Articles

Local and Landscape Habitat Associations of Shorebirds in Wetlands of the Sacramento Valley of California

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Abstract

The Sacramento Valley of California is a site of international importance for shorebirds despite having lost >90% of its historic wetlands. Currently both managed wetlands and flooded agriculture are important habitats for shorebird populations, but the extent of flooded agriculture may be declining in early winter when shorebirds need to acquire resources postmigration to survive winter. We employed long-term shorebird monitoring data to evaluate factors influencing abundance and species richness of shorebirds using the Sacramento National Wildlife Refuge Complex in early winter (November–December) between 2000 and 2009. We quantified the effect of local attributes of the wetland management unit (wetland type, size, and topography) as well as factors in the surrounding landscape (proportion of surface water and housing density) using generalized linear mixed models. We assessed a local-scale model, including covariates representing the area of six wetland types within the management unit, an index to the proportion of the management unit that had a tapered-edge (i.e., topography where flooded areas grade to exposed shoreline then upland), and a year effect. In this local-scale model, shorebird abundance had a significant positive association with the area of seasonally flooded marsh (SFM) and summer water. Topographical variation, characterized by the amount of tapered-edge, also had a significant positive effect on the abundance of shorebirds and species richness. Because >70% of the shorebirds were counted in SFM, we removed all wetland types except SFM to evaluate landscape covariates. Using only SFM-dominated units, there was a significant nonlinear association with the area of SFM within a management unit, with 40–95-ha wetlands having the highest shorebird abundance and species richness. On a landscape scale, the amount of flooding within a 10-km buffer was the best supported model of shorebird abundance and suggested the highest shorebird abundance in a management unit to be expected when 15–45% of the surrounding landscape was flooded. Species richness was positively associated with the proportion of surface water within 2- and 5-km buffers. We identified zones with a predicted high shorebird response to SFM, and assessed that only 6% of potential wetland areas in those zones have permanent conservation status. Our analyses suggest that shorebird abundance and species richness vary nonlinearly as a function of both local and landscape factors, and thus both spatial scales should be considered when developing conservation and management strategies.

Keywords: conservation; landscape; shorebirds; wetland management

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Introduction

Shorebirds are some of the most highly migratory birds in the world and rely on a network of wetland habitats distributed across broad landscapes. Many shorebird populations in the Western Hemisphere are declining and both local-level habitat management and broader scale conservation actions to generate new habitat are used to support shorebird populations (Brown et al. 2001). Given their wide-ranging ecology, shorebirds often respond to habitat features at multiple spatial scales (Elphick and Oring 1998; Niemuth et al. 2006; Taft and Haig 2006; Elphick 2008), which can make decisions on how to allocate conservation and management resources challenging. To develop management and conservation strategies for shorebirds that balance limited resources for local-scale management and landscape-scale conservation requires an understanding of the factors, from both within an individual wetland and from the surrounding landscape, influencing shorebird distribution and abundance.

The Sacramento Valley of California, which lies in the northern Central Valley, is a landscape of international importance for shorebirds in the Western Hemisphere (www.whsrn.org). More than 500,000 shorebirds use the Sacramento Valley annually, with approximately 200,000 shorebirds occurring there during the winter (Shuford et al. 1998). This is despite the fact that >90% of the natural wetlands that historically occurred have been lost, primarily to agriculture and urbanization (Frayer et al. 1989). However, since 1990, >8,000 ha of a total 30,000 ha of managed wetlands (largely anthropogenic restorations of previously existing wetland water, vegetation, and wildlife regimes) have been created throughout the Sacramento Valley by federal and state agencies, nongovernmental organizations, and private individuals (CVJV 2006). Additionally, in 1991, the state of California enacted a law that reduced the acreage farmers could burn to clear their fields of residual postharvest rice stubble (Rice Straw Burning Act, AB 1378 1991). This law resulted in increased postharvest flooding as an alternative strategy to decompose residual rice stubble (Fleskes et al. 2000; Miller et al. 2010). Currently, both managed wetlands and flooded agriculture (particularly postharvest flooded rice fields) are important for wintering shorebird populations (Elphick and Oring 1998; Shuford et al. 1998; Elphick 2000) and are relied upon to meet habitat conservation objectives for shorebirds set by the Central Valley Joint Venture (CVJV 2006)—a partnership initially established under the North American Waterfowl Management Plan to help conserve Central Valley waterfowl populations and habitats.

Though research has documented the importance of postharvest flooded rice as a resource for nonbreeding shorebirds in the Sacramento Valley (Day and Colwell 1998; Elphick and Oring 1998), considerably less attention has been directed at the role of managed wetlands to support shorebirds (however, see Shuford et al. 1998). There are likely many factors that influence shorebird distribution and abundance among wetlands, including water management (Taft et al. 2002), water depth (Colwell and Taft 2000; Strum et al. 2013), salinity (Warnock et al. 2002), and average distance to vegetation (Isola et al. 2000). Previous research completed on waterbirds and wetland management in the San Joaquin Basin, south of the Sacramento Valley, suggests wetland size, water depth, and water depth variation are significant factors influencing shorebird use of a managed wetland (Colwell and Taft 2000; Isola et al. 2000). We hypothesized that similar local-scale factors may be important for wetland use by shorebirds in the Sacramento Valley, a region where associations have not been evaluated. Moreover, few studies have evaluated shorebird use of different types of wetlands (e.g., seasonally flooded marsh [SFM] vs. vernal pool, etc.) to help determine which may be best suited to support shorebirds.

Landscape context, particularly the amount of flooded wetland habitat, has been shown to be important to predicting shorebird abundance in wetland–agriculture mosaics (Taft and Haig 2006; Elphick 2008). These studies suggest that there is a relationship between shorebird habitat use locally and the total amount of habitat across a broader landscape. Whether similar associations exist between shorebird use of wetlands in the Sacramento Valley and the extent of flooded habitat on the surrounding landscape is not known. Understanding the influence of landscape context on wetland use by shorebirds is needed to identify high-priority areas for wetland restoration. This is especially true given the dynamic availability of flooded habitat (Reiter and Liu 2011) and the continuing urbanization of the largely rural landscape of the Sacramento Valley (Theobald 2005), which may limit habitat availability for shorebirds.

The early winter (November–December) is an important period in the shorebird annual cycle because they are completing their southward migration and typically seek to acquire fat reserves to survive the winter (Pienkowski et al. 1979; Johnson et al.1989; Piersma and Jukema 2002). Although the current availability of flooded habitat in early winter (November–December) in the Sacramento Valley is substantial (>100,000 ha; Miller et al. 2010; Reiter and Liu 2011), several factors, including changing postharvest management strategies for rice (specifically declines in winter flooding), climate change (Snyder et al. 2004; Flint et al. 2013), urbanization (Bierwagen et al. 2010), land conversion from non-irrigated and irrigated rangeland to orchards and vineyards (Holland 2011), and competition for water, will lead to reductions in early winter surface-water habitat. The impacts of agricultural wetland habitat loss may be particularly problematic during this time period when there has traditionally been a rapid increase in shorebird habitat as the rice lands are flooded postharvest. During recent drought years, there have been declines and delays in postharvest flooded rice due to water policy, specifically Term 91 (California Water Code Section 85230(d)), which protects water quality and can curtail water allocations. Given the uncertainty of water resources in the face of climate change and potentially extreme drought, postharvest flooding of rice may be further reduced in the future, which would affect habitat availability on the landscape. Consequently, it is critical to understand the factors associated with shorebird use of managed wetlands during early winter to improve conservation and management decision-making.

