and P fractions. Conversion from arid agricultural land to paddy land decrease soil P leaching change point (0–20 cm: 59.63 mg kg^{-1} vs. 35.35 mg kg^{-1}; 20–40 cm: 24.31 mg kg^{-1} vs. 17.20 mg kg^{-1}; 40–60 cm: 32.91 mg kg^{-1} vs. 10.45 mg kg^{-1}); 30.9% of arid agricultural soils were at risk of P leaching into the shallow groundwater, compared to 87.5% of paddy soils, implying a high risk of P leaching after land-use conversion. P fraction analysis using the Hedley sequential extraction method showed that moderately active P, including NaOH-P_i, NaOH-P_o, and HCl-P_i, were the dominant fractions in the tested soils. HCl-P_i and NaOH-P_i were the major P fraction of moderately active P in arid agricultural land and paddy land, respectively, indicating that land-use change leads to the conversion from Ca-bound P to P associated with Fe and Al.

Conclusions The soil P leaching change point decreased due to land-use conversion from arid agricultural soils to paddy soils, which may lead to higher P leaching risk. Therefore, it is necessary to strengthen the management and control of soil P loss in areas with large-scaled conversion from arid agricultural land to paddy fields.

Keywords Land-use change, Arid agricultural land, Paddy land, Phosphorus leaching, SPOLERC, Xingkai Lake Basin

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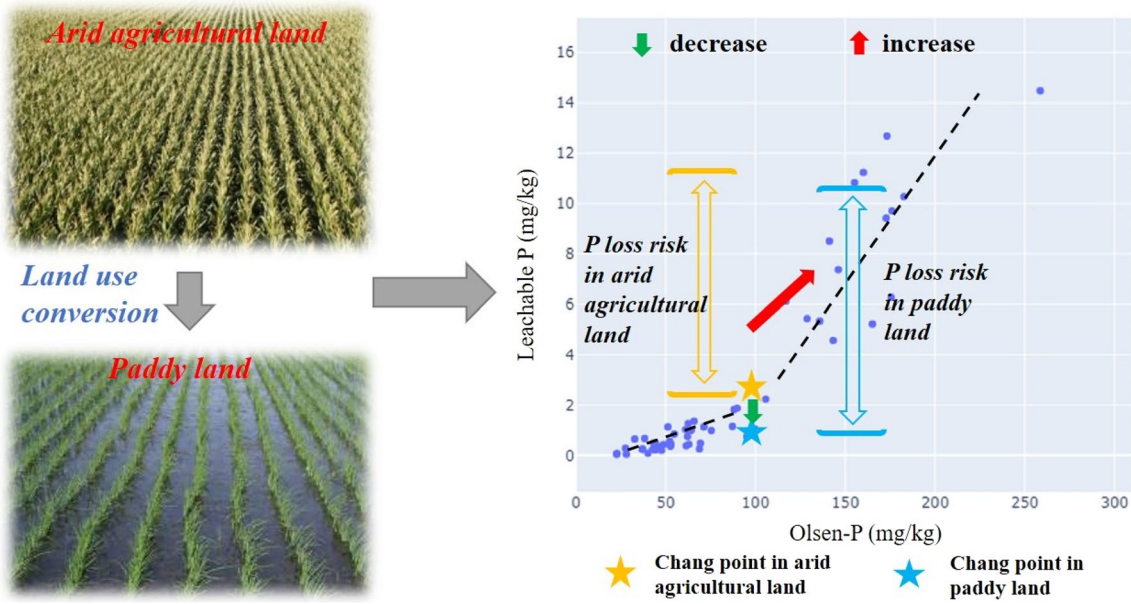
Graphical Abstract



Background

The black soil region of northeast China is the major grain producing area in China. The total area of black soil in northeast China is about 1.02×10^6 km^2, accounting for approximately 20% of global black soil [14]. Since the 1950s, large areas of black soil in China have been reclaimed from natural land to farmland, which has had a negative effect on the loss of black soil [58]. After reclamation, the thickness of black soil layer lost by 0.3–1.0 cm yr^{-1}, which has decreased from 60–70 cm in 1950s to 20-30 cm presently; the organic matter (OM) content in natural black soil was 40–60 g kg^{-1}, but it might decrease to 20–30 g kg^{-1} after reclamation [15, 34, 58]. Agricultural reclamation may increase soil phosphorus (P) losses from black soil. Long-term excess P fertilizer application and low P fertilizer utilization leads to the abundant accumulation of P in the top-layer soil of agricultural region [50, 52, 70]. In addition, the loss of OM decreased adsorption ability of inorganic phosphate and organic phosphorus compounds by soil [6, 56]. Excessive soil P accumulation and the reduction of soil P adsorption capacity can lead to large P losses from soils to the surface or ground waters because of rainfall and irrigation, which in turn contributes to eutrophication of waterbodies [16, 18, 22]. Consequently, it is important to investigate factors influencing P loss from black soils so that remedial actions can be identified and implemented.

The most common methods for P leaching risk evaluation include leachate collection drum, lysimeter, P adsorption capacity, P leaching change point [4, 20, 26, 39]. The leachate collection drum and lysimeter can effectively quantify P leaching flux, but it is time-consuming, costly, and difficultly applied to the assessment in a large-scale area [20, 26, 28]. The parameters related to P adsorption capacity include degree of P saturation, P sorption index, P sorption maximum; these indicators can effectively interpret the adsorption capacity for soil P [4, 26, 39], but limited to explain the relation between soil P and water environment P. $CaCl_2$-P, soil leachable P extracted by 0.01 M $CaCl_2$ solution, is an indicator of the potential for soluble P to enter soil interstitial water and be lost by leaching or surface runoff [18, 22, 54]. $CaCl_2$-P has a significant positive linear relationship with soil P leachates [18, 28, 53]. Based on the split-line relationship between soil Olsen-P and $CaCl_2$-P, a change point can be estimated, and used to predict soil P leaching risk [18, 22, 66]. Subsequently, Xie et al. [53] identified that $CaCl_2$-P and change point could be used to evaluate P leaching risk in soil profiles by a cascading extraction method. The cascade extraction method mimics P leaching processes in the soil profile by using the extracting solution from the soil layer above as the extractant. A soil P leaching risk calculator (SPOLERC) has been developed, which has considerably improved the efficiency of change point calculation and risk assessment for soil P leaching [55].

Compared with traditional soil P leaching risk assessment methods, such as columns or field lysimeters, the cascade extraction method combined with the SPOL-ERC model is a cost-effective approach to evaluate soil P leaching risk [18, 28, 53].

Determining soil P species is important to improve understanding of the process of soil P migration and transformation [1, 47]. The Hedley P sequential extraction method is the most commonly used method for extracting soil P forms [19, 49]. According to the Hedley P sequential extraction method, soil P can be classified into three categories: active P, moderately active P and stable P [47, 49]. The Hedley P sequential extraction method aims to evaluate how tightly the P is bound within the soil matrix and estimates the availabilities and stabilities of P species [19, 47]. Exploring the variable characteristics of different P components is beneficial for analyzing soil P loss risk and regional P availability management.

Sanjiang Plain is the core area of black soil in northeast China. It is a wetland plain formed by alluvial waters of the Heilong River, Songhua River and Wusuli River. Xingkai Lake is the main water source of the Wusuli River and the largest border lake between China and Russia. Over the past two decades, water quality in Xingkai Lake has continuously deteriorated. Total P (TP) in Xingkai Lake exceeded the level III of environmental quality standards for surface water (≥ 0.05 mg L^{-1}). The proportion of agricultural land in this area was 37.7% (23.8% in paddy land and 76.2% in arid agricultural land). As the major region of Xingkai Lake Basin, the agricultural land in Mishan City increased from 3008 to 3251 km^2 during 1990–2020. Paddy land increased from 856 to 1142 km^2 and mainly converted from arid agricultural land (Fig. 1).

