USE OF REMOTE SENSING TO ASSESS CHANGES IN WETLAND PLANT COMMUNITIES OVER AN 18-YEAR PERIOD: A CASE STUDY FROM THE BEAR RIVER MIGRATORY BIRD REFUGE, GREAT SALT LAKE, UTAH

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ABSTRACT.—Successfully managing wetlands requires monitoring changes in plant community composition. We used remote sensing techniques to document the replacement of desirable native wetland vegetation with invasive species in response to catastrophic flood disturbance in the 1980s and to evaluate wetland vegetation management between 1998 and 2010. We conducted our study at the Bear River Migratory Bird Refuge wetlands, which are located on the north-eastern arm of the Great Salt Lake in northern Utah. We acquired high-resolution airborne multispectral imagery of the refuge in 1992 and 2010 to quantify changes in vegetation cover over time. We produced classified vegetation maps for the years 1992 and 2010 to calculate vegetation and water cover in management units. Classification results indicate that invasive vegetation is fast replacing native species in areas adjacent to the water delivery canals. We also compared vegetation changes to historical data provided by refuge managers, and these data contained information about the management activities undertaken in the wetland units over the study period. We found that the managers' efforts to control the expansion of invasive species—such as keeping the units full of water throughout the year, adjusting water depth to manage salinity levels and aquatic vegetation, and undertaking burning or mechanical disturbance when needed—were successful in most of the units, although some units were still invaded by undesirable plants. Here we demonstrate how wetland managers can use remote sensing and historical data of vegetation cover to understand which native plant species are most susceptible to replacement by invasive species, how vegetation responds to management actions, and ultimately how managers can promote diverse plant communities with high wildlife value.

RESUMEN.—Las actividades efectivas de conservación de los pantanos exige el monitoreo de los cambios en la composición de la comunidad de plantas. Utilizamos técnicas de detección remotas para documentar las condiciones en las que la vegetación nativa del pantano es reemplazada por especies invasoras en respuesta a las perturbaciones ocasionadas por las inundaciones de la década de los ochenta y las actividades de conservación de la vegetación del pantano entre los años 1998 y 2010. Realizamos el estudio en los pantanos del Refugio *Bear River Migratory Bird*, que se encuentran en el área noreste del Gran Lago Salado al norte de Utah. Tomamos imágenes aéreas multispectrales de alta resolución del refugio en el año 1992 y en el año 2010 para cuantificar los cambios en la cobertura vegetal a lo largo del tiempo. Elaboramos mapas de vegetación clasificada correspondientes a los años 1992 y 2010 para calcular la cobertura vegetal y de agua en las unidades. Los resultados de la clasificación indican que la vegetación invasora está reemplazando las especies nativas en áreas adyacentes a los canales de distribución de agua con rapidez. Además, comparamos los cambios de vegetación con la información histórica que ofrecieron las autoridades a cargo del refugio, donde se explica la gestión de conservación de los pantanos durante la duración del estudio. Encontramos que las actividades de los encargados para controlar la extensión de las especies invasoras, tales como, mantener las unidades llenas de agua a lo largo del año, ajustar la profundidad del agua para controlar los niveles de salinidad y la vegetación acuática, y tomar las medidas de quemar o generar una alteración mecánica en caso de ser necesario, tuvieron buenos resultados en la mayor parte de las unidades, aunque en algunas unidades se registró la invasión de plantas en condiciones no saludables. En este estudio mostramos de qué manera los encargados de los pantanos pueden utilizar la detección remota y la información histórica sobre la cobertura vegetal para comprender qué especies de plantas nativas tienen mayor predisposición a ser reemplazadas por especies invasoras, cómo responde la vegetación a las actividades de conservación, y, en última instancia, de qué manera los encargados de la conservación pueden promover comunidades de plantas diversas con alto valor de vida silvestre.

