

Controlling Tailwater Sediment and Phosphorus Concentrations with Polyacrylamide in the Imperial Valley, California

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ABSTRACT

External loading of phosphorus (P) from agricultural surface discharge (tailwater) is the main cause of excessive algae growth and the eutrophication of the Salton Sea, California. Continuous polyacrylamide (PAM) applications to agricultural irrigation water inflows were evaluated as a means of reducing sediment and P in tailwater. Zero (control) and 1 mg L⁻¹ PAM (PAM₁) treatments were compared at 17 Imperial Valley field sites. Five and 10 mg L⁻¹ PAM treatments (PAM₅, PAM₁₀) were conducted at one site. The particulate phosphorus (P_p) fraction was determined as the difference between total phosphorus (P_t) and the soluble phosphorus (P_s) fraction. We observed 73, 82, and 98% turbidity reduction with PAM₁, PAM₅, and PAM₁₀ treatments. Although eight field sites had control tailwater sediment concentrations above the New River total maximum daily loads (TMDL), all but one were made compliant during their paired PAM₁ treatments. While PAM₁ and PAM₁₀ reduced tail water P_p by 31 and 78%, none of the treatments tested reduced P_s. This may have been caused by high irrigation water Na concentrations which would reduce Ca adsorption and Ca-phosphate bridging on the PAM. The PAM₁ treatments resulted in <0.5 mg L⁻¹ drain water polyacrylamide concentrations 1.6 km downstream of PAM addition, while PAM₅ and PAM₁₀ treatments produced >2 mg L⁻¹ drain water polyacrylamide concentrations. We concluded that, although PAM practically eliminates Imperial Valley tailwater sediment loads, it does not effectively reduce tailwater P_s, the P fraction most responsible for the eutrophication of the Salton Sea.

EXTERNAL LOADING of phosphorus (P) is the main cause of excessive algae growth and the eutrophication of the Salton Sea, a 980-km² saline lake located in California near the U.S.–Mexico border. Schroeder et al. (2002) stated that “the Salton Sea is highly eutrophic and that P is by far the limiting nutrient relative to N” noting that there is 26 times more nitrogen (N) than P in the Salton Sea relative to ocean water. Most of the external P load to the sea comes from agricultural surface discharge (tailwater) from the Imperial Valley. Due to fertilizer application, P is one of the few constituents in Imperial Valley soils that has increased over the last 50 yr (De Clerck et al., 2003). The Alamo and New Rivers deliver roughly 80% of the sea’s inflow of 1.6 km³ yr⁻¹. Data from Holdren and Montañó (2002) indicate that two-thirds of the rivers’ total P load to the sea is soluble phosphorus (P_s) which is the most bioavailable fraction for algae growth. The other third is particulate phosphorus (P_p). It bears to reason that reductions in

suspended solids and P in agricultural discharges should eventually abate the Sea’s eutrophic cycles.

With the existing sediment total maximum daily loads (TMDLs) and impending nutrient TMDLs for the New and Alamo Rivers, there is growing interest in on-field best management practices (BMPs) that are inexpensive and simple to implement. In addition to the typical BMPs that slow runoff flow rates, anionic polyacrylamide polymers (PAMs) added to irrigation water at rates of 1 to 10 mg L⁻¹ can provide numerous benefits (Sojka et al., 2005).

Each year in the United States, 1 to 10 mg L⁻¹ PAM solutions are applied to 400000 irrigated hectares for tailwater nutrient and sediment removal (Orts et al., 2000; Entry and Sojka, 2003). Removing sediment particles from tailwater has the added benefit of removing adsorbed pesticides (Agassi et al., 1995; Singh et al., 1996). Another benefit is bacteria removal: PAM, PAM + CaO, and PAM + Al₂(SO₄)₃ treatments decreased enteric bacteria in animal wastewater surface flows and in leachate from soil columns (Sojka and Entry, 2000; Entry and Sojka, 2000; Entry et al., 2002, 2003). Polyacrylamide treatments can also decrease the spread of weed seeds in irrigated furrows by 62 to 90% (Sojka et al., 2003). Furthermore, anionic PAM treatments improve soil structure and increase net irrigation water infiltration (Shainberg et al., 1990; Lentz and Sojka, 1994; Sojka et al., 1998). Finally, the toxicity effects of anionic PAM to mammals and fish are small to none (Barvenik, 1994).

