Fate and Transport Potential of Phosphorus in Sandy Soils Under Long-Term Municipal Wastewater Irrigation

Grant B. Weinkam, Mark T. Brown, David Kaplan, Mark Clark, and Matthew Cohen

Treated wastewater can be applied to landscapes to reduce direct surface water discharges of effluents that contain elevated nutrient concentrations and provide a reliable source of irrigation water (Feigin, Ravina, & Shalhevet, 1991; NRC, 2012; USEPA, 2012). This practice should become increasingly common as reuse of wastewater is expected to increase throughout the United States and globally (Bixio et al., 2008; Miller, 2006). Under long-term application, however, physical, chemical, and biological characteristics of the receiving environment can be altered, influencing phosphorus (P) retention and transport (Jaiswal & Elliott, 2011). While it is critical to understand the long-term implications of slow-rate application to receiving soil and water systems, many studies are conducted only on small spatial and temporal scales; for instance, from one to five years. (Falkiner & Smith, 1997; Rosabal et al., 2007).

Phosphorus applied to landscapes can be chemically adsorbed onto soil particles, taken up by plants, or leached from the soil profile (Barton et al., 2005). While the leached fraction of P in terrains with high slopes and silty clay-rich soils can usually be considered a negligible amount for watershed loading calculations (Aronsson et al., 2014; Liu et al., 2012), in areas with coarse-textured soils of low P-sorbing capacity and shallow groundwater, downward movement of P from organic wastes and other fertilizer sources is potentially significant (Eghball, Binford, & Baltensperger, 1996; Lu & O'Connor, 2001; O'Connor et al., 2005).

Under short-term, low-loading scenarios, the majority of applied effluent P can be expected to surface-sorb onto soil particles in the top 15 cm, or centimeters (Lin & Banin, 2005). Under continuous, long-term application, however, soil layers can become saturated and leach undesirable concentrations of P (Andres & Sims, 2013; He et al., 1999) to groundwater and surface water systems (Kim et al., 2007; McCobb et al., 2003). In early estimates (USEPA, 1981) and field studies (Elliott & Jaiswal, 2011; Lin & Banin, 2005; Moura et al., 2011), receiving surface soils have been shown to become P-

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saturated in as little as 10 to 15 years. This saturation can limit the useful life of a site, with coarse-textured soils and shallow underdrains discharging to sensitive water bodies, to 20 to 60 years (USEPA, 1977; USEPA, 2006).

The soil's ability to sequester P is dependent on physical and chemical soil characteristics, effluent constituent makeup, hydraulic loading rates, and management practices (Agyin-Birikorang, O'Connor, & Brinton, 2008; Lin & Banin, 2005; Walter et al., 1996). The most influential factors affecting fate and transport of dissolved P in sandy soils are the concentrations of charged molecules that can generate surface coatings of iron (Fe) and aluminum (Al) oxides and hydroxides, precipitate with calcium (Ca) and magnesium (Mg), or interact with clays and organic matter (Parfitt, Atkinson, & Smart, 1975; Rajan, 1975; Ryden & Pratt, 1980). Under acidic conditions, phosphates will preferentially bind with Al and Fe (Eckert & Watson, 1996; Kleinman & Sharpley, 2002), with an increased precipitation of Ca and Mg phosphate minerals with increasing soil alkalinity (Hu et al., 2005; Lin & Banin, 2005). As pH can be elevated under effluent loading (Hu et al., 2006), this shift can be expected to influence, and potentially weaken, the strength and formation of P complexes in receiving soils. As these chemicals can strongly control P sorption and transportability, there is a need to better define how long-term effluent irrigation can influence these key constituents. Additionally, as the incorporation of Al-rich amendments have been shown highly effective at
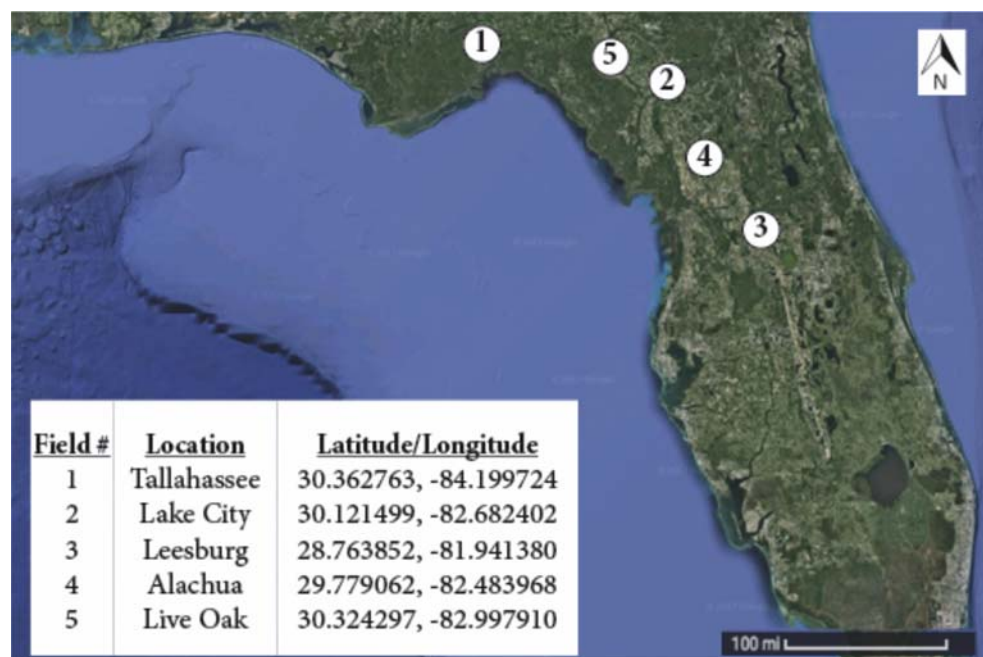