Succession is the change in plant communities through time, particularly after a disturbance (Horn 1974). Primary succession is the direct response of one or more species to permanent changes in the environment, whereas

secondary succession is the process of reestablishment of the original community after a temporary disturbance (Horn 1974). During secondary succession, new species—particularly invasive plants—that can compete with

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resident plants often emerge and may come to dominate the community (Bergen and Dronova 2007).

Wetland managers need to understand the succession and interactions of native plant species and nonnative invasive plant species (hereafter native vs. invasive) to predict species progressions and determine whether management actions are meeting management goals and objectives. Invasive species, such as introduced *Phragmites australis* (common reed, hereafter *Phragmites*) and *Typha* spp. (which include nonnative and hybrid cattail species, hereafter jointly referred to as *Typha*), present major challenges for wetland managers because such species impact floral and faunal diversity, as well as ecosystem functions and services (Galatowitsch et al. 1999, Saltonstall 2002). Refuge managers invest significant time and financial resources to manually and mechanically remove invasive species. Methods such as prescribed fire, herbicides, biological control agents, and native competitors are used to control the spread of invasive wetland plants and avoid the loss of important wetland functions and services (Randall 1996, Mitsch and Gosselink 2007). Use of remote sensing techniques to monitor changes in plant community composition can help managers successfully manage wetland ecosystems.

Airborne multispectral videography is a useful and relatively inexpensive remote sensing tool to classify and map native and invasive wetland and riparian species (Neale and Crowther 1994, Akasheh et al. 2008). Extensive research has used remote sensing techniques to map and classify wetland vegetation (Barrette et al. 2000, Ozesmi and Bauer 2002, Shuman and Ambrose 2003, Rodrigues and Souza 2011); evaluate efficacy of aquatic plant management (Santos et al. 2009); describe wetland temporal changes (Richardson and Harris 1999); study wetland vegetation response to flooding (Sims and Colloff 2012); and assess vegetation change and distribution (Haack 1996, Munyati 2000, Rebelo et al. 2009, Zhang et al. 2009). This prior research has been conducted primarily in wetlands outside of North America. A few North American studies have used remote sensing to assess changes in invasive wetland vegetation cover, particularly for *Phragmites*. However, these North American studies focus on invasion patterns in Atlantic coastal areas, the St. Lawrence River (Brisson

et al. 2010), the California Delta ecosystem (Andrew and Ustin 2008, Hestir et al. 2008) or the Great Lakes region (Pengra et al. 2007). Here, we present a unique data set that (1) considers interactions of native and invasive species in an inland wetland ecosystem over an 18-year period; (2) illustrates one of the dynamic and understudied wetlands of the Intermountain West; and (3) identifies and quantifies the native wetland species that invasive species have replaced over the 18-year period.

Our main objective is to use remote sensing techniques to document changes in desirable native wetland vegetation and undesirable invasive wetland vegetation after a major flood disturbance in the Bear River Migratory Bird Refuge, Utah, which is located along the northeastern arm of the Great Salt Lake (GSL). In 1987, the GSL reached a historic high water level of over 1280 m above mean sea level (Foote 1991) due to higher-than-normal winter precipitation and cooler summers with low evaporation in the preceding 4 years. These factors resulted in an increased water level that flooded with saline water approximately 175,000 ha of wetlands around the GSL margins, including the refuge (Foote 1991). The floods destroyed existing wetland vegetation and rendered the refuge inoperable. In the early 1990s, the waters of the GSL receded, and refuge managers began restoration efforts and management actions, such as controlling water depths and salinity levels in the 25 refuge management units. Thus, a secondary goal of our study was to evaluate the relationship between management objectives, management actions, and wetland vegetation responses. To identify and quantify secondary vegetation succession, we acquired and compared airborne multispectral imagery of the refuge between 1992 and 2010.