However, there is little information published on PAM’s potential to remove P_s and P_p while furrow irrigating heavy-textured, salt-affected soils like those found in the Imperial Valley.

In this study we tested the efficacy of PAM-treated irrigation water to decrease suspended solids, P_p, and P_s on Imperial Valley agricultural fields.

MATERIALS AND METHODS

The Imperial County Farm Bureau (ICFB) identified growers willing to volunteer agricultural field sites located throughout the Imperial Valley. Given free PAM treatments and an assurance of confidentiality, growers allowed us to conduct controlled experiments to evaluate the effects of PAM amended irrigation water on turbidity, total suspended solids (TSS), P_t and P_s. Throughout the study, Colorado River water was used to furrow irrigate fields with bare, tilled soils (Table 1). Soils at the 17 selected field sites were silty clays and silty clay loams (hyperthermic Typic Torrifluvents) with high shrink–swell potential (Zimmerman, 1980). Field slopes

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Abbreviations: DOM, dissolved organic matter; OPP, liquid orthophosphate phosphorus fertilizer; PAM, water-soluble anionic polyacrylamide powder; P_p, particulate phosphorus; P_s, soluble phosphorus; P_t, total phosphorus; TMDL, total maximum daily load; TSS, total suspended solids.

Table 1. Comparison of Idaho's Snake River Valley and California's Imperial Valley irrigation waters.[†]

	EC	SAR	Na	K	Ca	Mg	DOM
	dS m ⁻¹		mg L ⁻¹				
Lower Colorado River	1.1	3.1	136	6	92	35	5.1
Snake River [‡]	0.5	0.6	31	0.4	4	1.2	0.4

[†] EC, electrical conductivity; SAR, sodium adsorption ratio; DOM, dissolved organic matter.

[‡] Snake River water data from Entry and Sojka (2003) is for comparison.

were 2 to 3 m km⁻¹; side falls (drain ditch slopes) were 0 to 1.5 m km⁻¹ except for the last 10 m before the drain boxes where they often steepened to 50 to 100 m km⁻¹ (Al Kalin, Imperial County Farm Bureau, personal communication). Irrigation water inflow rates ranged from 167 to 507 L s⁻¹. Based on Imperial Valley water use data from 1988 to 1998, tailwater runoff averaged 17% of the irrigation water applied to fields (Steven Charlton, Imperial Irrigation District, personal communication).

Locally distributed anionic polyacrylamides were selected for these field experiments. Stockopam (Stockhausen, Greensboro, NC) and Soilfix IR (Ciba Specialty Chemicals, Basel, Switzerland), were used interchangeably due to the products' similarity. Both manufacturers reported their PAM to have 30 to 35% negative charge substitution, medium molecular weight (12–15 × 10⁶ g mol⁻¹), 99% active ingredient (w/w), and <0.05% residual monomers. Both of these finely ground, dry PAM products are designed to be added directly to free-flowing irrigation water and intended to ameliorate tailwater quality. The retail cost for the PAM products used ranged from US\$6 to \$11 kg⁻¹ which included product delivery and use of a rotary applicator (Fig. 1). All PAM concentrations reported herein are as mg L⁻¹ of active ingredient. Except where noted, PAM application rates were 1 mg L⁻¹ or, if based on



Fig. 1. Battery-powered, rotary feed polyacrylamide (PAM) applicator dosing inflow water.

average total irrigation volume and field site area, approximately 2.3 kg ha⁻¹.

The experimental design shown in Fig. 2 was repeated at each field site. Control treatments and PAM treatments were conducted sequentially on separate sets of border controlled lanes on the same fields.

First, control treatments were done by opening the head ditch valves on a set of three to five border-controlled lanes to irrigate with unamended canal water. Once water had advanced to the drain box, triplicate water samples were taken at five field positions: the inflow, at three points along the center lane, and at the drain box. These water samples were designated "control" (0 mg L⁻¹ PAM) samples. Once water advanced to the field tail, the head ditch valves were closed to stop irrigation.

