

Use of Drinking Water Treatment Residuals as a Potential Best Management Practice to Reduce Phosphorus Risk Index Scores

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ABSTRACT

The P risk index system has been developed to identify agricultural fields vulnerable to P loss as a step toward protecting surface water. Because of their high Langmuir phosphorus adsorption maxima (P_{\max}), use of drinking water treatment residuals (WTRs) should be considered as a best management practice (BMP) to lower P risk index scores. This work discusses three WTR application methods that can be used to reduce P risk scores: (i) enhanced buffer strip, (ii) incorporation into a high soil test phosphorus (STP) soil, and (iii) co-blending with manure or biosolids. The relationship between WTR P_{\max} and reduction in P extractability and runoff P was investigated. In a simulated rainfall experiment, using a buffer strip enhanced with 20 Mg WTR ha^{-1} , runoff P was reduced by from 66.8 to 86.2% and reductions were related to the WTR P_{\max} . When 25 g kg^{-1} WTR was incorporated into a high STP soil of 315 mg kg^{-1} determined using Mehlich-3 extraction, 0.01 M calcium chloride-extractable phosphorus ($\text{CaCl}_2\text{-P}$) reductions ranged from 60.9 to 96.0% and were strongly ($P < 0.01$) related to WTR P_{\max} . At a 100 g kg^{-1} WTR addition, Mehlich 3-extractable P reductions ranged from 41.1 to 86.7% and were strongly ($P < 0.01$) related to WTR P_{\max} . Co-blending WTR at 250 g kg^{-1} to manure or biosolids reduced $\text{CaCl}_2\text{-P}$ by >75%. The WTR P_{\max} normalized across WTR application rates ($P_{\max} \times \text{WTR application}$) was significantly related to reductions in $\text{CaCl}_2\text{-P}$ or STP. Using WTR as a P risk index modifying factor will promote effective use of WTR as a BMP to reduce P loss from agricultural land.

APPROXIMATELY 2.3 million Mg of manure phosphorus (P) is generated in the United States annually, according to the National Program 206 annual report (USDA, 1999). Historically, manure application recommendations have been based on crop nitrogen requirements resulting in a two- to threefold excess in P application (Parry, 1998). The USEPA Water Quality Inventory (USEPA, 2000) reported that >33% of U.S. rivers, lakes, wetlands, and estuaries had degraded water quality (USEPA, 2000) largely due to agricultural practices. As a result of increased concern about surface water degradation, many states have developed manure application limits or guidelines to restrict manure, biosolids, or P fertilizer application (Sharpley et al., 2003; Shober and Sims, 2003; USEPA and USDA, 2003). The Natural Resources Conservation Service (NRCS) Practice Standard 590-Nutrient Management, which focuses on land application of manure, has been revised to include P-based guidelines. There are 238 000 concentrated

animal feeding operations (CAFOs) in the United States (USDA, 1999). As a result of a joint effort by the USDA and the USEPA (USEPA and USDA, 2003), CAFOs will be regulated as point-source polluters and will be required to implement a comprehensive nutrient management plan by 2008. These plans address manure handling, storage, and land application issues. In addition, CAFOs will be required to have a national pollutant discharge elimination system permit. Achieving the typical effluent limitation guideline of $<1.0 \text{ mg P L}^{-1}$ in runoff is expected to impose a substantial burden on many CAFOs.

Phosphorus buildup in soil from land-applied biosolids is also coming under scrutiny. Shober and Sims (2003) reviewed the status of U.S. regulations pertaining to the land application of biosolids. Of the 51 states and territories that responded, 24 reported having regulations or guidelines restricting land application of biosolids based on soil or biosolids P levels. Of the 24 reporting states, 13 use soil test phosphorus (STP) limits or thresholds as criteria for guidelines or regulations. Land application of biosolids is stopped once the threshold STP value is reached. In most cases, the threshold STP level adopted is considerably higher than the agronomic critical STP value (Shober and Sims, 2003).

