Phosphorus Sorption and Sedimentation in a Multipond System within a Headstream Agricultural Watershed

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The movement, transformation, and assimilation of phosphorus (P) in a watershed are often related to the physicochemical characteristics of the sediments of the water storage structures in the watershed. Field and laboratory experiments were conducted to study P sorption by pond sediments in a multipond system in a watershed of 6.9 km² in southeastern China. Being scattered in different ambient land uses, and receiving soil particles from various sources, the multipond system provides an excellent research example for P sorption as well as the relationship between pond properties and their location in a watershed. It was found that the pond sediments showed largely variable P sorption capacities in this watershed. The P sorption maximum (SMAX) ranged from 390.0 to 729.2 mg/kg, and equilibrium P concentration (EPC,) ranged from 0.007 to 0.023 mg/L. The mean degree of P saturation (DPS) of the pond sediments was only 9.5%. Due to the large annual delivery of sediments to the ponds by soil erosion, the pond sediments never reached saturation. The location of the ponds could influence the physicochemical properties of sediments and their sorption characteristics. For SMAX, the Hill pond value was significantly higher than that for the Village pond. For EPC,,, the result was reverse. The oxalate-extractable Fe (Ox-Fe) and KCl-extractable P (KCl-P) were the key factors that influenced P sorption of the pond sediments. Ox-Fe could explain 84% of the variability in SMAX and KCl-P could explain 82% of the variability in EPC,. The acidic and high Fe contents of laterite soil in southeastern China was the most important cause of the high P sorption capacity of the pond sediments.

Key words: phosphorus, sorption, sediment, pond, watershed

Introduction

The nonpoint-source pollutant (NPS) phosphorus from agricultural regions accounts for more than half of the phosphorus (P) load in many water bodies in southeastern China (Shu et al. 1996). Excessive P loading from watersheds into surface waters can result in accelerated eutrophication and algal bloom in receiving waters (Wetzel 1983). Agricultural practices in upland areas are a major contributor of NPS nutrient contamination to freshwater ecosystems (U.S. EPA 1992).

The generation and transport processes of P pollution from agricultural fields are complicated and depend on many factors (Novotny and Olem 1994; Sims et al. 1998). In watersheds, wetlands function as important links between the terrestrial and aquatic systems and have a significant impact on downstream water quality. Removal mechanisms for P in wetlands include sedimentation, sorption by surface sediments, microbial degradation, as well as plant uptake (Reddy et al. 1999). P storage in the above ground portion of aquatic vegetation is temporary, and long-term P retention is primarily controlled by the characteristics of bottom sediments and the P concentration in the water column (Reddy et al. 1996; Reddy and Sacco 1981).

Soil factors affecting P retention include the amount and type of clay, the amount of Fe, Al, and Ca compounds, and pH (Khalid et al. 1977; Richardson 1985; Gale et al. 1994). In general, double acid extractable Ca, Mg, and poorly crystalline forms of Fe and Al play important roles in the P sorption of the wetland sediments, and different wetland types have different controlling factors (Nair et al. 1998; Reddy et al. 1998; Pant and Reddy 2001).

The multipond system, a typical agricultural landscape in southeastern China, is a special kind of wetland system. Many ponds are scattered in agricultural fields and linked together with small ditches to form a pond network. The original purpose of the multipond system was water storage and irrigation. Studies in recent years indicated that the multipond system could effectively retain NPS P from agricultural watersheds through sedimentation, sorption, and plant uptake (Yin et al. 1993; Yan et al. 1998; Yin and Shan 2001). However, the P sorption mechanisms and the controlling factors in the pond sediments are poorly understood. Due to the importance of the pond sediments in the watershed, it is necessary to determine their sorption characteristics and to find out the relationship to the selected physicochemical properties of the pond sediments for better management. Being scattered in different ambient land uses, and receiving sediments from various sources, the ponds in different locations in a watershed probably provide different P sorption capacities. The objectives of the study were: (i) to determine the potential P sorption capacities of different pond sediments in the multipond system; and (ii) to establish the relationship between P...
sorption characteristics and selected physicochemical properties of the pond sediments.

**Materials and Methods**

**Study Sites**

This study was conducted in an agricultural watershed of Liuchahe in Chaohu Lake, a hypereutrophic lake, during 2003. Located on the northern bank of the lake, the Liuchahe watershed covers an area of 691.6 ha with a typical landscape of a multipond system (Tu et al. 1990). There are 193 artificial ponds with an average water depth of 1.5 m and surface area from 0.01 to 1 ha. These ponds can receive and store runoff from ambient land uses. In this process, the water, soil particles, and P can enter the ponds and be stored, accumulated, and sorbed.

