LINKING SHOREBIRD AND MARSH BIRD HABITAT USE TO WATER MANAGEMENT IN ANTHROPOGENIC AND NATURAL WETLANDS IN THE COLORADO RIVER DELTA

By

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As members of the Dissertation Committee, we certify that we have read the dissertation prepared by Martha M. Gómez Sapiens, titled Linking shorebird and marsh bird habitat use to water management in anthropogenic and natural wetlands in the Colorado River delta, and recommend that it be accepted as fulfilling the dissertation requirement for the Degree of Doctor of Philosophy.

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Table of Contents

ABSTRACT

I estimated patterns of shorebird abundance and species diversity in the Colorado River Delta and Upper Gulf of California wetlands in order to determine the relative contribution of intertidal wetlands and non-tidal anthropogenic wetlands to support shorebird habitat use. Species richness varied from 15 to 26 species among sites and 29 species were detected across sites. Density during the peak migration month was higher at the anthropogenic wetland Cienega de Santa Clara (mean = 168 ind/ha, 95% C.I. 29-367), and the intertidal Golfo de Santa Clara (mean = 153 ind/ha, 95% C.I. 17-323). Anthropogenic wetlands (playa and lagoons) supported high abundance of shorebirds along with intertidal wetlands in the Colorado River Delta (mudflats). In contrast, intertidal wetlands farther south on the Sonoran Coast presented lower abundance but higher diversity of shorebird, likely as a result of the higher diversity of habitats (rocky shore, sandy beach, estuary). I modeled water management scenarios for the Cienega in order to determine the response of the dominant vegetation (southern cattail, *Typha domingensis* Pers.) and the area of the outflow pool below the marsh to different scenarios of water management. The model indicates that if the inflow rate is reduced below the current 4-5 $m^3 s^{-1}$ the vegetated area of the Cienega that supports habitat for marsh birds would decrease in proportion, as would the area of the outflow pool in the Santa Clara Slough identified previously as shorebird habitat. Increases in salinity will also reduce the vegetated area due to the low salt tolerance of *T. domingensis*. In winter about 90% of inflow water exits the Cienega into the Santa Clara Slough due to low evapotranspiration contributing to inundate areas that are used by wintering and migrating shorebirds. Lastly, I explored the feasibility of using Vegetation Indices (NDVI and EVI) to model Yuma Clapper Rail detections in the Cienega de Santa Clara as well as the

effects of adding other habitat variables and the presence of fire events in the performance of linear models based on NDVI. Both NDVI and EVI were positively related to the Yuma Clapper Rail detections. The relationship was weak to moderate, but significant ($P < 0.001$), which suggests other factors besides the vegetation condition play an important role in the bird distribution pattern. A model including all the variability among years was a better predictor of the rails detected per transect, than models for fire and non-fire years. We did not find a significant effect from adding habitat features (water % or vegetation %), and we recommend to include variables at both microhabitat level and landscape level, relevant before and during the breeding season in order to increase the explanatory power of models.

INTRODUCTION

Wetlands include an array of ecosystems that present as a main property standing water or saturated soils continuously, seasonally or periodically (Finlayson and van der Valk., 1995). Wetlands provide a diversity of physical and biological resources (habitat) that organisms dependent on wetlands use for foraging, cover, nesting, escape from predators and other life history traits. Temporal and spatial strategies in resource use are defined as habitat use (Hall et al., 1997).

Bird groups that use wetland resources include marsh birds, shorebirds and waterfowl. These groups can play a critical role in wetland management since their habitats are sensitive to hydrological fluctuations and to other variables (Kushlan, 1986). According to estimations half of all wetland area has been lost globally and part of the remaining wetland area is degraded (Zedler and Kercher, 2005). In the case of birds it is estimated that 44% of bird populations that depend on wetlands are currently of conservation concern (Eddleman et al., 1988; Morrison, et al., 2006; Wetlands International, 2006). This underscores the importance of integrating knowledge of avian habitat distribution, habitat use patterns and ecological processes that, in concert with management actions, can maximize benefits for bird species (Mcfarland et al., 2012).

The decision-making process in wetland conservation confronts the ecosystem substitution paradox, under which less valuable systems in theory could be replaced (Mitsch and Gosselink, 2000). Providing accurate information about the ecological values and wetlands attributes that support bird habitat use is a priority to guide the decision-making process in the allocation of resources to conserve wetlands.

The Colorado River Delta (CRD) and the Upper Gulf of California support 37,890 ha of intertidal and non-tidal brackish wetlands (Glenn et al., 2006). Even though biotic and abiotic characteristics of this delta region have been modified by the reduction in water flow due to upstream water diversion (Glenn et al., 2001, 2006), the CRD and Upper Gulf area still have high productivity wetlands that are habitat for resident and migratory waterbird communities and species of conservation concern (Hinojosa-Huerta et al., 2004; Hinojosa-Huerta et al., 2013a; Hinojosa-Huerta et al., 2013b). This has led to the establishment of long-term monitoring programs focused on marsh birds, shorebird and land birds (Hinojosa-Huerta et al., 2000; 2001; Román-Rodríguez, 2004; Gómez-Sapiens and Soto-Montoya, 2006). The purpose of these programs is to detect changes in population trends and integrate biological monitoring into wetlands management (Conroy et al., 2010; Conway, 2011) as well as to document habitat use in both natural and anthropogenic wetlands.

The Cienega de Santa Clara is the largest marshland of the Sonoran Desert and a critical component of the Colorado River delta. This wetland is habitat for 75% of the endemic Yuma Clapper Rail population (Hinojosa-Huerta et al., 2001) and provides habitat for 71% of the bird species in the Colorado River Delta region (Hinojosa-Huerta et al., 2013b).

The Cienega is an anthropogenic wetland accidentally created as a result of brackish water diverted to Mexico from the United States (Glenn et al., 1992; Carrillo-Guerrero, et al., 2013). The source of water for the Cienega is not guaranteed by legal mechanisms, and as a consequence is susceptible to variation in volume and quality. This will be especially pronounced if the Yuma Desalting Plant in the US state of Arizona, begins operation to treat the water that is currently sent to the Cienega. The fact that this wetland is habitat for the endangered Yuma Clapper Rail (*Rallus longirostris yumanensis*), a species of concern in both Mexico and the United States, and other secretive marsh birds has been determinant for the implementation of comprehensive monitoring programs (Flessa et al., 2012) and water management actions in order to preserve ecosystem ecological functions. This has forced the integration of interdisciplinary studies to provide a more comprehensive understanding of the interactions between the hydrology, water quality, vegetation dynamics, bird communities, and the effects of natural events such as fires and earthquakes, to predict the responses of the system to differing scenarios of water management (Flessa et al., 2012; Glenn et al., 2013a; Glenn et al., 2013b; Hinojosa-Huerta et al., 2013b, García-Hernández, 2013; Mexicano et al., 2013).

In the Cienega de Santa Clara marsh birds occupy the vegetated zones composed by cattail (*Typha domingensis*). The Cienega water drains into the Santa Clara Slough, a lower and shallow intertidal basin where the brackish water mixes with intertidal water during high tide events (Glenn et al., 2013a). The Santa Clara Slough has been detected as a shorebirds high aggregation zone (Morrison, 1993). Therefore, the Cienega de Santa Clara-Santa Clara Slough wetlands represent the ideal system to study patterns of marsh birds and shorebirds habitat use.

Marsh bird and shorebird habitat requirements

Marsh birds and shorebirds have different habitat requirements. The marsh bird group requires inundated areas with cattail patches of different ages that provide nesting and feeding habitats. In the case of shorebirds, habitat requirements includes the availability of playas and areas without vegetation with a water depth less than 20 cm preferentially.

In the Cienega de Santa Clara marsh birds occupy the vegetated zones with the highest densities in the range of 8-12 ind/ha in the northwest part of the Cienega (Hinojosa-Huerta et al., 2013). The presence of dense, old vegetation has been related to a higher risk to predation since the birds have to walk over the top of dense vegetation mats (Conway et al., 2010). Yuma Clapper Rail nesting habitat requires high density vegetation with shallow water, and the areas of the marsh with low density and deep water also seems to be used during the winter as foraging sites (Conway et al., 1993).

In contrast, shorebird aggregation zones include the southern portion of the Cienega or Santa Clara Slough (with a range of 50 to 75% of total shorebirds counted in the Delta and Upper Gulf of California) (Hinojosa-Huerta et al., 2013b). The most abundant group of species are small sandpipers (*Calidris sp*.), especially Western sandpiper (Mellink et al., 1997). Microhabitats for this group of shorebirds are restricted to open mudflats and shallow areas with a low proportion of emergent vegetation. The preference for this habitat by small sandpipers could be associated with highest density of invertebrates and a lower risk of predation (Fernandez and Lank, 2006).

The data suggest the Cienega de Santa Clara habitats might be an important site in terms of food sources for Western Sandpipers and/or an alternative when the coastal mudflats are unavailable due to the extreme high tides that characterize the Upper Gulf. Further studies are needed to determine the biological and physical characteristics that determine shorebird habitat use in the Cienega and how water management can impact their suitability for shorebirds.

Medium size shorebirds such as Dowitchers and Phalaropes and large shorebirds such as American Avocets and Black-necked Stilts use the open areas with deeper water level (10-17 cm) and little or no exposed substrate (Boettcher et al., 1997).

Effect of water volume and quality in marsh birds and shorebirds habitats

Shorebird and marsh bird microhabitats result from a diversity of landscape features that can be modified by the volume and quality of the water inflows. In the case of marsh birds, water volume and quality can alter the availability of healthy vegetation patches and open water areas. In the case of shorebirds this results in the reduction of the availability of playas used as feeding and roosting sites. Water volume and quality (salinity) affects prey availability and is also a critical factor.

Salinity increases can cause changes in the abundance and composition of the invertebrate communities, benefiting some bird species or promoting shifts in prey consumption in species that show diet flexibility (Velazquez, 1992).

Data about the effect of different salinity levels on the individual fitness response even for a single species is very scarce, making difficult to predict critical salinity levels in the case of shorebirds (Rubega and Robinson, 1996).

According to the Huckelbridge et al. (2010) model developed for the Cienega, the wetland can be considered a stable ecosystem. However, negative impacts on vegetation can be predicted from decreases in water inflow and from increases in salinity, the latter being the principal driver for vegetation dynamics. Based on this model it could be expected that marsh bird habitats will be compromised by increases in salinity or decreases in volume affecting vegetation, and to shorebird habitats due to the decreases in water volume that will reduce the availability of playas and shallow open habitats. Moreover, decreases in water volume would decrease the Cienega effluent for the Santa Clara Slough that supports habitat for shorebirds.

Dissertation Format

My study has three objectives: 1) To determine the contribution of intertidal and anthropogenic wetlands in supporting shorebird habitat use in the Colorado River delta and Upper Gulf of California Region, 2) To integrate water budget variables into simplified models to predict the responses of marsh bird and shorebird habitat size to different scenarios of water volume and quality in the Cienega de Santa Clara and Santa Clara Slough and 3) To analyze empirical relationships between Vegetation Indices (NDVI and EVI) and the number of Yuma Clapper Rails detected in the Cienega de Santa Clara to determine the feasibility of predicting and explaining Yuma Clapper Rail habitat use patterns from Vegetation Indices.

The three objectives resulted in three scientific papers and a note. The first paper is an analysis of the shorebird abundance and diversity patterns in anthropogenic, anthropogenic/intertidal and intertidal wetlands. This paper resulted from the analysis of a four year dataset collected from 2005-2008 during the Shorebird Monitoring Program in the Reserva Alto Golfo de California and Delta del Río Colorado (Upper Gulf of California and Colorado River Delta Biosphere Reserve) Sonora, Mexico. My participation involved the proposal writing in collaboration with the Reserve staff and Pronatura Noroeste staff, as well as the field work for data collection and analysis, and preparation of the manuscripts.

The second paper presents a simplified water budget model to predict the extent of the dominant vegetation (*Typha domingensis* Pers.) in the marsh and the area of the outflow pool below the marsh in response to variations of water volume and salinity. We based the water management scenarios on the operation of the Yuma Desalting Plant by using data collected during the Cienega Monitoring Program (Flessa et al., 2012) and complemented the model with an analysis of satellite images from MODIS to determine inundation patterns in the Santa Clara Slough. First and second papers were published in the journal Ecological Engineering, Special Issue Colorado Delta Wetlands.

In the third paper we explored the strength of the relationship between Vegetation Indices (EVI and NDVI), and analyzed the spatiotemporal variation of VIs and detections. Lastly, we explored the effects of adding other habitat variables and the presence of fire events in the performance of linear models based on NDVI. This manuscript has not been published and it is prepared in the format for the Journal Wetlands. Yuma Clapper Rail data were collected during the marsh bird monitoring program of Pronatura A.C. in the Colorado River delta (Hinojosa-Huerta et al., 2001). I integrated the data into a spatial database and performed spatial and non-spatial analysis.

The note is supplementary information from shorebird aerial surveys conducted from 2006 to 2011 in the Colorado River Delta and Upper Gulf wetlands. This complemented the data from ground surveys and provided an overview of the number of shorebirds grouped by size that use the wetlands on a large scale.

THE PRESENT STUDY

The methods, results, and conclusions of this study are presented in the manuscripts appended to this dissertation. The following is a summary of the most important findings in this study.

1. Shorebird Abundance and Species Diversity in Natural Intertidal and Non-Tidal Anthropogenic Wetlands of the Colorado River Delta, Mexico.

Shorebirds (Families: *Charadriidae, Recurvirostridae, Haematopodidae* and *Scolopacidae*) constitute the highest abundance group of birds that use the Upper Gulf of California and Colorado River Delta (CRD) wetlands for nesting, spring stopover and overwintering sites (Mellink et al., 1997; Hinojosa-Huerta et al., 2004).

The intertidal wetlands are extensive due to the shallow slope of the seabed and the high amplitude (up to 5 m) of the mixed diurnal tides (two tide cycles per day) in the CRD. However, the Colorado River no longer carries seasonal fresh water flows due to upstream dams and water diversions, and as a consequence the salinity regime of the estuary has been altered from seasonally mesohaline to consistently euryhaline, which has induced changes in the flora and fauna (e.g., Avila-Serrano et al., 2006).

On the other hand, a series of non-tidal, anthropogenic brackish wetlands have been created in the delta from the disposal of agricultural drainage water and other management activities. The goal was to determine the overall importance of the CRD in supporting shorebirds, and in particular the role of the anthropogenic wetlands, which face uncertain futures.

Species richness varied from 15 to 26 species among sites and 29 species were detected across sites. The most abundant species was *Calidris mauri*, which was most abundant in Isla Montague and Golfo de Santa Clara in winter and spring, while it was most abundant in the Cienega de Santa Clara and Mesa de Andrade wetland in spring and fall. Cienega de Santa Clara and Golfo de Santa Clara had the highest bird density with 168 and 105 individuals/ha in the peak migration month. Birds tended to use the intertidal wetlands during the winter and spring migration period while the inland wetlands were most used during spring and fall. The Cerro Prieto geothermal power plant wetlands were most used by Phalaropes species during fall migration. Bahia Adair, an extensive intertidal wetland system south of the CRD, had a low density of shorebirds (10 ind/ha) compared to CRD sites, but it had higher species diversity and the highest proportion of large size shorebirds. This study shows the importance of both intertidal and anthropogenic wetlands in supporting shorebirds along the Pacific Flyway. Management decisions that might impact these wetlands should consider their habitat value for migratory shorebirds, as documented here.

