Managing Biosolids Runoff Phosphorus Using Buffer Strips Enhanced with Drinking Water Treatment Residuals

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Vegetated buffers strips typically have limited ability to reduce delivery of dissolved phosphorus (DP) from agricultural fields to surface waters. A field study was conducted to evaluate the ability of buffer strips enhanced with drinking water treatment residuals (WTRs) to control runoff P losses from surface-applied biosolids characterized by high water-extractable P (4 g kg−1). Simulated rainfall (62.4 mm h−1) was applied to grassed plots (3 m × 10.7 m including a 2.67 m downslope buffer) surface-amended with biosolids at 102 kg P ha−1 until 30 min of runoff was collected. With buffer strips top-dressed with WTR (20 Mg ha−1), runoff total P (TP = 2.5 mg L−1) and total DP (TDP = 1.9 mg L−1) were not statistically lower (a = 0.05) compared to plots with unamended grass buffers (TP = 2.7 mg L−1; TDP = 2.6 mg L−1). Although the applied WTR had excess capacity (Langmuir P maxima of 25 g P kg−1) to sorb all runoff P, kinetic experiments suggest that sheet flow travel time across the buffers (~30 s) was insufficient for significant P reduction. Effective interception of dissolved P in runoff water by WTR-enhanced buffer strips requires rapid P sorption kinetics and hydrologic flow behavior ensuring sufficient runoff residence time and WTR contact in the buffer. Substantial phosphate-adsorbent contact opportunity may be more easily achieved by incorporating WTRs into P-enriched soils or blending WTRs with applied P sources.

PHOSPHORUS transport to sensitive water bodies can stimulate eutrophication and reduce water quality. A variety of best management practices (BMPs) are available to minimize the impacts of agricultural P loss on aquatic systems. Practices can be divided into source BMPs that aim to minimize the amount of P available to runoff and drainage, and transport BMPs that limit P movement pathways from the field to the receiving stream (Sharpley et al., 2006). Changing the method of application (e.g., subsurface placement) and establishment of riparian buffers are examples of source and transport BMPs. Vegetated filter strips are widely used to buffer valuable aquatic resources from nutrients lost via runoff from adjacent agricultural fields. Some state P indexing tools provide credits for buffer strips in developing nutrient management planning strategies. The effectiveness of vegetated buffers can, however, be quite variable. The minimum buffer width necessary to protect a water body depends on the type and condition of vegetation (Palone and Todd, 1998). Subsurface flow and channelized surface flow can bypass the renovative capacity of buffers and reduce their effectiveness (Sharpley et al., 2006).

Vegetated filter strips are most effective in removing P when the majority of the total P is sediment bound (Abu-Zreig et al., 2003). However, buffer zones are not considered an effective best management practice for decreasing dissolved P (DP) losses (Schmitt et al., 1999; Sharpley et al., 2006). Applying P-immobilizing materials (alum, water treatment residuals [WTRs], fly ash, gypsum) to edge-of-stream soils is one possible means of reducing soluble P losses to aquatic systems (Penn and Bryant, 2006).

Of possible P adsorbents, WTRs have received significant attention because they represent an environmentally benign municipal byproduct that has traditionally been landfilled, discharged to sanitary sewers, or lagooned for long-term storage (Elliott et al., 1990). Because Al and Fe salts are used to remove color and turbidity from surface water supplies, the residuals produced in drinking water treatment contain reactive hydrous Al and Fe oxides with substantial P-fixing ability (Zhang et al., 2006). The relative effectiveness of WTRs in reducing soluble P depends on their chemical and physical properties which, in turn, depend on source water characteristics, treatment operational parameters.

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Abbreviations: BMP, best management practice; BPR, biological phosphorus removal; DP, dissolved phosphorus; PP, particulate phosphorus; PSC, phosphorus source coefficient; PSI, phosphorus saturation index; RDP, runoff dissolved phosphorus; TDP, total dissolved phosphorus; TP, total phosphorus; WEP, water-extractable phosphorus; WTR, water treatment residual.
storage conditions, and the method and form of application to the soil.