Subsequently, PAM treatments were started on an adjacent set of lanes. Dry, powdered PAM was added directly to the head ditch with rotary applicators (Fig. 1) placed upstream of turbulent flow sections and at least 30 m from the field head to promote mixing. To ensure consistency of the PAM treatments, the initially dosed water was wasted down the head ditch, and inflow rates were regularly checked and PAM application rates adjusted to ensure a 1 mg L⁻¹ PAM application rate. Once PAM was well mixed with the inflow water, the head ditch valves were opened on a set of three to five border-controlled lanes adjacent to the "control" set. As with the control samples, PAM-treated water samples were taken from the inflow, the center lanes, and the drain box once water had advanced to the field tail.

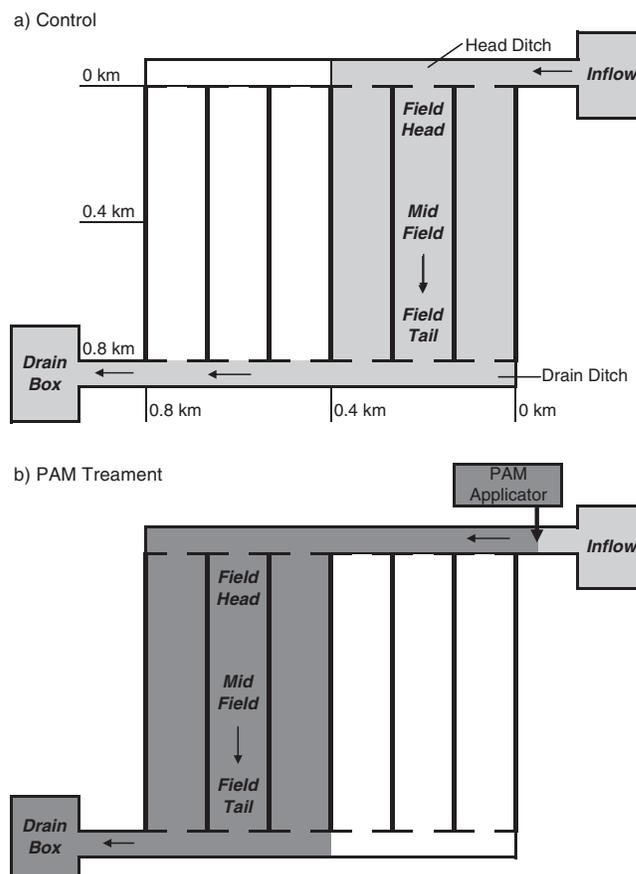


Fig. 2. Field plot showing water sampling locations for (a) control, then (b) polyacrylamide (PAM) treatment sets of border-controlled lanes at a single field site. Dimensions shown are approximate.

Turbidity was measured in-field with a DRT-15CE Portable Turbidimeter (HF Scientific, Fort Myers, FL). Total suspended solids (TSS) were determined by filtration and drying of the collected sediment material using standard methods (American Public Health Association, 1992).

Water samples intended for PAM concentration measurements were centrifuged in the field in 50-mL Teflon tubes and stored in 25-mL glass vials with Teflon caps at 4°C. These samples were then prepared and analyzed by size exclusion chromatography as described in Lu et al. (2003). Following 40 CFR Part 136 (USEPA, 2004), the calculated method detection limit was 0.5 mg L⁻¹ PAM.

Sample preparation and preservation protocols included filtration (<0.45- μ m Fisherbrand cellulose acetate membrane filters; Fisher Scientific, Hampton, NH) for P_s, acidification to pH of approximately 2.5 with concentrated H₂SO₄ for P_i, and storage in acid-washed polyethylene bottles at 4°C for both. The P_i sample preparation was modified from Method 4500-P B (American Public Health Association, 1992). Briefly, the oxidizing solution was made of 30 g boric acid, 350 mL 1 M NaOH, and 50 g K₂S₂O₈ diluted to 1 L with deionized water. Sample water and oxidizing solution were combined at the rate of 5:1 in 10-mL polyethylene mailing tubes and heated in a 100°C water bath for 2 h. All P_s and P_i samples were analyzed on an RFA 300 Autoanalyzer (Alpkem, Clackamas, OR) using the ascorbic acid–molybdate blue method described in American Public Health Association (1992). Particulate phosphorus (P_p) was determined by the difference between P_i and P_s.