Figure 1. Sprayfield reference numbers and sampling location information (Google Earth, 2016).

Field #	Location	Latitude/Longitude
1	Tallahassee	30.362763, -84.199724
2	Lake City	30.121499, -82.682402
3	Leesburg	28.763852, -81.941380
4	Alachua	29.779062, -82.483968
5	Live Oak	30.324297, -82.997910

reducing P leachability in other heavily loaded sites (Agyin-Birikorang, 2007), there is also value in determining the efficacy of such approaches for P control in effluent-irrigated soils having undergone chemical shifts.

As soils receiving effluent can chemically react differently than systems loaded with more traditional inorganic and organic P sources (Agyin-Birikorang et al., 2008; Holloway et al., 2001; Lombi et al., 2004), there is a need for further investigation into how P retention and transport potential are affected over extended timeframes.

To help address research needs (Jacangelo et al., 2012; National Research Council, 2012) and reduce uncertainties about systems under effluent irrigation (O'Connor, Elliott, & Bastian, 2008; Scott, Faruqui, & Raschid-Sally, 2004; Xu et al., 2010), this study aims to provide a more complete understanding of how wastewater application over long time periods (over 30 years) influence soil constituents and P retention in surface and subsurface horizons, up to 3 meters (m). The impact of chemical changes on leaching rates could then be deter-

mined through column experiments with reconstructed soil cores up to 2 m in depth. In total, significance of the long-term effluent irrigation was determined for influential soil constituents (pH, Fe, Al, Ca, Mg, water extractable P [WEP], Mehlich-3 P [M3P]), P leaching, P sequestration and uptake, and the ability of an aluminum soil amendment to improve P retention.

Materials and Methods

Site Description

The study was conducted at five wastewater sprayfields located across northern and central Florida having minimal topographic relief (Figure 1). Sprayfields, ranging in size from 45 to 810 hectares (ha), received secondarily treated municipal wastewater effluent through pivot irrigation for 28 to 38 years. Soils were made up primarily of fine and coarse sands, with predominant soil types identified in Table 1. Prior to effluent application, fields were used as low-intensity pasture operations, and applications of P and other soil amendments

were assumed to have been minimal. All sprayfields were under cultivation, predominantly growing and harvesting bahiagrass (*Paspalum notatum*). Effluent hydraulic loading rates ranged from 2.5 to 5 cm per week. Historic P concentration in effluents were unknown, but were assumed to exhibit water quality trends similar to other systems in the U.S., dropping substantially from system inception (as high as 10-15 mg/L[-1]) to present day (1-2 mg/L[-1]) (Jacangelo, 2012; Overman, 1982).

Soil Collection and Analysis

At each sprayfield, soils were collected at six sampling locations: three within the sprayfield and three from nearby (0.1-1.0 kilometers [km]) "control" sites considered to be representative of soil unaffected by effluent application or other major soil alterations. Locations for soil cores were randomly selected within, and just outside of, the sprayfield areas. At each location, two soil cores were collected to a maximum depth of 3 meters: one core for soil analysis and the other

Table 1. Sprayfield reference numbers, spatiotemporal characteristics, soil type, and sampling information.

