

Effect of Tomato Packinghouse Wastewater Properties on Phosphorus and Cation Leaching in a Spodosol

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Land application of wastewater is a common practice. However, coarse-textured soils and shallow groundwater in Florida present favorable conditions for leaching of wastewater-applied constituents. Our objective in this study was to determine phosphorus (P) and associated cations (Ca, Mg, K, Na) leaching in a Spodosol irrigated with tomato packinghouse wastewater. We packed 12 polyvinyl chloride soil columns (30 cm internal diameter × 50 cm length) with two soil horizons (Ap and A/E) and conducted 30 sequential leaching events by irrigating with wastewater at low (0.84 cm d⁻¹), medium (1.68 cm d⁻¹), and high (2.51 cm d⁻¹) rates. The control treatment received deionized water at 1.68 cm d⁻¹. Leachate pH was lower (6.4–6.5) and electrical conductivity (EC) was higher in the wastewater-treated columns (0.85–1.78 dS m⁻¹) than in the control treatment (pH 6.9; EC, 0.12 dS m⁻¹) due to the low pH (6.2) and high EC (2.16 dS m⁻¹) of applied wastewater. Mean leachate P concentrations were greatest in the control treatment (0.70 mg L⁻¹), followed by the high (0.60 mg L⁻¹) and low and medium wastewater-treated columns (0.28–0.33 mg L⁻¹). Leachate concentrations of Na, Ca, Mg, and K were significantly ($P < 0.05$) greater in wastewater-treated columns than in the control. Concentrations of P, Na, and K in leachate remained lower than the concentrations in the applied wastewater, indicating their retention in the soil profile. In contrast, leachate Ca and Mg concentrations were greater than in applied wastewater during several leaching events, suggesting that additional Ca and Mg were leached from the soil. Our results suggest that tomato packinghouse wastewater can be beneficially land-applied at 1.68 cm d⁻¹ in Florida's Spodosols without significant P and cation leaching.

LAND APPLICATION of agricultural and industrial residual byproducts, such as manures and composts, biosolids, and wastewater effluents, is a common method of recycling these materials. Reuse of wastewater has many benefits, including using the water for irrigation while improving the physical and chemical properties of soils and conserving potable water supplies. However, application of residuals at agronomic rates (typically N-based) to fields close to sensitive water bodies or application at very high rates (≥ 50 Mg ha⁻¹) can impair water quality (O'Connor et al., 2005). Many water bodies in the United States are phosphorus (P) limited, and wastewater discharges from domestic and industrial facilities and leaching and runoff from agricultural landscapes are the main sources of excess P in these systems. Because many Florida soils are coarse textured, are low in organic matter, and have limestone fractures, the leaching of P into groundwater is a concern in soils amended with residual byproducts (Elliott et al., 2002; Nair and Graetz, 2002). In addition, if the groundwater with high P discharges into surface waters, eutrophication may occur and cause water pollution in water bodies.

Long-term land application of liquid wastes has been shown to increase P concentrations in soil (Johnson et al., 2004; Sinaj et al., 2002). Therefore, to be environmentally sustainable, land application practices must consider the fate of P and its potential transport to groundwater. Liquid waste contains a diverse mix of elements. During land application, these elements may be leached from or retained in the soil or taken up by plants. Element fate may vary depending on soil type, wastewater properties, and application rate. For instance, 2-yr application of domestic wastewater (5.8 mg L⁻¹ total P) to intact clay loam cores (0.46 m wide and 0.7 m deep) resulted in 16% leaching of applied P, ostensibly due to low uptake by plants (Barton et al., 2005). Similarly, application of potato-processing wastewater (3.6 mg L⁻¹ total P) to intact sandy loam lysimeters (0.3 m wide and 1.5 m long) resulted in relatively higher P concentration in leachate (3.5–5.6 mg L⁻¹) than in wastewater, suggesting that additional P was contributed by soil reserves (Zvomuya et al., 2005).

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Abbreviations: EC, electrical conductivity; ICP–OES, inductively coupled plasma–optical emission spectroscopy; LI, leachate to irrigation; PV, pore volume;

Soil properties are known to affect P leaching; therefore, knowledge of P reactions with soil components can provide insights into the likelihood of P leaching. For example, only 3 to 5% of total P (36 mg L⁻¹ total P) applied in dairy wastewater to undisturbed silt loam lysimeters (0.5 m wide and 0.7 m long) was leached (leachate P: 1.1–1.4 mg L⁻¹), whereas 95 to 97% of the applied P was adsorbed by soil Fe and Al (Toor et al., 2005a). When potato-processing wastewater with low (1.1 mg L⁻¹) and high (21 mg L⁻¹) total P concentrations was applied to sandy loam lysimeters (0.16 m wide and 0.4 m long), the soil acted as a P source (~345% of P recovery in leachate) in the low wastewater treatment, whereas the soil acted as a P sink (38% P recovery in leachate) in the high wastewater treatment (Mamo et al., 2005). Application of dairy wastewater (70–164 kg total P ha⁻¹) to packed sandy soil lysimeters (0.05 m wide and 1.5 m long) resulted in low P concentration (≤0.08 mg L⁻¹) in the leachate due to retention of P in soil by Ca and Al added with the wastewater (Woodard et al., 2007). Soil organic matter, Mn oxides, and calcium carbonate can also fix P in a soil (Chen et al., 2003).

Leaching of cations, including Ca (3–13 mg L⁻¹), Na (20–40 mg L⁻¹), and Mg (<5 mg L⁻¹), has also been reported after the application of meat-processing wastewater (26 mg Ca L⁻¹, 255 mg Na L⁻¹, and 7 mg Mg L⁻¹) to undisturbed silty loam lysimeters (0.6 m wide and 1.2 m long) and was attributed to preferential flow in soil and removal of cations from soil during previous wastewater applications (Magesan et al., 1999). Therefore, the chemical characteristics of wastewater can play an important role in determining the net loss of a constituent from soil. In summary, soil characteristics such as texture, contents of P, and Fe and Al oxides (Toor et al., 2005a; Zvomuya et al., 2005); wastewater characteristics such as concentrations of P, Ca, and Al (Mamo et al., 2005; Woodard et al., 2007); and uptake of P by plants (Barton et al., 2005) can control the movement of P in the soil profile. In addition, the annual average rainfall of 135 cm in Florida, especially during the months of June to September, which receive 60% of the total rainfall, may enhance leaching of P in sandy soils.

