# Carbon Dynamics in Constructed Flow-Through Wetlands: A Chronosequence of Subaqueous Soils

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## Summary

The primary goal of this study was to evaluate the efficacy of using constructed wetlands (CW) to sequester organic carbon. Two CWs were monitored during the 2004 and 2005 irrigation season (April-September), a newly constructed CW (W-1) and 13-year-old CW (W-2). The initial stage of this project encompassed baseline sampling of seasonally submerged soils and identification of appropriate strategies for water quality and flow monitoring. Soil samples were analyzed for C, N, P, and particle size. Intricate input/output flow monitoring systems were designed and tested, which allowed us to calculate constituent loads for 2005. Input/output waters from CW were collected on a weekly basis and analyzed for the following constituents: total nitrogen (TN), total phosphorus (TP), dissolved organic carbon (DOC), particulate organic carbon (POC), total suspended solids (TSS), volatile suspended solids (VSS), and chlorophyll-a (a measure of algal biomass). Carbon, nutrient and sediment retention efficiency was evaluated from input/output concentration data. After comparing sediment plate data from 2004 and 2005, it was discovered that CWs have a great potential to store carbon.

Results indicate that both wetlands were efficient at removing POC and contaminants. Average POC retention, indicated by VSS, in 2004 was 75% in W-2 and 66% in W-1. In 2005, POC retention decreased in W-2 (48%) and remained the same at W-1 (68%). Chlorophyll-a (a bio-indicator of algae) tended to be higher at W-1 compared to W-2, especially in input water. In 2004, chlorophyll-a concentration [Chyll] at W-2 was regulated by light limitation from the wetland vegetation. Initially, output [Chyll] increased 15 fold in W-2, however, as emergent vegetation was established, chlorophyll-a decreased to 35% of input levels. In 2005, due to the absence of vegetative cover, there was a 7.7-fold increase in [Chyll] at the output. While W-1 was generally a sink for DOC in 2004, it became a source in 2005 likely due to organic matter build up from the previous year. W-2 was often a source of DOC, possibly due to leaching of DOC from vegetation and litter. Average TN removal efficiency was 31% at W-1 and 41% at W-2 in 2004. In 2005, removal efficiencies decreased to 21% at W-1 and 29% at W-2. After an initial release of P, due to establishment of reducing conditions in the wetland sediments, average TP removal efficiency was 63% at W-2 compared to 28% at W-1 during the 2004 irrigation season. While in 2005, TP removal efficiencies decreased to 35% at W-2 and 8% at W-1. CWs were most effective at removing TSS with average removal efficiency in 2004 of 84 and 97% for W-1 and W-2, respectively. Removal efficiencies were also high in 2005 with 88% removal at W-1 and 82% removal at W-2.

CWs are effective at capturing sediment and nutrients removed from irrigated farmland. Our results demonstrate that CWs act as sinks for organic carbon associated with eroded topsoil and POC, but in the early portion of the irrigation season, mature CWs may be a source of DOC.

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# Objectives

During the growing season, tailwater is the main water supply for constructed wetlands (CWs) in the Central Valley. Our goal was to evaluate the potential of CWs as a best management practice for irrigated agriculture to sequester organic carbon and improve water quality. The primary objective of this study was to understand the evolution of carbon, sediment, and nutrient flow within a spatial and temporal context using a chronosequence of seasonally submerged soils of constructed, flow-through wetlands in the San Joaquin Valley.

## *Specific objectives:*

1. Examine source and input/output budgets for particulate organic carbon, dissolved organic carbon, major nutrients (TN, TP, soluble-reactive $PO_4$, $NH_4$, $NO_3$), chlorophyll-a, and total suspended solids (TSS) in inflow/outflow waters.
2. Examine redox development in CW sediments and shallow ground waters along the hydrologic flowpath to assess the potential for methanogenesis.
3. Examine the spatial relationships between seasonally submerged soil properties, hydraulic flowpath, water quality, macrophyte, phytoplankton, periphyton NPP, and soil organic matter pools.

# Approach and Procedures

This monitoring and research project measured soil properties, vegetation biomass and input/output water volumes and contaminant concentrations at two CWs that receive irrigation return flows ultimately destined for the San Joaquin River (SJR). The sites were selected to compare differences in wetland design (age, size, configuration, and volume), emergent vegetation and hydrologic regime (e.g., quality and quantity of input/output water and hydrologic residence time). The CWs are located along the SJR between the confluences of the Merced and Tuolumne Rivers east of Patterson and Crows Landing and are representative of multiple projects funded by the Wetland Reserve Program and Environmental Quality Incentives Program (EQIP).

The two CWs were monitored during the 2004 and 2005 irrigation seasons (April-September) included a newly constructed wetland CW (W-1) and a 13-year-old CW (W-2). W-1 is a small wetland (1.3 ha) with a dendritic pattern of submerged micro lows and more well drained micro highs. W-2 is large (7.3 ha) with an open water design similar to a pond (*fig 1*). We also identified, instrumented, and surveyed a third wetland (W-3) of identical design and age to W-2, however, a breach in the system rendered it un-usable during the 2004 or 2005 irrigation seasons.

For all sites, soil, biomass and sedimentation samples were collected and analyzed for particle size, C, N, and P and were used to develop nutrient and carbon budgets. CW soils were sampled after the wetland was constructed and before it received water for W-1 and at the onset of the project for W-2. Additionally, sediment plates (over 50 per site) were placed throughout the CW in order to collect sediment each irrigation season and determine sedimentation rates and compare properties of each year's sediment with that of the original soil properties.

Input/output waters were collected on a weekly basis in 2004 and 2005 and analyzed for several water quality constituents, including total nitrogen (TN), total phosphorus (TP), dissolved organic carbon (DOC), total suspended solids (TSS), volatile suspended solids (VSS), and chlorophyll-a (a measure of algal biomass). Particulate organic carbon (POC) was calculated from carbon analysis on VSS samples. A great deal of time and effort was devoted to the design and testing of a flow monitoring system for input/output waters. This system was tested in 2004 and was fully operational at each CW in 2005. Input/output flow volumes were used to calculate input/output budgets for all water quality constituents.

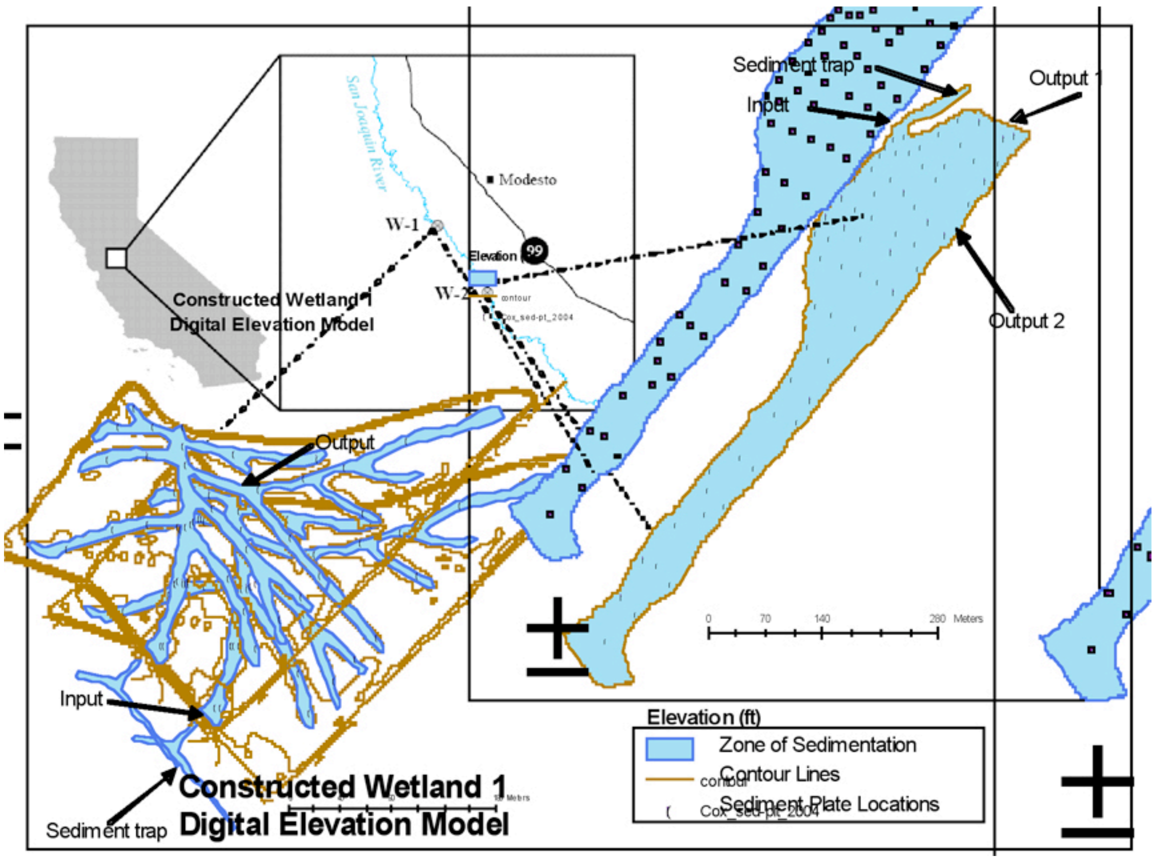


**Figure 1.** Diagram of W-1 and W-2, points indicate sediment plate sampling locations for W-1 and W-2.

In 2005, redox probes were installed at two locations within each CW in regions that reflected differences in depositional environment and water residence time. At each location eight redox probes were installed at three depths (three probes at 2.5 cm, three probes at 5 cm, and two probes at 10 cm). Redox probes were connected to data loggers, which recorded oxidation/reduction potential (ORP) readings every 15 minutes. We monitored ORP in each wetland to assess the potential for methanogenesis to occur. Monitoring began at the end of the irrigation season (early September) and continued past wetland dry-down (mid December).

