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Mechanical shredding of water hyacinth (*Eichhornia crassipes*): impacts to water quality in the Sacramento-San Joaquin River Delta, California

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Abstract

Management actions to control invasive aquatic species can have significant ecosystem-scale effects. We evaluated the water chemistry and nutrient effects of mechanical shredding to control water hyacinth (*Eichhornia crassipes*) in an agricultural slough and a tidal wetland on the Sacramento-San Joaquin River Delta, California. Shredding was conducted with two types of shredder boats in fall of 2003, and another boat in spring of 2004. Overall, shredding measurably affected water quality, but specific effects varied as a function of shredding site and season. Significant increases were observed for total Kjeldahl nitrogen and total phosphorus for all experiments. Dissolved oxygen impacts varied by site, decreasing after shredding at the agricultural slough but increasing at the tidal wetland. The increase in dissolved oxygen likely resulted from tidal incursions from the adjacent river. A year-long time series of dissolved oxygen data indicated a negative relationship between hyacinth abundance and dissolved oxygen concentrations. Hyacinth contained similar tissue concentrations of mercury (Hg) to underlying sediments, suggesting that plant harvesting could aid Hg remediation efforts. Simple mass calculations indicated that Delta-wide shredding operations could cause between 0.1% and 9.6% increases in the overall abundance of carbon, nitrogen, and phosphorus in the Delta water column. Results suggest that local effects of management actions to control invasive aquatic plants will vary widely as a function of site-specific hydrology, but that Estuary-wide effects would be limited.

Introduction

In shallow water habitats, invasive aquatic vascular plants (macrophytes) are ecosystem engineers with wide-ranging impacts. Community level impacts of invasive macrophytes include reductions in native plant abundance and diversity, and in habitat or prey availability for native fish (Madsen 1997; Killgore and Hoover 2001; Toft et al. 2003). Ecosystem level impacts include alterations in dynamics of productivity and contaminant partitioning (Madsen et al. 1988; Carignan and Neiff 1992; Madsen 1997; James et al. 2002; Riddle et al. 2002; Rommens et al. 2003) and changes to sediment dynamics and geomorphology (Scheffer et al. 1993; Craft et al. 2003). Infestations can spread rapidly, abetted by water currents, diverse recruitment strategies, and human institutional barriers to control resulting from unclear jurisdictional boundaries. Due to a general lack of public awareness or effective enforcement, macrophyte invasions are frequently abetted by the aquarium trade, nursery sales, and recreational boating activity (Kay and Hoyle 2001). These plant invasions have significant economic impacts by impeding boating activities, recreation, and delivery of culinary and irrigation water (Anderson 1990; Madsen 1997). Consequently, substantial economic resources are expended in control of these aquatic plants, predominantly via herbicide application directly to surface waters (Pimentel et al. 2000).

The invasive water hyacinth (*Eichhornia crassipes*) is considered to be one of the most noxious aquatic weeds, due to its rapid growth, adverse effects on native flora and fauna, and economic impacts. Native to tropical lowlands of South America, water hyacinth has invaded over 50 countries on five continents, and is particularly widespread in lakes and estuaries of Southeast Asia, Central and West Africa, Central America, and the Southeastern United States (Penfound and Earle 1948; Barrett 1989; Carignan and Neiff 1992; Charudattan et al. 1996; Rommens et al. 2003; Albright et al. 2005). As with many introduced aquatic plants, the primary control method for water hyacinth has been targeted application of aquatic herbicides (Charudattan et al. 1996). For example, in the Sacramento-San Joaquin River Delta, California (hereafter, the Delta), substantial infestations of water hyacinth and other invasive macrophytes have been controlled for decades, using primarily chemical herbicide applications (Anderson 1990; California Department of Boating and Waterways 2004). Due to recent legal challenges to aquatic herbicide application in the western U. S., permitting and monitoring requirements have increased (U. S. Ninth Circuit Court of Appeals 2001; Siemering et al. 2005). This has reduced the cost-effectiveness of herbicide application, resulting in reevaluation of alternative control methods (Greenfield et al. 2006).

Public perception is often favorable to non-chemical control methods. Mechanical plant harvesting is a frequently used alternative to herbicide application, but harvesting is relatively costly and time consuming (Madsen 1997; Greenfield et al. 2006). Shredding of hyacinth shoots, and leaving them in the water column to die and senesce, has lower control costs than harvesting (Stewart and McFarland 2000; Greenfield et al. 2006). Large-scale shredding operations (without vegetation removal) have recently been undertaken in Lake Victoria, Africa, and Lake Champlain, Vermont (James et al. 2002), and are presented in some statewide aquatic plant management plans (e.g., Texas Parks and Wildlife Department 2005). Transfer of nutrients to the water column, oxygen depletion, and associated water quality impacts may result from either mechanical shredding or chemical herbicide application (Tucker et al. 1983; Madsen 1997; James et al. 2002). If shredding were undertaken at a regional scale, releases of nitrogen, carbon, phosphorus, and trace metals could be substantial, possibly resulting in fundamental shifts in water body trophic state (Scheffer et al. 1993).

Water hyacinth are floating plants that absorb and immobilize nutrients directly from the water column (Klumpp et al. 2002; Rommens et al. 2003). Natural senescence of hyacinth is generally slow, with the majority of decay and nutrient release occurring during fall and winter (Carignan and Neiff 1992; Pinto-Coelho and Greco 1999; Battle and Mihuc 2000). Shredding during spring or summer would cause a pulse of bioavailable nutrients during periods of high algal production. As water hyacinth bioconcentrate and sequester mercury (Hg) in their tissues, the potential Hg pool in Delta hyacinth tissues may also be released (Lenka et al. 1992; Riddle et al. 2002). This sudden release of nutrients by plant shredding might be expected to cause eutrophication and consequent ecosystem stress (Scheffer et al. 1993; James et al. 2002). But in the highly turbid Delta, primary productivity is generally limited by light availability and activity of benthic grazers, rather than nutrient abundance (Jassby et al. 2002). The Delta also experiences strong tidal advection, which may rapidly disperse nutrients released by shredding or other ecosystem manipulations (Lucas et al. 2002). Organic carbon released by shredding could conceivably increase secondary production in the Delta, which is potentially carbon-limited (Jassby and Cloern 2000; Sobczak et al. 2002).

We evaluated the water chemistry impacts of large-scale experimental mechanical shredding operations on water hyacinth in two Delta water bodies. We compared conventional limnological parameters before versus after shredding to assess extent and duration of impacts. We also determined concentrations of Hg in plant tissues, water, and sediments, to assess the role that water hyacinth harvesting could play in Delta Hg remediation. Finally, we used the shredding experiment results in combination with Delta-wide hyacinth abundance information to estimate potential effect of Delta-wide shredding operations on overall nutrient mass in the water column.

Methods

Study Area

The Sacramento-San Joaquin River Delta is a network of tidal channels, sloughs, and lakes that drains from the confluence of the Sacramento and San Joaquin Rivers to San Francisco Bay. Over the past several decades, the Delta has been impacted by metal and pesticide contamination, introduced species invasion, aquatic habitat alteration, and shifts in primary and secondary production (Domagalski et al. 2000; Jassby and Cloern 2000; Sobczak et al. 2002; Foe 2003; Kimmerer 2004). Delta contamination includes Hg from historic mining operations causing elevated concentrations in water, sediments, and fish tissue (Domagalski et al. 2000; Davis et al. 2003). Recently, a combination of interacting factors has caused a reduction in Delta phytoplankton, zooplankton, and fish production. Competition with invasive bivalve species (*Corbicula fluminea* and *Potamocorbula amurensis*) for high quality organic carbon sources (i.e., phytoplankton) is a probable cause of the decline in zooplankton, and consequently fish (Jassby and Cloern 2000; Jassby et al. 2002; Müller-Solger et al. 2002; Kimmerer 2004). Therefore, management actions to increase bioavailable forms of carbon have been recommended (Sobczak et al. 2002).

Managers of the Delta and associated watersheds must contend with the complex and sometimes conflicting objectives of water delivery for human use, water quality, habitat restoration, and protection of ecosystem processes and native species (Kimmerer et al. 2005). Management is further complicated by spatial and temporal heterogeneity in variables such as river flow, tidal mixing, channel depth, vegetation density, and water quality (California Department of Boating and Waterways 2001; Lucas et al. 2002; Kimmerer 2004). For example, the lower San Joaquin River has relatively high nutrient concentrations and primary production, compared to other portions of the Delta (Lehman et al. 2004). In 1995, the CALFED Bay-Delta Program was established to integrate Delta management oversight (Kimmerer et al. 2005). Nevertheless, management of introduced aquatic plants is administered by private individuals and separate state agencies (e.g., California Department of Boating and Waterways, California Department of Food and Agriculture) that are not well coordinated with the CALFED program.