2. Modeling water management scenarios for the Cienega de Santa Clara, an anthropogenic coastal desert wetland system, based on inflow volumes and salinities.

The Cienega de Santa Clara in the Colorado River Delta, Mexico is a self-designed wetland system fed since 1977 by brackish groundwater diverted from the U.S. to Mexico. The vegetated upper portion of the Cienega provides habitat for endangered Yuma Clappers Rails and other marsh birds and fulfills other ecological functions. Outflow water pools in the Santa Clara Slough south of the Cienega and provides habitat for migratory shorebirds. Conditions in the Cienega and Santa Clara Slough could be altered by operation of the Yuma Desalting Plant (YDP), which will divert water from the Cienega and replace it with brine water resulting from the desalting process. Our objective was to integrate water budget components into models predicting the extent of the dominant vegetation (southern cattail, *Typha domingensis* Pers.) in the marsh and the area of the outflow pool below the marsh in response to different operating scenarios for the YDP. The models are intended to serve as tools for resource managers charged with maintaining this wetland complex. Unlike many wetland water budget models, this one explicitly takes into account salinity as a factor in the water budget.

We built a water budget model using surface water inflow (SW_{IN}), precipitation (P), evapotranspiration (ET) from vegetation (E_{VEG}) and open water areas (E_{water}) and surface water outflow (SW_{OUT}). Values for the water balance components were based on monthly measurements made over a 33-month period, 2009–2011 during the Cienega Monitoring Project.

The model indicates that if the inflow rate is reduced below the current 4-5 $m^3 s^{-1}$ the vegetated area of the Cienega would decrease in proportion, as would the area of the outflow pool in the Santa Clara Slough. Increases in salinity will also reduce the vegetated area due to the low salt tolerance of *T. domingensis*. In winter about 90% of inflow water exits the Cienega into the Santa Clara Slough due to low evapotranspiration, and on an annual basis 70% of inflows exit into the Santa Clara Slough. These flushing flows maintain the salt balance in the Cienega and contribute to the area of shorebird habitat. The Santa Clara Slough is periodically flushed by spring tides, making this a sustainable, open wetland system in its present state.

3. Spatiotemporal variation of NDVI and EVI and its suitability to model Yuma Clapper Rail detections in the Cienega de Santa Clara, Sonora, Mexico.

The determination of habitat features that have a key role in the species habitat preferences and fitness is crucial to generate habitat models. These models can be used to predict species probability of occupancy, to explain population trends and species habitat use pattern in response to changing variables (Naugle et al., 2001; Valente et al., 2011; Harms and Dinsmore, 2013).

The Normalized Difference Vegetation Index (NDVI) can be a good indicator of the quality of the vegetation, allowing relationships to richness to be established, distribution patterns, and abundance and life-history traits in birds (Pettorelli et al., 2005; 2011)**.** The Cienega de Santa Clara is an anthropogenic wetland dominated by cattail and provides habitat for 75% of the Yuma Clapper Rail population. In this wetland the NDVI has been positively correlated to the amount of water entering the wetland and to the presence of fire events (Glenn et al., 2013; Mexicano et al., 2013). Increments after fire events have been detected also in the marsh bird populations (Hinojosa-Huerta et al., 2013).

In light of this, we expected the NDVI and EVI could be used as a proxy of the vegetation condition and have an important contribution to explain the variance in the Yuma Clapper Rail detected spatially and temporally.

The objectives of this analysis were: 1) To determine the strength of the relationship between vegetation indices and Yuma Clapper Rail detections, 2) To conduct an exploratory spatial analysis to detect patterns of spatial association (clusters) in the number of Yuma clapper Rails and in the vegetation indices (NDVI and EVI) across years, 3) To determine if NDVI and additional habitat variables can be used to predict distribution patterns of Clapper Rails in the Cienega and 4) To explore the effect of fire in the relationship between NDVI and VIs.

Our approach was first to determine the strength of the relationship, then to analyze the spatiotemporal variation of VIs and detections. Lastly, we explored the effects of adding other habitat variables and the presence of fire events in the performance of linear models based on NDVI.

We found both NDVI and EVI were positively related to the Yuma Clapper Rail detections, the relationship was weak to moderate, but significant ($P < 0.001$), which suggest other factors besides the vegetation condition play an important role. We found NDVI was a better predictor than EVI. The analysis of the spatiotemporal variation in NDVI and detections showed that the wetland presents clusters of high NDVI values in the central lagoons in the north part of the wetland that are relatively stable across years. In contrast, the edges of the wetland, and particularly the eastern edge, tend to have more dynamic alternating clusters of high and low NDVI in response to an edge with higher fluctuations in water level due topographic effects. The clusters of high and low Yuma Clapper Rail detections did not follow the same temporal trend that the NDVI, however the effects of changes in the vegetation condition in the wetland were observed in subsequent years. Fire events were followed by increases in the maximum NDVI value and a high numbers of clapper rails were detected during the fire year and in the following years as a result of vegetation renewal and habitat improvement. A model that included all variability among years was a better predictor of Yuma Clapper Rails detected per transect than models for fire and non-fire years. We did not find a significant effect from adding habitat features (water % or vegetation %), and we recommend to include variables at both microhabitat level and landscape level relevant both before and during the breeding season in order to increase the explanatory power of models.

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APPENDIX A

Shorebird Abundance and Species Diversity in Natural Intertidal and Non-Tidal Anthropogenic Wetlands of the Colorado River Delta, Mexico

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Abstract

Shorebirds constitute the highest abundance group of birds that use the Upper Gulf of California and Colorado River Delta (CRD) wetlands for nesting, spring stopover and overwintering sites. From August 2005 to December 2008 ground surveys were conducted on three natural intertidal wetlands (Golfo de Santa Clara, Isla Montague and Bahia Adair) and three brackish anthropogenic wetlands (Cienega de Santa Clara, Cerro Prieto and Mesa de Andrade) in the Upper Gulf and CRD. The goal was to determine the overall importance of the CRD in supporting shorebirds, and in particular the role of the anthropogenic wetlands, which face uncertain futures. Species richness varied from 15 to 26 species among sites and 29 species were detected across sites. The most abundant species was *Calidris mauri*, which was most abundant in Isla Montague and Golfo de Santa Clara in winter and spring, while it was most abundant in the Cienega de Santa Clara and Mesa de Andrade wetland in spring and fall. Cienega de Santa Clara and Golfo de Santa Clara had the highest bird density with 168 and 105 individuals/ha in the peak migration month. Birds tended to use the intertidal wetlands during the winter and spring migration period while the inland wetlands were most used during spring and fall. The Cerro Prieto geothermal power plant wetlands were most used by Phalaropes species during fall migration. Bahia Adair, an extensive intertidal wetland system south of the CRD, had a low density of shorebirds (10 ind/ha) compared to CRD sites, but it had higher species diversity and the highest proportion of large size shorebirds. This study shows the importance of both intertidal and anthropogenic wetlands in supporting shorebirds along the Pacific Flyway. Management decisions that might impact these wetlands should consider their habitat value for migratory shorebirds as documented here.

Keywords. Shorebirds, Colorado River Delta, habitat use, anthropogenic wetlands, intertidal wetlands

1. Introduction

The Colorado River Delta (CRD) and the Upper Gulf of California support 37,890 ha of intertidal and non-tidal brackish wetlands (Glenn et al., 2006). Much of this area is protected by Mexican Law as a Biosphere Reserve and is designated as a Ramsar Site, an Important Bird Conservation Area (AICA), and a component of the Western Hemispheric Shorebird Reserve Network (WHSRN). Even though biotic and abiotic characteristics of this delta region have been modified by the reduction in water flow due to upstream water diversion (Glenn et al., 1996, 2006) the CRD and Upper Gulf area still have high productivity wetlands that support resident and migratory waterbird communities (Hinojosa-Huerta et al., 2004a).

Documented waterbird habitat uses for Upper Gulf and CRD region include reproduction sites for several species, and stopover and wintering sites for migratory species along the Pacific Flyway (Anderson et al., 2003). Shorebirds (Families: *Charadriidae, Recurvirostridae, Haematopodidae* and *Scolopacidae*) are the group with the highest abundance (Mellink et al., 1997; Hinojosa-Huerta et al., 2004b). An earlier aerial survey estimated 163,744 wintering shorebirds in the CRD (Morrison, 1992), whereas subsequent ground survey estimates ranged from 88,000 (Roman-Rodriquez, 2004) to 148,285 (Mellink et al., 1997). These studies indicated that the CRD region is equal to the Salton Sea and other high-density shorebird sites on the Pacific Flyway (Shuford et al., 2002). Shorebirds use the CRD wetlands mostly as a wintering site, with the mudflats and saltflats located near the mouth of the river being the most used areas (Mellink et al, 1997). According to Mellink et al, (1997) there is a possibility that individuals that use the Salton Sea could be the same individuals that use CRD wetlands since both sites are reported to be on the flyway for birds moving southward to the Pacific coastal areas during their migrations (Shuford et al., 2002).

The intertidal wetlands are extensive due to the shallow slope of the seabed and the high amplitude (up to 5 m) of the mixed diurnal tides (two tide cycles per day) in the CRD. However, the Colorado River no longer carries seasonal fresh water flows due to upstream dams and water diversions, and as a consequence the salinity regime of the estuary has been altered from seasonally miohaline to consistently euryhaline, which has induced changes in the flora and fauna (e.g., Avila-Serrano et al., 2006). On the other hand, a series of non-tidal, anthropogenic brackish wetlands have been created in the delta from the disposal of agricultural drainage water and other management activities. Our research question was, what is the relative contribution of the now-euryhaline intertidal wetlands and the brackish anthropogenic wetlands in supporting shorebird abundance and diversity? The answer to this question has management implications, because these anthropogenic wetlands were all inadvertently created by water management decisions in the U.S. and Mexico, and as a result they could be altered or eliminated by future management activities.

2. Methods

2.1. Study sites

Studies sites are shown in Figure 1 and their size and main features are described in Table 1. The intertidal zone (Fig. 1) includes the coastline located at El Golfo de Santa Clara village, Isla Montague, San Felipe and Bahia Adair. The first three localities are part of the CRD estuary, which is dissected by three channels bordering the Montague-Gore and Pelicano Islands inside the core zone of the Biosphere Reserve. The coastline consists of extensive mudflats that decrease in silt and clay with increasing distance from the river mouth. The vegetation includes patches of the endemic saltgrass *Distichlis palmerii* on the Isla Montague and adjacent coastline areas and other halophytic species at other locations in the intertidal zone (Yensen *et al*., 1983 and Felger, 2000).

The shoreline sites selected for sampling in the CRD consists of silty-clay (limo-arcilloso) sediments, with high benthic species richness (46), high density of benthic organisms (98.6 individuals per m^2) and high values for biomass (58.07 mg per m^2) (Villarreal and Carmona, 1999). Sampling was also conducted in Bahia Adair, an extensive estuary of tidal channels, mudflats and saltflats southeast of the CRD on the Sonoran shoreline (Felger, 2000). Sediments in that estuary sit on cemented seashell formations called coquinas, interrupted by tidal ponds and sandy areas. Halophytic species from Bahia Adair include *Allenrolfea occidentalis, Batis maritima, Salicornia bigelovii, S. subterminalis, S. virginica, Suaeda esteroa, S. puertopenascoa, Atriplex linearis, Atriplex barclayana, Cressa truxillensis, Frankenia salina, Distichlis palmeri, Monanthochloë littoralis* (Felger, 2000).

The first anthropogenic wetland complex selected for sampling was Cienega de Santa Clara (Fig. 1), a 4200 ha brackish wetland maintained by mildly saline (2-3 ppt) water releases from the Wellton-Mohawk Irrigation District in the U.S. (Glenn *et al*., 1992; Glenn et al., 2001). It is an emergent marsh dominated by cattail (*Typha domingensis*), with about 15% open water areas, and patches of *Phragmites australis*, *Juncus* , *Distichlis spicata*, *D. palmeri*, *Scirpus* spp., *Tamarix chinensis* and *Allenrolfea occidentals* within the marsh or along the sides of the marsh (Abarca et al., 1993). The Cienega also supports large quantities of submerged aquatics such as *Najas marina* and *Ruppia maritima* (Glenn *et al*., 1992), important food sources for some waterfowl. South of the Cienega is an extensive evaporation basin that receives effluent from the Cienega and occasional tidal flooding. The Cienega is within the core and buffer zone of the Biosphere Reserve.

The second anthropogenic wetland complex is the Cerro Prieto geothermal field (Fig. 1). It consists of approximately 4000 ha of evaporation ponds, mud volcanos (0.05-2 m high), hot springs, vents, salt pans and fault scarps (Mercado and Fernández, 1998). Water sources include ground water fed by the Cucapa Mountain Range alluvial fans and the Colorado River (Mercado and Fernández, 1998). The wetlands are salty and shallow with an important microcrustacean community and other benthic species (Varela-Romero *et al.,* 1998). *Allenrolfea occidentalis* is the dominant plant species. The main shorebird habitats are in a series of evaporation ponds where effluent water from the geothermal electric generating facility is discharged. These shallow ponds contain islands of dredged material, which provide protection from predators for nesting birds. The ponds also provide food in the form of submerged aquatic vegetation, fish and invertebrates growing in the brackish ponds. According to Hinojosa-Huerta *et al*. (2004a) the Cerro Prieto wetlands are a nesting site link for shorebirds, between the Lower Colorado River (Isla Montague) and Salton Sea nesting sites.

The third set of anthropogenic wetlands are the Mesa de Andrade wetlands (Fig.1), extending over an area of 6200 ha. These wetlands are formed by water seeping from the formerly unlined All American Canal in the United States. Seepage flows under the dune field comprising the San Andrade Mesa and surfaces along the foot of the dune field at its southern extent in Mexico, forming small, isolated wetlands separated by dunes. The wetlands consist of emergent marshes, playas and vegetated dunes, where the water table is within the reach of phreatophytic vegetation. Representative plant species in the marshes are *Distichlis spicata* and *Typha* *domingensis*, while the dunes support *Larreta tridentata*, *Prosopis glandulosa*, *P. pubescens* and *Atriplex* spp. and other halophytes (Hinojosa-Huerta *et al.,* 2002). With the lining of the All American Canal in 2008, seepage is not expected to continue and the future of these wetlands is in doubt.

2.2. Survey methods

The methods were based on the protocol of the Western Shorebird Survey (Bart, 2004). Surveys at each site were conducted at times of year previously identified as critical migration, wintering or nesting periods. Counts started at around 0800 h and continued throughout the day. At each site transects were set along the coastline or edge of the wetland and consisted of five or more point counts 400 m apart unless noted otherwise below. Each survey point had a radius of 250 m and an area of 12.5 ha. Surveyors used 10X binoculars and a 20X spotting scope to observe birds. In cases in which species identification was not possible, only the genus of the bird was recorded. In cases when larger concentrations than could be counted were encountered, numbers of shorebirds were determined by flock estimation following Kasprzyk and Harrington (1989) recommendations. Relative abundance (% composition) was calculated for each species.

At the intertidal zone habitats, the sampling dates corresponded to the lowest neap tide day for each month. At the non-tidal sites sampling dates were selected by chance keeping a similar time frame among surveys on each site. Surveys started in August 2005 in all the sites and in November for Bahia Adair and Isla Montague.

