Use of Hyperspectral Remote Sensing to Evaluate Efficacy of Aquatic Plant Management

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Invasive aquatic weeds negatively affect biodiversity, fluvial dynamics, water quality, and water storage and conveyance for a variety of human resource demands. In California's Sacramento–San Joaquin River Delta, one submersed species—Brazilian egeria—and one floating species—waterhyacinth—are actively managed to maintain navigable waterways. We monitored the spatial and temporal dynamics of these species and their communities in the Sacramento-San Joaquin River Delta using airborne hyperspectral data and assessed the effect of herbicide treatments used to manage these species from 2003 to 2007. Each year, submersed aquatic plant species occupied about 12% of the surface area of the Delta in early summer and floating invasive plant species occupied 2 to 3%. Since 2003, the coverage of submersed aquatic plants expanded about 500 ha, whereas the coverage of waterhyacinth was reduced. Although local treatments have reduced the coverage of submersed aquatic plants, Delta-wide cover has not been significantly reduced. Locally, multiyear treatments could decrease submersed aquatic plants spread, given that no residual plants outside the treated area were present. In contrast, the spread of waterhyacinth either has been constant or has decreased over time. These results show that (1) the objectives of the *Egeria densa* Control Program (EDCP) have been hindered until 2007 by restrictions imposed on the timing of herbicide applications; (2) submersed aquatic plants appeared to function as ecosystem engineers by enabling spread to adjacent areas typically subject to scouring action; (3) repeated herbicide treatment of waterhyacinth has resulted in control of the spread of this species, which also appears to have facilitated the spread of waterprimrose and floating pennywort. These results suggest that management of the Delta aquatic macrophytes may benefit by an ecosystem-level implementation of an Integrated Delta Vegetation Management and Monitoring Program, rather than targeting only two problematic species.

Nomenclature: Brazilian egeria, *Egeria densa* Planch.; floating pennywort, *Hydrocotyle ranunculoides* L. f.; waterhyacinth, *Eichhornia crassipes* (Mart.) Solms.; waterprimrose, *Ludwigia* L. spp.

Key words: Change detection, cover, exotic species, hyperspectral remote sensing, weed management.

Invasive aquatic plants compete with and alter native biodiversity, rework natural river dynamics, alter water quality, and compete with water demands for domestic and agricultural uses, recreation, and aquaculture (Anderson 1990, 2003). Invasive species are often successful because of their ability to capture resources (nutrients, light) through morphological and physiological plasticity, higher tolerance for their physical environment than native species, and being unrestricted by natural competitors and predators (Rejmánek 2000, Shea and Chesson 2002). This unrestricted spread may alter community dynamics in various ways: changes in plant growth rates (Center and Spencer 1981), amount and location of plant biomass (Callaway and Aschehoug 2000, Schierenbeck et al. 1994), density of vegetation patches (French and Chambers 1996) and their spatial complexity (Penfound and Earle 1948), shading by dense canopies (Eiswerth and Johnson 2002), build up of plant detritus (Lehman 1998), changed dissolved oxygen levels (Viaroli et al. 1996), and increased rates of evapotranspiration (Gordon 1998, Toft 2000, Toft et al. 2003). Alterations in community dynamics are often

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Interpretive Summary

The Sacramento–San Joaquin Delta provides a potable water source for more than 22 million Californians and irrigation water for multibillion dollar crop-production systems. Two nonnative aquatic weeds, waterhyacinth and Brazilian egeria, greatly impair the transport, quality, and uses of this vital resource. Multiyear aerial imagery, coupled with digital analysis of reflectance, has helped document the extent of aquatic weed infestations and the effectiveness of state-run programs aimed at managing these two invasive aquatic weeds. Delta-wide remote sensing showed that waterhyacinth populations have been reduced over the past several years through the use of foliar herbicide applications and that Brazilian egeria continues to expand, except where spring applications of a systemic herbicide, fluridone, were made in a 3,000 acre area called Frank's Tract. This remote-sensing method will help provide large-scale, long-term monitoring information on the location and cover of submersed, floating, and shoreline plants and, thus, can be an essential part of overall Delta restoration efforts.

irreversible and efforts have been made to predict the direction of change and its potential effects (Lodge et al. 2006, Zavaleta et al. 2001). Increased frequency of secondary effects because of increasing numbers of interacting invasive species is expected to occur (Zavaleta et al. 2001). Understanding these effects may inform us about the restoration potential of the system; however, restoration actions may not return the system to the desired condition either because the system lacks this potential, has achieved an alternative stable state (Beisner et al. 2003, Suding et al. 2004), or is subject to further invasions by release of other nonnative species already present in the system. Better understanding of these dynamics is particularly important because conservative estimates of costs of management of aquatic invasive species are US$110 million annually (Eiswerth and Johnson 2002, Pimentel et al. 2000, Pimentel et al. 2005). Economic impacts caused by aquatic weeds alone have been estimated to range from $1 to 3 billion when all the secondary and tertiary costs are included (Rockwell 2003).

Successful management of invasive weeds can best be achieved through an understanding of their ecological requirements (Rejmánek 2000) and through monitoring changes in community composition and dynamics. Large-scale monitoring programs that can assess changes in community composition and dynamics also facilitate adaptive management. However, data sets suitable for the analysis of spatial and temporal patterns of aquatic invasions are rare (Cohen and Carlton 1998). Remote sensing can overcome these limitations. By combining high spatial resolution and spectral resolution capabilities with broadscale sampling, airborne hyperspectral data are valuable tools for producing accurate maps of the distribution of invasive species over large spatial scales in a range of ecosystems (Hirano et al. 2003, Peñuelas et al. 1993, Schmidt and Skidmore 2003, Silvestri et al. 2003, Underwood et al. 2003, Ustin et al. 2002). Furthermore, remote-sensing products can be used in several ways: (1) to predict invasive species spread by detecting nascent patches at high spatial resolution (e.g., using pixels $<$ 5 by 5 m), (2) to identify the habitats at risk (Lodge et al. 2006), (3) to improve the accuracy of predictive spatial models of invasion (Bradley and Mustard 2006), and (4) to evaluate the effects of treatments over large spatial and temporal scales allowing for true adaptive management. Furthermore, recent developments have shown the potential of hyperspectral remote sensing to detect submersed aquatic plants (Hestir et al. 2008). However, the quality of the products is dependent on the timing of imagery acquisition, water quality and species depth in the water column, and, in some cases, a species-level detection has yet to be achieved (for example see Hestir et al. 2008).