Field #	Area (ha)	Years of Irrigation	Predominant Soil Types	Avg sample depth (cm)
1	810	32	Kershaw fine sand	300
2	140	28	Blanton fine sand and Bonneau-Blanton complex	135
3	130	34	Apopka sand and Candler sand	275
4	45	38	Arredondo fine sand	250
5	63	28	Falmouth-Bonneau-Blanton complex	90

Table 2. Analytical soil chemistry methods.

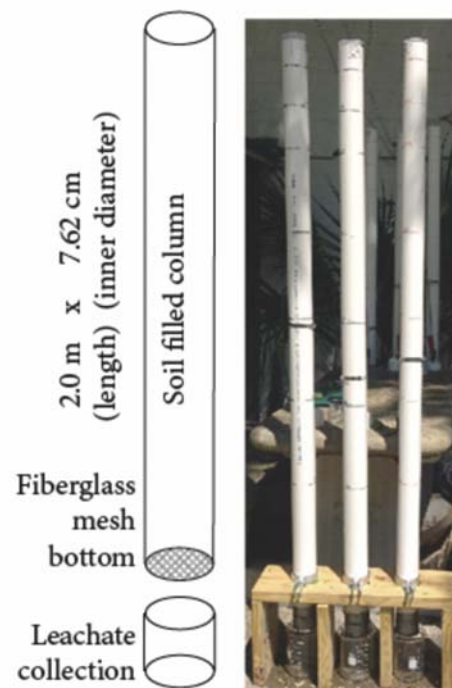| Analyte	Soil Extraction Method	Analytical Method	Analytical Instrument
Ca, Mg, Fe, Al, M3P	Mehlich-3	EPA 200.7	Inductively Coupled Plasma Spectrophotometer (Spectro Arcos)
WEP/DRP	Water	EPA 365.1	AQ2 Discrete Analyzer (Seal Analytical)
pH	Water	EPA 150.1	Orion Benchtop pH Meter
Organic matter	Oven Dry	Loss on ignition	N/A

Figure 2. Diagram of single reconstructed soil column (left). Experimental leaching column setup for three effluent irrigated soil cores. For each field, columns were dripped with 350 m/L of deionization water or wastewater effluent over one hour, every one to two days, for a total of three weeks (right).

for soil column leaching experiments. Soils were extracted with a 7.62-cm-diameter hand auger and sectioned into eight depth increments: 0-15, 15-30, 30-60, 60-90, 90-120, 120-150, 150-230, and 230-300 cm. All sites contained predominantly sandy soils with low organic matter content and minimal stratification or other apparent structure. When clay horizons were encountered, which occurred in Fields 2 and 5, sample depth was limited to 165 cm and 115 cm, respectively. In Field 2, one effluent-receiving site was misidentified, so two effluent-receiving soils and four control soils were collected.

From each depth increment, 500 to 1000 grams (g) of soil were homogenized, sieved (2 millimeters [mm]), oven-dried at 85°C for two to four hours, stored indoors at room temperature, and analyzed within 30 days, according to U.S. Environmental Protection Agency (EPA) methods (Table 2). Chemical concentrations were determined by the University of Florida Analytical Research Laboratory for pH, Ca, Mg, Fe, Al, WEP, dissolved reactive P (DRP), M3P, and organic matter. The WEP was extracted at a ratio of 20 g of soil to 40 m/L of water.

Leaching Experiment Methodology

Soil cores were reconstructed to emulate field conditions in a series of leaching experiments to determine effluent effects on phosphate leachability. Columns were constructed of polyvinyl chloride (PVC) cylinders at 200 x 7.62 cm (length times internal diameter), and fitted with fiberglass screen bottoms for comparative experimentation (Figure 2). Each soil depth fraction was added to the cylinder, with light repacking in the order that it was extracted.

Differences in DRP leaching between effluent-irrigated and nonirrigated soils were determined through three experiments:

◆ Experiment (A) involved the addition of only deionization (DI) water to determine DRP loss under pure water application.
◆ Experiment (B) involved the application of municipal treated wastewater to determine leaching and uptake of DRP applied with effluent irrigation waters. Effluent was collected and sampled weekly from the University of Florida campus wastewater treatment facility, where DRP concentrations were1.2 ± 0.6 mg L^{-1} on average.
◆ In Experiment (C), surface soils (15 cm) were amended with an aluminum-rich substrate

and then applied with effluent to determine amendment influence on DRP transport. Over the course of all experiments in total, 8 L (178 cm) were applied to each column. Liquids were dripped into columns as 350-m/L (7.62 cm) applications taking place over one-hour time periods. This application occurred every one to two days, for a total of three weeks.