For the period from August 2005 to December 2006 six sampling zones were selected in Golfo de Santa Clara, Isla Montague and Bahia Adair for the intertidal zone and Cienega de Santa Clara, Cerro Prieto and Mesa de Andrade for the anthropogenic sites. At El Golfo de Santa Clara five points each were set along the shoreline from northwest to southeast.

Around Isla Montague 15 points were set along the intertidal mudflat and surveys were done from a small boat using binoculars at a variable distance to the land (5-20m) according to the topography of the shoreline, weather and tidal conditions. At Bahia Adair two localities were established with a five point transect on each one with a separation of 500 m among points in the sandy shoreline and coquinas, and along the estuary. At the Cienega de Santa Clara surveys were conducted since April 2007 in the three points in the southwest part of the wetland which has open areas in the southern limit of the cattail distribution. These habitats are composed of mudflats and saltflats that extend toward the mouth of the river and points were separated by 500 m.

For the Mesa de Andrade wetlands three points were established, one for each of the lagoons. Visits to Mesa de Andrade Wetlands were discontinued in January 2007 due the lowest abundance of shorebirds recorded for the site. In the case of the Cerro Prieto Geothermal Field the surveys were done along the evaporation ponds by car to include individuals feeding in the shorelines of the ponds and using the spotting scope and binoculars to observe the flocks in distant areas. In Cerro Prieto the visits were restricted to the summer months (fall migration) since it was determined to be a more important season for shorebird use.

2.3. Data analysis

In order to compare density among the sites, annual mean density was estimated using the peak migration month for each site and the sampled area. In the case of Isla Montague two points were eliminated since they do not contain mudflat area and shorebirds visits were very infrequent (See table 1). To detect differences in mean abundance per site among years one-way analysis of variance (ANOVA) and t-tests were conducted on log transformed data, using SPSS version 13.0.

Shannon diversity index was estimated for each community's sampled year, and similarity indices (Jaccard, Sorenson, Morisita-Horn and Bray-Curtis) on the total of individuals detected per site during the period of survey. Correspondence Canonical Analysis (CCA) was used to determine the differences on community's structure and conducted on CANOCO version 4.5.

3. Results

3.1. Overall abundance and diversity

The sites with the highest shorebird density during peak migration months were the Cienega de Santa Clara and Golfo de Santa Clara with 168 and 105 individuals per ha, respectively (Table 1). Species richness ranged from 15 to 26 species among individual sites and 29 shorebird species were recorded among all the sites (Tables 2 and 3).

The site with the highest diversity of habitats (Bahia Adair) showed highest Shannon Diversity Index values from 1.90 to 2.62 and the lowest abundances (below 6,000 shorebirds per sampled year), playa habitat sampled in the Cienega had the highest abundance along with mudflats habitats in some of the sampled years (over 19,000 shorebirds) with diversity value in the middle of the spectrum found for this region (about 1.5), the lagoon habitats sampled in the Cerro Prieto Site showed diversity values below 1.3 and abundances in the range of 13,000- 18,000 shorebirds per year. Mudflats sampled in the Delta (Golfo and Isla Montague) showed the highest variation in terms of diversity and number of shorebirds (Figure 2).

Shorebird community assemblage differs mainly by type of wetland and diversity of habitats. According to the ordination model from a CCA (Figure 3), 53% of the variation in the shorebird community structure was explained by the sites Adair, Cienega and Cerro Prieto (*p*-value < 0.001), even though the axis that defines the Cienega site had a higher *p*-value (0.08). The axis defined by the shorebird community assemblage of El Golfo de Santa Clara and Isla Montague had higher *p*-values (> 0.3). Differences between the axis can be attributed in the case of Bahia Adair to the highest abundance of *Arenaria* species, Surfbird and American Oystercatcher. In the Cienega de Santa Clara the axis was defined by species detected in higher abundances (American Avocet and Black-necked Stilt) or exclusively at that site (Stilt Sandpiper and Killdeer). In the Cerro Prieto site the axis was defined by higher abundance of Phalarope species, Least Sandpiper and Spotted Sandpiper (Figure 3).

3.2. Shorebird composition and abundance in the intertidal zone

El Golfo de Santa Clara and Isla Montague had a similar distribution of shorebirds species. At both sites the most abundant shorebirds were small *Calidris* species, with *C. mauri* individuals accounting for about 60% of relative abundance (Table 3). Other dominant species included Dowitchers (*Limnodromus* spp.) with 12.7% and 10.5% of relative abundance respectively. After *C. mauri* the most common *Calidris* species was Red Knot (*Calidris canutus*) in the case of El Golfo site, and Willet (*Tringa semipalmata*) in Isla Montague. *Numenius* species were abundant at the Isla Montague accounting for 11.4% of relative abundance, and most were individuals of Long-billed Curlews (*Numenius americanus*). At the El Golfo site the percentage of the large size shorebirds like Willet, Long-billed Curlew and

Marbled Godwit was approximately 12%. Plovers represented 2.2% of relative abundance at El Golfo and nearly 1.9% at Isla Montague. In Bahia Adair the most abundant shorebirds were *Calidris* and *Numenius* species which accounted for 24% of relative abundance with *N. americanus* being the most common species from that genus. Willets (*Tringa semipalmata*) accounted for 17% of relative abundance while Marbled Godwits (*Limosa fedoa*) were slightly less dominant accounting for 11% of the total abundance. In this site a large flock (350 individuals) of American Oystercatchers (*Haematopus palliatus*) was detected in the "Coquina" areas during the winter months. From the three intertidal zone sites, Bahia Adair showed the highest relative abundance of plovers with 8.9% (Table 3).

3.3. Shorebird composition and abundance in the anthropogenic wetlands

In the southern portion of the Cienega, *Calidris* and *Limnodromus* species were the most abundant species, with relative abundances of approximately 56.9% and 19.3% respectively, followed by Black-necked Stilt and American Avocet (7.1%), and phalaropes (5.9%) (Table 4). 1,110 records of Marbled Godwit (*Limosa fedoa*) were recorded in this location and the Snowy Plover (*Charadrius nivosus*) was the most abundant plover with 89 records.

The most common shorebirds at the Andrade Mesa Wetlands were *Calidris,* or small sandpipers, representing 33% of the shorebird community. Dowitchers (*Limnodromus sp.*) and Black-necked Stilts (*Himantopus mexicanus*) were the second and third most abundant species with 24.3% and 23.8% respectively, with occurrence of phalaropes and American Avocets (*Recurvirostra americana*) being around 5% (Table 4). The most common plover was Killdeer (*Charadrius vociferus*) with 35 records.

The Cerro Prieto ponds were used mostly by phalaropes during the summer months or fall migration (relative abundance of 80%), with more than 20,000 individuals in three summers of surveys and a range of 3000-8000 individuals recorded during each visit. *Calidris* species were the second most abundant (11.7%) shorebird in these ponds. American Avocet (4.0%) and Black-necked Stilt (3.3%) were abundant all year round, and among the plovers the most recorded species were Snowy Plovers and Black-bellied Plovers with 121 and 66 individuals (Table 4).

3.4. Communities comparison based on similarity indices

Qualitative indices Jaccard and Sorenson suggest high similarity in terms of the number of shared species between Bahia Adair and El Golfo site, between the rest of the intertidal sites and also between Bahia Adair and Cerro Prieto with values higher than 0.6 for both indices (Table 5). Quantitative indices Morisita-Horn and Bray-Curtis suggest high similarity among the three communities close in distance to the delta (Cienega, Golfo and Isla Montague) and lower similarity between Bahia Adair and Cerro Prieto and El Golfo and Mesa de Andrade (Table 5).

3.5. Differences in seasonal abundance among sites

Seasonal abundance at El Golfo and Isla Montague intertidal sites showed the same pattern with highest abundance during winter months and over spring migration, which corresponds with *Calidris mauri* wintering and migration (Fig. 4.). In contrast, summer months showed a low abundance of *C. mauri* and an increase in the abundance of Red Knots (*C. canutus*).

In Bahia Adair seasonal abundance of shorebirds was high from November to January with another peak during spring migration (Figure 4), due mainly to visitations by *C. mauri* and *Numenius* spp.

In the Cienega de Santa Clara peak abundances were recorded in April and August as a result of the presence of both *Calidris* and *Limnodromus* species in April, and of *Calidris mauri* in August (Fig. 5). Important numbers of other species (American Avocet and Phalaropes) contributed to seasonal abundance in the summer months.

For the last three years of sampling at Cerro Prieto the months of highest abundances were August and September (Fig. 5) as a result of Phalarope migration. For the Mesa de Andrade wetlands abundance peaks were recorded in winter due the presence of *Limnodromus* spp. In August, bird abundance increased due to visitations by *Calidris* sp. (Fig. 5).

Differences in abundance among years (from 2006 to 2008) were non-significant ($p > 0.05$) for Bahia Adair, ($F = 0.671$, p-value = 0.522). La Flor in Cienega de Santa Clara ($F = 0.817$, p-value = 0.425 for years 2007 and 2008), Isla Montague, ($F = 0.537$, $p = 0.611$ for years 2006 and 2007). There was moderate evidence of a difference among years in the Golfo de Santa Clara ($F = 3.39$, $p = 0.046$).

4. Discussion

4.1. Overall abundance and species diversity of CRD and Upper Gulf wetlands

During the four years of the monitoring program we recorded 54,108 shorebirds at all localities. Mellink et al. (1997) recorded 170,700 shorebirds in 1993-1994 for the same localities. However, the area surveyed by Mellink et al. (1997) at each location was much larger than in our surveys, since they used wide-area estimates for their census versus our use of point counts which result in a smaller area being surveyed Nonetheless, both surveys show the CRD is a major stopover spot for migratory shorebirds on the Pacific Flyway. We did not detect decreases in habitat use over the years of our study. However, total abundance from specific sampling points should not be considered as the only indicator of population trends since other variables such as turnover rates and the use of other areas not sampled could be taking place (Warnock and Haig, 1998; Skagen et al., 2008).

4.2. Seasonal patterns of abundance in intertidal versus non-tidal wetlands

The seasonal patterns of abundance indicate that the intertidal zone had the highest use by shorebirds during winter and spring migration periods, whereas the anthropogenic wetlands had highest abundance during the spring and fall migration. Bahia Adair appears to be more important in the early winter months of November and December than at other times of year. Peak abundances of seasonal shorebird variation in winter and spring months at nearby staging sites in the Central Valley of California and Baja California (Palacios et al., 1991; Shuford et al., 1998) matched the seasonal patterns observed in our intertidal sites. On the other hand, sites such as the Salton Sea and Humboldt Bay show same pattern (peak abundance in spring and fall months) as our anthropogenic (non-tidal) wetlands (Colwell, 1994; Shuford et al., 2002 and Carmona et al., 2004). Thus, shorebird visitation in intertidal and anthropogenic wetlands were complimentary to each other, extending the regional habitat value of the CRD and Upper Gulf beyond what could be provided by the intertidal sites alone. They also appeared to be complimentary to seasonal visitation to other major shorebird visitation sites on the flyway. The CRD non-tidal wetlands and the Salton Sea provide inland stopover sites along the Pacific Flyway and studies suggest connectivity between both areas (Shuford et al., 2002).

4.3. Abundance of Western Sandpipers

The peak shorebird abundances in April and August at the Salton Sea and in February in the CRD are mainly the result of the high abundance of Western Sandpipers (Shuford et al., 2002). Our data suggest that part of the population of Western Sandpiper is wintering in the intertidal zone in the CRD and other studies indicate that the species also winters in the Central Valley of California and at southern stopover sites such as Ensenada Pabellones and Bahía Santa María in Sinaloa, Guerrero Negro and Ensenada de La Paz in Baja California Sur, México, where high relative abundance (60-80%) of this species has been reported (Carmona and Danemann, 1998; Engilis, et al., 1998 and Fernandez et al., 1998).

Western Sandpipers might also overwinter in the Salton Sea since highest abundances are reported in fall surveys (Shuford et al., 2002). These birds then move south to the intertidal zone at the CRD. Mellink et al. (1997) suggested that the same individuals might be using both areas. In spring, Western Sandpipers use the CRD coastal wetlands (Golfo de Santa Clara and Isla Montague) but they also use the Cienega during both spring and fall as well as the Salton Sea (Shuford et al., 1998). The migration of Western Sandpiper is reported to be more compressed in time and space in spring compared to fall (Butler et al., 1996). Clearly, Western Sandpipers use both intertidal and anthropogenic wetlands in the CRD, and display different seasonal preferences for these wetlands.

4.4. Abundance of Phalaropes

In the case of highly abundant phalarope species in summer, 1999 data for Salton Sea were lower than numbers observed in the present study (Shuford et al., 2002). They found fewer than half as many Red-necked and Wilson's Phalaropes as we found in the Cerro Prieto geothermal ponds. However, similar to our study, a high concentration of Wilson´s Phalarope were reported to be using evaporation ponds during late July in the Central Valley of California, along with Red-necked Phalarope, American Avocets, small sandpipers and Black-necked Stilts (Shuford et al., 1998). Although Great Salt Lake, Utah, along with Mono Lake, California and Lake Albert, Oregon are reported to be the main staging areas for Wilson's Phalarope, anthropogenic wetlands such as Cerro Prieto also appear to be important during fall migration, being the inland wetland with the highest abundance of *Phalaropus* (mostly *lobatus*) species in Northwest Mexico. The Cerro Prieto ponds represent protected feeding areas, similar to the saltworks at Guerrero Negro on the Baja California peninsula (Carmona and Danemann, 1998). The species use saltworks to recover during migration on their route to the South America wintering areas. However, our data suggest Phalaropes might use an alternate route during their northward migration, since they were not recorded in the CRD during the spring migration months.

4.5. Shorebird usage of specific habitat features in each wetland system

According to Mellink et al., (1997) the most frequently used parts of the intertidal zone in the CRD were the mudflats between Pelicano and Isla Montague and the mudflats between Pelicano Island and the coastline of El Golfo de Santa Clara. In our surveys, however, the areas with the highest numbers of shorebirds per sampling unit were Cienega de Santa Clara and Golfo de Santa Clara mudflats. Water enters the Cienega from the U.S. at about 2-3 ppt, and it exits at about 6 ppt at the southern end, where the shorebirds tend to concentrate. The reduced salinity associated with this effluent may support higher abundance or a more diverse invertebrate community on which shorebirds feed.

For the Cienega de Santa Clara there appear to have been changes in the dominant shorebird species assemblage, since previous studies reported highest abundance for the species *Limnodromus sp., Calidris mauri, Recurvirostra Americana* and *Himantopus mexicanus* (Abarca et al., 1993 and Román-Rodríguez, 2004 unpublished data)., whereas the most abundant group of species in our surveys were small sandpipers (*Calidris sp*. and mostly Western Sandpiper). The change in the community composition may be linked to dynamic vegetation changes. The sites that were monitored by this study are open shallow areas with less proportion of emergent vegetation, a habitat that may be preferred by small sandpipers due to a higher density of invertebrates and a lower risk of depredation (Fernandez and Lank, 2006). The proportion of area covered by vegetation has changed over time at this site, and the sites that were previously monitored were covered by vegetation with availability of open lagoons which are more often used by medium and large size shorebirds.

The data suggest that the Cienega de Santa Clara habitats might be an important site in terms of food sources for Western Sandpipers and/or an alternative site when the coastal mudflats are unavailable during high tides as well as for species that showed preference for brackish water habitats. Although further studies are needed to determine the biological and physical characteristics that determine shorebird habitat use in the Cienega, it has been documented that drops in the average water inflow (4.5 m^3/s) have effects on the availability of playa areas for shorebirds

(Gomez-Sapiens et al., 2012 this volume). This highlights the need of including the prevalence of this wetland feature as one of many conservation goals in the water management schemes for the Cienega.