The conversion from arid agricultural land to paddy land may alters soil physical structure, redox environment and microbial community characteristics, which then affect soil P loss risk [5, 11, 26]. Surface runoff and subsurface leaching are the common pathways for P losses [26]. Yang et al. [59] demonstrated that the TP loss by surface runoff from arid agricultural soils in China ranged from 0.045 ~ 0.473 kg·ha^{-1}, which were lower than that of paddy soils (0.585 ~ 3.015 kg·ha^{-1}). Fu et al. [17] demonstrated that the amount of TP loss by subsurface P leaching is 1.6 times higher than that by surface runoff from paddy land in China. Hence, it

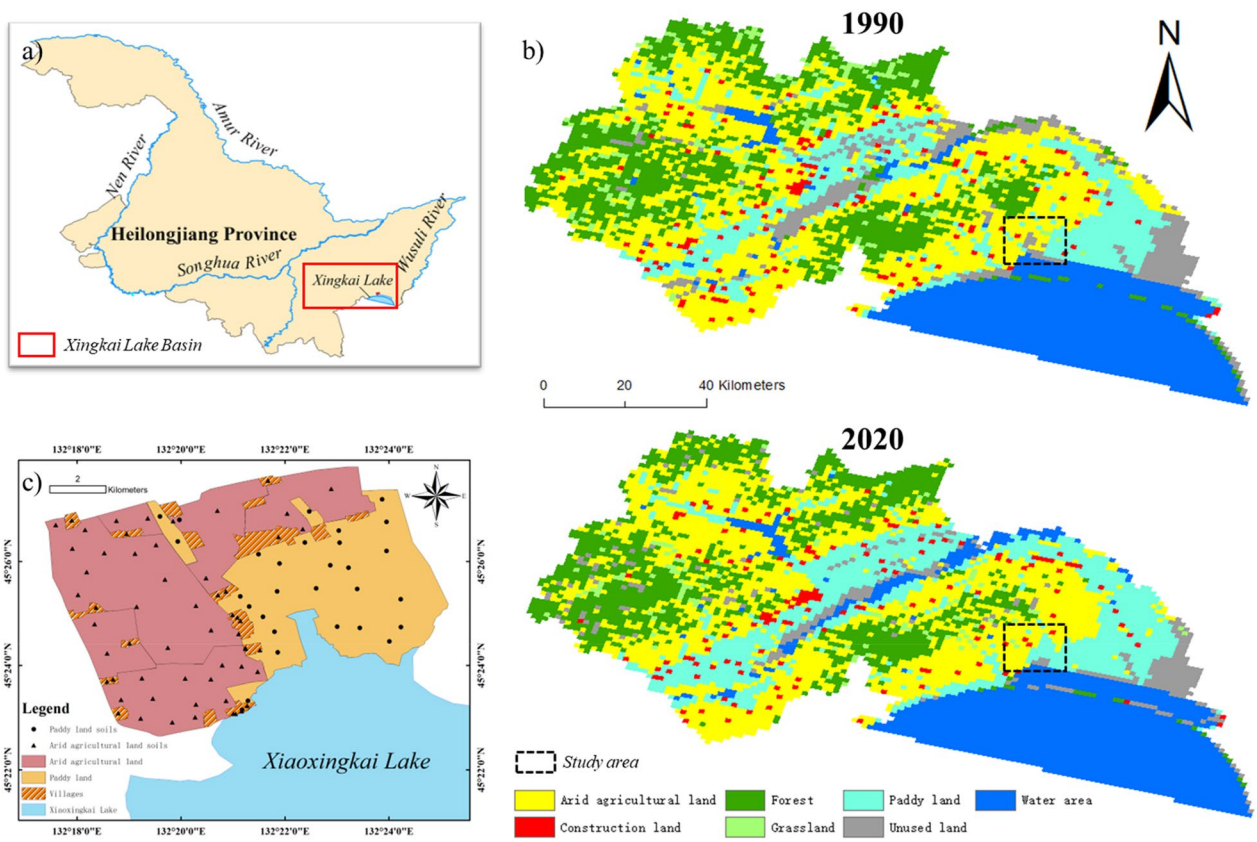


Fig. 1 Location and land use of the study area: **a** map of Heilongjiang Province, **b** land-use change from 1990 to 2020, **c** the sampling sites in the study area

is important to evaluate the effect of land-use change from arid agricultural land to paddy land on soil P leaching potential in the Xingkai Lake Basin.

The aims of this study were (1) to investigate the effect of land-use change from arid agricultural land to paddy land on soil P stocks and availability; (2) to analyze the change in P fractions due to the conversion from arid agricultural land to paddy land; and (3) to compare and evaluate the risk of P leaching as the land-use change from arid agricultural land to paddy land. It is expected that the current work will provide insight into water quality improvement and sustainable development of agriculture in the Xingkai Lake Basin.

Materials and methods

The study area and soil sample collection

The study area was in the northwest part of Xingkai Lake Basin, at an average elevation of 70 m. Arid agricultural land and paddy land are the major land use in this area. Most paddy land in the survey area was converted from arid agricultural land (Fig. 1), and had been cultivated for rice for an average of 26 years. This area belongs to temperate continental monsoon climate zone, and the average annual temperature and precipitation was 3 ℃ and 654 mm, respectively [33]. The dominant crops grown on arid agricultural land are corn (*Zea mays* L.) and soybean (*Glycine max (Linn.)* Merr.), while the dominant crop grown on paddy land is rice (*Oryza sativa* L.). Some arid agricultural land in urban areas has been planted with vegetables, including Chinese cabbage (*Brassica pekinensis (Lour.)* Rupr.) and carrot (*Raphanus sativus* L.). Before conversion, the amount of fertilizer applied to arid agricultural land were 75.0–168.7 kg N ha^{-1} yr^{-1}, 18.7–39.3 kg P ha^{-1} yr^{-1}, and 31.7–64.4 kg K ha^{-1} yr^{-1}; while in the paddy land, the applied fertilizer were 56.2–142.5 kg N ha^{-1} yr^{-1}, 13.1–27.1 kg P ha^{-1} yr^{-1}, and 21.5–52.3 kg K ha^{-1} yr^{-1}.

The soil was classified as phaeozems [21]. The sampling sites are presented in Fig. 1. After crop harvest (October 2020), a total of 261 soil samples were collected in this area from 87 sites (55 in arid agricultural land, and 32 in paddy land) at three soil depths (0–20 cm, top-layer; 20–40 cm, sub-layer; 40–60 cm, deep-layer). Collected soil samples were air-dried, ground, and passed through 2-mm and 0.15-mm sieves. The 2-mm sieved soil used for the analysis of pH, Olsen-P and leachable P, while the 0.15-mm sieved soil used for the analysis of organic matter (OM) total nitrogen (TN), total phosphorus (TP), and P fraction.

Determination of basic soil properties and P status

pH was measured by using a laboratory pH meter at a soil/water ratio of 1:2.5. OM was determined by the low temperature external thermal potassium dichromate oxidation colorimetric method [35]. Soil total nitrogen (TN) was determined by automatic Kjeldahl apparatus [35].

Olsen-P was determined by extracting soil with a 0.5 M $NaHCO_3$ (pH = 8.5) solution at a 1:20 soil/solution ratio [35, 40]. A cascade extraction method was used to determine soil leachable P at depths of 0–20 cm, 20–40 cm, and 40–60 cm [53]. Specifically, soil from 0 to 20 cm depth was extracted with 0.01 M $CaCl_2$, soil from 20 to 40 cm depth was extracted using an extract solution from 0 to 20 cm depth, and soil from 40 to 60 cm depth was extracted using an extract solution from 20 to 40 cm depth. In each case a soil:solution ratio of 1:5 was used. Soil TP was determined by digested in HNO_3–HF using microwave digestion [35]. The P in extracted or digested solution was determined by the molybdenum colorimetry method [38].

Soil P fractions were obtained using the Hedley sequential extraction procedure [19, 23, 36, 47, 49]. The P fractions included: (a) resin-P, 0.5 g soil was extracted with 25 mL deionized water and 0.5 g anion exchange resin; (b) $NaHCO_3$-P_i and $NaHCO_3$-P_o, the remaining soil was shaken with 25 mL 0.5 mol/L $NaHCO_3$ solution; (c) NaOH-P_i and NaOH-P_o, the remaining soil was shaken with 25 mL of 0.1 mol/L NaOH solution; (d) HCl-P_i, the remaining soil was shaken with 25 mL 1 mol/L HCl solution; and (e) residual P, the remaining soil was digested using 5 mL H_2SO_4 and 5 mL H_2O_2. Inorganic P (P_i) was determined directly using molybdenum colorimetry method [38]. Total dissolved phosphorus (P_t) in the filtrates was determined using molybdenum colorimetry method after digestion with alkaline potassium persulfate [23, 36, 47]. Organic P (P_o) was calculated as P_t minus P_i.

Data analysis

Statistical analysis, multiple comparisons (LSD) and 2-way ANOVA were conducted to test the difference in soil properties and P fractions by using SPSS v21.

The soil P leaching change point and risk probability was calculated by SPOLERC [55]. SPOLERC has developed two models for the calculation of P leaching change point based on a two linear relationship between soil Olsen-P and leachable P. Model 1 was used to calculate the change point according to two significant fitting lines, and the change point is the intersection between these two linear equations. While in model 2, if the linear relationship between soil Olsen-P and leachable P below the change point was not significant or it did not exist, the change point was calculated according to the intersection

between X axis and another line with significant correlation. Subsequently, the single factor index (SFI) method was used to evaluate the soil P leaching risk level. The risk index (RI) is equal to soil Olsen-P divided by P leaching change point, and the P leaching risk can be divided into four levels: no risk (RI ≤ 1), low risk (1 > RI ≤ 2), medium risk (2 > RI ≤ 3), high risk (RI ≥ 3).