Many strategies are being used to reduce P transport to surface water, including consideration of new best management practices (BMPs). One possible BMP is to remove dissolved P from runoff water and leachate by the land application of P-sorbing materials such as aluminum (Al) or iron (Fe) oxide. Drinking water treatment residuals (WTRs) may be effective due to their high P-sorbing capacity. Drinking water treatment residuals are often rich in amorphous Fe or Al oxides due to the use of Fe or Al salts as coagulants during drinking water treatment. Each treatment plant uses different source water and different treatment chemicals and processes, producing WTR with different chemical compositions and P sorption capacities. Dayton et al. (2003) examined the WTR components thought to contribute to WTR P sorption. Using batch equilibration, P sorption isotherms were generated for 21 Al-based WTRs, and the linearized Langmuir P_{\max} was determined. Dayton et al. (2003) found a significant ($r^2 = 0.69$, $P < 0.01$) relationship between P_{\max} and acid ammonium oxalate-extractable aluminum (Al_{ox}) in WTR. To further optimize the relationship between WTR Al_{ox} and P_{\max} , Dayton and Basta (2005) reported a method using Al_{ox} to predict WTR P_{\max} . Using 18 Al-based WTRs, they modi-

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Abbreviations: Al_{ox} , acid ammonium oxalate-extractable aluminum; BMP, best management practice; $\text{CaCl}_2\text{-P}$, 0.01 M calcium chloride-extractable phosphorus; CAFO, concentrated animal feeding operation; DRP, dissolved reactive phosphorus; Fe_{ox} , acid ammonium oxalate-extractable iron; P_{\max} , Langmuir phosphorus adsorption maximum; STP, soil test phosphorus; WTR, drinking water treatment residual.

fied the acid ammonium oxalate extraction method (McKeague and Day, 1993) for a more accurate determination of WTR Al_{ox} and modified experimental conditions used to generate P adsorption isotherms to determine WTR P_{max} . The improved linear relationship between WTR Al_{ox} and P_{max} ($r^2 = 0.91$, $P < 0.001$) could provide a useful tool for determining WTR P_{max} without the onus of the multipoint batch equilibrations necessary for the Langmuir model.

Several studies have suggested that using WTR as a P sorbent may be an effective BMP to reduce risk of P loss. Surface application of WTR has been successful at removing dissolved P from runoff water (Basta and Storm, 1997; Dayton et al., 2003; Gallimore et al., 1999; Hausteine et al., 2000; Peters and Basta, 1996). Incorporating WTR into a high P soil has been shown to reduce P solubility and P leaching (Codling et al., 2000; Elliott et al., 2002; Novak and Watts, 2004; O'Connor et al., 2002; Peters and Basta, 1996). Co-blending WTR with a manure or biosolids before land application reduces the solubility of P in the manure or biosolids (Codling et al., 2000; Elliott et al., 2002; Ippolito et al., 1999).

The objective of this work is to discuss three WTR land application methods, using five WTRs that can be used to reduce risk of P loss to surface water. The three methods include surface application of WTR to a buffer strip (enhanced buffer strip) to remove dissolved reactive P from surface runoff water, WTR incorporation into a high STP soil to reduce soil P solubility and STP, and finally co-blending WTR with organic soil amendments (poultry litter and biosolids) to reduce the P solubility of the amendment. The relationship between WTR P_{max} and the effectiveness of WTR as a P sorbent is also investigated.

MATERIALS AND METHODS

Material Collection and Preparation

Five Al-based WTRs were selected from the 18 WTRs characterized previously for Al_{ox} and P_{max} (Dayton and Basta, 2005) to provide a range of P_{max} . Residuals were air-dried and crushed to $<250 \mu m$. Slaking of WTR occurs naturally in the field, due to weathering, but differences in slaking during short-term incubations or runoff studies may be a source of variability. Therefore, WTRs were crushed and sieved to mitigate the effect of slaking. Tonti silt loam (fine-loamy, mixed,

active, mesic Typic Fragiudults) with a high soil test P concentration of 315 mg kg^{-1} determined using Mehlich-3 extraction was used in this experiment. This soil was chosen because it is a benchmark soil in an area of high poultry production and had received annual poultry litter application for the past 20 yr. The soil was air-dried and passed through a 2-mm sieve. The poultry litter, collected from the same farm, was air-dried and passed through a 2-mm sieve. Anaerobically digested biosolids was air-dried and passed through a 2-mm sieve. Properties of the poultry litter and biosolids are presented in Table 1.