To generate leachate in experiment (A), 700-1050 m/L was applied to columns over 24 hours and two samples were collected from the generated leachate. Following two days of nonliquid applications, to allow for all applied waters to completely percolate through the soils, a subsequent 700 m/L was added, and an additional two samples were collected. From the percolated waters, 20 m/L subsamples were filtered through a polytetrafluoroethylene (PTFE) 0.45 μm pore diameter membrane filter, stored below 4°C, and analyzed within 28 days.

Following the initial DI water flushes, for experiment (B), soil columns were loaded with 350 m/L (7.62 cm) of effluent per day for seven consecutive days. To collect random and representative samples, single-leachate samples were collected on two, nonconsecutive days following effluent application. After allowing two days for effluents to percolate through the system, the columns received a subsequent application of 700 m/L of DI water, and one sample was collected. The subsequent DI application was conducted to determine DRP concentrations following the application of effluents, and for comparative analysis of DI, only flushes for experimental conditions A, B, and C were done.

For experiment (C), an aluminum-rich drinking water treatment plant residual, found effective at reducing DRP loss in soils receiving manure, biosolid, and/or granular phosphate fertilizers (Agyin-Birikorang et al., 2007; O'Connor et al., 2002; Silveira, Miyittah, & O'Connor, 2006), was incorporated into the top 15 cm of column soils. This soil portion was augered out and homogenized with a relatively high rate (56 Mg ha^{-1}) of 2-mm sieved residual. Elemental concentrations in the residual were assumed equivalent to previously analyzed samples containing Al (11-89 g kg^{-1}), Fe (2-4 g kg^{-1}), and TP (2.8 g kg^{-1}) (Agyin-Birikorang et al., 2009; Miyittah-Kporgbe, 2004). Following a light repacking of the soils, columns were loaded exactly the same as experiment (B), resulting in the collection of two effluent samples and one DI water sample.

In total, for each field, effluent-irrigated soil and control soil columns yielded twelve DI water flush samples for experiment (A), six effluent flush and three DI flush samples for experiment (B), and six effluent flush and three DI flush samples for amended soils in experiment (C).

Table 3. Average and range of parameter concentrations for all depths sampled in all effluent irrigated soils and nonirrigated control soils.

Field #	Soil Condition	pH	WEP	M3P	Ca	Mg	Fe	Al
					mg kg^{-1}			
1	Effluent	7.5** (7.3-7.6)	2.9** (1.1-9.0)	57** (18-124)	316* (72-1217)	40 (8-140)	70 (26-136)	481 (251-731)
	Control	6.1 (5.0-7.4)	0.3 (0.0-0.9)	29 (6-80)	113 (18-699)	30 (1-179)	92* (38-145)	634* (225-1087)
2	Effluent	7.1** (6.7-7.5)	1.2** (0.1-2.4)	55** (8-138)	509* (180-1140)	88* (21-179)	66 (26-106)	552 (431-825)
	Control	6.0 (4.5-7.3)	0.3 (0.0-1.6)	10 (0-78)	229 (41-1148)	43 (0-179)	54 (25-106)	603 (396-946)
3	Effluent	7.7** (7.4-7.9)	2.2** (0.5-4.6)	96 (21-274)	357 (95-1246)	24 (1-73)	104 (26-181)	364 (112-591)
	Control	6.4 (5.6-7.3)	0.7 (0.0-3.0)	72 (11-206)	268 (55-1421)	22 (0-115)	95 (38-215)	454 (140-1666)
4	Effluent	7.7** (7.5-8.0)	5.8** (4.0-9.5)	235 (46-427)	1147 (151-3841)	111 (11-406)	113 (55-140)	735 (195-1363)
	Control	6.8 (5.5-7.6)	2.1 (0.1-4.7)	301* (210-461)	1131 (281-5725)	133 (39-291)	131* (88-206)	1481** (1134-1759)
5	Effluent	6.7** (5.7-7.7)	0.5 (0.1-2.2)	39 (2-150)	431 (75-1502)	21 (0-81)	98 (31-282)	404 (142-945)
	Control	5.8 (4.7-7.2)	0.5 (0.2-2.3)	29 (1-168)	546 (152-1988)	45* (7-126)	137 (50-496)	643** (350-964)
All Fields Avg	Effluent	7.3 ± 0.4	2.5 ± 2.1	96 ± 80	552 ± 340	57 ± 40	90 ± 21	507 ± 146
	Control	6.2 ± 0.4	0.8 ± 0.8	88 ± 121	457 ± 409	55 ± 45	102 ± 34	763 ± 408