Statistical significance tests for constituents in the water column were conducted using the Wilcoxon Rank Sum Test in the R statistical program. Geostatistical interpolation was performed on constituents of the wetland floor using the Regularized Spline with Tension interpolation method using GRASS software.

# Results

Results for the newer CW (W-1) and 13-year-old CW (W-2) suggest that CWs trap a variety of constituents in agricultural return flows. The CWs are particularly efficient at removing sediment, nitrogen, phosphorus and particulate organic matter (POC-as determined from VSS). Concentrations of constituents ([constituent X]) varied between years and between sites.

## *Nutrient and Suspended Sediment Removal from Agricultural Tailwaters*

### Nitrogen

Nitrogen levels were relatively high and variable in input waters. Encouraging N retention levels was achieved by the CWs, especially from W-2. In most instances, much of the total N was in the form of nitrate.

Variability in [TN] was highest at W-1 and corresponding TN removal efficiencies were low (*table 1*). In 2004, average [TN] at W-1 was 14.4±10.5 $mgL^{-1}$ (mean ± standard deviation) at the input and 10.0±3.6 $mgL^{-1}$ at the output. In 2005, average [TN] was 8.4±6.1 $mgL^{-1}$ at the input and 6.6±4.3 $mgL^{-1}$ at the output (*table 1*). There was no significant reduction in concentration between input and output in either year, despite modest retention efficiencies. Corresponding seasonal TN retention efficiencies were 31% for 2004 and 21% for 2005 (*table 1*).

The [TN] at W-2 was lower and less variable at input and output locations (*table 1*). In 2004, average [TN] removal was significant with a mean of 5.8±1.6 $mgL^{-1}$ at the input and 3.4±1.5 $mgL^{-1}$ at the output ($p<0.01$). In 2005, however, there was no significant change in concentration between locations with an average [TN] of 4.1±2.2 $mgL^{-1}$ at the input and 2.9±1.2 $mgL^{-1}$ at the output (*table 1*). Corresponding seasonal TN removal efficiencies were 41% for 2004 and 29% for 2005 (*table 1*).

### Phosphorus

The ability of the CWs to remove phosphorus varied between years and between sites, with W-2 having a higher efficiency than W-1 in both years. In 2004, average [TP] at W-1 was 0.432±0.246 $mgL^{-1}$ at the input and 0.313±0.122 $mgL^{-1}$ at the output. In 2005, average [TP] was 0.357±0.159 $mgL^{-1}$ at the input and 0.328±0.270 $mgL^{-1}$ at the output (*table 1*). Although output concentration was slightly lower than the input at W-1, there was no significant difference between input and output during either year. Corresponding seasonal TP removal efficiencies were 28% for 2004 and 8% for 2005 (*table 1*).

In 2004, input TP concentration at W-2 was lower and less variable when compared to W-1 (*table 1*). However, in 2005, the concentration and variability of TP were similar between W-1 and W-2 at inputs and outputs, although more P was removed by W-2 (*table 1*). W-2 was more efficient at removing TP in 2004 with an average [TP] of 0.275±0.080 $mgL^{-1}$ at the input, which was reduced to 0.103±0.068 $mgL^{-1}$ at the output. This resulted in a significant reduction in TP between the input and output ($p<0.01$). Similarly, in 2005, W-2 retained more TP than W-1 with an average [TP] of 0.386±0.165 $mgL^{-1}$ at the input and 0.253±0.127 $mgL^{-1}$ at the output (*table 1*), resulting in a significant reduction in [TP] ($p<0.01$). Corresponding seasonal TP removal efficiencies for W-2 were 63% for 2004 and 35% for 2005 (*table 1*).

***Table 1.*** *Seasonal mean concentration of water quality constituents at input and output locations at W-1 and W-2 (mean ± standard deviation).*

| | | | Total Nitrogen | Nitrate | Total Phosphorus | Dissolved Reactive Phosphorus | Total Suspended Solids |
|---|---|---|---|---|---|---|---|
| | | | TN | $NO_3$ | TP | DRP | TSS |
| | | | -----mg $L^{-1}$----- | | | | |
| W-1 | 2004 | | | | | | |
| | | Input | 14.4 ± 10.5 | 8.7 ± 3.5 | 0.432 ± 0.246 | 0.382 ± 0.181 | 469 ± 301 |
| | | Output | 10.0 ± 3.6 | 7.7 ± 3.1 | 0.313 ± 0.122 | 0.277 ± 0.117 | 47 ± 37 |
| | | Retention† | 31% (ns) | 13% (ns) | 28% (ns) | 28% (ns) | 90% (**) |
| | 2005 | | | | | | |
| | | Input | 8.4 ± 6.1 | 4.3 ± 2.8 | 0.357 ± 0.159 | 0.223 ± 0.154 | 285 ± 293 |
| | | Output | 6.6 ± 4.3 | 4.7 ± 3.9 | 0.328 ± 0.270 | 0.179 ± 0.222 | 35 ± 41 |
| | | Retention† | 21% (ns) | 9% (ns) | 8% (ns) | 20% (ns) | 88% (**) |
| W-2 | 2004 | | | | | | |
| | | Input | 5.8 ± 1.6 | 4.2 ± 1.5 | 0.275 ± 0.080 | 0.188 ± 0.049 | 305 ± 185 |
| | | Output | 3.4 ± 1.5 | 2.1 ± 1.2 | 0.103 ± 0.068 | 0.043 ± 0.067 | 7 ± 3 |
| | | Retention† | 41% (**) | 50% (**) | 63% (**) | 77% (**) | 98% (**) |
| | 2005 | | | | | | |
| | | Input | 4.1 ± 2.2 | 2.8 ± 2.1 | 0.386 ± 0.165 | 0.162 ± 0.063 | 430 ± 271 |
| | | Output | 2.9 ± 1.2 | 2.0 ± 3.3 | 0.253 ± 0.127 | 0.100 ± 0.097 | 78 ± 41 |
| | | Retention† | 29% (ns) | 29% (*) | 35% (**) | 38% (*) | 82% (**) |

†Retention efficiencies were calculated as the percent reduction in seasonal mean concentration between input and output locations. Significance level between input and output mean concentrations is given in parenthesis. ns= not significant, *=significant at $P<0.05$, **=significant at $P<0.01$.

Similar results were observed for dissolved reactive phosphorus (DRP). At W-1, DRP removal efficiencies were not significant with 28% retention in 2004 and 20% retention in 2005. At W-2, there was a significant removal of DRP with 77% removal in 2004 ($p<0.01$) and 38% in 2005 ($p<0.05$) (*table 1*).

## *Total Suspended Solids*

The mean [TSS] in output water at W-1 was lower than input water over both irrigation seasons (*table 1*). Input levels of total suspended solids were very high throughout much of the irrigation season in 2004. Episodes of low TSS at the input in 2005 were observed to correspond with the presence of PAM (a soil aggregating agent) in the irrigation water resulting in a high standard deviation in both years. In 2004, average [TSS] was 469±301 mg$L^{-1}$ at the input and 47±37 mg$L^{-1}$ at the output. In 2005, average [TSS] was 285±293 mg$L^{-1}$ at the input and 35±41 mg$L^{-1}$ at the output (*table 1*). Corresponding seasonal TSS removal efficiencies were 90% for 2004 and 88% for 2005, resulting in significant reductions in [TSS] in both years (p<0.01) (*table 1*).

At W-2, input levels of TSS were slightly lower than W-1 for 2004, but higher in 2005. In 2004, average [TSS] was 305±185 mgL$^{-1}$ at the input and 7±3 mgL$^{-1}$ at the output. In 2005, average [TSS] was 430±271 mgL$^{-1}$ at the input and 78±41 mgL$^{-1}$ at the output (*table 1*). Corresponding seasonal TSS removal efficiencies were 98% for 2004 and 82% for 2005, resulting in significant reductions in [TSS] in both years (p<0.01) (*table 1*).

## *Carbon Removal from Agricultural Tailwaters*

The fate of organic carbon in the water column was assessed through measurements of POC, DOC and chlorophyll-a concentration at input/output locations.