Results and discussion

Effects of land-use change on basic soil properties

The soil was weakly acidic down the entire profile. Soil pH values were lower in paddy land than in arid agricultural land. The pH values of arid agricultural soils in the top-layer and sub-layer were higher than that of paddy soils, and the difference was significant in the top-layer soils ($p < 0.05$) (Table 1). Moreover, the pH values of arid agricultural soils in the deep-layer were significantly lower than that of paddy soils ($p < 0.05$) (Table 1). During conversion from arid agricultural fields to paddy fields, more nitrate ions are produced due to the waterlogging and anaerobic conditions, which leads to a decrease of soil pH [51, 69]. Long-term application of urea is likely to an important factor influencing soil acidity [54, 69]. The rate of N fertilization in the arid agricultural soil and paddy soil was 75.0–168.7 and 56.2–142.5 kg N ha^{-1} yr^{-1}, respectively. Urea hydrolyzes to NH_4^+ under the catalysis of urease and then converts into NO_3^- [46]. The uptake of NH_4^+ by crops promotes the secretion of H^+ ions by roots, which results in a decline of soil pH [71].

Soil OM in the study area was classified as abundant (3 ~ 4%) and very abundant (> 4%), apart from the deep-layer soils in the paddy land (medium level, 2–3%) [8], Table 1). Higher OM in the top-layer soil was mostly due to crop residues returning to the field. Land-use conversion to paddy soils tended to significantly decrease soil OM ($p < 0.05$). OM in arid agricultural soils was higher than that of paddy soils across the entire profile ($p < 0.05$) and tended to decrease with increasing depth ($p < 0.01$) (Table 1). This difference could be attributed to freeze–thaw cycles and the transformation from arid agricultural soils to paddy soils. The moisture content in arid agricultural soils of 0–20, 20–40, and 40–60 cm were 31%, 30%, and 29%, and was lower than that of paddy soils (46%, 35% and 34%, respectively). The cycle of freezing and thawing can damage the soil aggregate structure, particularly for paddy soils with high moisture content. After the soil structure is broken, OM particles are more likely to be lost under the influence of snow melt in spring, heavy rain in summer and farmland water retreating in autumn [27, 67]. In addition, OM in the deep-layer of paddy soil is significantly lower than that of arid agricultural soils, which indicates that land-use change may accelerate the loss of OM in the black soil layer ($p < 0.05$) (Table 1).

Land-use change from arid agricultural field to paddy field tended to decrease the TN stocks in the survey area. TN in arid agricultural soils was higher than that of paddy soils across the entire profile and tended to decrease with profile depth ($p < 0.01$) (Table 1). While the application of N fertilization in the arid agricultural soils (75.0–168.7 kg N ha^{-1} yr^{-1}) was higher than paddy soils (56.2–142.5 kg N ha^{-1} yr^{-1}), waterlogging and recession in paddy soils may increase the dissolution and loss of soil soluble nitrogen [51, 69]. Soil TN in the all soil profiles of the study area was classified as abundant (1.5 ~ 2.0 g kg^{-1}) and very abundant (> 2.0 g kg^{-1}), apart from the deep-layer of paddy soils was classified as medium level (0.75 ~ 1.0 g kg^{-1}) [8], Table 1).

Table 1 Soil properties distribution with profile depth under arid agricultural soils and paddy soils

Land use	Profile depth (cm)	pH (–)	OM (%)	TN (mg kg^{-1})	C:N (–)	C:P	N:P
Arid agricultural soils	0–20	6.43 ± 0.94a*[†]	5.42 ± 0.79a	2648.4 ± 790.0a	11.94 ± 1.50a	24.04 ± 7.66a**	2.01 ± 0.61a*
	20–40	6.45 ± 0.75a	4.49 ± 1.06b	2036.3 ± 807.2ab	13.51 ± 2.42ab	25.13 ± 5.58a*	1.88 ± 0.32a*
	40–60	6.23 ± 0.64a	3.19 ± 1.17c*	1508.3 ± 853.6b	15.48 ± 3.21b	27.08 ± 7.10a	1.78 ± 0.45a
Paddy soils	0–20	6.05 ± 0.48a*	5.37 ± 0.83a	2480.2 ± 775.7a	13.04 ± 2.01a	34.19 ± 6.34a**	2.62 ± 0.24a*
	20–40	6.32 ± 0.48b	4.22 ± 1.03b	1821.6 ± 1057.1a	15.01 ± 4.21a	30.35 ± 4.13ab*	2.20 ± 0.79ab*
	40–60	6.37 ± 0.46bc	2.68 ± 0.70c*	946.7 ± 224.0b	16.82 ± 1.90b	28.51 ± 5.30b	1.70 ± 0.31b
Analysis of variance (*P value*)							
Land use		0.177	0.026	0.147	0.073	0.002	0.034
Profile depth		0.411	< 0.001	< 0.001	0.001	0.753	0.003
Land use × profile depth		0.054	0.317	0.716	0.975	0.115	0.112

[†] Means ± SD followed by a different lowercase letter indicate significant difference from different profile depth in the same land use

*, **indicate significant difference in the same soil depth between arid agricultural soils and paddy soils at $p < 0.05$ and $p < 0.01$, respectively

Soil C:N:P stoichiometry can be used to evaluate the supply of soil nutrients [30, 69]. Soil P mineralization rate and P availability were negatively correlated with C:N and C:P ratio [30]. The C:P ratio is an indicator of soil P mineralization potential and P release potential from OM [69]. The average C:N:P ratios in arid agricultural soils ranged from 24.0:2.0:1 to 27.6:1.8:1, while it ranged from 28.6:1.7:1 to 34.2:2.6:1 in paddy soils. Compared with the average soil C:N:P ratios at the global scale (287:17:1 or 186:13:1) [10, 57], or national scale (134:9:1) [48], soils in the Xingkai Lake Basin showed remarkably narrow C:N:P ratios, owing to the higher P accumulation in the agricultural area. The C:N and C:P ratio in arid agricultural soils was lower than that of paddy soils, and the difference in C:P ratio reached a significant level ($p < 0.01$) (Table 1). The N:P ratio in the arid agricultural soils and paddy soils is less than 14.5, which indicates that was restricted by N content and the activity of soil P is lower [69]. Since microbes have a competitive advantage over plants for N uptake, this N-limitation status could lead to a reduction in crop yields [24, 68]. Additionally, the N:P ratio in the top-layer and sub-layer of arid agricultural soil was significantly lower than paddy soil ($p < 0.05$), and the N:P ratio in the deep-layer of arid agricultural soil was higher than paddy soil (Table 1). Because chemical fertilizer is rich in N and P, intensive application of chemical fertilizer could lead to a higher accumulation of N and P than that of C; this unbalanced nutrient status may result in an enhanced risk of N and P loss from the soil [51, 69].

Effects of land-use change on soil TP, Olsen-P, and leachable P in soil profiles

Conversion from arid agricultural land to paddy land influences soil P storage, availability and loss potential. TP in the top-layer, sub-layer, and deep-layer of arid agricultural soils were 1423.9 ± 656.3, 1081.0 ± 361.5, and 870.1 ± 547.9 mg kg^{-1}, respectively, which were higher than that of paddy soils (949.1 ± 291.3, 783.6 ± 196.1 and 557.5 ± 103.9 mg kg^{-1}, respectively) (Table 2). The TP values tended to decrease with profile depth. Generally, TP in the study area was classified as abundant (800 ~ 1000 mg kg^{-1}) and very abundant (> 1000 mg kg^{-1}), apart from the deep-layer of paddy soils (deficient level, 400 ~ 600 mg kg^{-1}) [8], Table 2). Abundant soil P stocks are easily transformed into soil soluble P through desorption, dissolution, and mineralization; the soluble P in the top-layer soil is easily lost to the surface water through surface runoff or leachate to the sub-layer soil, deep-layer and shallow groundwater under the drive of rainfall and irrigation, resulting in a decrease of water quality in the surface water or shallow groundwater [26, 42].