Two sites on the Delta were chosen for shredding evaluation (Figure 1a), Lambert Slough (Elk Grove, CA; 38° 19.254’ N, 121° 28.686’ W) and Dow Wetland (Antioch, CA; 38° 01.242’ N, 121° 50.038’ W). These sites are representative of the variable conditions found in the Delta (see also Greenfield et al. 2006). Dow Wetland (Figure 1b) is strongly tidally influenced, densely infested with water hyacinth, and abuts the main-stem San Joaquin River. Lambert Slough is an irrigation ditch, divided into eastern and western channels by a dirt levee (Figure 1c), and connected by an underground culvert to a Delta backwater slough. Tidal influence is muted, and inflow and outflow are limited.

Shredding Operation

Mechanical shredding was conducted using three separate vessels over three operations in 2003 and one operation in 2004. In 2003, two shredders were evaluated, each built and operated by an independent contractor (Master's Dredging, Lawrence, KS; http://sunflower.com/~cleanh2o/). The "Amphibious Terminator," a modified airboat, having a set of flail chopper blades, and a standard airboat fan, was operated in East Lambert Slough on September 6 and 8, 2003. The "AquaPlant Terminator," an 8.5 m long barge, equipped with sets of shredding blades at the front and rear of the boat, was operated in West Lambert Slough from September 19 – 21, 2003, and in Dow Wetland from September 22 – 24, 2003. On June 3, 2004, a "Cookie Cutter," leased and operated by a local contractor (Clean Lakes, Inc., Martinez, CA; http://www.cleanlake.com/), was employed in Dow Wetland (Greenfield et al. 2006).

Chemistry Analysis

In 2003, water quality was monitored at one shredding station in Dow Wetland (D1) and four stations impacted by shredding at Lambert Slough (LE1, LE2, LW1, and LW2). In 2003, four reference (unshredded) stations were also monitored at Dow Wetland (R1, R2, R3, R4) (Table 1, Figure 1). For all stations, sampling was conducted on at least two dates prior to shredding and several dates following shredding. In 2004, it was expected that shredding effects would be more short-lived, and the sampling design was changed to estimate immediate water quality impacts with spatial replication. Specifically, water quality data were collected at four shredding stations in the Dow Wetland (D1, D2, D3, D4). These data were collected on three dates: June 1 (prior to shredding), June 3 (within one hour following shredding), and June 7 (four days after shredding). Additionally, a datalogging monitor (YSI Sonde 6920) was established to monitor turbidity, dissolved oxygen, and conductivity at 15 minute intervals at station D1, between May 22 and June 25, 2004.

On each sampling date, water grab samples were collected 0.3 m beneath the water surface in precleaned HDPE-plastic bottles (glass bottles for dissolved organic carbon), and shipped on ice to analytical labs for filtration and preparation within 48 hr. The following parameters were analyzed in the laboratory: total phosphate (TP), dissolved reactive orthophosphate (OP), total Kjeldahl nitrogen (TKN), dissolved nitrate + nitrite ($NO_3 + NO_2$), biochemical oxygen demand (BOD), dissolved organic carbon (DOC), total suspended solids (TSS), and turbidity. Laboratory analyses were performed using standard EPA and APHA protocols (U. S. EPA 1983; Clesceri et al. 1998). TKN was determined by sulfuric acid digestion followed by boric acid absorption and sulfuric acid titration. $NO_3 + NO_2$ was determined colorimetrically, after the reduction of nitrate to nitrite on a copperized cadmium column and subsequent reaction of nitrite with sulfanilamide and N-(1-naphthyl)ethlyenediamine dihydrochloride. Phosphorus was determined colorimetrically, after reduction to molybdenum blue. Prior to analysis, samples for OP were filtered with 0.45 micron filters, and TP samples were digested with sulfuric acid at 115°C. BOD was determined as the depletion of oxygen after five day dark incubation at 20°C. DOC was determined by persulfate ultraviolet oxidation. Turbidity was analyzed using a Hach Model 2100P turbidimeter. Analyses were performed at Sierra Foothill Laboratories (nutrients, BOD, and DOC in 2003; Jackson, CA), California Department of Fish and Game Water Pollution Control Laboratories (nutrients and BOD in 2004; Rancho Cordova, CA), and California Laboratory Services (DOC in 2004; Rancho Cordova, CA).

Prior to shredding, total Hg analyses were conducted on water, sediment, and plant samples collected from Dow Wetland station D1 (Figure 1b). Water samples were collected in plastic 500 ml acid washed bottles and sediment samples were collected using a 30 cm depth core sampler. Plant samples were collected by hand on April 23, 2004, using plastic gloves and stored in sealed plastic bags. Samples were digested with 30% HNO_3 (trace metal grade) and analyzed by cold vapor atomic absorption spectroscopy, using a Perkin Elmer 300 AA spectrophotometer. For all nutrient and Hg analyses, sample QA procedures included field and laboratory duplicates, field and laboratory blanks, laboratory matrix spikes and duplicates, and standard reference materials (Yee et al. 2004).

Statistical Analysis

For the 2003 sampling, a number of stations were relatively close to each other (Figure 1) creating the need to evaluate statistical independence for subsequent analyses. Spatial independence of separate sampling stations at each of the two sites (Dow Wetland and Lambert Slough) was ascertained by examining association in water quality parameters measured on the same dates. To limit pseudoreplication, results from separate stations were averaged when Pearson correlation coefficients were above 0.50 for any of the following parameters: BOD, DOC, OP, TKN, turbidity, or dissolved oxygen. Only comparisons having sample sizes of five or more paired samples were used. Once appropriate station partitioning was determined in 2003, it was possible to evaluate treatment effects for individual station categories, after averaging adjacent stations that were spatially correlated.

Time series data were analyzed for the effect of the mechanical shredding treatment, using a simple independent t-test, comparing samples collected prior to and after the perturbation. Variability of residuals was examined using Levene's test, and the Welch's t-test was performed when the residual variances were unequal among treatments (Stewart-Oaten et al. 1992). The project involved repeated sampling of individual stations, often with sample sizes (generally between 8 and 12 separate dates) insufficient to statistically model serial autocorrelation (Stewart-Oaten et al. 1992; Rasmussen et al. 2001). Serial autocorrelation of residuals was evaluated by examining autocorrelation and partial autocorrelation functions. If serial autocorrelation was present, and the direction of autocorrelation could cause changes in statistical significance (at $p < 0.05$), t-test results were not included.

For 2004, within-station differences in water quality over three sampling dates were examined using a one-way, repeated measures analysis of variance (ANOVA). All repeated measures F - tests were performed under the assumption of multivariate normality. Mauchly's Test of Sphericity was used to test for violations of the assumption of sphericity, and probability values adjusted for violations using the Huynh-Feldt epsilon (Von Ende 2001). Metrics were log transformed (metric + 1) to normalize the data and equalize variances.

The datalogging sonde dissolved oxygen and turbidity data exhibited strong daily and tidal patterns, and mean values were generated for each daily and tidal cycle, based on NOAA predicted tidal patterns for Antioch, CA (http://co-ops.nos.noaa.gov/tides04/tpred2.html). This resulted in sample sizes of 34 days or 66 tidal cycles for each parameter. Data exhibited significant serial autocorrelation (p <

0.05), which needed to be removed prior to evaluation of treatment effects (Rasmussen et al. 2001). Serial autocorrelation was accounted for by evaluating autoregressive (AR) and moving average (MA) models, selecting models based on: 1. successful removal of significant autocorrelation, 2. minimization of the Akaike Information Criterion (AIC) value, and 3. overall model parsimony (minimizing number of parameters) (Box et al. 1994). Serial autocorrelation was present for both daily and tidal results. The residual ACF values remained significant after applying combinations of first and second order AR and MA models to the tidally averaged data. In order to simplify the modeling and interpretation, analyses therefore focused on the daily averaged data, for which serial autocorrelation was readily removed with ARMA techniques. Turbidity data were log transformed to best approximate normal distribution and variance homoscedasticity. The residuals of the time series model were then examined for significant treatment effect using a t-test between samples collected before versus after shredding. All statistical analyses were performed using either SAS 9.1 or JMP 5.0.1 (SAS Institute Inc., Cary, NC), at significance level of 0.05.