Brazilian egeria (*Egeria densa* Planch.) was first reported in California's Sacramento–San Joaquin River Delta in 1946 (Light et al. 2005) as a release from aquaria, which rapidly expanded its distribution in the 1980s (Jassby and Cloern 2000). The primary productivity of Brazilian egeria in the Delta has been estimated as 7.4 tons C/day (Jassby and Cloern 2000), which granted the species a rank of A-2—Most Invasive Wildland Pests—in California by the California Invasive Plant Council (Cal-IPC 2003), thus supporting immediate action. The control of Brazilian egeria was then assigned through legislative action to the California Department of Boating and Waterways (CDBW) through the *Egeria densa* Control Program [EDCP]; CDBW 2005a). One of the EDCP management prescriptions includes control of the species by herbicide application in selected areas throughout the Delta. EDCP has treated 35 selected sites (total area, 756.7 ha; 3% of the Delta waterways) in the Delta since 2001, and through 2006, the maximum treated area in any given year was 387.8 ha (2% of the Delta waterways; CDBW 2005a). These treatments included the systemic herbicide fluridone and the contact herbicide diquat, with repeated applications to achieve the recommended contact time and concentrations. However, until 2007, optimal spring (April to May) application of fluridone was prohibited because of National Oceanic and Atmospheric Administration (NOAA) concerns for the endangered fish reproductive season. This restriction resulted in delaying applications of fluridone until July each year through 2006. However the EDCP is a single-species target control program, and the herbicides used (primarily fluridone) differentially affect other submersed species. For example, Eurasian watermilfoil (*Myriophyllum spicatum* L.) is much more sensitive to fluridone than is Brazilian egeria. Sprecher et al. (1998) showed that Eurasian watermilfoil decreased concentration of carotenes and subsequently increased phytoenes in much lower fluridone concentrations (1 μg/L) than Brazilian egeria (5 μg/L). Eurasian watermilfoil can be controlled with fluridone at 5 to 6 parts per billion (ppb) and a contact time of 5 to 6 wk (Getsinger and Madsen 2002).

Waterhyacinth [*Eichhornia crassipes* (Mart.) Solms] is considered one of the most harmful nonindigenous species in the United States, being included within the 100 worst invasive species (OTA 1993). Control programs date back to 1899 (OTA 1993). It was first detected in the Delta in 1904 and was presumed to have been intentionally introduced by horticulturists (Cohen and Carlton 1998) or possibly from garden escape (Light et al. 2005); its primary productivity in the Delta has been estimated as 4.9 tons C/day (Jassby and Cloern 2000). The control of waterhyacinth has been directed and implemented by CDBW, since 1983, through a targeted-species control program (Waterhyacinth Control Program [WHCP]; CDBW 2005b). The WHCP uses the systemic foliar-applied herbicides 2,4-D and the glyphosate application in waterhyacinth nursery sites (CDBW 2005b). A cumulative area of 10,360 ha (25,600 ac) has been treated since 1988; typically around 500 to 1,000 ha are treated annually.

These two single-target weed management programs provide a unique setting in which to study invasion biology in a managed system, through allying the Sacramento–San Joaquin River Delta invasibility history with its environmental heterogeneity and an ongoing in situ field experiment due to active management. We used these ongoing treatments to perform a spatial and temporal analysis of the effect of treatment prescriptions at local and Delta-wide spatial scales using a temporal series of hyperspectral remote-sensing products. Our study is unique for several reasons: (1) the spatial extent is one of the largest areas continuously sampled in the world and has been analyzed for invasive species using a consistent method and good accuracy (> 80%); (2) the temporal extent, which is one of a few examples with a temporal frame matching the timing of management activities; and (3) it addresses aquatic species, one of the most challenging communities to be assessed with these remote-sensing methods. Specifically, we addressed the following questions: (1) What are the dynamics in the spatial distribution of submersed, aquatic plant communities within and across treatment years? (2) What are the effects of posttreatment, residual plant material on subsequent regrowth? (3) What is the effect of discontinuing the repeated herbicide treatments? (4) Is there an interaction between control and the dynamics of the aquatic and floating species? and (5) How is single-species management affecting other species?

Study System

The greater San Francisco Estuary, which includes the Delta, has been identified as the gateway to approximately 25% of the invasions in the United States (Lodge et al. 2006). It has been described as one of the most invaded estuaries in the world in terms of the number of species ($n = 234$), the individuals, and the biomass, with an accelerating invasion rate (Cohen and Carlton 1998). These high rates of invasion are mainly due to its geographical location (Lodge et al. 2006), its depauperate biota (Cohen and Carlton 1998), its high frequency of invasion vectors (Light et al. 2005), and its high rates of disturbance (human activities and the mixing of tidal water and freshwater, which occurs twice a day).

The Sacramento–San Joaquin River Delta covers 2,500 km^2 and drains approximately 160,000 km^2 of California, including the San Joaquin, Sacramento, Mokelumne, Cosumnes, and Old rivers, and its proximity to the San Francisco Bay creates a unique system with strong tidal and marine influences. The dominant land cover in the Delta region is agriculture, with annual grasslands and oak woodlands in the foothills (Figure 1). In some areas, prime farmland has been replaced by suburbs, and several urban areas have developed and expanded along the margins of the Delta region.

Expanding anthropogenic activities in and around the Delta have increased water diversion (via dams and levee systems) and recreation, which has resulted in the loss of most of the original California bulrush [*Schoenoplectus californicus* (C.A. Mey.) Palla] marsh (Jassby and Cloern 2000). This has irreversibly changed the seasonal pattern of river dynamics and flows (Golet et al. 2003) and has increased toxic contaminants (Jassby and Cloern 2000, Pereira et al. 1996, Werner et al. 2000) and exotic plant and animal species (Alpine and Cloern 1992, Moyle and Light 1996, Pereira et al. 1996, Underwood et al. 2006).

The exotic species have greatly decreased the distribution and abundance of the native plant community (Anderson 2003, Golet et al. 2003, Toft 2000, Toft et al. 2003). Today, the aquatic plant community of the Delta is composed of exotic and native, submersed, floating, and emergent plant species. Coontail (*Ceratophyllum demersum* L.), sago pondweed [*Stuckenia pectinata* (L.) Boerner], American pondweed (*Potamogeton nodosus* Poir), and common elodea (*Elodea canadensis* Michx.) are native, submersed species that experience increasing competition from nonnative plants, including Brazilian egeria, Eurasian watermilfoil, fanwort (*Cabomba caroliniana* Gray), and curlyleaf pondweed (*Potamogeton crispus* L.). The emergent (shoreline/levee) plant community includes natives species, such as floating pennywort, California bulrush, common cattail (*Typha latifolia* L.), and some small stands of common reed [*Phragmites australis* (Cav.) Trin. ex Steud.]. Native and some nonnative waterprimrose species (*Ludwigia* spp.), as well as other nonnative species, including giant reed (*Arundo donax* L.), perennial pepperweed (*Lepidium latifolium* L.), and Himalaya blackberry (*Rubus armeniacus* Focke), also occur along levee banks. Floating species