To our knowledge, the effects of land application of tomato packinghouse wastewater on the transport of P and other elements in coarse-textured soils have not been studied. In Florida, there are about 70 tomato packinghouses that pack fresh-market tomatoes, with a typical facility packing about 1.1 million kg of tomatoes in a day. In a packinghouse, approximately 3700 to 68,000 L d⁻¹ freshwater is used for rinsing, washing, and sanitizing tomatoes before packing, which results in the generation of approximately 3800 to 18,000 L d⁻¹ of wastewater (Florida Tomato Committee, 2007). This equates to about 231 million L of wastewater each year in all tomato packinghouses in Florida. Approximately 54% of this wastewater is disposed of on agricultural fields (Florida Tomato Committee, 2007). Thus, our objective was to determine leaching of P and cations (Ca, Mg, K, Na) in a typical Florida spodosol after irrigation with tomato packinghouse wastewater because leaching of P into groundwater and then discharge into surface waters may cause eutrophication, and the leaching of soluble salts may increase the salinity and sodicity of the groundwater. We provide recommendations for the land

application rate of wastewater to reduce P leaching and prevent groundwater contamination in Florida.

Materials and Methods

Study Site

We obtained soil used in column packing from a field located at the University of Florida Gulf Coast Research and Education Center in Wimauma, Florida. The field was in citrus production before 2000 but has not been in cultivation since 2000. The soil at the collection site was a Zolfo fine sand (sandy siliceous, hyperthermic Oxyaquic Alorthods) and was somewhat poorly drained (seasonal water table: 60–106 cm), and the depth to the restrictive layer (Bh horizon) was more than 200 cm (USDA–NRCS, 2009). Soil samples were collected from the surface soil horizon (Ap; 0–17 cm) and the subsurface horizon (A/E; 17–50 cm). We air-dried the collected soil samples for about 1 to 2 wk and sieved using a 2-mm sieve (US sieve No. 10), and a subsample was taken for analyses.

Column Packing, Setup, and Equilibration

We built soil columns by cutting polyvinyl chloride (PVC) pipe (30-cm internal diameter) into 50-cm-long sections. In each PVC column (total surface area: 730 cm²), we packed about 45 kg of air-dried and sieved soil from the A/E horizon into the lower portion (17–50 cm; measured field bulk density = 1.87 g cm⁻³) and 22 kg of soil from the Ap horizon into the upper portion (0–17 cm; measured field bulk density = 1.77 g cm⁻³) to achieve field bulk density values. Columns were packed in 5-cm increments, alternating packing with tapping for 3 min. A similar procedure has been used by other researchers to pack columns (Ashworth et al., 2008; Gao and Trout, 2006; Mamo et al., 2005).

We covered the bottom of each column with an end cap and placed a section of cheesecloth at the bottom of each end cap to prevent sand loss. The end cap was packed to 6.3 cm depth with a mixture of deionized water-washed sand (2.3 kg) and pea gravel (6.25 kg) to create free drainage; this resulted in a bulk density of 1.78 g cm⁻³. We securely attached each packed end cap to the bottom of a PVC column and applied a sealant to prevent water leakage. A hole was drilled at the center of the end cap, and a plastic pipe was attached to allow leachate collection. All packed soil columns were placed in a greenhouse under controlled temperature conditions (30–35°C).

Soil columns were equilibrated by wetting the soil with 11.1 L (15.06-cm depth) of deionized water per column, an amount equivalent to 1 pore volume (PV) of soil. Equilibration removed air bubbles entrapped during the packing process and ensured homogeneity of moisture in columns before experiments (Table 1). Soil columns were freely drained for about 2 wk and then re-wetted by adding deionized water equivalent to a total of 0.78 PV (11.73 cm water depth) at a rate of 0.111 PV (1.68 cm d⁻¹) for 7 d; this application rate prevented water ponding and ensured uniform water infiltration. After three wetting events, the coefficient of variation of leachate volume was <10%, suggesting that all columns had achieved uniform water flow.

Treatments and Leachate Collection

We arranged 12 packed soil columns in a completely randomized design in the greenhouse. Columns received four

treatments, in triplicate, including one control and three wastewater treatments. We collected approximately 400 L of wastewater from a representative tomato packinghouse located in central Florida that washes and sanitizes field-harvested tomatoes for commercial distribution. The collected wastewater was filtered through cheesecloth to remove plant material (leaves, stems) and stored at 4°C for use in wastewater treatments. A subsample was removed and analyzed from stored wastewater each week during the study period.

Wastewater was applied to soil columns at three rates: low (0.84 cm d⁻¹), medium (1.68 cm d⁻¹), and high (2.51 cm d⁻¹). The control treatment received deionized water at the medium rate (1.68 cm d⁻¹). In our study, the four treatments were subjected to compare the effects of irrigation rates (low, medium, and high) and irrigation source (medium rate of wastewater vs. medium rate of deionized water or control). We selected the medium rate (1.68 cm d⁻¹) to be comparable to seepage irrigation rates for tomato crops in Florida, which receive approximately 101 cm of water over a growing season (Santos, 2009). For example, considering a 60-d growing period of tomato crop, the seepage irrigation of 101 cm equates to 1.69 cm d⁻¹. Seepage irrigation, often called subsurface irrigation, is a common method in the sandy soils of southern Florida and consists of elevating the water table through the use of lateral field ditches to allow upward water movement through capillarity in soils with an underlying impermeable layer. Soil columns were irrigated once each day for 30 d, adding a total of approximately 55 L (5 PV) of irrigation water to the high wastewater treatment, 36.3 L (3.3 PV) to the medium wastewater and control treatments, and 18.7 L (1.7 PV) to the low wastewater treatment. We applied water daily for 30 d to mimic the field practice of using land application to recycle packinghouse wastewater; packinghouse operation is seasonal, typically lasting 4 to 6 wk, two times a year. We collected wastewater drained from each column daily (24 h after wastewater application) in 2-L amber glass bottles for a total of 30 leachate sampling events. Leachate volume was measured each day, and about 200 mL as subsample was taken for analysis.

Soil and Water Analysis

We analyzed soil samples from both horizons for sand, silt, and clay using the hydrometer method (Day, 1965). Field bulk density of undisturbed soil cores was determined as described by Blake and Hartge (1986) in samples collected at 5-cm depth intervals from the 0- to 50-cm layer. Particle density of soil horizons was measured using the pycnometer method (Blake and Hartge, 1986). We used the bulk density and particle density of soil samples to calculate the porosity of each soil horizon.

Soil pH was measured with a digital meter (Accumet XL60 dual channel pH/ion/conductivity/dissolved oxygen meter; Fisher Scientific, Pandan Crescent, Singapore) after equilibrating 10 g of soil with 20 mL of deionized water (1:2) for 1 h. Soil electrical conductivity (EC) was measured in a 1:1 (mass/volume) soil to deionized water suspension with the

above digital meter. Total soil organic matter was determined by the wet oxidation method of Walkley and Black (1934). Total P, Al, Ca, Fe, K, Mg, and Na were extracted from soils, in triplicate, using HNO₃ and H₂O₂ (USEPA, 1996) followed by analysis on inductively coupled plasma–optical emission spectroscopy (ICP–OES) (PerkinElmer Optima 2100 DV; PerkinElmer, Shelton, CT). We extracted 4 g of air-dried soil with 40 mL of deionized water (1:10 mass/volume soil to water ratio) by shaking on a reciprocating shaker for 2 h followed by centrifugation at 4000 rpm for 20 min to determine P and cation concentrations. We then filtered the sample through a 0.45-µm membrane filter paper and analyzed the filtrate by ICP–OES to determine water-extractable P and cation concentrations.

