Phosphorus Availability in Soils Amended with Dewatered Water Treatment Residual and Metal Concentrations with Time in Residual

J. E. Hyde and T. F. Morris*

ABSTRACT

The objective of this laboratory study was to measure the effect of dewatered water treatment residual (WTR) on extractable P and Al in soils with above-optimum P concentrations. A secondary objective was to document the variability of the metal content of WTR during 1 yr. Two soils, a Paxton fine sandy loam (coarse-loamy, mixed, active, mesic Oxyaquic Dystrudept) and an Enfield silt loam (coarse-silty over sandy or sandy-skeletal, mixed, active, mesic Typic Dystrudept) with above-optimum Mehlich 3 P concentrations (833 mg kg^{-1} and 630 mg kg^{-1}, respectively) were amended with four dewatered WTRs. The WTRs were dewatered using three treatments: (i) WTR dewatered at 40°C in a forced-air oven (DRY), (ii) WTR dewatered by freezing at -4°C and then dried at 40°C in a forced-air oven (FROZEN), and (iii) WTR dewatered to 4.5% solids at 40°C in a forced-air oven (RAW). The WTRs were added to the soils at rates of 20 or 60 g kg^{-1} and incubated for 21 d. The WTRs in the RAW treatment significantly reduced Mehlich 3 P concentrations compared with the DRY and FROZEN treatments. The RAW treatment reduced soil P concentrations an average of 64% compared with a reduction of 28% for the DRY treatment and 23% for the FROZEN treatment. The results suggest that the method used to dewater WTR will alter its ability to decrease Mehlich 3 P soil concentrations. The secondary objective involved collection of two of the WTRs every 3 wk for 1 yr and subsequent analysis for metal concentrations. The metal concentrations changed little during the year and only Cu, due to its use as an algicide, was elevated.

D RINKING water treatment facilities produce millions of liters of potable water every day. The process for producing potable water from surface water supplies typically involves coagulation and flocculation of suspended solids. Common coagulants include aluminum sulfate ($Al_2(SO_4)_3$), ferric chloride ($FeCl_3$), and ferric sulfate ($Fe_2(SO_4)_3$). In some cases, organic polymers are added. The coagulants cause precipitation of suspended solids in the water, which are mostly suspended organic material. The precipitated material is filtered, and the solid material is known as water treatment residual (WTR). This residue is comprised of Al or Fe hydroxide flocs and materials, both soluble and insoluble, from the water source.

Water treatment residual often is disposed in a landfill. Research to find a beneficial use for WTR has included use as a soil amendment to improve the physical properties of potting soils (Rengasamy et al., 1980; Bugbee and Frink, 1985), use as a liming substitute (Elliott and Singer, 1988), use as a material to reduce N and P in runoff (Gallimore et al., 1999), and for specific conditions, as a supplier of macro- and micronu-trients (Heil and Barbarick, 1989). In many studies, however, the benefits of WTR were offset by a need for applications of P fertilizer because of reductions in P availability caused by the WTR (Bugbee and Frink, 1985; Elliott and Singer, 1988; Lucas et al., 1994; Cox et al., 1997). Reductions in P availability probably were caused by the formation of relatively insoluble aluminum- or iron-complexed P (Lucas et al., 1994). There was an increased need for P fertilizer in these experiments because the soils had low to optimum concentrations of soil test P.

Addition of WTR to soils with soil test P concentrations much above optimum, however, may provide a beneficial decrease in soil test P without reducing plant growth or yield (Elliott and Dempsey, 1991; Peters and Basta, 1996; Gallimore et al., 1999). Alum WTR has been shown to be particularly effective for reducing extractable P in soils with above-optimum P concentrations (Peters and Basta, 1996). In this study, application of alum WTR at 100 g kg^{-1} reduced Mehlich 3 P concentrations 30 to 60%. The WTR did not decrease the extractable soil P concentration below the critical level for good plant growth, nor was there evidence of deleterious effects to soil characteristics such as pH, soil salinity, and extractable Al and heavy metals. A decrease in soil test P would be beneficial on soils with large concentrations of extractable P, because the greater the soil test P concentration the greater the concentration of dissolved P in runoff (Sharpley, 1995). Dissolved P in agricultural runoff can cause eutrophication of lakes and streams and reducing P in agricultural runoff should improve the quality of our environment.

Dewatering WTR can affect its ability to decrease extractable soil P (Lunetta, 1993). Application of raw WTR reduced extractable soil P by an average of 85% compared with an average reduction of 65% when WTR that had been dewatered by freezing and thawing was applied (Lunetta, 1993). Most published research studies that estimated the benefits of WTR application to soils used either air-dried or frozen-dried material or the method of dewatering was not noted (Rengasamy et al., 1980; Bugbee and Frink, 1985; Elliott and Singer, 1988; Heil and Barbarick, 1989; Elliott et al., 1990; Lucas et al., 1994; Peters and Basta, 1996; USEPA, 1996). More complete knowledge about the effect of dewatering WTR on its capacity to reduce soil test P concentrations is needed before WTR is applied to agricultural land to reduce above-optimum soil test P concentrations.

Application of WTR to agricultural land should not

Department of Plant Science, Univ. of Connecticut, 1376 Storrs Road, Storrs, CT 06269. Storrs Agric. Exp. Stn. Journal Paper no. 1914. Received 30 Sept. 1999. *Corresponding author (tmorris@canr.cag.uconn.edu).