Methods employing WTRs to reduce P loss from agricultural land fall into three categories: (i) incorporation of WTRs into soil with excessive soil test P (STP); (ii) Co-blending of WTRs with the nutrient source (biosolids, manures); and (iii) surface application of WTRs to capture and remove dissolved P from runoff (Zhang et al., 2006). Several studies have reported that WTRs are effective in reducing STP and decreasing P release from soils (Elliott et al., 1990; Codling et al., 2000; Eaton and Sims, 2003). Ippolito et al. (1999) reported that mixing WTR and biosolids at an 8:1 ratio adsorbed all soluble biosolids P.

While some researchers (Eaton and Sims, 2003) have suggested that WTR as soil amendments in buffer strips should decrease the risk of nonpoint source P pollution of surface waters, supporting field studies are few. Gallimore et al. (1999) studied the use of WTR-enhanced buffers to reduce runoff soluble P from pastures treated with poultry litter. One WTR significantly reduced runoff total dissolved P (TDP), while a second WTR with lower P sorption capacity (as indicated by oxalate-extractable Al) failed to provide statistically lower TDP. Although a potentially useful management practice, no state P indices currently allow adjustment for transport reductions due to buffer strips enhanced with WTRs (Dayton and Basta, 2005b). Quantitative estimates of the reduction of P risk associated with WTR type and application method are needed (Zhang et al., 2006).

The overall purpose of the research was to evaluate the use of buffers strips amended with WTRs as a management practice for reducing P loss from surface-applied biosolids. The biosolids were generated from a biological P removal (BPR) wastewater treatment process; an increasingly important type of treatment to meet stringent effluent nutrient discharge limits (Daigger, 2003). Among biosolids types, BPR materials produce the highest runoff P when land applied (Penn and Sims, 2002; Elliott et al., 2005). Thus, management strategies will be needed to sustain land-based recycling of BPR materials under P-based nutrient management policies. Soil incorporation, widely recognized as a P loss reduction strategy, was also evaluated for comparison. The specific goal was to address the practical use of WTR-enhanced buffer strips to reduce soluble P loss from agricultural land.

### Materials and Methods

#### Phosphorus Source and WTR Materials

A BPR biosolids, Largo pellets (LP), was used due to its relatively high runoff potential compared to other biosolids (Elliott et al., 2005). Largo pellets are a heat-dried, class-A biosolids generated at the Largo, Florida wastewater treatment plant. Di-ammonium phosphate fertilizer (DAP, 18-46-0), obtained from a local distributor, was included for comparison. The WTR used in this study was acquired from the Manatee County water treatment plant in Bradenton, Florida. This plant employs a conventional coagulation-flocculation process using alum. The Bradenton WTR has a reported P adsorptive capacity of greater than 6000 mg P kg⁻¹ WTR (Makris et al., 2004).

Phosphorus sources and the WTR were analyzed for total P, Fe, Al, Ca, and Mg by inductively coupled plasma atomic emission spectroscopy (ICP) (USEPA Method 6010) following acid digestion (USEPA Method 3051). Total N was determined by combustion at 1010°C using a Carlo Erba (Milan, Italy) NA-1500 CNS analyzer. Oxalate-extractable P, Fe, and Al were reported from previous studies (Elliott et al., 2002; Makris et al., 2005). Organic matter contents were determined by loss on ignition, percent solids by drying at 110°C, and pH at 1:2 solids/water ratio. Water-extractable P (WEP) was measured at a 1:200 solids/solution ratio achieved by adding distilled/deionized water to 0.5 dry g of material (Wolf et al., 2005). Particle size analysis of the WTR was accomplished by the hydrometer method and dry sieving (Klute, 1986).

#### Field Runoff Study

Plots measuring 3 by 10.7 m were established in a field of mixed grasses (cut to approximately 10 cm) with an average slope of 9.5%. Soil at the site was an Andover channery loam (fine-loamy, mixed, active, mesic Typic Fragiaquults) with a fragipan at ~40-cm depth (Braker, 1981). Mehlich-3 STP values for the plots (14–22 mg kg⁻¹) were below optimum for crop production (Beegle, 2002) and indicated no recent P fertilization.