* Significant at the 0.05 probability level
** Significant at the 0.01 probability level

Calculations and Statistical Approach

Data generated from soil sampling and leachate experiments were analyzed to determine differences in soil constituents that can influence P transportability (pH, Ca, Mg, Fe, Al) and DRP leaching from effluent irrigated and nonirrigated soils. Soil concentrations for WEP, M3P, Ca, Mg, Fe, Al, and pH were compared to identify significant differences among different effluent-irrigated and control soils, as well as between different fields and depths. Differences between effluent-irrigated and control soils were calculated by subtracting averaged effluent values from averaged control values at each field and depth increment. Positive and negative values indicate higher or lower analyte masses, respectively, in effluent-irrigated soils. Significance was determined using a two-tailed T-test at the 0.05 and 0.01 probability level.

Leached DRP concentrations were calculated for each field and irrigation condition by averaging all samples. The effects of the different soil conditions and treatments on DRP uptake and leaching under effluent application were compared by calculating the percent DRP removed, relative to the initial effluent concentration added. Significance was determined using a one-tailed T-test at the 0.01 probability level.

Results

Effluent Irrigated Soil and Control Soil Concentrations

Fields have inherent natural and management-induced variations in soil chemical properties, but the effects of effluent loading were clearly identifiable for some constituents (Table 3). Average pH for all effluent-receiving fields was higher (7.3 ± 0.4) than corresponding control sites (6.2 ± 0.4). In Field 5, pH was the only parameter showing a significant difference. In Fields 1 through 4, average WEP concentrations were significantly higher in irrigated fields (3.0 ± 2.0 mg kg-1) than control sites (0.9 ± 0.9 mg kg^{-1}). Other parameters with significant differences in multiple fields included higher M3P (Fields 1, 2), higher Ca (Fields 1, 2), and lower Al (Fields 1, 4, 5). In Field 4, naturally present concentrations of M3P in the soils were 10 times higher than other fields, especially in deeper horizons and controls. No significant differences were observed for Mg, Fe, and organic matter content. For all surface soils (0-0.3 m), organic matter content ranged from 2 to 5 percent, and no clear differences were identified between effluent irrigated and control sites.

Soil Differences With Depth

To highlight surface and subsurface specific differences, mean soil parameter concentrations were separated by depth increment (Table 4). Positive and negative values indicate higher or lower analyte mass when irrigated soils were subtracted from control soils. Results show that in all surface soils, to a depth of 0.6 m, higher M3P concentrations were identified in effluent receiving fields. In Fields 1, 2, 3, and 5, 60 percent of increases were in the upper 0-0.6 m. Except in Field 4, higher M3P was observed to maximum depths under effluent-receiving conditions. In Field 4, naturally occurring phosphatic-rich soil deposits in the deeper, control soil horizons, resulted in the M3P values observed.

The trend is similar to other P loaded systems, where an average 26 percent of the increased mass was present in the upper 0.15 m, followed by substantial accumulation in soils directly below the surface (0–0.6 m), where 63 percent of the total elevated M3P was stored. Similarly, higher WEP was identified to maximum sampled depth in all fields, except Field 5. Contrary to the substantial increases in M3P in surface soils, however, on average, only 8 percent of the total elevated WEP was located in the upper 0-0.15 m, with only 29 percent found in the top 0.6 m of soil. Results indicate that upon accumulation of significant P as M3P in surface horizons, more rapid transport can occur to lower horizons through P in the form of loosely bound WEP.

In all effluent-receiving systems, excluding Field 4, samples were identified as having higher Ca concentrations to a minimum of 0.6 m depth. In Fields 1, 2, and 3, where Ca was found higher at all sampled depths, 43 percent of increases were located in the upper horizons (0-0.6 m). Lower extractable Al was found to maximum depths of all soils under effluent irrigation. Minimal differences were identified for Fe and Mg.

Column Leaching

Due to issues encountered with high concentrations of suspended colloidal solids and the failure of one column for Field 2, the number of replicates was reduced for some experiments. For experimental condition (A), a minimum of eight samples were collected, and for each part (B) and (C), a minimum of four effluent samples and two DI samples were collected.

Of the DI water samples collected, DRP concentrations from experiments (A), (B), and (C) showed significantly higher values from ef-

Table 4. Cumulative differences between mean effluent irrigated soils and control soils analyte concentrations with depth.