Bahía Adair represents a different system with the least influence from the Colorado River. Even though this site showed a low density of shorebirds there was a high diversity and a high proportion of large size shorebirds using Bahia Adair. The different composition in the shorebird assemblage for this area is the result of the diversity of habitats at the bay i.e.saltflats, mudflats, estuaries, rocky intertidal areas “coquinas” and sandy shores. Some of the concerns for this recently monitored area include the construction of a new highway that will facilitate increased human access to nesting habitats for at least two shorebird species (Snowy Plover and Wilson Plover) and the endangered Least Tern. Additionally, the highway will likely enable human development along the coastline area close to the shorebird´s feeding and resting areas between Puerto Peñasco and El Golfo de Santa Clara, including Bahia Adair, which combined represents about 90% of the northern Gulf of California wetlands that currently have no major human impact (Glenn et al., 2006). It is remarkable that during one survey event 100% of the American Oystercatcher population reported by Wetlands International (2006) for the Gulf of California and Western Mexico (350 individuals) was detected in this area. This highlights the importance of documenting the shorebird use of these habitats in undisturbed conditions in order to keep track of any changes in shorebirds habitat use as a result of human impacts.

Cerro Prieto geothermal ponds offer protection to birds nesting on mounds of dredge spoils which formed islands within the brackish ponds. They also provide food in the form of small fish and aquatic plants such as *Ruppia maritima*. They support a high density and diversity of shorebirds. On the other hand, the Andrade Mesa Wetlands appear to be less important areas for shorebirds, perhaps due to their intermittent inundation patterns, smaller lagoons for phalarope species and less availability of playa areas for small and medium size shorebird species.

4.6. Conclusions

Our study confirms the relevance of the CRD natural and artificial wetlands for shorebirds along the Pacific Flyway. Some species appear to be specialists regarding habitat selection, having been detected only in the inland artificial wetlands (Cerro Prieto geothermal power plant, Cienega de Santa Clara and Mesa de Andrade Wetlands). Moreover non-intertidal wetlands represent alternative feeding sites for nearly all the species of shorebirds observed. They provide important brackish habitats and food chains that were formerly associated with the intertidal zone wetlands when seasonal fresh water flows still reached the intertidal zone. Hence, maintaining the habitat value of the CRD for shorebirds will require active management steps to preserve the size and habitat quality of the anthropogenic wetlands in the CRD.

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Table 1. Shorebird survey sites

Site	Area (Ha)	Type of Wetland	Habitats
Bahia Adair (BA)	28,709	Intertidal	Sandy /Rocky shore/estuaries
Golfo de Santa Clara (GSC)	969	Intertidal	Mudflat
Isla Montague (IM)	16,050	Intertidal	Mudflat
Cienega de Santa Clara (CSC)	22,104	Anthropogenic	Playa areas/saltflats
Cerro Prieto (CP)	1,739	Anthropogenic	Lagoons/saltflats
Mesa de Andrade (MA)	42	Anthropogenic	Lagoons

Table 2. Shorebird mean density during the peak migration month of each site

Site	Month	Mean Abundance	95% CI	N	Sampled Area (ha)	Density (ind/ha, CI)
BA	November	1,021 (SD ± 250)	674-1377	2	98.1	10 (7-14)
GSC	February	7,518 (SD ± 7,363)	814-15,850	3	49.0	153 (17-323)
IM	February	4,846 (SD ± 4,353)	80-9,772	3	127.6	38 (1-76)
CSC	August	4,960 (SD ± 4,202)	863-10,783	2	29.4	168 (29-367)
CP	August	7,245 (SD ± 3,751)	3,569-10,921	4	300	24 (12-36)
MA	August	520 (SD ± 300)	104-936	2	29.4	18 (3-32)

Table 3. Total and relative abundance on intertidal zone sites from August 2005 to December 2008, see appendix for species code.

Species	Bahia Adair		Golfo de Santa Clara		Isla Montague	
	Total	Relative Abundance	Total	Relative Abundance	Total	Relative Abundance
AMAV	184	1.30	251	0.46	778	2.29
AMGP	4	0.03	4	0.01	0	0
AMOY	685	4.84	53	0.10	84	0.25
AREN	3	0.02	0	0	0	0
BBPL	606	4.28	871	1.61	489	1.44
BLTU	56	0.40	25	0.05	1	0
BNST	16	0.11	0	0	0	0
CALI	2489	17.58	23357	43.17	18282	53.89
CHAR	84	0.59	17	0.03	13	0.04
DUNL	177	1.25	11	0.02	117	0.34
GRYE	3	0.02	1	0	0	0
KILL	1	0.01	6	0.01	1	0
LBCU	1551	10.95	1124	2.08	2541	7.49
LBDO	0	0	0	0	3	0.01
LESA	1	0.01	0	0	6	0.02
LEYE	11	0.08	6	0.01	0	0
MAGO	1564	11.04	1872	3.46	660	1.95
NUME	1641	11.59	1099	2.03	863	2.54
PAGP	0	0	1	0	0	0

PHAL	0	0	0	0	0	0
PLUV	313	2.21	161	0.30	134	0.39
REKN	186	1.31	3470	6.41	1853	5.46
RNPH	0	0	0	0	0	0
RUTU	227	1.60	18	0.03	0	0
SAND	97	0.68	371	0.69	188	0.55
SBDO	11	0.08	116	0.21	4	0.01
SEPL	52	0.37	10	0.02	8	0.02
SESA	0	0	0	0	0	0
SNPL	121	0.85	76	0.14	2	0.01
STSA	0	0	0	0	0	0
SPSA	0	0	0	0	1	0
SURF	6	0.04	0	0	0	0
TRIN	78	0.55	54	0.10	11	0.03
UNDO	525	3.71	6891	12.74	3582	10.56
WESA	734	5.18	11701	21.63	1003	2.96
WHIM	258	1.82	287	0.53	482	1.42
WILL	2393	16.90	2198	4.06	2803	8.26
WIPH	0	0	0	0	0	0
WIPL	84	0.59	57	0.11	16	0.05
Total	14161		54108		33925	

Table 4. Total and relative abundance on inland wetlands from August 2005 to December 2008 in the Cienega and Cerro Prieto sites and from August 2005 to December 2006 in the Mesa de Andrade site (see appendix for species code).

Species	Cienega		Cerro Prieto		Mesa de Andrade	
	Total	Relative Abundance	Total	Relative Abundance	Total	Relative Abundance
AMAV	3002	7.11	2146	4.08	189	4.61
AMGP	0	0	0	0	0	0
AMOY	0	0	0	0	0	0
AREN	0	0	0	0	0	0
BBPL	0	0	66	0.13	0	0
BLTU	0	0	0	0	0	0
BNST	2998	7.10	1738	3.30	977	23.83
CALI	20940	49.61	4579	8.70	1000	24.39
CHAR	2	0	7	0.01	0	0
DUNL	319	0.76	12	0.02	2	0.05
GRYE	8	0.02	23	0.04	10	0.24
KILL	54	0.13	4	0.01	35	0.85
LBCU	3	0.01	20	0.04	1	0.02
LBDO	0	0	0	0	4	0.10
LESA	9	0.02	599	1.14	117	2.85
LEYE	0	0	5	0.01	16	0.39
MAGO	1110	2.63	3	0.01	0	0
NUME	4	0.01	21	0.04	0	0

PAGP	0	0	0	0	0	0
PHAL	2355	5.58	24365	46.28	200	4.88
PLUV	0	0	0	0	0	0
REKN	0	0	12	0.02	0	0
RNPH	71	0.17	15431	29.31	105	2.56
RUTU	0	0	1	0	0	0
SAND	0	0	8	0.02	28	0.68
SBDO	0	0	0	0	6	0.15
SEPL	9	0.02	12	0.02	4	0.10
SESA	0	0	0	0	0	0
SNPL	89	0.21	121	0.23	0	0
STSA	3	0.01	0	0	0	0
SPSA	0	0	9	0.02	0	0
SURF	0	0	0	0	0	0
TRIN	177	0.42	100	0.19	19	0.46
UNDO	8176	19.37	4	0.01	1156	28.20
WESA	2773	6.57	988	1.88	227	5.54
WHIM	10	0.02	1	0	0	0
WILL	26	0.06	72	0.14	4	0.10
WIPH	74	0.18	2299	4.37	0	0
WIPL	0	0	2	0	0	0
Total	42212		52648		4100	

Table 5. Shorebird similarity indices

1st Sample	2nd Sample	Shared species	Jaccard	Sorenson	Morisita-Horn	Bray-Curtis
BA	CP	19	0.68	0.81	0.08	0.11
BA	CSC	13	0.46	0.63	0.50	0.19
BA	IM	18	0.67	0.80	0.61	0.45
BA	GSC	21	0.84	0.91	0.60	0.34
BA	MA	13	0.50	0.67	0.37	0.22
CP	CSC	15	0.60	0.75	0.25	0.26
CP	GSC	17	0.61	0.76	0.15	0.12
CP	IM	16	0.57	0.73	0.15	0.15
CP	MA	13	0.52	0.68	0.24	0.10
CSC	GSC	12	0.44	0.62	0.91	0.69
CSC	IM	13	0.52	0.68	0.94	0.64
CSC	MA	12	0.60	0.75	0.80	0.17
GSC	IM	17	0.65	0.79	0.91	0.71
GSC	MA	11	0.42	0.59	0.66	0.09
IM	MA	11	0.44	0.61	0.63	0.14

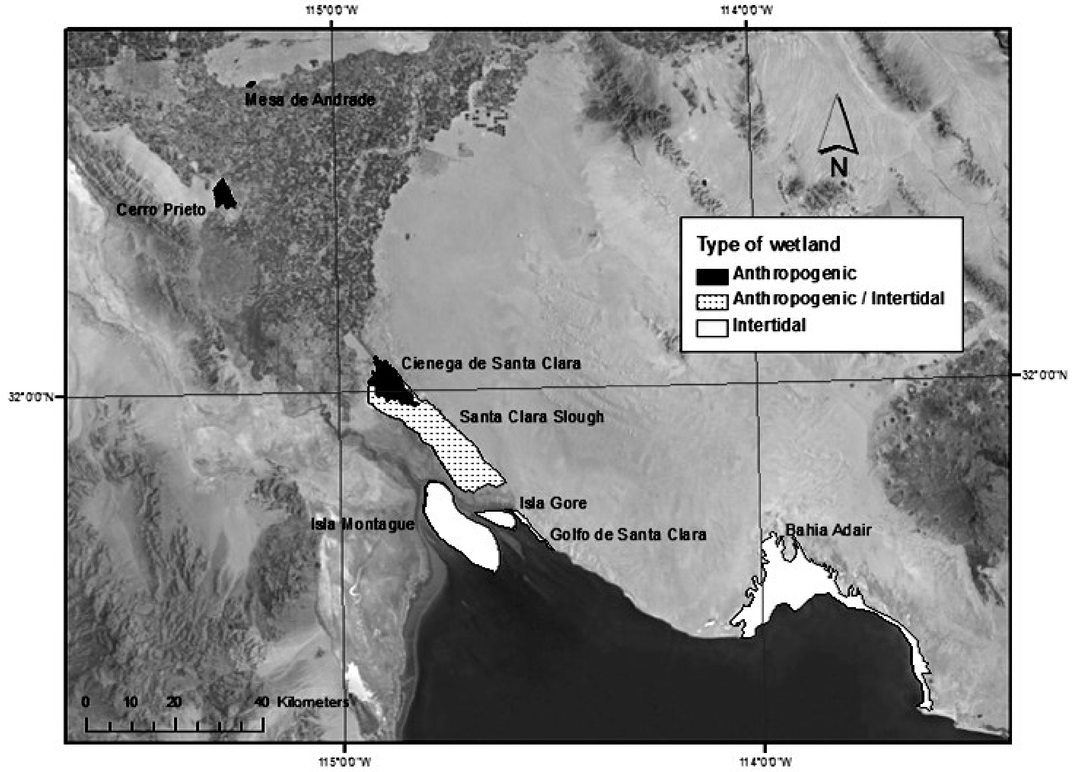

Figure 1. Survey sites at the Colorado River Delta.

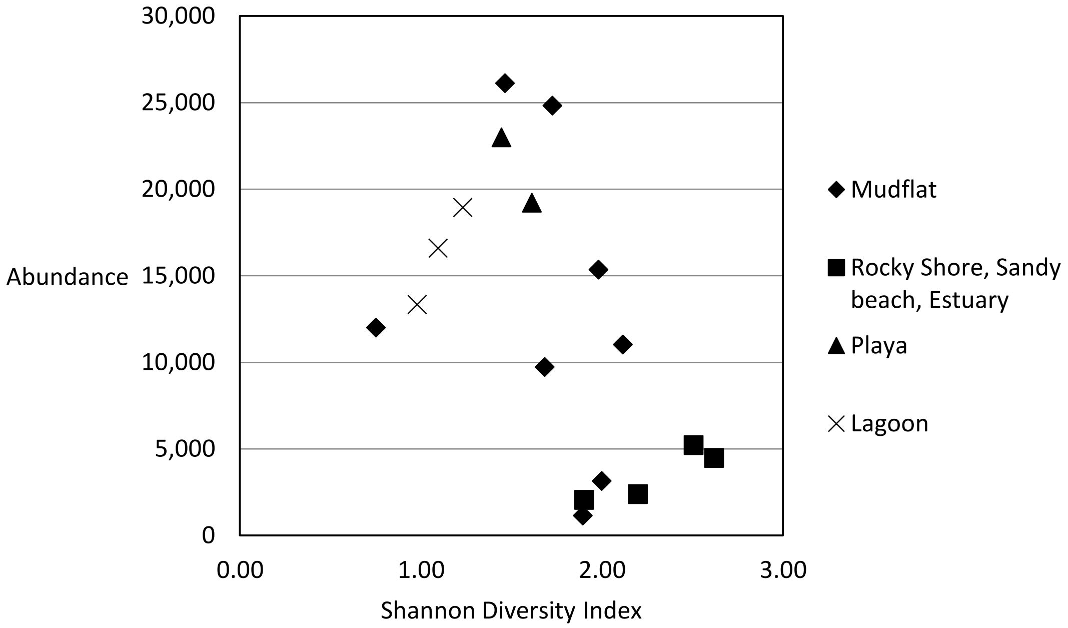

Figure 2. Relationship between the Shannon diversity index and abundance per year in the wetlands, mudflat corresponds to Golfo de Santa Clara and Isla Montague, Rocky shore, Sandy beach and Estuary to Bahia Adair, Playa to the Cienega de Santa Clara and Lagoon to Cerro Prieto site.