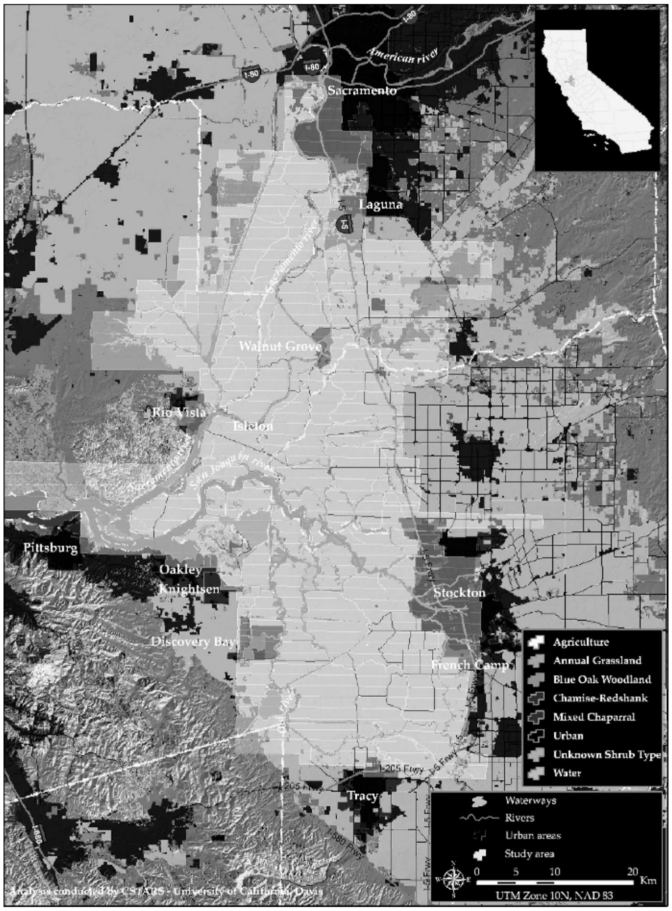

Figure 1. Map of the Sacramento–San Joaquin River Delta and its land cover and administrative boundaries in California. The land cover map includes a 100-m resolution statewide multisource land cover data following the classes defined in the California Wildlife–Habitat Relationships (CWHR; current version 02_2 obtained at http://frap.cdf.ca.gov/data/frapgisdata/select.asp).

include native mosquito ferns (*Azolla* spp.), duckweeds (*Lemna* spp.), and nonnative waterhyacinth.

CDBW actively manages the 2,139 km^2 of waterways in the Delta, divided into 226 CDBW management units. The EDCP has treated 757 ha (3% of the Delta waterways) in the Delta since 2001. The program treated 159, 239, 386, and 388 ha in 2003, 2004, 2005, and 2006, respectively (CDBW 2005a). In 2007, the management program was permitted to apply fluridone herbicide to a large (1,400 ha), inundated island (Franks Tract) in early spring. Because of the significant differences between timing, efficacy, and application regimes between 2007 and the previous years, we initially focused on management from 2003 to 2006. We then compared the 2007 and 2006 distribution to determine the effect of the early spring applications of fluridone. Application rates for control of Brazilian egeria were targeted initially (first week) at about 50 ppb, with final residues, thereafter, targeted at about 10 ppb for 5 to 6 wk (CDBW 2008). The WHCP treatment prescription includes treatment of all waterhyacinth nurseries, and a total of 10,360 ha have been treated since 1988.

Materials and Methods

Mapping Species Distributions. We used high spatial and spectral resolution airborne imagery to create distribution maps of submersed aquatic, floating, and emergent plant species in the Delta region. Hyperspectral imagery[1] (15 to 20 nm spectral resolution) was collected at 3-m ground resolution in June to July 2003, 2004, 2005, 2006, and 2007 and in October 2005. The image data acquired in 2003 and October 2005 were limited to the central Delta (381 km^2), which contained most of CDBW's treatment sites. All other image dates were for the entire Delta (3,183 km^2). Selecting the best remote-sensing approach for our analysis represented a trade-off between applying a single consistent method for the entire Delta vs. local-scale analyses with accuracy reflecting local environmental conditions. Because we aimed to produce a repeatable method applied to the full imagery, we adopted the first approach. We used a decision-tree approach, establishing thresholds using various spectral indices in a hierarchical framework to produce distribution maps for submersed aquatic plants, floating plants, and emergent communities for each date of the imagery. The decision tree consisted of various steps: (1) select pixels that were within Delta waterways; (2) select pixels with high fractions of vegetation to discriminate between water and emergent and submersed plants; (3) for submersed plants, select pixels with chlorophyll-absorption features to discriminate algae, turbid water, and submersed plants; (4) for emergent plants, select pixels with high fractions of target species (waterhyacinth), as determined by linear spectral unmixing and spectral angle analysis. For more details on the methods see Hestir et al. (2008). It was not possible to differentiate between species of submersed aquatic plants because of similarities in their spectral signatures. Therefore, the patterns observed refer to submersed aquatic plants as a functional group and not specifically to Brazilian egeria. Thus, our map results are for all submersed aquatic plants at or near the surface (< 2.5 m deep at low tides). In addition, light penetration is limited in these waters because of high (and variable) turbidity, and detection becomes problematic when plants are too far below the water surface. In contrast, the spectral signal of floating and emergent species was sufficient to distinguish these species. To maximize imagery of submersed plants, aerial imagery transects were flown when tides were low. Simultaneously with imagery acquisition, we conducted field surveys of the target species. We conducted boat surveys throughout all

Table 1. Confusion matrices itemizing accuracy assessments per year: producer's accuracy, user's accuracy, omission and commission errors, κ statistic and total map accuracy.[a]

Year	Plant type	Producer	User	Omission	Commission	κ	Total
2003	SAP	83.96	91.75	16.04	8.25	0.6161	69.63
	Waterhyacinth	100	100	0	0	0.1756	
2004	SAP	66.54	98.12	33.46	1.88	0.9656	77.37
	Waterhyacinth	90.04	69.55	9.96	30.45	0.6462	
2005 (June)	SAP	72.48	94.33	27.52	5.67	0.9077	77.99
	Waterhyacinth	64.33	78.42	35.67	21.58	0.7398	
2005 (October)	SAP	79.16	84.24	20.84	15.76	0.7797	75.43
	Waterhyacinth	45.79	62.55	54.21	37.45	0.5504	
2006	SAP	71.98	94.8	28.02	5.2	0.8985	76.17
	Waterhyacinth	55.51	68.33	44.49	31.67	0.6485	
2007	SAP	83.49	83.43	16.51	16.57	0.761	86.29
	Waterhyacinth	39.29	17.89	60.71	82.11	0.17	

[a] SAP, submersed aquatic plants.

Delta waterways and collected field control points of submersed aquatic plants (SAP), waterhyacinth, other emergent vegetation (OEM), and water. During the field campaign, points were selected a priori of the remote-sensing analysis. Points were randomly selected in the field, ensuring full coverage of all the Delta waterways and ensuring a high number of points of all the submerged and emergent species detected in the field, to be used as groundtruthing of the remote-sensing features of interest. At each location, we recorded its geographical position using a Geographic Positioning System (GPS; Trimble GeoXT)[2] unit and recorded the species composition and percentage of cover. Field points were then screened to select points with $> 50\%$ cover to perform accuracy assessment of the distribution maps. A total of 13,085 points were collected during the 5 yr of survey ($n_{2003} = 135$, $n_{2004} = 1{,}728$, $n_{2005\_jun} = 2{,}085$, $n_{2005\_oct} = 1{,}921$, $n_{2006} = 2{,}753$, $n_{2007} = 4{,}463$). The accuracy of the areal cover maps produced was assessed using producer's and user's accuracy estimates and κ statistics (Bloch and Kraemer 1989, Cohen 1960). The total classification accuracy was $> 70\%$ for all years, except 2003 (Table 1). This map accuracy is within the ranges reported by other studies using similar techniques (Bachmann et al. 2002, Glenn et al. 2005, Phinn et al. 2000). We conclude that our classification is conservative because it is significantly affected only by omission errors, resulting in a likely underestimation of the mapped, emergent, floating, and submersed aquatic plant communities.