In the watershed, there are five major types of land use: rice fields (45.2%), nonirrigated farmlands (25.7%), forests (15.7%), villages (7.2%) and ponds (6.2%) (see Table 1). According to the location of ponds and ambient land uses, the ponds can be classified into five types: village ponds (near or within villages); hill ponds (located in foothills); nonirrigation ponds (within nonirrigated farmlands); rice ponds (within rice fields); and river ponds (in the trunk of the Liuchahe river). The multipond system in the Liuchahe watershed and the experimental pond sites are shown in Fig. 1. Two village ponds, four rice ponds, one hill pond, one nonirrigation pond, and one river pond were selected for study (see Fig. 1).

**Field Experiment**

A field experiment was conducted to study the variations in P concentration when runoff passed through different ponds. Water samples were taken at the inlets, outlets, and centers of the nine ponds in 4 h intervals for heavy rainfall events and in 30 min intervals for light rainfall events when surface runoff was generated. After the end of the rainfall event, water samples were taken from the centers of the ponds at 0, 1/4, 1/2, 1, 2, 4, 8, and 16 days. All water samples were collected from 10 cm below the water surface and brought to the laboratory for chemical analysis. Samples were analyzed for total phosphorus (TP) by the method of peroxodisulfate oxidation (Ebina et al. 1983), and for soluble reactive phosphorus (SRP) by the ascorbic acid method (Murphy and Riley 1962).

**Sorption Experiment**

**Sample collection.** Triplicate sediment cores (ca. 12-cm depth) were obtained from each of the nine ponds at representative locations using polyvinyl chloride pipe. The sediment samples were then immediately homogenized, air dried, and crushed to pass through a 20-mesh sieve to form the initial soil sample. The samples were placed in sealed plastic bags and kept at 25 ± 3°C in a darkroom until chemical analysis.

![Fig. 1. The multipond system in Liuchahe watershed and the experimental pond sites.](https://iwaponline.com/wqrj/article-pdf/44/3/243/229594/wqrjc0440243.pdf)
Extraction and analytical methods. The pH value was measured in a 1:2 (wt:vol) sediment:water mixture after equilibrating for 30 min using a composite electrode. KCl-extractable P (KCl-P) in the sediment was measured after extracting the leachate from the sediment with a 0.01 M KCl solution at a sediment:water ratio of 1:20 after 24 h using the molybdenum blue method (Murphy and Riley 1962). Oxalate-extractable Fe, Al, and P (Ox-Fe, Ox-Al, and Ox-P) were determined by the oxalate method as described by Khalid et al. (1977). Another sample of air-dried sediment was mixed with 0.1 M H₂C₂O₄ + 0.175 M (NH₄)₂C₂O₄ (pH 3.5) at a sediment:water ratio of 1:50 by shaking on an end-over-end mechanical shaker for 4 h. The suspensions were then filtered through a 0.45-μm membrane filter. The filtrates were analyzed for Fe, Al, and P by ICP (inductively coupled plasma) spectrometry. MehlichI-extractable Ca, Mg, and P were determined by the method described by Reddy et al. (1998). The air-dried sediment was mixed with a weak acid mixture (0.05 M HCl + 0.0125 M H₂SO₄) at a sediment:water ratio of 1:5 to determine Mehlich-extractable Ca, Mg, and P. After 5 min of shaking, the suspension was filtered through a 0.45-μm membrane filter, and filtrates were analyzed for Ca and Mg using atomic absorption spectrophotometry. The total organic carbon (TOC) content of sediments was determined by wet oxidation (SSSA and ASA 1996). Total thickness of sediment was directly measured from the sediment profiles when ponds were dried to the bottom in winter. Average annual sedimentation was calculated through total thickness of the sediment divided by the years since last dredging (Yan et al. 1998).

Sorption isotherms. One gram of sediment was placed in a 50-mL plastic centrifuge tube and treated with 20 mL of 0.01 M KCl solution containing several levels of P (0, 0.05, 0.1, 0.5, 1.0, 3, 10, 50, 100 mg/L) as KH₂PO₄. Two drops of chloroform were added to each tube during the equilibration period. Then, the tubes were sealed and shaken for a 24 h in the dark using an end-over-end mechanical shaker at 25 ± 3°C. Following equilibration, sediment samples were allowed to settle for about 1 h, and then the supernatant was removed and filtered through a 0.45-μm membrane filter. Filtrates were determined for SRP using the ascorbic acid method (Murphy and Riley 1962). Phosphate not recovered in solution was considered the amount sorbed by the sediments.