Abbreviations: WTR, water treatment residual; AL1 and AL2, aluminum-based water treatment residuals from two different facilities; FE1 and FE2, iron-based water treatment residuals from two different treatment facilities.

substantially increase the metal concentration of the soil. Previous studies have shown that most WTRs have low concentrations of regulated metals (Bugbee and Frink, 1985; Elliott and Singer, 1988; Elliott and Dempsey, 1991; Lucas et al., 1994; Peters and Basta, 1996). Most of these studies reported the metal concentrations of one or two WTRs, and only one sample of WTR was collected. In the largest study of the metal concentrations of WTR, Elliott and Dempsey (1991) showed that eight different WTRs had metal concentrations that were similar to soils, and the concentrations were primarily controlled by the concentrations of the metals in the coagulant. Systematic sampling of WTR to describe the variability of metal concentrations in WTR during 1 yr has not been reported to our knowledge. Metal concentrations in WTR could change during 1 yr due to changes in the metal concentrations of the lake from summer stratification and fall circulation (Wetzel, 1983).

The objective of this laboratory study was to measure the effect of dewatered WTR on extractable P and Al in soils with above-optimum P concentrations. A secondary objective was to document the variability of the metal content of WTR during 1 yr.

MATERIALS AND METHODS

We collected samples of WTR from four water treatment facilities in Connecticut. The AL1 (Lake Gaillard) and FE1 (Lake Saltonstall) samples were collected directly from the filters. The AL2 (Lake Waterbury) and FE2 (West River) samples were collected from settling basins located immediately after the filters. The AL1 and AL2 facilities used alum as the coagulant, and the FE1 and FE2 facilities used ferric chloride as the coagulant (Table 1). Various amounts of cationic polymer also were added with the alum and ferric chloride (Table 1). The solids contents of the WTRs were determined using Method 2540G of the American Public Health Association (1989). The solids content and the pH of each material are listed in Table 1. Additional samples of WTR were collected from the AL1 and FE1 facilities to characterize the metal concentrations of WTR during 1 yr. The AL1 facility was sampled every 3 wk between August 1997 and August 1998, for a total of 18 samples. The FE1 facility was sampled every 3 wk between August 1997 and July 1998, for a total of 16 samples. All samples were collected directly from filter backwash.

The WTRs were dewatered using three regimes. The DRY treatment involved dewatering the WTRs in a forced-air oven at 40°C until a stable weight was reached. The FROZEN treatment involved freezing the WTRs at −4°C until the material was frozen solid as indicated by visual inspection of the plastic bucket containing the WTR. The frozen WTRs were

thawed at room temperature (22°C), and the water was decanted by siphoning. The thawed and decanted WTRs were further dewatered in a forced-air oven at 40°C until a stable weight was reached. After drying, both the DRY and FROZEN WTRs were ground by mortar and pestle to pass through a 2-mm screen. The RAW treatment consisted of dewatering the WTRs in a forced-air oven at 40°C until each material contained 4.5% (±0.1%) solids.

Soil samples were collected from the surface 15-cm layer of two soils with long-term histories of large additions of P. Selected information about the soils is shown in Table 2. The Enfield soil had received applications of inorganic P fertilizer for at least 20 yr, and the Paxton soil had received applications of poultry manure for at least 20 yr. The soil samples were air-dried at room temperature and passed through a 2-mm screen.

The experimental design was completely randomized with treatments in a factorial arrangement (soil by WTR by dewatering method by rate) with three replications. There were three controls (no amendments) for each soil. The dewatered WTRs were added to each soil at rates of 20 and 60 g kg^{-1} soil on a dry weight basis. Each WTR–soil mix was created by adding the correct amount of WTR to 80 g of soil in a plastic cup. The WTR–soil mixes were wetted to field capacity, mixed thoroughly, partially covered to reduce evaporation, and placed in a warm room (26°C) for 21 d. A 21-d incubation was used because previous research had shown minimal changes in Mehlich 3 P and pH values after 21 d (Peters and Basta, 1996; Butkus et al., 1998). The treatments were thoroughly mixed at 7, 14, and 21 d. Water was added as needed to maintain field capacity. At the end of the 21-d period, all treatments were air-dried at room temperature and passed through a 2-mm screen.

Extractable P concentrations were measured using the Mehlich 3 extraction method (Wolf and Beegle, 1995). The procedure included shaking 2 g of soil with 20 mL of Mehlich 3 solution for 15 min with filtration of the extract through Whatman no. 2 filter paper. This shaking time is 10 min longer than the procedure in the original description of the Mehlich 3 method (Mehlich, 1984). Because the longer time of shaking may extract a different amount of P, we compared the P concentrations of triplicate samples of the soil–WTR incubated mixes using a 5-min shaking time and a 15-min shaking time. The P concentrations for the two shaking times were highly correlated ($R^2 = 0.99$), and the equation for the relationship was:

P concentration (mg P L^{-1}) 5-min =

20.9 + 0.97 (P concentration 15–min).

Table 2. Selected information about the two soils used in this study.

Variable†	Soil	
	Paxton	Enfield
Sand, g kg^{-1}	566	222
Silt, g kg^{-1}	340	630
Clay, g kg^{-1}	94	148
Mehlich 3 P, mg kg^{-1}	829	621
CaCl$_2$ soluble P, mg L^{-1}	0.31	0.45
pH	7.02	6.53
Modified-Morgan K, mg kg^{-1}	407	968
Modified-Morgan Ca, mg kg^{-1}	9165	1121
Modified-Morgan Mg, mg kg^{-1}	392	221
Organic matter, LOI, g kg^{-1}	82	26

† Percent sand, silt, and clay by the hydrometer method (Gee and Bauder, 1986). pH in a soil to water ratio of 1:2. Modified-Morgan K, Ca, and Mg (Wolf and Beegle, 1995). LOI = loss on ignition at 375°C for 16 h (Ball, 1964).