Treatments included a no-till unamended control (CNT), a tilled unamended control (CT), surface-applied DAP fertilizer (DAP), surface-applied LP (LPS), incorporated LP (LPB), and surface-applied LP with a surface-applied WTR-enhanced buffer strip (LPB). All treatments were randomly paired and replicated three times. Materials were surface applied with a drop-style lawn spreader at a targeted rate of 100 kg P ha⁻¹. Actual application rates were 101.4 and 102.2 kg P ha⁻¹ for the DAP and LP, respectively. Plots where the P source was incorporated (and the corresponding controls) were tilled to approximately 15-cm depth with a rototiller and then rolled to smooth and slightly compact the soil.

Treatments were applied to the upslope 75% of each plot 48 h before rain simulations. The lower 25% (2.67 m) was reserved for a grassed buffer strip. All plots included the grassed buffer strip to allow for comparison between treatments. Because of plot geometry and study emphasis on runoff flow direction, the term “length” will be used to describe what is called buffer width in practice. For the LPB treatment the air-dried WTR was applied in the grassed buffer at a rate of 20 Mg ha⁻¹ (dry weight equivalent) similar to other recent studies (Dayton and Basta, 2005b). Following treatments, steel borders were installed to isolate plot runoff. Steel weirs were installed at the downslope end of plots to allow runoff to flow freely into collection troughs.

Simulated rain was generated using the Water Erosion Prediction Project (WEPP) simulator closely following the protocol described by Sharpley and Kleiman (2003). The WEPP simulator is a trailer-mounted, rotating boom simulator. Ten 7.6-m booms rotate (~6 rpm) from a central hub at a height of approximately 2.4 m. A total of 15 Teejet 2HH-SS50WSQ nozzles were utilized of the available 30. This produced an average rainfall intensity of 62.4 mm h⁻¹ (standard deviation of 3.4) with a 93% coefficient of uniformity. Intensity-duration-frequency curves...
Table 1. Chemical characteristics of phosphorus sources and water treatment residual.

<table>
<thead>
<tr>
<th>Material</th>
<th>Total elemental</th>
<th>Oxalate-extractable</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>N</td>
<td>P</td>
</tr>
<tr>
<td>Largo pellets</td>
<td>67.4</td>
<td>34.6</td>
</tr>
<tr>
<td>DAP fertilizer</td>
<td>181</td>
<td>233†</td>
</tr>
<tr>
<td>Bradenton WTR</td>
<td>5.95</td>
<td>4.04</td>
</tr>
</tbody>
</table>

† Percent volatiles by loss on ignition.
‡ From Elliott et al. (2002).
§ From Makris et al. (2005).
¶ From O’Connor and Elliott (2006).

(Aron et al., 1986) indicated that the simulated rainfall events exceeded the 100-yr storm for south central Pennsylvania. Water was supplied by a nearby irrigation pond located upslope of the plots. Water was pumped from the surface of the pond directly to the WEPP simulator. Simulated rainfall had average TDP and total P (TP) concentrations of 0.022 and 0.055 mg L⁻¹, respectively. No measurable natural rainfall occurred during simulations and plots were covered overnight with tarps.

The WEPP simulator simultaneously wets two plots (Sharpley and Kleinman, 2003). Plots were paired randomly and blocked with one replication per day. Each pair received one rainfall event sufficient to capture the initial 30 min of runoff. Plot runoff was captured by two collection troughs and fed to 0.12 m low-flow flumes equipped with water-level recorders, calibrated to calculate discharge flow using the flume rating equation (Holtan et al., 1962). During the 30-min runoff event, discharge flow was recorded every minute and six runoff samples were taken at 5-min intervals.

Collected runoff volume was measured and runoff TP, TDP, and total P were determined. Flow-weighted composite samples were made from the six samples to represent the entire runoff from each plot. Filtered (0.45 μm) and unfiltered samples of each composite were used to quantify TDP and TP with the difference interpreted as particulate-bound P (PP). Filtered samples were analyzed directly, and unfiltered samples were acid digested (USEPA Method 3051) before ICP analysis. Statistical analyses were performed using Analysis of Covariance (ANCOVA) and Tukey’s Pairwise Comparison with statistical difference determined using a critical value (α) of 0.05.