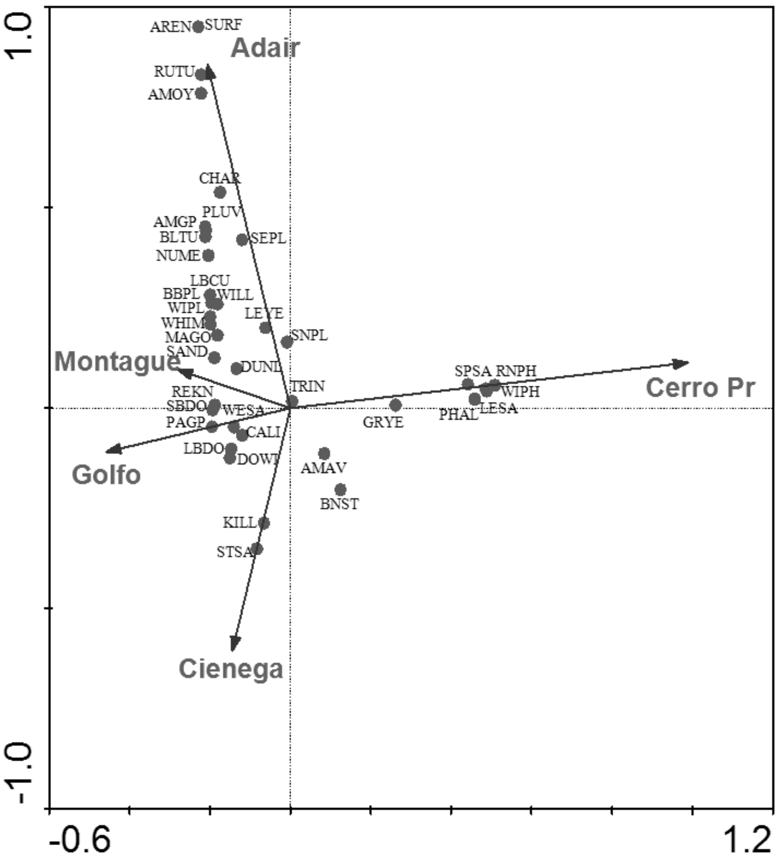

Figure 3. Ordination diagram of species/taxa presence and abundance on the sites from Canonical Correspondence Analysis. The letters refer to species/taxa code showed on appendix 1.

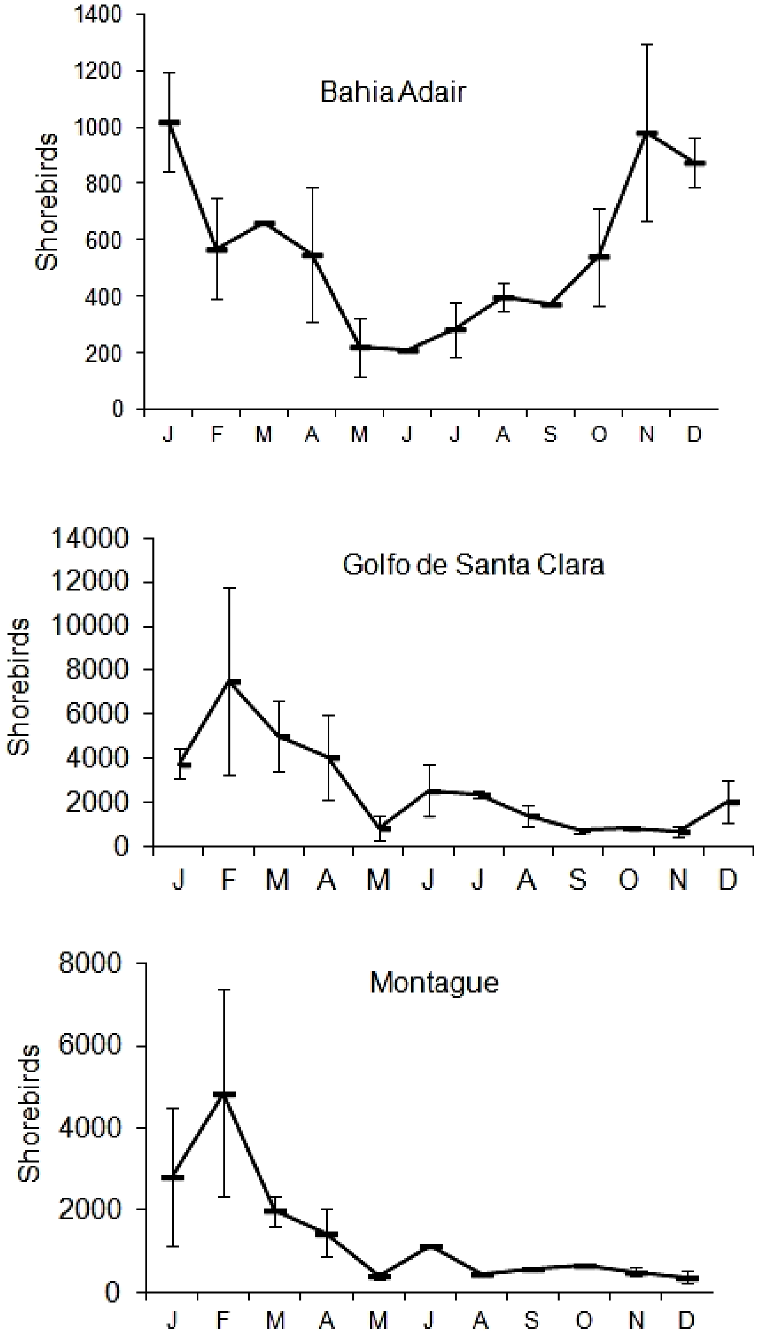

Figure 4. Shorebirds mean seasonal abundance and standard error at the intertidal wetlands, from August 2005 to December 2008.

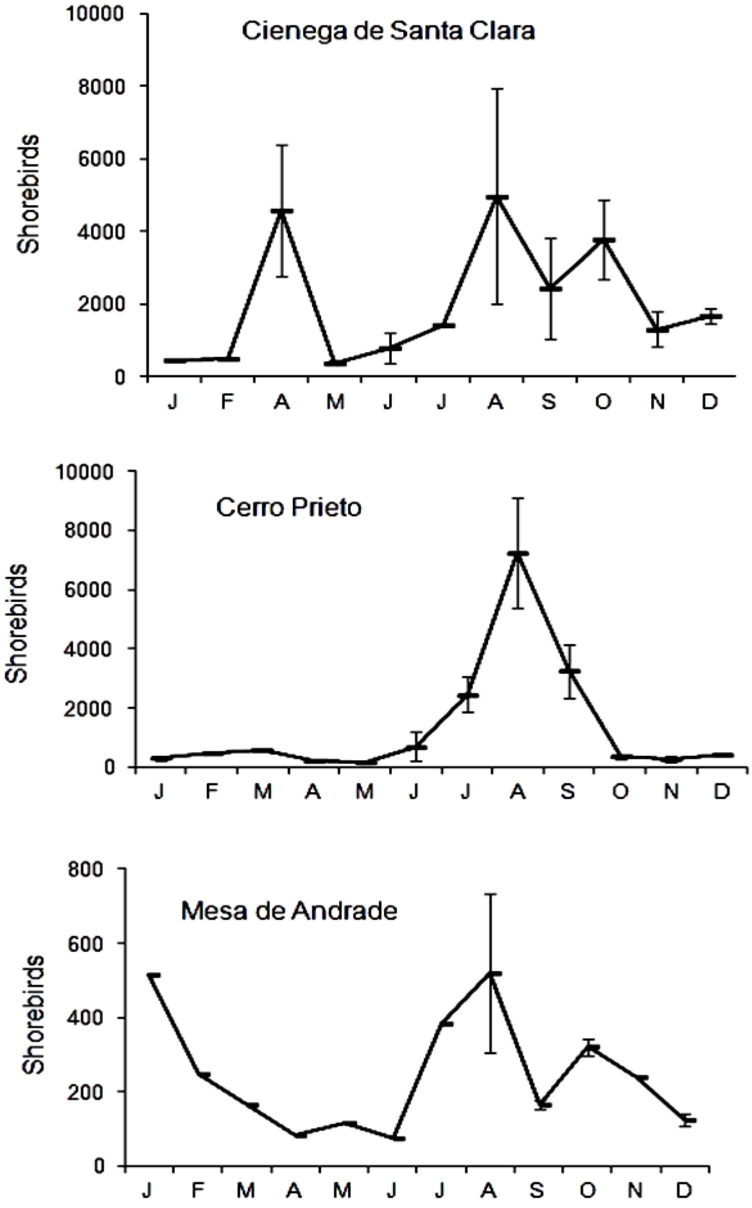

Figure 5. Shorebirds mean seasonal abundance and standard error at the inland wetlands, from August 2005 to December 2008.

Appendix 1. List of species

Code	Common name	Scietific name
AMAV	American Avocet	*Recurvirostra americana*
AMGP	American Golden Plover	*Pluvialis dominica*
AMOY	American Oystercatcher	*Haematopus palliatus*
AREN	Turnstones	*Arenaria sp.*
BBPL	Black bellied plover	*Pluvialis squatarola*
BLTU	Black Turnstone	*Arenaria melanocefala*
BNST	Black necked Stilt	*Himantopus mexicanus*
CALI	Sandpipers	*Calidris mauri,minutilla, alba, alpina* and others sandpipers shorebirds size
CHAR	Plovers	*Charadrius semipalmatus, C. melodus, C. nivosus* and plovers of similar size.
DUNL	Dunlin	*Calidris Alpina*
GRYE	Greater Yellowlegs	*Tringa Melanoleuca* and *T. flavipes*
KILL	Killdeer	*Charadrius vociferus*
LBCU	Long-billed Curlew	*Numenius americanus*
LBDO	Long-billed Dowitcher	
LESA	Least Sandpiper	*Calidris minutilla*
LEYE	Leaser Yellowlegs	*Tringa flavipes*
MAGO	Marbled Godwit	*Limosa Fedra*
NUME	Long-billed Curlew and Whimbrel	*Numenius americanus* or *N. phaeopus*
PAGP	Pacific golden Plover	*Pluvialis fulva*
PHAL	Pharalopes	*Phalaropus tricolor, P. lobatus* and *P. fulicaria*
PLUV	Plovers	*Pluvialis sp.*
REKN	Red Knot	*Calidris canutus*
RNPH	Red-necked Phalarope	*Phalaropus lobatus*
RUTU	Ruddy Turnstone	*Arenaria interpres*
SAND	Sanderling	*Calidris alba*
SBDO	Short-bill Dowitcher	*Llimnodromus griseus*
SEPL	Semipalmated Plover	*Charadrius semipalmatus*
SESA	Semipalmated Sandpiper	*Calidris pusilla*
SNPL	Snowy Plover	*Charadrius alexandrinus*
STSA	Stilt Sandpiper	*Calidris himantopus*
SPSA	Spotted Sandpiper	*Actitis macularia*
SURF	Surfbird	*Aphriza virgata*
TRIN	Yellowlegs	*Tringa flavipes* or *melanoleuca*
UNDO	Unknown Dowitcher (long-billed or short-billed)	*Limnodromus scolopaceus,* and *L. griseus*
WESA	Western Sandpiper	*Calidris mauri*
WHIM	Whimbrel	*Numenius phaeopus*
WILL	Willet	*Tringa semipalmata*
WIPH	Wilson Phalarope	*Phalaropus tricolor*
WIPL	Willson´s Plover	*Charadrius wilsonia*

APPENDIX B

Modeling water management scenarios for the Cienega de Santa Clara, an anthropogenic coastal desert wetland system, based on inflow volumes and salinities

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Abstract

The Cienega de Santa Clara in the Colorado River delta, Mexico is a self-designed wetland system fed since 1977 by brackish groundwater diverted from the U.S. to Mexico. The vegetated upper portion of the Cienega provides habitat for endangered Yuma Clappers Rails and other marsh birds and fulfills other ecological functions. Outflow water pools in the Santa Clara Slough south of the Cienega and provides habitat for migratory shorebirds. Conditions in the Cienega and Santa Clara Slough could be altered by operation of the Yuma Desalting Plant (YDP), which will divert water from the Cienega and replace it with brine water resulting from the desalting process. Our objective was to integrate water budget components into models predicting the extent of the dominant vegetation (southern cattail, *Typha domingensis* Pers.) in the marsh and the area of the outflow pool below the marsh in response to different operating scenarios for the YDP. The models are intended to serve as tools for resource managers charged with maintaining this wetland complex. Unlike many wetland water budget models, this one explicitly takes into account salinity as a factor in the water budget. We modeled inflow rates ranging from 1 to 6 m^3 s^{-1} and inflow salinities ranging from 0 to 6 g L^{-1} Total Dissolved Solids. The model indicates that if the inflow rate is reduced below the current 4-5 m^3 s^{-1} the vegetated area of the Cienega would decrease in proportion, as would the area of the outflow pool in the Santa Clara Slough. Increases in salinity will also reduce the vegetated area due to the low salt tolerance of *T. domingensis*. In winter about 90% of inflow water exits the Cienega into the Santa Clara Slough due to low evapotranspiration, and on an annual basis 70% of inflows exit into the Santa Clara Slough. These flushing flows maintain the salt balance in the Cienega. The Santa Clara Slough is periodically flushed by spring tides, making this a sustainable, open wetland system in its present state.

Keywords: Wetland water budget, salinity, Colorado River delta, wetland water management, agricultural return flows, wetland model, arid zones wetland.

1. Introduction

1.1. Background

The Cienega de Santa Clara is a large self-designed wetland system of about 20,000 ha in the eastern portion of the Colorado River delta in Mexico that depends in part on brackish water delivered from United Stated to Mexico (Mexicano et al., 2013; Glenn et al., 2012; Greenberg and Schlatter, 2012) (Figure 1). The wetland system consists of a vegetated marsh of about 5000 ha dominated by southern cattail (*Typha domingensis* Pers.) (designated the Cienega in this paper) and a shallow, unvegetated pool of variable area south of the vegetation, fed by Cienega outflows and tide water (designated the Santa Clara Slough in this paper) (Glenn et al., 1992, 1995, Zengel et al., 1995). Both are within the core area of the Biosphere Reserve of the Upper Gulf of California and Delta of the Colorado River, and are internationally significant for providing marsh bird habitat in the Cienega in spring and summer and shorebird habitat in the Santa Clara Slough in winter (Hinojosa-Huerta, 2001a, b; Hinojosa-Huerta et al., 2002; Greenberg and Schlatter, 2012; Gomez-Sapiens et al., 2013). Other threatened and endangered species are also found in the Cienega (e.g., Zengel and Glenn, 1996).

Most water enters the Cienega in the Main Outlet Drain Extension (MODE) canal (also called the Bypass Drain), which since 1977 has sent brackish agricultural return flows from the Wellton-Mohawk Irrigation District in the U.S. to the delta in Mexico for disposal (Greenberg and Schlatter, 2012). A much smaller flow enters from the Riito Drain, which collects local agricultural drain water (see Table 1). The MODE discharge was intended to be temporary pending completion of the Yuma Desalination Plant (YDP), which was designed to desalinate MODE water for delivery to Mexican agricultural fields. The MODE was then intended to carry brine from the YDP, containing salts from the desalination process, for disposal in Mexico (Leitz and Ewoldsen, 1977; Haugseth, 1978; Lohman, 2003). However, since its completion in 1992 the YDP has only been operated during a few brief test runs and the flow of brackish agricultural return water has continued largely without interruption, contributing to development of a wetland of international importance in the delta (Greenberg and Schlatter, 2012).

US and Mexican agencies collaborated to implement a monitoring program from 2009 to 2011 with the aim of gathering information about the hydrology, weather, physical attributes and biological resources of the wetland (Peters et al., 2009; Greenberg and Schlatter, 2012). This information is expected to be used to understand the main ecosystem processes and analyze management alternatives for the Cienega. In particular, information is needed on effects to the Cienega should the YDP become operational. A range of operating scenarios for the YDP have been contemplated (e.g., Blank, 2007), with different potential impacts on the volume and salinity of flows into the Cienega. The present study developed simple models to predict the changes in the area of the dominant vegetation type, *Typha domi*ngensis (southern cattail) in the Cienega and the surface area of the outflow pool in the Santa Clara Slough in response to different operating scenarios of the YDP.