Wastewater and leachate pH and EC were measured with the digital meter described above. Chloride concentration in wastewater and leachate samples was determined using a discrete analyzer (AQ2+; Seal Analytical Inc., Mequon, WI). Phosphorus and cation concentrations in wastewater and leachate samples were measured using ICP–OES. We calculated the ionic strength of wastewater using the Marion and Babcock (1976) equation for samples with an EC range from 0.05 to 12.9 dS m⁻¹ as follows:

$$\text{Log } I_c = -1.841 + (1.009 \times \text{Log EC})$$

where I_c is ionic strength (mol L⁻¹), and EC is electrical conductivity (dS m⁻¹).

Statistical Analysis

We calculated descriptive statistics, including mean, standard deviation, range, and coefficient of variation of parameters for leachate samples in Microsoft Excel 2007. The data met the ANOVA assumptions of normal distribution and homogeneity of variance and thus required no transformations. Mean concentration (mg L⁻¹) of each element was multiplied by daily leachate volume (L) to calculate daily loads. We used the least significant difference method to evaluate the statistical significance ($\alpha = 0.05$) of differences in concentrations and daily loads among treatments using the SAS PROC GLM procedure (SAS Institute, 2007).

Results and Discussion

Soil Properties

Soil used in our study was dominated by sand (924–935 g kg⁻¹) with low clay content (2–4 g kg⁻¹) (Table 2). The

Table 1. Prewetting irrigation events with deionized water in all soil columns.

Irrigation event	Application rate cm d ⁻¹	Depth of applied water	Leachate depth	
			Mean ± SD	Range
1	15.06	15.06 (1.0 PV)†	1.35 ± 0.41 (9)‡	0.70–1.97
2§	1.68	11.73 (0.78 PV)	7.26 ± 0.32 (62)	6.45–7.62
3§	1.68	11.73 (0.78 PV)	8.87 ± 0.2 (76)	1.22–1.30
4§	1.68	11.73 (0.78 PV)	9.65 ± 0.21 (82)	9.16–9.88

† PV, pore volume.

‡ Values in parentheses indicate percent recovery of applied water.

§ Each of the irrigation events 2, 3, and 4 were conducted continuously for 7 d, and leachate was collected on Day 8.

Table 2. Selected physical and chemical properties of the surface (0–17 cm) and subsurface (17–50 cm) horizons in the packed soil columns.

Parameter	Horizon	
	Ap	A/E
Sand, g kg ⁻¹	924 ± 1.3†	935 ± 0.4
Silt, g kg ⁻¹	74 ± 1.3	61 ± 0.01
Clay, g kg ⁻¹	2 ± 0.4	4 ± 0.4
Bulk density, g cm ⁻³	1.77 ± 0.04	1.87 ± 0.02
Particle density, g cm ⁻³	2.58 ± 0.01	2.62 ± 0.1
Porosity, %	31 ± 1.5	28 ± 0.9
pH	6 ± 0.08	5.5 ± 0.01
EC, ‡ dS m ⁻¹	0.065 ± 3	0.039 ± 11
Organic matter, g kg ⁻¹	23 ± 1	8 ± 1

† Mean ± SD.

‡ Electrical conductivity.

surface horizon (Ap) had lower bulk density and particle density than the subsurface horizon (A/E). Total porosity was similar between horizons (28–31%). The Ap horizon had higher pH, EC, and organic matter than the A/E horizon. The Ap horizon had higher total Ca, Mg, K, and P contents than the A/E horizon, but Na and Fe did not vary between horizons (Table 3). In contrast, total Al was lower in the Ap horizon (1284 mg kg⁻¹) than in the A/E horizon (1701 mg kg⁻¹). Water extraction of different elements varied between horizons, with greater recovery of Mg and K in the Ap horizon (17–27%) than in the A/E horizon (14–17%), whereas Ca and Na were more water extractable in the A/E horizon (20–33%) than in the Ap horizon (9–27%) (Table 3). Water-extractable P was greater (8%) in the A/E horizon than in the Ap horizon (4%), indicating that labile P may have leached from the surface to lower depths due to the sandy nature and low P sorption capacity of the soil (Harris et al., 1996, 2010).

Chemical Characteristics of Packinghouse Wastewater Applied to Soil Columns

Wastewater pH ranged from 6.0 to 6.9 (mean, 6.2), which was lower than several other wastewater types, including potato-processing plant wastewater (pH 7.4) (Zvomuya et al., 2005), dairy manure wastewater (pH 7.0) (Harris et al., 2008), and municipal wastewater (pH 7.2) (Woertz et al., 2009). Wastewater EC was 1.94 to 2.44 dS m⁻¹ (mean, 2.16 dS m⁻¹) (Table 4). The high EC in our wastewater was due to the high chloride concentration of 551 to 638 mg L⁻¹. The

high chloride in the wastewater is attributed to the formation of chlorides when chlorine as sanitizer undergoes oxidation in the water. Total P concentration in wastewater ranged from 4 to 4.4 mg L⁻¹. This was slightly higher than the total P concentration of 3.6 mg L⁻¹ reported for treated potato-processing wastewater (Zvomuya et al., 2005) but was much lower than the reported values of >10 and 28 mg L⁻¹ for domestic wastewater (Vaillant et al., 2004) and dairy wastewater (Harris et al., 2008), respectively.

Wastewater cation concentration was greatest for Na (349–377 mg L⁻¹), followed by Ca (45–50 mg L⁻¹), K (32–34 mg L⁻¹), and Mg (21–23 mg L⁻¹). Harris et al. (2008) reported much higher concentrations of Ca (138 mg L⁻¹), Mg (64 mg L⁻¹), and K (248 mg L⁻¹) but lower Na (75 mg L⁻¹) in dairy wastewater than we found in our wastewater. Howe and Wagner (1996) observed higher concentrations of Na (422 mg L⁻¹) and Ca (108 mg L⁻¹) in paper-mill wastewater than we found in our wastewater.

Drainage from Soil Columns

Leachate volume varied significantly among treatments during the 30-d application period (Table 5). As expected, the high wastewater treatment yielded greater drainage of leachate (2.31 cm d⁻¹), followed by the medium wastewater and control treatment (1.45–1.51 cm d⁻¹) and the low wastewater application rate (0.66 cm d⁻¹).