Cover Estimates and Herbicide Treatment Conditions. We estimated submersed aquatic plants area and percentage of cover at local and Delta-wide scales. At the local scale, we considered the CDBW treatment polygons, corresponding to actual treatment locations each year. At the Delta-wide analysis, we considered all Delta waterways. We overlaid each of these data sets (treatment sites, management sites, and Delta waterways) on the remote-sensing distribution maps and calculated the submersed aquatic plant area (pixel-based) and percentage of cover at each scale yearly between 2003 and 2007.

Additionally, for the local scale of analysis, we created a set of matching control (no-treatment) sites. These sites mimicked the treatment sites in proximity to treatment site; initial (2003) submersed aquatic plants distribution, size, river, waterway, and island characteristics; and salinity gradients. We verified our no-treatment site choice with the management agency. For each of these sites, submersed aquatic plants' area and cover was estimated for all the years.

Finally, local treatment sites were assigned a treatment condition: (1) herbicide treatment applied in only 1 yr, (2) treatment applied in 2 consecutive yr, and (3) treatment applied in 3 consecutive yr. In total, 53 treatment sites were chosen for subsequent analysis.

We estimated waterhyacinth only at the Delta-wide scale because WHCP includes all detected occurrences of this species. We calculated the waterhyacinth area (pixel-based) and percentage of cover annually during 2003 to 2007.

Effects of Herbicide Treatment Priority. Herbicide applications for control of Brazilian egeria in the Delta are highly regulated and were limited to those sites with the greatest negative impacts on navigation and recreational uses caused by the target plant (CDBW 2005a, 2006). This created a nonrandom, directional selection of treatment sites to meet regulatory standards and programmatic goals; however, if this spatial distribution is not accounted for, it could bias the results when comparing with control (untreated) sites. Thus we could not assume that sites not chosen for treatment had less cover of the target weed

Table 2. Generalized linear model results of comparison of treatment and control sites area and percentage cover by submersed aquatic plants at time t_0.

Year	Parameter	Treatment	Control	χ^2	P value
2003	Area (ha)	7.45 ± 2.12	5.82 ± 2.12	0.32	0.57
	Cover (%)	0.43 ± 0.09	0.31 ± 0.09	0.99	0.32
2004	Area (ha)	4.69 ± 1.52	5.05 ± 1.52	0.03	0.86
	Cover (%)	0.33 ± 0.06	0.3 ± 0.06	0.09	0.76
2005	Area (ha)	4.2 ± 1.49	7.03 ± 1.49	0.01	0.89
	Cover (%)	0.31 ± 0.06	0.36 ± 0.06	0.28	0.6
2006	Area (ha)	7.27 ± 2.29	9.45 ± 2.29	0.47	0.5
	Cover (%)	0.44 ± 0.05	0.39 ± 0.05	0.37	0.54

because the criteria were driven by impact, not solely density or cover. To address this issue, we took a conservative approach to test the reliability of the site selection. Our first step was to verify the assumption that treatment and no-treatment sites had similar submersed aquatic plant areas across all years. To assess whether the treatment site was a true subset of the total sites, we compared the range of variability between treated and control sites and statistically assessed the differences using a generalized linear model. We found no significant differences in either area or cover of submersed aquatic plants between control and treatment sites for any of the years (Table 2), indicating that the initial conditions were similar. The second step was to standardize the measurements to detect the effect of site area. We used percentage cover of submersed aquatic plants for the remaining analysis because it reduces the errors associated with using a quadratic term (area) in linear statistical methods. The third step was to assess whether the cover of the previous year (t_0) had a clear influence on the measured differences between treatment and control sites. Because invasive species are known to have slower spread at higher densities, we expect a negative slope in a regression between cover at time t_0 and the change in % cover from t_0 to t_1. We then tested the parameter estimates using a t test and estimated the P value. If the slope of the regression was significantly different from zero and negative, we assumed that the results were independent of the cover at time t_0. If the regression had a positive slope, it resulted from an effect on t_0 cover. Our results showed negative slopes for both treatment and control sites for the pooled analysis across years and for each year individually, except for 2003 (Table 3), indicating that the results were independent of the cover at time t_0.

Multiple assessments (i.e., monitoring) of invasive species cover that extends over large spatial scales is likely to be affected by both spatial and temporal autocorrelation. Positive spatial autocorrelation (Legendre 1993) occurs because infested pixels increase the likelihood of infestation of neighboring pixels, more than farther distant pixels, reducing the independence of data locations (Legendre 1993) and inflating the significance of statistical tests (Legendre et al. 2002). Temporal autocorrelation results from acquiring multiple measures at spaced intervals in time at the same location. This is particularly important for invasive species, which have inherent lag times (Crooks 2005). To account for autocorrelation, we used the mixed-model approach to assess autocorrelation in repeated measures of treatment and control sites using percentage of cover as the dependent variable; year and treatment type (treatment or control) were included as fixed factors. We used the interaction between treatment type and year, and the spatial location of treatment site as random effects. Our results showed no significant effect on the random factors ($t = -0.8$; $P = 0.43$), which confirmed that the data are not affected by spatial and temporal dependency. Because the data passed all autocorrelation tests, they were considered reliable for further analysis.

Effect of Herbicide Type. To test the effect of the herbicide type, we compared the change in cover between all pairs of 2 consecutive yr and between 2004 and 2007. We restricted the analysis to treated sites and assessed the effect of treatment with diquat or fluridone using a general linear model (GLM). Despite the differences in modes of action and uptake characteristics of the two herbicides (diquat is a rapid-acting contact herbicide, and fluridone is a slow-acting systemic herbicide), there were no significant differences in cover in the overall area treated with either diquat or fluridone between all years ($\chi^2 = 0.349$; $P = 0.55$) and for pairwise comparisons (Figure 2). We grouped both herbicides as the effect of treatment.

Similarly, even though two types of systemic herbicides were applied to control waterhyacinth (i.e., 2,4-D and glyphosate), we grouped both herbicides into the effect of treatment because they were often both used at a given site over the control season.

Spatial Dynamics within a Treatment Season. Spatial dynamics of aquatic ecosystems are hard to assess because snapshots of species distributions often fail to incorporate

Table 3. Regression between cover at time t_0 and difference in cover between time t_0 and t_1 for both treatment and control sites.