Water Treatment Residual Characterization

Noncrystalline reactive Al (Al_{ox}) and Fe (Fe_{ox}) oxide content of the materials was determined by a modified acid ammonium oxalate extraction (McKeague and Day, 1993) with subsequent analysis by inductively coupled plasma atomic emission spectroscopy (ICP-AES). The oxalate solution to soil ratio was increased from the recommended 40:1 to 100:1 to account for the high oxide content of the WTR (Dayton and Basta, 2005). The pH of all materials was measured in a 1:1 material to deionized water solution using a glass electrode (Thomas, 1996). Soil test P was determined by Mehlich-3 extraction (Mehlich, 1984). Soil or WTR (2 g) was combined with 20 mL Mehlich-3 extraction solution, equilibrated for 5 min, and filtered ($0.45 \mu m$). Soluble P, estimated as $CaCl_2$ -P (0.01 M), was determined by shaking 1 g material with 25 mL of 0.01 M $CaCl_2$ for 2 h. Soil extracts were filtered ($0.45 \mu m$) and analyzed by colorimetry using the modified Murphy and Riley method (Kuo, 1996).

Sorption isotherms were generated by batch equilibration to determine each WTR P_{max} . Water treatment residuals were shaken with standard P solutions (1 g WTR to 10 mL P solution ranging from 0 to 3.5 g P L^{-1} , as KH_2PO_4) for 6 d (Dayton and Basta, 2005). The isotherm data were fit to the linearized Langmuir model (Sparks, 1995), and the P_{max} values were determined.

Soil Incorporation Study

Residual (0, 25, 50, and 100 g kg^{-1}) was mixed with the high P soil. All soil treatments were replicated three times. The soil-WTR blends were incubated at $30^\circ C$ until there was no change in $CaCl_2$ -P or Mehlich 3-extractable P (4 wk). To ensure aerobic conditions incubation samples were subjected to wet-dry cycles by watering and mixing well twice a week. Soil-WTR blends were sampled weekly and $CaCl_2$ -P and STP using Mehlich-3 extraction (Mehlich, 1984) were determined on air-dried samples. Soil-WTR extracts were filtered ($0.45 \mu m$) and analyzed by colorimetry using the modified Murphy

Table 1. Characterization results of drinking water treatment residuals (WTRs), Tonti soil, poultry litter, and biosolids.

| WTR | Oxalate-extractable | | | M3† | $CaCl_2$ -P‡ | P_{max} § | pH |
|----------------|---------------------|-------|-------|--------------|--------------|---------------|------|
| | Al | Fe | P | | | | |
| | g kg^{-1} | | | mg kg^{-1} | | g P kg^{-1} | |
| A | 29.1 | 3.52 | 2.20 | 24.0 | BDL¶ | 18.6 | 7.24 |
| B | 86.6 | 0.223 | 0.279 | 1.50 | BDL | 23.7 | 7.51 |
| C | 95.8 | 1.90 | 1.15 | 6.00 | BDL | 27.1 | 7.04 |
| D | 114 | 12.3 | 1.46 | 7.00 | BDL | 29.3 | 6.98 |
| E | 130 | 1.84 | 1.58 | 6.50 | BDL | 32.6 | 6.93 |
| Tonti soil | 1.11 | 1.01 | 0.590 | 315 | 13.6 | NA# | 5.69 |
| Poultry litter | 0.621 | 2.51 | 19.4 | NA | 2054 | NA | 6.80 |
| Biosolids | 8.90 | 10.67 | 15.3 | NA | 62.5 | NA | 6.61 |

† Mehlich 3-extractable phosphorus.

‡ 0.01 M calcium chloride-extractable phosphorus.

§ Langmuir phosphorus adsorption maximum of WTR.

¶ Below detection limit of 0.1 mg kg^{-1} .

Not applicable.

and Riley method (Kuo, 1996). There was no change in either Mehlich 3- or CaCl_2 -extractable P in the control soils as a result of incubation.

Runoff Study

Small plot studies using simulated rainfall were used to evaluate the ability of WTR to reduce runoff P losses. Small-scale box plots were established in a greenhouse. Each plot (0.5 m wide, 1 m long, and 10 cm deep) was filled with 75 kg of Tonti silt loam and packed to a depth of 10 cm. Each box had fifteen 6.4-mm drainage holes, and landscape weed cloth was placed in the bottom of the boxes to minimize soil loss. Each box was planted with a mix of perennial ryegrass (*Lolium perenne* L.), fescue (*Festuca arundinacea* Schreb.), and bermudagrass [*Cynodon dactylon* (L.) Pers.]. Grass was allowed to grow for 1 mo and was clipped each week.