Quality assurance protocols (prompt sample refrigeration, preservation, preparation, and analysis) were consistently performed on all field-collected water samples. Preservation and preparation protocols were proven effective by comparing P concentrations on selected samples stored for up to 22 d after collection. To avoid possible changes in the P_i concentrations in unoxidized samples, phosphorus analysis was completed within 1 wk after collection, or the samples were promptly oxidized before storage in the refrigerator.

Quality control protocols included the analysis of one check standard, one replicate analysis, and one spiked sample for every 10 samples analyzed. We set the control limits for these quality controls at $\pm 15\%$ of their expected concentrations. Our calculated method detection limit for this procedure was 0.20 mg P L⁻¹ (USEPA, 2004). Replicates that fell outside of the control limits, but had concentrations below the method detection limit of 0.20 mg P L⁻¹ were not used to discriminate unacceptable runs.

Triplicate analyses were averaged and paired (control versus PAM treatments) by field site and field position. Statistical analyses were performed using SatView software (SAS Institute, 1998). Our low number of field sites ($n = 17$) gave us non-Gaussian data sets. Therefore, we used Wilcoxon Signed Rank, a nonparametric alternative to the t test, to analyze the statistical significance of the differences between our control and PAM treatments. Error bars shown in the figures indicate the standard error of the mean (SEM).

RESULTS AND DISCUSSION

Turbidity and Total Suspended Solids Removal

As previously reported in the literature (Lentz and Sojka, 1994; Orts et al., 2000; Entry and Sojka, 2003), our continuous PAM applications to irrigation water inflows significantly reduced drain water sediment. Figure 3 shows average turbidity and TSS measurements

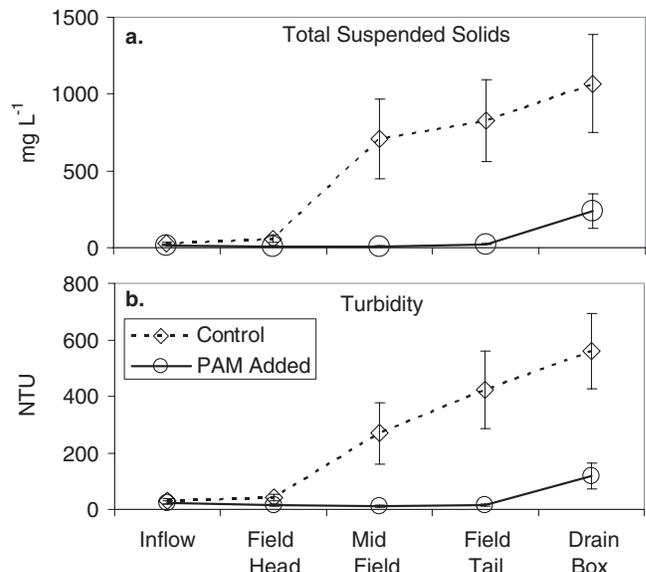


Fig. 3. Overall average reductions in irrigation water total suspended solids (TSS) and turbidity for control versus polyacrylamide (PAM) added (1 mg L⁻¹) treatments. Error bars indicate standard error of the mean.

for control and 1 mg L⁻¹ PAM-treated irrigation water. Wilcoxon Ranked Sign analysis indicates that PAM significantly decreased turbidity by $74 \pm 5\%$ and TSS by $82 \pm 4\%$ from mid-field to the drain boxes ($P < 0.05$). Control drain box TSS exceeded the 230 mg L⁻¹ New River sediment TMDL at 8 out of 17 field sites. However, with continuous 1 mg L⁻¹ PAM treatment, all but one field site's drain box TSS met the TMDL. High TSS at this one site was probably caused by the steep drain ditch slope in the last 10 m before the drain box.

Even with PAM treatment, there was a slight but consistent increase in TSS and turbidity from the field tail to the drain box (Fig. 3). These increases in suspended solids can be attributed to excessively steep drain ditch slopes that allowed the tailwater to gain enough velocity to suspend sediments (field observation). This suggests that modification to the slope of drain ditches could greatly reduce the TSS load delivered to the drains and rivers flowing into the Salton Sea.