We used shorebird survey data from the Sacramento National Wildlife Refuge Complex (SNWRC) between 2000 and 2009 to better understand the influence of local and landscape factors on shorebird use of managed wetlands in early winter (November–December). The SNWRC is an important wintering site for waterfowl (Anseriformes), waterbirds (Ciconiiformes), and shorebirds (Charadriiformes) in the Sacramento Valley (Gilmer et al. 1996; Shuford et al. 1998), and is particularly important when postharvest flooded rice is not available as a food resource. Using these data, we addressed the following research questions: Is shorebird abundance and species richness in a wetland associated with 1) the wetland type? 2) the size and topographical variation of the wetland? 3) the amount of flooded habitat or urban development in the surrounding landscape and at what spatial scale? We selected variables to include in our analysis a priori based on previous literature and plausible hypotheses of factors influencing shorebirds (see Methods). Given the retrospective approach in this study, we were unable to consider several variables known to influence shorebirds (e.g., water depth, vegetation). However, our selection of covariates is a relevant set for shorebirds that can help guide local and landscape wetland management and conservation. In particular, we applied the results of our analyses in a spatial prediction framework to identify areas and strategies for wetland restoration to promote shorebird use during early winter in the Sacramento Valley of California.

Study Site

The Sacramento Valley extends approximately 180 km from Red Bluff in the north to the city of Sacramento in the south. The Sacramento Valley is largely rural, although urbanization (housing density) has increased over the previous 20 y, and is expected to increase even further (Theobald 2005; Bierwagen et al. 2010). There are >200,000 ha of rice grown annually in the Sacramento Valley of which >100,000 ha are flooded postharvest for stubble decomposition and waterfowl hunting (CVJV 2006; Miller et al. 2010). There are an additional 30,000 ha of managed wetlands on federal, state, and private lands (CVJV 2006). Few unmanaged, natural wetlands exist in the Sacramento Valley and are mostly vernal pools and a few seminatural sloughs. Our study site included six areas (hereafter, refuges) within the SNWRC: Sacramento, Delevan, Colusa, and Sutter National Wildlife Refuges; and the Butte Sink and Llano Seco Units, which are part of the Butte Sink and North Central Valley Wildlife Management Areas, respectively (Figure 1). The SNWRC consists of 11,000 ha of managed wetland and upland habitats, distributed throughout approximately 210,000 ha of the Sacramento Valley. We distinguished managed wetlands, as opposed to unmanaged wetlands, as those that have the capability of being flooded and drained through water control structures. Within SNWRC, about 66% of the habitat is SFM. The remaining wetland types typically each comprise ≤15% of SNWRC land. Wetland types for our study were defined by the timing and duration of flooding, as well as the structure (height and extent) and composition (common species) of vegetation cover (Table 1; USFWS 2009). All wetlands within SNWRC were generally classified as palustrine per Cowardin et al. (1979). On the broader landscape, SNWRC is surrounded by extensive rice agriculture and privately owned managed wetlands (Figure 1).

Methods

The SNWRC conducted wildlife surveys of all management units twice per month during November and December 2000–2009. During each survey, observers drove standardized survey routes and all shorebirds observed were counted and recorded by species within each management unit. Management units were defined as areas with common levees and water control structures that can be used to manage for desired characteristics. Management units varied in size from 1 to 243 ha (mean = 29; SD = 26). Observers used binoculars and spotting scopes to identify all birds to species. There was not a specified time limit to survey a management unit, but efforts were made to complete surveys rapidly to limit bird movement. Surveys of all management units on an individual refuge were completed on the same day. Additional data tracked for each management unit included the total area (ha) of each wetland type within the management unit (Table 1), the percent of the management unit that was flooded, and the total management unit area (ha).

For each species, we summarized the total birds counted, the proportion of the total birds counted of all species it represented, the coefficient of variation (SD/mean) of the count per management unit, and the probability of >1 bird occurring to better understand the distribution of the data used in our analysis. We employed mixed-effects Poisson regression to model our data and included random effects to account for overdispersion and correlation within management units, refuges, and years (Gelman and Hill 2007). We considered the total number of shorebirds of all species observed in each management unit on each survey (abundance) and the total number of shorebird species observed in each management unit on each survey (species richness) as response variables. We evaluated local-scale management unit models and combined models with local- and landscape-scale covariates for both abundance and richness. Our models of abundance weighted the more abundant species higher in our analyses, whereas our assessment of species richness provided increased weight for species that were less abundant but occurred more frequently. However, our application of random effects also down-weighted rare, large flocks of shorebirds that were observed for some species (e.g., 6,000 dowitcher spp. in one observation), limiting the influence of these outlier observations on our inference (Gelman and Hill 2007).