STUDY AREA

The Bear River Migratory Bird Refuge (BRMBR) is located in northern Utah where the Bear River flows into the northeastern arm of the Great Salt Lake (Fig. 1). As a federally managed national wildlife refuge, the BRMBR formally protects the wetlands found at the mouth of the Bear River. These wetlands are the largest freshwater component of the Great Salt Lake ecosystem and serve as globally significant resting and breeding grounds for

Fig. 1. Location of the Bear River Migratory Bird Refuge, Brigham City, Utah. The refuge's wetlands are subdivided into multiple units via a vast dike system. The numbered areas correspond to the refuge's wetlands units. The shaded areas represent the management units under study. Adapted from Alminagorta et al. 2010.

Fig. 2. A *Phragmites* stand in a northern unit at the Bear River Migratory Bird Refuge. Image by Melina Santos Vanderlinder.

multiple migratory bird species (Olson et al. 2004, Evans and Martinson 2008).

The refuge is divided into 25 management units that are separated by dikes. Our study focuses on 4 wetland units (1A, 2A, 2B and 5C; see shaded areas in Fig. 1). We chose these management units because they are where refuge managers first started noticing the growth of *Phragmites* after the 1980s floods (shown in Fig. 2; Al Trout, former refuge manager, personal communication, 2010). Over 18 years, we focused on mapping 2 invasive species (*Phragmites* and *Typha*), native emergent plants, and native wet meadows species (Table 1) to

identify which native species were being replaced by invasive species. Although native *Phragmites australis* subsp. *americanus* is widespread in Utah, it is not considered invasive and is very uncommon along the GSL (Kulmatiski et al. 2011, Kettenring et al. 2012, Kettenring and Mock 2012); therefore, it is not considered in our study. The introduced, invasive European lineage of *Phragmites australis*, however, is a dominant presence in Great Salt Lake wetland vegetation and is the target of intensive control efforts (Kettenring 2012); hereafter, we use *Phragmites* to note the invasive, nonnative lineage. Although nonnative

TABLE 1. List of native and invasive species included in classification.

Common name	Scientific name	Native/invasive	Function	Reference
Common reed	<i>Phragmites australis</i>	Invasive	Invasive grass that grows rapidly and decreases plant and animal biodiversity in wetlands throughout the U.S. and worldwide	Tucker 1990
Cattail	<i>Typha</i> spp.	Invasive	Listed as an invasive weed by some regulatory agencies due to its ability to rapidly dominate a wetland plant community and create monotypic stands that can reduce the overall habitat value	Perry et al. 2009
Alkali bulrush	<i>Schoenoplectus maritimus</i>	Native	A perennial wetland plant whose seeds and rhizomes provide food for waterfowl, game birds, and songbirds, as well as muskrat and beaver	Hoag et al. 2001
Hardstem bulrush	<i>Schoenoplectus acutus</i>	Native	A perennial wetland plant that provides cover for numerous birds and mammals, valuable nesting cover and escape cover for a variety of passerines and waterfowl throughout its range, and staple food for muskrat and other small mammals	Boggs et al. 1990
Saltgrass	<i>Distichlis spicata</i>	Native	A perennial of great significance in the salt marshes because it provides nesting grounds for birds, fish, and larvae of many species of marine invertebrates	Skaradek and Miller 2010
Pickleweed	<i>Salicornia</i> spp.	Native	A perennial herb that occurs under high salinity conditions in wetlands	Calflora 2012

and hybrid cattails are purported to be widespread along the GSL, along with native cattails, the relative covers of these different lineages are unknown and we were not able to distinguish different lineages with our remote sensing work. Because managers often consider any lineage of *Typha* as undesirable (Kettenring 2012), we consider *Typha* an invasive species complex in our study.

METHODOLOGY

Overview

We used airborne multispectral videography data from 1992 and 2010 to develop an 18-year overview of vegetation cover changes in 4 refuge management units in response to management actions taken after a major flood disturbance. The 1992 and 2010 data sets provide a unique historical perspective and provide valuable insights on specific vegetation changes in this understudied region. The data sets do not quantify the continuous and dynamic vegetation change at intermediary points during the time period (Narumalani et al. 2004).