Table 1. Rates of alum, iron chloride, and cationic polymer used as a coagulant and the pH and solids content of the water treatment residual from each treatment facility.

Coagulant component	Water treatment facility			
	AL1	AL2	FE1	FE2
Alum rate, mg L^{-1}	8.3	10 to 18	0	0
Iron chloride rate, mg L^{-1}	0	0	25.2	3.5
Cationic polymer rate, mg L^{-1}	1.32	0	0.83	2.6
pH	5.9	5.7	7.2	6.4
Solids content, g kg^{-1}	8.7	7.3	6.0	42.2

Our original 15-min Mehlich 3 concentrations were converted to 5-min Mehlich 3 concentrations using the above equation. The values reported are the converted values.

Soluble P soil concentrations were determined using 0.01 M CaCl$_2$ (Kuo, 1996). The procedure for the soluble P determination consisted of shaking 2 g of soil with 20 mL of CaCl$_2$ for 1 h with filtration of the extract through Whatman no. 42 filter paper. The extracts were measured for concentrations of P using ascorbic acid (6%) and molybdate–antimony reagents and a Scientific Instruments continuous flow analyzer (Westco, Danbury, CT). The detection limit for CaCl$_2$–soluble P was 0.025 mg P L^{-1}, and values below detection were recorded as one-half the detection limit. We used one-half the detection limit for values below detection rather than a no detect value because we wanted to perform calculations and statistical analysis on the data. Reporting values that are below detection as one-half the detection limit is a common practice for data of this type (Cohen et al., 1999). Aluminum was extracted from the soil using 1 M KCl (Bertsch and Bloom, 1996), and the Al concentrations in the extracts were determined using inductively coupled plasma. Metal concentrations of the WTRs were determined using the EPA 3050 method (USEPA, 1986). Inductively coupled plasma atomic emission spectroscopy was used to determine concentrations of Ag, Al, Ba, Ca, Cr, Cu, Fe, Ni, P, V, and Zn; graphite furnace atomic absorption was used to determine concentrations of As, Be, Cd, Pb, Sb, Se, and Tl; and cold-vapor atomic absorption was used to determine the concentration of Hg.

All statistical analyses were conducted with the SAS software (SAS Institute, 1990) using the generalized linear model (GLM) procedure for the factorial analysis. Treatment means for Mehlich 3–extractable P concentrations and CaCl$_2$–extractable P concentrations were separated from the mean values of the control samples using Dunnett's two-tailed t-test for a single control (SAS Institute, 1990). An alpha value of 0.05 was used to declare significant differences for all measurements in this paper.

Sixteen separate one-way analyses of variances within the soils, WTRs, and rates were performed to determine if the dewatering treatments had a significant effect on Mehlich 3 P concentrations. Separate analyses were conducted because all interaction terms were significant. Calcium chloride–

soluble P concentrations in the Enfield soil were not statistically analyzed because 72% of the values were below the detection limit, and statistical analysis of data with a large number of values below the detection limit is problematic (Cohen et al., 1999). Eight one-way analyses of variance were performed on the CaCl$_2$–soluble P concentrations in the Paxton soil. Mean values for the DRY, FROZEN, and RAW treatments for all the variables were separated using Duncan's multiple range test (α = 0.05) (SAS Institute, 1990). A linear-plateau model (Goodnight and Ihnen, 1990) was used to describe the relationship between CaCl$_2$–extractable P and Mehlich 3 P concentrations.

A nonparametric procedure, the Kruskal–Wallis Test (Pierce and Sall, 1990), was used to test for significant differences in the seasonal fluctuations of the metal concentrations of the WTR collected from the AL1 and FE1 facilities. The analysis was done by grouping the samples by seasons of the year. Samples collected between 21 March and 20 June were designated spring, samples collected between 21 June and 20 September were designated summer, samples collected between 21 September and 20 December were designated fall, and samples collected between 21 December and 20 March were designated winter.

RESULTS AND DISCUSSION

Metal Concentrations of Water Treatment Residual

The Al and Fe concentrations of the four WTRs reflect the coagulant used at each water treatment facility (Table 3). The metal concentrations of all the WTRs were lower than the USEPA Part 503 regulatory limits for sewage sludge (USEPA, 1995). The metal concentrations of WTR are not regulated by the EPA 503 statute for sewage sludge and domestic septage, but there are similar concerns about the accumulation of metals in soils from WTR, and for this reason many states invoke the 503 regulations for WTR (USEPA, 1996). The AL1 and FE2 materials had elevated Cu concentrations, which probably were caused by additions of CuSO$_4$ to

Table 3. Elemental concentrations of the four water treatment residuals used in this study determined by the EPA 3050 method.