Estimated Nutrient Mass Released by a Delta Wide Shredding Operation

To assess the potential ecosystem impact of wide-scale hyacinth treatment on the Delta, we estimated total hyacinth mass of carbon, nitrogen, phosphorus, and Hg. We compared these estimates to the total estimated nutrient mass in the Delta water column or annual Hg loading to the Delta (Foe 2003), in order to determine whether particular biogeochemical impacts of the treatment method could conceivably affect overall water quality in the Delta or other similar ecosystems. Nutrients were calculated in two fashions: as the total available mass in plant tissue and as the total mass transferred to the water column after mechanical shredding. Mass in plant tissue was calculated as the product of standing crop (kg dw m^{-2}), area covered by hyacinth (m^2), and tissue percent of the nutrient (Table 2). Tissue percent phosphorus data were not available from Delta hyacinth; peer-reviewed literature indicated tissue percent phosphorus for hyacinth collected in natural waters generally ranges between 0.1% and 0.55% (Pinto-Coelho and Greco 1999; Klumpp et al. 2002; Rommens et al. 2003; Xie et al. 2004; de Neiff et al. 2006). Based on previous observations of elevated Hg and other metals in hyacinth roots (Lenka et al. 1992; Klumpp et al. 2002; Riddle et al. 2002), tissue concentration of Hg was separately calculated for roots and stem base versus leaves and shoots, and combined using estimated percent dry mass from each tissue type (Table 2). Penfound (1948) indicates 60% of dry mass to be in leaves and shoots. Similarly, the percent of dry mass in leaves and shoots for hyacinth collected at a Delta site ranged from 50% to 75% (N = 7 biweekly collection dates, from April 28, 1997 to July 21, 1997; D. F. Spencer, *unpublished data*). Model input estimates varied widely in some cases (Table 2), so mass calculations spanned the full range of potential conditions, propagating through the minimum or maximum estimated values of each input parameter.

Nutrient release was based on water column concentration changes observed in this study, depth at shredding locations, and Delta wide hyacinth area coverage. Jassby and Cloern (2000) estimate aerial coverage to be 302 ha, based on median area chemically treated from 1983 to 1998. However, the California Department of Boating and Waterways (CDBW) indicate that aerial coverage has increased substantially over the past several years due to chemical treatment permitting difficulties, resulting in current aerial coverage estimates of 360 to 2200 ha (California Department of Boating and Waterways 2004; A. Morrill, CDBW, *pers. comm.*). Calculations therefore spanned the range from 300 to 2200 ha (Table 2).

Calculated hyacinth nutrient mass was compared to an estimated present mass in the water column. Minimum and maximum concentration increases in the water column resulting from shredding were obtained based on the range of pre-treatment versus post-treatment differences among the shredding treatment locations. Total Delta water column nutrient mass was estimated as the product of total water volume in the Delta and average water column concentration (Table 2). Total Delta water volume was obtained from N. E. Monsen (USGS, *pers. comm.*), using bathymetry in a Delta hydrodynamic model (Monsen 2001). Concentration data was obtained from the publicly accessible Bay Delta and Tributaries (BDAT) database (http://baydelta.ca.gov/). The California Department of Water Resources Estuary Monitoring Project and Municipal Water Quality Investigations programs, and the Port of Stockton's San Joaquin River Monitoring Program collect this water quality data. Data from 1995 through 2002 were assembled and averaged by station for dissolved organic carbon (2535 collections at 30 stations), total phosphorus (1487 collections at 38 stations), and TKN (1377 collections at 36 stations).

Results

General site conditions

All shredding locations had dense coverage of water hyacinth, but plant size was greater when shredding occurred in fall 2003 than spring 2004 (Table 1). In September 2003, plant standing crop was 1.8 kg m^{-2} dry weight in East Lambert Slough (station LE1;

SD = 0.4; N = 10) and 4.3 kg m^{-2} dry weight at Dow Wetland (station D1; SD = 1.3; N = 10). Plant dry weight to fresh weight ratio averaged 0.045 (SD = 0.018; N = 20). In general, each shredding event resulted in a significant reduction in plant standing crop, and number of live plants at a site (e.g., Figure 3c), but many viable fragments remained, and plant regrowth rate was elevated in shredded sites (Spencer et al. 2006).

Measured total Hg concentrations collected from station D1 were 1.25 $\mu g\ g^{-1}$ wet weight in hyacinth roots (SD = 0.36; N = 6 samples), 1.31 μg/g wet weight in hyacinth shoots (SD = 0.36; N = 6), and 0.27 $\mu g\ g^{-1}$ in sediments (SD = 0.075; N = 10). Of nine unfiltered water samples collected at the station, only three had detectable residues of Hg, with concentrations equaling 0.45, 0.52, and 0.77 $\mu g\ l^{-1}$. The remaining six water samples were below the detection limit for total Hg (i.e., less than 0.2 $\mu g\ l^{-1}$).

In 2003, measured water quality parameters generally were correlated between the two stations in West Lambert Slough (LW1, LW2), and also between the two stations in East Lambert Slough (LE1, LE2). For these pairwise comparisons, correlation coefficients (r) were positive and ranged between 0.59 and 0.99, with the exception of turbidity in East Lambert Slough, which exhibited $r = –0.56$. Water quality parameters were generally not correlated between West and East Lambert Slough stations for DO, DOC, BOD, OP, or TKN, with $–0.49 < r < 0.42$. Correlations were found between stations from the separate sloughs for TP ($0.62 < r < 0.98$) and turbidity ($-0.32 < r < 0.66$), though the associations were generally driven by a single data point. Based on the lack of independence between stations within East or West Lambert Slough, averages for each slough were used in the t-test, creating a total of three independent sampling locations for evaluation of shredding impacts in 2003 [East Lambert Slough (i.e., LE1), West Lambert Slough (LW1 and LW2, hereafter combined into LW), and the Dow Experimental Station (D1)]. Although East Lambert Slough stations were correlated, one of the two East Lambert Slough stations (LE2) was not included in t-tests because it was only sampled once prior to treatment. A t-test was also performed on the average of the Dow reference stations (i.e., DR) to ascertain whether water quality traits in the wetland changed significantly, independent of shredding.

Water chemistry trends

Overall, t-tests indicated significant changes in water quality after shredding at experimental stations treated with the AquaPlant Terminator (LW and D1) and the Amphibious Terminator (LE1) in 2003 (Table 3). Significant ($p < 0.05$) increases were observed in OP and TP for treatment stations in 2003 (Table 3, Figure 2). DO decreased significantly at station LE1, exhibited no significant trend at station LW, and increased significantly at station D1. In contrast to treatment stations, the Dow reference stations (R) did not exhibit changes in DO, OP, or TP (Table 3). Mean TKN concentrations were higher at LE1, LW, and D1 after shredding, though this pattern was only statistically significant at LW. Mean DOC concentrations were also higher at LE1 and LW after shredding, with a statistically significant increase at LE1. Following shredding, average DOC increased by 17.6 mg l^{-1} at LE1, and 10.7 mg l^{-1} at LW (Table 3). Conductance increased significantly at D1, R, and LW, suggesting a salinity influx during the experimental treatment (Table 3). Significant serial autocorrelation impeded statistical analysis for four parameter versus station combinations: turbidity at station R, TP in both Dow stations, and TKN in D1 (Table 3).

Graphical analysis suggested that nutrient increases and biochemical oxygen demand at the individual Lambert Slough stations (LE1, LE2, LW1, LW2) were sustained for several weeks after treatment in 2003 (Figure 2). The average TP increase (i.e., average concentration after treatment minus the average concentration before treatment) was high for all stations, equaling 0.38 mg l^{-1} at station LE1, 0.54 mg l^{-1} at LW, and 0.25 mg l^{-1} at D1. OP also increased: 0.15 mg l^{-1} at LE1, 0.30 mg l^{-1} at LW, and 0.063 mg l^{-1} at D1. Biochemical oxygen demand was extremely high after shredding in Lambert Slough, reaching maximum post-shredding concentrations of 76, 47, 48, and 54 mg l^{-1}, at LE1, LE2, LW1, and LW2, respectively (Figure 2d). Consequently, dissolved oxygen significantly declined after treatment at station LE1 (Table 3, Figure 2c).