The leachate-to-irrigation (LI) ratio in all treatments was <1 during all leaching events ($n = 30$), indicating that some of the applied irrigation water was retained in soil columns (Fig. 1A and 1B). The initial eight leaching events showed a continuous and gradual increase in LI ratio from 0.70 to 0.86 to 0.96 in all but the low wastewater treatment. Initial leaching events are known to play an important role in the emergence and development of water flow paths in soil columns, with continuous irrigation applications resulting in an apparent steady-state flow (Kleinman et al., 2005), as seen in our soil columns. At the last leachate sampling event (i.e., Day 30), the LI ratio was greatest in the high wastewater treatment (0.94), followed by the medium-rate and control (0.88–0.90) and low-rate (0.75) treatments. Due to variation in irrigation water application rates among treatments, leachate totals were also greatest in the high irrigation treatment (4.3 PV), followed by control and medium (2.9–3.0 PV) and low (1.4 PV) treatments (Fig. 1B).

Table 3. Total and water-extractable (soil/water ratio, 1:10) elements in surface (0–17 cm) and subsurface (17–50 cm) horizons of the packed soil columns.

Element	Ap horizon		A/E horizon	
	Total	Water-extractable	Total	Water-extractable
	mg kg ⁻¹			
Al	1284 ± 39†	13 ± 2 (1)‡	1701 ± 65	43 ± 14 (3)
Ca	517 ± 15	46 ± 20 (9)	89 ± 1	18 ± 1 (20)
Fe	454 ± 23	4 ± 0.6 (1)	446 ± 19	11 ± 4 (3)
P	268 ± 6	12 ± 1 (4)	186 ± 9	14 ± 2 (8)
Mg	75 ± 3	12 ± 1 (17)	40 ± 3	6 ± 0.9 (14)
K	59 ± 21	16 ± 1 (27)	23 ± 2	4 ± 0.4 (17)
Na	15 ± 6	4 ± 0.5 (27)	10 ± 1	3 ± 1 (33)

† Mean ± SD.

‡ Values in parentheses indicate percentage of total.

Chloride Breakthrough Curves in Wastewater-Irrigated Soil Columns

Chloride is a nonreactive tracer and therefore can be used to determine the flow of water in a soil profile. In our study, chloride breakthrough curves had a sigmoid (S) shape in all wastewater-treated columns (Fig. 1C), indicating that chloride and water flow behavior were similar despite different application rates in three treatments. At 1 PV, 50% (i.e., C/Co = 0.5) of the chloride applied with wastewater appeared in the leachate (Fig. 1C), suggesting uniform convective transport or lack of preferential transport in our soil columns (Mamo et al., 2005). This was probably due to the lack of preferential flow pathways due to homogeneous packing of soil columns and the use of application rates (0.06–0.17 PV per application or day) that were less than the infiltration capacity of soil in all treatments. If preferential flow had occurred, chloride would have been detected in leachate earlier and at a high concentration approaching that of the applied wastewater (Mamo et al., 2005). However, this was not the case in our study because chloride began appearing in leachate only after 0.7 PV and because chloride concentrations were always lower than in wastewater, suggesting matrix flow in soil columns.

During Phase 1 (<0.7 PV), leachate collected from all soil columns consisted of pre-event water (i.e., deionized water stored from prewetting events), which was displaced by applied irrigation water (Fig. 1C); thus, the chloride C/Co ratio in Phase 1 leachate was <0.01. In Phase 2 (0.7–1.4 PV), wastewater began to appear in the leachate, resulting in a gradual increase in leachate chloride C/Co ratio in wastewater treatments. In Phase 3 (>1.1–1.4 PV), leachate from the columns consisted predominantly of wastewater flow (chloride C/Co ratio: 0.7–1.0). Previous studies have reported similar results in homogenized soil columns, with chloride breakthrough observed after 0.5 to 0.7 PV of leachate and C/Co ratio continuing to increase to equivalence, indicating soil matrix flow (Camobreco et al., 1996). Therefore, our breakthrough curves are typical of homogeneous soil media and indicate soil matrix flow in our columns.

Leachate pH and Electrical Conductivity in Wastewater-Irrigated Soil Columns

Mean pH in all leachate sampling events ($n = 30$) was significantly ($P < 0.05$) lower in medium (6.53) and high (6.44)

Table 4. Selected chemical properties of tomato packinghouse wastewater applied in the soil columns ($n = 6$).

Parameter	Mean \pm SD	Range
pH	6.2 \pm 0.3	6–6.9
EC, † dS m ⁻¹	2.16 \pm 0.23	1.9–2.4
Chloride, mg L ⁻¹	593 \pm 34	551–638
	mg L ⁻¹	
Na	358 \pm 10	349–377
Ca	47 \pm 2	45–50
K	33 \pm 0.6	32–34
Mg	22 \pm 0.7	21–23
P	4.2 \pm 0.1	4–4.4
Al	–‡	–

† Electrical conductivity.

‡ Below detection limit.

wastewater treatments than in the control treatment (6.91) due to lower pH of applied wastewater; however, pH was similar between the low wastewater and control treatments (Table 5). Cationic resins containing hydrogen ions are frequently used to remove Na and Ca from the solution phase of wastewater (Skogley and Dobermann, 1996). The addition of Ca, Mg, and Na in our applied wastewater may have caused ion exchange (soil pH: 5.5–6), resulting in leaching of soil hydrogen ions and reduction of leachate pH. During Phase 1 of sampling, control and wastewater treatments leached pre-event water, and leachate pH was neutral (6.6–7.1) (Fig. 2). In Phase 2, leachate pH declined gradually in wastewater treatments and began to approach the pH of applied wastewater. During Phase 3, pH was slightly lower in the high wastewater treatment than in the medium wastewater treatment.

Leachate EC was significantly ($P < 0.05$) greater in wastewater treatments (0.85–1.78 dS m⁻¹) than in the control treatment (0.12 dS m⁻¹) because applied wastewater had a higher EC (1.9–2.4 dS m⁻¹) than deionized water (0.001 dS m⁻¹) (Table 5). During Phase 1, while pre-event water was draining, EC was <0.1 dS m⁻¹ in the control and 0.1 to 0.3 dS m⁻¹ in wastewater treatments because this was the pre-event (deionized) water (Fig. 2). Continuous application of wastewater (of high EC) in Phase 2 displaced pre-event water (of low EC) stored in the soil profile, resulting in a gradual increase in leachate EC from 0.3 to 2.5 dS m⁻¹. In Phase 3, leachate EC was constant in the medium and high treatments and was similar to applied wastewater EC, suggesting that equilibrium was

Table 5. Irrigation rate, mean drainage depth, and mean leachate pH, chloride, and electrical conductivity in 30 leaching events.