Field #	Depth (m)	WEP	M3P	Ca	Mg	Fe	Al
		Differences between effluent irrigated and control soils (mg kg^{-1})					
1	0-0.15	1.2	59	610	17	-6	-471
	0-0.3	2.7	101	892	11	-31	-939
	0-0.6	4.6	133	1068	11	-61	-1201
	0-3.0	21.3	225	1622	80	-173	-1223
2	0-0.15	1.6	101	491	61	28	-23
	0-0.3	2.9	174	859	103	52	-79
	0-0.6	3.7	206	1010	117	55	-169
	0-1.5	4.9	252	1678	291	58	-212
3	0-0.15	1.3	34	70	-19	-32	-46
	0-0.3	2.8	60	202	-19	-20	-66
	0-0.6	4.5	100	366	-10	4	-77
	0-3.0	11.6	173	668	15	48	-728
4	0-0.15	4.2	44	-561	-17	9	-552
	0-0.3	9.1	74	-835	-88	16	-1172
	0-0.6	14.4	67	-761	-132	32	-2020
	0-3.0	37.9	-615	312	-146	-188	-6349
5	0-0.15	0.6	41	172	-30	-84	-156
	0-0.3	0.7	61	93	-61	-96	-392
	0-0.5	0.4	68	183	-66	-108	-564
	0-1.0	0.0	81	-457	-133	-130	-1310

† Negative values indicate lower concentrations for effluent irrigated soils.

†† For each field, three effluent and three control samples for 0-0.15m. Total samples at maximum depth as (Field #: effluent samples: control samples); (1: 24: 24), (2: 11: 24), (3: 23:24), (4: 24:22), (5: 17:14).

fluent irrigated soils when compared to control soils (Figure 3). In Fields 1 through 4, leached DRP concentrations were significantly higher (alpha = 0.01) than paired controls, with Field 5 being the only site showing no significant differences. These discrepancies clearly show the influence that the long-term effluent loading has had on P transport behavior in the receiving soils.

Averaging all values, mg DRP L^{-1} for Fields 1 to 5 were 1.0 ± 0.3, 0.6 ± 0.2, 2.1 ± 0.5, 2.5 ± 0.7, and 0.1 ± 0.1, respectively (average, 1.3 ± 0.4 mg DRP L^{-1}). Averaged DRP for control soils (identified by the horizontal lines across each series of experiments) were considerably lower (0.01, 0.01, 0.37, 0.14, and 0.19; average, 0.14 ± 0.09 mg DRP L^{-1}) and had small variability (standard deviation [SD] = 0.15 mg L^{-1}, variance = 0.02). Under the assumptions that the tests are representative of soil conditions, current irrigation practices, and a percolation rate of 0.64 m $year^{-1}$, effluent irrigated soils are estimated to leach an average ± SD of 8.0 ± 6.4 kg P ha^{-1} per year compared to control soils leaching only 0.9 ± 0.9 kg P ha^{-1} per year.

Higher values were identified in some initial experiment (A) flushes. Values were possibly associated with the exposure of soil surfaces and flushing of loosely bound P generated from soil disturbance and transfer. Additionally, as Styles & Coxon (2006) showed that dry soils can extract two to three times more DRP than wet soils, the higher level of water saturation in (B) and (C) can also partially explain the higher releases identified in (A). The DRP leached from nonamended (B) and aluminum-amended soils (C) were found to not be significantly different (alpha = 0.01).

The percentage change in effluent DRP concentration after percolating through the soil columns emphasizes that a significant change (alpha = 0.01) has occurred in the P sequestration capabilities of Fields 1 through 4 when compared to controls (Figure 4). Field 5 was the only field where no significant difference in uptake efficiency was identified. Effluent was applied to columns with an average DRP of 1.2 ± 0.6 mg L^{-1} (range 0.1-2.1 mg L^{-1}), and exited Field 1 through 4 columns at 0.6-2.3 mg L^{-1}, for an average increase of 16 percent ± 75 percent. All control soils decreased DRP by an average of 88 percent ± 8 percent, equivalent to a percolated DRP of 0.13 mg L^{-1}. Through these results the leaching experiment methodology was shown to be valid and representative, as indicated by the control soil DRP uptake values producing results similar to previous researchers, where unimpacted soil P sequestration was in the range of 90 to 95 percent under in situ conditions.