Year	Treatment or control	Regression equation	R^2	F	P value	Slope	t ratio	P value
Total	Treatment	$\Delta_{SAPt0-t1} = 0.215 - 0.535 \times SAP_{t0}$	0.3	33.34	< 0.0001	-0.535	-5.77	$<$ **0.0001**
	Control	$\Delta_{SAPt0-t1} = 0.241 - 0.594 \times SAP_{t0}$	0.31	33.84	< 0.0001	-0.594	-5.82	$<$ **0.0001**
2003	Treatment	$\Delta_{SAP03-04} = 0.137 - 0.407 \times SAP_{03}$	0.15	1.36	0.28	-0.407	-1.17	0.278
	Control	$\Delta_{SAP03-04} = 0.079 + 0.0009 \times SAP_{03}$	0.0007	0.006	0.94	0.0009	0.08	0.942
2004	Treatment	$\Delta_{SAP04-05} = 0.263 - 0.541 \times SAP_{04}$	0.32	8.42	0.01	-0.541	-2.9	**0.009**
	Control	$\Delta_{SAP04-05} = 0.339 - 0.716 \times SAP_{04}$	0.3	7.64	0.01	-0.716	-2.76	**0.013**
2005	Treatment	$\Delta_{SAP05-06} = 0.286 - 0.465 \times SAP_{05}$	0.29	9.21	0.006	-0.465	-3.04	**0.006**
	Control	$\Delta_{SAP05-06} = 0.226 - 0.465 \times SAP_{05}$	0.29	9.21	0.006	-0.465	-3.04	**0.006**
2006	Treatment	$\Delta_{SAP06-07} = 0.158 - 0.547 \times SAP_{06}$	0.3	9.25	0.006	-0.547	-3.04	**0.006**
	Control	$\Delta_{SAP06-07} = 0.239 - 0.816 \times SAP_{06}$	0.47	19.87	0.0002	-0.816	-4.46	**0.0002**

[a] SAP, submersed aquatic plants.

Bolded values represent significant differences at P $<$ 0.05

short-term, transient changes. To overcome this limitation, we used a multitemporal approach to test the effect of treatment in changes of percentage of cover (horizontal spread) of the submersed, aquatic plant community within and across treatment years. We used a remote sensing–derived, pixel-based classification of total submersed aquatic plant cover to assess the rates of change. Within a treatment year, we compared the change in submersed, aquatic plant cover between June (before treatment) and October (after treatment) in 2005 treatment and control sites and assessed statistical significance using a GLM. Finally, we assessed the horizontal spread rates by fitting a regression to the submersed, aquatic plant cover in June and October for both treatment and control sites. If the slope was significantly positive, we concluded the submersed aquatic plants were spreading; if it was negative, they were regressing, and if it was zero, then no changes were observed.

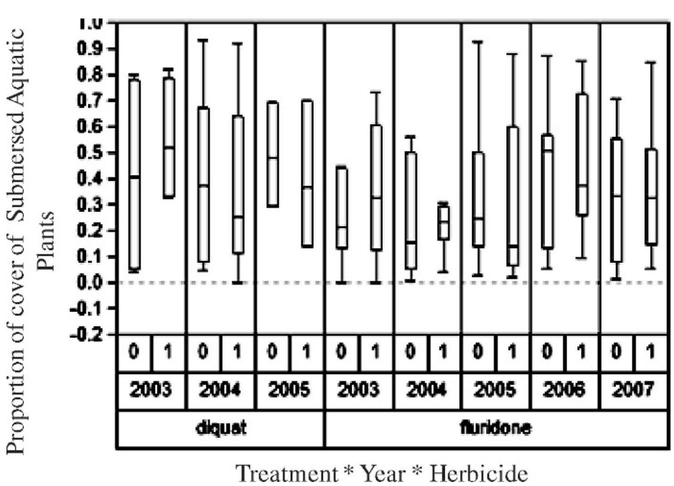

Figure 2. Box plot of the submersed aquatic plants percent cover in treatment and control sites by year and herbicide type.

Because waterhyacinth treatment occurred in every nursery site, no control sites were available for comparison. We limited our analysis to across–treatment-season comparisons.

Spatial Dynamics across Treatment Seasons. To assess the pattern of change of submersed aquatic plants across treatment seasons, we used a GLM to compare the annual change in cover across years, for treatment and control sites, with treatment as a fixed factor. For waterhyacinth, the comparison was done for the yearly area estimates. These analyses were done for all pairs of consecutive years (2003 to 2004, 2004 to 2005, 2005 to 2006, and 2006 to 2007) and for the entire data set (2003 to 2007). To assess the rates of spread across years, we fitted a regression to the submersed aquatic plants cover at time 0 and time 1, for both treatment and control sites. A similar analysis was done for waterhyacinth considering cover at time 0 and 1. If the slope of the regression was significantly positive then the submersed aquatic plants were spreading; if it was negative, then they were regressing, and if it was zero, then no changes were observed.

Temporal Dynamics across Treatment Seasons. We examined the effect of interrupting the treatment temporally by comparing among treatment sites treated once, twice, or three times. We also compared the results for treatment sites with gaps of a year within their treatment prescription. The significance of change was tested using a GLM. Because all waterhyacinth infested sites were treated independent of previous treatment and because of waterhyacinth's floating characteristics, it does not allow assessing the effect of continued treatment in the same sites over years.

Effect of Residual Plant Material on Recolonization. We compared the area covered by submersed aquatic plant residuals in October 2005 to the area covered in June 2006 to assess the effect of treatment residuals on submersed aquatic plants distribution in the following year. We used a GLM to test the significance of change in both treatment and control sites. For waterhyacinth, we compared the area covered by waterhyacinth in October 2005 with that covered in June 2006, using a GLM to test for significance.

Effect of Multiple-Species Treatments. Our last objective was to understand whether the simultaneous application of treatments for different species types (submersed plants and waterhyacinth) had cumulative impacts or whether the treatments resulted in release of one species (e.g., opening up available surface area). We used the time series of pixels classified in 2003 as either submersed aquatic plants or waterhyacinth and assessed their coverage in 2004, 2005, and 2006. We define species change in terms of proportion no-change and change from submersed aquatic plants to waterhyacinth and vice versa. The proportion was calculated by dividing the area in 2004, 2005, and 2006 by the area of either submersed aquatic plants or waterhyacinth in 2003.

Image analysis and map production was performed in Definiens[3] Professional 5 and ENVI[4] version 4.3. All statistical analyses were performed in JMP,[5] version 7, at a significance level of 0.05. We used Bonferroni corrections for P values in multiple comparisons using the same data set.

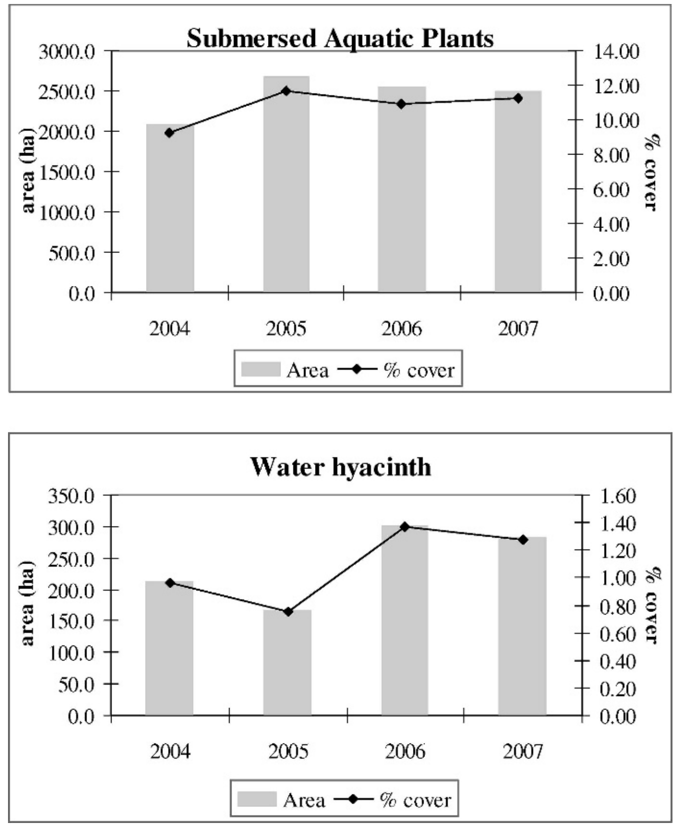

Figure 3. Area and percentage of cover by submersed aquatic plants and water hyacinth in the Sacramento–San Joaquin River Delta in 2004, 2005, 2006, and 2007. Please note that 2003 values were estimated for the central Delta (381.1 km^2) and, therefore, were not represented in this graph.