We used the historical data available in the refuge's annual habitat management plans (Olson et al. 2004, Olson 2005, 2006, 2007, 2008, 2009) to put the remotely sensed vegetation changes from 1992 to 2010 in context and to help quantify vegetation responses during the interval period. The data include the management goals, the management actions, and a description of the habitat responses between 2003 and 2009. We initially created tables for each year containing the management goal, management actions, and habitat response in terms of percent cover of water and native and invasive vegetation. These tables allowed us to compare the percentages of vegetation cover and water in each unit before and after management actions were taken. Next, we noted whether the changes in vegetation concurred with the managers' expected outcomes. Lastly, using 2 methods, we evaluated whether the managers' goals were met. First, we looked at the reports from previous years and noted the current vegetation percentages as compared to the desired outcomes. Second, we used the remotely sensed vegetation from 1992 and 2010 and observed vegetation changes. Doing so allowed us to identify and quantify the effects of management actions on changes in vegetation cover in the management units.

Image Analysis

We acquired airborne multispectral imagery of the refuge at 1-m pixel resolution in May 2010, using the latest version of the USU airborne digital system (Cai and Neale 1999). Spring is the best time to acquire multispectral imagery to map wetland vegetation cover because the green-up of *Phragmites* and other wetland species begins in Utah between late April and June (depending on weather conditions), and the species can be differentiated fairly easily while dormant. In addition, the dormant wetland species contrast well with wet meadow and upland species, which are already greening up at that point in time.

The USU airborne system acquires high-resolution imagery in the green, red, and near-infrared bands, similar to Landsat Thematic Mapper bands TM2, TM3, and TM4, respectively. Geometric correction was applied to individual images to correct for lens geometric radial distortion by using previously developed functions (Sundararaman and Neale 1997). The correction facilitated the alignment and registration of multiband images, as well as the mosaicking of multiple images along a flightline and adjacent lines (Neale 1997). The 2010 images were rectified to a NAIP image base map using 20–30 common control points (pixels) visible in both images. This rectification procedure maintained the rms error below one pixel. The 1992 images were rectified to 1:24000 black and white orthophotos placed on a table digitizer, also using common control points. The individual rectified and registered 3-band images were mosaicked along the flightlines to produce rectified image strips. The 2010 image strips were calibrated to a reflectance standard using incoming radiation (measured with an EXOTECH radiometer placed at nadir over a standard fused halon reflectance panel [Spectralon]) and system calibration equations relating image digital numbers to radiance obtained in a separate experiment (Neale and Crowther 1994). The calibrated strips were then mosaicked together to form a multispectral image map of the refuge.

Ground surveys were conducted in June 2010 using printed copies of the 2010 imagery to visually observe and verify the vegetation types in the studied units. We visited 260 locations in the refuge, noted each location with a GPS, observed the vegetation present, and used this information to identify the spectral

signatures of the observed vegetation types. The combination of aerial digital photography and ground-based methods provided an accurate and efficient means of sampling vegetation cover and succession of native and invasive species.

The images were processed using ERDAS Imagine software and analyzed using unsupervised and supervised classification techniques (IMAGINE 2010) to estimate the vegetation cover and spread of invasive and native vegetation species (Table 1). During the unsupervised classification, isodata clustering was used in which pixels were clustered based on similarities in spectral information and then automatically grouped into a category. In the supervised classification, we selected several groups of training pixels that were representative of the 8 vegetation classes and later classified the image using the maximum likelihood technique.

After classifying the 2010 imagery, we processed multispectral airborne video imagery acquired in May 1992, also at 1-m resolution, using the airborne system described in Neale and Crowther (1994) and Neale (1997). The 1992 imagery was acquired with bandwidths similar to those used by the airborne system in 2010. We processed and rectified the 1992 imagery using techniques similar to those used with the 2010 imagery (explained above). In the classification of the 1992 imagery, we used knowledge obtained from the spectral signatures of the 2010 imagery. It is possible to do such correlation because the 1992 and 2010 over-flights both occurred around the same time in early spring (end of April), when the wetland vegetation species were at similar stages of phenological development.