***Table 2.*** *Seasonal mean concentration of carbon constituents at input and output locations at W-1 and W-2 (mean ± standard deviation).*

| | | | Dissolved Organic Carbon | Particulate Organic Carbon | Chylorophyll-a |
|---|---|---|---|---|---|
| | | | DOC | POC | Chyll-a |
| | | | ------------------mg L$^{-1}$------------------ | | -------µg L$^{-1}$------ |
| W-1 | 2004 | | | | |
| | | Input | 4.3 ± 2.3 | 7.0 ± 2.6 | 21.1 ± 18.1 |
| | | Output | 3.7 ± 1.0 | 2.4 ± 0.9 | 26.5 ± 25.0 |
| | | Retention† | 14% (ns) | 66% (**) | -26% (ns) |
| | 2005 | | | | |
| | | Input | 3.3 ±0.8 | 5.6 ± 3.2 | 18.7 ± 7.6 |
| | | Output | 3.8 ± 1.4 | 1.8 ± 1.0 | 22.5 ± 24.0 |
| | Retention† | | -12% (ns) | 68% (**) | -20% (ns) |
| W-2 | 2004 | | | | |
| | | Input | 4.9 ± 1.9 | 5.7 ± 1.9 | 4.2 ± 2.3 |
| | | Output | 5.1 ± 1.3 | 1.4 ± 0.3 | 13.5 ± 12.6 |
| | | Retention† | -4% (ns) | 75% (**) | -221% (**) ¶ |
| | 2005 | | | | |
| | | Input | 4.0 ± 1.0 | 7.0 ± 2.8 | 8.0 ± 10.1 |
| | | Output | 4.4 ± 1.2 | 3.6 ± 1.4 | 69.5 ± 42.4 |
| | | Retention† | -10% (ns) | 48% (**) | -769% (**) § |

†Retention efficiencies are calculated as the percent reduction in seasonal mean concentration between input and output locations. Significance level between input and output mean concentrations is given in parenthesis. ns= not significant, *=significant at P<0.05, **= significant at P<0.01.

¶Corresponds to ~2 fold increase

§Corresponds to 7.7 fold increase

## Chlorophyll-a

Chlorophyll-a was used as a bio-indicator of algae. High algal loads have been linked to low dissolved oxygen levels in the Stockton Ship Channel acting as a barrier to spawning fish. CWs have the potential to serve as bioreactors for algae when the hydraulic residence times and nutrient levels are high.

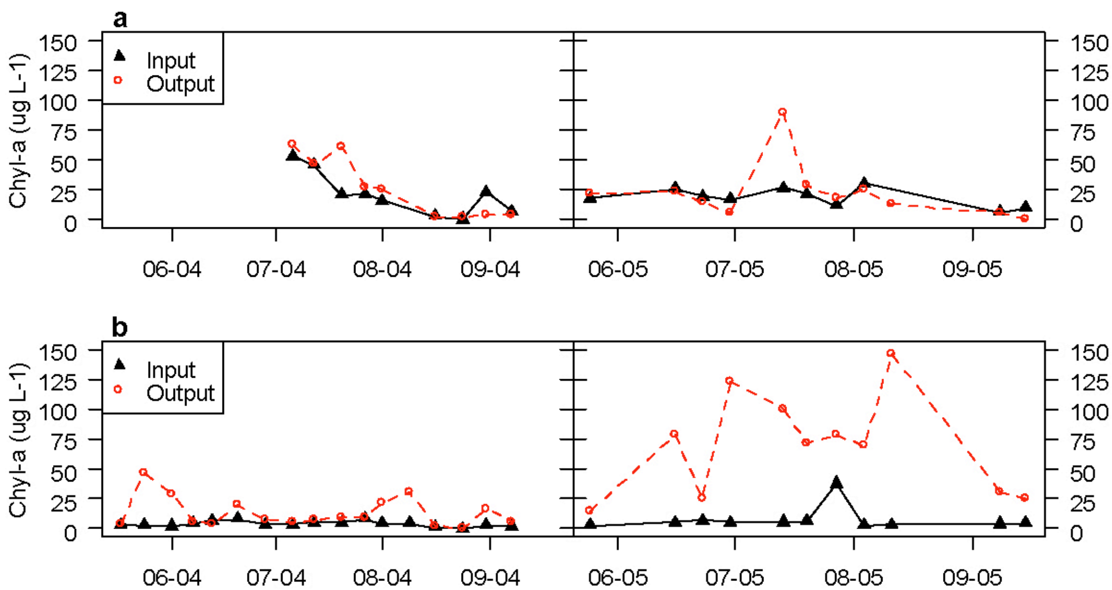


**Figure 2.** Chlorophyll-a, a bio-indicator of algae, measured at (a) W-1 and (b) W-2 at input and output locations over the 2004 and 2005 irrigation seasons.

In general, trends in chlorophyll-a concentration ([Chyll]) were similar at input and output locations at W-1 with episodes in time where it was a source of algae (*fig. 2a*). In 2004, average [Chyll] was 21.1±18.1 $\mu gL^{-1}$ at the input and 26.5±25.0 $\mu gL^{-1}$ at the output. In 2005, average [Chyll] was 18.7±7.6 $\mu gL^{-1}$ at the input and 22.5±24.0 $\mu gL^{-1}$ at the output (*table 2*). When considering seasonal averages, W-1 was a source of algae with seasonal removal efficiencies of -26% for 2004 and -20% for 2005 (*table 2*). The increase in [Chyll] at outputs, however, was not significant in either year.

Trends in chlorophyll-a concentration were even more variable at W-2, particularly the output (*fig. 2b*). In 2004, average chlorophyll-a concentration was very low, 4.2±2.3 $\mu gL^{-1}$ at the input and 13.5±12.6 $\mu gL^{-1}$ at the output. In 2005, average [Chyll] was low at the input, 8.0±10.1 $\mu gL^{-1}$, but much higher at the output 69.5±42.4 $\mu gL^{-1}$ (*table 2*). When considering seasonal averages, W-2 was a major source of algae, increasing over two fold ($p<0.01$) at output locations in 2004 and almost 8 fold ($p<0.01$) in 2005 (*table 2*).

In 2005, YSI data sondes were deployed at W1 and W2 output locations to assess the temporal variability of [Chyll] (*fig. 3*). A continuous record of data [Chyll] throughout the season was not possible due to sensor malfunction during certain periods. Data from W-1 output shows a muted signal with only slight fluctuations in [Chyll]. Conversely, at W-2 output, [Chyll] showed dramatic fluctuations through the season, as well as the presence of a diel signal, indicating a longer residence time than W-1 (*fig. 3b*).

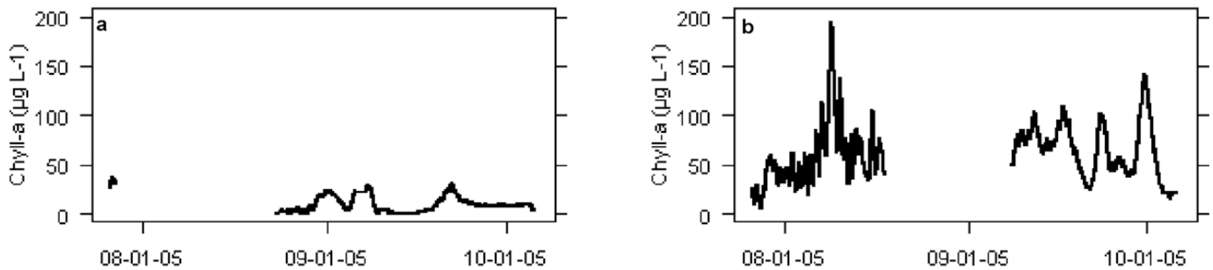


**Figure 3.** Chlorophyll-a, a bio-indicator of algae, measured at 30min increments using a YSI 6600 Sonde. (a) W-1 and (b) W-2 at output locations over the 2005 irrigation seasons.

# Dissolved Organic Carbon

Dissolved organic carbon (DOC) was measured because some forms of DOC react with chlorine during the water purification process to form carcinogenic disinfection byproducts. DOC can also constitute a major pool in the carbon budget in wetlands. There were no clear trends in [DOC] through the 2004 and 2005 irrigation season at W-1, and [DOC] was relatively low for wetland systems (*fig. 4a*). In 2004, average [DOC] was 4.3±2.3 $mgL^{-1}$ at the input and 3.7±1.1 $mgL^{-1}$ at the output. In 2005, average [DOC] was 3.3±0.8 $mgL^{-1}$ at the input and 3.7±1.3 $mgL^{-1}$ at the output (*table 2*). Corresponding seasonal DOC removal efficiencies were 14% for 2004 and -12% for 2005 (*table 2*). W-2 showed no clear trends in [DOC] over the 2004 and 2005 irrigation season (*fig. 4b*). In 2004, average [DOC] was 4.9±1.9 $mgL^{-1}$ at the input and 5.1±1.3 $mgL^{-1}$ at the output. In 2005, average [DOC] was 4.0±1.0 $mgL^{-1}$ at the input and 4.4±1.2 $mgL^{-1}$ at the output. Corresponding seasonal DOC removal efficiencies were -4% for 2004 and -10% for 2005 (*table 2*).