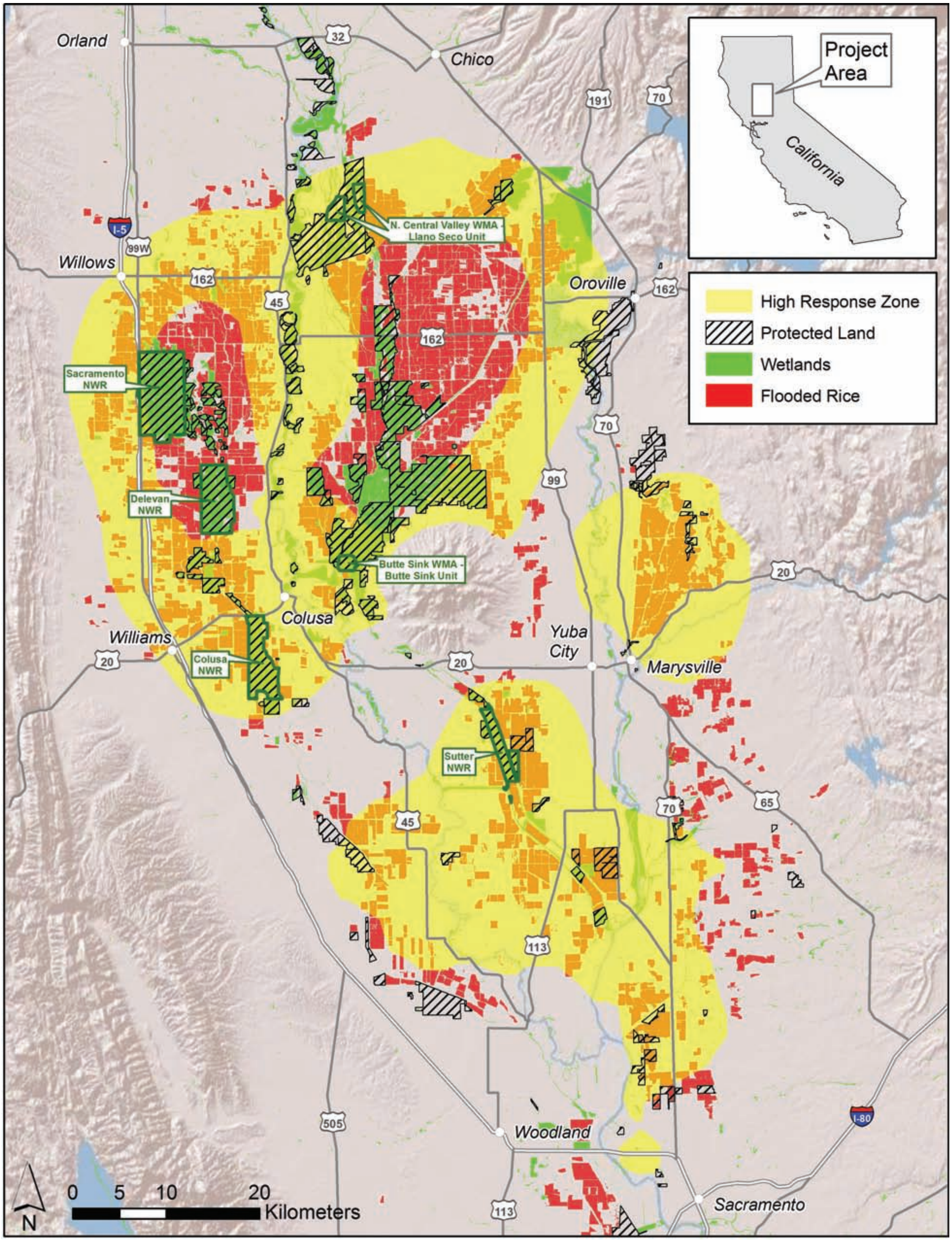

Figure 1. Distribution of Sacramento Valley National Wildlife Refuge Complex study areas, flooded rice and wetlands, and the zones where wetland creation is predicted to generate high shorebird response in the Sacramento Valley, California. High response zones are areas where 15–45% of the surrounding landscape, on average, is flooded during November and December.

Table 1. Six focal wetland types considered in analysis of shorebird abundance and species richness at Sacramento National Wildlife Refuge Complex, California, 2000–2009. Descriptions modified from USFWS (2009) except where indicated.

Wetland type	P[a]	Inundation period	Water depth	Plant species, composition, and height
Seasonally flooded marsh	0.62	Managed freshwater wetland[b] that is flooded September through mid-April.	Shallow (<30 cm) with small portions of some units up to 90 cm	At least 50% of area contains vegetation ≤15 cm, such as swamp timothy *Crypsis schoenoides* and pricklegrass *Crypsis vaginiflora*. Remaining vegetation varies from 15 to 180 cm, including Bermuda grass *Cynodon dactylon,* jointgrass *Paspalum distichum,* alkali bulrush *Bolboschoenus maritimus,* tuberous bulrush *Bolboschoenus glaucus,* cocklebur *Xanthium* sp., smartweed *Polygonum* sp., cattail *Typha* sp., and hardstem bulrush *Schoenoplectus acutus*.
Watergrass	0.09	Managed freshwater wetland that is flooded September through May. Irrigated for 1–3 wk mid-May to mid-July.	Shallow (<30 cm) with small portions of some units up to 90 cm	At least 90% of area contains vegetation ≥60 cm and is dominated by watergrass *Echinochloa crus-galli* and smartweed, with small patches of Bermuda grass and jointgrass.
Vernal pool–alkali meadow complex	0.12	Naturally occurring wetland basins that flood and dry naturally through precipitation or flood events.	Shallow (2.5–30 cm)	At least 95% of the natural vegetation is <15 cm and best characterized as saline vernal pools of the Colusa-Solano Region (Barbour et al. 2003) and alkali meadow plant communities (Griggs et al. 1992; Silveira 2000).
Summer water (semipermanent)	0.05	Managed freshwater wetland that is flooded October through late July.	30–120 cm, maintained at consistent levels in summer, but often shallower during winter	At least 50% of area contains emergent vegetation ≥90 cm, including cattail, hardstem bulrush, alkali bulrush, and tuberous bulrush. Floating-leaved emergent and submergent vegetation includes burhead *Echinodorus cordifolius,* arrowhead *Sagittaria* sp., water primrose *Ludwigia* sp., sago pondweed *Stuckenia pectinata,* and horned pondweed *Zannichellia palustris*.
Permanent pond	0.02	Managed freshwater wetland that is flooded year-round	60–120 cm, maintained at consistent levels in summer, but often shallower during winter	At least 50% of area contains emergent vegetation ≥90 cm, including cattail, hardstem bulrush, alkali bulrush, and tuberous bulrush. Floating-leaved emergent and submergent vegetation includes burhead, arrowhead, water primrose, sago pondweed, and horned pondweed.
Unmanaged freshwater wetland	0.01	Wetlands, other than vernal pools, that have little or no artificial water management capabilities. Flooding is highly variable.	Variable	Variable

[a] P = the proportion of total area of all surveyed wetland types across all Sacramento National Wildlife Refuges that was composed of each wetland type indicated. The remaining 10% of the survey areas were composed of wetland types that are not considered suitable for wetland-dependent shorebirds.
[b] Managed freshwater wetlands = wetlands that are largely artificial in nature, having the capability of being flooded and drained through water control structures. Most of these wetlands impound water entirely or partially within levees that are maintained.

At the management unit scale we assessed two covariate categories. First, we modeled the effect of the area (ha) of each of the six wetland types within a management unit that comprised ≥1% of the surveyed area across SNWRC and could plausibly serve as wetland habitat for shorebirds (Table 1). We are aware of no other study that has evaluated the relative importance of this combination of wetland types for shorebirds, all of which are relatively common across the Sacramento Valley. We hypothesized that wetland types with less vegetation and that were flooded during the time of the surveys would be more likely to be associated with shorebirds (Ma et al. 2010).

Second, previous studies have indicated that the amount of topographical and depth variation can influence both the abundance and diversity of waterbirds using a managed wetland (Colwell and Taft 2000; Taft et al. 2002; see Ma et al. 2010 for review). To capture this variability, we included a variable to index the amount of tapered-edge or sloped-edge (hereafter, tapered-edge) for each management unit. Tapered-edge characterized wetland topography and identified a zone where wetlands grade gradually to exposed shoreline and then upland and where water is not usually confined by constructed levees. We used the following scale for our index based on the percent of the wetland perimeter: 0 = 0% tapered-edge; 1 = 1–10% tapered-edge; 2 = 11–50% tapered-edge; and 3 = 51–100% tapered-edge. These values were derived using Geographic Information Systems (GIS), aerial photos, and expert knowledge of the units once for the entire study time period because we believe the index did not change. Management units with a low tapered-edge index had levees on most or all of the wetland boundaries, whereas management units with a larger tapered-edge index had a greater proportion of wetland boundaries that transitioned into natural topography toward upland areas before a levee. Though year-to-year variation in flooding could influence the availability of tapered-edges, our data come from a consistent time period each year resulting in little year-to-year variation in the percent of a wetland unit that was flooded. We considered tapered-edge to be a factor in our models and hypothesized that management units with higher tapered-edge index values would be correlated with a greater diversity of shallow water depths, and therefore have relatively greater shorebird use and higher species richness. We also included a year-effect variable (Year) to account for temporal changes in abundance and richness.

Based on preliminary summaries, >70% of the shorebirds occurred in SFM during early winter, and >60% of the total area surveyed was SFM (Table 1). Seasonally flooded marsh is also the most common type of restored wetland in this region and is considered, along with flooded rice, in bioenergetics models for shorebirds in this region (CVJV 2006). We therefore removed all management units that did not have >80% of their area as SFM (hereafter, SFM-dominated units or SFM-units) from the landscape-scale analysis.