To illustrate more clearly the changes in vegetation from 1992 to 2010, we developed a method similar to the change detection feature of Leica Geosystem's ERDAS IMAGINE™ 10.0 software (IMAGINE 2010). This ERDAS feature calculates the difference between pre- and post-event images and highlights the changes (Bitelli et al. 2004). The new image is obtained by comparing corresponding pixels between images and determining the type of change with a change detection matrix (Shaoqing and Lu 2008). Each image had 8 classes corresponding to a specific vegetation type; we assigned a unique pixel value (1–8) to each class. Using Model Maker in ERDAS Imagine, we

performed our own image change detection by multiplying classified pixel values from the 1992 image by 10 and adding pixel values from the 2010 image. This manipulation was done to produce a new, derived image where pixel values ranged from 10 to 88. The pixel values simultaneously encoded the original 1992 (tens digit) and final 2010 (ones digit) vegetation classes, and represented all possible changes in vegetation classifications over the 18-year period. We then re-coded the resulting pixel values to more general categories—native, water, and invasive vegetation—to simultaneously show the types and extent of vegetation changes from 1992 to 2010. We did not analyze temporal trends in units 2A and 5C because annual refuge management plans did not record vegetation cover, survey habitat, list management actions undertaken, or present data on the effects of management actions in these units.

Mapping *Phragmites* in GSL wetlands is particularly challenging due to the dynamic lake levels and diversity of plant species (Pengra et al. 2007). Further, researchers have discovered that the presence of *Typha* can lead to wetland vegetation classification errors (Maheu-Giroux and de Blois 2005). The similar ecological requirements and behaviors of *Typha* and *Phragmites* (Brisson et al. 2010), the species' structural resemblance, and the species' similar location within wetlands result in the intermixing of the spectral signatures of the vegetation classes (Pengra et al. 2007). As a result, we do not differentiate *Phragmites* and *Typha*, and in our analysis we consider them together as one invasive vegetation class.

Accuracy Assessment

Accuracy assessment is an essential tool in determining the quality of information obtained from remotely sensed imagery and in identifying sources of error within the analyzed data. According to Congalton and Green (1998), the error matrix is the standard medium for reporting the accuracy of maps derived from remotely sensed imagery. We performed an accuracy assessment for the 2010 imagery using the ground-truth data and multispectral imagery. Our error matrix consisted of a square array of numbers set out in rows and columns that indicated the number of sample points randomly assigned to each of the 8 vegetation

categories relative to the ones verified on the ground (Table 2). We used a total of 377 points distributed randomly among the 8 vegetation classes. ERDAS IMAGINE offers 5 sampling schemes for collecting the sample points; we selected stratified random sampling because it is the most appropriate method for satisfactorily representing important minor categories (Van Genderen et al. 1978) like *Phragmites* and *Typha*.

We did not perform a formal accuracy assessment on the 1992 classified imagery because no ground-truthing points were collected in 1992. Instead, we qualitatively compared the 1992 classified imagery to descriptions refuge managers provided for locations where they observed *Phragmites* first appearing in 1992 (Al Trout, former Refuge Manager, personal communication, 2010).

RESULTS

Imagery from 1992

Our analysis shows that in 1992, over 93% of the area in units 1A and 2A was open water, while units 2B and 5C had 30% and 58% open water, respectively (see Fig. 3 for classified and unclassified images; see Fig. 4a for vegetation cover breakdown). None of the units showed the presence of invasive *Phragmites* and *Typha*, and in all units, *Schoenoplectus acutus* was the dominant species, with a cover of 3%–17%. Meanwhile, *S. maritimus* was the second-most dominant species, with 2%–8% vegetation cover. In units 2A, 2B, and 5C, unvegetated playa covered 2%–4% of the land area, whereas playa covered with *Salicornia* spp. ranged from 10% to 12% in units 2B and 5C. In 1992, the only unit that showed a substantial percentage of *Distichlis spicata* (8% of all vegetation) was unit 2B.