Furthermore, Olsen-P in arid agricultural soils were higher than that of paddy soils across the entire profile, and the significant difference occurred in top-layer and sub-layer ($p < 0.05$) (Table 2). Olsen-P in the top-layer, sub-layer, and deep-layer of arid agricultural soils were 85.3 ± 53.4, 45.7 ± 41.8, and 30.1 ± 22.0 mg kg^{-1}, respectively, which was higher than that of paddy soils (34.1 ± 10.2, 19.6 ± 9.4, and 25.4 ± 7.0 mg kg^{-1}) (Table 2). Olsen-P tended to decrease with depth in arid agricultural soils; while in paddy soils, the values in the sub-layer soil were lowest. Rice roots are mostly distributed in the top-layer and sub-layer soils; roots adsorption leads to the decrease of P in the sub-layer soils, while in the deep-layer soils, P is difficult to adsorb by roots [63]. Nearly 62.1–96.6% of arid agricultural soils and 60.0–93.3% of paddy soils were higher than the critical value of 20.0 mg kg^{-1} (Olsen-P), which could be classified into P-enriched soils in China [44]. Moreover, the concentration of leachable P in arid agricultural soils

Table 2 Soil TP, Olsen-P and leachable P distribution with profile depth under arid agricultural soils and paddy soils

Land use	Profile depth (cm)	TP (mg kg^{-1})	Olsen-P	Leachable P
Arid agricultural soils	0–20	1423.9 ± 656.3a[†]	85.3 ± 53.4a**	2.80 ± 3.87a**
	20–40	1081.0 ± 361.5ab	45.7 ± 41.8b**	0.78 ± 1.10b**
	40–60	870.1 ± 547.9b	30.1 ± 22.0c	0.38 ± 0.83bc*
Paddy soils	0–20	949.1 ± 291.3a	34.1 ± 10.2a**	0.10 ± 1.56a**
	20–40	783.6 ± 196.1a	19.6 ± 9.4b**	0.03 ± 0.03b**
	40–60	557.5 ± 103.9b	25.4 ± 7.0c	0.08 ± 0.05a*
Analysis of variance (*P value*)				
Land use		0.003	< 0.001	< 0.001
Profile depth		0.007	< 0.001	< 0.001
Land use × profile depth		0.788	< 0.001	< 0.001

[†] Means ± SD followed by a different lowercase letter indicate significant difference from different profile depth in the same land use

*, **indicate significant difference in the same soil depth between arid agricultural soils and paddy soils at $p < 0.05$ and $p < 0.01$, respectively

was significantly higher than for paddy soils ($p < 0.01$) (Table 2). Leachable P in arid agricultural soils of 0–20 cm, 20–40 cm, and 40–60 cm were 2.80 ± 3.87, 0.78 ± 1.10, and 0.38 ± 0.83 mg kg^{-1}, respectively, while the values in the paddy soils were 0.10 ± 1.56, 0.03 ± 0.03, and 0.08 ± 0.05 mg kg^{-1}, respectively (Table 2). Leachable P tended to decrease with soil depth in arid agricultural soils; while in paddy soils, the values in the sub-layer soil were lowest.

Conversion from arid agricultural land to paddy land will lead to the change of fertilization, crop planting type, and irrigation mode (sprinkler irrigation and flooding irrigation), then affect the P stocks and its migration and transformation process [43, 69]. The arid agricultural soil is fertilized for almost 40–60 years, while for paddy soil, it's fertilized for almost 5–30 years. The application of P fertilizer in arid agricultural soil was 18.7–39.3 kg P ha^{-1} yr^{-1}, and in paddy soil, it was 13.1–27.1 kg P ha^{-1} yr^{-1}. The amount of fertilizer is chosen based on the type of crop and the availability of soil fertility. Previous studies demonstrate that the average crop yield in arid agricultural land was 11183 kg ha^{-1}, with a range from 9731 ~ 12057 kg ha^{-1}, which was higher than that in paddy land (8506 kg ha^{-1}, 7365 ~ 9546 kg ha^{-1}); moreover, the phosphate fertilizer utilization efficiency in arid agricultural soil (28.0%) was higher than that in paddy soil (15.6%) [61, 64]. Chi et al. [7] have also found that soil Olsen-P in arid agricultural soils was significantly higher than in paddy soils. Furthermore, P absorbed by Fe- and Al-oxides becomes more soluble under flooded conditions, and soil pH decreases with flooding, increasing the solubility of Ca-P [25, 31]. However, when flooded soil is drained, iron can re-oxidize to form complex compounds with phosphate. Large quantities of dissolved P can be lost via surface runoff during the drainage process, which can result in a decrease of Olsen-P and leachable P [41, 69].

Effects of land-use change on soil P fractions in soil profiles

Fractionation of P analysis using the Hedley scheme identified that residual P was the dominant P fraction in arid agricultural soils and paddy soils across the entire soil profile. The NaOH-Pi, NaOH-Po, and residual P in arid agricultural soils were lower than that of paddy soils. The resin-Pi, $NaHCO_3$-Pi, $NaHCO_3$-Po, and HCl-Pi in arid agricultural soils were significantly higher than that of paddy soils ($p < 0.05$) (Table 3). In arid agricultural soils, residual P tended to increase with depth, while other P fractions tended to decrease with depth. For paddy soils, residual P in the sub-layer soils was highest, and other P fractions in the sub-layer soils was lowest.

Table 3 Content and percentage of Hedley P fraction distribution with profile depth under arid agricultural soils and paddy soils

Land use	Profile depth (cm)	Resin-Pi (mg kg^{-1})	$NaHCO_3$-Pi	$NaHCO_3$-Po	NaOH-Pi	NaOH-Po	HCl-Pi	Residual P
Arid agricultural soils	0–20	49.9 ± 40.0a**†	239.4 ± 169.6a*	81.7 ± 55.1a	225.1 ± 90.6a	146.1 ± 68.2a	374.3 ± 267.1a**	307.5 ± 128.9a
	20–40	19.4 ± 17.6b	131.1 ± 81.1b**	69.0 ± 52.7a	154.1 ± 67.5ab	138.1 ± 48.3a	211.0 ± 153.5a	358.2 ± 70.5a
	40–60	13.9 ± 20.1bc	89.9 ± 92.3bc	63.0 ± 34.7a	146.7 ± 102.0b	73.4 ± 46.3b	190.4 ± 258.3a	292.7 ± 125.5a
Paddy soils	0–20	5.4 ± 2.8a**	88.3 ± 33.9a*	51.3 ± 16.4a	210.7 ± 78.6a	163.1 ± 53.4a	120.5 ± 38.3a**	310.0 ± 102.7a
	20–40	4.0 ± 2.1a*	45.3 ± 23.6bc**	47.0 ± 20.5a	122.9 ± 52.2ab	127.6 ± 61.0a	109.5 ± 66.6a	327.4 ± 45.0a
	40–60	3.5 ± 1.1a	38.6 ± 12.8c	43.3 ± 11.5a	101.0 ± 27.5b	55.8 ± 9.1b	90.4 ± 67.3a	224.9 ± 37.3b
Percentage of P (%)								
Arid agricultural soils	0–20	3.1 ± 1.6	15.5 ± 4.9	5.7 ± 2.2	16.7 ± 5.5	12.2 ± 8.4	24.5 ± 9.1	22.2 ± 6.0
	20–40	1.6 ± 1.1	11.3 ± 4.6	6.0 ± 2.4	14.1 ± 4.0	13.7 ± 5.3	17.9 ± 8.6	35.4 ± 9.9
	40–60	1.3 ± 1.2	9.0 ± 4.1	8.5 ± 3.3	16.5 ± 2.4	11.2 ± 6.9	16.6 ± 10.6	37.1 ± 10.0
Paddy soils	0–20	0.6 ± 0.2	9.3 ± 1.7	5.4 ± 0.5	22.2 ± 3.3	17.3 ± 2.9	12.9 ± 2.6	32.5 ± 4.1
	20–40	0.5 ± 0.2	5.5 ± 1.9	6.1 ± 2.4	15.2 ± 3.6	16.0 ± 5.1	13.2 ± 6.7	43.7 ± 11.1
	40–60	0.6 ± 0.1	6.8 ± 1.5	7.8 ± 1.4	18.0 ± 2.7	10.4 ± 2.6	15.0 ± 9.1	41.4 ± 9.6
Analysis of variance (*P value*)								
Land use		< 0.001	< 0.001	0.024	0.145	0.790	0.003	0.220
Profile depth		0.020	0.006	0.571	0.001	< 0.001	0.174	0.035
Land use × profile depth		0.039	0.270	0.906	0.824	0.568	0.343	0.542

† Means ± SD followed by a different lowercase letter indicate significant difference from different profile depth in the same land use

*, **indicate significant difference in the same soil depth between arid agricultural soils and paddy soils at $p < 0.05$ and $p < 0.01$, respectively

Resin-Pi, $NaHCO_3$-Pi and $NaHCO_3$-Po are generally considered as the most biologically available P fractions (defined the active P) [36, 47]. The average content of active P in arid agricultural soils of 0–20, 20–40, and 40–60 cm was 371.0, 219.5, and 166.8 mg kg^{-1}, respectively, while the values in paddy soils of 0–20, 20–40, and 40–60 cm was 145.0, 96.3, and 85.4 mg kg^{-1}, respectively (Table 3). $NaHCO_3$-Pi was the dominant form of active P, accounting for 4.4–11.0% and 2.5–6.7% in arid agricultural soils and paddy soils, followed by the $NaHCO_3$-Po and resin-Pi (Table 3).

NaOH-Pi, NaOH-Po, and HCl-Pi are often associated with iron and aluminum oxides and easily bound to calcium (defined as moderately active P) [36, 47]. The average content of moderately active P in arid agricultural soils of 0–20, 20–40, and 40–60 cm was 745.5, 503.2, and 410.5 mg kg^{-1}, respectively, and were higher than that of paddy soils in the soil depth of 0–20, 20–40, and 40–60 cm (494.3, 360 and 247.2 mg kg^{-1}, respectively) (Table 3). HCl-P_i was the dominant form of moderately active P in the arid agricultural soils, accounting for 8.6–16.9%, while the NaOH-Pi was the leading moderately active P in the paddy soils, accounting for 6.8–16.4% (Table 3).