At a landscape scale, previous studies suggest that the amount of flooded cover (Taft and Haig 2006; Elphick 2008) and the extent of urbanization (Zhenming et al. 2006; LeDee et al. 2008) may influence shorebirds positively and negatively, respectively. We quantified the effect of these two landscape features using existing GIS data. We chose to follow Elphick (2008), who evaluated 2-, 5-, and 10-km buffers around rice fields to assess landscape effects on waterbird use in this region, but we also included a 20-km buffer. We calculated the proportion of the landscape that was flooded within each buffer surrounding the management unit (Table 2). We derived these data for each year 2000–2009 from Landsat images that were classified to delineate flooded and nonflooded areas during early winter to assess long-term availability of shorebird habitat (Reiter and Liu 2011). We did not distinguish between the types of land cover that were flooded (e.g., rice vs. wetland) because we did not have a time-series of land-cover types comparable to our water data across the landscape. The largest percentage of flooded cover was postharvest rice (Reiter and Liu 2011). We evaluated the effects of urbanization using the mean housing density (units/ha) within 2-, 5-, 10- and 20-km buffers surrounding the management unit (Table 2). We calculated the mean housing density within each buffer distance using a GIS layer developed by the Environmental Protection Agency (Bierwagen et al. 2010).

Based on the results of Elphick (2008) and Taft and Haig (2006), we hypothesized that smaller scale relationships (i.e., 2-km buffer) would be more important for shorebirds than would larger scale with regard to the amount of flooding, and that shorebirds would be positively associated with the amount of flooding on the surrounding landscape. Further we hypothesized that there could be an optimal range of values at which the proportion of the landscape that is flooded may influence shorebird abundance in wetlands, and thus we included linear and quadratic forms of the landscape flooding variables at each of the four buffer distances. We had no information with which to generate hypotheses regarding the effective scale of urbanization, though we hypothesized that overall there would be a negative linear association between shorebird abundance and richness and the amount of urbanization (LeDee et al. 2008). We used ArcMap 9.3.1 (© 2010 ESRI, Inc.) and the Geospatial Modeling Environment extension (©2009–2012 Spatial Ecology LLC) to derive spatial covariates.

We developed 32 models a priori to evaluate associations of shorebird abundance and richness with landscape covariates using data from the SFM-units in a model selection framework (Burnham and Anderson 2002). All models, except for the intercept-only model, included local variables: area (ha) of SFM within the management unit; a tapered-edge index; a year effect (trend); and random effects of year, refuge, and management unit. To reduce spatial autocorrelation, we did not include covariates from different spatial buffers in the same models. We hypothesized that shorebird abundance and species richness may asymptote at large values of SFM (i.e., after a certain size the added area of habitat does not increase shorebird abundance or richness); thus, we evaluated both a linear or quadratic form of this covariate.

To facilitate model convergence, we rescaled wetland type covariates by dividing area values (ha) by 1,000, and we rescaled Year to the mean of Year equal to 0. The remaining variables were already scaled between 0 and 1 because they represented proportions or very small values (e.g., housing density; Table 2). We ranked competing landscape-scale models using Akaike's Information Criterion corrected for small sample sizes (AIC_c; Burnham and Anderson 2002). We determined models within two AIC_c units of the top model to be part of the competing model set and evaluated relative support for each model and model selection uncertainty with Akaike weights (w_i). We considered 95% confidence intervals (CI) of model parameter estimates that did not overlap zero to be significant. Post hoc, we evaluated a model for shorebird abundance and species richness with the SFM-unit data only and none of the landscape variables to better understand the contribution of landscape effects on SFM-units once local-level variation was controlled. We examined residual plots for evidence of lack of fit as well as residual spatial and temporal correlation in our models. We calculated the percent of variance explained by the fixed-effects in our models following the methods of Nakagawa and Schielzeth (2013).

We illustrated the modeled relationships and the strength of the association of covariates with shorebird abundance and species richness by plotting the marginal multiplicative effect of covariates from the best-supported models. The marginal multiplicative effect represents how many times larger, on average, the count of birds or species is expected to be for each level of a plotted covariate, given all other covariates are fixed as constants. Marginal multiplicative effect values <1 indicated a negative effect of the selected covariate on the expected abundance or species richness. We plotted the 95%

Table 2. Definition and summary of variables used in models of shorebird abundance and species richness in seasonally flooded marsh at the Sacramento National Wildlife Refuge Complex, California, 2000–2009. Distribution of quadratic form of variables not presented. All variable values summarized per management unit.

Category	Variable (alias)	Values
Habitat—local	Hectares of seasonally flooded marsh ha (SFM)	Mean = 24
	Quadratic term (SFM2)	Min. = 0, Max. = 160
	Tapered-edge (TE) index	0 = 0%; 1 = 1–10%; 2 = 11–50%; 3 = 51–100%
Habitat—landscape		
Flooding	Proportion flooded <2 km of a management unit (FLD2k) Quadratic term (FLD2k2)	Mean = 0.47 Min. = 0, Max. = 0.79
	Proportion flooded <5 km of a management unit (FLD5k) Quadratic term (FLD5k2)	Mean = 0.39 Min. = 0.10, Max. = 0.66
	Proportion flooded <10 km of a management unit (FLD10k) Quadratic term (FLD10k2)	Mean = 0.30 Min. = 0.09, Max. = 0.54
	Proportion flooded <20 km of a management unit (FLD20k) Quadratic term (e.g., FLD10k2)	Mean = 0.21 Min. = 0.09, Max. = 0.35
Urbanization	Houses per m^2 <2 km of a management unit (URB2k)	Mean = 0.0001 Min. = 0, Max. = 0.002
	Houses per m^2 <5 km of a management unit (URB5k)	Mean = 0.0001 Min. = 0.00001, Max. = 0.003
	Houses per m^2 <10 km of a management unit (URB10k)	Mean = 0.0001 Min. = 0.00002, Max. = 0.0002
	Houses per m^2 <20 km of a management unit (URB20k)	Mean = 0.00004 Min. = 0.00002, Max. = 0.0002
Trend	Year	2000–2009
Random effect		
	Management Unit	238 units
	Refuge	6 refuges
	Year	10 y

confidence envelope of the marginal multiplicative effect to highlight uncertainty in our modeled associations. We conducted all analyses using the statistical program R v.14.1 (© The R Foundation for Statistical Computing) and fit all models using the lme4 package (Bates et al. 2012).

We applied the best-supported models based on AIC_c to assess local and landscape capacity to maximize shorebird habitat in wetlands. First, we plotted the observed distribution of SFM-unit sizes (ha) compared with the effect of the area (ha) of SFM on shorebird abundance to identify whether the management unit sizes at the SNWRC were maximizing the potential for shorebird use. Next, we employed the best supported model of shorebird abundance, the spatial distribution of covariates from that model, and ArcMap 9.3.1 to predict locations where SFM-units would promote the largest number of shorebirds during early winter in the Sacramento Valley (''high response zone''). We then used the California Central Valley Joint Venture Protected Lands 2009 GIS layer (©Ducks Unlimited 2009) to identify permanently protected lands, and determined what fraction of the predicted high shorebird response zones already had permanent conservation status.

