Efficacy of constructed wetlands to mitigate non-point source pollution from irrigation tailwaters in the San Joaquin Valley, California, USA

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Abstract The efficacy of using constructed wetlands (CWs) to sequester organic carbon and nutrients from irrigation tailwaters was studied in the San Joaquin Valley, California. Two CWs were monitored during the 2004 irrigation season, a new CW (W-1) and 10-year-old CW (W-2). Input/output waters from CW were collected weekly and analyzed for a variety of water quality contaminants. Organic carbon, nutrient and sediment retention efficiencies were evaluated from input/output concentrations. Characteristics of sediment were examined spatially at W-2. Results indicate that W-2 was more efficient at contaminant removal. Average particulate organic carbon retention, was 70 ± 13% (mean ± standard deviation) in W-2 and 48 ± 32% in W-1. Chlorophyll-a, a measure of algal biomass, was higher at W-1, especially in input waters. Initially, output concentration of chlorophyll-a increased 15-fold in W-2, however over time, as emergent vegetation established, chlorophyll-a decreased to 35% of input levels. Average total N removal efficiency was 45 ± 18% for W-2 compared to 22 ± 32% in W-1. Total P removal efficiency was 72 ± 14% at W-2 compared to 18 ± 26% at W-1. CWs were most effective at removing total suspended solids, 84 ± 15% and 97 ± 2% for W-1 and W-2, respectively. Results demonstrate that CWs are effective at capturing POC, sediment and nutrients from irrigation tailwaters.

Keywords Best management practices; carbon sequestration; ecosystem services; nutrient removal; water quality

Introduction

The San Joaquin River (SJR) is a hypereutrophic tributary of the Sacramento–San Joaquin Delta, the hub of California’s water supply system. Low dissolved oxygen (DO) conditions, frequently occurring between July and October (> 50% of the time), often violate the water quality objectives for DO in the lower SJR: 6.0 mg L⁻¹ from September through November to facilitate salmon migration and 5.0 mg L⁻¹ during the remainder of the year. The oxygen deficit can stress and kill aquatic organisms and has delayed the upstream migration of fall-run Chinook salmon (Oncorhynchus tshawytscha). An important component of the oxygen demand originates from high algal biomass loading from upstream sources. Algal loads are a result of excess nutrient supply, largely from non-point sources associated with irrigated agriculture (Kratzer et al., 2004). In the lower SJR, summer concentrations of mineral nitrogen (NH₄⁺ + NO₃⁻) range between 2.0 to 2.5 mg N L⁻¹ and soluble reactive phosphorus range between 0.10 and 0.15 mg P L⁻¹.

New regulations are requiring that all irrigated land managers develop and implement water quality management and water quality monitoring plans in order to obtain a waiver to discharge irrigation return flows from their properties. As a result, agricultural land managers are searching for cost-effective management practices to improve irrigation tailwaters before disposal to waterways. One possible solution is the use of flow-through wetlands, which have been shown in other regions to be effective for improving water quality (Phipps and Crumpton, 1994; Woltemade, 2000; Jordan et al., 2003; Zedler, 2003).
If effective in the San Joaquin Valley, constructed wetlands would provide an important best-management practice for enhancing water quality of tailwaters. Wetland treatment of irrigation tailwaters could provide a valuable tool for addressing the downstream hypoxia problem and the needs of land managers to address the agricultural discharge waiver.

The primary objective of this study was to understand the evolution of carbon, sediment, and nutrient dynamics within a spatial and temporal context in constructed, flow-through wetlands in the San Joaquin Valley. This will allow us to evaluate the potential of constructed wetlands (CW) of differing ages for sequestering organic carbon and nutrients in California’s irrigated agriculture. Specific objectives were to: first, examine the spatial variability of sediment deposition and associated carbon pools. And secondly, examine the retention efficiencies of these wetlands for particulate organic carbon, dissolved organic carbon, major nutrients (total N and P), chlorophyll-a, and total suspended solids.