Poultry litter was applied at 8.8 Mg ha⁻¹ to the upslope 75% of the plot area, and air-dried WTR was applied in a buffer strip to the remaining downslope area at 0, 5, 10, or 20 Mg ha⁻¹. The slope of the boxes was set at 5%; simulated rain was supplied at a rate of 70 mm h⁻¹ by a solenoid-operated, variable-intensity rainfall simulator, and runoff water was collected for 30 min after it commenced. Runoff water for the entire 30-min simulation from each plot was collected and mixed in one 100-L container. Each runoff sample was filtered with a 0.45- μm membrane filter. Dissolved reactive P was determined by colorimetry using the modified Murphy and Riley method (Kuo, 1996).

Co-Blending Study

Water treatment residuals were blended with poultry litter at 0, 100, 250, and 500 g kg⁻¹, for a total of 50 g, and with biosolids at 0, 125, 250, and 375 g kg⁻¹, for a total of 100 g and incubated at 25°C for 60 d (30 d after no change in extractable P). To ensure aerobic conditions incubation samples were subjected to wet-dry cycles by watering and mixing well twice a week. Incubated blends were sampled weekly and CaCl_2 -P was determined on air-dried samples. The CaCl_2 -P extracts were filtered (0.45 μm) and analyzed by colorimetry using the modified Murphy and Riley method (Kuo, 1996). There was no change in CaCl_2 -P in the control poultry litter or biosolids during the incubation.

RESULTS

Characterization

Table 1 gives an overview of characteristics related to P adsorption and P status for the materials used in this study. In the five Al-based WTRs (A–E), Al_{ox} ranged from 29.1 to 130 g kg⁻¹ with a mean of 91.1 g kg⁻¹. The Al_{ox} range of the WTR selected for study falls within the typical range (13.9–165 g kg⁻¹) for Al-based WTRs (Dayton and Basta, 2005). The Al_{ox} ranged from 20 to more than 100 times greater in the WTR than in the Tonti soil, the poultry litter, or the biosolids (Table 1). The Fe_{ox} was considerably lower than Al_{ox} , ranging from 0.223 to 12.3 g kg⁻¹ with a mean of 3.96 g kg⁻¹. The Fe_{ox} content of the WTR was generally comparable to the Fe_{ox} content of the Tonti soil and the poultry litter. The Fe_{ox} content of the biosolids, however, was generally greater than the Fe_{ox} content of the WTR (Table 1). The P_{max} values ranged from 18.6 to 32.6 g P

Table 2. Drinking water treatment residual (WTR) Langmuir phosphorus adsorption maximum (P_{max}), and percent reductions in runoff P, 0.01 M calcium chloride-extractable phosphorus (CaCl_2 -P), or Mehlich 3-extractable P as a result of the three WTR land application methods.

| | WTR | | | | |
|---|--|------|------|------|------|
| | A | B | C | D | E |
| P_{max} , g P kg ⁻¹ WTR | 18.6 | 23.7 | 27.1 | 29.3 | 32.6 |
| | % P reduction | | | | |
| WTR level | Method: WTR enhanced buffer strip | | | | |
| Mg ha ⁻¹ | Runoff P | | | | |
| 5 | 2.90 | 8.60 | 32.5 | 37.6 | 32.5 |
| 10 | 24.8 | 40.4 | 46.2 | 48.7 | 50.5 |
| 20 | 70.9 | 66.8 | 78.7 | 84.4 | 86.2 |
| | Method: Soil incorporation | | | | |
| g kg ⁻¹ | 0.01 M CaCl_2 -P | | | | |
| 25 | 60.9 | 82.5 | 88.4 | 94.5 | 96.0 |
| 50 | 73.6 | 91.5 | 94.4 | 97.1 | 97.6 |
| 100 | 83.9 | 95.5 | 97.2 | 92.3 | 98.6 |
| | Mehlich 3-extractable P | | | | |
| 25 | 6.93 | 27.5 | 26.4 | 25.7 | 28.4 |
| 50 | 21.2 | 32.7 | 44.6 | 57.6 | 55.4 |
| 100 | 41.1 | 48.2 | 66.3 | 86.5 | 86.7 |
| | Method: Co-blended with poultry litter | | | | |
| g kg ⁻¹ | 0.01 M CaCl_2 -P | | | | |
| 100 | 58.1 | 77.6 | 64.2 | 72.7 | 44.2 |
| 250 | 82.9 | 91.5 | 93.8 | 81.8 | 89.4 |
| 500 | 95.5 | 96.5 | 95.3 | 81.6 | 98.4 |
| | Method: Co-blended with biosolids | | | | |
| g kg ⁻¹ | 0.01 M CaCl_2 -P | | | | |
| 125 | 78.0 | 67.4 | 56.9 | 58.8 | 71.6 |
| 250 | 82.1 | 76.6 | 77.4 | 85.7 | 84.1 |
| 375 | 85.6 | 89.7 | 86.2 | 85.7 | 83.1 |

kg WTR⁻¹ (Table 1). The WTR pH ranged from 6.93 to 7.51.