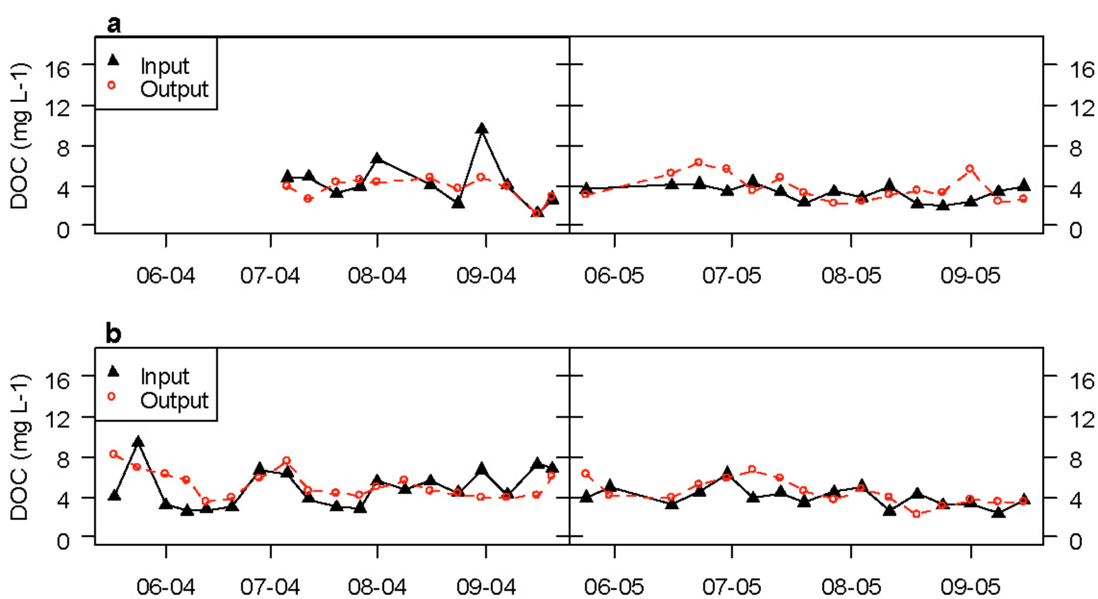


**Figure 4.** Dissolved organic carbon measured at (a) W-1 and (b) W-2 at input and output locations over the 2004 and 2005 irrigation seasons.

## Particulate Organic Carbon

Both CWs were sinks for particulate organic carbon (POC) (*figs. 5a & 5b*). In 2004, average [POC] at W-1 in input water was 7.0±2.6 mg $L^{-1}$ and 2.4±0.9 mg $L^{-1}$ in output water (*fig. 5a*). In 2005, average [POC] was 5.6±3.2 mg $L^{-1}$ at the input and 1.8±1.0 mg $L^{-1}$ at the output. Corresponding seasonal POC removal efficiencies were 66% for 2004 and 68% for 2005 (*table 2*). The [POC] in output water at W-2 was consistently low in 2004, but increased during the 2005 irrigation season (*fig 5b*). In 2004, average [POC] was 5.7±1.9 mg $L^{-1}$ in input water and 1.4±0.3 mg $L^{-}1$ in output water. In 2005, average [POC] was 7.0±2.8 mg $L^{-1}$ at the input and 3.6±1.4 mg $L^{-1}$ at the output. Corresponding seasonal POC removal efficiencies were 75% for 2004 and 48% for 2005 (*table 2*).

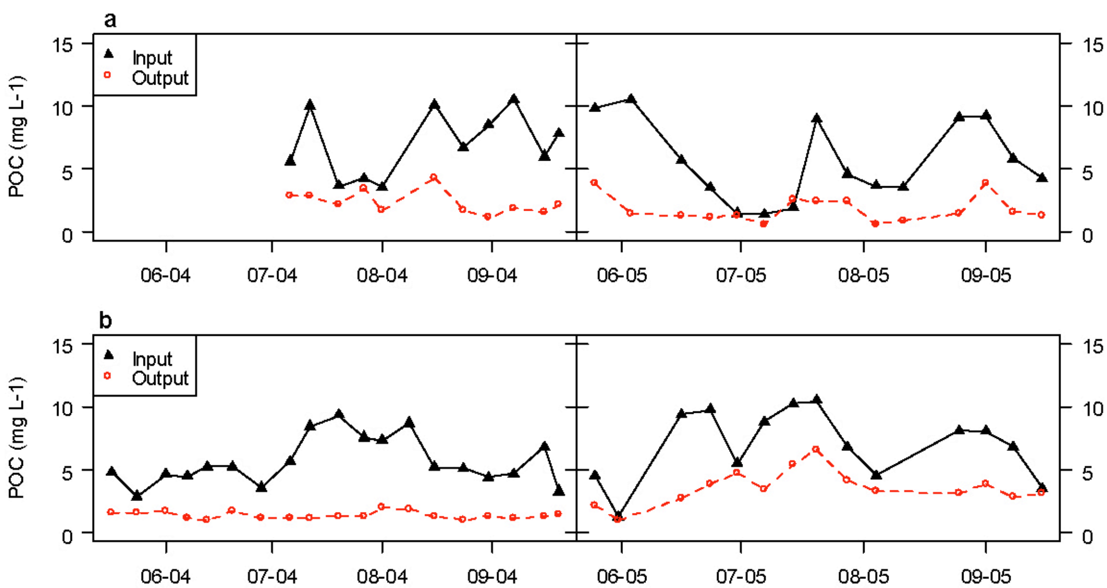


**Figure 5.** Particulate organic carbon measured at (a) W-1 and (b) W-2 at input and output locations over the 2004 and 2005 irrigation seasons.

## *Carbon Fluxes in the Wetland Water Column*

Carbon loads were calculated at input and output locations in 2005. Loads were calculated by multiplying the concentration of a constituent at a given time interval by the corresponding flow rate. In general, flow rates ranged from 1 to 6 cubic feet per second (cfs) but varied significantly throughout the irrigation season (*fig. 6*). Flow rates at W-1 were controlled by input weirs, which diverted a portion of total flow from the adjacent drainage ditch into the wetland. During the first half of the irrigation season, flow was low due to the regulation of input flows by the land manager, which were increased later in the season (*fig. 6*). At W-2, input and output flows were not regulated by control structures. Due to late spring flooding, flow monitoring at W-2 was not possible until late June.

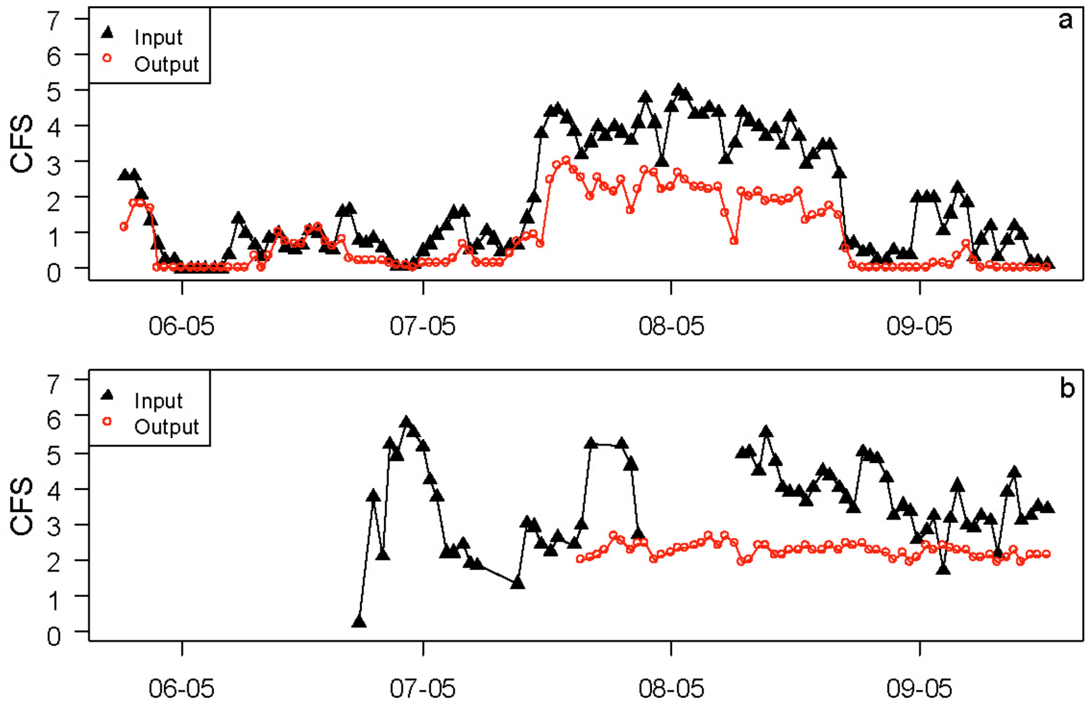


**Figure 6.** Input and output hydrograph for (a) W-1 and (b) W-2 during the 2005 irrigation season.

Seasonal loads and retention efficiencies for DOC, POC, and chlorophyll-a are presented in table 3. Retention efficiencies were calculated as the percent difference between input and output seasonal loads for each wetland. Percent loss between net seasonal in-flowing and out-flowing water fluxes was 47% for W-1 and 35% for W-2. W-1 was more effective at removing DOC, POC and chlorophyll-a loads than W-2 (*table 3*). This higher retention is believed to be due to both a shorter hydraulic residence time and thus less in-situ carbon production, and a higher reduction in total flow between input and output. There was moderate removal of DOC loads at both wetlands with 49% reduction at W-1 and 31% reduction at W-2. Retention of POC was high with 84% reduction at W-1 and 67% reduction at W-2. Retention efficiency for chlorophyll at W-1 was moderate with 36% retention, while at W-2, it was a source of chlorophyll with over a four-fold increase (-438%) in the output load relative to the input. The optimal conditions for the removal of carbon from surface waters differ, depending on the constituent. For example, the removal of POC requires a long residence time for particles to settle. While in contrast, a long residence time can cause chlorophyll to proliferate when light is not limiting. Thus finding wetland conditions that maximize the removal of carbon from the water column is important. One possible scenario would be to increase the residence time to decrease POC, while at the same time encouraging the establishment of a wetland canopy to minimize algae production, as seen at W-2 in 2004.