Imagery from 2010

The classification of management units in year 2010 yielded results very different from year 1992 (Fig. 3, 4b). For example, in 2010 the units were visibly more vegetated. Less than 40% of units 1A, 2A and 2B were open water, whereas 75% of Unit 5C was open water. Every unit showed the presence of *Phragmites* and *Typha*, with percentages ranging from 1% to 4%. In Unit 1A, *S. acutus* was the dominant vegetation, whereas in the other units, *S. maritimus* dominated.

TABLE 2. Classification accuracy error matrix for the 2010 land cover map using ground-truthing data.

Habitat class	Ground-truthing points								Total points	User's accuracy (%)	Producer's accuracy (%)	Kappa coefficient (%)
	1	2	3	4	5	6	7	8				
Alkali bulrush	24						3		27	88.89	96	88
Hardstem bulrush		11	1				1	2	15	73.33	61.11	72
<i>Phragmites</i> and <i>Typha</i>			22	1				5	28	78.57	95.65	77.18
Dry playa				16			1		17	94.12	88.89	93.82
Playa with <i>Salicornia</i>				1	59			9	69	85.51	93.65	82.60
Saltgrass		7			3	41		2	53	77.36	97.62	74.52
Water					1		134	1	137	97.81	95.71	96.52
Water-covered playa						1	1	29	31	93.55	60.42	92.61
Total	25	18	23	18	63	42	140	48	377			
Commissions (%)	4	39	4	11	6	2	4	40		336	Total correct	
Omissions (%)	11	27	21	6	14	23	2	6		89.12	% Accuracy overall	86.32

Accuracy Assessment Results

An overall accuracy of 89% was obtained for the 2010 vegetation classification across the 4 wetland units (Table 2), with a Kappa coefficient of agreement of 86%. We calculated a user's and producer's accuracy for each vegetation class. The user's accuracy indicates the probability that a pixel classified on the map corresponds to the actual vegetation class on the ground and assesses errors of commission. The producer's accuracy indicates the probability that a reference pixel is correctly classified and assesses errors of omission (Story and Congalton 1986). Vegetation classes such as *S. maritimus*, *Phragmites* and *Typha*, and saltgrass have a producer's accuracy of >95% but a user's accuracy of <90%. This result implies that, for example, >95% of the pixels observed as *S. maritimus* in the ground-truthing were labeled as *S. maritimus* in the classification map. At the same time, several additional pixels that were actually *Phragmites* and *Typha* or *D. spicata* were inadvertently classified as *S. maritimus*. The user's accuracy for the *Phragmites* and *Typha* class was 79% and indicates that 79% of all pixels in the classification map labeled as *Phragmites* and *Typha* were actually *Phragmites* and *Typha*.

Changes in Vegetation Cover from 1992 to 2010

To evaluate vegetation succession over the last 18 years, we compared the vegetated areas between 1992 and 2010. Results suggest that *S. maritimus* was replaced by *Phragmites* and *Typha*, with changes concentrated along the water delivery canals to wetland units and water intakes. We also found that many areas previously covered with open water are now covered with native and invasive emergent vegetation. For example, units 1A and 2A had expansive growth of *S. acutus* and some *S. maritimus* into areas that formerly were open water. Unit 2B used to be wet playa with *D. spicata*, but in 2010, the unit was dominated by *S. maritimus*, as well as *Phragmites* and *Typha* along the water delivery canals. Unit 5C showed the greatest increase of *Phragmites* and *Typha* coverage, from 0 ha in 1992 to 25.9 ha in 2010.

When we compared native and invasive species cover over the 18-year period, we found that the most undesirable changes occurred along the water delivery canals (Fig. 5). Unit 5 exhibited the most changes that involved invasive vegetation replacing native vegetation.