Results and Discussion

About 12 to 15% of the Sacramento–San Joaquin River Delta surface area was covered every year by invasive aquatic species during the period of this study. In the past 5 yr submersed aquatic plants covered 10 to 12%, and waterhyacinth covered about 1 to 2% of the Delta waterways. The area covered by waterhyacinth (160 to 300 ha) was much lower than that of submersed aquatic plants (about 2,500 ha) for all years (Figure 3). From year to year, the Delta-wide area of submersed aquatic plants increased, particularly in 2005 (Figure 3). Waterhyacinth, on the other hand, showed a decrease in 2005 and 2007 (Figure 3).

Spatial and Temporal Dynamics of Submersed Aquatic Plants. At the scale of treatment sites, we found no significant effect of treatment in the aerial cover of submersed aquatic plants within a treatment season (from June to October 2005; $\chi^2 = 0.073$; $P = 0.79$) and across treatment seasons (from 2003 to 2007; Table 4). Within a treatment season, some treatment sites showed an average decrease in submersed aquatic plants cover (−0.3%), and control sites showed an average increase 10-fold higher (3%). Across treatment seasons, some treatment sites decreased in submersed aquatic plant cover between 2003 and 2004 and 2006 to 2007 (Table 4). The rate of spread of submersed aquatic plants was not significantly different in either treatment or control sites within a treatment season. Similarly, the horizontal spread of submersed aquatic plant cover was not significantly different from zero in either treatment or control sites (Table 5). Across treatment seasons, however, the rates of spread always showed a positive slope, which was significantly different from zero in treatment sites between 2004 and 2005, 2005 and 2006, and 2006 and 2007. In control sites, the slope was significantly positive between 2003 and 2004 and between 2005 and 2006 (Table 6). Our data show no treatment effect on residual plant recolonization for treatments through 2006. When comparing the change in cover of submersed aquatic plants between October 2005 and June 2006, no significant differences were found due to treatment ($\chi^2 = 0.6$, $P = 0.45$). Control sites showed, on average, a higher net increase in submersed plants of 8 ± 3%, compared with the treatment sites' increase of 5 ± 3%. Even though the distribution maps are not species specific, we believe that our results from the SAP community matched those of the target species from 2003 to 2006 because (1) the herbicide-treated sites (and

Table 4. Change in submersed aquatic plant (SAP) percentage of cover in pairwise comparisons of the years and generalized linear model χ^2 values between treatment and control sites.

Year	χ^2	P	SAP cover Treatment	SAP cover Control
			%	%
2003–2004	1.66	0.197	8.2	−3.8
2004–2005	0.142	0.706	12.1	8.6
2005–2006	0.000	0.99	7.9	7.9
2006–2007	0.0003	0.987	−8.4	−8.2

the control sites) were selected by CDBW in areas where Brazilian egeria was the dominant species and required treatment; (2) if Brazilian egeria released other SAP from competition, they would colonize the Brazilian egeria–freed areas, and the net effect would be an undetected decrease in Brazilian egeria cover; and (3) the relative selectivity of the herbicide to Brazilian egeria and Eurasian watermilfoil (Sprecher et al. 1998) would make these two species more likely to change than other SAP species. CDBW surveys and our field data indicated that Brazilian egeria returned every year, and it was the dominant species. The only exception was in 2007, where there was an observed shift in the SAP community composition where the Brazilian egeria–freed areas were indeed being replaced by other SAP species, and there was a measurable increase in the co-occurrence between Brazilian egeria and other SAP species (See Santos et al. 2009). For example, the low dose (about 10 ppb) did not prevent native sago pondweed from sprouting from tubers and establishing successfully during the late spring and summer months (Ruch 2008), whereas no Eurasian watermilfoil was detected in any of the numerous rake samples taken posttreatment, even though it had been present pretreatment.

Treatment sites treated for 2 or 3 consecutive yr had on average a trend toward negative net change in submersed aquatic plants cover, although this change was not statistically significant (χ^2 = 1.89; P = 0.16). In sites treated only once, submersed aquatic plants cover increased on average 15 ± 7%, whereas in sites treated twice, submersed aquatic plants cover decreased on average −5 ± 10%, and in sites treated three times, submersed aquatic plants cover decreased on average −5 ± 13% (Figure 4). However, the timing and scale of fluridone applications were clearly a major determinant in level of efficacy as illustrated in the data from Franks Tract. No significant reduction in submersed plants was observed in Franks Tract until 2007 when the first large-scale (1,500 ha), spring (April) applications were made within the tract (Figure 5). Because of restrictions before 2007 on herbicide applications to July 1, prior year applications occurred far too late in the submersed aquatic plants growth cycle. In 2007, there was a significant reduction in submersed aquatic plants (Figure 5).

The relative specificity of the herbicide activity toward Brazilian egeria and Eurasian watermilfoil, compared with all submersed aquatic plants and the small area treated (3% of the Delta waterways), does not allow extension of these results to the entire Delta. Nonetheless, our results indicate that the first treatment regime for submersed aquatic plants did not result in significant differences between treatment sites compared with control sites. However, when early (spring) applications of herbicide to submersed aquatic plants were allowed, a 47% decrease in areal cover was observed (Figure 5).

Vegetative growth and dispersal of submersed aquatic plants, including Brazilian egeria, curlyleaf pondweed and Eurasian watermilfoil occur through a variety of vegetative propagules, such as shoot fragments containing nodes, rhizomes, root crowns, and turions (Anderson 1999, Haramoto and Ikusima 1988, Sculthorpe 1965). This creates a high propagule pressure, which allows plants to exploit changes in habitats in the Delta from year to year. Shifts in relative abundance, therefore, can result from a suite of changes: (1) high nutrient availability in the sediment can promote disproportionate rates of increase, (2) changes in plant canopy may favor one species over another, (3) plants may respond differentially to herbicide effects, (4) spatial environmental variability (e.g., hydraulic and bathymetric) can make it difficult for consistently effective herbicide treatment, (5) timing of herbicide

Table 5. Regression equations of submersed aquatic plant (SAP) spread from June to October 2005 in treatment and control sites by R^2, model fit, and slope estimate tests. Rates of change are estimated by regression slopes, positive slopes significantly different from zero indicate yearly increase (parameter estimate tests P value).