Sorption parameter calculations. Sorption parameters were calculated using linear, Langmuir, and Freundlich isotherms as follows (Reddy et al. 1998):

\[ S = S' + S_0 \] (1)
\[ S' = K_d \times C_s - S_0 \] (2)

where \( S \) = total sorbed P in sediments (mg/kg), \( S' \) = amount of added P sorbed by sediments (mg/kg), \( S_0 \) = initial or native sorbed P in sediment (mg/kg), \( C_s \) = solution P concentration measured after a 24 h equilibration (mg/L), and \( K_d \) = the linear coefficient (L/kg).

The equilibrium P concentration (\( EPC_{eq} \)) is defined as the concentration of P in a solution, where no net P sorption or desorption occurs, and is equal to the value of \( C_s \) when \( S' = 0 \):

\[ EPC_{eq} = S_0 / K_d \] (3)

Langmuir isotherm:

\[ C_d / S = 1 / k + S_{max} / C_d / S_{max} \] (4)

where \( S_{max} \) = P sorption maximum (mg/kg) and \( k \) = a constant related to P bonding energy (L/mg).

Freundlich isotherm:

\[ S = K_f \cdot C_d^n \] (5)

where \( K_f \) = P sorption coefficient with sorption energy (L/kg), and \( n \) = a correction factor.

Curve fitting, statistical analysis, and multiple regression analysis were done using the Origin software (OriginLab Corporation) and SAS software (SAS Institute 1990). The univariate model was a completely randomized one-way analysis of variance. The Duncan’s test was used to identify significant differences among pond types. Unless otherwise noted, all significant differences were at a probability of 0.05 or less.

Results and Discussion

P Trapping by the Multipond Systems During Rainfall Events

<table>
<thead>
<tr>
<th>Land use</th>
<th>Area (ha)</th>
<th>TP in surface soil (mg/kg)</th>
<th>TP in surface runoff (mg/L)</th>
<th>SS in surface runoff (g/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Village</td>
<td>49.6</td>
<td>535</td>
<td>1.632</td>
<td>8.68</td>
</tr>
<tr>
<td>Hill forest</td>
<td>108.7</td>
<td>130</td>
<td>0.046</td>
<td>1.43</td>
</tr>
<tr>
<td>Nonirrigated farmland</td>
<td>177.7</td>
<td>337</td>
<td>0.141</td>
<td>6.75</td>
</tr>
<tr>
<td>Rice field</td>
<td>312.3</td>
<td>278</td>
<td>0.070</td>
<td>1.52</td>
</tr>
</tbody>
</table>

*Values measured in the rainfall event on April 29, 2002; SS = suspended solids.*
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Inlet and outlet P concentrations of five major types of ponds were determined in five main rainfall events in 2002. Their mean values are shown in Fig. 2(i). The concentrations of TP and SRP at the outlets were significantly lower than those at the inlets for all types of ponds. Previous studies indicated the removal by aquatic plants in the multipond system was little (Yan et al. 1998). The decrease of TP and SRP concentrations at the outlets was mainly caused by the accumulation of soil particles as well as the sorption of SRP by the pond sediments, which are similar to the physicochemical mechanisms for P removal reported by Yin and Shan (2001). The changes of TP and SRP concentrations with time in the rice pond 2 after the end of a rainfall event are shown in Fig. 2(ii).

The data indicate the influence of sedimentation and sorption of ponds. The concentrations of TP and SRP in pond water decreased rapidly with time and reached equilibrium in two days after the rainfall events ended. Because of sedimentation, the ratio of SRP to TP increased with time and reached the maximum after two days. After two days, sorption became dominant. Soil particles and particulate P in the pond water could settle on the bottom of the ponds rapidly and form new pond sediments. The new sediment layers could further sorb SRP from the pond water. Sedimentation and sorption were two main aspects in a study of the retention mechanism of the multipond system described earlier by Yin and Shan (2001).

The multipond system could effectively trap solid particles from the surrounding fields due to the huge water storage volume. When storm runoff passed through the ditch-pond system, large amounts of solid particles settled in the different ponds. Annual sediment accumulation could reach 14.4 x 10^6 kg, and form a 3-cm sediment layer (Yin and Shan 2001). Different pond types had different accumulative rates of sediments due to the different export loadings of the ambient land uses (Yan et al. 1998).