Phosphorus Adsorption Kinetics

Adsorption of P by the WTR was investigated with batch kinetic experiments following the procedures of Nair et al. (1984) and Dayton and Basta (2005a). One gram (dry weight equivalent) of WTR was added to 25 mL of 10 mg L⁻¹ P solution in 0.01 mol L⁻¹ CaCl₂ (pH=6). Samples were agitated on an end-over-end shaker and removed at the following times: 1, 1.5, 2, 2.5, 5, 15, and 30 min, and 1, 2, 5, 12, and 24 h. Ten mL of solution were removed, filtered (0.45μm), acidified, and analyzed for TDP by ICP. The amount of P adsorbed was calculated by difference (initial minus final).

Results and Discussion

Phosphorus Source and WTR Properties

Table 1 shows the results of chemical analyses for both P sources and the WTR. Both the LP and the DAP were relatively dry with high solids contents. The DAP fertilizer had the greatest TP (233 g kg⁻¹) and WEP (172 g kg⁻¹). High WEP levels were expected from the DAP since chemical fertilizers are designed to be highly soluble. The LP had WEP levels ~4 g kg⁻¹. This is a typical value for BPR biosolids, yet high in terms of biosolids in general (Brandt et al., 2004). The TP of the WTR (4.04 g kg⁻¹) was typical although greater than average (~1.4 g kg⁻¹) for alum-based WTRs (Dayton et al., 2003; Dayton and Basta, 2005a). However, the extremely low WEP and degree of P saturation [Pox/(Alox+Feox)] imply this material is a strong P sink rather than source (Elliott et al., 2002).

The WTR was high in total Al (116.1 g kg⁻¹), reflecting the use of alum as the primary coagulant in the drinking water treatment process. Previous research has found that the P-sorbing potential of WTRs depends on the non-crystalline fraction of total Al as measured by oxalate extraction (Elliott et al., 2002; Dayton and Basta, 2005a), and this WTR had an Alox of 91 g kg⁻¹ (Makris et al., 2005). Dayton and Basta (2005a) found that Alox for 18 different WTRs ranged from 13.9 to 165 g kg⁻¹ with a mean value of 73.1 g kg⁻¹. The Alox is highly correlated to the P adsorption capacity of WTRs. The Langmuir P adsorption maximum (Pmax) for the Bradenton WTR was estimated as 25.5 g P kg⁻¹ using the Pmax vs. Alox regression equation developed by Dayton and Basta (2005a).

Particle-size analysis indicated the WTR particles were sand sized or larger (~94%) with silt- and clay-sized particles accounting for only 4.2 and 1.7%, respectively. Sieving was used to further subdivide sand sized particles and resulted in ~32% of total material mass larger than 2.0 mm in size, and ~23% between 1.0 and 2.0 mm (very coarse sand). By difference, 38% of the WTR particles were in the 0.05 to 1 mm range.

Runoff Results

Figure 1 shows runoff TDP and TP concentrations for the treatments. The DAP reference treatment (not shown) was excluded from statistical analyses as its large runoff P concentrations (TP = 9.4 mg L⁻¹, TDP = 9.3 mg L⁻¹) would cause overestimation of the mean square error in ANCOVA. Runoff P from all treatments was primarily TDP, as indicated by the similar TDP and TP magnitudes in Fig. 1. A lack of appreciable PP...
Effect of WTR-Enhanced Buffer Strips

Reductions in runoff TP (7.4%) and TDP (27.8%) from plots with WTR-amended buffer strips were not sufficient to give a significant $\alpha = 0.05$ treatment effect. The inability of WTR-amended buffer strips to dramatically reduce runoff P was surprising in light of the documented high P sorption potential of WTR (Dayton and Basta, 2005a), the ability of incorporated WTRs to significantly reduce Mehlich-3 P (Peters and Basta, 1996), and studies showing that runoff P from fields high in STP can be decreased by WTR applications (Haustein et al., 2000). In marked contrast to the present study are the findings of Dayton and Basta (2005b) who reported that buffer strip application of WTRs having P-sorbing properties comparable to the Bradenton material showed substantial (66.8–86.2%) reductions in runoff P in small-scale indoor rainfall simulations.