Runoff Study

Mean dissolved reactive phosphorus (DRP) in runoff from the control plots that received poultry litter but not WTR was 31.1 mg P L⁻¹. Addition of WTR as an enhanced buffer strip reduced mean DRP in runoff water by 2.90 to 37.6% for the 5 Mg ha⁻¹ WTR addition, 24.8 to 50.5% for the 10 Mg ha⁻¹ addition, and 66.8 to 86.2% for the 20 Mg ha⁻¹ WTR addition (Table 2). The WTR P_{max} normalized across WTR application rates ($P_{\text{max}} \times \text{WTR application}$) was highly correlated ($r^2 = 0.89$, $P < 0.001$) with reductions in runoff DRP (Fig. 1).

Soil Incorporation

Addition of WTR to the high P Tonti soil reduced both soluble (0.01 M CaCl_2) and Mehlich 3 (STP)-extractable P values at all WTR application levels. The CaCl_2 -P in the control soil was 13.6 mg kg⁻¹. After 4 wk of incubation, CaCl_2 -P was reduced by 60.9 to 96% for the 25 g kg⁻¹ WTR addition, by 73.6 to 97.6% for the 50 g kg⁻¹ WTR addition, and by 83.9 to 98.6% for the 100 g kg⁻¹ WTR addition (Table 2). The WTR P_{max} normalized across WTR application rates ($P_{\text{max}} \times \text{WTR application}$) was significantly ($r^2 = 0.52$, $P < 0.01$) related to reductions in CaCl_2 -P (Fig. 2A).

The initial Mehlich 3-extractable P (315 mg P kg⁻¹) was reduced by 6.93 to 28.4% at the 25 g kg⁻¹ WTR

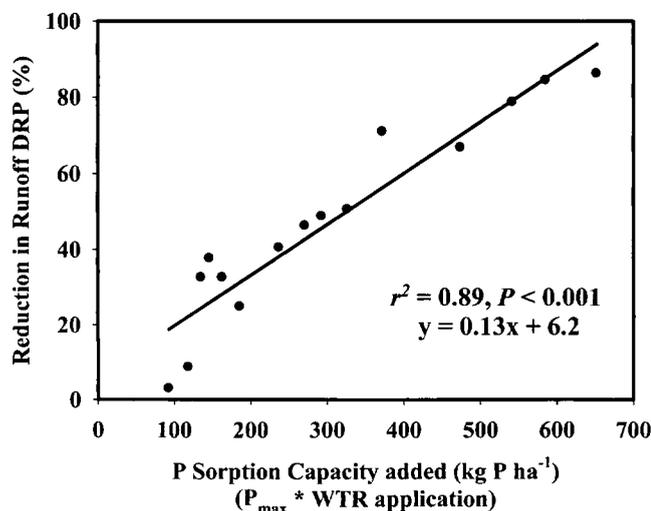


Fig. 1. Relationship between reduction in runoff P and drinking water treatment residual (WTR) Langmuir phosphorus adsorption maxima (P_{\max}) normalized across WTR application of 5, 10, and 20 Mg ha^{-1} ($P_{\max} \times$ WTR application).

addition, by 21.2 to 57.6% at the 50 g kg^{-1} WTR addition, and by 41.1 to 86.7% at the 100 g kg^{-1} WTR addition (Table 2). The WTR P_{\max} normalized across WTR application rates ($P_{\max} \times$ WTR application) was highly correlated ($r^2 = 0.86$, $P < 0.001$) to reductions in Mehlich 3-extractable P (Fig. 2B).

Co-Blending Water Treatment Residual

Addition of WTR to poultry litter substantially reduced $CaCl_2$ -P (Table 2). The poultry litter used in this study had 2.05 g kg^{-1} $CaCl_2$ -P. After an 8 wk incubation, P solubility was reduced by 44.2 to 77.6% at the 100 g kg^{-1} WTR addition, by 81.8 to 93.8 at the 250 g kg^{-1} WTR addition, and by 81.6 to 98.4% at the 500 g kg^{-1} WTR addition. The WTR P_{\max} normalized across WTR application rates ($P_{\max} \times$ WTR application) was highly correlated ($r^2 = 0.88$, $P < 0.001$) to reductions in poultry litter $CaCl_2$ -P (Fig. 3).