Results

We used data from surveys completed between 15 November and 17 December after selecting those observations that were closest in time each year to the date that the water availability data were obtained by the Landsat satellite. Across all wetland types between 2000 and 2009, there were 2,838 unit surveys of 303 management units, representing 80,503 ha. A total of 34,266 shorebirds representing 12 species were counted (see Table S1 and S2, *Supplemental Material,* for source data and metadata). Dowitcher spp. (primarily long-billed *Limnodromus scolopaceus,* but also some short-billed *Limnodromus griesus*) dominated the survey, accounting for 66% of the birds counted. Black-necked stilt *Himantopus mexicanus*, long-billed curlew *Numenius americanus*, greater yellowlegs *Tringa melanoleuca*, dunlin *Calidris alpina,* and western *Calidris mauri* and least *Calidris minutilla* sandpipers combined accounted for an additional 30% of all shorebirds observed. Killdeer *Charadrius vociferus,* American avocet *Recurvirostra americana,* lesser yellowlegs *Tringa flavipes,* and Wilson's snipe *Gallinago gallinago* were also observed (Table 3). There was high variation in the average count per management unit within and among species. Furthermore, although dowitcher spp. was most abundant, black-necked stilt, greater yellowlegs, and killdeer were the most frequently occurring species (Table 3). Using just the SFM-unit data from 2000 to 2009, there were 1,767 unit surveys of 238 management units, representing 49,076 ha. We counted 24,962 shorebirds (73% of total; 0.51 shorebirds per ha) and observed all 12 species in SFM-units that were seen

Table 3. Summary of shorebird counts by species on management units at the Sacramento National Wildlife Refuge Complex, California, in November and December 2000–2009.

Species	Total[a]	Percent of total[b]	CV[c]	Occurrence[d]
Killdeer *Charadrius vociferus*	658	2	19	0.056
American avocet *Recurvirostra americana*	5	<1	44	0.001
Black-necked stilt *Himantopus mexicanus*	3,137	10	6	0.053
Greater yellowlegs *Tringa melanoleuca*	158	<1	9	0.021
Lesser yellowlegs *Tringa flavipes*	1	<1	53	0.000
Long-billed curlew *Numenius americanus*	806	2	18	0.012
Long-billed or short-billed dowitcher *Limnodromus scolopaceus* or *griesus*	22,938	66	18	0.018
Dunlin *Calidris alpina*	1,088	3	16	0.011
Western sandpiper *Calidris mauri*	34	<1	47	0.001
Least sandpiper *Calidris minutilla*	124	<1	22	0.004
Western or least sandpiper *Calidris* spp.	5,300	15	19	0.012
Common snipe *Gallinago gallinago*	17	<1	21	0.003

[a] Total = total birds counted over all surveys.
[b] Percent of Total = percent of all birds counted represented by each species.
[c] CV = coefficient of variation of the counts.
[d] Occurrence = probability of ≥1 bird occurring during survey of a management unit.

across all wetland types on the SNWRC (see Table S2 and S3, *Supplemental Material,* for source data and metadata).

Local-scale models that included effects of the wetland type area (ha), the tapered-edge index, and Year suggested there was a significant positive association of shorebird abundance with SFM (Table 4). In addition, the area of the summer water wetland type had a significant positive effect on shorebird abundance (Table 4). There was not a significant association of shorebird abundance with permanent pond, vernal pool–alkali meadow complexes, unmanaged freshwater wetlands, or watergrass wetland types (Table 4), though sample sizes were relatively low for these wetland types, which made parameter estimates imprecise. Through a comparison of the 95% CI of cover-type parameter estimates from this model, both SFM and summer water had a significantly larger effect on the expected abundance of shorebirds than did other wetland types (Table 4). The 95% CI of parameter estimates for summer water and SFM overlapped, indicating these wetland types were not significantly different from each other as measured by shorebird abundance. The amount of tapered-edge had a significant positive effect on shorebird abundance, with abundance increasing for each level of the factor representing increasing tapered-edge. The Year effect was significant and suggested an increase in shorebird abundance over the past 10 y (Table 4). Species richness was significantly associated with the SFM wetland type and the tapered-edge index at the management unit scale (Table 4), but had no significant temporal trend.

Of the 32 landscape-scale models evaluated using data from only SFM-units, one model was substantially better than the others evaluated and received 100% of the Akaike weight when describing shorebird abundance (Table 5; see Table S4, *Supplemental Material,* for

Table 4. Parameter estimates and 95% confidence intervals for shorebird abundance and species richness models using data from all cover types at the Sacramento National Wildlife Refuge Complex, California, in November and December, 2000–2009. Confidence intervals that do not overlap zero (in bold) were considered significant.

	Abundance			Richness		
Parameter	Estimate	95% Lower	95% Upper	Estimate	95% Lower	95% Upper
Intercept	**−5.87**	**−7.39**	**−4.35**	**−4.56**	**−5.18**	**−3.94**
Seasonally flooded marsh	**66.60**	**46.60**	**86.61**	**25.49**	**15.58**	**35.40**
Vernal pool–alkali meadow	14.28	−1.09	29.68	11.23	−19.93	42.40
Permanent pond	−15.03	−68.18	38.13	−25.80	−124.24	72.63
Summer water	**71.07**	**51.10**	**91.10**	13.27	−7.47	34.00
Watergrass	4.62	−23.97	33.21	−35.17	−83.72	13.38
Unmanaged freshwater wetland	177.16	−46.85	401.18	67.47	−47.09	182.04
Tapered edge 1 (1–10% of unit perimeter)	**1.75**	**0.57**	**2.92**	**1.18**	**0.61**	**1.76**
Tapered edge 2 (11–50% of unit perimeter)	**3.85**	**2.32**	**5.37**	**1.90**	**1.14**	**2.65**
Tapered edge 3 (51–100% of unit perimeter)	**4.61**	**2.75**	**6.47**	**2.00**	**1.05**	**2.94**
Year	**0.23**	**0.04**	**0.43**	0.04	−0.04	0.11

Table 5. Summary of model selection results for models of landscape factors influencing the abundance and species richness of shorebirds in seasonally flooded marsh in early winter on the Sacramento National Wildlife Refuge Complex, California, 2000–2009. In addition to landscape variables specified below, all models, except Intercept only, included an intercept and the local covariates (quadratic form of the area of seasonally flooded marsh; tapered-edge index; year; and random effects for management unit, refuge, and year). We present landscape models with the lowest AIC_c or <2 AIC_c units of the top model, with the exception of the Intercept only model and the Intercept plus local covariates model (see Tables S4 and S5, *Supplemental Material,* for full model results).

Response	Model	ΔAIC_c[a]	w_i[b]	K[c]
Abundance	FLD10k[d] + FLD10k2[e]	0.00	1.00	11
	Intercept + local covariates	5,428.38	0.00	9
	Intercept only	5,540.00	0.00	4
Richness	FLD5k[f]	0.00	0.26	10
	FLD5k + FLD5k2[g]	0.52	0.20	11
	FLD2k[h]	1.13	0.15	10
	Intercept + local covariates	2.71	0.07	9
	Intercept only	71.88	0.00	4

[a] ΔAIC_c = the difference between each model in a model set and the model in that set with the lowest AIC_c.
[b] w_i = Akaike weight (the probability that the model is the best given the model set and the data).
[c] K = the number of parameters in the model.
[d] FLD10k = proportion flooded within 10-km buffer.
[e] FLD10k2 = quadratic term of proportion flooded within 10-km buffer.
[f] FLD5k = proportion flooded within 5-km buffer.
[g] FLD5k2 = quadratic term of proportion flooded within 5-km buffer.
[h] FLD2k = proportion flooded within 2-km buffer.

complete model list and ranks). The best supported model included the quadratic effect of the area (ha) of SFM, the tapered-edge index, a quadratic effect of the proportion of flooded area within 10 km of the management unit, and Year (Table 5). The parameter estimates were all significantly different from zero, except for Year (Table 6). The second-best supported model (though $>$2,000 AIC_c units from the top model) included the quadratic effect of the proportion of the area that was flooded within 5 km of the management unit. Urbanization was not included in any of the top 10 models of abundance and had a larger AIC_c than a model with the local covariates only (Table S4, *Supplemental Material*).