Treatment or control	Regression equation	Regression line R^2	F	P	Estimates Intercept (P)	Estimates Slope (P)
Treatment	$SAP_{Oct} = 0.337 + 0.06 \times SAP_{Jun}$	0.009	0.175	0.68	0.0001**	0.68
Control	$SAP_{Oct} = 0.414 - 0.0203 \times SAP_{Jun}$	0.0006	0.012	0.916	0.0002**	0.92

**represent highly significant differences at P < 0.01

Table 6. Rate of spread of submersed aquatic plants (SAP) percentage of cover across years by regression line R^2, model fit, and slope estimate tests. Rates of change are estimated by regression slopes, positive slopes significantly different from zero indicate yearly increase (parameter estimate tests P value).

Year	Treatment or control	Regression equation	R^2	F	P	Intercept (P)	Slope (P)
2003 to 2004	Treatment	$SAP_{2004} = 0.137 + 0.593 \times SAP_{2003}$	0.27	2.89	0.13	0.457	0.127
	Control	$SAP_{2004} = 0.079 + 1.009 \times SAP_{2003}$	0.89	65.89	0.0001**	0.157	0.0001**
2004 to 2005	Treatment	$SAP_{2005} = 0.263 + 0.459 \times SAP_{2004}$	0.25	6.07	0.02	0.004**	0.024*
	Control	$SAP_{2005} = 0.339 + 0.284 \times SAP_{2004}$	0.06	1.19	0.29	0.004**	0.29
2005 to 2006	Treatment	$SAP_{2006} = 0.226 + 0.535 \times SAP_{2005}$	0.35	12.21	0.002*	0.002**	0.002**
	Control	$SAP_{2006} = 0.226 + 0.535 \times SAP_{2005}$	0.35	12.21	0.002*	0.002**	0.002**
2006 to 2007	Treatment	$SAP_{2007} = 0.158 + 0.453 \times SAP_{2006}$	0.22	6.34	0.02*	0.09	0.02*
	Control	$SAP_{2007} = 0.239 + 0.184 \times SAP_{2006}$	0.04	1.02	0.32	0.01*	0.32

*represent significant differences at $P < 0.05$

**represent highly significant differences at $P < 0.01$

applications may not be optimal, and (6) positive feedback loops created by submersed aquatic plants can reinforce newly established populations. As colonies become established, they (1) create more stable substrate; (2) reduce water flows (up to 41%; Champion and Tanner 2000), which facilitate establishment and expand colonies along their margins; and (3) increase water temperature (1 to 5 C higher) within the plant canopy because it absorbs more solar radiation. The current hyperspectral analysis can only document changes in areal cover but cannot allow us to ascribe a particular "driver" of the observed changes, except for the clear and significant differences of pre-2006 and post-2006 timing of herbicide applications. The effective use of the systemic herbicide fluridone was clear only when it was applied in early spring, during the period of rapid plant growth (Figure 5). The proper timing, coupled with effective herbicide concentrations (attainable via multiple applications), together resulted in the very successful reduction in Brazilian egeria cover (our results) and biovolume (ReMetrix 2007) in Franks Tract during the 2007 season.

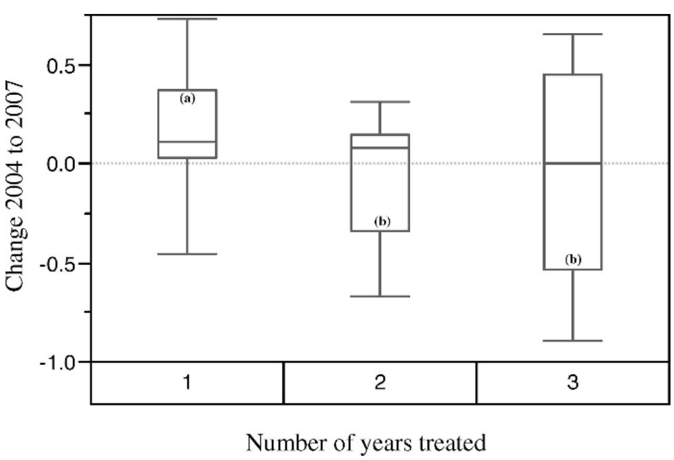

Figure 4. Box plot of the change in submersed aquatic plants cover from 2004 to 2007 by the number of years any given site was treated. (a) and (b) indicate treatment sites treated only once were significantly different from treatment sites treated two or three times.

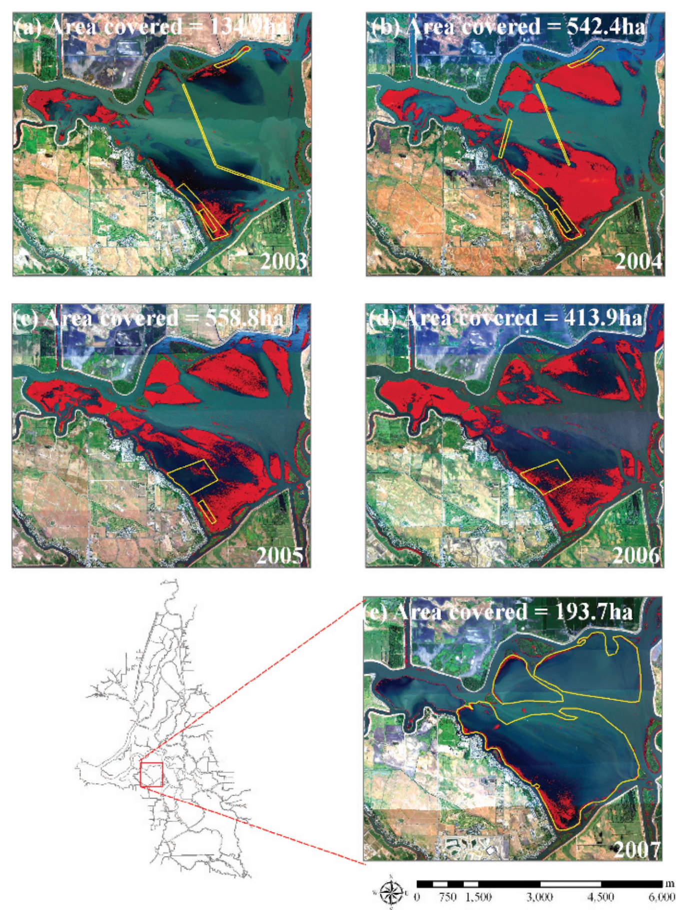

Figure 5. Franks Tract submersed aquatic plants coverage from 2003 to 2007. Yellow lines represent the extent of the treatment sites in each year.

Table 7. Change from submersed aquatic plants (SAP) to waterhyacinth and vice versa in image products for the Sacramento–San Joaquin River Delta in 2003, 2004, 2005, and 2006. Note: 2003 values were estimated for 11 flight lines, whereas the remaining year values were estimated for 64 flight lines.