Element	Water treatment facility				Regulatory limit†	Typical range for soils‡
	AL1	AL2	FE1	FE2		
			mg kg^{-1}			
Ag	ND§	ND	135	ND	–	0.01–5.0
Al	110 000	139 210	3 033	3 277	–	10 000–300 000
As	6.2	5.9	4.4	ND	41	1–50
Ba	59	59	274	125	–	100–3 000
Be	0.8	0.4	0.5	0.8	–	0.1–40
Ca	1 303	2 044	18 398	2 270	–	7 000–500 000
Cd	0.3	0.2	0.1	0.55	39	0.036–0.78
Cr	9	150	106	133	1 200	1–1 000
Cu	1 451	98	53	927	1 500	3.8–94.9
Fe	7 373	15 855	289 354	214 852	–	7 000–550 000
Hg	0.07	0.07	0.07	0.13	17	0.01–0.30
Ni	7	9	13	24	420	4.1–56.8
P	1 672	1 472	1 764	1 585	–	200–5 000
Pb	ND	5.2	26	24	300	4.0–23.0
Sb	ND	ND	1 593	ND	–	–
Se	ND	6	14	3.2	36	5–50
Tl	ND	1 101	ND	ND	–	–
V	23	55	89	70	–	20–500
Zn	64	79	46	127	2 800	8.0–126.0

† EPA 40 CFR Part 503 land application concentration limits for sewage sludge.
‡ Lindsay (1979); Cd, Pb, Zn, Cu, Ni in Holmgren et al. (1993).
§ ND = below detection limit.

the reservoirs for control of algae (John Hudak, Environmental Officer, South Central Regional Water Authority, New Haven, CT, personal communication, 1999). Addition of $CuSO_4$ to reservoirs for the control of algae in the summer is a common practice in reservoir management (Lucas et al., 1994). Another benchmark that may be used to evaluate the metal concentrations of WTR is the typical concentration in soils. The metal concentrations of these WTRs, except for the elevated Cu concentrations, are similar to those typically found in soils (Table 3). These results demonstrate that the

coagulants and the reservoir waters both had low concentrations of regulated metals immediately before these samples were collected.

Mehlich 3–Extractable Phosphorus Concentrations

The addition of WTR to the soils caused significant reductions in Mehlich 3 P concentrations compared with the unamended control soil (Fig. 1). In the amended soils, all statistical analyses within soils, WTRs, and rates
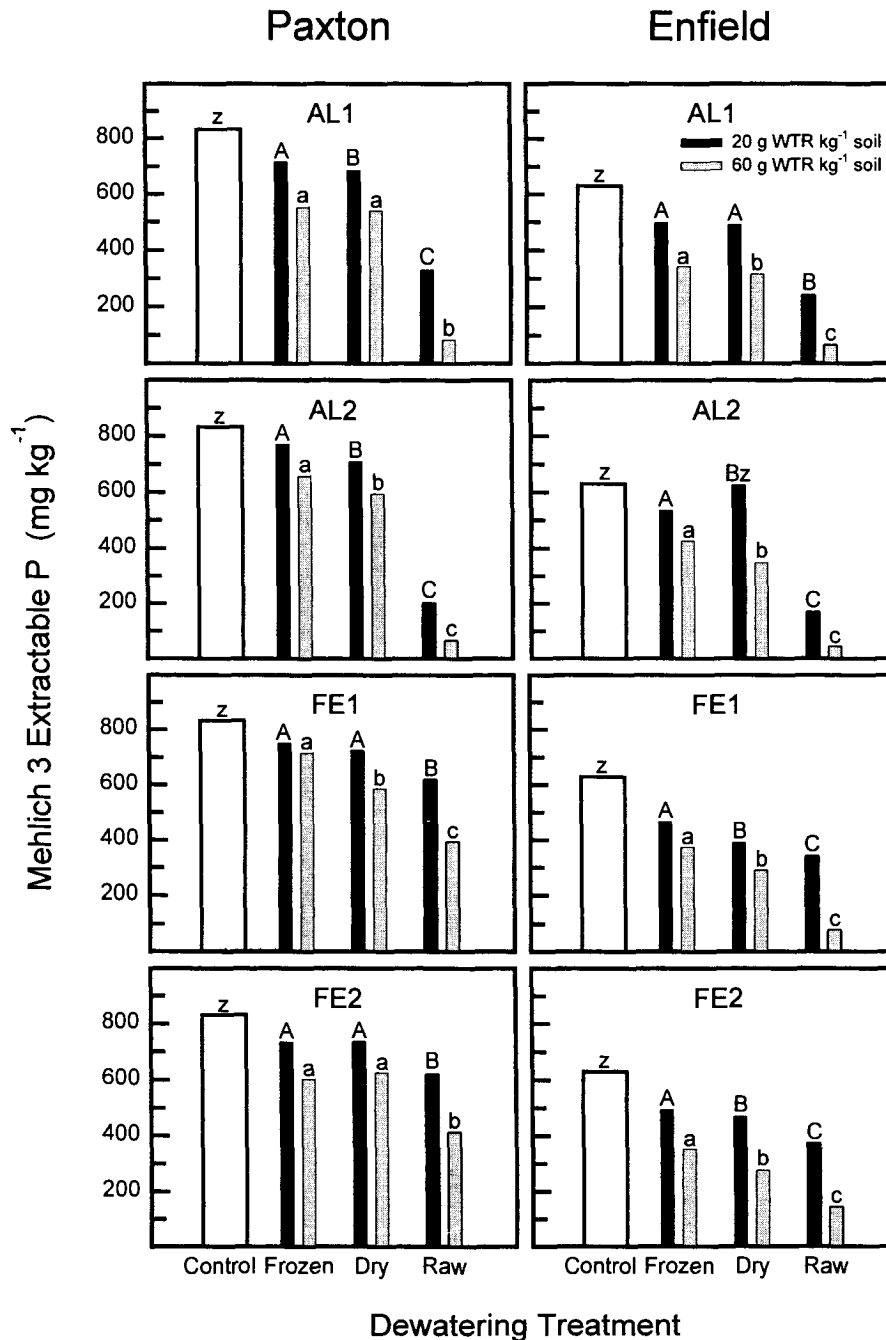