Treatment	Application rate	Drainage depth†	Leachate		
			pH	Chloride	EC‡
		cm d ⁻¹		mg L ⁻¹	dS m ⁻¹
Control	1.67	1.45 \pm 0.03b (88)§	6.91 \pm 0.01b	1.6 \pm 0.03a	0.12 \pm 0.01a
Low	0.84	0.66 \pm 0.1a¶ (79)	6.87 \pm 0.01ab	195 \pm 5b	0.85 \pm 0.06b
Medium	1.67	1.51 \pm 0.01c (91)	6.53 \pm 0.01a	427 \pm 6c	1.59 \pm 0.01c
High	2.51	2.31 \pm 0.01d (92)	6.44 \pm 0.04a	477 \pm 2d	1.78 \pm 0.02d

† Mean \pm SD.

‡ Electrical conductivity.

§ Values in parentheses indicate the percent recovery of applied water.

¶ Values followed by same letters in a column are not significantly different at $P < 0.05$ using Fisher's LSD.

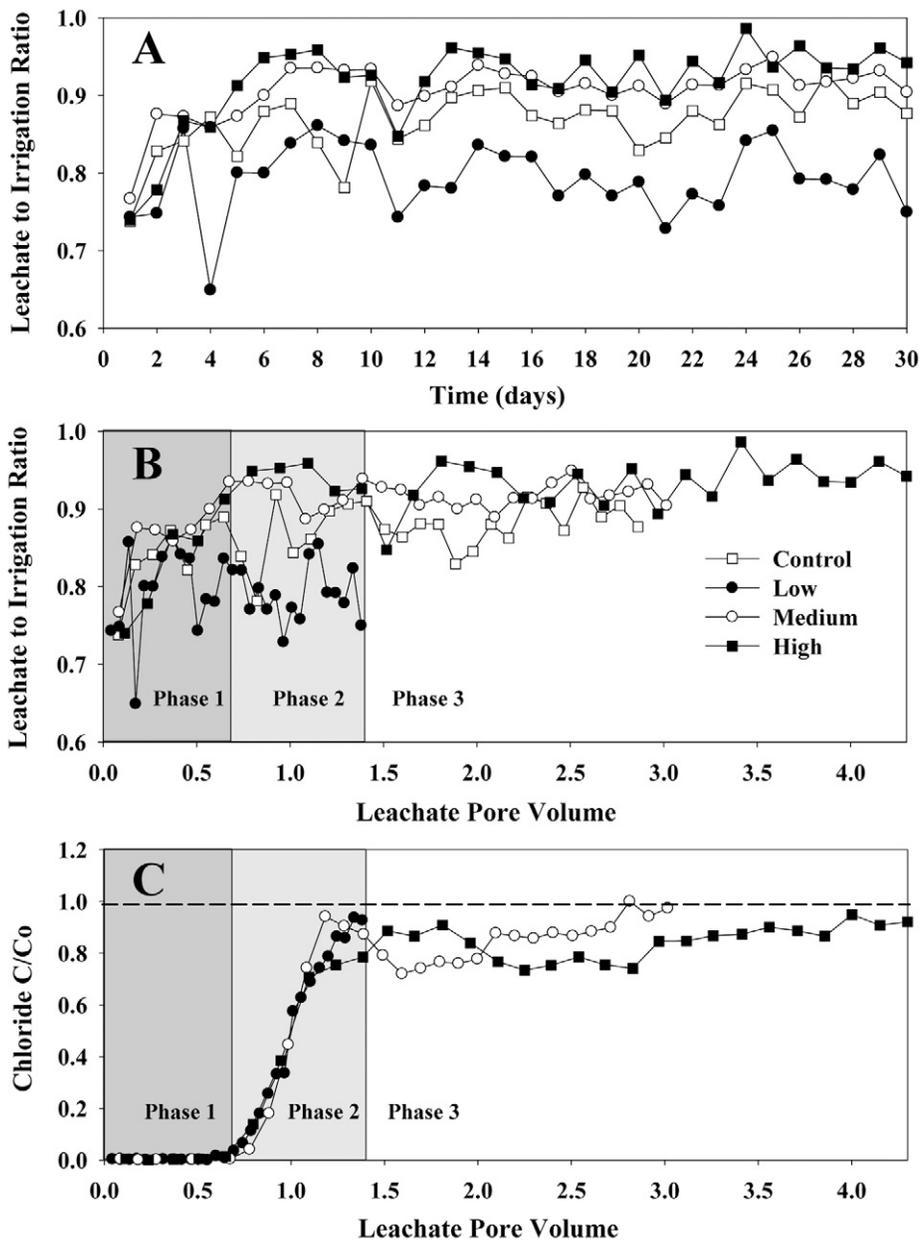


Fig. 1. Leachate volume recovery with (A) time and (B) pore volume and (C) chloride breakthrough showing three phases where C is the leachate concentration and C_o is the irrigation concentration in soil columns during 30 leaching events. Phase 1: pre-event water. Phase 2: mixture of pre-event water and wastewater. Phase 3: wastewater flow only.

achieved in the soil columns. The breakthrough curves (Fig. 2) of chloride were similar to those of EC.

Effect of Wastewater Application on Phosphorus Transport in Soil Columns

During all leaching events ($n = 30$), wastewater application did not have a significant effect on mean leachate P concentration in the control (0.7 mg L^{-1}) and/or among wastewater ($0.33\text{--}0.60 \text{ mg L}^{-1}$) treatments (Table 6). Mean P concentration in applied wastewater (4 mg L^{-1}) was much greater than in leachate from the wastewater treatments ($\leq 0.7 \text{ mg L}^{-1}$), indicating that most of the wastewater-applied P was retained in the soil.

During the study, P concentration gradually increased from 0.38 to 0.96 mg L^{-1} in leachate from the control treat-

ment, whereas P concentration in leachate from all wastewater treatments increased only in Phase 1 ($0.2\text{--}0.8 \text{ mg L}^{-1}$) (Fig. 3). The increase in leachate P in Phase 1 in all treatments was probably because of the removal of a part of the water-soluble P present in the soil profile with pre-event (deionized) water. For example, water-extractable P concentrations in the soil (surface and subsurface horizons) were 12 to 14 mg L^{-1} (Table 3), suggesting that the observed increases in leachate P were the result of movement of soluble P present in the soil profile in pre-event water. The release of water-soluble P from the soil is dependent on the amount of P available and the soil to water ratio used for extraction (Toor et al., 2007). The soil to water ratio used to extract P from soil samples was much narrower ($1:10$; see Table 3) than the ratio experienced in soil columns, which ranged from $1:110$ in the low wastewater treatment to $1:55$ in the control and medium and $1:36$ in high treatments. As a result, leachate P concentrations were much lower ($< 1 \text{ mg L}^{-1}$) and were greatest in the high wastewater treatment. This finding has practical implications for irrigation management in soils with high water-soluble P because high irrigation rates result in greater leaching losses of soluble P.

It is also likely that the addition of deionized water (with zero P) diluted and reduced the P in the soil solution, which promoted the concentration gradient to be low in the soil solution phase and high on the soil solid phase. To keep the electrical balance in the soil solution, the solid phase released P to equilibrate the reaction, allowing more P in the soil solution with the potential for leaching. Therefore, the higher P leaching in the control than in the wastewater-treated columns could be attributed to P depletion from the solid phase of soil.