The area of SFM within a management unit with the highest expected abundance of shorebirds was between 40 and 95 ha (Figure 2). Management units of this size had a 1,000 to 3,500 times greater expected count of shorebirds than in units outside of this interval. The predicted abundance of shorebirds generally increased with the overall area of SFM within the management unit, though rapid increases did not occur until SFM was close to 35 or 40 ha. Shorebird abundance started to decrease when the overall area of SFM within a management unit exceeded approximately 95 ha (Figure 2). On account of the low sample size of large management units ($>$95 ha), there was considerable uncertainty in our identification of an upper threshold of management unit size as indicated by the width of 95% confidence envelope. The quadratic effect of the proportion of the landscape that was flooded also identified a threshold, with shorebird abundance in SFM-units being the greatest when 15–40% of the landscape was flooded within 10 km (Figure 3). Within this range, shorebird abundance was expected to be 250–1,250 times greater than similar management units located in a landscape with the amount of flooding within 10 km outside of the 15–40% interval. Overall, models of shorebird abundance including landscape-level covariates were a substantial improvement over models with local-level covariates only (Table 5).

There were two best supported models ($\Delta AIC_c \leq 2$) for species richness within SFM-units, and both included the quadratic parameterization of the amount of SFM (Table 5). The effect of SFM was statistically significant (Table 6) and species richness was predicted to peak in management units with approximately 100 ha of SFM (Figure 2). In contrast to abundance models, the effect of landscape variables in the best supported species-richness models were linear rather than quadratic and included the proportion of water within 2 km and 5 km (Table 5). The effect of both landscape variables was significant (Table 6), and indicated that increasing species richness is associated with an increasing proportion of flooded habitat on the surrounding landscape (Figure 3). Overall, there was substantial uncertainty in the landscape models of species richness as the top two models received only a combined 41% of the Akaike weight (Table 4; see Table S5, *Supplemental Material,* for complete richness model list and ranks). An urbanization covariate (URB2k) model was just outside the top model set (ΔAIC_c = 2.66; Table S5, *Supplemental Material*), though the 95% CI for the parameter estimate overlapped zero, which suggested no significant association of species richness with urbanization. Overall, landscape-level models of species richness were only a moderate improvement (ΔAIC_c = 2.66) compared with models with only local-level covariates included (Table 5).

In both abundance and richness analyses, the top models were an improvement over intercept-only models (Table 5). Residual variation, as specified by the random effects, was still highest among wetland units compared with among years or refuges (Table 6). This result highlights the challenge of predicting highly clustered and abundant shorebirds (particularly dowitcher spp.) and potentially the need to consider additional management-unit-level covariates. Overall, our fixed effects in the best supported abundance model explained 32% of the variation in the data, whereas the best supported species richness model explained 25%.

Our models indicated that relatively few (28 of 238) of the SNWRC's SFM-units between 2000 and 2009 fell within the optimal size range to maximize shorebird abundance in the early winter (Figure 4); the majority of the SFM-units were too small. The best supported abundance model predicted several areas in the Sacramento Valley landscape for wetland restoration or

Table 6. Parameter estimates (SE) for variables in best supported landscape-scale models of shorebird abundance (Abundance) and species richness (Richness) models for seasonally flooded marsh at the Sacramento National Wildlife Refuge Complex, California, in November and December, 2000–2009. Confidence intervals (95%) for all fixed effect parameters did not overlap zero and were considered significant except for Year in models of abundance.

Parameter		Abundance		Richness	
Fixed effect	SFM[a]	252.0 (30.4)	78.8 (15.5)	78.1 (15.4)	77.7 (15.8)
	SFM2[b]	−1,930.0 (227.0)	−454.2 (125.7)	−450.5 (125.4)	−449.5 (128.0)
	FLD2k[c]	—	—	—	1.6 (0.8)
	FLD5k[d]	—	3.6 (1.5)	−2.4 (5.0)	—
	FLD5k2[e]	—	—	7.9 (6.3)	—
	FLD10k[f]	50.0 (1.3)	—	—	—
	FLD10k2[g]	−90.0 (1.7)	—	—	—
	TE1[h]	2.4 (0.7)	1.3 (0.3)	1.3 (0.3)	1.2 (0.3)
	TE2[i]	6.3 (1.2)	3.0 (0.4)	2.9 (0.4)	2.9 (0.4)
	Year[j]	0.07 (0.1)	0.1 (0.05)	0.1 (0.05)	0.1 (0.04)
Random effect[k]	Refuge[l]	3.8	0.8	0.8	0.5
	Unit[m]	12.8	1.1	1.1	1.2
	YearV[n]	1.0	0.1	0.1	0.1

[a] SFM = seasonally flooded marsh.
[b] SFM2 = quadratic term of seasonally flooded marsh effect.
[c] FLD2k = proportion flooded within 2-km buffer.
[d] FLD5k = proportion flooded within 5-km buffer.
[e] FLD5k2 = quadratic term of proportion flooded within 5-km buffer.
[f] FLD10k = proportion flooded within 10-km buffer.
[g] FLD10k2 = quadratic term of proportion flooded within 10-km buffer.
[h] TE1 = tapered-edge index level 1.
[i] TE2 = tapered-edge index level 2.
[j] Year = continuous fixed-effect of year.
[k] Standard error of variance of random effects could not be estimated.
[l] Refuge = variance of random effect of refuge.
[m] Unit = variance of random effect of management unit.
[n] YearV = variance of random effect of year.

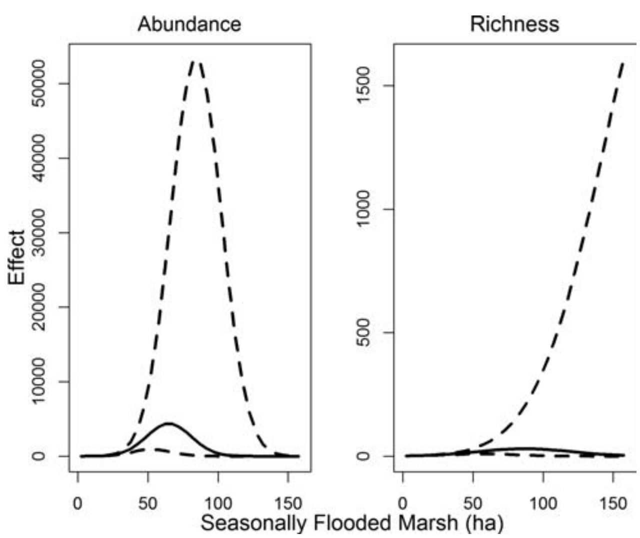

Figure 2. Marginal multiplicative effect (Effect) of the area (ha) of seasonally flooded marsh within a management unit on shorebird abundance (left) and species richness (right) at the Sacramento National Wildlife Refuge Complex (California) in November and December, 2000–2009, from the best supported model. Solid line is the mean estimate and dashed lines are the 95% confidence envelope. The marginal multiplicative effect represents how many times larger the count of birds (per ha) or species (per ha) is expected to be for each level of a plotted covariate, given all other covariates are fixed as constants; values <1 indicate a negative effect on abundance and richness.