Vegetation change		Area		
		2003–2004	2004–2005	2005–2006
		ha		
SAP	Waterhyacinth	8.87	16.1	21.9
Waterhyacinth	SAP	6.28	30.2	22.6

Spatial and Temporal Dynamics of Floating and Emergent Aquatic Vegetation. We found no significant differences between waterhyacinth percentage of cover across treatment seasons ($\chi^2 = 1.2$; $P = 0.31$), covering 2 to 3% of the waterways. However, there were significant differences in the rate of change across years ($\chi^2 = 2.738$; $P = 0.018$). Change in waterhyacinth percentage of cover between 2005 and 2006 is significantly higher than the detected change in 2003 to 2004 ($P = 0.032$), 2004 to 2005 ($P = 0.003$), and 2006 to 2007 ($P = 0.001$). Regression analysis did not reveal any significant effect of previous-year cover on the waterhyacinth spread. In fact, all analyses showed that the slope of the regression was not significantly different from zero (P value > 0.05). No significant effect of October 2005 residual plant material was detected in the June 2006 cover of waterhyacinth ($\chi^2 = 0.02$; $P = 0.9$). In addition, the slope of the regression was not significantly different from zero.

A common trait of invasive species is to compete and spread through space over time; if left undisturbed they exhibit rapid growth rates and spread (Eiswerth and Johnson 2002). For example, waterhyacinth mats are known to expand at a rate of 0.61 m/mo and are found to double every 2 wk through the production of offshoots, with a mat of 10 plants easily producing 650,000 plants in one growing season (Penfound and Earle 1948). Our analysis showed a net decrease in the surface area invaded by waterhyacinth from 2003 to 2006, with the exception of a small increase from 2005 to 2006. Moreover, the spatial location of the colonized areas changed from year to year, as a result of within- and across-season drifts in the floating colonies. There are various potential explanations for this pattern: (1) climatic conditions, (2) nutrient load, (3) herbicide applications, and (4) interactions between these factors. Waterhyacinth is sensitive to variable climate conditions, such as number of days of cloud cover (Williams et al. 2005) and light availability (Methy et al. 1990) and decreases growth when exposed to chronic (2 to 3 wk), below-freezing temperatures (Owens and Madsen 1995). However, these two conditions do not seem sufficient to explain the observed increase in areal extent from 2005 to 2006, because both years had similar number of days of cloud cover and because waterhyacinth in the Delta withstands the short periods of below-freezing temperatures (L.W.J. Anderson, personal observation). Most likely, the very cold rainy spring that was near flood conditions in 2005 resulted in fast flows that may have swept some waterhyacinth out of the system. We, therefore, conclude that the sustained application of foliar herbicides on waterhyacinth is the most plausible explanation for the observed multi-year decline.

Effect of Multiple Species Management. We found a high interchange in cover by submersed aquatic plants and waterhyacinth. In fact, between 10 and 20% of the area covered by submersed aquatic plants in any given year is replaced in the following year by waterhyacinth, the same happening between waterhyacinth and submersed aquatic plants, with a replacement between 6 and 30% (Table 7). These results suggest the need for integrated vegetation management and not simply single-target species management.

In spite of the high level of invasion of the Sacramento–San Joaquin River Delta, it is one of the least-managed waterways of its kind for control of these species. In this nutrient-rich system, the effect of physical environmental modifications and introductions of nonnative species has made the system highly susceptible to invasion, especially now because of elevated rates at which new invasive species are entering this system (Cohen and Carlton 1998). Invasive species seem to be sequentially replaced by others as either the earlier ones lose their competitive advantage or through management induced reduction. The observed dominance of Brazilian egeria in the submersed aquatic community indicates that this species is a major competitor. Brazilian egeria, acting as an ecosystem engineer by reducing water velocity, may promote species turnover through a positive-feedback loop, enhancing habitat characteristics for establishment of alternative submersed aquatic plant species (e.g., Eurasian watermilfoil), anchorage for floating species, and a rooting substrate to emergent species (Champion and Tanner 2000, Ji 2008). In addition, submersed aquatic species can persist, albeit in low densities, under floating mats of waterhyacinth, particularly at edges and as a consequence of tidal and wind drifts in waterhyacinth (L.W.J. Anderson personal observation). Furthermore, persistence of above-sediment vegetative growth during the winter (Pennington 2007; Pennington and Systma 2009) can provide a competitive advantage over native species. If Brazilian egeria were excluded from the Delta system, it is likely that alternative submersed species may invade any empty niches. Today this appears to be mainly Eurasian watermilfoil, coontail, curlyleaf pondweed, and sago pondweed. Although Eurasian watermilfoil is recognized as an aggressive invasive species in other aquatic systems in the United States, its spread in areas targeted for control of Brazilian egeria would be unlikely. Eurasian watermilfoil is highly sensitive to current herbicide concentrations (Netherland and Getsinger 1995) and so are other species of submersed plants (Marcondes et al. 2003, Nelson et al. 2002, Netherland et al. 1997). Despite coontail's invasive potential (Anderson 2003, Brown and Michniuk 2007), if either coontail or sago pondweed or both replaced Brazilian egeria, the Delta SAP would be returned to a native-dominated community. On the other hand, 14% of areas cleared from waterhyacinth mats have become dominated by submersed aquatic plant communities or by other floating and emergent species, such as floating pennywort and water primrose. Our hyperspectral data show a 6 to 18% increase in cover of floating pennywort and water primrose and a 2 to 3% increase in California bulrush, following effective waterhyacinth reduction over the years of monitoring. Furthermore, the Sacramento–San Joaquin Delta has not been in a ''natural'' state since the extensive levee systems was constructed in the late 1800s nor since the expansion of agriculture and attendant nutrient and sediment inputs. Thus, there is no a priori reason to assume that shifts to native populations of SAP or emergent and floating plants will not also have undesirable effects under the current set of environmental drivers. The question, thus, remains: Is the present management approach affecting the succession of invasive species and are there alternatives?

We suggest that a solution to reduce the potential of a cascade of released species' impacts is to develop and implement a fully Integrated Delta Vegetation Management and Monitoring Program (IDVMMP). This will require simultaneously monitoring multiple invasive and native species, the effectiveness of targeted management actions, and possible feedbacks, in a Delta-wide approach. Our data suggest that single-target weed control approaches are insufficient to provide sustained management of the Delta plant community. Fully integrated vegetation programs (sensu Ehler 2006) have been mostly applied to terrestrial communities; however, few cases were applied to aquatic systems (Gibbons et al. 1994, Lee-II et al. 2008, Van-Damme et al. 2005). This approach reinforces the need for integrating and modeling species presence and responses to environmental parameters, which can be obtained by remote-sensing products, such as those presented here. Thus, a population-level distribution mapping at a high spatial resolution via hyperspectral remote sensing, coupled with other assessments such as point sampling and hydroacoustic imaging (Winfield et al. 2007) should support an effective adaptive management strategy.

Sources of Materials

[1] Hyperspectral imagery, HyVista, Inc., Sydney, Australia.
[2] Trimble GeoXT, Trimble Navigation Ltd., Sunnyvale, CA.
[3] Definiens Professional 5, Definiens, Munich, Germany.
[4] ENVI, Version 4.3, ITT Industries, Inc., Colorado Springs, CO.
[5] JMP, Version 7, SAS Institute, Cary, NC.

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