Residual P is difficult to decompose (defined as stable P) [12, 47]. As the leading P fraction, the residual P content in arid agricultural soils of 0–20, 20–40, and 40–60 cm were 307.5 ± 128.9, 358.2 ± 70.5, and 292.7 ± 125.5 mg kg^{-1}, respectively, and the values in paddy soils of 0–20, 20–40, and 40–60 cm were 310.0 ± 102.7, 327.4 ± 45.0, and 224.9 ± 37.3 mg kg^{-1}, respectively (Table 3). Although the residual P content in arid agricultural soils was higher than that of paddy soils across the entire profiles, the P percentage in the paddy soils (32.5–43.7%) was higher than that of arid agricultural soils (22.2–37.1%) (Table 3).

The shift from arid agricultural land to paddy land can impact on soil P fractions [2, 47, 62, 65]. Fertilization directly replenishes the active P, the active P decreased with the extension of time, and alters to moderately active P and stable P under the action of precipitation, adsorption, and biological retention [26, 47, 69]. Compared with paddy soils, arid agricultural soils applied more fertilizer (18.7–39.3 vs. 13.1–27.1 kg P ha^{-1} yr^{-1}). As the paddy soil is mostly flooded during fertilization, the applied fertilizer dissolves quickly and produces large-amounts of active P; similarly, the recession of water in paddy land will take away more active P, resulting in poor surface water quality [41, 69]. Additionally, the freeze–thaw cycle can damage the aggregate structure of farmland soil, particularly the paddy soil with high water content, and decrease the stability of moderately active P and stable P, as well as increase soil P activity [13, 47].

Moreover, the acidity and alkalinity of the soil can affect the fixation way of P. P is likely retained by soil clay minerals and oxyhydroxides according to ligand exchange in acidic soils, while in alkaline soils, P is likely adsorbed by the precipitation of calcium phosphates and by sorption on calcium carbonates and clay minerals [3]. Paddy soil has a lower pH value than arid agricultural land (Table 1). The decrease of pH resulted in the solubility of the HCl-P_i, which was closely bound to Ca [62]. NaOH-P, including NaOH-Pi and NaOH-Po, was the main P fraction in the tested area, indicating that most P was associated with Fe and Al oxides instead of being bound to Ca, which was similar to results from previous studies [47]. Additionally, soil P adsorbed by Fe- and Al-oxides becomes more soluble under flooded conditions, which may explain why NaOH-P in paddy soils was lower than that in arid agricultural land [25, 31] (Table 3). The organic P (NaOH-P_o and $NaHCO_3$-P_o) difference between arid agricultural soils and paddy soils is relatively small, suggesting that land-use change has a limited effect on soil organic P [2] (Table 3).

Effects of land-use change on soil P leaching change point in soil profiles

There was a significant correlation between soil leachable P and Olsen-P in the top-layer, sub-layer and deep-layer of arid agricultural soils and paddy soils (Fig. 2). The P leaching change point in the arid agricultural soils of 0–20, 20–40, and 40–60 cm was 59.63, 24.31, and 32.91 mg kg^{-1}, respectively, which was higher than that of paddy soils (35.35, 17.20 and 10.45 mg kg^{-1}, respectively) (Fig. 2). The change point tended to decrease with depth in paddy land. While in arid agricultural land, the value was lowest in the sub-layer soil. Originally, Heckrath et al. [18] found the linear split-line model between soil Olsen-P and dissolved reactive P in drainage water in Broadbalk, and obtained the P leaching change point was 60 mg kg^{-1}. Then, Heskrath and Brookes [20] confirmed that special linear relationship between the soil Olsen-P and dissolved reactive P in drainage water could also be found between the soil Olsen-P and 0.01 M-$CaCl_2$-extractable P, and the P leaching change point was predicted in a wide range of 10–119 mg kg^{-1} in the UK. McDowell et al. [37] also investigated different soil from the United Kingdom, New Zealand, and the United States and evaluated the change point for Olsen-P was 20–112 mg kg^{-1}. Additionally, Zhao et al. [72] found that the change point in Chinese agricultural soils ranged from 30–160 mg Olsen-P kg^{-1}; this result is consistent

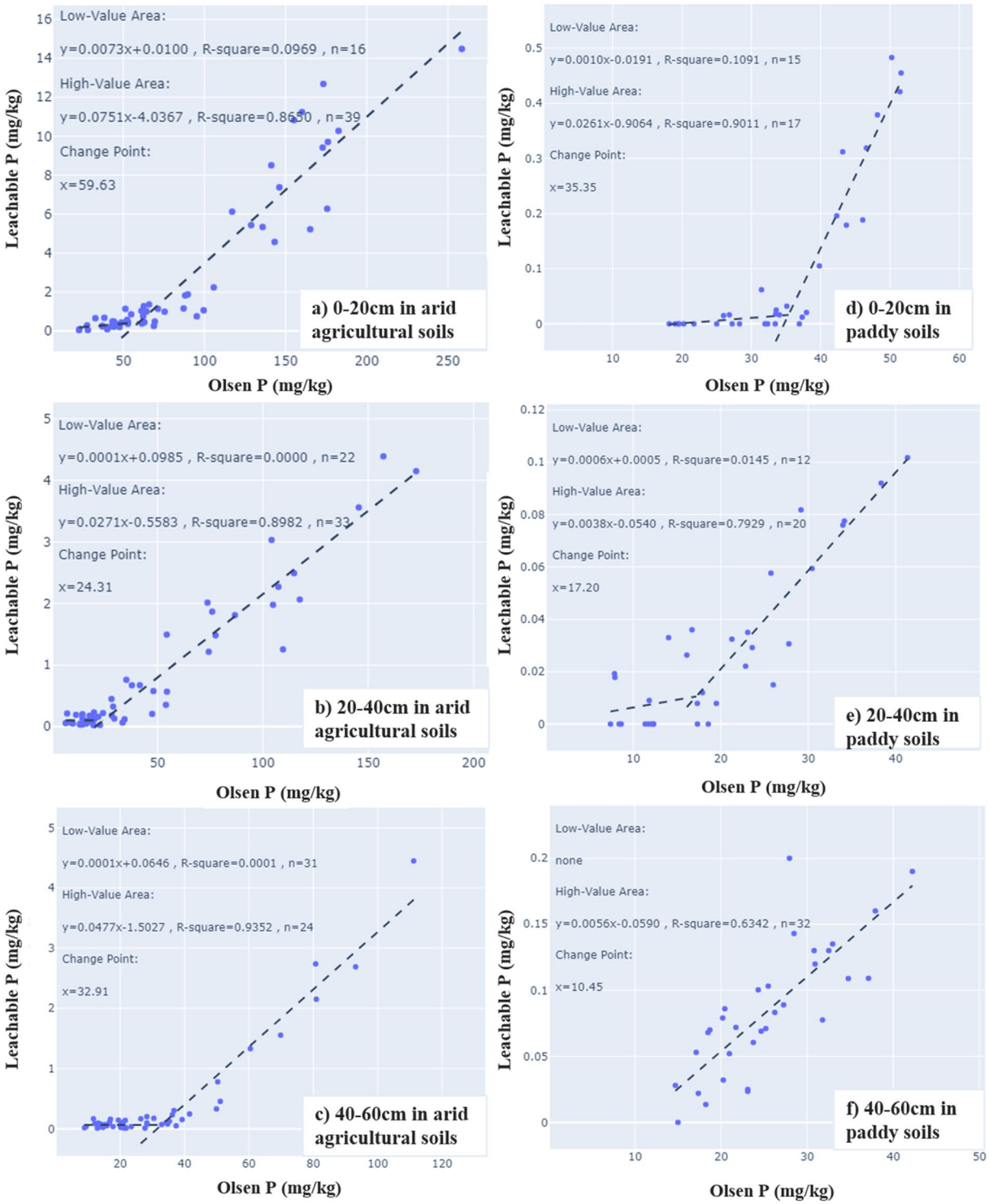


Fig. 2 Relationships between Olsen-P and leachable P in arid agricultural soils (**a**, **b**, **c**) and paddy soils (**d**, **e**, **f**) calculated by the SPOLERC

with the change point in the top-layer soil in the present study.