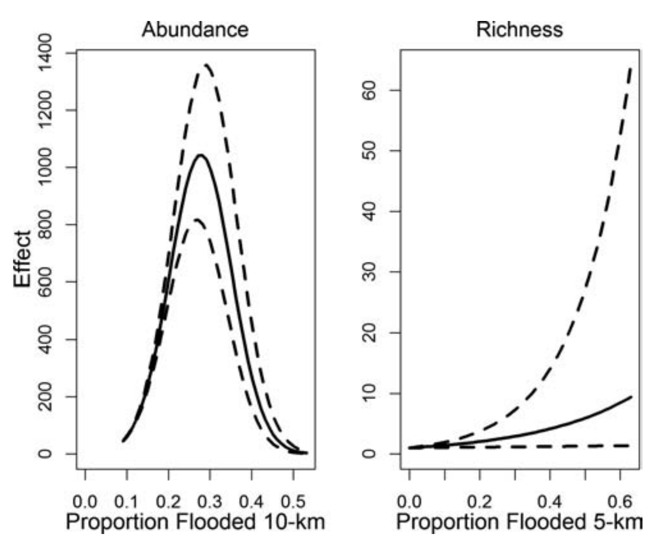

Figure 3. Marginal multiplicative effect (Effect) of landscape variables on shorebird abundance (left) and species richness (right) at the Sacramento National Wildlife Refuge Complex (California) in November and December, 2000–2009, from the best supported model. Solid line is the mean estimate and dashed lines are the 95% confidence envelope. The marginal multiplicative effect represents how many times larger the count of birds (per ha) or species (per ha) is expected to be for each level of a plotted covariate, given all other covariates are fixed as constants; values <1 indicate a negative effect on abundance and richness.

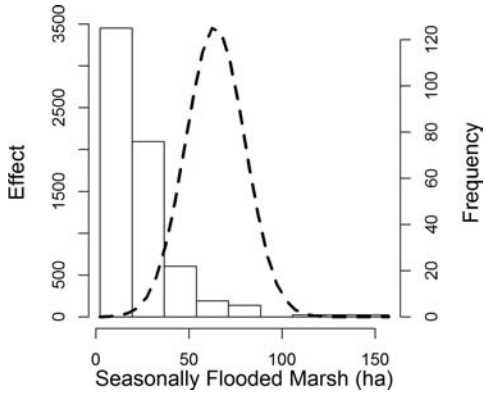

Figure 4. The distribution of the size (ha) of seasonally flooded marsh management units (Frequency; bars) at the Sacramento National Wildlife Refuge Complex (California) in November and December, 2000–2009, and the marginal multiplicative effect (Effect) of wetland size from the model (dashed line). The marginal multiplicative effect represents how many times larger the count of birds is expected to be for each level of a plotted covariate, given all other covariates are fixed as constants. The taller the dashed line, the more shorebirds are expected to occur.

enhancement that would likely maximize shorebird response under the average flooding conditions experienced 2000–2009 (Figure 1). The high response zones are locations where on average 15–40% of the surrounding landscape within 10 km of a 65 ha pixel (potential location for an optimally sized wetland) is flooded. High response zones for wetland establishment represented approximately 300,000 ha of land, of which only 6% currently has permanent conservation status, though there is also a significant amount of flooded rice (Figure 1).

Discussion

Managed wetlands are important habitat for shorebirds throughout North America and particularly in the Sacramento Valley of California. Our analysis of 10 y of early winter surveys, a period of time during which the availability of flooded habitat may become increasingly limited as drought years extend and water available for postharvest flooding of rice is reduced, identified both local- and landscape-level variables influencing shorebird abundance. These results were largely consistent with other studies (see Colwell and Taft 2000; Taft and Haig 2006; Elphick 2008). We provide guidance for wetland managers on wetland type, wetland size, and topography, and as well to conservation planners on where to position a wetland on the landscape to generate the greatest use by shorebirds in the Sacramento Valley during early winter. Overall, shorebirds were most strongly associated with SFM with high topographic variability during this time of year, although another much less common wetland type (summer water) also had densities significantly greater than the overall mean at the SNWRC. The spatial location of the wetland on the landscape relative to other flooded areas also significantly influenced shorebird abundance and species richness. Although, the marginal effect of local wetland size was relatively greater than the effect of landscape-level variables, models including covariates from both scales were selected over models with only local-level covariates, indicating the importance of landscape context.

Seasonally flooded (SFM) and semipermanent managed wetlands are assumed to have value for shorebirds in bioenergetics models in the Central Valley Joint Venture Implementation Plan (CVJV 2006), which is used to set conservation objectives for shorebirds. Our study confirms that SFM appears to be one of the most used habitats when considering wetland management for shorebird abundance and richness during early winter. This is not the first study to identify the relative value of SFM compared with other cover types (Shuford et al. 1998; Elphick 2000), though we evaluated it against a comparatively larger suite of wetland types, characterized by large variation in vegetation and water regimes. The summer water cover type also contained significant numbers of shorebirds. Over the past decade, the SNWRC has been increasing emphasis on managing shallow water habitat in semipermanent wetlands during winter months and providing late summer water that lasts into the early autumn (SNWRC, unpublished data), which likely resulted in the observed shorebird use of these cover types during the early winter. Shorebirds were not associated with watergrass, despite an inundation period and water depth regime similar to SFM (Table 1). The more abundant and taller vegetation characteristics of the watergrass likely resulted in lower use than the comparable but less vegetated SFM. Although vernal pool wetlands were shown to be important in other studies (Silveira 1998), shorebirds were not significantly associated with such pools, but we suspect this is due to lack of flooding of this cover type in early winter.

Our data suggest that SFM management units should be 40–95 ha in size to maximize shorebird abundance and richness. We are aware of no other studies that identify potential for an optimal wetland unit size for shorebirds, or that have evaluated a nonlinear association. However, given the uncertainty in our model due to low sample size of larger management units, an upper threshold of the amount of SFM is difficult to define. Generally, other studies have concluded that larger wetlands are better than smaller wetlands (e.g., Colwell and Taft 2000). Our model uncertainty with regard to the upper threshold of wetland size does not change our assessment that many management units are smaller than optimally sized at SNWRC (Figure 4). These results highlight the need to consider establishing minimum-area recommendations when developing new wetlands to maximize both shorebird abundance and species richness. However, from a wetland management and restoration perspective, there are limitations. The size of a wetland unit is often dictated by the capacity for water management, the existing topography of the landscape prior to restoration, and the habitat objectives for other species. Furthermore, larger units are often more difficult to manage to precise specifications (e.g., water depth, vegetation composition) than are smaller units. These practical constraints limit the capacity to manage for a strictly defined management-unit area threshold, though our results suggest that a beneficial range may be quite large once a minimum size has been reached.