Addition of WTR also reduced biosolids $CaCl_2$ -P (Table 2) which was initially 62.5 mg P kg^{-1} . After an 8-wk incubation, $CaCl_2$ -P was reduced by 56.9 to 78% at the 125 g kg^{-1} WTR addition, by 76.6 to 85.7% at the 250 g kg^{-1} WTR addition, and by 83.1 to 89.7% at the 375 g kg^{-1} WTR addition. The WTR P_{\max} normalized across WTR application rates ($P_{\max} \times$ WTR application) was highly correlated ($r^2 = 0.77$, $P < 0.001$) to reductions in biosolids $CaCl_2$ -P in (Fig. 4).

DISCUSSION

Addition of WTR as an enhanced buffer strip greatly reduced DRP in runoff water and the reduction was related to WTR P_{\max} . The greatest WTR application (20 Mg ha^{-1}) reduced runoff P by 66.8 to 86.2% from the control runoff level of 31.1 mg P L^{-1} . Enhancing a buffer strip with this level of WTR should be easily accomplished with commonly used agricultural spreading equipment. The amount of WTR required will depend on the desired amount of excessive P to be sorbed.

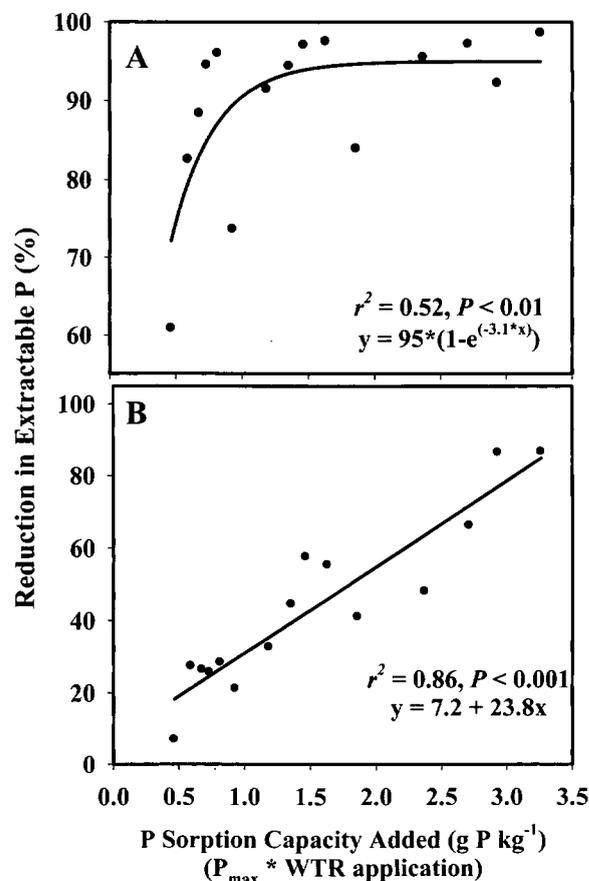


Fig. 2. Relationship between (A) percent reduction in 0.01 M calcium chloride-extractable phosphorus ($CaCl_2$ -P) and drinking water treatment residual (WTR) Langmuir phosphorus adsorption maxima (P_{\max}) normalized across WTR application ($P_{\max} \times$ WTR application) for incorporation of WTR at 25, 50, and 100 g kg^{-1} and (B) percent reduction in Mehlich 3-extractable P and P_{\max} normalized across WTR application ($P_{\max} \times$ WTR application) for incorporation of WTR at 25, 50, and 100 g kg^{-1} .

Accurately determining WTR P_{\max} is essential to effectively using WTR to achieve target P reductions. Other studies have shown reductions in runoff DRP (Basta and Storm, 1997; Dayton et al., 2003; Gallimore et al., 1999; Haustein et al., 2000; Peters and Basta, 1996) and that the reductions in runoff DRP were related to WTR P_{\max} (Dayton et al., 2003).

Incorporating WTR into a high P soil greatly reduces P solubility. Many studies have shown reductions in STP due to WTR addition (Codling et al., 2000; Elliott et al., 2002; Novak and Watts, 2004; O'Connor et al., 2002; Peters and Basta, 1996). The WTR application levels used in this study (25, 50, and 100 g kg^{-1}) correspond to 55, 110, and 220 Mg ha^{-1} applications, respectively, assuming a 15-cm depth of incorporation. Concentration of P in runoff is strongly related to P accumulation in the upper 2 cm of soil (Sharpley and Halvorson, 1994). Application levels of WTR could be substantially reduced while still achieving target P reductions, with reductions in the desired depth of incorporation from 15 cm to 2 cm depth.