**Methods**

The SJR has a perennial drainage area of 19,158 km² in California’s Central Valley, including portions of the Sierra Nevada (11,192 km²), Coast Ranges (2,078 km²) and Valley Basin (2,273 km²) (Kratzer *et al.*, 2004). The valley basin is among the most productive agricultural regions of California, in large part due to the availability of irrigation water. The intensive utilization of tributary runoff for irrigation commonly results in 40–50% of the summer flows in the SJR originating from surface and subsurface agricultural drainage.

Two sites were selected to compare differences in wetland design and age on wetland biogeochemistry. The CWs are located along the SJR between the confluences of the Merced and Tuolumne Rivers (Figure 1). A new CW (W-1) having an area of 1.3 ha and receiving irrigation tailwaters from about 450 ha was compared to a 10-year-old CW (W-2) wetland.
with an area of 7.3 ha and receiving tailwaters from about 2,300 ha. Both wetlands receive tailwater ultimately destined for the SJR.

Sediment deposition plates (25 × 25 cm) were placed throughout the wetland floor to measure net sedimentation over the irrigation season. At the end of the year sediment was collected from the plates, weighed and analyzed for total sediment deposition, particle size, and organic carbon.

Input and output water samples were collected on a weekly basis during the 2004 irrigation season (April–Sept.) and analyzed for several water quality constituents, including total nitrogen (TN), total phosphorus (TP), dissolved organic carbon (DOC), particulate organic carbon (POC) determined by volatile suspended solids, total suspended solids (TSS), and chlorophyll-a (a measure of algal biomass). For a description of methods, see Ahearn et al. (2004).

Results
Sediment deposition

Figure 2a illustrates spatial patterns in net sedimentation accumulated over the 2004 irrigation season for the W-2 wetland. Net sedimentation displays a spatial pattern that correlates with the water flowpath. Net sedimentation rate ranged from 0.024 to 63 kg m⁻² yr⁻¹ (Figure 2a). Sediment deposition rates were used to identify three zones with contrasting depositional patterns: depositional zone, passive zone, and transitional zone. Areas of high sedimentation, termed the depositional zone, were located around the input and follow the flowpath towards the outlets, diminishing with increasing distance from the input. A passive zone, characterized by low sediment deposition was located in areas distal to the input location. This zone appears to be a stagnant water zone disconnected from the main hydrologic flowpath. A transitional zone, areas of intermediate sediment deposition rates, was located between the depositional zone and the passive zone and had moderate sediment deposition rates.

Figure 2b summarizes spatial patterns in organic carbon content measured from the sediment plates. Organic carbon content was greatest at the two output locations and

![Figure 2](https://iwaponline.com/wst/article-pdf/55/3/55/430859/55.pdf)

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throughout the passive zone where vegetative biomass was high. Organic carbon was lowest near the input where net sedimentation was highest (Figure 2b).

Water quality effects
First-year monitoring results for the new CW (W-1) and 10-year-old CW (W-2) indicated that CWs trap a variety of constituents in agricultural return flows. The CWs were particularly efficient at removing TSS, TN, TP, and POC.

The fate of organic carbon in the water column was assessed through measurements of POC, DOC, and chlorophyll-a concentration at input and output locations. Chlorophyll-a concentration fluctuated throughout the season in W-1 ranging from 0.7 to 53.5 μg L⁻¹ at the input and 2.4 to 62.2 μg L⁻¹ at the output (Figure 3a). Average chlorophyll-a concentration was 21.1 ± 18 μg L⁻¹ (mean ± standard deviation) in input water and 26.5 ± 25 μg L⁻¹ in output water. Chlorophyll-a concentration tended to be higher at W-1 compared to W-2, especially in the tailwater inputs. Input concentrations at W-2 were relatively consistent throughout the season ranging from 0.3 to 8.2 μg L⁻¹ while output waters ranged from 0.7 to 47.3 μg L⁻¹ (Figure 3b). Average chlorophyll-a concentration was 4.2 ± 2.2 μg L⁻¹ in input water and 13.5 ± 12.6 μg L⁻¹ in output water.