The soil P leaching change point decreased after land-use change from arid agricultural land to paddy land. Previous analysis demonstrated that soil types, pH, and OM content were the main factors affecting the P leaching change point, and that land-use change may influence these soil basic properties [28, 54, 72]. For example, soil pH can influence soil P leaching change point. A negative relationship was found between soil pH and the P leaching change point at soil $pH > 6.0$, while in soil with $pH < 6.0$, the relationship was found to be positive [72]. Given that the conversion from arid agricultural land to paddy land decreased soil pH, and more than two-thirds of soils have a $pH < 6.0$ (Table 1), a decrease in the soil P leaching change point could result. Secondly, there was a significant positive relationship between soil OM and P leaching change point [72]. Xie et al. [54] showed that biochar application increased soil OM, and soil P leaching change point. In this study, land-use change from agricultural soil to paddy soil resulted in a decrease in OM (Table 1), which could also lead to a decline in P leaching change point. Higher P leaching change point in arid agricultural soil may be due to high soil P adsorption capacities [28]. The soil with high OM has a more stable aggregate structure and higher P adsorption capacity [26, 32]. Many authors, including Yao et al. [60] have shown that soils with higher the clay content have greater P adsorption capacity, because of clay mineralogy and much greater surface area. In the area of this study, the clay content of arid agricultural soil was higher than that of paddy soil (21.9% vs. 14.1%) [45]. In addition, recession of water in paddy land can increase the loss of OM and clay particles, especially in the top-layer soil [41, 59, 69].

Effects of land-use change on soil P leaching risk in soil profiles

The single factor index (SFI) was introduced to classify the soil P leaching risk level [29]. The evaluation index was obtained by dividing soil P leaching change point by soil Olsen-P content. When soil Olsen-P is higher than the change point, significant loss of P through leaching could occur; otherwise, P leaching risk is not observed. Additionally, the higher the evaluation index value is, the more serious the risk level is. The calculation formula and risk rate of P leaching evaluation index are shown in Table 1 of Xie et al. [55]. Based on the soil P leaching change point and the SFI evaluation method, the classification of soil P leaching risk level in arid agricultural soils and paddy soils is shown in Table 4. SPOLERC was used to calculate the soil P leaching risk in soil profiles.

Generally, a higher P leaching change point indicates a lower P leaching risk at the same soil Olsen-P content

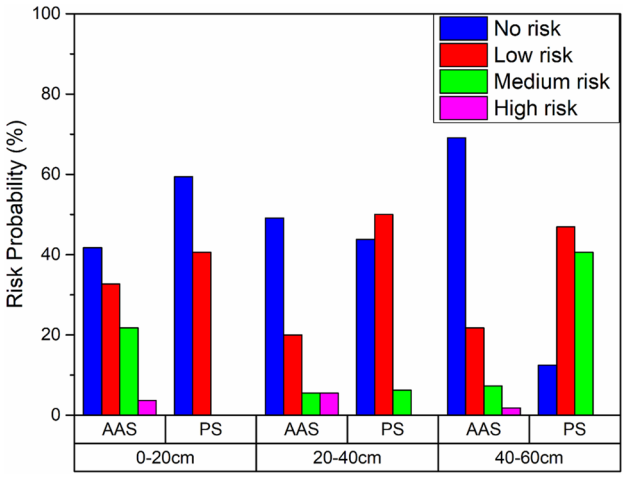


Fig. 3 Soil P leaching risk probability in arid agricultural soils (AAS) and paddy soils (PS) measured by the SPOLERC

Table 4 Classification of soil P leaching risk in arid agricultural soils and paddy soils

Risk level	Profile depth (cm)	No risk (mg kg^{-1})	Low risk	Medium risk	High risk
Arid agricultural soils	0–20	≤ 59.63[a]	59.63 ~ 119.26	119.26 ~ 178.89	> 178.89
	20–40	≤ 24.31	24.31 ~ 48.62	48.62 ~ 97.24	> 97.24
	40–60	≤ 32.91	32.91 ~ 65.82	65.82 ~ 131.64	> 131.64
Paddy soils	0–20	≤ 35.35	35.35 ~ 70.70	70.70 ~ 106.05	> 106.05
	20–40	≤ 17.20	17.20 ~ 34.40	34.40 ~ 68.80	> 68.80
	40–60	≤ 10.45	10.45 ~ 20.90	20.90 ~ 41.80	> 41.80

[a] When the soil Olsen-P is below the change point, the soil is classified as no risk area. In addition, soils are classified as risk areas when soil Olsen-P is higher than the change point

[28, 53]. According to soil P leaching change point and the Olsen-P content at sampling points, the P leaching risk probability in the arid agricultural soil sampling points of 0–20 cm, 20–40 cm, and 40–60 cm were 58.2%, 50.9%, and 30.9%, respectively, while in paddy soil sampling points, they were 40.6%, 56.3% and 87.5%, respectively (Fig. 3, Additional file 1: Figure S1). The P leaching risk probability for the top-layer of arid agricultural soils was higher than that of paddy soils. Both the sub-layer and deep-layer of arid agricultural soils had a lower probability of P leaching risk than that of paddy soils. Furthermore, 25.5% and 11.0% of arid agricultural soils at soil depth of 0–20 cm and 20–40 cm had a medium risk and high risk, which was higher than that for paddy soils (0% and 6.3%) (Fig. 3, Additional file 1: Figure S1). The medium and high P leaching risk probability of the deep-layer of arid agricultural soils was higher than that of paddy soils (9.1% vs. 40.6%) (Fig. 3, Additional file 1: Figure S1). In conclusion, 30.9% of arid agricultural soils and 87.5% of paddy soils were at risk of P leaching into shallow groundwater, while 58.2% of arid agricultural soils and 40.6% of paddy soils were at risk of P into the surface runoff or the sub-layer soil. In this study, the conversion from arid agricultural land to paddy land has led to a change in basic soil properties, such as pH, and OM, which could lead to a decrease of P leaching change point and an increase of P leaching risk. Cho [9] showed that over 60% of annual P loss occurred during rice growing season while approximately 20% of P loss occurred during the non-cropping season after harvest as surface runoff. The recession of water in paddy land will take away more active P. Since our investigation occurred in the non-cropping season, the P leaching risk from paddy soil is likely to be much higher. Hence, the priority to reduce P loss should be to focus on the application rate and improve the utilization efficiency of P fertilization.

Conclusions

This research focused on the impact of land-use change from arid agricultural land to paddy land on soil P leaching risk. The soils tested in the Xingkai Lake Basin were P-enriched. Results simulated by SPOLERC indicated that there was a significant relationship between soil leachable P and Olsen-P in the top-layer, sub-layer and deep-layer of arid agricultural soils and paddy soils. The P leaching change point in the arid agricultural soils of 0–20 cm, 20–40 cm, and 40–60 cm (59.63, 24.31, and 32.91 mg kg^{-1}, respectively) were higher than that of paddy soils (35.35, 17.20 and 10.45 mg kg^{-1}, respectively). 30.9% of arid agricultural soils and 87.5% of paddy soils were at risk of P leaching into the shallow groundwater, while 58.2% of arid agricultural soils and 40.6% of paddy soils were at risk of P loss via surface runoff. The recession of water from paddy land will take away more active P, resulting in paddy soil having a higher leaching risk. Meanwhile, moderately active P, including NaOH-P_i, NaOH-P_o, and HCl-P_i, is the leading P fraction in the tested soils. HCl-P_i and NaOH-P_i was the major P fraction of moderately active P in arid agricultural land and paddy land, suggesting that land-use change leads to the conversion from Ca-bound P to P associated with Fe and Al. In conclusion, land-use change from arid agricultural land to paddy land decreased the soil P leaching change point, which in turn resulted in increased soil P leaching risk.

Supplementary Information

The online version contains supplementary material available at https://doi.org/10.1186/s40538-023-00420-9.

Additional file 1: Figure S1. The pie chart of soil P leaching risk probability in arid agricultural soils (**a**, **b**, **c**) and paddy soils (**d**, **e**, **f**) measured by the SPOLERC.

Acknowledgements
Not applicable

Author contributions
Zi-jian Xie: conceptualization, methodology, formal analysis, investigation, writing—original draft; Dong Zhu: formalanalysis, investigation; Wei-Wei Wei: investigation; Chun Ye: validation, resources, writing—review and editing, supervision;Hao Wang: investigation; Chun-hua Li: validation, resources, writing—review and editing, supervision

Funding
This project was supported by the Open Research Fund of State Environmental Protection Key Laboratory for Lake Pollution Control (2022HPYB-12), the National Key Research and Development Project of China (Watershed non-point source pollution prevention and control technology and application demonstration, 2021YFC3201502) and the Lake Xingkai Basin Ecological Environment Survey Project.

Availability of data and materials
The datasets used and/or analyzed during the current study are available from the corresponding author on reasonable request.

Declarations

Ethics approval and consent to participate
Not applicable.

Consent for publication
Not applicable.

Competing interests
The authors declare that they have no competing interests.