Figure 4a shows the effects of varying concentrations of PAM on turbidity and TSS. Although the 5 and 10 mg L⁻¹ PAM treatments were not replicated, they suggest that a 99% sediment removal rate can be achieved with PAM concentrations between 5 and 10 mg L⁻¹. However, more replicates of these treatments are needed to prove their statistical significance.

The fact that TSS and turbidity closely track each other in both Fig. 3 and 4 is indicative of how well correlated these two lines of data are. For our data, the equation:

$$\text{turbidity (NTU)} = 1.35 \times \text{TSS (mg L}^{-1}\text{)}$$

has an r^2 of 0.86. This supports the use of in-field turbidity readings in place of more time consuming laboratory TSS measurements.

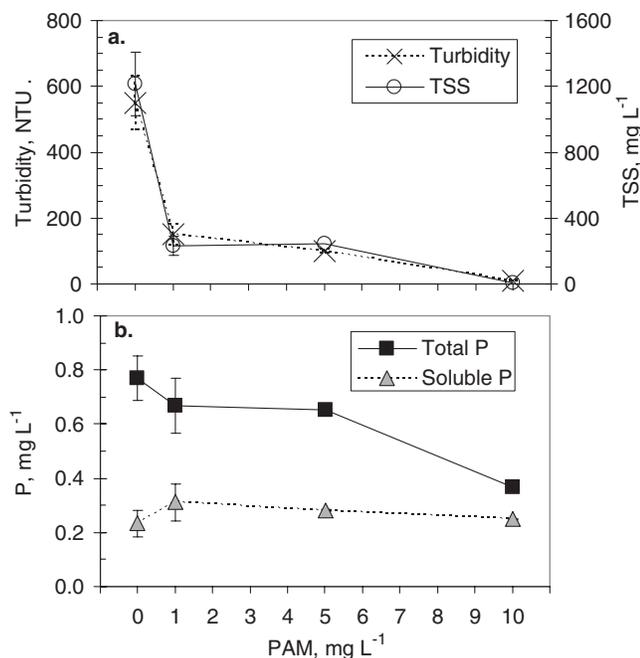


Fig. 4. Variable polyacrylamide (PAM) treatment effects on drain box (a) turbidity and total suspended solids (TSS) and (b) average total phosphorus (P_t) and soluble phosphorus (P_s). Error bars indicate standard error of the mean.

Phosphorus Removal

The data shown in Fig. 5a and 5b were pooled from 17 Imperial Valley field sites and averaged for each treatment–field location. Compared to the control treatments, Wilcoxon Signed Rank analysis indicates that there were no significant reductions in drain box P_s with 1 mg L⁻¹ PAM ($P > 0.5$) (Fig. 5a and 5b). There were significant reductions in P_p at mid-field, field tail, and drain box ($P < 0.05$). However, these reductions were partially offset by sediment resuspension in excessively steep drain ditches, as noted earlier.

Figure 4b shows how drain box P was affected by varying PAM addition rates. Data for 0 and 1 mg L⁻¹ PAM treatments represent 17 Imperial Valley field sites, while 5 and 10 mg L⁻¹ PAM treatments represent only one field site. In general, suspended solids removal had a smaller effect on P_t removal than expected (Fig. 4b). The reason for this is unclear; perhaps variability in soil properties, cropping history, and fertilizer management among the field sites affected the relative loss of sediment and P. However, a 10 mg L⁻¹ treatment reduced P_p by 78%; this may suggest the efficacy of a higher initial PAM concentration (Fig. 4b). It is notable that none of the treatments evaluated in this study significantly decreased concentrations of P_s , the most bioavailable fraction of phosphorus load to the Salton Sea (Fig. 4b). This is contrary to the 50% reductions in P_s reported in Idaho's Snake River Valley (Lentz et al., 1998). This may be due to irrigation water salinity differences between Idaho's Snake River Valley and Imperial Valley's Colorado River water (Table 1). Higher TSS and Na concentrations in the Imperial Valley may block PAM anionic adsorption sites and reduce its efficiency to remove P_s .