At the low WTR application of 25 g kg^{-1} , $CaCl_2$ -P was reduced by 60.9 to 96%. With the exception of the

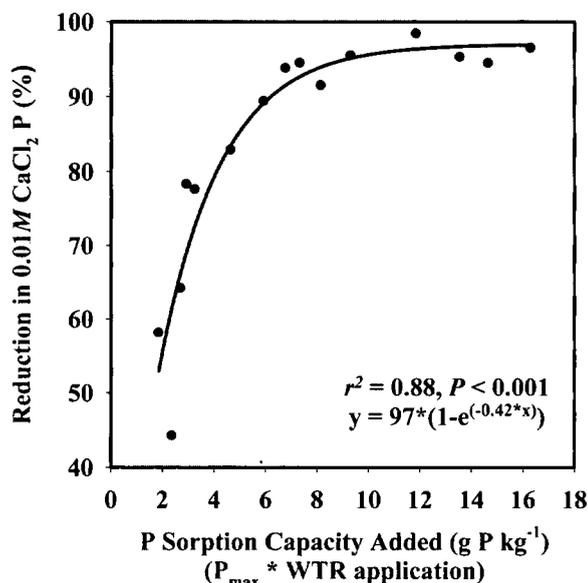


Fig. 3. Relationship between reduction 0.01 M calcium chloride-extractable phosphorus ($\text{CaCl}_2\text{-P}$) and drinking water treatment residual (WTR) Langmuir phosphorus adsorption maxima (P_{max}) normalized across WTR application ($P_{\text{max}} \times \text{WTR application}$) for WTR co-blended with poultry litter at 100, 250, and 500 g WTR kg^{-1} .

low P_{max} WTR, higher WTR applications (50 and 100 g kg^{-1}) resulted in only small additional $\text{CaCl}_2\text{-P}$ reductions. Using a high P-sorbing WTR, significant reductions in $\text{CaCl}_2\text{-P}$ can be achieved at WTR applications less than 25 g kg^{-1} .

Reductions in Mehlich 3-extractable P were smaller than reductions in $\text{CaCl}_2\text{-P}$ at the same WTR application. At the low WTR application of 25 g kg^{-1} (55 Mg ha^{-1}), reductions in STP ranged from 6.93 to 28.4%. Greater WTR applications (50 and 100 g kg^{-1}) resulted in significant additional reductions in STP. Mehlich 3 is a strong acid fluoride-containing solution used to measure P sufficiency and make fertilizer recommendations.

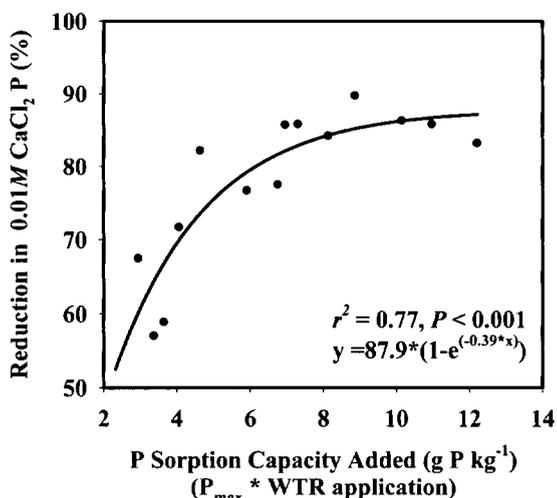


Fig. 4. Relationship between reduction 0.01 M calcium chloride-extractable phosphorus ($\text{CaCl}_2\text{-P}$) and drinking water treatment residual (WTR) Langmuir phosphorus adsorption maxima (P_{max}) normalized across WTR application ($P_{\text{max}} \times \text{WTR application}$) for WTR co-blended with biosolids at 125, 250, and 375 g WTR kg^{-1} .