In 2004, at Dow Wetland (stations D1, D2, D3, and D4), total nutrient concentrations increased immediately after Cookie Cutter treatment, and then declined to pretreatment conditions (Table 4). Repeated measures ANOVA indicated a significant change over the three sampling periods for TKN ($F_{2,6} = 5.947$, $p = 0.038$) and TP ($F_{2,6} = 6.312$, $p = 0.033$). The average observed nutrient increase was 0.37 mg l^{-1} for TP and 1.3 mg l^{-1} for TKN. Average concentrations were higher immediately after treatment for TSS and BOD (Table 4), although this trend was not statistically significant after Huynh-Feldt epsilon adjustment for sphericity violation (for TSS, $F_{1,3} = 8.313$, $p = 0.061$; for BOD, $F_{1,3} = 7.569$, $p = 0.071$). No trend was observed in dissolved nutrient concentrations (DOC, OP, or $NO_3 + NO_2$) or DO (Table 4).

DO and conductivity were collected from station D1 on an intermittent basis between April 2002 and February 2004. Over three dates in April and May of 2002, DO concentration declined (Figure 3); this coincided with an invasion of water hyacinth into the site (S. P. Andrews, personal observation). Mechanical shredding on September 22, 2003 resulted in a decline in the number of hyacinth plants present at station D1 (Figure 3c). DO generally increased over the course of 25 measurement dates between June 2003 and February 2004 (Figure 3a). There was a significant increase in DO after mechanical shredding occurred on September 22, 2003 (t-test, $p = 0.02$; no significant serial autocorrelation), with average concentrations increasing from 3.7 to 5.1 over that time period. Conductivity increased between June and November 2003, but declined sharply in February 2004 (Figure 3b), when DO remained relatively high (Figure 3a).

DO and turbidity were continuously monitored at station D1 from May 22 through June 25, 2004 (Figure 4). Significant serial autocorrelation was observed for daily averaged values of both DO and turbidity ($N = 34$). For DO, the model with the best fit, based on absence of residual autocorrelation, lowest AIC, and high R^2 (0.74) was an AR(1) model (i.e., containing a 1st order autoregressive term). For turbidity (log transformed), the best fit ($R^2 = 0.32$) was also achieved with an AR(1) model. No significant difference was observed between pre-treatment and post-treatment samples for the model residuals of either DO ($t = 1.24$; $p = 0.23$; 31 df) or turbidity ($t = -0.15$; $p = 0.89$; 31 df). For the raw data (Figure 4), DO declined and turbidity increased during the 24 hr immediately following the primary shredding event. Evaluation of the residuals of the AR(1) models confirmed the graphical results. On June 3, the DO residual was 1.46 SD below the mean (probability of selecting a random sample of this value; $p < 0.1$) and the turbidity residual was 2.79 SD above the mean ($p < 0.005$).

Estimated Nutrient Mass Released By a Delta Wide Shredding Operation

Using compiled data for the Delta, the average TP, TKN, and DOC concentrations were 0.19, 0.78, and 3.5 mg l^{-1}, respectively. These concentrations are comparable to Delta concentrations reported elsewhere (e.g., Jassby et al. 2002; Sobczak et al. 2002; Schemel et al. 2004). For carbon, nitrogen, and phosphorus, the range of estimated total mass present in water hyacinth tissue spanned the estimated current mass in the Delta water column (Table 5). For total phosphorus and total nitrogen, nutrient increases to the water column resulting from shredding were similar in magnitude to the current concentrations in the Delta water column (Table 2, Table 5). For dissolved organic carbon, the increase resulting from shredding at Lambert Slough stations (17.6 mg l^{-1} at LE1, and 10.7 mg l^{-1} at LW) was three to five times the Delta average water column concentration (3.5 mg l^{-1}). Nevertheless, based on these concentration increases resulting from the shredding experiments, the maximum calculated nutrient mass released due to Delta wide shredding operations would be only 3.1% to 9.6% of the mass present in the water column. Minimum calculated nutrient mass releases were extremely low, ranging from 0.1% to 0.4% of current water column nutrient concentrations (Table 5). The total Hg mass present in water hyacinth in the Delta in a given year was estimated to be between 7 and 242 kg. The higher estimate of this value is comparable to the estimated annual riverine Hg load to the Delta in 2000 (180 kg) and 2001 (99 kg) (Foe 2003).

Discussion

Our results indicated increases in water column nutrients, DOC, and BOD after mechanical shredding of water hyacinth. However, the extent and duration of these effects varied considerably among the different shredding operations; changes were greater at the irrigation ditch (Lambert Slough) than the tidal wetland (Dow Wetland), and were more apparent during the fall 2003 operations, when plants were larger.

The greater nutrient increases and oxygen depletion observed at Lambert Slough than Dow Wetland likely resulted from the substantial differences in water residence times between these two locations. Lambert Slough exhibits limited flow-through and weak tidal influence, with water exchange occurring via small drainage pipes on the west end of the Slough. In stagnant locations, such as Lambert Slough, shredding would result in decomposition of organic carbon and anoxia, leading to fish mortality (Killgore and Hoover 2001), and production of bioavailable methyl-Hg (Gilmour et al. 1992). In 2003, biochemical oxygen demand at Lambert Slough reached greater than 48 mg l^{-1}, which is about 10 to 20 times the level typically observed in the nearby San Joaquin River (Lehman et al. 2004), and is more similar to the BOD of treated sewage effluent (Moss 1988). The substantial oxygen demand at East Lambert Slough (stations LE1 and LE2) resulted in the site going completely anoxic for several weeks after shredding. On September 24, 2003, 16 days after shredding occurred, about 20 dead bluegill sunfish (*Lepomis macrochirus*) and one dead carp (*Cyprinus carpio*) were observed along the banks of the East Lambert Slough (B. Greenfield, personal observation). Presumably, the anoxic conditions resulted in this fish kill.

At Dow Wetland, DO concentrations declined during the water hyacinth invasion in spring of 2002, and then increased after hyacinth was shredded in fall of 2003. The Dow Wetland is directly off the mainstem San Joaquin River, experiencing one to two meter tide height variation, with the complete dewatering of many locations during low tides. Other studies have also demonstrated oxygen depletion associated with water hyacinth (Penfound and Earle 1948; Rommens et al. 2003; Perna and Burrows 2005), likely resulting from the dense floating vegetation impeding wind and tidal mixing (Madsen 1997; James et al. 2002). Additionally, phytoplankton and bacterial respiration increase, as a result of reduced light penetration, and increased organic material production and decomposition beneath the plants (Carignan and Neiff 1992; Battle and Mihuc 2000; Rommens et al. 2003). By breaking the barrier of floating vegetation, thereby allowing light penetration and wind and tide driven circulation, shredding can increase DO (James et al. 2002), and also increase available habitat for sensitive fish species, such as the Sacramento splittail and Chinook salmon (Moyle et al. 2004).

In spring of 2004, when hyacinth stands at Dow Wetland were chopped with the Cookie Cutter, water quality impacts were short lived. Although TKN and TP increased one hour after shredding, they returned to pretreatment conditions within three days. Continuous water quality monitoring at the D1 station indicated that a decline in DO and increase in turbidity only persisted for a single day. These findings suggest that spring treatments are likely to have fewer water quality impacts, presumably because the plants are smaller and much less dense early in the growing season (Penfound and Earle 1948; Bock 1969; Spencer and Ksander 2005).

Due to the difficulty obtaining simultaneous measurements at untreated reference stations, statistical treatment-control comparisons (e.g., BACI or related designs, Stewart-Oaten et al. 1992) could not be achieved in this study. Rather, a “weight of evidence” approach must be used to confirm that chemistry changes observed in this study resulted from shredding, rather than unrelated changes in background conditions. In this study, nutrient concentration increases were observed after four separate shredding events (Table 1), indicating that the pattern was robust to different environmental settings, dates, and shredding boats. A set of reference stations during one of the shredding operations (the Dow Wetland reference station, R) did not exhibit changes in nutrient concentrations. A conductivity increase was also observed during the Dow Wetland experiments (Figure 3), suggesting an influx of saline water from downstream within the Estuary, due to variations in tide strength and riverine inputs (reviewed in Kimmerer 2004). However, dissolved oxygen at one station (D1) remained at elevated post-treatment levels in spring of 2004, after conductivity returned to pretreatment levels (Figure 3). Since the primary exogenous nutrient sources to the Delta are upstream tributary inputs (Jassby and Cloern 2000), and total organic carbon generally decreases with increasing salinity (Murrell and Hollibaugh 2000), the influx of saline water may in fact account for the short duration of the nutrient increase and oxygen depletion during the spring 2004 shredding trial. In tidally influenced systems such as Dow Wetlands, local impacts of mechanical shredding are not likely to be a major management concern.