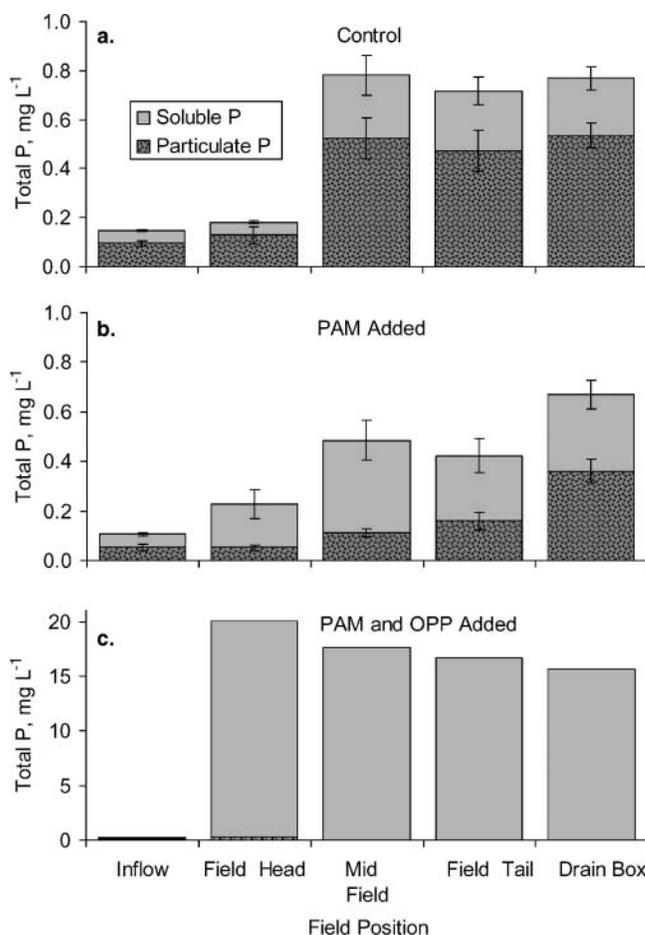


Fig. 5. Overall average P versus distance for (a) control, (b) 1 mg polyacrylamide (PAM) L⁻¹ added, and (c) a 1 mg PAM L⁻¹ plus liquid orthophosphate phosphorus (OPP) fertilizer treatment. Error bars indicate standard error of the mean.

Specifically, it has been suggested that sodium impairs Ca^{2+} adsorption and the ion bridging needed for PAM to adsorb HPO_4^{2-} (Entry and Sojka, 2003). Conversely, sodium adsorbed to anionic PAM and dissolved organic matter (DOM) exchange sites may subsequently inhibit P_s adsorption to PAM floc or even cause further dispersion of P_s from soil surfaces. Evidence for this is seen in Fig. 4b where the P_s increased slightly (albeit, not significantly) with 1 mg L⁻¹ PAM.

In this study we did not measure drain box outflow rates during the field trials. Therefore, estimates of net irrigation water infiltration and outflow P loads are not possible. However, literature suggests that infiltration in finer textured soils increases with PAM application (Shainberg et al., 1990; Sojka et al., 1998).

Effects of Liquid Phosphate Fertilizer Application on Irrigation Water

We sampled one field site while liquid orthophosphate phosphorus (OPP) fertilizer was being injected into the irrigation water at the inflow end of the field (Fig. 5c). The fertilizer concentration was 20 mg L⁻¹ P at the field head. The majority of the OPP injected left the field unadsorbed, whether in the presence of PAM or

not. Furthermore, P content of OPP amended irrigation water at the drain box was 100% P_s , the most bioavailable fraction of P.

The release of tailwater during water-run OPP fertilizer applications mainly represents a large P_s input to the Salton Sea. During our observation of a 173-ha field fertilized with 84 kg P as water-run liquid OPP fertilizer, 65 kg of P (worth US\$100) left the field at the drain box. Since the area's drains are connected to the Alamo River, this P_s was conveyed directly into the Salton Sea. In the future, offering technical assistance to growers in managing their P fertilizer usage could ameliorate this kind of situation.

Irrigation Water Polyacrylamide Concentrations

The PAM concentrations were measured in the irrigation water from inflow to drain box at three field sites (Fig. 6). Note that the apparent down-field increases in PAM concentrations for the control treatments can be attributed to high molecular weight dissolved organic matter (DOM) interference. Average DOM concentrations (in terms of total organic carbon in solution) at eight of our field sites were 5.1 and 9.0 mg L⁻¹ for inflow and drain box waters, respectively. Based on these data and the methods of Lu and Wu (2001), the variability in PAM concentration data in Fig. 6 due to DOM interference is negligible.