***Table 3.*** *Seasonal Carbon Loads at W-1 and W-2 in 2005*

| | | | Dissolved Organic Carbon | Particulate Organic Carbon | Chylorophyll-a |
|---|---|---|---|---|---|
| | | | DOC | POC | Chyll-a |
| | | | -------------------------kg $yr^{-1}$-------------------------- | | |
| W-1 | | | | | |
| | | Input | 1637 | 2823 | 10.2 |
| | | Output | 836 | 453 | 6.5 |
| | | Retention† | 49% | 84% | 36% |
| W-2 | | | | | |
| | | Input | 3687 | 6600 | 7.9 |
| | | Output | 2549 | 2163 | 42.6 |
| | | Retention† | 31% | 67% | -438% |

†Retention efficiencies are calculated as the percent reduction in seasonal load between input and output locations.

## *Potential for Methanogenesis in Wetland Sediments*

Methanogenesis is a pathway leading to loss of organic carbon from CWs. Methane is produced in flooded wetland soils by the anaerobic decomposition of organic matter. The oxidation/redox potential (ORP) of a soil has a direct relationship to $CH_4$ formation, with low soil Eh values, indicating the potential for methane production. The microbial oxidation of organic matter requires an electron acceptor, which under idealized conditions follows a sequence of preferential utilization (e.g. $O^2 \rightarrow NO^3 \rightarrow Mn \rightarrow Fe \rightarrow SO^4 \rightarrow CO^2$). Under these idealized conditions, the reduction of $CO^2$ and production $CH_4$ is predicted to occur in the ORP range of -250 to -350 (Mitsch and Gosselink 1993). However, these predicted values do not always accurately predict the upper limit of $CH_4$ production, due to the heterogeneity of soils and ability of several electron acceptors to be reduced simultaneously. Several studies have found $CH_4$ to occur at higher Eh values with and Eh of -150 mV considered to be the critical value for the initiation of methane production (Wang et al. 1993). Thus, we set -150 mV as the beginning range for potential methane production. At W-1, the ORP at both Sites 1 and 2 were highly reduced (Eh<-150) at 2.5 cm for a short period at the end of the irrigation season (September) (*figs. 7a & 7d*), followed by an increase in ORP during dry-down at all depths (*figs 7a-7f*). Site 2 showed a slower increase in ORP than Site 1, due its lower topographic position. The sediments at W-2 were more highly reduced than those at W-1 as shown by the lower ORP readings at both sites. W-2 was highly reduced at 2.5cm and 5cm depths at Site 1 (*figs. 7g & 7h*) and 5cm and 10cm depths at Site 2 (*figs. 7k & 7l*).

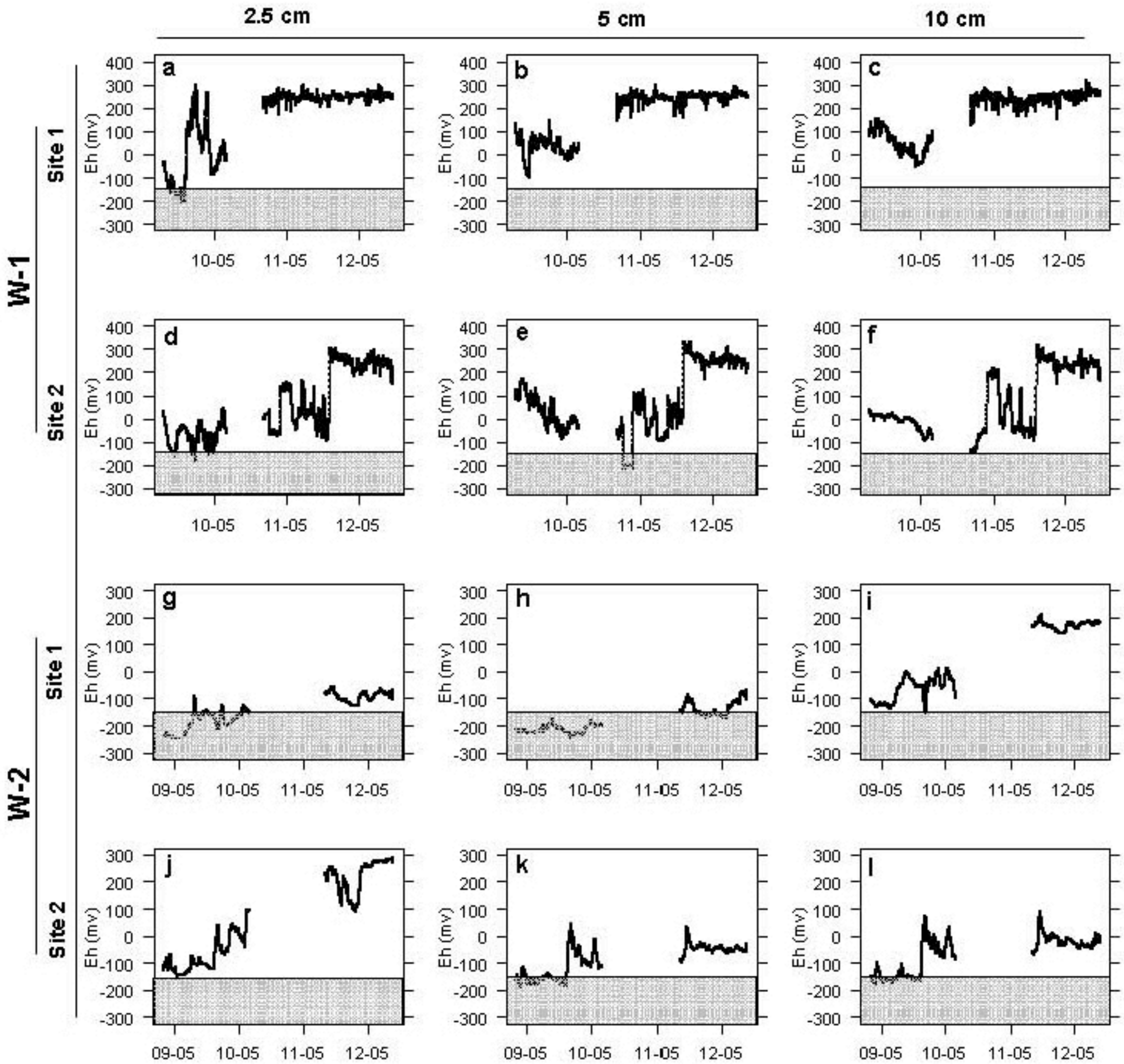


**Figure 7.** Redox potential measurements (Eh) taken at 30-min. increments at three depths (2.5cm, 5cm, and 10cm) in the wetland sediment. Measurements at each of the locations are averages based on triplicate measurement at 2.5cm and 5cm depths and duplicate measurements at 10cm. Shaded area indicates potential zone of methanogenesis.

## *Spatial and Temporal Distribution of Carbon and Sediment*

Analysis of sediment plates was used to understand the role of sedimentation on carbon dynamics in CWs. Interpolation maps illustrating the spatial distribution of sediment in 2004 and 2005 at W-1 were generated using the Regularized Spline with Tension (RST) interpolation method (*figs. 8a & 8b*). All subsequent interpolation maps were also generated using the RST method. In 2004, spatial patterns show the highest rates of sedimentation occurring along the main flowpath (channel running from input to output), with moderate accumulation occurring in the eastern side-channels and low rates of accumulation occurring in the western side-channels (*fig. 7a*). The average rate of sedimentation in 2004 was 14.0 kg $m^{-2}$ $y^{-1}$, with rates ranging from 228 kg $m^{-2}$ $y^{-1}$ close to the input, to rates of less than 0.03 kg $m^{-2}$ $y^{-1}$ in the western side-channels.

In 2005, rates of accumulation increased slightly along the main channel, in addition to providing a relatively more even distribution of sedimentation rates across both the eastern and western side-channels (*fig. 8b*). Rates of sediment accumulation nearly doubled from 2004, with an average deposition rate of 27.8 kg m$^{-2}$ y$^{-1}$. Sedimentation rates ranged from 465 kg m$^{-2}$ y$^{-1}$ in the area close to the input, to rates of 0.83 kg m$^{-2}$ y$^{-1}$ in portions of the southeastern side-channels.