As observed previously, measured Hg concentrations in hyacinth were several times the concentrations in sediments, and several thousand-fold greater than total water column concentrations, suggesting bioconcentration of water column Hg (Lenka et al. 1992; Riddle et al. 2002). Mass estimate calculations indicated a relatively high Hg mass in Delta water hyacinth, compared to upstream loading. Furthermore, other studies indicate that high rates of bioavailable methyl-Hg production occur on hyacinth roots (Mauro et al. 2001), as well as possibly in anoxic sediments beneath hyacinth stands (Gilmour et al. 1992). Locations impacted by both Hg contamination and introduced aquatic plants include the Sacramento-San Joaquin River Delta (Bock 1969; Davis et al. 2003), the Florida Everglades (Duvall and Barron 2000; Pimentel et al. 2000), Lake Victoria (Ramlal et al. 2003; Albright et al. 2005), and many northern temperate lakes (e.g., Gilmour et al. 1992). Although mechanical harvesting may be an appropriate method for Hg remediation in highly contaminated locations (Riddle et al. 2002), allowing unchecked growth, or employing control methods that allow plant decay in the water column (e.g., herbicide application or mechanical shredding without removal), could augment Hg release into the water column and methylation. The relative impacts of different control methods on Hg cycling and bioavailability merits further research.

Although the shredding operation was associated with increases in water column DOC, TP, OP, and TKN at the monitoring stations, calculated effects to Delta-wide nutrient budgets were small. For example, despite the large biomass of plant material, we estimated that wide-scale shredding would increase water column organic carbon by only 0.4% to 9.6%. The low impact on overall nutrient budgets is due to the fact that the area of the Delta covered by hyacinth in a given year (i.e., 300 to 2200 ha) is only 1% to 10% of total Delta water surface area (26,000 ha). Similarly, Rommens et al. (2003) found low impact of water hyacinth on whole-lake nutrient levels, due to relatively low areal coverage. In the Delta, tidal action would be expected to dilute nutrient additions (see Lucas et al. 2002), with limited region-wide nutrient impacts.

Metazoan production in the Delta is believed to be primarily dependant on abundance of bioavailable (i.e., labile) organic carbon, such as that produced by phytoplankton, rather than total bulk carbon (Jassby and Cloern 2000; Müller-Solger et al. 2002; Sobczak et al. 2002). Bioavailability and quality of organic material produced by shredding were not measured in this study, but prior studies indicate relatively slow decay rates of hyacinth, compared to other macrophytes (reviewed in Battle and Mihuc 2000). It may be beneficial to evaluate whether the pool of organic carbon produced by shredding macrophytes would be readily utilized by Delta primary consumers. If so, shredding could be combined with other management actions to increase plankton and fish production, such as managed flooding of riparian areas, and minimizing fish entrainment mortality at water pumping plants (Moyle et al. 2004; Schemel et al. 2004). Nevertheless, for shredding to be an effective long-term management method for water hyacinth, the method must be improved to reduce regrowth of shredded plant material (Spencer et al. 2006).

Although these results did not indicate that large-scale shredding operations would substantially increase water column nutrients on a Delta-wide basis, shredding might have adverse impacts in localized areas. These include large inputs of DOC at culinary water canal intakes posing a risk of trihalomethane formation (Fujii et al. 1998; Brown 2003), as well as increased biochemical oxygen demand to localized anoxic zones. In particular, the Stockton Deepwater Shipping Canal is impaired due to DO loss (Lehman et al. 2004), and management activities there should focus on reducing oxygen demand.

A goal of this study was to employ the ecosystem experiment approach recommended by scientists in the CALFED Bay-Delta program and elsewhere (Kimmerer et al. 2005; Zedler 2005). This included a regional scale assessment of how a community-level management action (destruction of an introduced species) could affect ecosystem processes (nutrient cycling). Current management of aquatic plants in the

Delta focuses primarily on control efficiency for a single management objective (nuisance vegetation removal). Given the varied impacts of introduced aquatic species, expense of control programs currently underway (Pimentel et al. 2000), and complexity of the Delta and other managed estuaries (Lucas et al. 2002; Kimmerer 2004; Kimmerer et al. 2005), we believe that more efforts towards ecosystem-scale experiments and system integration are warranted in aquatic plant management.

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Table 1. Description of shredding and reference sites, including site conditions, treatment and monitoring dates, and statistical analyses applied. ANOVA = repeated measures Analysis of Variance.

Site (Stations)	Treatment	Treatment Dates	Site Conditions	Shredded Area (ha)	Monitoring Dates	Data Analysis
East Lambert Slough (LE1, LE2)	Amphibious Terminator	9/6, 9/8/2003	Dense 2' stem height	1.4	6/5 - 10/8/2003	t-test
West Lambert Slough (LW1, LW2) [a]	AquaPlant Terminator	9/19 - 9/21, 9/26 - 9/27/2003	Dense 3'-4.5' stem height	4.7	8/7 - 10/8/2003	t-test
Dow Wetland (D1)	AquaPlant Terminator	9/21 - 9/24/2003	Dense 4'-4.5' stem height	0.36	8/8 - 11/10/ 2003 [b]	t-test
Dow Wetland (R)	Average of 4 reference stations on Figure 1 (R1, R2, R3, and R4)	None (ref. for 2003)	Open water		8/8 - 10/7/2003	t-test
Dow Wetland (D1)	Cookie Cutter	6/3/2004	1' stem height	0.53	6/1, 6/3, 6/7/2004 [b]	ANOVA
Dow Wetland (D2)	Cookie Cutter	6/3/2004	1' stem height	0.24	6/1, 6/3, 6/7/2004	ANOVA

Dow Wetland (D3)	Cookie Cutter	6/3/2004	1' stem height	0.12	6/1, 6/3, 6/7/2004	ANOVA
Dow Wetland (D4)	Cookie Cutter	6/3/2004	1' stem height	0.45	6/1, 6/3, 6/7/2004	ANOVA

a. These two stations were not statistically independent for some parameters; therefore, results were averaged and described in the text as LW.

b. Additional data were collected at station D1: DO and conductivity between 2002 and 2/2004; Hg in water, sediment, tissues; and continuous field chemistry measurement from 5/22 - 6/25/2004

Table 2. Range of values used for estimating total mass of Hg, carbon, nitrogen, and phosphorus in Delta water hyacinth, mass released by a large-scale shredding operation, and mass of nutrients currently in the Delta water column.

E. Crassipes Parameter	Value	Reference
Standing crop (biomass/unit area) [a]	1.8 - 4.3 kg m^{-2}	This study
Coverage in Delta	300 - 2200 ha	Jassby and Cloern (2000); CDBW (2004); A. Morrill, CDBW, *pers. comm.*
Tissue percent nitrogen	1.5 - 2.5%	Spencer and Ksander (2004)
Tissue percent carbon	37%	Spencer and Ksander (2004)
Tissue percent phosphorus	0.1 – 0.55%	Klumpp et al. (2002); Rommens et al. (2003); Xie et al. (2004); de Neiff et al. (2006)
Tissue Hg concentration [a]		
Shoots and leaves	1.31 mg kg^{-1}	This study
Stem base and roots	1.25 - 4.44 mg kg^{-1}	This study; Riddle et al. (2002)
Percent of hyacinth tissue dry mass in		
Shoots and leaves	50 – 75%	Penfound (1948); D. F. Spencer, *Unpublished data*
Stem base and roots	25 – 50%	Penfound (1948); D. F. Spencer, *Unpublished data*
Water depth [b]	0.5 – 1 m	This study
Delta Water Quality Parameter	Value	Reference
Total water volume	1.2 x 10^{9} m^{3}	N. E. Monsen, USGS, *pers. comm.* based on Monsen (2001); Kimmerer (2004)

Total Kjeldahl nitrogen	0.78 mg l^{-1}	This study, using BDAT database
Dissolved organic carbon	3.45 mg l^{-1}	This study, using BDAT database
Total phosphorus	0.19 mg l^{-1}	This study, using BDAT database
Estimated total annual Hg input to Delta	99 – 180 kg	Foe (2003)

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a. Dry weight basis.

b. Used to convert post-shredding nutrient concentration increases to mass release estimates

Table 3. Water chemistry results in 2003. Average concentrations are presented for each of four stations before and after shredding. DO = dissolved oxygen. TP = total phosphorus. OP = dissolved orthoreactive phosphorus. TKN = total Kjeldahl nitrogen. DOC = dissolved organic carbon. SD = standard deviation. N = number of sampling dates. The last column presents results of the statistical analysis for differences before versus after shredding. Boldfaced results indicate a significant difference ($p < 0.05$).