After wastewater began to appear in the leachate (Phase 2 and 3), P concentrations decreased and were always lower than in the control (Fig. 3). We attribute this differential P leaching behavior to the greater ionic strength of applied wastewater (0.031 mol L^{-1}) as compared with deionized water (0 mol L^{-1}). When a high-ionic-strength wastewater is used for irrigation, it can increase P sorption in soil because of an increase in net surface charge (i.e., surface potential) in the soil (Jensen et al., 1998; Pardo et al., 1992). Moreover, in the treatments that received wastewater (with P added in

each irrigation event), the P depletion from soil solid phase to soil solution phase (unlike control treatment) was reduced because more P was added in the soil solution and the contribution of depletion in the P leaching was low. The high Ca added via wastewater could have played an important role to precipitate P; for example, in low wastewater treatment, a considerable amount of Ca (twofold more than the control) and the high pH of the leachate (6.87, similar to 6.91 in the control treatment) probably increased the P precipitation, thus reducing P leaching. On the other hand, in the high wastewater treatment, high amounts of P and Ca were added; however, the pH was reduced (6.44), which may have decreased P adsorption and resulted in higher P leaching.

Therefore, the lower leachate P concentrations in our wastewater treatments might be due to increased adsorption of wastewater P onto the surface of soil colloids and the presence of Ca in wastewater. Because natural adsorbents present in the soil organic matter, such as Fe/Mn and Al oxides, are known to adsorb P (Chen et al., 2003), the presence of Al (1284–1701 mg kg⁻¹), Fe (454–446 mg kg⁻¹), and Ca (89–517 mg kg⁻¹) in our soil may be important adsorption sites for wastewater P. Soil column irrigation with P-rich dairy wastewater has been shown to result in rapid adsorption of P as the solution penetrates soil aggregates containing Fe and Al, resulting in low P in leachate (Toor et al., 2005b). Soil can adsorb P added with wastewater and reduce leachate P concentration. Application of wastewater containing 21 mg P L⁻¹ to soil with high P has been shown to reduce P leaching as compared with application of lower P wastewater (Mamo et al., 2005).

In our study, leachate P concentration in the high wastewater treatment decreased in the early part of Phase 2 but then began to increase, albeit at a slower rate than seen in the control

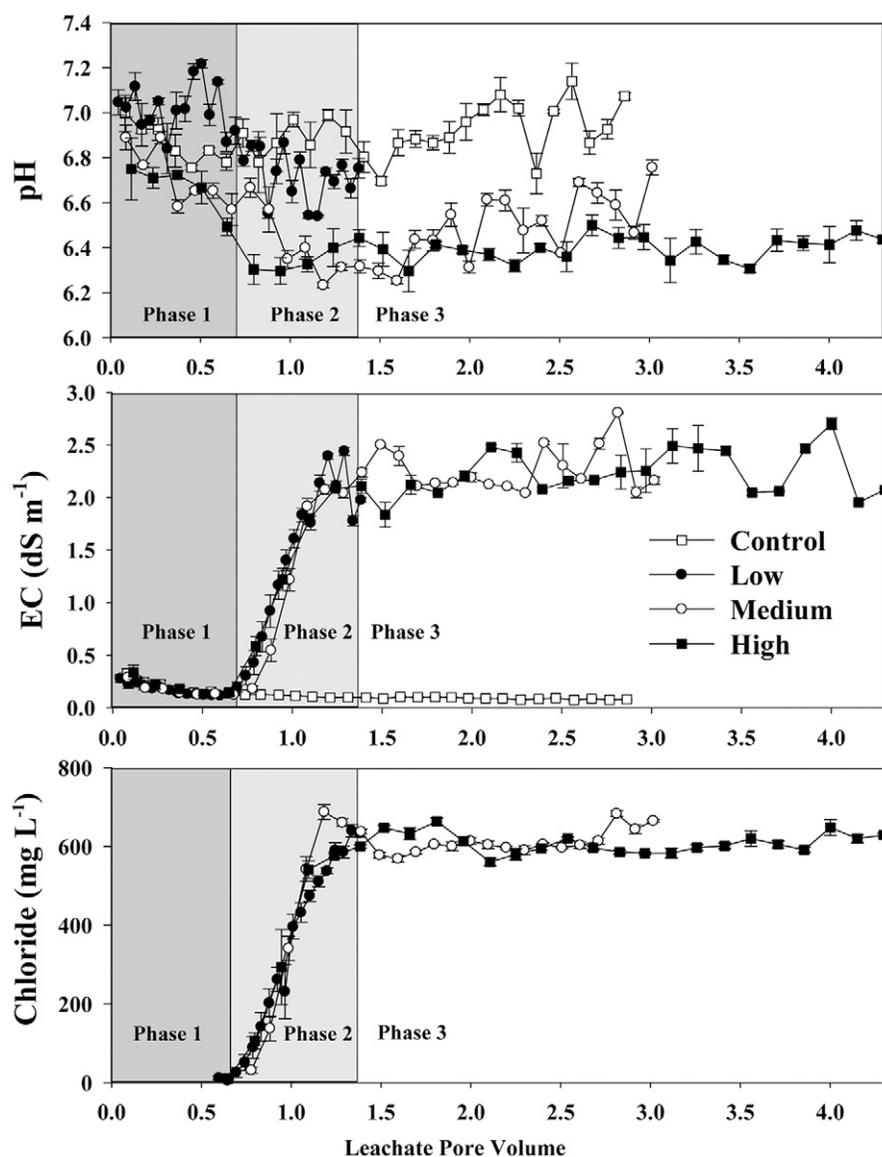


Fig. 2. Leachate pH, electrical conductivity (EC), and chloride in four treatments during 30 leaching events. Error bars indicate SEM. Chloride concentrations in the control and wastewater treatments (up to 0.7 pore volume) were below the 5 mg L⁻¹ detection limit.

treatment. This suggests a decline in the capacity of the soils used in our study to retain wastewater P, which could result in an increase in leachate P with long-term, continuous application of wastewater. Nonetheless, leachate P concentrations from the high wastewater treatment (0.4–0.8 mg L⁻¹) remained lower than for the control and well below the P concentration of applied wastewater (4.2 mg L⁻¹).

Table 6. Mean leachate concentrations of phosphorus and cations in the control and wastewater treatments in 30 leaching events.

Treatment	P	Na	mg L ⁻¹			K
			Ca	Mg		
Control	0.7 ± 0.3b†	0.18 ± 0.1a‡	15 ± 1.3a	2.7 ± 0.2a	7 ± 1a	
Low	0.28 ± 0.1a	35 ± 8b	66 ± 4b	17 ± 0.5b	14 ± 1.2b	
Medium	0.33 ± 0.2a	188 ± 2c	71 ± 1c	19.4 ± 1.2c	14 ± 1.1b	
High	0.6 ± 0.1ab	235 ± 2d	66 ± 2b	16 ± 1b	18 ± 0.7c	
LSD	0.41	7.91	4.83	1.60	1.96	

† Mean ± SD.