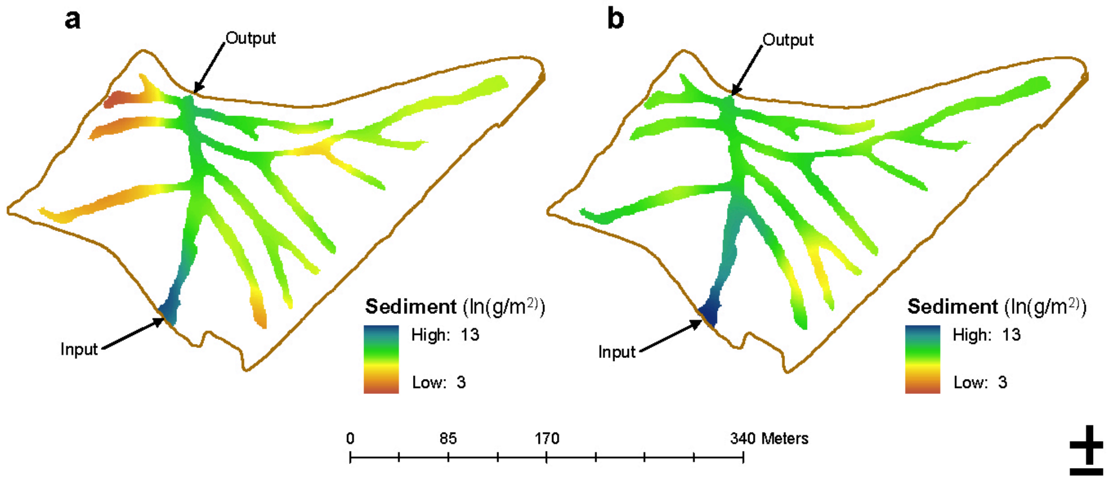


**Figure 8.** Spatial patterns in sedimentation in (a) 2004 and (b) 2005 at W-1.

The spatial distribution of carbon showed a similar pattern to that of sediment in 2004 with the majority of accumulation occurring along the main channel, moderate accumulation along the eastern side-channels and minimal accumulation along the western side-channels (*fig. 9a*). The average rate of carbon accumulation in 2004 was 0.26 kg m$^{-2}$ y$^{-1}$, with rates ranging from 3.15 kg m$^{-2}$ y$^{-1}$ close to the input, to rates as low as 0.026 kg m$^{-2}$ y$^{-1}$ in the western side-channels (*table 4*). In 2005, carbon distribution followed similar spatial patterns to that of sediment, with slightly higher rates of accumulation along the main channel near the input and a fairly even distribution throughout the remainder of the wetland (*fig. 9b*). Average rates of carbon accumulation increased from 2004 with an average accumulation rate of 0.33 kg m$^{-2}$ y$^{-1}$ (*table 4*). Rates of accumulation ranged from 1.09 kg m$^{-2}$ y$^{-1}$ in the area close to the input, to rates as low as 0.053 kg m$^{-2}$ y$^{-1}$ in certain portions of the southeastern side-channels.

The average sediment accumulation rate at W-2 nearly doubled from 2004 to 2005, with rates of 5.8 kg m$^{-2}$ y$^{-1}$ (range: 0-80 kg m$^{-2}$ y$^{-1}$) in 2004 and 11.9 kg m$^{-2}$ y$^{-1}$ (range: 0-93 kg m$^{-2}$ y$^{-1}$) in 2005 (*table 4*). In 2004, the establishment of a dense canopy of wetland macrophytes confined the zone of active sedimentation to the regions proximal to input and output locations (*fig. 10a*). In 2005, due to early spring flooding, wetland vegetation failed to establish resulting in an open water environment throughout the wetland. The absence of vegetation allowed sediment to travel to the farther reaches of the wetland before eventually settling out (*fig. 10b*).

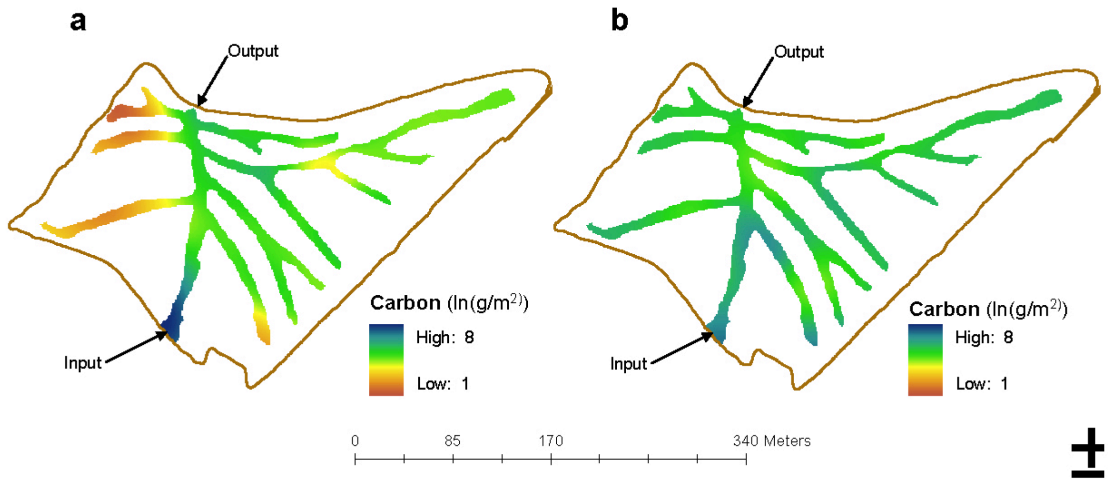


**Figure 9.** Spatial patterns in the annual soil organic carbon accumulation rate in (a) 2004 and in (b) 2005 at W-1.

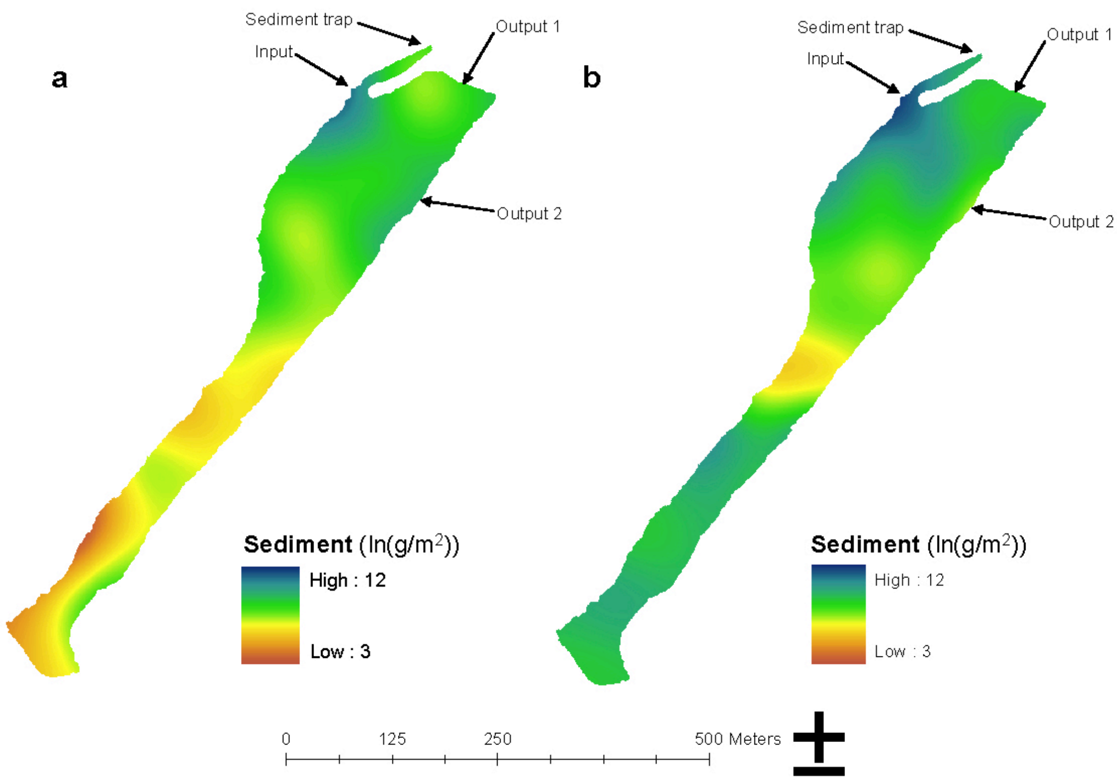


**Figure 10.** Spatial patterns in sedimentation in (a) 2004 and (b) 2005 at W-2.

At W-2, the carbon accumulation rate, averaged for the entire wetland, did not change between years, with rates of 0.290 kg $m^{-2}$ $y^{-1}$ in 2004 and 0.294 kg $m^{-2}$ $y^{-1}$ in 2005 (*table 4*). In 2004, spatial patterns of C accumulation were dependant on the addition of both allochthonous and autochthonous sources of organic matter. Allochthonous sources originate from terrestrial plants and soil organic carbon from fields, while autochthonous sources originate from organic matter associated with wetland vegetation. The contribution of each of these sources can be seen by comparing the distribution of sediment (*fig. 10*) to the distribution of carbon (*fig. 11*). Areas of high carbon accumulation that correlate with high sediment accumulation indicate that allochthonous organic matter is the dominant source of carbon. Conversely, areas of high carbon accumulation with corresponding low sediment accumulation rates indicate that autochthonous sources of carbon dominate.

We measured C/N ratios at W-2 to help identify organic matter sources (*figs. 12a & 12b*). The C/N ratio of sediment is influence by the presence or absence of cellulose in organic matter. Nonvascular plants such as algae contain low C/N ratios (between 4 and 10), whereas vascular plants, such as terrestrial or emergent wetland plants, contain higher C/N ratios (20 and greater) due to the presence of cellulose (Meyer and Ishiwatari 1993). In 2004 and 2005, carbon accumulation was greatest surrounding the input, due to high sediment accumulation and the addition of allochthonous organic matter. The spatial patterns of C/N ratios differed dramatically between years, indicating potentially different sources of organic matter. In 2004, C/N values averaged 10.0, with values ranging from 7.2 to 12.0 and zones of particularly high C/N were scattered at opposite ends and in the middle of the wetland. In 2005, C/N ratios decreased with an average of 7.8 and values ranging from 3.8 to 10.8 and higher C/N ratios were present towards the distal end of the wetland.

In 2004, the output locations and areas at the far reaches of the CW also contained high carbon accumulation rates due to the high density of emergent vegetation, providing enrichment of the wetland sediments with autochthonous organic matter. When the distribution of carbon is examined in terms of concentration (g $kg^{-1}$) (data not shown), carbon concentration tends to be lowest at the input location and along the sedimentation path, and highest at output locations and at the far reaches of the CW where water tends to be shallow. This indicates that areas high in autochthonous organic matter provide higher amounts of carbon per unit mass than areas high in allochthonous sources. Due to the absence of vegetation in 2005, patterns of carbon accumulation in W-2 correlate with patterns of sediment deposition, indicating that allochthonous organic matter associated with sediment may contribute to the majority of carbon accumulation (*figs. 10b & 11b*). However, lower C/N ratios in 2005 indicate that algal sources of carbon may contribute a significant amount of total carbon inputs.