The high variability of the WTR-enhanced buffer plots warrants comment. The WTR application rate (20 Mg ha$^{-1}$) translated into ~16 kg of WTR applied in the 3 × 2.67 m buffer, making uniform application of WTR difficult. As such, we speculate the variability in runoff P for the WTR-enhanced plots would be higher because of the greater impact of flow channelization on runoff P mitigation. That is, if the surface micro-topography resulted in flow moving predominantly through a portion of the buffer area with a localized concentration of WTR, the P mitigation effect would be greater than for flow being channeled through a portion of the buffer strip receiving little or no WTR. For the plots without WTR in the buffer, P reduction should be relatively independent of the actual portion of the buffer subject to short circuiting.

While our plot size was limited by the rainfall application area of the WEPP simulator, field plots with longer buffers and the use of higher WTR application rates would be useful for investigating this phenomenon. In a recent study of surface-applied swine manure, Allen and Mallarino (2008) observed that large flow variations at the small rainfall-simulator scale reduced the statistical significance of treatment effects on P loads.

Gallimore et al. (1999) conducted one of the few comparable field studies using WTR-amended buffers. Their P application rates were similar (104 kg P ha$^{-1}$ upslope of the buffers) but poultry litter was the P source. They reported a statistically significant reduction in runoff TP and DP concentrations when buffers were amended with a WTR having an Al$_{ox}$ of 50.5 g kg$^{-1}$. However, application of a WTR with lower Al$_{ox}$ (11.7 g kg$^{-1}$) failed to show a statistically lower runoff DP level. Runoff DP from their control (poultry litter only) plots was dramatically greater (14–17 mg L$^{-1}$) than that of the comparable (LPS) plots (our Fig. 1). The plots of Gallimore et al. (1999) had flatter slopes (< 5%), which likely resulted in lower runoff velocities, and thus longer WTR-runoff contact time in the buffers.

In explaining our observed behavior, it was important to verify the existence of adequate P sorption capacity in the WTR-enhanced buffer (LPS) plots. If the untreated buffer had a negligible effect on TDP removal, the TDP observed for the LPS plots (2.6 mg L$^{-1}$) should approximate the TDP entering the WTR-enhance buffers for the LPS plots. Using this assumed TDP value and the average volume of runoff for the LPS plots, only 0.24%
of the P sorption capacity (25.5 g kg⁻¹) of the WTR in the buffer strip would have needed to adsorb all the P in the runoff. While the WTR apparently had sufficient P-fixing capacity to substantially reduce runoff P, inadequate contact time could conceivably explain the inappreciable adsorption of P by the WTR. To investigate this possibility, the residence time of the runoff in the buffer strip was estimated. Numerous empirical methods are available to estimate runoff travel velocity under sheet flow conditions (Seybert, 2006) but some are inappropriate for small-scale experiments or require unavailable input data. Given these constraints, the curve number and NRCS segmental methods were used to quantify the rate at which runoff was conveyed across the buffers (Table 2). These two methods resulted in an average travel time over the 2.67-m WTR-amended buffer of 30.4 s which is equivalent to a surficial flow velocity of 0.09 m s⁻¹. This estimate is reasonable in light of Dougherty et al. (2004), who reported plot runoff velocities typically do not exceed 0.15 m s⁻¹, which equates to a travel time of 17.8 s for our plots.

WTR-Phosphorus Sorption Kinetics

A batch sorption experiment was conducted to investigate the kinetics of P removal from solution by the Bradenton WTR (Fig. 2). The extent of adsorption increased abruptly over a short time period with 50% reduction of TDP in 2 min and was followed by a slower adsorption phase resulting in 90% removal in 0.25 h and nearly complete removal after 24 h. Others have observed similar behavior (Makris et al., 2005; Novak and Watts, 2005). Makris et al. (2005) postulated that the initial fast adsorption process is due to binding to readily accessible surface sites while the slower adsorption stage is associated with diffusion of phosphate ions into interstitial pores of the WTR. Two-phase sorption kinetic patterns have been observed for many adsorbates and oxyhydroxide adsorbents (Zhang and Stanforth, 2005). This behavior has led some investigators (Makris et al., 2004; Dayton and Basta, 2005a) to use long reaction times (up to 80 d) when determining Pmax values for WTRs.