‡ Values followed by same letter in a column are not significantly different at *P* < 0.05 using Fisher's LSD.

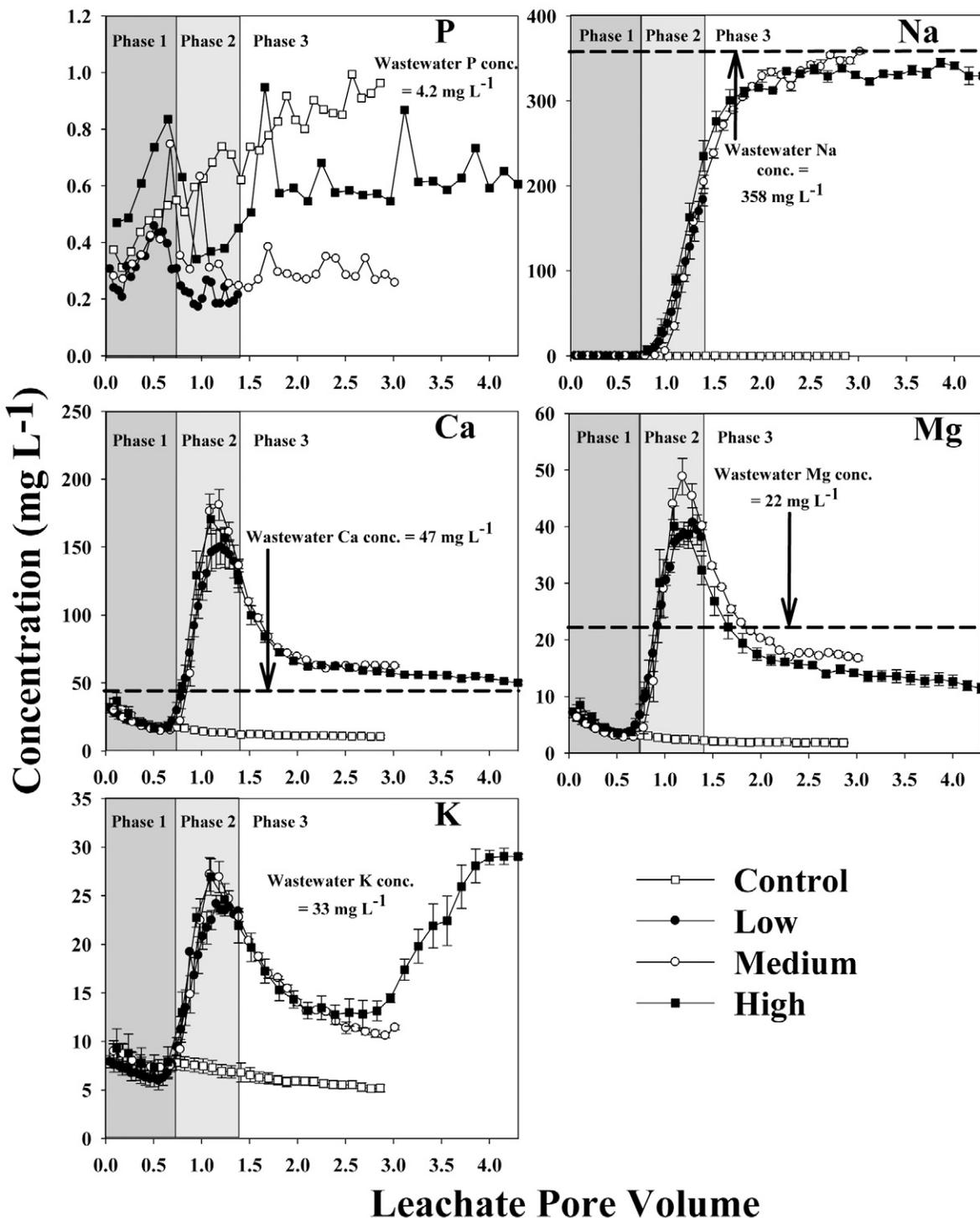


Fig. 3. Concentrations of P, Na, Ca, Mg, and K in leachate collected in four treatments during 30 leaching events. Error bars indicate SEM. Note the scale for concentrations of different elements.

Effect of Wastewater Application on Transport of Cations in Soil Columns

Leachate Na concentration was $\leq 0.4 \text{ mg L}^{-1}$ in the control treatment throughout the study, whereas in the wastewater treatments, leachate Na concentration was low only in Phase 1 during leaching of pre-event water (Fig. 3). In Phase 2, leachate Na concentration gradually increased from <0.4 to $>200 \text{ mg L}^{-1}$ in wastewater treatments. In Phase 3, Na concentrations in medium and high wastewater treatments reached 335 ± 20

mg L^{-1} but remained lower than Na concentrations of applied wastewater ($349\text{--}377 \text{ mg L}^{-1}$). The transport behavior of Na suggests that Na was freely leached from the soil profile, likely due to low levels of water-soluble Na ($3\text{--}4 \text{ mg kg}^{-1}$) in the soil profile (Table 4).

Calcium and Mg exhibited similar transport behavior in soil columns (Fig. 3) because both are divalent ions with similar chemical properties. Mean concentrations of Ca in control leachate varied from 11 to 36 mg L^{-1} , whereas for Mg and K, concentrations ranged from 1.8 to 7.2 mg L^{-1}

and 5.2 to 8.7 mg L⁻¹, respectively. Similar concentrations were observed in Phase 1 leachate from wastewater treatments due to leaching of soil Ca, Mg, and K with pre-event water. Concentrations of water-extractable Ca, Mg, and K in soil surface horizon were 46, 12, and 16 mg L⁻¹, respectively (Table 3). Most Florida soils originated from sandy marine deposits and limestone minerals such as calcite (CaCO₃) and dolomite [CaMg(CO₃)₂] and have high amounts of Ca and Mg (Watts and Collins, 2008). Because most of Florida's soils are coarse textured, they also have high amounts of K, which can be present in soil solution and on exchangeable sites on the surface of minerals. The natural abundance of Ca, Mg, and K in our soil probably resulted in high initial leachate concentrations.

Applied wastewater was also rich in Ca (45–50 mg L⁻¹), Mg (21–33 mg L⁻¹), and K (32–34 mg L⁻¹). During Phase 2, concentrations of these cations in leachate increased as wastewater appeared in the leachate, with Ca (150–180 mg L⁻¹) and Mg (40–49 mg L⁻¹) concentrations reaching two to four times the wastewater concentration (Fig. 3). The concentrations of Ca and Mg in leachate from wastewater treatments likely originated from soil sources due to ion exchange with Na applied with wastewater or due to the past accumulations of Ca and Mg applied with previous wastewater irrigations. In contrast, maximum K concentration in leachate from wastewater treatments (24–27 mg L⁻¹) never exceeded applied wastewater concentration, suggesting that some of the K in the applied wastewater K was retained in the soil.