![Figure 3](https://iwaponline.com/wst/article-pdf/55/3/55/430859/55.pdf)
Temporal dynamics of algal biomass in W-2 may be explained by interactions with the growing vegetation. Initially, output concentration of chlorophyll-a increased 15-fold, yet over time, as emergent vegetation established, chlorophyll-a in output water decreased to 35% of input levels, likely due to shading by the plant canopy (Figure 3b).

Both CWs were sinks for POC (Figures 3c, d). At W-1, average POC concentration of input water was $7.1 \pm 3.2 \text{mg L}^{-1}$ (range 3.5–12.7 mg L$^{-1}$) and output water was $3.2 \pm 1.7 \text{mg L}^{-1}$ (range 1.1–6.9 mg L$^{-1}$) (Figure 3c). At W-2, average POC concentration of input water was $5.4 \pm 1.8 \text{mg L}^{-1}$ (range 3.0–9.4 mg L$^{-1}$) and output water was $1.5 \pm 0.52 \text{mg L}^{-1}$ (range 0.8–2.6 mg L$^{-1}$) (Figure 3d). On average, 70 ± 13% of POC was sequestered in W-2, while 48 ± 33% was sequestered by W-1.

The new CW (W-1) appeared to be a sink for DOC while the mature wetland (W-2) may be a source during certain times of the year (Figures 3e, f). At W-1, average DOC concentration of input water was $4.8 \pm 2.1 \text{mg L}^{-1}$ (range 2.2–9.5 mg L$^{-1}$) and output water was $4.1 \pm 0.7 \text{mg L}^{-1}$ (range 2.5–4.7 mg L$^{-1}$) (Figure 3e). At W-2, the average DOC concentration of input water was $4.6 \pm 1.9 \text{mg L}^{-1}$ (range 2.5–9.4 mg L$^{-1}$) and output water was $5.2 \pm 1.4 \text{mg L}^{-1}$ (range 3.5–8.1 mg L$^{-1}$) (Figure 3f). On average, W-1 was a sink for DOC in 5 out of the 9 weeks sampled, while W-2 was a DOC sink 7 out of the 17 weeks sampled (Figure 3f).

Sediment and nutrient removal efficiencies
The results illustrate that the older wetland (W-2) was more efficient than W-1 for removal of sediments and nutrients. Average total N removal efficiency was 45 ± 18% for W-2 compared to 22 ± 32% in W-1 (Figures 4a and 4b). At W-1, removal efficiency ranged from a low of −30% to a maximum of 85% throughout the irrigation season. In comparison, W-2 removal efficiency ranged from a low of 10% to a maximum of 75% (Figures 4a and 4b). A comparison of TN concentrations in input waters and TN removal efficiency indicates that the magnitude of N removal was greatest when N input concentrations were high (data not shown).

Similar trends were observed for total P removal. Average TP removal efficiency at W-1 was 18 ± 14% compared to 72 ± 14% (following the initial wet-up) at W-2 (Figures 4c and 4d). At W-1, removal efficiency ranged from a low of −24% to a maximum of 50%, while at W-2, removal efficiency ranged from a low of −145% during initial wet-up to a high of 85% (Figures 4c and 4d). Removal efficiency at W-1 fluctuated throughout the season, whereas TP was consistently retained in W-2 after the first week. The large release of P during initial flooding of W-2 was likely due to release of P following reduction/dissolution of iron oxides in the CW sediments.

Both W-1 and W-2 were highly effective at removing suspended solids from input waters. Average removal efficiency for TSS at W-1 and W-2 was 84 ± 15% and 97 ± 2%, respectively. Although similar, TSS removal efficiency was consistently higher at W-2 (Figures 4e and 4f). Wetland size and vegetation density are greater in W-2, which may promote particle settling through reduced water velocity and increased hydraulic residence time (Braskerud, 2002).