Fig. 1. Mehlich 3–extractable P concentrations for two soils when two alum water treatment residuals (AL) and two ferric water treatment residuals (FE), dewatered using three methods, were applied at two application rates. Z indicates Dunnett's test for control, A through C indicate Duncan's multiple range test (α = 0.05) within the 20 g kg^{-1} rate, and a through c indicate Duncan's multiple range test (α = 0.05) within the 60 g kg^{-1} rate.

showed significant decreases in Mehlich 3 P concentrations due to dewatering treatment. All interaction terms for the soils, dewatering methods, WTRs, and rate of WTR were significant. All the interaction terms for the analysis of variance for Mehlich 3–extractable P within a soil also were significant. The interactions may be explained by the different properties of the soils (clay content, organic matter content; Table 2) and because the WTRs contain different amounts of Al, Fe, and polymer (Table 1). Different amounts of oxidized Fe and Al added to soil would be expected to cause differential changes in extractable P concentrations, and cationic polymers can adsorb a large quantity of P fertilizer added to WTR (Butkus et al., 1998). The amount of cationic polymer added to the WTRs in this study varied considerably (0 to 2.6 mg L^{-1}; Table 1).

A comparison of the mean concentrations of Mehlich 3 P for the three dewatering treatments, based on the analyses of variance for drying treatment effects, showed that all values for the RAW treatment were significantly different from the DRY and FROZEN treatments (Fig. 1). In 11 of the 16 mean comparisons, the DRY and FROZEN treatments had statistically different Mehlich 3 P concentrations. These results indicate that the method used to dewater WTR can significantly alter the ability of WTR to reduce Mehlich 3 P concentrations.

The large differences in Mehlich 3 P reductions for the dewatering treatments probably are related to changes in surface area caused by dewatering the WTR. Others have found that dewatering WTR by air-drying or freezing changes the surface area of WTR, and the dewatered WTR can be granular, flaky, or stone-like (Martel and Diener, 1991; Elias, 1993). The average differences in Mehlich 3 P reductions across soils, WTRs, and rates were large. The RAW treatment had an average reduction of 64%, compared with an average reduction of 28% for the DRY treatment and 23% for the FROZEN treatment. The WTRs in this study were stone-like after drying and granular after freezing and drying. They were ground to pass through a 2-mm screen, which would make the surface areas somewhat uniform for the DRY and FROZEN treatments. For this reason it is impossible to know whether the small but significant differences in P adsorption between the DRY and the FROZEN treatments were due to differences in the size of the particles caused by the grinding technique, or due to differences in the adsorption potential caused by the dewatering method.

The Al- and Fe-WTRs produced similar average reductions of about 26% in Mehlich 3 P for both the DRY and FROZEN treatments (Fig. 1). In the RAW treatment the Al-WTR produced a much greater average reduction in Mehlich 3 P (79%) compared with the Fe-WTR (49%). The Al-based materials were more effective even though the concentration of Al in the Al-based materials was much lower than the concentration of Fe in the Fe-based materials (Table 3). Iron and Al both have a great affinity for PO_4 (Lindsay, 1979), and the affinity is dependent on many factors (e.g., pH, pE, presence of other compounds, and the crystalline to

amorphous ratio of the material). We did not measure the effect of dewatering WTR on these factors, but we believe our data suggests that the ratio of crystalline to amorphous forms in the Fe-WTR was greater than the ratio in the Al-WTR.

Large differences in the ability of amorphous Al and Fe to adsorb P compared with structured forms of Al and Fe have been documented in previous research with pure Al and Fe compounds (McLaughlin et al., 1981). Their results showed that amorphous Al adsorbed 70 times as much P compared with its structured form, while an amorphous Fe gel adsorbed 10 times as much P compared with its structured form. Our Al- and Fe-WTRs in the RAW treatment adsorbed only two to three times as much P compared with the structured materials in the DRY and FROZEN treatments, not nearly as much as reported by McLaughlin et al. (1981), probably because McLaughlin et al. worked with pure compounds. The rate of solid phase formation from amorphous to crystalline (structured) forms is a complex process controlled by many factors. Dewatering WTR clearly is one of the major factors increasing the rate of solid formation, which subsequently reduces the amount of P adsorbed when the WTRs are dewatered.

The WTRs dewatered by the DRY and FROZEN methods did not reduce Mehlich 3 P below 150 mg kg^{-1} (Fig. 1), which in some states is a recommended guideline for excessive P (Gartley and Sims, 1994). The lowest Mehlich 3 value was 41 mg kg^{-1} for the AL2 WTR in the RAW treatment at the 60 g kg^{-1} rate in the Enfield soil. This is considerably greater than the 30 mg kg^{-1} P concentration often used as a critical concentration, or the concentration where one is unlikely to observe increases in crop yield from additional P applications (Mehlich, 1984). It is evident from these results that large amounts of dewatered WTR (via drying or freezing) could be applied to soils before plant growth deficiencies might occur.