During Phase 3, Ca and Mg concentrations in leachate from medium and high wastewater treatments continued to decline; Ca concentration in leachate was always greater than that in wastewater, whereas leachate Mg concentration decreased below that in the applied wastewater at 1.7 to 1.8 PV (Fig. 3). Continuous addition of Mg to soil may result in the solubility product of Mg compounds (e.g., magnesite, struvite) being exceeded, thereby causing precipitation of Mg from the

solution phase (Miyamoto and Pingitore, 1992); this explains the decline in Mg concentration in Phase 3 of our study. Also, the high total Ca/Mg ratio in our soil (7:1) and applied wastewater (2:1) may have resulted in less retardation to Ca flow as compared with Mg. During Phase 3, after an initial decrease, K concentration gradually increased in leachate from the high wastewater treatment. Our observations suggest that native soil Ca and Mg reserves increased leachate Ca and Mg concentrations during our study, whereas most of the applied Na and K was retained by the soil.

Mass Balances of Phosphorus and Cations in Soil Columns

In the control treatment, approximately 3 kg P ha⁻¹ was leached from the soil profile (Table 7). In the low, medium, and high wastewater treatments, approximately 5, 7, and 13% of applied P was leached, respectively. This suggests that, at low application rates, wastewater-applied P may have more time to react with soil components, resulting in increased P fixation and decreased P leaching. This is consistent with the findings of Mamo et al. (2005). Total amounts of P leached during our study (0.6–4.2 kg ha⁻¹) were lower than the P leaching losses observed in the gley to allophanic soils of New Zealand (2–49 kg ha⁻¹) after the application of domestic wastewater (Barton et al., 2005) but were greater than the P losses observed in monolith soil columns (1.7–2.2 kg ha⁻¹) irrigated with dairy wastewater (Toor et al., 2004). Previous research reported total P leaching losses from grassland soils ranging from 0.4 to 0.7 kg ha⁻¹ yr⁻¹ (Sims et al., 1998; Turner and Haygarth, 2000). However, a comparison of leaching losses across studies and systems is difficult because net leaching depends on the number of irrigation events, the type of soil columns used, the rate of wastewater application, and soil properties. No plants were grown in our soil columns, and most (87–95%) of the wastewater-applied P was retained in the soil profile. However, forage grasses

Table 7. Amounts of P and cations (Na, Ca, Mg, and K) applied in the soil columns, amounts leached, and percent leaching of applied amounts in four treatments in 30 leaching events.

Treatment	P	Na	Ca	Mg	K
Amount applied					
kg ha ⁻¹					
Control	0	0	0	0	0
Low	11	898	119	55	82
Medium	21	1796	238	109	165
High	32	2708	359	165	248
Amount leached					
kg ha ⁻¹					
Control	3.0 ± 1.4bc†‡	0.79 ± 0.3a	65 ± 5a	12 ± 1a	29 ± 4a
Low	0.56 ± 0.2a	70 ± 17b	130 ± 8b	33 ± 1b	27 ± 3a
Medium	1.5 ± 1.1ab	866 ± 6c	326 ± 4c	89 ± 5c	64 ± 5b
High	4.16 ± 0.4c	1658 ± 23d	463 ± 10d	114 ± 7d	125 ± 4c
LSD	1.75	27.17	13.29	8.42	7.55
Percent leached					
Low	5	8	110	60	33
Medium	7.2	48	137	82	39
High	13	61	128	69	50

† Mean ± SD.

‡ Values followed by same the letter in a column are not significantly different at $P < 0.05$ using Fisher's LSD.

have been shown to remove up to 81 to 141 kg P ha⁻¹ from the soil (Adriano et al., 1975), and tomato plants are also known to take up to 23 kg ha⁻¹ in 121 d (Adams, 1986). This suggests that agricultural fields that use packinghouse wastewater for crop irrigation are less likely to have a large P surplus but should consider reducing fertilizer P inputs for economic and environmental benefits.

Leaching of wastewater-applied Na was relatively high and ranged from 8 to 61% (CV = 62%) in our study, whereas leaching of Mg was relatively less variable (CV = 14%) in treatments where about 60 to 82% of wastewater-applied Mg was leached (Table 7). In the control treatment, 65 kg ha⁻¹ of Ca was leached from the soil profile. Leaching of wastewater-applied Ca equaled or exceeded wastewater-applied amounts (100–137%), suggesting that high amounts of Ca were leached from soil reserves. In the control treatment, 12 kg ha⁻¹ of Mg was leached from the soil profile, which was several-fold lower than the wastewater treatments that leached at a rate of 33 to 114 kg ha⁻¹. Potassium leaching from the control columns amounted to 29 kg K ha⁻¹, similar to that from the low wastewater treatment (27 kg ha⁻¹) but much lower than K leaching from the medium and high wastewater treatments (64–125 kg ha⁻¹). Overall, significant fractions of wastewater-applied P, Na, Mg, and K were retained in the soil profile, whereas additional Ca was leached from native soil reserves.

Conclusions

The pH of tomato packinghouse wastewater was in the suggested range of 6.5 to 8.0 for irrigation water, but EC and chloride were elevated. This suggests that care should be taken when selecting crop varieties and irrigation practices (drip vs. flood) in fields using wastewater. The current soil acted as a source when leached with deionized water but acted as a sink when leached with wastewater. In addition, application of wastewater at the medium rate (1.67 cm d⁻¹) did not increase P leaching. This suggests that packinghouse wastewater land-applied to Spodosols may enhance soil fertility, resulting in beneficial effects on crop growth. Although the concentrations of Ca, Mg, and K in leachate were elevated in wastewater-treated columns compared with the control, there are no current water quality guidelines for soluble Ca, Mg, and K that restrict the use of groundwater for various purposes, such as drinking water, recreation, or protection of aquatic life. Thus, these elements are not of critical concern in groundwater pollution. The maximum concentrations of Na in leachate at 50 cm depth (345–358 mg L⁻¹) were greater than Florida groundwater cleanup target levels of 160 mg L⁻¹. The calculated sodium adsorption ratio values in leachate from low, medium, and high wastewater treatments were 1.0, 5.1, and 6.7, respectively. These values are much less than the <10 sodium adsorption ratio value of no harmful effects of irrigation water on soil, suggesting that wastewater can be safely used for irrigation. The current study design did not include plants, so the effect of nutrient plant uptake was not taken into account, which may modify the leaching behavior of P. We suggest that farms using packinghouse wastewater

should conduct routine P mass balances (inputs and outputs) to ensure environmental sustainability.

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