Discussion
There are several mechanisms acting in CWs that influence the fate of carbon and other water quality contaminants including: (1) sedimentation and burial (adsorbed-P, pesticides, POC, pathogens); (2) microbial transformations to gaseous forms (denitrification, methanogenesis); and (3) plant uptake of nutrients. As a result of these processes, wetlands are considered to have a beneficial effect on water quality (Jordan et al., 2003; Zedler, 2003). Important factors controlling carbon sequestration and the water purification capacity of
wetlands include rate of contaminant inflows, hydraulic residence time, availability of organic matter and other substrates for microbial growth, and nutrient uptake demand by plants (Phipps and Crumpton, 1994; Woltemade, 2000).

The hydraulic residence time appears to play a major role in nutrient and sediment capture in CWs. Removal of N from the water column occurs through denitrification and plant uptake. P removal from the water column occurs through plant uptake and sorption to mineral particles. Removal of sediment and POC from the water column occurs through deposition. W-2 is an older and larger CW, and as a result, vegetation is more established and hydraulic residence time is longer. In addition to increasing nutrient uptake, an established plant community can increase water residence time by decreasing water velocity (Braskerud, 2002). Thus, conditions that optimize the degree of denitrification, plant nutrient uptake, sorption and sedimentation are more prevalent in W-2 compared to W-1. In time, as vegetation becomes established in W-1, we expect to observe greater removal efficiencies.

While wetlands may be a major sink for POC, nutrients and sediments, they may also be a source of algal biomass and DOC (Figures 3c–f; Tockner et al., 1999). Dissolved organic carbon (DOC) in the San Joaquin–Sacramento River Delta is a water quality concern because of the production of mutagenic and carcinogenic disinfection by-products during water treatment. In addition, these components contribute to biological oxygen demand (BOD) in wetland drainage waters and could add to the BOD load causing hypoxia in the wetland.

Figure 4 A comparison of contaminant removal efficiency for total nitrogen (TN, a and b), total phosphorus (TP, c and d) and total suspended solids (TSS, e and f) at a newly constructed- and 10-yr-old wetland in 2004.
lower SJR. The high hydraulic residence time coupled with warm water temperatures in the shallow wetlands could enhance algae production in wetlands. Similarly, removal of suspended solids by wetlands may result in less turbid conditions in the SJR that in turn could result in enhanced algal growth due to greater light availability. Therefore, processing of irrigation tailwaters in flow-through wetlands may conceivably enhance hypoxia through increased algal production both within wetlands (algae exported to river) and within the main stem of the river. Thus, flow-through wetlands could simply serve as an incubator, transforming nutrients to algal biomass, resulting in no beneficial effect on BOD loads and DO in the lower SJR. The first year results for chlorophyll-a indicate that CWs may not serve as incubators for algal growth to any large extent. Initial output concentrations at W-2 were high, but decreased over time, possibly due to the establishment of a plant canopy which decreased the light penetration needed for algal growth (Figure 3b).

Preliminary results for DOC suggest that the mineral-soil dominated CW systems in the Central Valley are not significant sources of DOC; however, additional data such as DOC loads are needed to verify these findings. W-1 was not the DOC source probably because it was newly constructed and soil organic carbon reserves were low. In contrast, W-2 tended to be a source of DOC early in the irrigation season likely due to flushing of DOC from stored humic substances that decomposed in the soil profile and decaying litter during the spring when the CW was dry.

Conclusions
The conversion of flood plain agroecosystems to flow-through wetlands is becoming a popular land-use practice nationwide, yet little information exists to document how these systems function in California where CWs dry out in late winter and spring. This project directly addresses the needs of agricultural land managers who will be required to obtain a waiver for disposal of agricultural tailwaters and total maximum daily load efforts related water quality impairment in the lower SJR. Information gained from this research will allow us to identify factors that may improve the functionality of CWs as carbon sinks and water purifiers. Constructed wetlands have the potential to be excellent organic carbon and contaminant sinks and represent the last opportunity for treatment before tailwaters are discharged back to the SJR.

References