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Received: 10 November 2022 Accepted: 30 May 2023
Published online: 11 July 2023

References

1. Ahlgren J, Djodjic F, Borjesson G, Mattsson L. Identification and quantification of organic phosphorus forms in soils from fertility experiments. Soil Use Manag. 2013;29:24–35. https://doi.org/10.1111/sum.12014.
2. Annaheim KE, Doolette AL, Smernik RJ, Mayer J, Oberson A, Frossard E, et al. Long-term addition of organic fertilizers has little effect on soil organic phosphorus as characterized by P-31 NMR spectroscopy and enzyme additions. Geoderma. 2015;257(2013 Ecological Significance of Soil Organic Phosphorus Meeting):67–77. https://doi.org/10.1016/j.geoderma.2015.01.014.
3. Bai Z, Li H, Yang X, Zhou B, Shi X, Wang B, et al. The critical soil P levels for crop yield, soil fertility and environmental safety in different soil types. Plant Soil. 2013;372(1–2):27–37. https://doi.org/10.1007/s11104-013-1696-y.
4. Barrow NJ, Roy D, Debnath A. Evaluating the benefits of legacy phosphate. Plant Soil. 2022;480(1–2):561–70. https://doi.org/10.1007/s11104-022-05601-3.
5. Bergkemper F, Schöler A, Engel M, Lang F, Krüger J, Schloter M, Schulz S. Phosphorus depletion in forest soils shapes bacterial communities towards phosphorus recycling systems. Environ Microbiol. 2016;18:1988–2000.
6. Bornø ML, Müller-Stöver DS, Liu F. Contrasting effects of biochar on phosphorus dynamics and bioavailability in different soil types. Sci Total Environ. 2018;627:963–74. https://doi.org/10.1016/j.scitotenv.2018.01.283.
7. Chi GY, Chen X, Shi Y. Effects of land use type on the profile distributions of soil amorphous iron and total phosphorus. Adv Sci Lett. 2011;4(1):156–8. https://doi.org/10.1166/asl.2011.1175.
8. China National Soil Census Office. Chinese soil. Beijing: Agric. Press China; 1998. (In Chinese)
9. Cho JY. Seasonal runoff estimation of N and P in a paddy field of central Korea. Nutr Cycl Agroecosystems. 2003;65(1):43–52. https://doi.org/10.1023/A:1021819014494.
10. Cleveland CC, Liptzin D. C:N: P stoichiometry in soil: is there a "redfield ratio" for the microbial biomass? Biogeochemistry. 2007;85(3):235–52. https://doi.org/10.1007/s10533-007-9132-0.
11. Dai Z, Liu G, Chen H, Chen C, Wang J, Ai S, Wei D, Li D, Ma B, Tang C, Brookes P, Xu J. Long-term nutrient inputs shift soil microbial functional profiles of phosphorus cycling in diverse agroecosystems. ISME J. 2020;14:757–70.
12. Daroub SH, Pierce FJ, Ellis BG. Phosphorus fractions and fate of phosphorus-33 in soils under plowing and no-tillage. Soil Sci Soc Am J. 2000;64(1):170–6. https://doi.org/10.2136/sssaj2000.641170x.
13. Drosos M, Nebbioso A, Mazzei P, Vinci G, Spaccini R, Piccolo A. A molecular zoom into soil Humeome by a direct sequential chemical fractionation of soil. Sci Total Environ. 2017;586:807–16. https://doi.org/10.1016/j.scitotenv.2017.02.059.
14. Fan H, Cai G, Wang H. Condition of soil erosion in phaeozem region of Northeast China. J Soil Water Conserv. 2004;18(2):66–70.
15. Fan H, Cai Q, Chen G, Cui M. Comparative study of the soil erosion and control in the three major black soil regions in the world. J Nat Resour. 2005;20(3):387–293.
16. Fink G, Alcamo J, Flörke M, Reder K. Phosphorus loadings to the world's largest lakes: sources and trends. Global Biogeochem Cy. 2018;32(4):617–34. https://doi.org/10.1002/2017GB005858.
17. Fu J, Wu Y, Wang Q, Hu K, Wang S, Zhou M, et al. Importance of subsurface fluxes of water, nitrogen and phosphorus from rice paddy fields relative to surface runoff. Agric Water Manag. 2019;213:627–35. https://doi.org/10.1016/j.agwat.2018.11.005.
18. Heckrath G, Brookes PC, Poulton PR, Goulding KWT. Phosphorus leaching from soils containing different phosphorus concentrations in the broadbalk experiment. J Environ Qual. 1995;24(5):904–10. https://doi.org/10.2134/jeq1995.00472425002400050018x.
19. Hedley MJ, Stewart JWB, Chauhan BS. Changes in inorganic and organic soil phosphorus fractions induced by cultivation practices and by laboratory incubations. Soil Sci Soc Am J. 1982;46(5):970–6. https://doi.org/10.2136/sssaj1982.03615995004600050017x.
20. Hesketh N, Brookes PC. Development of an indicator for risk of phosphorus leaching. J Environ Qual. 2000;29:105–10. https://doi.org/10.2134/jeq2000.00472425002900010013x.
21. IUSS Working Group WRB. World Reference Base for Soil Resources 2014, update 2015 International soil classification system for naming soils and creating legends for soil maps. World Soil Resources Reports No. 106. FAO, Rome. 2015.
22. Jalali M, Jalali M. Assessment risk of phosphorus leaching from calcareous soils using soil test phosphorus. Chemosphere. 2017;171:106–17. https://doi.org/10.1016/j.chemosphere.2016.12.042.
23. Koch M, Kruse J, Eichler-Lobermann B, Zimmer D, Willbold S, Leinweber P, et al. Phosphorus stocks and speciation in soil profiles of a long-term fertilizer experiment: evidence from sequential fractionation, P K-edge XANES, and P-31 NMR spectroscopy. Geoderma. 2018;316:115–26. https://doi.org/10.1016/j.geoderma.2017.12.003.
24. Kuzyakov Y, Xu X. Competition between roots and microorganisms for nitrogen: mechanisms and ecological relevance. New Phytol. 2013;198(3):656–69. https://doi.org/10.1111/nph.12235.
25. Kwon H, Koh DC, Cho BW, Jung YY. Nutrient dynamics in stream water and groundwater in riparian zones of a mesoscale agricultural catchment with intense seasonal pumping. Agric Water Manag. 2022;261:107336. https://doi.org/10.1016/j.agwat.2021.107336.
26. Li BG. Soil science and life (the 14th version). Beijing: Science press; 2019.
27. Li L, Meng QY. Reviews of phosphorus transport and transformation in soil under freezing and thawing actions. Ecol Environ Sci. 2013;22(6):1074–8.
28. Li M, Hu Z, Zhu X, Zhou G. Risk of phosphorus leaching from phosphorus-enriched soils in the Dianchi catchment. Southwestern China Environ Sci Pollut Res. 2015;22(11):8460–70. https://doi.org/10.1007/s11356-014-4008-z.
29. Li W, Zhang X, Wu B, Sun S, Chen Y, Pan W, Zhao D, Cheng S. A comparative analysis of environmental quality assessment methods for heavy metal-contaminated soils. Pedosphere. 2008;18(3):344–52. https://doi.org/10.1016/s1002-0160(08)60024-7.
30. Li Y, Wu J, Liu S, Shen J, Huang D, Su Y, et al. Is the C:N: P stoichiometry in soil and soil microbial biomass related to the landscape and land use in southern subtropical China? Global Biogeochem Cycles. 2012;26(4):1–14. https://doi.org/10.1029/2012GB004399.
31. Li Z, Zhang R, Liu C, Zhang R, Chen F, Liu Y. Phosphorus spatial distribution and pollution risk assessment in agricultural soil around the Danjiangkou reservoir. China Sci Total Environ. 2020;699:134417. https://doi.org/10.1016/j.scitotenv.2019.134417.
32. Liang B, Lehmann J, Sohi SP, Thies JE, O'Neill B, Trujillo L, et al. Black carbon affects the cycling of non-black carbon in soil. Org Geochem. 2010;41(2):206–13. https://doi.org/10.1016/j.orggeochem.2009.09.007.
33. Liu X, Zhang Y, Dong G, Hou G, Jiang M. Landscape pattern changes in the Xingkai Lake area, northeast China. Int J Environ Res Public Health. 2019. https://doi.org/10.3390/ijerph16203820.
34. Liu XY, Yan BX. Soil loss and food safety in Northeast China. Soil and Water conservation in China. 2009;1:17–9.
35. Lu RK. Analytical methods of soil agrochemistry. Beijing: Chinese Agric Sci Technol Press; 1999.
36. Maranguit D, Guillaume T, Kuzyakov Y. Land-use change affects phosphorus fractions in highly weathered tropical soils. CATENA. 2017;149:385–93. https://doi.org/10.1016/j.catena.2016.10.010.
37. McDowell R, Sharpley A, Brookes P, Poulton P. Relationship between soil test phosphorus and phosphorus release to solution. Soil Sci. 2001;166(2):137–49.
38. Murphy J, Riley JP. A modified single solution method for the determination of phosphate in natural waters. Anal Chim Acta. 1962;21:11–22.
39. Nair VD. Soil phosphorus saturation ratio for risk assessment in land use systems. Front Environ Sci. 2014;2(APR):1–4.
40. Olsen SR, Cole CV, Watanabe FS, Dean LA. Estimation of available phosphorus in soils by extraction with sodium bicarbonate. USDA Circ. 939. U.S. Gov. Print Office, Washington, DC; 1954.
41. Ouyang W, Shan Y, Hao F, Chen S, Pu X, Wang MK. The effect on soil nutrients resulting from land use transformations in a freeze-thaw agricultural ecosystem. Soil Tillage Res. 2013;132:30–8. https://doi.org/10.1016/j.still.2013.04.007.
42. Pierzynski GM, McDowell RW, Sims JT. Chemistry, cycling, and potential movement of inorganic phosphorus in soils. pp. 53–86. In: Sims J.T. and