Discussion

Soil Changes in Long-Term Effluent Irrigated Landscapes

Results highlight that changes to soil chemistry and P leaching can be significant in systems with heavy effluent loading, high hydraulic conductivity, and minimal topographic relief. Given the higher surface M3P concentrations identified for effluent irrigated sites, it's unlikely that the harvested biomass is regularly taking up all effluent P applied to soils. Additionally, as a substantial portion of the elevated WEP identified (69 percent) is occurring below typical soil sampling and plant rooting depths (>0.6 m), it is reasonable to assume that the nutrient is therefore available for subsurface leaching and loss. These results show that the chemical changes that have occurred in the system can increase P lability, regardless of the

M3P increase in the system. This is especially evident in Field 4, where high concentrations of M3P were identified to maximum depths in both effluent and control soils, with only effluent soils showing significant increases in WEP.

In Field 5 the influence of effluent loading was evident in some soil parameters (pH, Al) but not for the WEP or P leached. Possible explanations could be associated with lower effluent P concentrations or application rates, or the specific physical characteristics of the site. The location of this field contained a thick clay horizon at depths as shallow as 0.7 m that likely produced more regular soil saturation and flushing of soil elements when compared to other fields.

The elevated surface soil Ca concentrations finding is in general agreement with other researchers (Hu et al., 2006), who also identified similar trends in soils under effluent application. Lower Al concentrations in effluent-receiving soils is possibly associated with an increase in insoluble Al precipitates formed under increased contribution of wastewater anions or attributed to podzolization due to organic ligands added through effluent applications (Jaiswal & Elliott, 2011). Organic matter content showed little variation between effluent and control sites, and therefore was determined to have little influence on the differences identified in P accumulation and transport. Changes in soil pH likely had a significant impact on P transport, and may have been heavily influenced by sodium (Na) loading associated with the effluent irrigation. Measurement of sodium loading and accumulation could help determine ion-specific contributions to pH shifts.

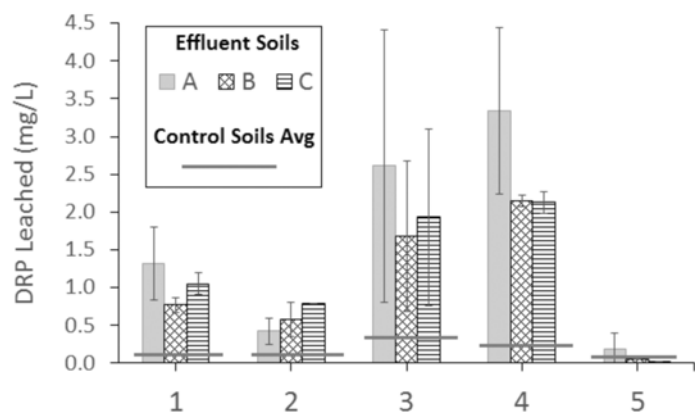These findings can be important, as these elemental shifts can directly influence the appli-

Figure 3. Dissolved reactive phosphorus concentration (±1 standard deviation) leached from effluent irrigated soils and control soils for experiments (A), with deionization water added; (B), with wastewater effluent added; and (C), with wastewater effluent added after surface soils amendment with aluminum substrate.
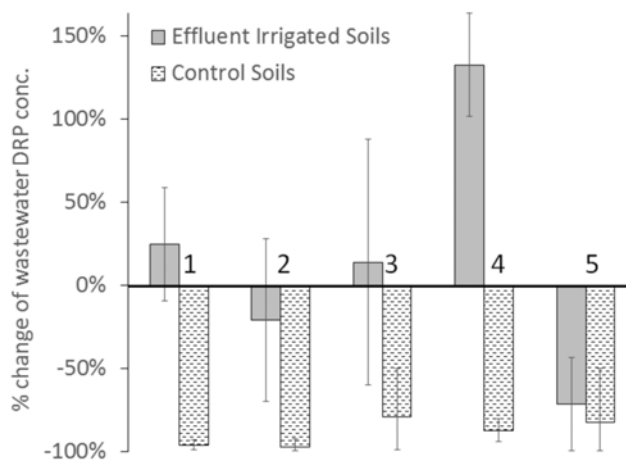

Figure 4. Percentage change (±1 standard deviation) of dissolved reactive phosphorus concentration from applied wastewater after percolating through effluent irrigated soils and control soils.

cability and accuracy of commonly applied P saturation calculations that do not typically take into account changes in pH, Ca, or WEP, such as the P saturation ratio (PSR) and the soil P storage capacity (SPSC). Validation of these measures and associated land management determinations may warrant further conformational studies in similarly loaded landscapes.