Parameter (units)	Station	Value Before Mean (SD, N)	Value After Mean (SD, N)	Transform	2 tail t-test $p > t$
DO (mg l^{-1})	LE1	1.49 (1.25, 5)	0.07 (0.019, 6)	Log	**0.008**
DO	LW [a]	1.32 (0.78, 8)	0.86 (1.07, 4)	Log	0.16
DO	D1	4.08 (0.75, 4)	5.10 (0.42, 6)	Log	**0.012**
DO	R	6.77 (2.23, 5)	6.00 (0.84, 4)	Log	0.60 [b]
Turbidity (NTU)	LE1	7.5 (3.7, 5)	13.0 (12.7, 6)	Sqrt	0.93
Turbidity	LW [a]	24.3 (20.2, 8)	133 (240, 4)	1/Sqrt	0.96
Turbidity	D1	33.0 (8.6, 4)	25.1 (25.5, 6)	Log	0.17 [c]
Turbidity	R	11.8 (5.8, 5)	6.2 (1.2, 4)	NA [d]	NA [d]
TP (mg l^{-1})	LE1	0.10 (0.05, 5)	0.48 (0.29, 4)	Log	**0.009**
TP	LW [a]	0.10 (0.05, 6)	0.64 (0.18, 4)	1/Sqrt	**< 0.001**
TP	D1	0.13 (0.04, 4)	0.38 (0.41, 6)	NA [d]	NA [d]
TP	R	0.07 (0.01, 5)	0.07 (0.02, 4)	NA [d]	NA [d]
OP (mg l^{-1})	LE1	0.012 (0.009, 2)	0.16 (0.064, 4)	None	**0.036**

OP	LW [a]	0.017 (0.010, 4)	0.32 (0.28, 4)	1/Sqrt	**0.006**
OP	D1	0.026 (0.008, 3)	0.089 (0.056 6)	Log	**0.012**
OP	R	0.042 (0.010, 3)	0.043 (0.010, 4)	None	0.85
TKN (mg l^{-1})	LE1	0.76 (0.36, 4)	2.13 (1.44, 4)	Log	0.072
TKN	LW [a]	0.61 (0.23, 5)	1.70 (1.16, 4)	1/Sqrt	**< 0.02**
TKN	D1	0.57 (0.23, 4)	1.15 (1.25, 6)	NA [d]	NA [d]
TKN	R	ND (NA, 4)	ND (NA, 4)	NA	NA
DOC (mg l^{-1})	LE1	3.9 (NA, 1)	21.5 (4.4, 4)	None	**<0.005**
DOC	LW [a]	3.9 (0.6, 2)	14.6 (6.7, 4)	Arcsin(Sqrt)	0.08
DOC	D1	5.0 (0.3, 2)	4.9 (0.6, 6)	1/X	0.71
DOC	R	3.7 (0.8, 2)	3.7 (0.8, 4)	Log	0.99
Conductance (umhos/S)	LE1	254 (72, 5)	281 (44, 6)	None	0.47
Conductance	LW [a]	158 (12, 8)	249 (35, 4)	1/X	**< 0.0001**
Conductance	D1	1219 (530, 4)	3250 (244, 6)	Arcsin(Sqrt)	**< 0.0001**
Conductance	R	1226 (531, 5)	3339 (198, 4)	Arcsin(Sqrt)	**< 0.001**

a. Average of values from LW1 and LW2 (Figure 1)

b. Variances may be unequal (Levene's test $p < 0.10$); Used Welch ANOVA assuming unequal variances

c. Errors not normally distributed. Used Kruskal-Wallis ranked sum evaluation (Wilcoxon)

d. Result not available due to serial autocorrelation of t-test residuals

ND = all samples were below the detection limit (0.5 mg l^{-1})

Table 4. Water chemistry results for four stations monitored at Dow Wetland during Cookie Cutter treatment in 2004 (stations D1, D2, D3, and D4; Figure 1). Average concentrations are presented for all of the stations before shredding, immediately after shredding, and four days after shredding. SD = standard deviation. Boldfaced results indicate a significant change in that chemistry parameter over the three sampling dates ($p < 0.05$).

Parameter (mg l^{-1})	Before Average (SD)	After One Hour Average (SD)	After Four Days Average (SD)
Biochemical oxygen demand (BOD)	1.5 (0)	5.3 (3)	1.5 (0)
Dissolved organic carbon (DOC)	2.9 (0.6)	2.5 (0.3)	1.7 (0.8)
Dissolved oxygen (DO)	5.2 (1.2)	4.8 (2.6)	4.4 (1.3)
Total phosphorus (TP)	**0.12 (0.05)**	**0.49 (0.31)**	**0.09 (0.02)**
Orthoreactive phosphate (OP)	0.06 (0.03)	0.02 (0.03)	0.04 (0)
Total Kjeldahl nitrogen (TKN)	**0.46 (0.14)**	**1.76 (1.09)**	**0.43 (0.06)**
Dissolved nitrates ($NO_3 + NO_2$)	0.22 (0.09)	0.26 (0.13)	0.25 (0.09)
Total suspended solids (TSS)	22 (14)	321 (259)	13 (9)

Table 5. Potential impact of large-scale shredding operation on Delta nutrient budget, based on study results and compiled data (Tables 2 and 3). Total hyacinth biomass = total biomass of water hyacinth currently present in the Delta in the form of organic carbon, nitrogen, or phosphorus (range of values in metric tons, T). Shredding material released = total amount of shredded material released into the water column, based on study results. Current mass in water column = total mass of nutrients in entire Delta water column; carbon as dissolved organic carbon and nitrogen as total Kjeldahl nitrogen. Shredding addition = range of percent increase in nutrient as a result of Delta wide shredding operation.

Estimate (units)	Carbon	Nitrogen	Phosphorus
Total hyacinth mass (T)	2,000 – 35,000	81 – 2365	5.4 – 520
Water column increase after shredding (mg l^{-1})	$11_a – 18_b$	$0.6_c – 1.3_a$	$0.25_c – 0.54_b$
Shredding material released (T)	17 – 396	0.9 – 29	0.4 – 12
Current mass in water column (T)	4,100	940	230
Shredding addition (% range)	0.4 - 9.6%	0.1 - 3.1%	0.2 - 5.2%

a. Based on LE1.

b. Based on LW.

c. Based on D1.

Figure captions

Fig. 1. Study site and sampling station locations. Dark gray = open water; light gray = floating vegetation. a. Location of Dow Wetland and Lambert Slough Sites. b. Sampling stations on Dow Wetland. c. Sampling stations on Lambert Slough.

Fig. 2. Results of the water chemistry monitoring over time for Lambert Slough monitoring stations in 2003. The dashed vertical line indicates the date of Amphibious Terminator shredding in East Lambert Slough. The two East Lambert Slough stations are LE1 (▽) and LE2 (○). The solid vertical line indicates the date of AquaPlant Terminator shredding in West Lambert Slough. The two West Lambert Slough stations are LW1 (♦) and LW2 (■). Note log axes, plots b and c.

Fig. 3. Dissolved oxygen, specific conductance, and plant abundance monitoring over time at the Dow Wetland 2003 mechanical shredding station (D1). The black rectangle before the scale break indicates a time period when water hyacinth invaded the station area (April – May 2002). The black vertical line indicates when mechanical shredding was conducted on the station with that AquaPlant Terminator.

Fig. 4. Continuous monitoring of the Dow Wetland mechanical shredding station (D1) during a one month period in 2004. The vertical hash lines indicate points when mechanical shredding was conducted on the site using the Cookie Cutter. Inset plots present on an expanded time scale the 48 hr period when the shredding operation was conducted, with the circled area referred to in the text. a. Dissolved oxygen (mg l^{-1}). b. Turbidity (NTU).

Figure 1.