***Table 4.*** *Annual sediment and carbon loads and accumulation rates for W-1 and W-2 seasonally submerged soils in 2004 and 2005.*

| | | Annual Sediment Load | Annual Carbon Load | Average Sediment Accumulation Rate | Average Carbon Accumulation Rate |
|---|---|---|---|---|---|
| | | ----------------kg $yr^{-1}$---------------- | | ------kg $m^{-2}$ $yr^{-1}$---- | ------kg $m^{-2}$ $yr^{-1}$---- |
| W-1 | 2004 | 181,498 | 3,433 | 14.0 | 0.264 |
| | 2005 | 361,058 | 4,338 | 27.8 | 0.334 |
| W-2 | 2004 | 423,952 | 21,178 | 5.8 | 0.290 |
| | 2005 | 871,595 | 21,480 | 11.9 | 0.294 |

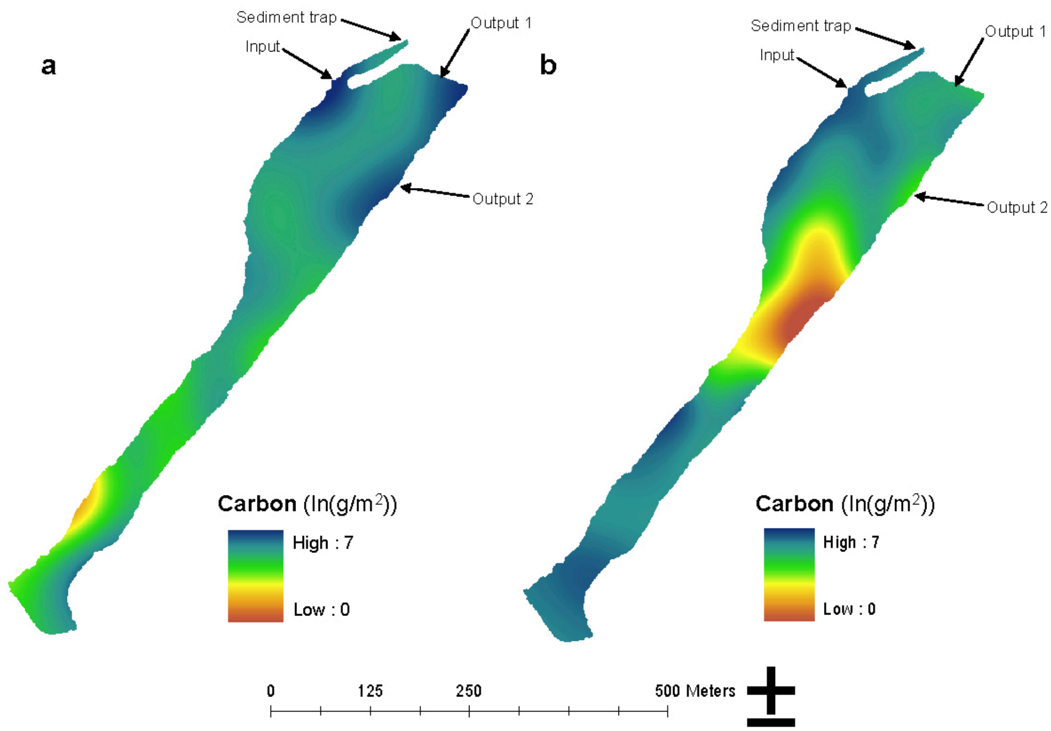


**Figure 11.** Spatial patterns in the annual soil organic carbon accumulation rate in (a) 2004 and (b) 2005 at W-2.

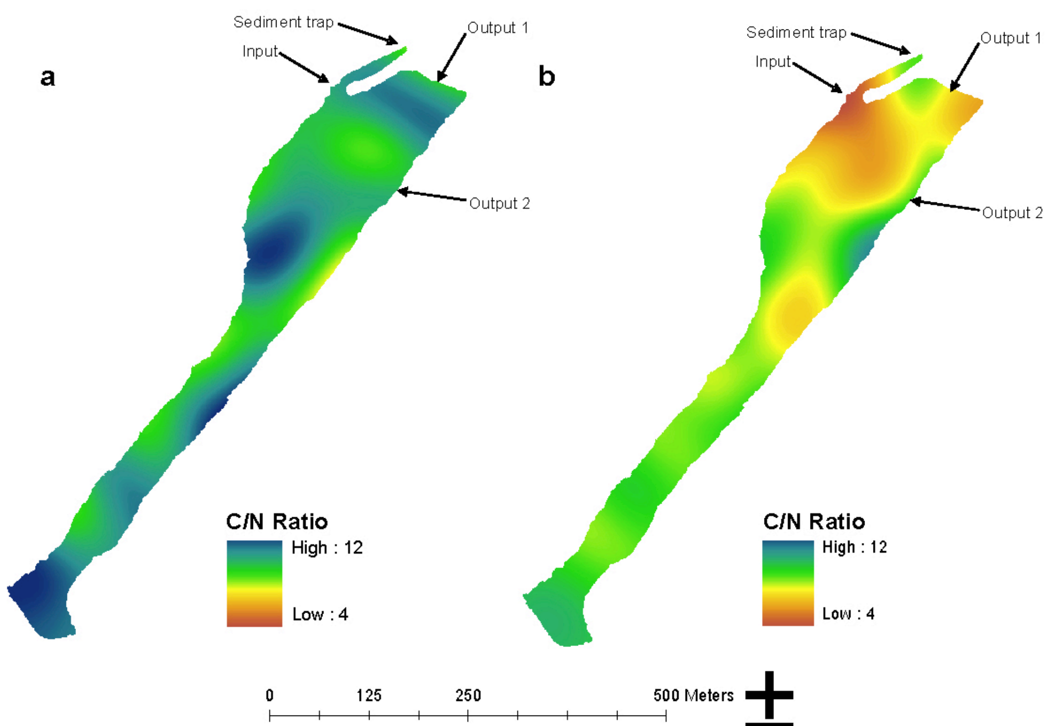


**Figure 12.** Spatial patterns in carbon to nitrogen ratios in (a) 2004 and (b) 2005 at W-2.

# Discussion

The biogeochemical cycling of carbon in wetland ecosystems occurs in both the water column and the soil/sediment. These carbon pools constitute key components in a number of physical, chemical and biological processes. From a water quality standpoint, the cycling of carbon in the water column is important because it regulates dissolved oxygen concentration and in some cases is a precursor of DBPs. Carbon cycling in the soil/sediment layer is important in driving soil redox reactions and in serving as a potential reservoir for long-term carbon storage.

There are several mechanisms acting in constructed wetlands 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, dimethylselinide); and 3) plant uptake of nutrients. As a result of these processes, it is commonly considered that wetlands 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 growth of microbes, and nutrient uptake demand by plants (Phipps and Crumpton 1994; Woltemade 2000).

## *Nutrient and Sediment Removal*

Constructed wetlands are an effective management practice to reduce non-point source pollution in irrigated agriculture. Although residence time was not directly measured in the CWs, the sensitivity of fluctuations in output flow to changes in input flow is a relative indication of residence time. At W-1, output flows changed according to changes in input flows within a short lag time ( t<1day) (*fig. 6*). In contrast, output flow at W-2 was invariant to fluctuation in input flow, suggesting that the system is highly buffered, and thus, has a relatively long residence time. The hydraulic residence time of CWs appears to play a major role in determining the efficiency of nutrient and sediment capture. The removal of N, for instance, requires a residence time, long enough for water to make contact with the sediment layer where, under reduced conditions, denitrification can occur. Residence time also affects the infiltration and/or diffusion of N into the sediment layer, which is required for plant uptake to occur. Lower residence times in W-1 may have contributed to the lower TN and $NO_3$ retention efficiencies.

In addition to hydraulic residence time, dentrification is also dependant on an adequate supply of labile organic carbon. Thus, low TN and nitrate removal efficiencies at W-1 may also be attributed to the lower concentration of carbon in sediment (Average SOC in top 10cm=10.7±2.9 g $kg^{-1}$). While in contrast, W-2 had a significant reduction in TN and nitrate in 2004 ($p<0.01$) and a significant reduction in nitrate in 2005 ($p<0.05$), likely due to its higher concentration of SOC (Average SOC in top 10cm=16.9±2.9 g $kg^{-1}$). This suggests that as constructed wetlands age and carbon accumulates in sediments, the retention of N may increase.

P removal from the water column occurs through plant uptake and sorption to mineral particles. Thus, long residence times and the establishment of wetland vegetation are important factors in improving P retention. This can be seen in W-2, where in 2004, a long residence time combined with a dense stand of vegetation, resulted in significant reductions in both TP and dissolved reactive P (DRP) ($p<0.01$) (*table 1*). In 2005, the absence of vegetation at W-2 reduced retention efficiencies, but retention of P forms remained significant (TP, $p<0.01$; DRP,

$p<0.05$) (*table 1*). Significant P removal efficiencies in 2005 at W-2 could be a result of higher TSS in input waters or a greater hydraulic residence time compared to W-1. Due to its lack of vegetative cover and shorter residence time, W-1 had no significant retention of P during either year (*table 1*). This inability to retain P at W-1 could be due to: 1) less than optimal water management at the site, 2) minimal emergent vegetation, 3) small wetland size and low water residence time, or 4) may suggest that an ageing effect plays a role in nutrient removal capacity.