Phosphorus Leaching Changes in Long-Term Effluent Irrigated Landscapes

As DRP leached from Fields 1 through 4 showed significant, yet reasonable, differences between the two field conditions, the methods used were shown to be valid and representative of hypothetical field conditions. The values identified from effluent irrigated plots were also above stated critical water quality concentrations for shallow groundwaters (Breeuwsma, Reijerink, & Schoumans, 1995) and many surface waters (FDEP, 2012; USEPA, 1986). These results highlight that the P sequestration capabilities have decreased to such a point that the soils used as environmental buffers are no longer functioning as effective water quality protection measures (USEPA, 1977; USEPA, 2006). While similar soil trends of accumulation and downward transport of P have been recognized up to 10 meters at sites disposing of effluent at high rates (Lin et al., 2006; Moura et al., 2011), such transport is typically not reported at depths greater than 0.5 m in systems under long-term effluent irrigation.

While columns were loaded with liquids at a rate higher than typically-applied-for-irrigation purposes, under field conditions, some mobilized DRP may have been retarded or resorbed within upper soil layers. As frequent storms in the region can provide substantial pulses that saturate soils to depths sampled, results could be considered representative of such circumstances. Installation of field lysimeters could yield additional information on how changes in water application rates or storm intensity influence P leaching in undisturbed landscapes.

Given the unique chemical shifts in the receiving systems, where altered pH, Ca, and Al have influenced strength, binding preference, and ultimately, lability and transportability of P (Cho, 1991; Isensee & Walsh, 1972; Nair, Graetz, & Portier, 1995), numerous passive and active management recommendations are suggested to increase terrestrial P sequestration.

Managing Phosphorus in Long-Term Effluent Irrigated Landscapes

While the aluminum amendment in experiment (C) had previously been shown effective at controlling P transport in other P loaded systems, it was not identified as a worthwhile control strategy under these experimental conditions. As an increase in pH can decrease the sorptive strength of Al to P, this shift may have partially driven the inadequacy of this P control method. Additionally, as highly leachable soil, WEP was elevated deep into the receiving soils (up to 3 m), and such an approach to P control may have increased success under different application scenarios. Suggestions for improved P uptake and control include incorporation of the amendment to depths greater than tested in this experiment (15 cm), integration of amendment prior to significant effluent loading has taken place, and/or its incorporation as a component of subsurface permeable barrier systems lining sensitive receiving surface waters (McCobb et al., 2003; Sibrell & Tucker, 2012).

As many landscape loading models assume that nearly all effluent P is accumulated and freely available for plant uptake in receiving surface soils, these findings indicate that actual accumulation and plant available amounts can be substantially lower than the total load applied. Given these considerations, recommended vegetative management strategies include control and harvest of plant biomass (Martinez et al., 2011) and planting of deep rooted vegetation (Falkiner & Polglase, 1997; Minogue et al., 2012; Stewart et al., 1990) to provide stable vegetative sinks and increase uptake from lower soil horizons (Chakraborty et al., 2011; Ibrikci et al., 1994; Nair & Graetz, 2004). Lastly, as irrigation with effluent, even at low P concentrations, has been found sufficient to meet turf nutritional needs (Fan et al., 2014; Martinez et al., 2011; Morgan et al., 2008), increased monitoring of irrigation system performance and education on appropriate irrigation rates can help avoid undesirable vegetative growth and potential groundwater loading (Arrington & Melton, 2010; Jacangelo et al., 2012).

Conclusions

This research provides a more complete understanding of P attenuation in sandy soils receiving treated wastewater, and highlights potential limitations in long-term operation. In these systems, irrigation with wastewater has resulted in significant alterations to soil chemical properties and P transportability, both in surface soils (0-0.3 m), and, in most sites, to depths up to 3 meters. Increased WEP with depth was the most important finding, as the results indicate unique system behavior where substantial accumulation of soil M3P is not necessarily required prior to significant increases in leachable P. High hydraulic and P loading rates over extended periods of time, and effluent-induced soil chemistry changes such as lower extractable Al and higher pH, help explain the occurrence of higher WEP deep in soil profiles. Increases in effluent-associated Ca were expected to limit P leachability, but appeared to have little influence on deep (>2 m) DRP transport.

As many of the soil analytes impacted can heavily influence P transportability, incorporation of these findings into P saturation and transport calculations can help increase the impact of legislative and remedial actions taken to maintain and improve nutrient-sensitive basins with strong groundwater and surface water connectivity.

Acknowledgments

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