Although PAM was added at the rate of 1 mg L⁻¹, the maximum concentration measured was consistently less

than 0.5 mg L⁻¹ indicating that some PAM adsorbed to the inflow canal bottom before reaching the field head (Fig. 6a). The fact that peak concentrations occurred at different field locations for different field sites may be attributed to two factors: (i) powdered PAM does not immediately dissolve when sprinkled into the flowing water, thus concentration is dependent on mixing energy, and (ii) PAM adsorption may have varied with soil texture, salinity, and organic matter content. Although the rates of PAM adsorption varied between sites, adsorption was nearly complete once water reached the drain box as is evident from the near-baseline PAM concentrations at that location for all three field sites. The distance required to reduce 1 mg L⁻¹ PAM treatments to baseline levels was approximately 10 times greater than those reported for Snake River Valley soils (Lentz et al., 2002). This may be attributed to the higher salt content of the irrigation water and soils in the Imperial Valley.

At one site, PAM application rates of 5 and 10 mg L⁻¹ were tested. The PAM concentrations decreased by 40 to 50% but concentrations of >2 mg L⁻¹ were lost to the agricultural drain (Fig. 6b).

CONCLUSIONS

In this study, we demonstrated that anionic PAM applied to irrigation water can effectively reduce tailwater sediment and particulate-bound P leaving Imperial Valley agricultural fields. Continuous additions of 1 mg L⁻¹ PAM to irrigation inflows could reduce drain water TSS by 71 to 91% and P_p by 44 to 52% from bare, tilled fields.

Drain box PAM concentrations should fall below detection limits (<0.2 mg L⁻¹) following continuous 1 mg L⁻¹ PAM treatments to Imperial Valley fields. However, PAM application rates of 5 and 10 mg L⁻¹ to irrigation water could result in >2 mg L⁻¹ PAM inputs to agricultural drains.

Our original hypothesis, that PAM could remove P_s from Imperial Valley tailwater as it does in Idaho's Snake River Valley, proved false. Higher irrigation water sodium concentrations may have inhibited divalent cation (Ca²⁺) adsorption to anionic PAM. This would have reduced the cation bridging needed to adsorb P_s from tailwater.

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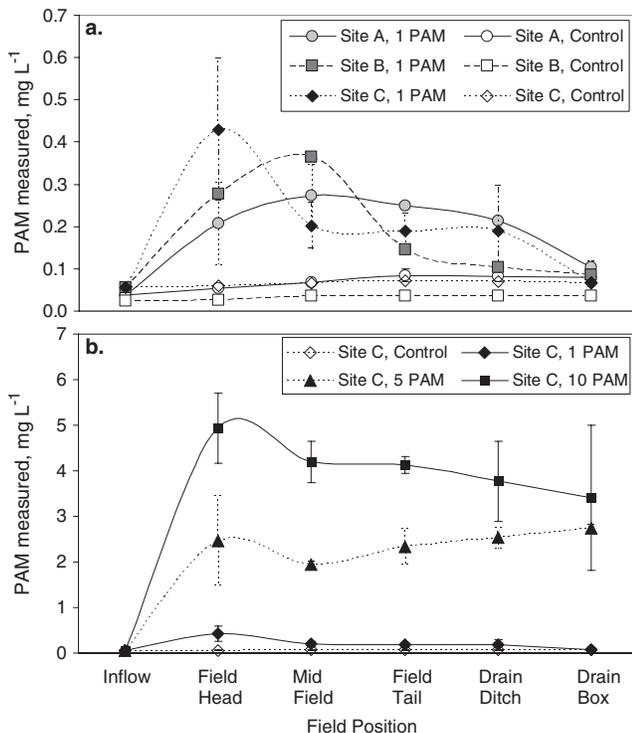


Fig. 6. Measurements of irrigation water polyacrylamide (PAM) concentrations (mg L⁻¹) for control and PAM treatments. (a) Control and 1 mg L⁻¹ PAM treatments at Sites A, B, and C; (b) control, 1, 5, and 10 mg L⁻¹ treatments at Site C only. Error bars indicate one standard error of the mean.

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