Sharpley A.N. (ed.) Phosphorus: Agriculture and the environment. Agron. Monogr. 46. ASA, CSSA, and SSSA, Madison, WI; 2005.
43. Rodrigues M, Withers PJA, Soltangheisi A, Vargas V, Holzschuh M, Pavinato PS. Tillage systems and cover crops affecting soil phosphorus bioavailability in Brazilian Cerrado Oxisols. Soil Tillage Res. 2021. https://doi.org/10.1016/j.still.2020.104770.
44. Shen SM. Fertility of soil in China. Agric. Beijing: Press China; 1998.
45. Shi XZ, Yu DS, Warner ED, Pan XZ, Petersen GW, Gong ZG, Weindorf DC. Soil database of 1:1,000,000 digital soil survey and reference system of the Chinese genetic soil classification system. Soil Survey Horizons. 2004;45(4):129–36.
46. Sigurdarson JJ, Svane S, Karring H. The molecular processes of urea hydrolysis in relation to ammonia emissions from agriculture. Rev Environ Sci Bio/Tech. 2018;17:241–58. https://doi.org/10.1007/s11157-018-9466-1.
47. Sun X, Li M, Wang G, Drosos M, Liu F, Hu Z. Response of phosphorus fractions to land-use change followed by long-term fertilization in a sub-alpine humid soil of Qinghai-Tibet plateau. J Soils Sediments. 2019;19(3):1109–19. https://doi.org/10.1007/s11368-018-2132-y.
48. Tian HQ, Chen GS, Zhang C, Melillo JM, Hall CAS. Pattern and variation of C:N: P ratios in China's soils: a synthesis of observational data. Biogeochemistry. 2010;98(1–3):139–51. https://doi.org/10.1007/s10533-009-9382-0.
49. Tiessen HJ, Moir JO. Characterization of available P by sequential extraction, soil sampling and methods of analysis. Boca Raton: Lewis Publishers; 1993.
50. Turan V, Schröder P, Bilen S, Insam H, Fernández-Delgado JM. Co-inoculation effect of Rhizobium and *Achillea millefolium* L. oil extracts on growth of common bean (*Phaseolus vulgaris* L.) and soil microbial-chemical properties. Sci Rep. 2019;9(1):1–10. https://doi.org/10.1038/s41598-019-51587-x.
51. Wang W, Sardans J, Zeng C, Zhong C, Li Y, Peñuelas J. Responses of soil nutrient concentrations and stoichiometry to different human land uses in a subtropical tidal wetland. Geoderma. 2014;232–234:459–70. https://doi.org/10.1016/j.geoderma.2014.06.004.
52. Wang Z, Yang S, Zhao C, Bai J, Lou H, Chen K, et al. Assessment of non-point source total phosphorus pollution from different land use and soil types in a mid-high latitude region of china. Water. 2016;8(11):1–17. https://doi.org/10.3390/w8110505.
53. Xie Z, Li S, Tang S, Huang L, Wang G, Sun X, et al. Phosphorus leaching from soil profiles in agricultural and forest lands measured by a cascade extraction method. J Environ Qual. 2019;48(3):568–78. https://doi.org/10.2134/jeq2018.07.0285.
54. Xie Z, Yang X, Sun X, Huang L, Li S, Hu Z. Effects of biochar application and irrigation rate on the soil phosphorus leaching risk of fluvisol profiles in open vegetable fields. Sci Total Environ. 2021. https://doi.org/10.1016/j.scitotenv.2021.147973.
55. Xie Z, Zhang F, Ye C, Wang H, Wei W, Li C, Shi X. A tool (SPOLERC) to guide the evaluation of phosphorus leaching for agricultural soil by using the change point value in the Xingkai. Environ Sci. 2021. https://doi.org/10.1186/s12302-021-00572-1.
56. Xu G, Sun JN, Shao HB, Chang SX. Biochar had effects on phosphorus sorption and desorption in three soils with differing acidity. Ecol Eng. 2014;62:54–60. https://doi.org/10.1016/j.ecoleng.2013.10.027.
57. Xu X, Thornton PE, Post WM. A global analysis of soil microbial biomass carbon, nitrogen and phosphorus in terrestrial ecosystems. Glob Ecol Biogeogr. 2013;22(6):737–49. https://doi.org/10.1111/geb.12029.
58. Xu XZ, Xu Y, Chen SC, Xu SG, Zhang HW. Soil loss and conservation in the black soil region of Northeast China: a retrospective study. Environ Sci Policy. 2010;13(8):793–800. https://doi.org/10.1016/j.envsci.2010.07.004.
59. Yang W, Xia Y, Jiang X, Yan X. Influencing factors and estimation of total phosphorus runoff from farmlands in China. J Agro-Environ Sci. 2015;34(2):319–25.
60. Yao QZ, Du JT, Chen HT, Yu ZG. Particle-size distribution and phosphorus forms as a function of hydrological forcing in the Yellow River. Environ Sci Pollut Res. 2016;23(4):3385–98. https://doi.org/10.1007/s11356-015-5567-3.
61. Zhang L, Du J, Kong L, Yin C, Zhao Y, Hou Y, Li Q, Li S. Effects of phosphorus fertilizer application on yield, phosphorus absorption and utilization, soil phosphorus balance of spring maize in black soil region of Northeast China. J Northeast Agric Sci. 2020;45(5):38–42.
62. Zhang M, Li C, Li YC, Harris WG. Phosphate minerals and solubility in native and agricultural calcareous soils. Geoderma. 2014;232:164–71. https://doi.org/10.1016/j.geoderma.2014.05.015.
63. Zhang Y. Spatial distribution characteristics of root of rice and maize and cultivation regulation. Nanning: Guangxi University; 2013.
64. Zhang Z, Sun D, Li T, Qi Z, Du S, Wang B, Huang Y. Effects of water-saving irrigation and reducing phosphorus fertilizer on rice yield, phosphorus utilization and soil phosphorus balance in black soil. Trans Chin Soc Agric Eng. 2022;38(4):67–74.
65. Zhang ZJ, Zhang JY, He R, De WZ, Zhu YM. Phosphorus interception in floodwater of paddy field during the rice-growing season in TaiHu Lake Basin. Environ Pollut. 2007;145(2):425–33. https://doi.org/10.1016/j.envpol.2006.05.031.
66. Zhao XR, Li D, Kong J, Lin QM. Does biochar addition influence the change points of soil phosphorus leaching? J Integr Agric. 2014;13(3):499–506. https://doi.org/10.1016/S2095-3119(13)60705-4.
67. Zhao Y, Li YL, Yang F. Critical review on soil phosphorus migration and transformation under freezing-thawing cycles and typical regulatory measurements. Sci Total Environ. 2021. https://doi.org/10.1016/j.geoderma.2017.05.021.
68. Zheng L, Chen H, Wang YQ, Mao QG, Zheng MH, Su YR, et al. Responses of soil microbial resource limitation to multiple fertilization strategies. Soil Tillage Res. 2020. https://doi.org/10.1016/j.still.2019.104474.
69. Zheng S, Xia Y, Hu Y, Chen X, Rui Y, Gunina A, et al. Stoichiometry of carbon, nitrogen, and phosphorus in soil: effects of agricultural land use and climate at a continental scale. Soil Tillage Res. 2021;209:104903. https://doi.org/10.1016/j.still.2020.104903.
70. Zhou J, Zhang Y, Wu K, Hu M, Wu H, Chen D. National estimates of environmental thresholds for upland soil phosphorus in China based on a meta-analysis. Sci Total Environ. 2021;780:146677. https://doi.org/10.1016/j.scitotenv.2021.146677.
71. Zhang Y, de Vries W, Thomas BW, Hao X, Shi X. Impacts of long-term nitrogen fertilization on acid buffering rates and mechanisms of a slightly calcareous clay soil. Geoderma. 2017;305:92–99. https://doi.org/10.1016/j.geoderma.2017.05.021
72. Zhao X, Zhong X, Bao H, Li H, Li G, Tuo D, Lin Q, Brookes P.C. Relating soil P concentrations at which P movement occurs to soil properties in Chinese agricultural soils. Geoderma. 2007;142:237–44. https://doi.org/10.1016/j.geoderma.2007.07.012.

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