Removal of sediment from the water column occurs through particle settling. Significant reductions in TSS ($p<0.01$) occurred in both wetlands during both years. High TSS retention efficiencies were achieved at W-1 despite a shorter residence time than W-2 and a lack of vegetation. The beneficial effect of vegetation on TSS removal can be seen at W-2, where 98% of TSS was removed when dense vegetation was present in 2004, while this was diminished to 82% with the absence of vegetation in 2005. An established plant community increases water residence time by decreasing water velocities, which promotes particle settling (Braskerud 2002). In evaluating W-1 and W-2, it is clear that the 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.

Sharp decreases in nutrient removal efficiencies were observed in both wetlands in 2005. This decrease in removal efficiency is the result of compounding factors. W-2 for instance, did not have an extended dry period before the irrigation season in 2005 and, as a result, dense vegetation did not emerge. Dense vegetation increases water residence time, which promotes particle settling and nutrient uptake. We attribute the decrease in removal efficiency at W-1 to poor flow regulation; in 2005 water residence time was observed to be much shorter than in 2004. The variation in nutrient removal from 2004 and 2005 indirectly suggests that wetland management can be optimized to enhance removal efficiencies.

## *Carbon Dynamics in Water Samples*

Low dissolved oxygen conditions frequently occur in the lower SJR due to excessive amounts of oxygen demanding substances. 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. Flow-through wetlands have the potential to mitigate non-point source nutrient loads and, thus, prevent algal loading and low DO conditions. However, the 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 biological oxygen demand (BOD) loads and DO in the lower SJR.

Temporal trends in [Chyll] at W-1 indicate that no significant increase in [Chyll] occurred within the wetland in either 2004 or 2005. This is likely due to a short hydraulic residence time. In contrast, at W-2 highly significant ($p<0.01$) increases in [Chyll] occurred, most notably in 2005 (*table 2, fig. 2b*). Temporal dynamics of algae primary productivity in W-2 may be explained by interactions with the growing vegetation. Initially in 2004, output [Chyll] 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 (*fig. 2b*). In 2005, the vegetative canopy failed to establish due to early spring flooding resulting in elevated [Chyll] at the output throughout the season (*fig. 2b*). High-resolution sampling of [Chyll] in 2005 further illustrates the difference in wetland characteristics influence chlorophyll-a dynamics. The muted [Chyll] signal at W-1 (*fig. 3a*), indicates a short residence time, preventing the growth of algae within the system. While in contrast, W-2 shows dynamic fluctuations in [Chyll] indicative of longer residence times allowing algae to proliferate.

Dissolved organic carbon (DOC) in the SJR/Delta system is a water quality concern because of production of mutagenic and carcinogenic disinfection by-products (DBP) 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 lower SJR (e.g., Stockton Deep Water Ship Channel). Results from [DOC] data suggest that the mineral dominated CW systems in the Central Valley are not significant sources of DOC. There was no significant difference ($p<0.05$) between input and output locations at either site or during either year (*table 2*). Additionally, no clear seasonal trend was detected at either site (*figs. 4a & 4b*).

Both wetlands were major sinks for POC, with significant reductions ($p<0.01$) in concentration at output locations (*table 2*). POC retention followed a similar trend to that of TSS at W-1, due to similar retention mechanisms. W-2 deviated from this slightly, due to higher rates of *in situ* organic matter production. This was particularly evident in 2005 with much higher concentrations of chlorophyll-a.

## *Methanogenesis*

The redox potential of the soil provides insight into the type of constituents being reduced. Due to the limited supply of electron acceptors in highly reduced soils, methanogenesis is often the primary pathway driving organic matter decomposition. Factors controlling methane emissions include, soil redox conditions, the availability of substrate, and temperature (Trettin and Jurgensen 2003). When an abundant supply of organic carbon is present in reduced soils, strong gradients in redox potential can develop across short distances (e.g., 2mm) (Howeler and Bouldin 1971).

Wetland vegetation can have a variety of effects on both total methane emissions and the mechanisms by which it is released. Plants serve as sources of substrate for decomposition, as active site for methane oxidation, and as conduits for the release of soil methane (Bartlett and Harriss 1993). Rates of methane flux vary seasonally with the greatest losses exhibited during the summer months (Trettin and Jurgensen 2003). Due to the absence of emergent vegetation at either site during the sampling period, it is believed that methane loss was minimal. The release of methane through plants is one of the dominant exchange mechanisms for many wetland ecosystems, with seasonal variations in flux attributable to the dynamics of plant growth and senescence.

The redox potential of soils at W-1 were highly reduced at the beginning of the monitoring period, but soon became more oxidized as the soils began to dry-down (*figs. 7a-7f*). At W-2, the soils were more highly reduced than those at W-1 and remained highly reduced for a longer duration (*figs. 7g-7l*). This is most likely due to the higher availability of SOC to fuel reduction reactions. A soil Eh value of -150mV was chosen as the critical value for the initiation of methane production in reduced soils. However, we recognize that considerable variability exists in reduced soils, with $CH_4$ production potentially occurring at Eh values higher or lower than our identified value. Based on this estimate, we can see that both wetlands reached Eh values below -150 mV, indicating the potential for methane production. However, since Eh values were only slightly below Eh -150 mV and for a short duration, it does not appear that $CH_4$ production was an important process during our monitoring in 2005. Further monitoring of ORP is required to more accurately assess the potential for methane production during years with emergent vegetation as well as earlier in the season to assess seasonal variation.

## *Spatial Temporal Distributions of Carbon and Sediment*

Wetlands serve as carbon sinks within a landscape through their retention of sediment and associated organic matter. Wetlands are also highly productive ecosystems, capable of producing and incorporating large amounts of biomass into sediment. Studies examining sedimentation dynamics in constructed wetlands have shown that newly constructed wetlands have higher rates of sedimentation than older wetlands (Fennessy et al. 1994; Braskerud 2001; Craft 2003). This agrees with our findings in which the younger wetland, W-1, had higher rates of sedimentation than the older wetland, W-2. However, in addition to differences in wetland age, W-1 and W-2 also vary considerably in wetland design, with differing hydrologic flow paths.

The effects of CW design (volume, shape, and configuration of input/outputs) on hydrologic flow path and residence time may play an important role in carbon and nutrient capture. The spatial distribution of sediment and nutrient accumulations within a wetland can be highly heterogeneous, depending upon preferential flow-paths and proximity to inputs (Reddy et al. 1993). In W-1 changing micro-topography played a dominant role in shaping sedimentation patterns. In 2004, varying inflow volume and changing management strategies resulted in a low standing water depth. This resulted in lower sediment accumulation rates in the western side-channels due to their higher topographic position. In 2005, higher inflow volumes and a higher standing water depth created a more even distribution of sediment and organic carbon throughout the wetland.

Annual variation in vegetative cover can also significantly effect the spatial distribution of both sediment and carbon within a wetland. This was illustrated in W-2, where in 2004, high rates of sedimentation where confined to the area proximal to the input, where vegetative cover was dense. Emergent vegetation is known to increase sedimentation by slowing water velocities, providing a substrate for particles to adhere, and preventing re-suspension (Braskerud 2001). While in 2005, sediment accumulation was dispersed throughout the wetland likely due to the absence of emergent vegetation (*figs. 9a & 9b*). Thus, without the presence of vegetation, suspended sediments entering the wetland in 2005 had the potential to remain in suspension longer as well as being more susceptible to re-suspension.

The presence of aquatic vegetation affects carbon accumulation rates by enriching sediments with autochthonous organic matter. In 2004, the presence of dense vegetation provided high amounts of autochthonous organic matter in areas outside of the main zone of deposition (*fig. 11*). Autochthonous forms of organic matter tend to be more structurally intact than allochthonous forms, thus supplying the sediment layer with a greater long-term source of carbon (Wetxel 2001). In our analysis, we found that allochthonous OM had a lower concentration of C per unit mass relative to autochthonous OM.

# Conclusions

The conversion of flood plain agroecosystems to flow-through wetlands is becoming a popular land-use practice nation wide, yet little information exists to document how these systems function in California where CWs dry out in late winter and spring. This project directly addressed the needs of the Kearney mission, several TMDL efforts and related water quality issues in the lower SJR. Information gained from this research and monitoring program has allowed 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 re-circulated back to the SJR.

Human-induced disturbances to the geomorphic/hydrologic landscape have transformed the fluvial sediment transport system of the continent. Accelerated erosion, largely attributable to agriculture, and the construction of sediment sinks (e.g., water catchment reservoirs, farm ponds, wetlands and other terrestrial deposits) are responsible for much of this change (Renwich et al. 2005). Approximately 2.6 million small (surface areas smaller than approx. $10^4$ $m^2$), artificial water bodies are scatted across the continental United States. These small features are believed to receive one third of all eroded materials in the U.S. (Smith et al. 2002). Although the fate of organic carbon in transported sediments is unknown, it is clear that once sediments reach these subaqueous depositional areas, rates of sediment respiration are suppressed. Thus, carbon accumulation in small subaqeous environments can have a significant effect on carbon storage within a watershed.

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