Analysis of the kinetics experiment data (Fig. 2) revealed an extremely good fit to a second-order rate process. A linear plot of [TDP]⁻¹ versus time (R = 0.99) for the first 5 min of the adsorption experiment had a slope (second-order rate constant) of 8.14 × 10⁻¹ L mg⁻¹ s⁻¹. Makris et al. (2005) reported a second-order P sorption rate constant for the Bradenton WTR of 2.4 × 10⁻¹ L mg⁻¹ s⁻¹. Such fits should be viewed mathematically and mechanistic interpretation in terms of second-order dependence is usually not justified (Stumm and Morgan, 1996). For this reason, such reactions are often called pseudo-second-order processes. In a review of numerous reported adsorption studies, Ho and McKay (1999) found that pseudo-second-order reaction kinetics provided the best correlation of experimental data when a surface chemical reaction was the rate-controlling step in the adsorption process.

Assuming the TDP entering the LPB-plot buffers was equal to that observed for the LPS plots (2.6 mg L⁻¹), the WTR accounted for a net P removal of 0.7 mg L⁻¹. If this removal occurred in the estimated buffer travel time and followed a second order process, the effective pseudo-second order rate constant for the buffer would be 4.66 × 10⁻³ L mg⁻¹ s⁻¹. This value differs from that calculated in the complete-mix batch kinetic experiments because the laboratory conditions do not directly simulate the flow-through process occurring in the field.

Buffer Design Considerations

Federal regulations (CFR 40 Part 503) for land application of biosolids (USEPA, 1993) prohibit spreading within 10 m of surface waters, although many states mandate wider buffers. The federal standard was used in conjunction with our data to interpret the practical significance of the findings. The incoming buffer TDP concentration (TDP) was assumed to be 2.6 mg L⁻¹, and a target reduction to 1 mg L⁻¹ was chosen to illustrate the interplay of factors affecting the P removal effectiveness of buffers.

To quantify the effect of slope on P reduction over various buffer lengths, all parameters used in the travel time analysis (Table 2) were held constant except slope and length to develop the following relationship for average travel time (t):
If adsorption rate increased by approximately 15% (to 5.35 \times 10^{-3} L \text{mg}^{-1} \text{s}^{-1}), the 1 mg L^{-1} runoff dissolved P could be achieved with a 10 m long buffer. However, it should be noted the Bradenton WTR had the one of the highest rate constants reported by Makris et al. (2005) and the availability of WTR with faster P sorption kinetics is uncertain.

The kinetics of P fixation by porous WTRs is complicated by simultaneous diffusion-adsorption phenomena. Logically, the kinetics of P adsorption should be faster for WTRs having more of their P-sorbing sites on highly accessible external surfaces rather than in micropores. The external-internal binding site distribution varies with WTR (Makris et al., 2005), and those with a high density of external sites would be preferred as buffer amendments.

Decreasing the effective particle size should increase the kinetics of P adsorption because the total external surface area per unit mass should vary with the reciprocal of the WTR particle diameter, and the diffusive path to internal P-binding sites should decrease as particles become smaller. Novak and Watts (2005) investigated P sorption kinetics for five WTR aggregate size ranges and found the fastest sorption for the smallest WTR aggregates (< 0.5 mm) and that adsorption rate decreased with increasing aggregate size. They concluded that WTR aggregate size is an important P-reduction performance criterion.

Besides accelerating sorption kinetics, applying more finely divided WTRs should promote more uniform surface coverage in the buffer strips, thereby enhancing the likelihood that runoff passing through the buffer would contact the WTR. Haustein et al. (2000) demonstrated that WTRs applied to soils containing high STP significantly reduced runoff P. In their field plots, opportunity for P sorption was facilitated by application of the WTR as slurry (8% solids) in contrast to our study where WTR aggregates, sand-sized and larger, were surface applied.