The ponds scattered in the watershed can significantly delay the water flow process and reduce runoff velocity (Yin and Shan 2001). The massive ponds and vegetated small ditches in the fields also constitute the transport and storage network for water and soil particles. Thus the ponds scattered in the various land uses form multipond sedimentation. Through this process, over 80% of the eroded soil particles can be retained in the ponds, and a large amount of nutrients can also be retained in the ponds and sorbed by the sediments. The multipond system provides an effective way for controlling the loss of water and soil in watersheds.

Physicochemical Properties of the Pond Sediments

Sediment pH, TOC, and concentration of various forms of P are given in Table 2. In the multipond system, sediment pH values ranged from 4.3 to 7.5 and indicated strong acidity, except for the river pond. The lowest pH value was found in the hill pond (<4.5) and the highest was in the river pond (>7.0). The acidity of sediments probably came from the acidity of laterite soil in southeastern China and may also be contributed to by needles from the pine trees.

The highest TOC was found in the sediments of the hill pond and village pond. In the surface soils of hill forest in the watershed, there is abundant humus from decayed rests of pine trees (Yan et al. 1998). Massive organic and animal manure accumulate on the village surface soils. In the hill pond and village pond, greater TOC concentration probably reflected C accumulation during the storm in runoff from hill forest and village surface soils, similar to the findings of Reddy et al. (1998).

Ammonium oxalate-oxalic acid extracts amorphous forms of Fe and Al, and preferentially attacks selected poorly crystalline oxides and hydroxides of Fe (Reddy

Fig. 2. (i) TP and SRP concentrations of pond inlet and outlet in five rainfall events in 2002. (ii) The changes of TP and SRP concentrations with time in the rice pond 2 after the end of the rainfall event.
In the watershed, the Ox-Fe contents were very high (>4,700 mg/kg); the highest content was found in the sediments of the hill pond. There was significant correlation between Ox-Fe and organic C contents \((r = 0.60, p < 0.01)\). High Ox-Fe concentrations in the sediments were also associated with high organic C contents (Reddy et al. 1998).

In the pond sediments, Ox-Al concentrations were generally higher than 1,100 mg/kg. The highest values were in the nonirrigation pond, and no significant difference was found among other pond types. In general, Ox-Al concentrations represented less than 20% of the Ox-Fe content, suggesting that Ox-Fe was probably the dominant cation regulating P sorption in the pond sediments in the multipond system (Reddy et al. 1998).

Mehlich-I-Ca ranged from 4,895 to 14,820 mg/kg (Table 2). The Ca concentrations in the river pond sediments were significantly higher (>13,000 mg/kg) than those in the other pond types (<8,000 mg/kg). High Ca concentrations in the river pond sediments reflected higher pH and Ca accumulation in the stream. There was significant correlation between Ca concentration and pH in the pond sediments \((r = 0.93, p < 0.001)\). Low Ca concentrations in hill pond were probably due to the acidic pH conditions of the sediments. Mehlich I-Mg contents ranged from 336 to 474 mg/kg. The lowest Mg content was found in the hill pond sediments, but no significant difference was found between the other pond types.

**P Sorption Characteristics of the Pond Sediments**

The relationship between P sorbed by the sediment and equilibrium P concentrations in different pond sites is shown in Fig. 3. Pant and Reddy (2001) found that every P compound in sediments could maintain equilibrium independently with the P in solution and behaved as a mixture of homogeneous surfaces. For pond sediments, P sorption behaviour could also be well described by the three models \((r^2 > 0.95)\): linear model, Langmuir model, and Freundlich model (see Table 3 and Fig. 3).

In the multipond system, ponds were scattered among the different land uses. The different solid particles in runoff from different land uses settled in the ponds and composed the different types of pond sediments. The sorption characteristics of pond sediments were greatly influenced by the ambient land uses. Studies had indicated that the location of wetlands in watersheds could also influence their characteristics (De Laney 1995). The P sorption maxima, equilibrium concentrations, and degree of P saturation in different types of ponds are given in Fig. 3.