1.2. Importance of salinity in wetland water budgets

One of the basic approaches in the understanding of wetlands structure and function is to estimate the water balance (Mitsch and Gosselink, 2000). Once the water balance is established it is possible to model the response of the wetland to hydrological and weather parameters, to simulate the implementation of management schemes and to understand the response of the ecosystem to natural events (Carter, 1986; Zacharias et al., 2005; Zhang and Mitsch, 2005; Huckelbridge et al., 2010; Jia et al., 2011).

In arid and semiarid ecosystems, high evapotranspiration rates can increase the salinity of wetlands and other sensitive ecosystems (Cramer and Hobbs, 2002; Jolly et al., 2008). Salinity effects are especially important in wetlands such as the Cienega that receive saline agricultural return flows. Problems of elemental toxicity and elevated salinities have developed in some wetlands receiving saline agricultural return flows, calling into question their sustainability and safety (Lemly et al., 1993; Lemly, 1994). Even though the transport of salinity is critical in these systems, in most cases, the salt transport pathways in wetlands are not very clear and are rarely incorporated in water budget models (Bauer et al., 2006; Jolly et al., 2008; Jia et al., 2011). In the Cienega, the salts (mainly sodium, calcium, chlorine and sulfate) enter through the inflow at a 2-3 g L^{-1} Total Dissolved Solids (TDS) average concentration (Haugseth, 1978). Due to the relatively low salt-tolerance of the dominant vegetation (Baeza et al., 2013), we considered salt inputs need to be included in a water budget model for the Cienega.

1.3. Aims and objectives of the study

The aim of this work was to integrate water budget variables, to include inflow rate, outflow volume, salinity concentration of the inflow, vegetation salinity threshold and evapotranspiration (ET), into simplified models that can be useful to predict the responses of the Cienega and Santa Clara Slough to different scenarios of water quality and volumes. The Cienega de Santa Clara wetland system is isolated from surrounding systems with relatively well-defined inputs and outputs, and baseline data were available from 2009-2011 to parameterize and partially validate the models. The results present a case study of how alterations in flow and salinities due to land use change or climate change can impact both anthropogenic and natural brackish coastal wetlands, and how management practices can mitigate damage.

1. Methods

2.1. System description

The Cienega is an emergent marsh, 89% of which is vegetated. The remainder of the marsh is composed of 543 open-water lagoons scattered amidst the emergent plant stands, ranging in size from < 1 ha to about 20 ha in area. *T. domingensis* accounts for about 90% of vegetation cover and patches of common reed (*Phragmatis australis*) make up about 7% of the vegetation, growing in shallow areas within the marsh (Glenn et al., 1992; Zengel et al., 1995; Mexicano et al., 2012). Twenty other hydrophytes occur within and around the Cienega (Glenn et al., 1995; Zengel et al., 1995). Mean water depth is 0.32 m but exceeds 1 m in the open water lagoons, and is only a few cm deep along the perimeter, which changes location according to inflow rates and rates of evaporation (Greenberg and Schlatter, 2012). Open water lagoons cover 11% of the Cienega and support submerged aquatic plants such as *Najas marinas* and *Ruppia maritima*. The thick *T. domingensis* stands support 80% of the remaining Yuma Clapper Rails (*Rallus longirostris yumanensis*), an endangered marsh bird that nests in the Cienega in spring and summer (Hinojosa-Huerta et al., 2001a, 2002). A smaller number of endangered black rails are also found in the Cienega (Hinojosa-Huerta et al., 2001b).

The lower basin Santa Clara Slough is a shallow (ca 0-0.3 m depth) evaporation lagoon that varies in size in response to the season, tides and water entering from the Cienega (Nelson et al., 2012). In this unvegetated wetland the incoming brackish water mixes with sea water during occasional high tide events (above 5.2 meters, occurring approximately 8-10 times per year) (Nelson et al., 2012). Much of the water that enters the Santa Clara Slough evaporates within the basin, making this a hypersaline wetland when flooded. It is an important feeding station for migratory shorebirds in winter, with annual visitation by several hundred thousand birds per year (Hinojosa Huerta et al., 2013).

The Cienega and Santa Clara Slough are in the most arid portion of the Sonoran Desert in the Lower Colorado River Valley Subdivision (Shreve and Wiggins, 1964). Average annual precipitation is 68 mm, and average temperature of 25°C with rain events in both summer and winter (Miranda-Reyes, et al., 1990).

2.2. General modeling approach

Models were based on the steady-state water balance equation for wetlands (Mitsch and Gosselink, 2000):

$$SW_{IN} + P + GW_{IN} - ET - SW_{OUT} - GW_{OUT} - \Delta S = 0 \qquad (1)$$

where SW_{IN} is surface water inflow, P is precipitation, GW_{IN} is ground water in, ET is evapotranspiration, SW_{OUT} is surface water outflow, GW_{OUT} is ground water outflow and ΔS is change in storage in the wetland (shown schematically in Fig. 2).

Salinity relationships were based on equations developed by Jia et al. (2011), who defined a coefficient, β, as:

$$\beta = SW_{OUT}/SW_{IN} = C_{IN}/C_{W} \qquad (2)$$

where C_W is the mean salinity in the wetland, and C_{IN} is the salinity of the inflow water. Equation (2) accounts for the increase in salinity that occurs as water from the wetland is discharged in ET, which reduces the outflow volumes and increases the salinity in the wetland, because pure water is discharged in ET. As an example, if half the inflow water exits the marsh as surface flows or net groundwater discharge, and the other half is discharged in ET, $\beta = 0.5$, and if the inflow salinity is 2 g L^{-1}, the mean wetland salinity is 4 g L^{-1}. In the Jia (2011) model C_W is corrected for the uptake of salt into vegetation, which is harvested in their model system, but in the Cienega salts in vegetation are expected to be a small term in the overall water budget. Based on an annual production rates and salt contents of *T. domingensis* reported in Baeza et al. (2013), uptake of salts into vegetation could account for only 1.2% of salts in inflow water, and these salts are expected to be recycled within the wetland as litter decays or burns, hence this factor is not included in Equation (2). Groundwater exchange is also not included in Equation (2) due to lack of data (Flessa and Garcia-Hernandez, 2007). However, groundwater exchanged is also expected to be a small term compared to inflows, and no net exchange of salts between surface water and groundwater is expected at equilibrium (Jia et al., 2011). Equation (2) allowed us to estimate ET and SW_{OUT} from C_{IN} and C_W.

Values for the water balance components were based on monthly measurements made over a 33-month period, 2009-2011, and are listed in Table 1 (Greenberg and Schlatter, 2012; Glenn et al., 2012). The area of vegetation and open water were determined on a April 24, 2009 Quickbird (0.6 m resolution, Digital Globe, Inc., Longmont, CO) satellite image (Mexicano et al., 2012). The total area (4913 ha) ($A_{WETLAND}$) contained 4395 ha of vegetation (A_{VEG}), mostly *T. domingensis*, and 518 ha of open water lagoons (A_{WATER}). Total flows into the Cienega were approximately 2.7

x 10^8 m^3 during the 33 month measurement period, 17 times greater than the volume of the Cienega (approximately 1.6 x 10^7 m^3), justifying the assumption that changes in storage volume (ΔS) were negligible compared to inflows and outflows over annual or longer time periods, a requirement for steady-state water budgets (Jia et al., 2011). GW_{IN} and GW_{OUT} were not measured, but were assumed to be in equilibrium over annual or longer time periods (Jia et al., 2011). This assumption was supported by a mass balance of surface flows, showing that inflows minus outflows could be accounted for by losses due to ET as estimated by a remote sensing technique (Glenn et al., 2012).

ET was divided into two components, transpiration by vegetation (E_{VEG}) and open water evaporation (E_{WATER}). E_{VEG} and E_{WATER} were expressed in rate units of m^3 m^{-2} $month^{-1}$, and total ET in units of m^3 $month^{-1}$ was calculated as:

$$ET = (0.89\ E_{VEG} + 0.11\ E_{WATER})A_{WETLAND} \qquad (3)$$

Equation (3) holds the proportion of vegetation to open water, measured in 2009, constant over the different scenarios modeled in this study.

E_{VEG} was estimated at 16-day intervals using the Enhanced Vegetation Index (EVI) from the Moderate Resolution Imaging Spectrometer (MODIS) sensors on the Terra satellite and estimates of potential ET (ET_o) calculated by the Blaney-Criddle formula (Brouwer and Heibloem, 1986) from temperature data from the Yuma Valley AZMET station (AZMET, 2012), about 100 km from the Cienega in Arizona, U.S. (Glenn et al., 2012). E_{VEG} was calculated using the method developed by Nagler et al. (2009):

$$E_{VEG} = 1.22ET_o(EVI^*) \qquad (4)$$

where EVI* is EVI scaled between bare soil EVI (EVI_{min}), assumed to have zero transpiration, and the EVI of dense, unstressed vegetation (EVI_{max}), assumed to be transpiring at the rate of ET_o. EVI* was calculated as:

$$EVI^* = 1-(EVI_{max} - EVI)/(EVI_{max} - EVI_{min}) \tag{5}$$

EVI values of 0.091 and 0.542 were used for EVI_{min} and EVI_{max} based on a large data set from previous studies (Nagler et al., 2005a,b). 1.22 is a regression coefficient relating E_{VEG} to ETo and EVI* based on ground values of E_{VEG} measured for riparian and agricultural plants on the Lower Colorado River in a previous study (Nagler et al., 2009). It had a standard error of 20% in predicting E_{VEG} of different plant types in that study.

E_{WATER} was assumed to be equal to ET_o (Huckelbridge et al., 2010; Glenn et al., 2012). As justification, annual evaporation from the Salton Sea (150 km from the Cienega), for which evaporation data are available, is 1798 mm per year (Ponce, 2005), compared to our estimate of ET_o of 1820 mm per year for the Cienega based on Yuma AZMET data (AZMET, 2012). SW_{IN} was the sum of MODE canal water (data from the International Boundary and Water Commission, El Paso, Texas), precipitation (from the Yuma Valley AZMET station) and sporadic flows in the Riito canal carrying local irrigation return flows (gaged during the study period) (Greenberg and Schlatter, 2012). During the study period, MODE canal water accounted for over 95% of SW_{IN}. Salinity data for MODE water was from the International Boundary and Water Commission. Salinity in the Riito canal and at 22 points in the Cienega were measured during field surveys monthly with hand-held electrical conductivity meters calibrated with gravimetric measurements of TDS in the laboratory (Greenberg and Schlatter, 2012).

2.3. Modeling vegetated area in response to inflow volumes and salinities

Different approaches were used to model the areal extent of *T. domingensis* vegetation in the Cienega and the outflows from the Cienega. The Cienega vegetation model was a steady state, empirical model based on the salt tolerance limit of *T. domingensis*, which determined the fraction of inflow water that could support *T. domingensis* growth and ET:

$$SW_{IN\text{-}Usable} = SW_{IN}\,(C_{Threshold} - C_{IN})/C_{Threshold} \qquad (6)$$

where $SW_{IN\text{-}usable}$ is the flow rate of inflow water that can support transpiration of *T. domingensis* (e.e., E_{VEG}), C_{IN} is salinity of inflow water and $C_{Threshold}$ is the salinity limit for growth and transpiration of *T. domengensis*. Eq. (6) was originally developed to predict annual ET and yield of crop plants grown on saline water based on the salt tolerance threshold of a given crop (Shani and Dudley, 2001; Dudley et al., 2008). Its application to the Cienega was validated in Glenn et al. (2012) which used mass balance and remote sensing methods to compute water balance components of the marsh. Based on greenhouse and field studies in the Cienega (Glenn et al., 1995; Baeza et al., 2013) and literature values (Hocking, 1981; Beare and Zedler, 1987), $C_{Threshold}$ was set at 6.0 in our model. As a sample calculation, if water enters at 3 g L^{-1} TDS, the usable fraction is (6-3)/6, or 0.5. The remainder is assumed to exit the Cienega as outflow, or to be discharged as evaporation from open water lagoons (E_{WATER}). When C_{IN} is 0 g L^{-1} TDS, $SW_{IN\text{-}Usable}$ is 1.0, while when C_{IN} is 6.0 g L^{-1} TDS, $SW_{IN\text{-}Useable}$ is 0.0.

The maximum area of vegetation that could be supported by a given volume and salinity of inflow water was calculated based on the period 2009-2011, taken as a baseline, during which time SW_{IN} was 4.97 $m^3 s^{-1}$, C_{IN} was 2.62 g L^{-1} TDS, area of vegetation was 4395 ha, and $SW_{In\text{-}Usable}$ was calculated by Equation (6) as 0.56 over the June-September period of maximum E_{VEG}, when water availability was assumed to be limiting for determining the area of the wetland (Glenn et al., 2012). Then for any given scenario the area of vegetation was calculated as:

$$A_{VEG} = 4395\,[(6 - C_{In})/6]/0.56\,(SW_{IN}/4.97) \qquad (7)$$

and $A_{WETLAND}$ is 1.12 (A_{VEG}) to account for open water areas

Several simplifying assumptions are embedded in Eq. (7). First is that *T. domingensis* will continue to be the dominant vegetation type determining vegetated area under different salinity scenarios. However, the wetland also supports *Schoenoplectus* spp. (bulrushes), which are more salt tolerant than *Typha* spp. (Baeza et al., 2013) and they could replace *T. domingensis* in portions of the Cienega. Dense *T. domingensis* stands are required for support of breeding Yuma Clapper Rails (Hinojosa-Huerta et al., 2001a, 2002; Conway et al., 1993), hence the model provides a fair description of habitat availability for this and other marsh birds that require dense *Typha* stands, but cannot predict the eventual species composition of the marsh under altered salinity conditions. A second assumption is that E_{VEG} can be treated as a constant within the usable salinity range and A_{VEG} is only determined by the amount of usable water available in summer. However, at elevated salinities, E_{VEG} could be expected to decrease, leading to lower water use on a unit area basis and thus more areal coverage of vegetation at the high end of the salinity range than predicted by the model.

Conversely, at lower salinities E_{VEG} could increase, leading to overestimates of vegetation cover at reduced salinities. However, E_{VEG} under reference conditions was only 50% of ET_o based on MODIS imagery and meteorological data, and appeared to be light-limited due to the accumulation of thatch within the stands (Glenn et al., 2012). Furthermore, when thatch was removed by fire in other years, peak ET over the wetland was equal to ET_o despite salinity constraints (Glenn et al., 2012). Hence, a reduced rate of E_{VEG} is already built into the reference conditions and no correction was applied to E_{VEG} under altered salinity conditions. Model assumptions and sources of error and uncertainty are discussed further in Section *4.3*. Note also that Eqs. (2), (6) and (7) offer means of estimating E_{VEG} and E_{WATER} for comparison with MODIS estimates (see Section *3.1*).