**WTRs and Phosphorus-Based Management**

Incorporation of WTRs into high P soils has been shown to reduce STP (Peters and Basta, 1996) and increase P sorption potential (Novak and Watts, 2004). Runoff (Haustein et al., 2000) and leachable (Eaton and Sims, 2003) P are accordingly reduced. Co-blending of Al and Fe-based materials, like WTRs, with the P source before application is also effective (Ippolito et al., 1999). In both cases, adequate mixing is necessary for intimate adsorbent-P contact (Jacobs, 2003). These practices should involve sufficient contact time between soluble P and the WTRs barring a significant runoff-generating event immediately following incorporation or blending.

The use of WTRs to enhance the normally minimal capacity of grassed filter strips to remove soluble P (Sharpley et al., 2006) requires careful design and management. Buffer slope and length should be such that concentrated flow is avoided. In this study, surficial flow velocities through the buffer were estimated to be ~0.1 m s^{-1}. While buffer entry velocities of less than 0.5 m s^{-1} are suitable for suspended solids removal (Palone and Todd, 1998), they may be too high for adequate contact between runoff dissolved P and P-fixing amendments spread in buffers. Candidate WTRs should exhibit rapid P sorption kinetics. Since adsorption is surface-area dependent, more finely divided materials should perform better. However, fine particles will be more susceptible to erosional loss. From a practical perspective, the form in which the WTR is applied depends on the economic hauling distance, the form of the WTR (lagooned or stockpiled), and the available spreading equipment.

Dayton and Basta (2005b) propose the use of WTR-based management practices for modifying source and transport factors in state P risk indices. Incorporation of WTR into high P soils to lower STP and co-application with highly soluble organic P fertilizers are proven source BMPs that lower P risk. The former practice will be accounted for in most state assessment tools since STP is a common input factor in P indices (Sharpley et al., 2003). Co-blending of WTR with nutrient sources to reduce
their soluble P content is credited as a P reduction strategy in state indices that allow testing of the applied nutrient source to differentiate P loss potential (Elliott et al., 2006).

A few state P indices account for P loss reduction due to vegetated buffer strips that intercept eroded sediment. The effectiveness of vegetated strips as nutrient buffers is highly variable and can be compromised by concentrated flow where runoff velocities circumvent buffer renovative capacity. In the Vermont P index (Jokela, 2005) adjustment is permitted only if sheet flow is maintained across the buffer. The ability of WTR-enhanced buffers to remove DP is likewise diminished as water flow concentrates and increases in velocity. Nevertheless, WTRs will increase buffer P sorption capacity and provide benefit when soils are nearly saturated with water but not generating runoff. No state P indices currently allow adjustment for transport reductions due to buffer strips enhanced with WTRs (Dayton and Basta, 2005b). Such adjustments should be contingent on a quantitative understanding of the conditions under which enhanced buffer strips provide the desired level of water quality protection.

Conclusions

The results of this field study failed to demonstrate conclusively that surface-applied WTRs in buffer strips will reduce delivery of DP and TP from agricultural fields to surface waters. With buffer strips top-dressed with WTR (20 Mg ha⁻¹), runoff TP (2.5 mg L⁻¹) and total DP (1.9 mg L⁻¹) were not statistically lower (α = 0.05) compared to plots with unamended grass buffers (TP = 2.7 mg L⁻¹; TDP = 2.6 mg L⁻¹). This was unexpected since, given the average runoff P concentrations and volumes, < 1% of the P sorption capacity (25.5 g kg⁻¹) of the WTRs in the buffer strips would have been needed to adsorb all P in the runoff. Based on WTR-P sorption kinetic experiments and estimates of the runoff residence time in the short buffers of the experimental plots, we attribute the lack of observed effect to insufficient contact opportunity between the WTRs and P-laden runoff. Buffer strips amended with WTRs to intercept runoff P should be designed to ensure adequate contact to meet water quality protection goals.

References


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