\( S_{\text{max}} \) P sorption maxima \((S_{\text{max}})\) ranged from 390.0 to 729.2 mg/kg for the multipond system sediments (see Table 3). Generally, sediments from the hill pond exhibited the highest \( S_{\text{max}} \) values, and the village pond sediments had lowest \( S_{\text{max}} \), suggesting a P loading effect (greater in the village soils and smaller in the hill forest soils) on sorption capacity. High NPS P loadings could reduce P sorption capacities of the receiving water column (Reddy et al. 1980; Nair et al. 1998). Correlation analysis suggested that there was negative correlation between \( S_{\text{max}} \) and Mehlich-I-P \((r = -0.48, p < 0.05)\), and positive correlation between \( S_{\text{max}} \) and Ox-Fe \((r = 0.92, p < 0.001)\).
Thus, the P sorption maxima in the pond sediment was probably controlled by two factors: interior factors (Ox-Fe contents in sediments) and exterior factors (P loading into the ponds). The hill pond had the highest Ox-Fe content, while P loading into the hill pond was relatively lower (see Table 1), which resulted in the highest $S_{\text{max}}$ in the hill pond sediments. The village pond had the lowest Ox-Fe contents in sediments and the highest P loading. Therefore, the $S_{\text{max}}$ was lowest in the village pond.

$E_{\text{PCo}}$. Generally, $E_{\text{PCo}}$ was very low in the multipond system (<0.05 mg/L), which reflected a strong buffer intensity of the pond sediments. The $E_{\text{PCo}}$ of the village pond was significant higher than those of the other ponds (see Table 3). In opposition to $S_{\text{max}}$, $E_{\text{PCo}}$ was generally higher in the sediments of the heavily P-impacted intensive components (village pond) than in the less P-impacted components (hill pond). Correlation analysis and multiple linear regression analysis indicated that $E_{\text{PCo}}$ was mainly affected by KCl-P of the pond sediments. High influent P loading could increase the equilibrium P concentration of pond sediments. No significant relationship was found between $E_{\text{PCo}}$ and Ox-Fe, Ox-Al, MehlichI-Ca, nor Mg.

A similar result was reported by Reddy et al. (1998).

**DPS.** The degree of P saturation (DPS) is defined as the ratio of sorbed P to the P sorption maxima (Breeuwsma et al. 1995). There is no standardized method to estimate DPS because it is difficult to rapidly extract sorbed P (Sallade and Sims 1997). Combined determination of oxalate extractable Fe, Al, and P is commonly adopted to make this estimation (Van der Zee et al. 1988; Sallade and Sims 1997). In this experiment, the mole ratio of oxalate extractable P, Fe, and Al was also used to estimate DPS: $DPS = [\text{Ox-P}] / [\alpha(\text{Ox-Fe + Ox-Al})] \times 100\%$. $\alpha(\text{Ox-Fe + Ox-Al})$ is the P sorption capacity and $\alpha$ is the scaling factor. Reported values of $\alpha$ in the literature have ranged from 0.34 to 0.68, and we used 0.5 as proposed by Maguire et al. (2001a, 2001b) and Schoumans (2000).

The range of DPS in the pond sediments was 5.0 to 21.9%. In the multipond system, all the pond sediments still had a large amount of sorption sites. The highest DPS values were found in the sediments of the village pond and it meant the village pond sediments were closer to saturation than the other pond types for the high P loadings from the villages. The hill pond, nonirrigation pond, and rice pond sediments with 5% of DPS were far from saturation. To prevent adverse effects on surface and ground water quality, sediments exceeding a DPS of 25% (threshold value of critical saturation) should not receive further P inputs (Breeuwsma et al. 1995). In the multipond system, the DPS of all pond sediments were generally far below this threshold, except for the village pond sediments. To protect surface and groundwater quality, the sediments of the village pond should be dredged up in time.

**Fig. 3.** Relationship between P sorbed by the sediments and the equilibrium P concentration according to the pond types.

**TABLE 3. P sorption characteristics of the pond sediments.**

<table>
<thead>
<tr>
<th>Pond</th>
<th>$S_{\text{max}}$ (mg/kg)</th>
<th>$E_{\text{PCo}}$ (mg/L)</th>
<th>$K_d$ (L/kg)</th>
<th>$K_f$ (L/kg)</th>
<th>DPS (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hill</td>
<td>729.2 ± 89.8a</td>
<td>0.007 ± 0.002a</td>
<td>766.1 ± 127.7a</td>
<td>88.1 ± 11.5a</td>
<td>5.0 ± 0.2c</td>
</tr>
<tr>
<td>Nonirrigation</td>
<td>587.8 ± 14.5ab</td>
<td>0.008 ± 0.003a</td>
<td>440.4 ± 110.6bc</td>
<td>69.8 ± 6.7a</td>
<td>5.1 ± 0.4c</td>
</tr>
<tr>
<td>Rice</td>
<td>559.7 ± 21.9ab</td>
<td>0.010 ± 0.003a</td>
<td>529.4 ± 51.9ab</td>
<td>71.0 ± 4.7a</td>
<td>5.3 ± 0.3c</td>
</tr>
<tr>
<td>River</td>
<td>542.8 ± 14.4ab</td>
<td>0.011 ± 0.005a</td>
<td>147.9 ± 26.2c</td>
<td>43.5 ± 2.7b</td>
<td>9.3 ± 0.4b</td>
</tr>
<tr>
<td>Village</td>
<td>390.0 ± 54.1c</td>
<td>0.023 ± 0.005b</td>
<td>198.9 ± 70.4c</td>
<td>45.0 ± 9.3b</td>
<td>21.9 ± 0.4a</td>
</tr>
</tbody>
</table>