Calcium Chloride–Soluble Phosphorus Concentrations

Calcium chloride–soluble P concentrations in both soils were significantly reduced by the addition of WTR when compared with the unamended control (Fig. 2). A large percentage (72%) of the values for the Enfield soil were lower than the detection limit of 0.025 mg P L^{-1}. For this reason, statistical analysis of the Enfield data was not included. An overall analysis of variance in the Paxton soil showed that all interaction terms for the dewatering methods, WTRs, and rates were significant. The reason for the interactions is the same as previously noted for the interactions observed in the Mehlich 3 P concentrations. Analyses of variance for the Paxton soil within WTRs and rates showed significant decreases in $CaCl_2$–soluble P concentrations for the dewatering treatments. The RAW treatment produced significantly lower $CaCl_2$–soluble P concentrations compared with the DRY and FROZEN treatments (Fig. 2). The DRY and FROZEN treatments produced statistically similar $CaCl_2$–soluble P concentrations in five of
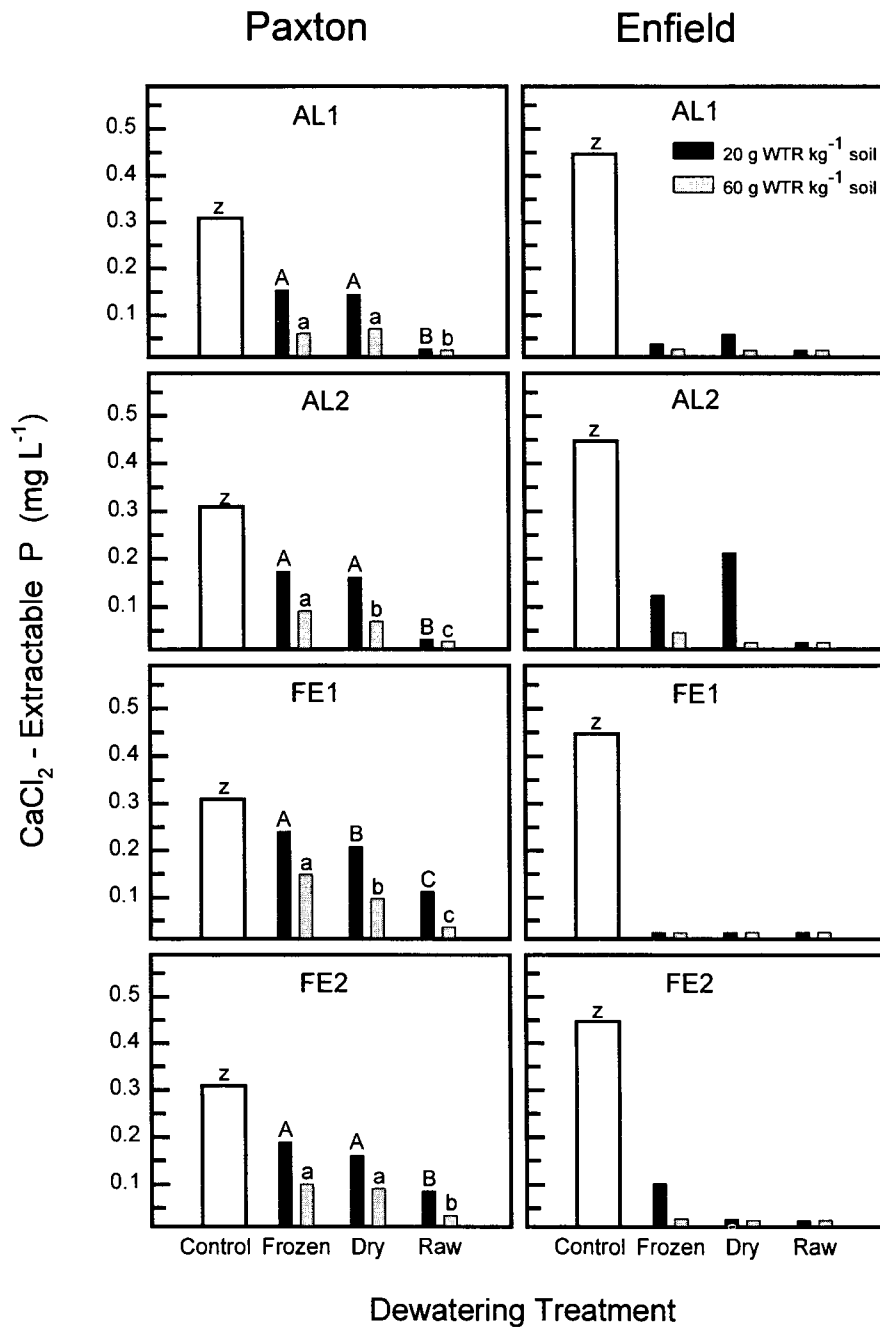
Fig. 2. Calcium chloride–soluble P concentrations for two soils when two alum water treatment residuals (AL) and two ferric water treatment residuals (FE), dewatered using three methods, were applied at two application rates. Z indicates Dunnett's test for control, A through C indicate Duncan's multiple range test (α = 0.05) within the 20 g kg^{-1} rate, and a through c indicate Duncan's multiple range test (α = 0.05) within the 60 g kg^{-1} rate.

eight mean comparisons. These results for the Paxton soil are similar to the results for the Mehlich 3 P concentrations and show that the method of dewatering WTR can significantly change the capacity of WTR to absorb P from a soil.