The influence of wetland size in our study may be an indicator of wetland depth or proximity to vegetated cover, both of which are known to affect shorebird use patterns (Colwell and Taft 2000; Ma et al. 2010). Generally shorebirds prefer water depths from >0 to 20 cm in flooded Sacramento Valley rice fields (Elphick and Oring 1998; Strum et al. 2013) and across a similar depth gradient in managed wetlands of the San Joaquin Basin (Isola et al. 2000). We could not measure water depth in this study, but suspect that lower use of the largest wetland units at SNWRC may be due to the fact that overall, they tended to have less tapered-edge features. Our analysis demonstrated that the amount of tapered-edge was positively related with shorebird abundance and species richness. This is likely the result of increased diversity of water depths when there is a gradual transition to upland compared with a ''hard'' transition defined by a levee. Colwell and Taft (2000) found that depth variability correlated with topographical variation. Additionally, shorebirds are known to stay away from vegetated edges to reduce predation risk in a tidal ecosystem (Pomeroy 2006), and this also likely occurs in managed wetlands. It may be that smaller than optimally sized wetlands, on average, have a shorter distance to emergent vegetation or a levee edge than larger than optimally sized wetlands in our study, possibly making them less desirable for shorebirds. Combined, low topographic variation in larger management units and closer proximity to edges in smaller management units may constrain shorebird use and produce the observed pattern of an optimal management-unit size.

The abundance of shorebirds in SFM-units was nonlinearly associated with the amount of surface water within a 10-km buffer, with optimal conditions occurring when 15–40% of the surrounding landscape was flooded (approx. 11,000 ha; 30% of an averaged-sized 10-km buffer). Elphick (2008) highlighted a significant positive association between the amount of refuge wetlands within a 2-km buffer and shorebird use of flooded rice fields in the Sacramento Valley. Taft and Haig (2006) found a significant association between the amount of flooding within a 2-km buffer and the abundance of shorebirds in the Willamette Valley in Oregon. Although Taft and Haig (2006) only used a 2-km buffer, Elphick (2008) used the same range of buffers used in our study (except for the 20-km buffer). The cause of the difference in the spatial scale that had a significant effect on shorebirds between our study and Elphick (2008) is not known. However, in our assessment we also found a significant effect of both linear and nonlinear parameterizations of the landscape water variables at 2- and 5-km scale; although, when comparing with AIC_c, models with smaller scale and linear forms of buffer covariates were not competitive. Because neither Taft and Haig (2006) nor Elphick (2008) evaluated nonlinear models for the effect of landscape covariates, we are unable to compare directly. Our data propose that shorebird conservation through managed wetlands should consider targeting regions with a mosaic of flooded and dry cover at multiple scales during early winter.

Our inclusion of nonlinear model forms provides additional guidance for managers and likely represents real thresholds in bird–habitat associations that are common in ecology (Wiens et al. 2002). Previous analyses of waterbird use in wetlands in the San Joaquin Valley did not consider nonlinear model forms (Colwell and Taft 2000). Although the threshold concept in ecology and its applicability to restoration has been debated (see Bestelmeyer 2006; Groffman et al. 2006), our study shows the advantages of understanding where a threshold may occur. By targeting areas for conservation, management, or restoration that are optimal (e.g., meet a minimum size threshold), the benefit may be much greater.

Our models helped identify potential priority areas for wetland restoration to support use by shorebirds given the landscape context of early winter flooding over 10 y in the Sacramento Valley. These modeled associations of shorebird abundance using largely coarse-scale variables can be valuable for planning and spatial allocation of resources within a refuge or on a landscape. However, because we evaluated coarser level variables, our models still had the largest uncertainty among individual management units (Table 6) even after accounting for local and landscape factors, and overall models only accounted for 25–32% of the variation in the data. Finer resolution information on management within SFM-units (e.g., recent vegetation management, water depth and source, salinity) would be valuable to better understand how specific management practices influence shorebirds within SFM (e.g., Colwell and Taft 2000). In addition to SFM-units being optimally sized and positioned on the landscape to promote shorebirds, we emphasize previous studies that highlight the need to have largely vegetation-free (Taft and Haig 2006) flooded habitat that is <15 cm deep (Elphick and Oring 1998; Isola et al. 2000; Strum et al. 2013).

There are several considerations when seeking to further apply our results. Species richness during early winter increased linearly in SFM-units with increasing water on the landscape at small spatial scales (2–5 km), whereas alternatively abundance was maximized when 30% of the surrounding landscape was flooded within 10 km. This finding indicates that, when determining where to create a wetland on the landscape, the management objectives (abundance vs. richness) need to be clearly articulated because it may affect strategies and decision-making. Additionally, our abundance results, despite controlling for large flocks and being representative of the composition of shorebirds in the Central Valley (Shuford et al. 1998), were strongly influenced by dowitcher spp.; thus, if another species is the target for conservation and management, additional information and analyses may be needed. Lastly, although our model identified an optimal landscape condition for a specific time period within the year (early winter), we advise caution when interpreting these findings because we are unable to compare the importance of cover types and landscape habitat covariates on shorebirds from other seasons.

Our study provides an assessment of habitat associations of shorebirds in managed wetland habitats in the Sacramento Valley during the transition from the end of the dry season into the rainy winter season (November–December). This is a critical time in the shorebird annual cycle. Birds have just completed a long migration and need to acquire fat reserves for the winter. It is also a time when water availability may become increasingly limited in the Sacramento Valley. We identified optimal wetland sizes, particularly a minimum size threshold, and delineated potential high shorebird response zones for wetland conservation. These results can be used in conjunction with other conservation planning models (see CVJV 2006; Stralberg et al. 2011) and economic models to direct the allocation of conservation and management resources to maximize the return on investment (Naidoo et al. 2006). Ongoing assessment of data collected by long-term monitoring programs (such as surveys conducted at the SNWRC), as well as increased covariate data collection and assessment of habitat association models during other seasons, can provide needed guidance to promote successful data-driven management and conservation strategies for shorebirds in wetlands.

Supplemental Material

Please note: The *Journal of Fish and Wildlife Management* is not responsible for the content or functionality of any supplemental material. Queries should be directed to the corresponding author for the article.

Table S1. Table of data used in analysis of local-scale models of shorebird abundance and species richness at Sacramento National Wildlife Refuge Complex, California, 2000–2009.

Found at DOI: http://dx.doi.org/10.3996/012014-JFWM-003.S1 (677 KB XLSX).

Table S2. Definition of column headings for Tables S1and S3.

Found at DOI: http://dx.doi.org/10.3996/012014-JFWM-003.S2 (11 KB XLSX).

Table S3. Table of data used in analysis of landscape-scale models of shorebird abundance and species richness at Sacramento National Wildlife Refuge Complex, California, 2000–2009.

Found at DOI: http://dx.doi.org/10.3996/012014-JFWM-003.S3.

Table S4. Summary of all models evaluated of shorebird abundance during November and December 2000–2009 at the Sacramento National Wildlife Refuge Complex, California. See Table 1 in manuscript for variable definition.

Found at DOI: http://dx.doi.org/10.3996/012014-JFWM-003.S4 (12 KB XLSX).

Table S5. Summary of all models evaluated of shorebird species richness during November and December 2000–2009 at the Sacramento National Wildlife Refuge Complex, California. See Table 1 in manuscript for variable definition.

Found at DOI: http://dx.doi.org/10.3996/012014-JFWM-003.S5 (11 KB XLSX).

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Found at DOI: http://dx.doi.org/10.3996/012014-JFWM-003.S6; also available at http://cran.r-project.org/web/packages/lme4/lme4.pdf (360 KB PDF).

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Found at DOI: http://dx.doi.org/10.3996/012014-JFWM-003.S12; also available at http://www.fws.gov/uploadedFiles/Region_8/NWRS/Zone_1/Sacramento_Complex/Sacramento/Uploaded_Files/Other/Sac%20Del%20Cls%20Sut%20CCP%20-%20Final.pdf (9420 KB PDF).

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