2.4. Model for determining outflows and area of the outflow pool

Unlike A_{VEG}, which is expected to change slowly in response to inflows, SW_{OUT} changes throughout the year according to environmental evaporation rates and seasonal water demands of the vegetation. Therefore, SW_{OUT} was modeled in monthly time steps. According to Equation (1), and assuming changes in groundwater and surface water storage are negligible at monthly time steps, SW_{OUT} can be estimated as:

$$SW_{OUT} = SW_{IN} - (E_{VEG} \text{ x } A_{VEG} + E_{WATER} \text{ x } A_{WATER}) \qquad (8)$$

where A_{VEG} is calculated by Eq. (8). Monthly rates of ET_{VEG} and ET_o used in the model are in Table 2. As examples, for the month of January, 2009 the calculation is:

SW_{OUT} (m^3 $month^{-1}$) = 1.237×10^7 m^3 – ($0.025 m^3 m^{-2} \times 4.395 \times 10^7 m^2 + 0.101 m^3 m^{-2} \times 5.18 \times 10^6 m^2$) = 1.075×10^7 m^3 $month^{-1}$

(9)

whereas for June, 2009 the calculation is:

SW_{OUT} (m^3 $month^{-1}$) = 1.33×10^7 – ($0.121 m^3 m^{-2} \times 4.395 \times 10^7 m^2 + 0.217 \times 5.18 \times 10^6 m^2$) = 6.442×10^6 m^3 $month^{-1}$

(10)

These examples show that in winter, when ET is low, most of the water exits the Cienega as outflow (87% in January, 2009), whereas in summer, when ET is high, outflow is reduced (51% in June, 2009), presumably representing the fraction that is too saline to support E_{VEG}.

2.5. Estimating the area of the outflow pool in the Santa Clara Slough

The area of the Santa Clara Slough tide basin is approximately 36,000 ha but the inundated area is normally much smaller. The area of the outflow pool is dependent on the rate of outflow from the Cienega, the water depth, the evaporation rate, and the geometry of the basin. None of these variables were directly measured. Furthermore, the Santa Clara Slough also receives water from extreme high tide events that bring in water from the Gulf of California. When flooded to its maximum extent the water surface of the Santa Clara Slough covers as much as 36,000 ha, but the pool area is typically much smaller, ranging from 0 in summer to approximately 15,000 ha following extreme spring tides in winter.

We approximated the pool size that could result from outflows from the Cienega and compared them to satellite imagery obtained at different times of year to attempt to determine the contribution that Cienega outflows make to shorebird habitat. We modeled the basin as a section of a sphere with a mean depth of 0.15 m, varying from 0 m at the margins to 0.3 m in the center (Fig. 3), based on estimates from aerial and ground observations at different times of year by anecdotal evidence from several observers who have canoed through the Slough when it is full (Nelson et al., 2012). The controlling equations (Fig. 3) show that as water evaporates, the surface area decreases in direct proportion to the decrease in water depth. An iterative procedure was used to estimate the potential water surface area attributable to Cienega outflows in monthly time steps. For the initial month (January 2009), an initial area was calculated based on the volume of inflows and assuming an initial maximum depth (**h** in Fig. 3) of 0.30 m. Then the monthly rate of ET_o (Table 1) was used to decrease the area in proportion to the amount evaporated during the month. The final pool size at the end of the month was then added onto the next month's initial area determined by that month's inflows, and the final area was adjusted downward according to that month's ET_o. The hypothetical pool areas were estimated for the period 2009-2010, using an initial value of 4400 ha based on January 2009 inflows and assuming no carry-over from the previous month. Then the December, 2010 pool area was used as input to subsequent iterations of the model. Three iterations of the calculation chain were sufficient to produce stable monthly values.

2.6. Santa Clara Slough pool area estimated by MODIS imagery

MODIS images are acquired at near-daily intervals at approximately 10:30 a.m. in descending mode in the Colorado River delta. A collection of daily images representing different tide conditions from 2009 to 2011 was obtained from NASA's Rapid Response website (http://earthdata.nasa.gov/data/near-real-time-data/rapid-response/modis-subsets). False color images combining Bands 7 (mid infrared), 2 (near infrared) and 1 (red) were analyzed as this band combination adequately differentiates water, soil and vegetation (Eyton, 2003). Water appears as false-color blue in this band combination, with light blue corresponding to shallow water (with light-colored soil visible underneath) and dark blue corresponding to water deeper than approximately 0.3 m in the case of the Cienega. We prepared a mask of the Santa Clara Slough area encompassing 36,000 ha, and used a supervised classification program in ERDAS software (ERDAS Imagine, Inc.,, Atlanta, GE) to divide the images into four classes, representing dry soil, wet soil, Water 1 (approximately 0.15-0.3 m or deeper) and Water 2 (0-0.15 m deep). Training sites were selected on images based on visual inspection of images and knowledge of water depths from ground and aerial surveys, and a signature file was developed that was then applied to all images analyzed.

3. Results

3.1. Inflows, outflows, salinities and model validation during the reference period

Measured inflow volumes and calculated outflows based on Eqs (3) and (4) (MODIS ET estimates) are in Fig. 4A, and measured inflow salinities and wetland salinities measured at 22 stations are in Figure 4B. Inflow volumes and salinities tended to be variable over the study period, with some months in which inflows were reduced due to low flows in the MODE canal, and periods of increased inflow salinity in summer, 2010 due to operation of the YDP. Salinities at the 22 stations ranged from 2.4 to 12 g L^{-1} TDS over the study. ET can also be estimated by Eqs. (1) and (2) by mass balance of salts and inflows (Glenn et al., 2012). Over the 33 month period in which inflows and salinities were monitored (2009-2011), mean inflow salinity was 2.62 g L^{-1} TDS and mean wetland salinity was 3.73 g L^{-1} TDS (Table 1), for a β value of 0.70. Therefore the fraction of inflow water that was lost in annual ET was estimated as (1.0-0.7), or 0.30. From Table 1 inflows were 4.97 m^3 s^{-1} and wetland area was 4913 ha, hence by Eq. (6) ET in m^3 m^{-2} $month^{-1}$ was:

$$4.97 \text{ m3 } s^{-1} (60 \text{ sec min}^{-1} \text{ x } 60 \text{ min hr}^{-1} \text{ x } 24 \text{ hr d}^{-1} \text{ x } 30.5 \text{ d month}^{-1}) \, 0.3 / 49{,}130{,}000 \text{ m}^2 = 0.080 \qquad (11)$$

Whereas mean ET by the MODIS method was 0.070 m^3 m^{-2} month over the same period, 13% lower (difference not significant at $P < 0.05$) (Glenn et al., 2012).

Eq. (6) can also be used to estimate E_{VEG} during summer (June-August) and compared to MODIS E_{VEG} for the same period:

$$E_{VEG} = SW_{IN\text{-}Usable}/A_{VEG} \qquad (12)$$

June-August SW_{IN} in 2009 (before operation of the YDP) was 4.01 m^3 s^{-1} and $SW_{IN\text{-}Usable}$ based on Equation (6) was 0.56, and A_{VEG} was 4395, hence E_{VEG} was 0.137 m^3 m^{-2} $month^{-1}$. E_{VEG} by MODIS over the same period was 0.131 m^3 m^{-2} s^{-1}, 4.4% lower. Given the sources of error and uncertainty in both types of estimates, no adjustments to the model were made based on these comparisons.

3.2. Dependence of vegetated area on inflow volumes and salinities

A nomogram relating the projected area of *T. domingensis* to inflow volume and salinities is in Fig. 5. Also shown on the graph is the area of vegetation under current conditions (2009-2010). As expected, at a given salinity the area of vegetation increases with increasing inflows, whereas at a given inflow volume, increasing salinity decreases the vegetated area due to the reduction in $SW_{IN\text{-}usable}$ according to Eq. (6). Note that reductions in salinity can to some extent compensate for decreases in volumes; for example, if salinity is reduced to 2.0 g L^{-1} TDS, the volume of inflows could be reduced from the present 4.97 m^3 s^{-1} to 4.0 m^3 s^{-1} while maintaining the same area of *T. domingensis*. Conversely, if salinity is increased to 4 g L^{-1} TDS, increasing the inflow rate to 6 g L^{-1} TDS would partially compensate for the increase in salinity, but vegetated area would still decrease to approximately 3500 ha. Thus, by this model salinity appears to be a major determinant of the area of *T. domingensis* in the Cienega.

3.3. Dependence of outflow rates on inflow volumes and salinities

Fig. 6 shows modeled monthly outflow volumes at three inflow rates and three salinities. Outflow volume as expected is highest during the winter, and is highly dependent on inflow rate but not on salinity. This is because *T. domingensis* is dormant in winter and E_{VEG} is very low, and about 90% of inflows exit into the Santa Clara Slough. On the other hand, in summer outflows are lower than in winter due to increased ET, but outflow volume increases in response to salinity because E_{VEG} is constrained by salinity. When C_{IN} is 5 g L^{-1} TDS, about 80% of inflow water exits to the Santa Clara Slough in summer, whereas all the water is consumed at C_{IN} of 0 g L^{-1} TDS.

3.4. Effect of outflows on surface area of water in the Santa Clara Slough

Modeling of the pool area in the Santa Clara Slough indicated that outflows from 2009 to 2010 could sustain a peak pool area of about 5000 ha in winter, decreasing to near zero in summer, due to lower outflow volumes from the Cienega and higher rates of E_{WATER} in summer (Fig. 7A). Based on data in Figs. 6 and 7A, projections of winter and summer pool areas due to Cienega outflows at three inflow volumes and salinities are in Fig. 7B. Winter pool areas are mainly dependent on inflow volumes, as expected. However, summer pool areas are also dependent on salinity, with an inflow of 5 g L^{-1} TDS supporting a large pool area, and a inflow of 0 g L^{-1} TDS supporting near-zero pool area regardless of inflow volume. This is because the model assumes that A_{VEG} will expand to consume nearly all of the inflows in summer, because $SW_{In\text{-}Usable}$ is 1.0 at 0 g L^{-1} TDS. This is a reasonable assumption because the vegetated area can expand to the south and along the western perimeter of the Cienega to use all available water.

January and June water surface areas determined by MODIS imagery for 2009 to 2011 are illustrated in Fig. 8 and summary statistics are in Table 3. The images were selected during neap tide conditions when contributions from recent tidal flooding were considered to be minimal. In winter, observed pool areas were about 14,400 ha, of which 50% was in the deeper Water 1 class, whereas summer pool area was only 2500 ha, consisting mostly of the shallow Water 2 class. For comparison, peak pool area attributed to Cienega outflows based on model predictions were 6400 ha in winter and the summer minimum was 100 ha. Thus under current conditions outflow from the Cienega appears to contribute 44% to the area of standing water in the Santa Clara Slough in winter in the absence of recent tidal inflows. Pool areas immediately following high tides > 5 m, occurring 6-8 times per year, are in Fig. 9. Pool areas are expanded, but in summer much of this water evaporates in the interval between flooding, and the contribution from the Cienega is near zero due to high ET.

4. Discussion

4.1. Factors controlling wetland dynamics

The inflow rate, the salinity of the inflow and the ET rate interact to determine the area vegetated by *T. domingensis* and the amount of water that is potentially drained to the lower basin. Changes in A_{VEG} are assumed to occur over annual or longer time periods (Huckelbridge et al., 2010), whereas changes in SW_{OUT} are more dynamic, showing seasonal variability. When a range of salinity values is simulated there is a response to salinity in both the outflow volume and the area of the marsh. As salinity increases there is an increase in the estimated summer outflow due to salinity constraints on *T. domingensis* E_{VEG}; this eventually leads to a contraction in A_{VEG} over time. On the other hand, outflow volumes in winter are high in all salinity scenarios because *T. domingensis* is dormant and ET is low.

Huckelbridge et al. (2010) modeled the response of the vegetation of the Cienega to scenarios of water inflow and salinity integrating the interaction between wetland hydrology, mixing processes, salinity dynamics and vegetation, using a fixed percentage of outflow volume. They concluded the wetland is more sensitive to changes in salinity than to the volume of the inflow. Our approach, based on data collected 2009-2011 during the Cienega de Santa Clara Monitoring program (Greenberg and Schlatter, 2012), supports this conclusion. The sensitivity to salinity is due to the narrow salinity range over which *T. domingensis* can grow, with an upper salinity limit of 6 g L^{-1} (Baeza et al., 2013).

Salts entering in a wetland can be stored in the water or soil, leave the system through the outflow or via plant uptake and harvest (Jia, et al., 2011). In systems where the ratio between the outflow and the inflow approaches 0 the accumulation of salts will increase to high levels. An example of an anthropogenic wetland that accumulated salts and toxic elements due to lack of outflow is Kesterson National Wildlife Refuge in the San Joaquin Valley in California (Wu, 2004). Selenium present in the agricultural return flows that fed the wetland accumulated to toxic levels and the wetland had to be closed. Rather than being an isolated case, Lemly (1994) estimated that about half of wetlands receiving agricultural return flows experience accumulation of salts and toxic elements over time due to inadequate flushing (see also Jia et al., 2011). Typically, drain water follows a seasonal pattern corresponding to the cropping cycle in the agricultural district in which it is generated. By contrast, MODE water is delivered at a relatively constant rate throughout the year (Greenberg and Schlatter, 2012; Mexicano et al., 2012), and the Cienega has an outflow/inflow ratio close to 1 during winter months and an annual ratio of 0.7. This high annual outflow/inflow ratio appears to be functioning to prevent salt accumulation in the Cienega over time (Mexicano et al., 2012). Excess salts from the Cienega flow into the Santa Clara Slough, which is periodically flushed by tides, although it also functions as an evaporation basin, with large amounts of salt crystals visible on the soil between flood events. Hence, the Cienega is an open wetland system discharging into a partially open evaporation basin that is naturally hypersaline.

4.2. Operating scenarios for the YDP

The YDP has an intake capacity of 4.22 $m^3\ s^{-1}$, and at full operation would produce 3.16 $m^3\ s^{-1}$ of desalted water and 1.06 $m^3\ s^{-1}$ of brine effluent containing 8-11 g

L^{-1} TDS depending on the salinity in the inflow water (Leitz and Ewoldsen, 1977; Lohman, 2003). Flows in the MODE averaged 4.30 m^3 s^{-1} (SD =0.46) and salinity averaged 2.59 g L-1 TDS (SD = 0.32) from 1995 to 2010, hence nearly the full volume of MODE water could be diverted to the YDP. Several operating scenarios have been proposed to protect the Cienega ecosystem while operating the YDP to provide desalted water for human use (Blank, 2007). Based on the models developed here, we evaluated four possible scenarios (Table 4). We took as the initial conditions the current inflow rates and salinities, and projected new scenarios to eventual equilibrium conditions. These included full operation of the YDP; operation at 1/3 capacity with placement of YDP brine in the MODE canal for delivery to the Cienega; operation at 1/3 capacity without placement of brine in the MODE canal; and operation at 1/3 capacity without placement of brine in the MODE and with replacement water from other sources provided with 2 g L^{-1} TDS. The third and fourth scenarios would require an alternate method of disposing of the brine.

Table 4 shows that any reduction in inflow volume will proportionately reduce the area of *T. domingensis*, as expected. Salinity also decreases the vegetated area, but because there is an upper limit of 6 g L-1 TDS for support of *T. domingensis*, even a moderate rise in salinity has a large impact on vegetated area. This is seen in the two operating scenarios at one-third capacity; in the first, when the effluent brine is placed back in the MODE, salinity increases by 0.87 g L^{-1} TDS, resulting in a decrease in vegetated area of 847 ha compared to the scenario in which brine is disposed of elsewhere. On the other hand, sources of less saline water from the U.S. and Mexico have been proposed as replacement water, which could support the Cienega but are not suitable for desalting (e.g. treated sewage effluent, containing about 1 g L^{-1} TDS). The fourth scenario shows that such water could actually increase the vegetated area in the Cienega.