*Values are means ± 1 SE and means with different letters are significantly different at $p = 0.05$.

* $S_{\text{max}}$ = P sorption maximum; $E_{\text{PCo}}$ = equilibrium P concentration; $K_d$ = the linear coefficient; $K_f$ = P sorption coefficient with sorption energy; DPS = degree of P saturation.

* $DPS = [P_{\text{sat}}/(0.5[Fe_{\text{sat}} + Al_{\text{sat}}])]$. 

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Relationship between Sediment Properties and P Sorption

Significant correlations were observed between the P sorption maximum ($S_{\text{max}}$) and pond sediment properties (see Table 4). The most striking relationship existed between $S_{\text{max}}$ and Ox-Fe, which represented the amorphous and some poorly crystalline forms of Fe. The amorphous form of Fe is highly reactive and tends to retain P by sorption onto surfaces and by forming insoluble complexes and precipitates (McCallister and Logan 1978). Though amorphous forms of Al are highly reactive, and can also sorb P on surfaces (McCallister and Logan 1978), no significant relationship was found between sorption parameters and amorphous forms of Al ($p > 0.1$). In the sediments of the multipond system, Ox-Al content was less than 20% of the Ox-Fe content. Thus, the key factor controlling the P sorption capacity was probably the Ox-Fe, which is similar to the findings of Reddy et al. (1998). The correlation between Ox-Fe and $S_{\text{max}}$ was $r = 0.92$ ($p < 0.001$), whereas the correlation between Ox-Fe + Al and $S_{\text{max}}$ was $r = 0.93$ ($p < 0.001$). Adding Al to the regression did not significantly improve the relationship. This result strongly suggested that Fe was the dominant regulator of P sorption in the sediments of the multipond system.

A significant correlation between $S_{\text{max}}$ and TOC ($r = 0.65, p < 0.001$) suggested that organic matter could also play an important role in P sorption (see Table 4). In the multipond system, high Fe concentrations in the pond sediments were associated with high organic C contents ($r = 0.60, p < 0.01$). Organic matter could complex Fe and indirectly affect P sorption (Traina et al. 1986; Börling et al. 2001). The role of organic matter in augmenting P sorption in some soils has often been attributed to the association of the soil organic matter with hydrous sesquioxides forming Fe and Al chelates, which provide an active surface for P sorption. Positive correlation between organic matter content in the wetland sediment and P sorption has also been shown by previous studies (Sanyal et al. 1993; Reddy et al. 1998).

Previous studies have indicated that ferric hydroxides, ferric oxyhydroxides, and amorphous Fe oxides and hydroxides play a significant role in P sorption in acidic soils. (Reddy et al. 1998). Multiple regression analysis of P sorption and selected independent variables (Table 5) suggested that $S_{\text{max}}$ was adequately predicted from Ox-Fe in pond sediments. About 84% of the P sorption variability could be accounted for by Ox-Fe. In general, the sum of Ox-Fe + Al was slightly better than Ox-Fe alone ($r^2 = 0.86$) (see Table 5). Thus, it is concluded that in the multipond system, Ox-Fe alone is sufficient to predict P sorption capacities of the pond sediments. Therefore, when new ponds are excavated to control the P output, the metal contents of the soils should be considered. High Fe content soil, such as the laterite in southeastern China, can provide a high P-retention effect.

Table 4 indicated that $EPC_0$ was independent of the physicochemical properties of the pond sediments, but had significant positive correlations with KCl-P concentrations. A multiple linear regression analysis indicated that KCl-P was a good indicator of $EPC_0$ and could explain 82% of the variability in $EPC_0$ (see Table 5). KCl-P is defined as the soluble P or available P for plant growth (Gale et al. 1994). Generally, higher KCl-P content was found in the pond sediments that had higher influent P loading. With the P loading increasing, the sorption threshold of the pond sediments increased and the buffer intensity decreased, and only the storm runoff that had higher P concentrations could be sorbed.