This extractant aggressively attacks and dissolves amorphous (Al_{ox}) surfaces releasing adsorbed P. Because WTR is rich in amorphous Al-oxide, a strong acid fluoride-containing extractant may substantially underestimate reductions in the risk of P loss if WTR is used as a BMP. Phosphorus is strongly adsorbed to Al_{ox} structures in WTR. This WTR P adsorption phenomenon results in a great deal of adsorption-desorption hysteresis with little desorption of P occurring in soil solution or other dilute salt aqueous solutions (Dayton and Basta, 2005). Results here suggest using water or a 0.01 M CaCl_2 extractant rather than a strong acid fluoride extractant like Mehlich 3 will more accurately depict reductions in the risk of P loss from a high P soil. Therefore, $\text{CaCl}_2\text{-P}$ and not an overly aggressive extractant, designed to measure soil fertility, such as Mehlich 3, is better suited to determine adequate WTR application to achieve target P reductions. We have included both measures here, because many state P risk index schemes use a strong acid fluoride-containing extractant and not water or $\text{CaCl}_2\text{-P}$ for initial screening to characterize the risk of P loss (Sharpley et al., 2003; Shober and Sims, 2003).

Co-blending WTR with a high P organic material (poultry litter or biosolids) substantially reduced $\text{CaCl}_2\text{-P}$. Co-blending WTR with manure or biosolids before land application has been shown to reduce soluble P in manure or biosolids (Codling et al., 2000; Elliott et al., 2002; Ippolito et al., 1999). In our study, WTR addition to a poultry litter or biosolids at ≤ 250 g kg^{-1} reduced $\text{CaCl}_2\text{-P}$ by $>75\%$. The reductions were similar with either poultry litter or biosolids at the same rate. However, $\text{CaCl}_2\text{-P}$ in the poultry litter (2.05 g kg^{-1}) was much greater than in the biosolids (62.5 mg kg^{-1}). Therefore, the $\text{CaCl}_2\text{-P}$ reduction in poultry litter was much greater than in biosolids. Greater P solubility in the poultry litter than in the biosolids is consistent with the substantially lower Al_{ox} and Fe_{ox} content in the poultry litter than in the biosolids. Because the WTR Al_{ox} is substantially higher than in either the biosolids or poultry litter, WTR Al_{ox} dominates the WTR-litter or biosolids system and significantly reduces $\text{CaCl}_2\text{-P}$ at less than 25% WTR blending rates (250 g kg^{-1} WTR).

In response to the continued degradation of surface water, the NRCS in each state has been mandated to choose a P-based nutrient management strategy. One of these approaches is establishing a P risk index system. Lemunyon and Gilbert (1993) first developed the P risk index to identify agricultural fields vulnerable to P loss. All site characteristics contributing to P loss are considered, and weighting factors are applied to account for differences in each characteristic's relative contribution to P loss. For example, in the Pennsylvania P risk index (Sharpley et al., 2003; Weld et al., 2003), the total P risk score is based on the product of site P source factors and transport factors. The source factors include STP, fertilizer or manure-biosolids application rate, method, and timing. Also considered is the availability (i.e., solubility) of the manure or biosolids P. Transport factors include site runoff potential, subsurface drainage potential, distance from surface water, and modified connec-

tivity (i.e., buffer strip or grassed waterway) (Sharpley et al., 2003).

Several state P risk indices (Sharpley et al., 2003) allow modification of source and transport factors to account for management practices when calculating P risk index scores. Use of WTR should be considered as a modifying factor when calculating P risk index scores. For example, incorporation of WTR into a high P soil (Table 2) can reduce STP, possibly to below the P risk index screening level of 200 mg kg⁻¹ STP used in Pennsylvania or the 150 mg kg⁻¹ STP used in Ohio (Sharpley et al., 2003). Surface application of WTR to pasture or as an enhanced buffer strip at the edge of fields can remove DRP from runoff water (Table 2). The Pennsylvania P risk index allows modification of transport factors, such as the presence of a buffer strip and distance to surface water. Because buffer strips enhanced with WTR will provide additional DRP transport reductions, perhaps additional adjustments should be considered when calculating P risk scores. Treated manure or biosolids is given a lower score than untreated in the Pennsylvania P risk index (Sharpley et al., 2003). Co-blending of WTR with an organic material reduces the P solubility of the organic amendment before land application (Table 2) and should be considered as a modifying factor when calculating P risk index scores. Reducing the solubility of manure or biosolids P by co-blending with WTR before land application allows a producer to take advantage of the nitrogen, micronutrient, and organic carbon content of the manure or biosolids without increasing the P risk. Inclusion of WTR as a P risk index modifying factor will promote effective use of WTR as a BMP to reduce P loss from agricultural land and CAFOs.

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