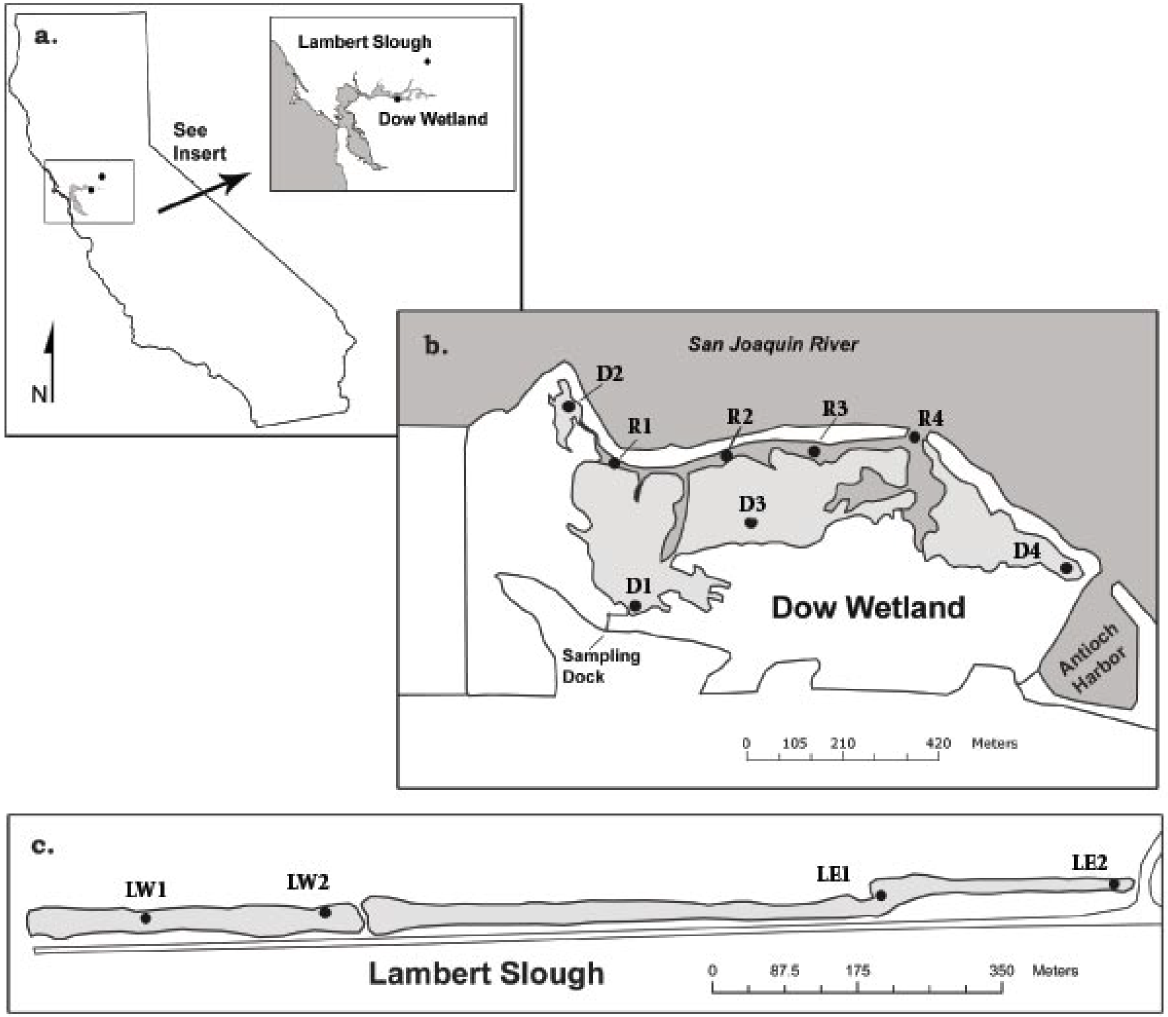

a.
Lambert Slough
Dow Wetland
See Insert
N
b.
San Joaquin River
D2
R1
R2
R3
R4
D3
D4
D1
Dow Wetland
Sampling Dock
Antioch Harbor
0 105 210 420 Meters
c.
LW1
LW2
LE1
LE2
Lambert Slough
0 87.5 175 350 Meters

Figure 2.

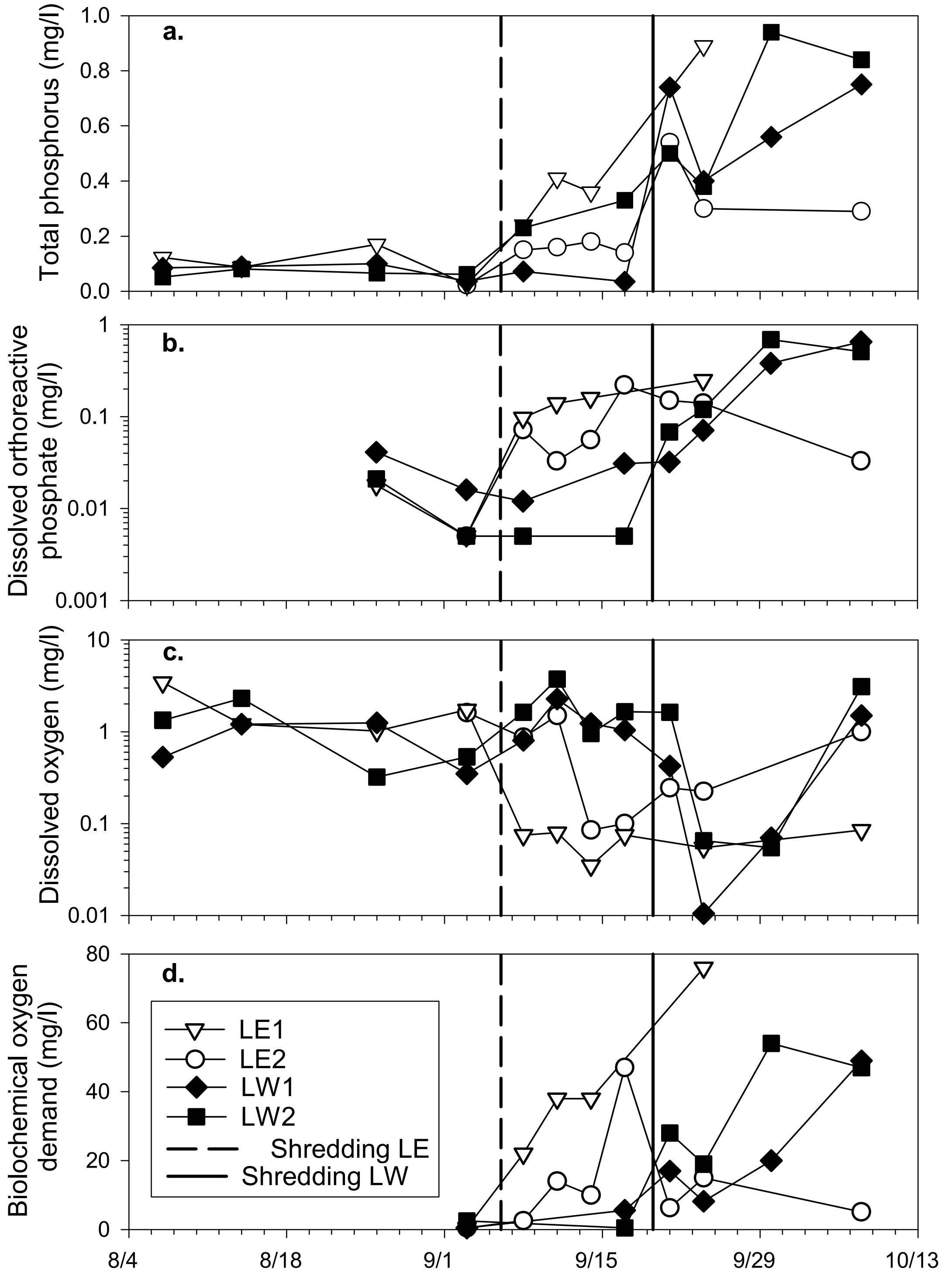

a.
b.
c.
d.
Total phosphorus (mg/l)
1.0
0.8
0.6
0.4
0.2
0.0
Dissolved orthoreactive phosphate (mg/l)
1
0.1
0.01
0.001
Dissolved oxygen (mg/l)
10
1
0.1
0.01
Biochemical oxygen demand (mg/l)
80
60
40
20
0
8/4
8/18
9/1
9/15
9/29
10/13
LE1
LE2
LW1
LW2
Shredding LE
Shredding LW

Figure 3.