### Table 4. Correlation coefficients of P sorption parameters and associated physicochemical properties of pond sediments

<table>
<thead>
<tr>
<th>Parameter</th>
<th>$S_{\text{max}}$</th>
<th>$K_d$</th>
<th>$K_f$</th>
<th>$EPC_0$</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
<td>-0.44*</td>
<td>-0.68***</td>
<td>-0.63***</td>
<td>NS</td>
</tr>
<tr>
<td>Mehlich-P</td>
<td>-0.48*</td>
<td>NS</td>
<td>NS</td>
<td>0.54**</td>
</tr>
<tr>
<td>KCl-P</td>
<td>-0.38*</td>
<td>-0.48*</td>
<td>NS</td>
<td>0.91***</td>
</tr>
<tr>
<td>Mehlich-Mg</td>
<td>-0.43*</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
<tr>
<td>Mehlich-Ca</td>
<td>NS</td>
<td>-0.53**</td>
<td>-0.49*</td>
<td>NS</td>
</tr>
<tr>
<td>Ox-Fe</td>
<td>0.92***</td>
<td>0.67***</td>
<td>0.81***</td>
<td>NS</td>
</tr>
<tr>
<td>Ox-Al</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
<tr>
<td>Ox-Fe + Al</td>
<td>0.93***</td>
<td>0.68***</td>
<td>0.81***</td>
<td>NS</td>
</tr>
<tr>
<td>TOC</td>
<td>0.65***</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
</tbody>
</table>

1 The number of samples assessed = 27.
2 $S_{\text{max}}$ = P sorption maximum; $K_d$ = the linear coefficient; $K_f$ = P sorption coefficient with sorption energy; $EPC_0$ = equilibrium P concentration;
3 Asterisks represent values that are significant at the 0.05(*), 0.01(**), and 0.001(***), and 0.001 levels.
4 NS = not significant.
5 Mehlich-P = Mehlich-extractable P; KCl-P = KCl-extractable P; Mehlich-Mg = Mehlich-extractable Mg; Mehlich-Ca = Mehlich-extractable Ca; Ox-Fe = oxalate-extractable Fe; Ox-Al = oxalate-extractable Al; Ox-Fe + Al = oxalate-extractable Fe + Al.
Liu et al.

TABLE 5. Multiple linear regression analysis of P sorption parameters and selected independent variables.

<table>
<thead>
<tr>
<th>Regression equation</th>
<th>$R^2$</th>
<th>SE</th>
</tr>
</thead>
<tbody>
<tr>
<td>$S_{\text{max}} = -297.6 + 0.13 \text{(Ox-Fe)}$</td>
<td>0.84</td>
<td>66.6</td>
</tr>
<tr>
<td>$K_T = 29.6 + 0.015 \text{(Ox-Fe)}$</td>
<td>0.66</td>
<td>12.4</td>
</tr>
<tr>
<td>$EPC_0 = 0.0014 + 0.046 \text{(KCl-P)}$</td>
<td>0.82</td>
<td>0.0035</td>
</tr>
<tr>
<td>$S_{\text{max}} = -446.9 + 0.13 \text{(Ox-Fe + Al)}$</td>
<td>0.86</td>
<td>60.8</td>
</tr>
<tr>
<td>$K_T = -45.9 + 0.015 \text{(Ox-Fe + Al)}$</td>
<td>0.67</td>
<td>12.2</td>
</tr>
</tbody>
</table>

*a The number of samples assessed = 27.

Potential P Sorption Capacity

In the Liuchahe watershed, assuming that P retention was limited to the upper 12 cm of the pond sediments, the P retention capacity of the pond sediments could also be determined by converting the value of $S_{\text{max}}$ to an area basis. The data presented in Table 6 show the P sorption maxima of pond sediments expressed on an areal basis. The hill pond showed the highest retention capacity and reached 902 kg/ha because of its highest $S_{\text{max}}$. Though the $S_{\text{max}}$ of the river pond was comparatively lower, its P-sorption capacity was only less than that of the hill pond and more than the capacity of other pond types because it had the highest sediment content (Yan et al. 1998). In the Liuchahe watershed, the mean potential retention capacity of the multipond system was 665.5 kg/ha (Table 6). Since the nonirrigation pond had the highest annual accumulation rate, the annually renewed P sorption capacity was significantly greater than the capacity of the other pond types (see Table 6).