Fig. 3. Multispectral image mosaic before and after classification, Bear River Migratory Bird Refuge. Color legend (bottom left) remains the same for all classified imagery. Refer to Figure 1 for scale.

Units 1A and 2A showed changes in which open water areas became populated with native vegetation, as well as a few areas where native vegetation replaced invasive vegetation. Vegetation cover in unit 1A increased from 6% to 51%, and open water areas in unit 2A decreased from 93% to 17%. In unit 2B, vegetation cover increased from 34% to 72%; the most dynamic changes occurred in this unit, with 4% of native vegetation becoming invasive, and water replacing native vegetation and vice versa. Unit 5C experienced the least change (17% vegetation cover increase) in open water areas

(Table 3). Units 2A and 2B showed much smaller increases in invasive *Phragmites* and *Typha* area over the 18-year period as compared to the larger increases in units 1A and 5C. Because we only qualitatively assessed the accuracy of the 1992 imagery, we cannot definitively state the accuracy of the change detection.

Management Actions and Vegetation Changes—Units 1A and 2B

Because there was no remote sensing data available for the years between 1992 and 2010,

Fig. 4. Vegetation cover of management units in 1992 (A) and 2010 (B), Bear River Migratory Bird Refuge.

we used the refuge management plans to understand vegetation dynamics during these years. By reviewing the refuge management plans for 2003 through 2009, we constructed a time series of vegetation cover in wetland units 1A and 2B (Fig. 6). In unit 1A, invasive vegetation appeared some time between 1992

and 2003. After 2006, invasive vegetation cover decreased. From 2003 to 2009, the management goal for unit 1A was to achieve 50% open water and 50% emergent vegetation. To achieve this goal, refuge managers (1) kept unit 1A full of water through the year; (2) managed salinity levels by managing water levels

Fig. 5. Images of change by pixel within each management unit. Change detection matrix (bottom right) and color legend apply to all units, Bear River Migratory Bird Refuge. Color-coding indicates the state change shown in the table, whereas numbers tabulate the total area changed (ha). Refer to Fig. 1 for scale.

TABLE 3. Change in vegetation cover over time (hectares) in the 4 wetland units at the refuge.

Unit	Total area	Native species		<i>P. australis</i> + <i>Typha</i> spp.		Open water	
		1992	2010	1992	2010	1992	2010
1A	220	14.6	102.8	0	9.1	204.4	91.8
2A	54.4	1.9	41.7	0	1	50.6	9.5
2B	119	11.4	82.3	0	4	35.9	27.8
5C	1029	230.4	165.9	0	26	592.3	767.2

(depth of water); and (3) periodically drew down water levels and, when needed, burned, mechanically removed, or sprayed herbicide on invasive plants. By 2010, management actions had nearly reached the desired reduction of invasive species vegetation cover and had achieved the approximately desired ratios of open water to emergent vegetation.

From 1992 to 2005, the vegetation in unit 2B was entirely native. In 2006, invasive vegetation appeared, and by 2010, the percent cover of invasive species was reduced. Before 2006, managers (1) maintained unit 2B with standing water throughout the year and (2) prohibited herbicide use in order to reach the management goal of 75% *S. maritimus* cover to 25%

Fig. 6. Time-series plot for management units 1A (top) and 2B (bottom), Bear River Migratory Bird Refuge. Plots show remotely sensed vegetation cover from 1992 and 2010. The other years represent management goals for vegetation cover described in the refuge's habitat management plan.

open water cover. To counteract the expansion of invasive vegetation in unit 2B in 2006, managers subsequently manipulated water depth to control salinity levels. As a result, invasive vegetation cover after 2006 decreased and approached the target (Fig. 6).

DISCUSSION

We used high-resolution airborne imagery and remote sensing techniques to identify large-scale and long-term changes in wetland vegetation at the refuge. The refuge is a dynamic wetland system, and we documented dramatic changes in the vegetation over 18 years. Said changes include emergent vegetation now covering areas previously flooded with water, arrival of invasive species, and invasive species replacing native species.