The $CaCl_2$–soluble P concentrations in the amended Enfield soil were much lower than concentrations in the amended Paxton soil. It is surprising that addition of WTR reduced the $CaCl_2$–soluble P concentrations to a greater extent in the finer-textured Enfield soil. Finer-textured soils have a greater buffering capacity for P

(Olsen and Watanabe, 1963), and it would be expected that soluble P concentrations would decrease more slowly in the Enfield soil. One explanation could be the difference in the source of the P in the two soils. The above-optimum Mehlich 3 P concentrations in the Paxton soil were created by long-term additions of poultry manure, while the Enfield soil received only long-term additions of fertilizer P. Manure applications have been shown to increase the ability of a soil to supply P compared with soils receiving only fertilizer P applications by increasing the organic P content and the P buffering
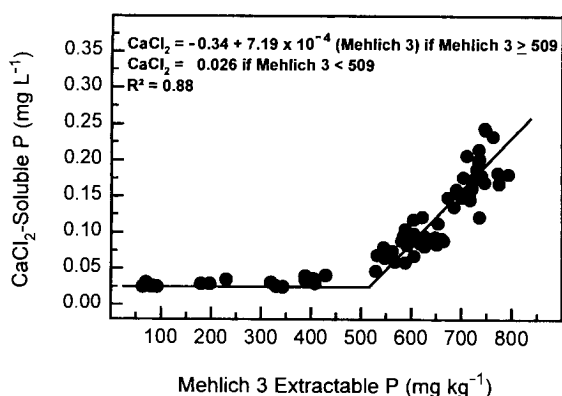
Fig. 3. Relationship between Mehlich 3–extractable P and CaCl₂–soluble P for the Paxton soil.

capacity of the soil (Abbott and Tucker, 1973; Habib et al., 1994).

Large reductions in concentrations of $CaCl_2$-soluble P should greatly reduce the potential for transport of dissolved P in leachate or runoff (Stout et al., 1998). The low concentrations of many of the WTR treatments, especially the RAW treatment (Fig. 2), however, were lower than the concentration needed for good plant growth. Concentrations of $CaCl_2$-soluble P required for good plant growth will vary with the crop, the soil texture, and the method used to extract the P (Kamprath and Watson, 1980; Kuo, 1996), but concentrations less than 0.07 mg P L^{-1} would be considered deficient for most crops and soils (Kamprath and Watson, 1980).

There was a significant two-phase relationship between $CaCl_2$-soluble P concentrations and Mehlich 3 P concentrations in the Paxton soil (Fig. 3). The values for the Enfield soil were not plotted because of insufficient values greater than the detection limit. In the Paxton soil, when the $CaCl_2$-soluble P concentration is equal to the lower limit (0.07 mg L^{-1}) for good plant growth, the Mehlich 3 P concentration is 570 mg P kg^{-1}. This

Mehlich 3 P concentration is much greater than the critical P concentration of 30 mg P kg^{-1} recommended for good plant growth. These data suggest that use of the Mehlich 3 extracting solution on WTR-amended soils may not provide accurate information about the availability of P for plant growth. Field experiments would be required to verify whether $CaCl_2$-soluble P or Mehlich 3-extractable P provides the best estimate of the P status in WTR-amended soils.

Extractable Aluminum

The amount of extractable Al in the unamended soils was extremely low and the addition of WTR to the soils caused only small changes in the extractable Al content (data not shown). We did not statistically analyze the extractable Al concentrations because a large number of samples were less than the detection limit, and because most of the values greater than the detection limit were extremely low. The extractable Al concentrations in 33% of the amended soils were lower than the detection limit of 0.50 mg kg^{-1} soil. Sixty-eight percent of the amended soils had concentrations less than 1.0 mg Al kg^{-1}, and the greatest concentration of extractable Al was 3.9 mg kg^{-1}. Problems with Al toxicity do not occur until the concentration of extractable Al is >60 mg kg^{-1} for wheat (*Triticum aestivum* L.) (Sloan et al., 1995), and other crops more sensitive than wheat probably would not be affected by the low concentrations in our study (Kamprath, 1970). The low concentrations of extractable Al probably are due to the formation of sparingly soluble Al–phosphate compounds.

Seasonal Changes in Metal Concentrations in Water Treatment Residual

Samples of WTR were collected from the AL1 and the FE1 facilities every 3 wk for 1 yr to describe the variability of metal content of WTR with season of the

Table 4. Average concentrations of 15 metals in the AL1 water treatment residual determined by the EPA 3050 method. Samples collected every 3 wk for 1 yr.

| Metal | AL1 | | | | | | Reg. limit†† |
	Range	Mean†	Median	SD‡	CV§	Seasonal¶	
		mg kg⁻¹			%	P>F	mg kg⁻¹
Ag	0.25–11.4	1.74	0.65	2.68	153	0.17	–
As	1.8–12.9	5.1	4.6	2.9	56	0.30	41
Ba	28.0–466	103.7	59.8	109.2	105	0.03	–
Be	0.30–2.94	1.34	1.36	0.60	45	0.06	–
Cd	0.05–1.75	0.63	0.68	0.54	87	0.01	39
Cr	6.6–23.5	11.6	10.4	4.5	39	0.03	1 200
Cu	360–2780	943.2	560.7	760.5	81	0.11	1 500
Hg	0.01–0.99	0.14	0.11	0.22	151	0.39	17
Ni	7.9–20.2	12.1	11.0	3.5	29	0.60	420
Pb	1.5–12.1	5.8	5.7	3.0	51	0.01	300
Sb	0.03–0.30	0.06	0.04	0.06	110	0.56	–
Se	0.3–7.2	2.6	2.3	2.0	194	0.08	36
Tl	0.03–0.25	0.07	0.03	0.07	99	0.01	–
V	13.6–37.5	22.8	22.0	6.8	30	0.06	–
Zn	38–482	97.1	63.5	106.9	110	0.35	2 800

† Eighteen samples collected from Aug. 1997 to Aug. 1998.
‡ SD=standard deviation.
§ CV=coefficient of variation.
¶ Seasonal=samples grouped by season of the year. The Kruskal–Wallis Test was used to test for significant differences in seasonal fluctuations of the metal concentrations.
†† Reg. limit=USEPA 40 CFR Part 503 concentration limits for land application of sewage sludge.