In the Liuchahe watershed, due to the crisscrossed ditches in the nonirrigated farmlands and rice lands, as well as the artificial disturbance by farmers in the precipitation and irrigation events, it was difficult to calculate the influent P loading of the different pond types. However, the total P loading into the pond system could be roughly calculated through the surface runoff volume and average nutrient concentration. The result for the average annual P loading into the multipond system was 21.6 kg/ha. Subsurface runoff only accounts for 5 to 8% of the total runoff in one rainfall event (Tu et al. 1990). Thus, the subsurface runoff was not included in the influent loading. At this annual input rate, the sediments of the multipond system could continue to remove P from NPS runoff for about 30 years. But annual renewed sorption capacity [The sorption capacity of the sediments which enter ponds from ambient land uses every year = annual mass accumulation (kg) × $S_{\text{max}}$/ area (ha)] of pond sediments could reach 98.6 kg/ha (see Table 6). Therefore, in general, pond sediments probably never reached sorption saturation and could continue to remove P from NPS runoff until the ponds were full of sediments about 50 years later. The multipond system provides a type of long-term P sorption.

TABLE 6. Potential P sorption capacity and annual renewed capacity in the Liuchahe watershed

<table>
<thead>
<tr>
<th>Type</th>
<th>Area (ha)</th>
<th>Annual Accumulation rates (m)</th>
<th>Gravity (x10³ kg/m³)</th>
<th>Annual mass Accumulation (kg)</th>
<th>Average retention capacity a (kg/ha)</th>
<th>Annual renewed capacity b (kg/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hill pond</td>
<td>3.8</td>
<td>0.012</td>
<td>1.03</td>
<td>$0.47 \times 10^6$</td>
<td>901.9</td>
<td>90.2</td>
</tr>
<tr>
<td>Nonirrigation pond</td>
<td>1.2</td>
<td>0.058</td>
<td>1.01</td>
<td>$0.70 \times 10^6$</td>
<td>709.4</td>
<td>342.9</td>
</tr>
<tr>
<td>Rice pond</td>
<td>36.3</td>
<td>0.016</td>
<td>0.96</td>
<td>$5.63 \times 10^6$</td>
<td>644.5</td>
<td>85.9</td>
</tr>
<tr>
<td>River pond</td>
<td>1.1</td>
<td>0.025</td>
<td>1.20</td>
<td>$0.33 \times 10^6$</td>
<td>781.6</td>
<td>162.8</td>
</tr>
<tr>
<td>Village pond</td>
<td>1.9</td>
<td>0.040</td>
<td>1.08</td>
<td>$0.81 \times 10^6$</td>
<td>498.8</td>
<td>166.3</td>
</tr>
<tr>
<td>Liuchahe watershed</td>
<td>44.3</td>
<td>—</td>
<td>—</td>
<td>$7.90 \times 10^6$</td>
<td>665.5</td>
<td>98.6</td>
</tr>
</tbody>
</table>

a Potential P sorption capacity of 0 to 12 cm sediments.

b Annual renewed capacity = annual mass accumulation (kg)$\times S_{\text{max}}$/area (ha).
Conclusions

In this multipond system, many ponds are scattered in a headstream agricultural watershed, which provides conveniences to research the different effects and characteristics of the different ponds in the watershed.

Due to the huge water storage volume, the multipond system could effectively settle the solid particles from the storm runoff. The average annual sediments could form a 3-cm sediment layer. The greatest accumulation rate was in the village pond and nonirrigation pond (Table 1). Although the hill pond had the lowest accumulation, they had the greatest P retention capacity ($S_{max}$) and a lower DPS.

In the headstream watershed in the south of China, the amorphous and poorly crystalline form of Fe is key in controlling the pond sediments' retention capability and spatial variability, as shown by highly significant correlation ($r > 0.9$). No significant relationship was found between adsorption capacities and amorphous and poorly crystalline forms of Al, or double acid extractable Ca or Mg. When new ponds are excavated to control the P output, the metal content of the soils should be considered. In acidic soil, such as laterite in the south of China, soil with high Fe content can provide a higher P-retention effect.

Agricultural wetlands that receive large volumes of P inputs can become saturated with P. But, due to the large annual delivery of sediments to the ponds by soil erosion in the multipond system, the pond sediments could not reach the P sorption saturation until the ponds were full of sediments about 50 years later. The multipond system provides a type of long-term P sorption.

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References


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