Our large-scale spatial analysis allowed us to identify patterns of wetland plant invasions.

More specifically, in 1992, no invasive species were detected at the refuge as compared to 2010, when every unit had *Phragmites* and *Typha* present. Image change detection showed that invasive vegetation was grouped in large, homogeneous areas instead of random spatial patterns. For example, we found the largest increases of *Phragmites* and *Typha* cover around specific wetland unit features like canals and water intakes. These patterns are supported by prior work that suggests that (1) diking wetlands fosters plant communities that are characteristic of lower salinity levels, including *Phragmites* and *Typha* (Weinstein et al. 2001); (2) waterways such as canals can serve as a conveyance network to disperse plant propagules, including those of invasive species (Wilson et al. 2009); and (3) invasive species management can be particularly challenging along the steep slopes of dikes and canals. Experimental evidence will be required to determine

the causal factors of the specific vegetation patterns in our study.

We examined the relationship between management actions and vegetation response following the major flooding of the Great Salt Lake. These dynamics identify important temporal and spatial patterns that managers can use to more effectively manage wetlands and reduce future invasion (Bradley and Mustard 2006). Observed cover of open water, native vegetation, and invasive vegetation nearly matched management goals and suggests that management actions such as regulating water depths, managing salinity levels, and controlling the spread of invasive species had the desired outcomes. These management actions capitalized on known abiotic controls on wetlands plants, such as the importance of water depth for preventing seed germination in *Typha* (Travis et al. 2010) and salinity for reducing invasion of *Phragmites* (Lissner and Schierup 1997, Brisson et al. 2010). These findings highlight the importance of basic ecological studies in understanding the drivers of wetland plant interactions and integrating such findings into active wetland management.

Suggestions for Future Work

Two time-framed aerial-image acquisitions were used to obtain an 18-year historical perspective and insight on vegetation changes. To further understand timescales of wetland succession and invasive succession patterns, future researchers should observe changes at higher temporal resolution, such as annually. Observing changes at higher temporal resolution would detect and quantify the continuous and dynamic phenomena of vegetation change over the entire time period. Where budgets allow, we recommend continuing observations of vegetation change through use of high resolution remote-sensed imagery. Managers can alternatively use free, low-resolution images such as those provided by Landsat (30-m resolution). However, the classification accuracy will be lower, and it will be harder and take longer to identify areas of emerging invasive vegetation. It will also be important to simultaneously observe the hypothesized drivers of vegetation change, such as water depth, salinity levels, and management activities in order to understand vegetation changes in context.

Despite intense research efforts, succession patterns of invasive species following a

disturbance still remain poorly understood (Odland and del Moral 2002, Murphy et al. 2010). This fact raises the questions of (1) which areas are more prone to invasion post-disturbance and (2) to what extent invasive species will continue to expand and take over areas of native vegetation if no preventive measures are taken. Future work could experimentally manipulate sizes, types, and extents of disturbances while assessing invasive (and native) vegetation response.

Conclusion

We used historical airborne multispectral photos and remote sensing techniques to document the potential replacement of desirable native wetland vegetation with invasive species, and we evaluated the correlation between management objectives and vegetation response at the Bear River Migratory Bird Refuge, Utah. Results suggest that native *S. maritimus* was replaced by *Phragmites* and *Typha*, with changes concentrated along the water delivery canals to wetland units. Historical records show that management actions maintained a low area of invasive vegetation. The BRMBR managers and other wetlands managers can use remote sensing techniques and historical vegetation data to (1) better monitor where invasive plant species are increasing in cover so that they can focus management in those areas and understand which native species are most susceptible to replacement by invasive species; (2) determine how vegetation responds to management actions; and (3) better understand how to manage for a diversity of wetland plant species with high wildlife value. Such monitoring will allow managers to track trends and focus on the most at-risk areas.

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