year (Tables 4 and 5). Most of the metals showed no significant differences with season of the year. All the metals had concentrations within the typical range for soils, or the concentrations were much lower than the regulatory limit, with the exception of Cu in the AL1 material. Some of the metal concentrations were quite variable. A major portion of the variability probably is due to changes in the metal concentration of the coagulant because the metal concentrations of WTR usually are controlled by impurities in the coagulant (Elliott and Dempsey, 1991). Other possible sources of variability in the metal concentrations are from variability in the distribution of the metals within the WTR, which we did not estimate, and from seasonal changes in the metal concentrations within the water column of the lake. Not much is known about the distribution of metal concentrations in the water column of lakes, but Wetzel (1983) reports that there could be small differences in the metal concentrations of the water column due to summer stratification and fall circulation. Slight changes in the metal concentrations of the water column could substantially change the metal concentrations in WTR because water treatment facilities usually process millions of liters of water per day. Our results for two WTRs show no pattern of seasonal changes in metal content that could limit the use of the WTR on agricultural land.

The elevated Cu concentrations in the AL1 WTR reflect the addition of $CuSO_4$ for control of algae in the lake supplying the AL1 facility. A total of 13 000 kg of $CuSO_4$ was applied to AL1 lake in May and June, 1997, with the last application on 14 June. The Cu concentration in the WTR was greater than the EPA 503 regulatory limit of 1500 mg kg^{-1} until 4 mo after the last Cu application (Fig. 4). Elevated Cu concentrations in the WTR 4 mo after application of $CuSO_4$ to the lake suggest that $CuSO_4$ should not be applied to a water body if the WTR will be applied to agricultural land, because even one application in the spring may contaminate a
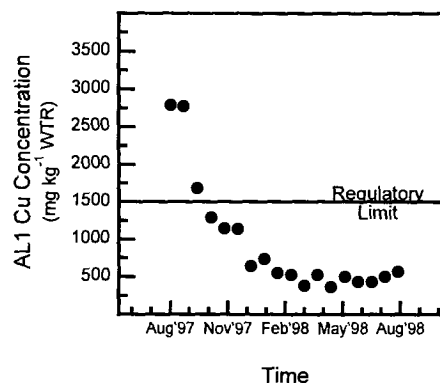

Fig. 4. Copper concentrations in the AL1 water treatment residual determined by the EPA 3050 method from August 1997 to August 1998.

significant percentage of the WTR produced at a facility. The use of Cu also would increase the cost of land application due to increased WTR sampling and analysis cost to determine when Cu concentrations are lower than the regulatory limit.

CONCLUSIONS

The method used to dewater WTR can alter its ability to decrease extractable P concentrations in soils with above-optimum P. Dewatering WTR by freezing or drying at 40°C created a structured material that was much less efficient at reducing Mehlich 3 P concentrations and $CaCl_2$-soluble P concentrations compared with WTR dewatered to only 4.5% solids. Large decreases in Mehlich 3 P concentrations and $CaCl_2$-soluble P concentrations, however, were observed for all WTRs and dewatering methods. These results indicate that above-optimum P concentrations on agricultural land can be rapidly reduced by application of WTR. Reduction of extractable and soluble P concentrations should reduce the amount of P that is transported from a field

Table 5. Average concentrations of 15 metals in the FE1 water treatment residual using the EPA 3050 method. Samples collected every 3 wk for 1 yr.

Metal	FE1 Range	Meant	Median	SD‡	CV§	Seasonal¶	Reg. limit††
	mg kg^{-1}				%	$P>F$	mg kg^{-1}
Ag	0.25–2.7	1.08	0.65	0.79	73	0.35	–
As	1.9–10.3	5.8	4.3	3.0	51	0.58	41
Ba	66.2–353	175.5	185.8	77.5	44	0.82	–
Be	0.03–2.18	0.81	0.78	0.60	74	0.04	–
Cd	0.03–9.31	1.56	0.26	3.04	195	0.01	39
Cr	8.1–253	105.8	63.1	103	97	0.03	1 200
Cu	120–475	247.5	232.2	113	46	0.48	1 500
Hg	0.01–0.12	0.04	0.03	0.03	88	0.84	17
Ni	8.1–62	29.2	18.8	21.3	73	0.09	420
Pb	1.5–13.5	6.6	6.6	3.3	50	0.70	300
Sb	0.03–0.80	0.13	0.03	0.24	196	0.07	–
Se	0.05–5.9	2.0	1.7	1.7	87	0.39	36
Tl	0.03–0.25	0.08	0.05	0.07	92	0.01	–
V	14.0–130	57.3	34.0	45.1	79	0.10	–
Zn	44.4–206	90.4	70.9	50.3	56	0.25	2 800

† Sixteen samples collected from Aug. 1997 to July 1998.
‡ SD=standard deviation.
§ CV=coefficient of variation.
¶ Seasonal=samples grouped by season of the year. The Kruskal-Wallis Test was used to test for significant differences in seasonal fluctuations of the metal concentrations.
†† Reg. limit=USEPA 40 CFR Part 503 concentration limits for land application of sewage sludge.

in runoff or leachate if all other factors affecting P losses are equal. Seasonal differences in the metal concentrations of WTR were small, and all the metal concentrations, except for Cu, were much lower than regulatory limits. These results suggest that seasonal changes in metal concentrations would not be an impediment to agricultural use of WTR, but Cu should not be added to reservoirs to control algae if the WTR will be applied to agricultural land.

ACKNOWLEDGMENTS

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