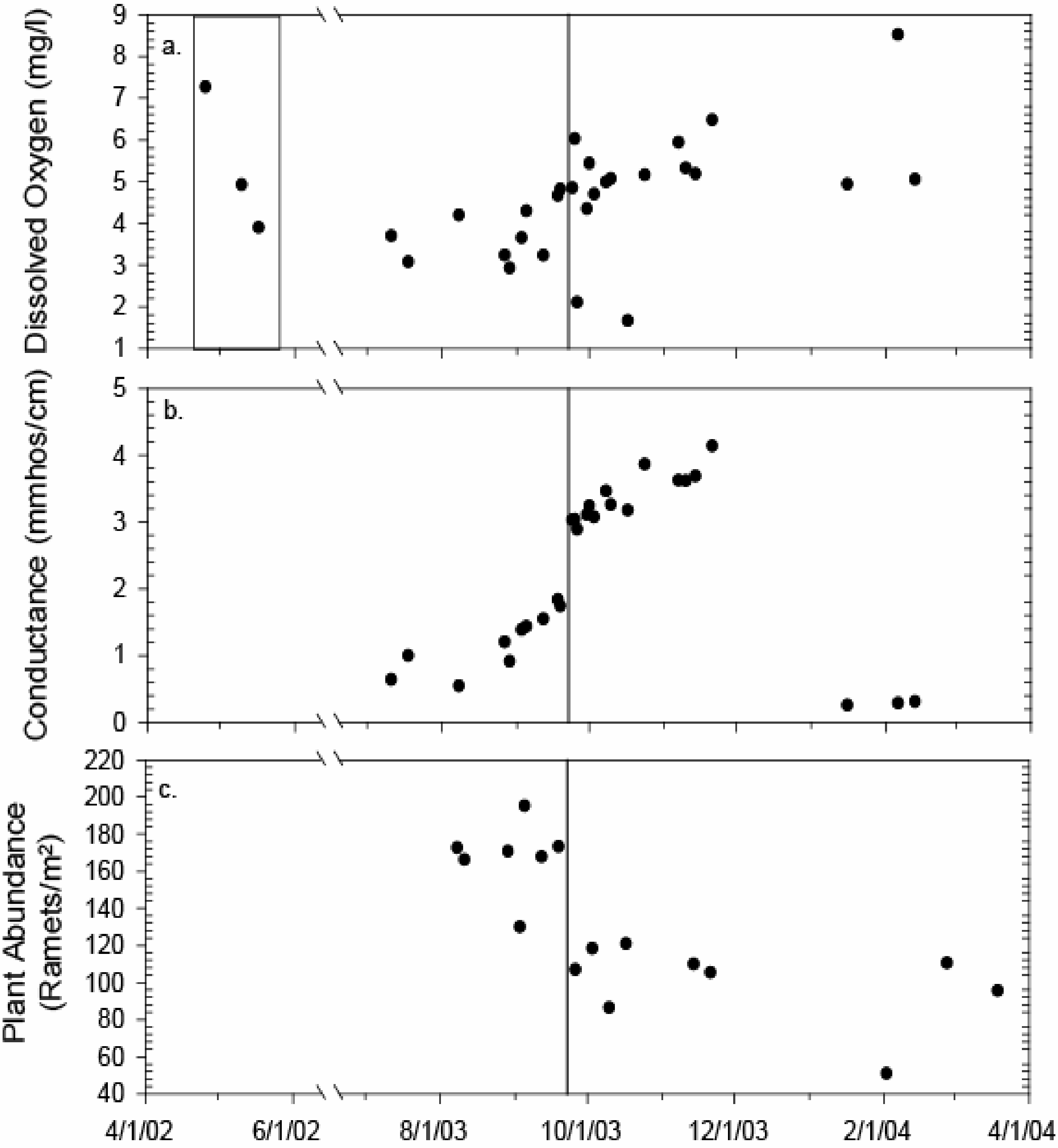
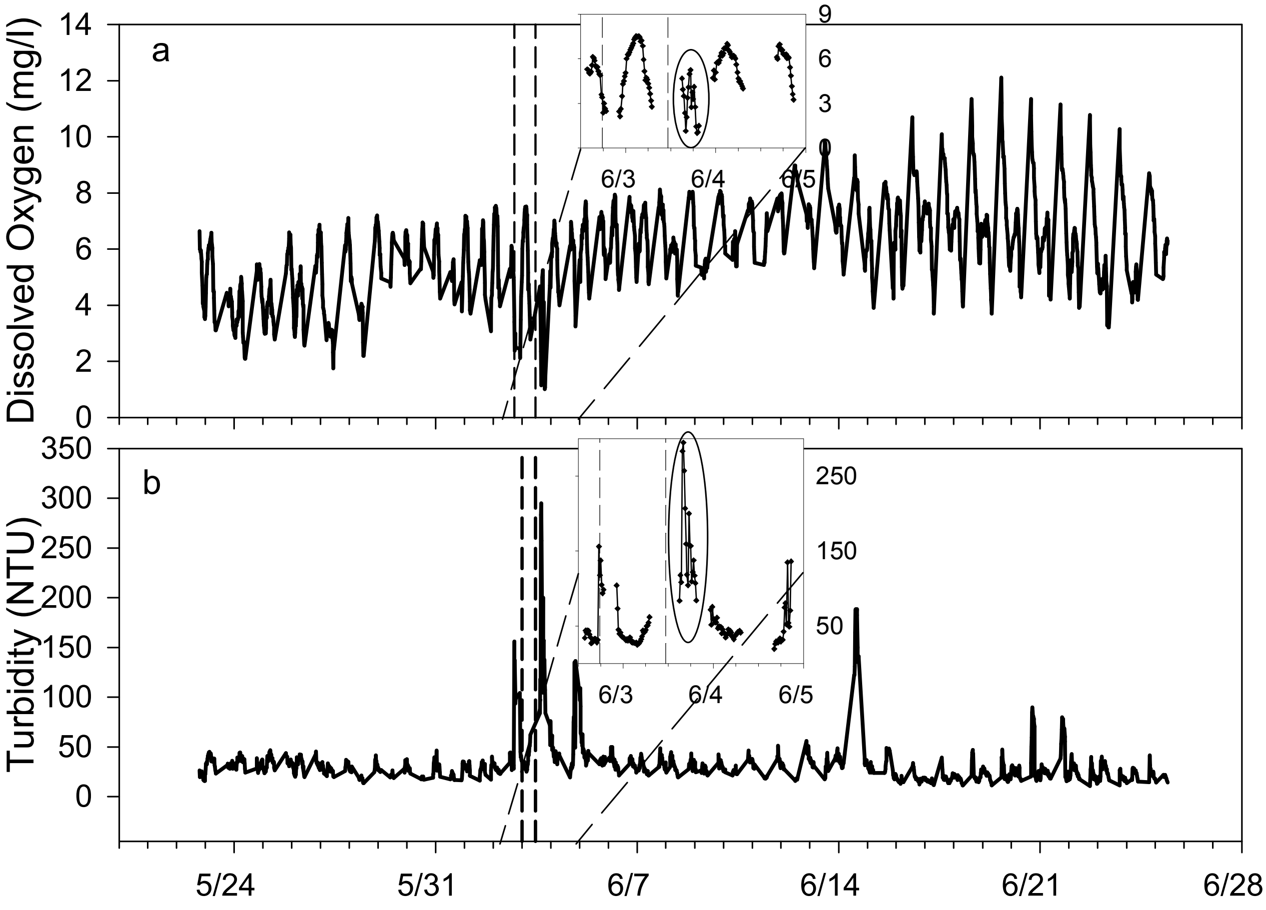

a.
Dissolved Oxygen (mg/l)
1
2
3
4
5
6
7
8
9
b.
Conductance (mmhos/cm)
0
1
2
3
4
5
c.
Plant Abundance (Ramets/m^2)
40
60
80
100
120
140
160
180
200
220
4/1/02
6/1/02
8/1/03
10/1/03
12/1/03
2/1/04
4/1/04

Figure 4


a
Dissolved Oxygen (mg/l)
14
12
10
8
6
4
2
0
9
6
3
0
6/3
6/4
6/5
b
Turbidity (NTU)
350
300
250
200
150
100
50
0
250
150
50
6/3
6/4
6/5
5/24
5/31
6/7
6/14
6/21
6/28