4.3. Model potential errors and limitations

The models have two main sources of error and uncertainty. The first source is due to errors inherent in measurements of inflows and salinities and estimates of E_{VEG} and E_{WATER} by MODIS imagery and meteorological data. Despite the apparent good agreement between ET estimated by MODIS and by salinity measurements, both remote sensing and ground methods for estimating wide-area ET in wetlands have errors on the order of 20-30%. (Kalma et al., 2008; Glenn et al., 2011). A detailed comparison of ET estimates by MODIS and mass-balance methods for the Cienega are in Glenn et al. (2012), and they differed by 13%. Hence the error in projected vegetated area in different operating scenarios is likely < 15%. Greater error is likely associated with the estimates of the contribution of Cienega outflows to the surface area of water in the Santa Clara Slough, because outflows were not measured directly, and the topology of the receiving basin is poorly known. Errors in estimating the mean depth of the Santa Clara Slough could be as great as 50%, for example. It can be concluded that Cienega outflows probably make a significant contribution to the surface area of water in the slough, but the exact magnitude of this contribution cannot be estimated based on the relatively coarse approximations in the present model.

The second source of error and uncertainty is the validity of the assumptions built into the models, which determines how well they predict future conditions if inflow volumes and salinities to the Cienega are altered. The predictions of vegetated area depend on how well Eqs. (6) and (7) describe the relationship between A_{VEG} and C_{IN}. Dudley et al. (2008) found that while E_{VEG} of crop plants might decrease on a rate basis due to increased salinity, the final annual yield and crop water use could still be predicted from Eq. (6), in which the limiting factor over a growing season is the total amount of usable water available. Lowered rates of E_{VEG} were compensated by slightly longer crop cycles in their field experiments. We assume similar considerations apply to the Cienega. In support, June-August values of salinity in the Cienega are in the range of 5-8 g L^{-1} near the discharge points (Greenberg and Schlatter, 2012), similar to the value of 6 g L^{-1} assigned to $C_{Threshold}$ in Eq. (6), indicating the vegetation is consuming the usable fraction of water in summer. However, there were insufficient reference data on which to test the full range of predictions modeled here, because inflow volumes and salinities have been relatively constant since creation of the Cienega in 1977, and interruptions in flow have been temporary. At higher salinities, it is not certain that the Cienega would remain a *Typha* marsh since other emergent species already present in the Cienega have higher salt tolerance and could at least partially replace *T. domingensis* if salinity increases. These include *Schoenoplectus americanus*, *S. maritimus* and *P. australis* (Mexicano et al., 2012). Hence, the models likely provide a reasonable approximation of the response of *T. domingensis* to altered inflows, but the eventual species composition of the marsh under altered inflows is unknown.

4.4. Implications for management

The results suggest some points that should be considered in designing an operating protocol for the YDP while preserving ecological functions in the Cienega and Santa Clara Slough:

- The area of *T. domingensis* is directly determined by the volume and salinity of inflows during the April to October growing season, and reductions in flows or increases in salinity will reduce the area of *T. domingensis*.

-Salinity is especially important in determining the area of *T. domingensis* due to its relatively low $C_{Threshold}$; therefore, disposing of YDP brine in the Cienega would appear to be especially detrimental to the marsh.

-Conversely, sources of replacement water with lower salinity than MODE water could enhance the area of *T. domingensis*.

-High outflow values in winter contribute to the area of shorebird habitat in the Santa Clara Slough and help maintain the salt balance in the Cienega by flushing salts accumulated in summer into the slough.

-High outflow volumes also prevent the accumulation of toxic elements such as boron and selenium to levels of concern (Greenberg and Schlatter, 2012).

-Reduction in size or fragmentation of coastal marshes generally reduces species abundance and diversity in the marsh (e.g. Craig, 2008; Guadagnin et al., 2009). Given the already much diminished wetland area in the delta, preserving the present area of the Cienega and Santa Clara Slough is important to maintaining avian habitat value in the delta

-Stability of the Cienega de Santa Clara ecosystem appears to be sustainable over time due to its connection to the Santa Clara Slough evaporation basin and occasional connection to the sea at high tides.

Unlike some other anthropogenic wetlands supported by agricultural drain water (e.g., Lemly, 1994, Wu, 2004), the Cienega represents a success story that has maintained high-quality wildlife habitat for over 30 years. In summary, its stability can be attributed mainly to the high volume of winter outflows that keeps salts and toxic elements in balance, and its connection to the sea, which makes it an open rather than closed wetland system. Future management scenarios should be designed to preserve and enhance these attributes.

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Table 1. Mean and standard errors of hydrological parameters used in estimating ET by a water budget equation, compared to ET estimating by MODIS satellite imagery (from Glenn et al., 2012).

Parameter	Mean (Std. Error)
Flows In (m^3 s^{-1})	
MODE	4.75 (0.18)
Riito	0.21 (0.08)
Precipitation	0.013
Total In (m^3 s^{-1}):	4.97(0.19)
TDS In (g L^{-1})	
MODE	2.59 (0.06)
Riito	3.46 (0.15)
Precipitation	0.0
Weighted Mean TDS In (g L^{-1}):	2.62 (0.07)
TDS Mean Cienega (g L^{-1})	3.73 (0.08)
ET by Mass Balance (mm d^{-1})	2.62 (0.11)
ET by MODIS (mm d^{-1})	2.97 (0.16)

Table 2. Values for E_{VEG} and E_{WATER} used in models in this study. For a given month, E_{WATER} was assumed to be equal to ET_o estimated by the Blaney-Criddle method based on meteorological data from the Yuma Valley, Arizona, AZMET station.

Month	**E_{VEG} (m^3 m^{-2} $month^{-1}$)**	**E_{WATER} (m^3 m^{-2} $month^{-1}$)**
January	0.025	0.101
February	0.022	0.125
March	0.024	0.138
April	0.040	0.162
May	0.070	0.210
June	0.121	0.217
July	0.152	0.215
August	0.121	0.199
September	0.089	0.170
October	0.057	0.134
November	0.041	0.115
December	0.021	0.095

Table 3. Surface area of water in the Santa Clara Slough, 2009-2011, in January and July under neap tide conditions, based on MODIS imagery. Number in parentheses are standard errors of means.

Pool Area (ha)		
Class	**Winter**	**Summer**
Pool area (ha)		
Water 1	6906 (335)	752 (502)
Water 2	7502 (1184)	1757 (562)
Wet Soil	7888 (550)	3881 (557)
Dry Soil	14,419 (1349)	30,083 (1567)

Table 4. Projections of *Typha domingensis* cover and area of water in the Santa Clara Slough attributable to Cienega outflows under different operating scenarios of the Yuma Desalination Plant.

YDP operating scenario	**Inflow volume to Cienega (m^3 m^{-2} s^{-1})**	**Inflow salinity to Cienega (g L^{-1} TDS)**	**Area *T. domingensis* (ha)**	**Summer outflow pool (ha)**	**Winter outflow pool (ha)**
Full capacity	1.06	10.3	0	1320	1320
One-third capacity, brine placed in MODE	3.16	3.46	2097	2400	2970
One-third capacity, brine not placed in MODE	2.81	2.59	2944	1460	2600
No operation of YDP	4.22	2.59	4395	2200	3900
One-third capacity, brine not placed in MODE, replacement water of 1 g L^{-1} TDS added	4.22	2.19	5074	1947	3850

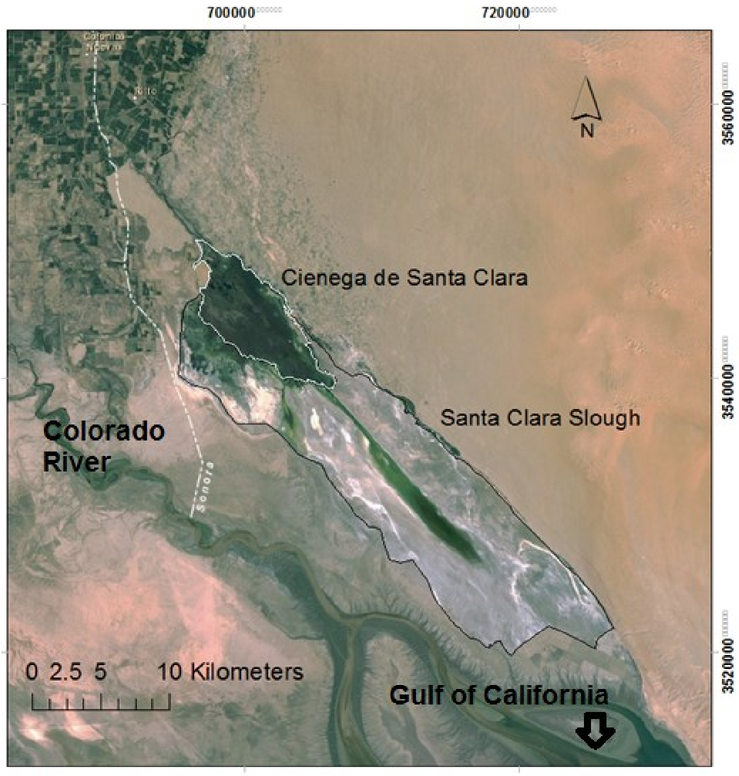


Fig. 1. Satellite image of Cienega and Santa Clara Slough in the Colorado River delta.

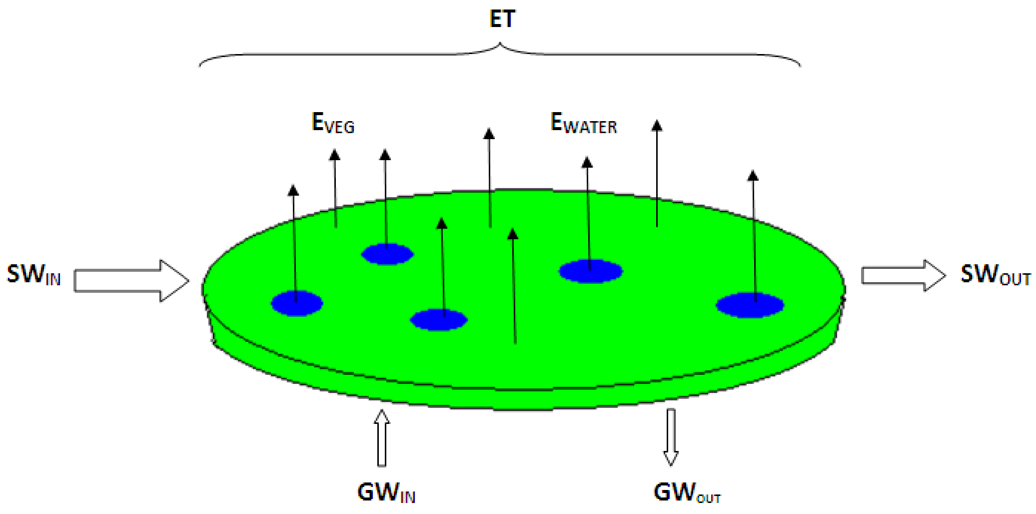
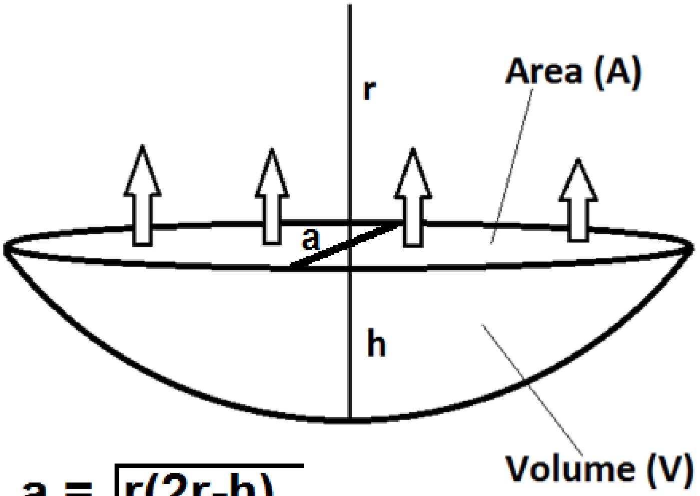


Fig. 2. Schematic of the Cienega vegetated area showing water budget components. SW_{IN}, surface water inflow; SW_{OUT}, surface water outflow; E_{VEG}, transpiration by vegetation; E_{WATER}, evaporation from open water areas; ET, total evapotranspiration; GW_{IN}, groundwater inflows; GW_{OUT}, groundwater outflows.



$$a = \sqrt{r(2r-h)}$$

$$A = \pi a^2$$

$$V = \frac{\pi h^2}{3}(3r-h)$$

Fig. 3. Method for calculating surface area of the Santa Clara Slough supported by outflows from the Cienega, based on volume and area relations of a spherical section. Small letter **a** is the radius of the cap; h is the height of the cap; r (including segment h) is the radius of the sphere. Arrows represent water lost to evaporation; as water evaporates Area **A** decreases in proportion to the decrease in **h**. Formulas from Wolframs Mathworld, http://mathworld.wolfram.com/SphericalCap.html.

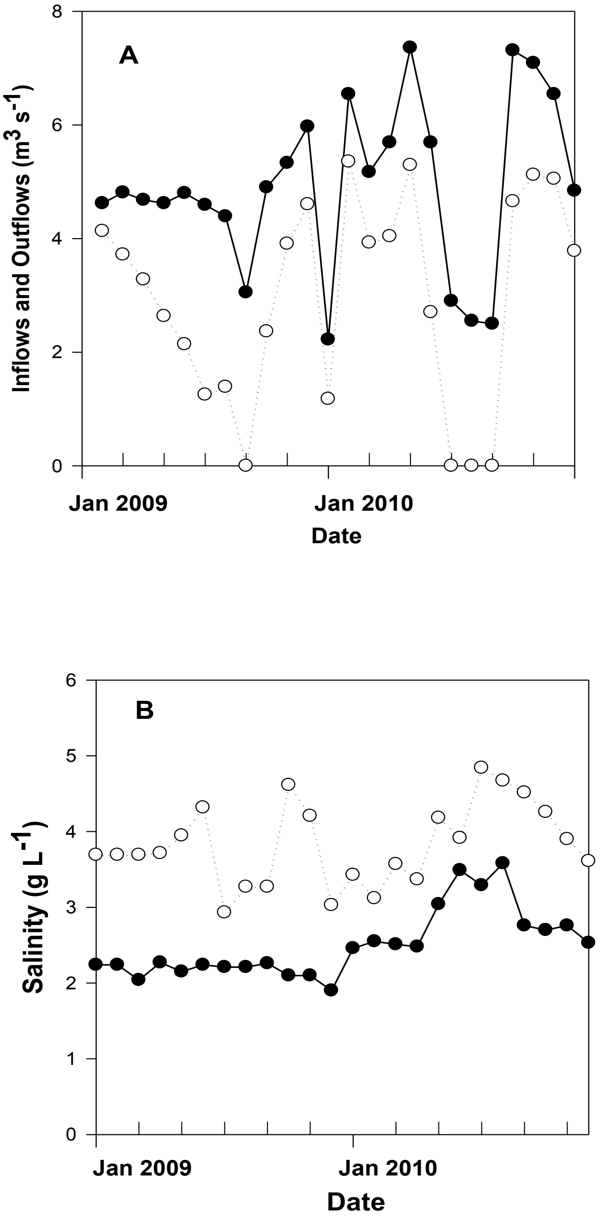


Fig. 4. Baseline data used in constructing models. (A) Measured inflows (closed circles) and calculated outflows based on ET (open circles). (B) Measured inflow salinity (closed circles) and mean wetland salinity measured at 22 stations within the Cienega (open circles).

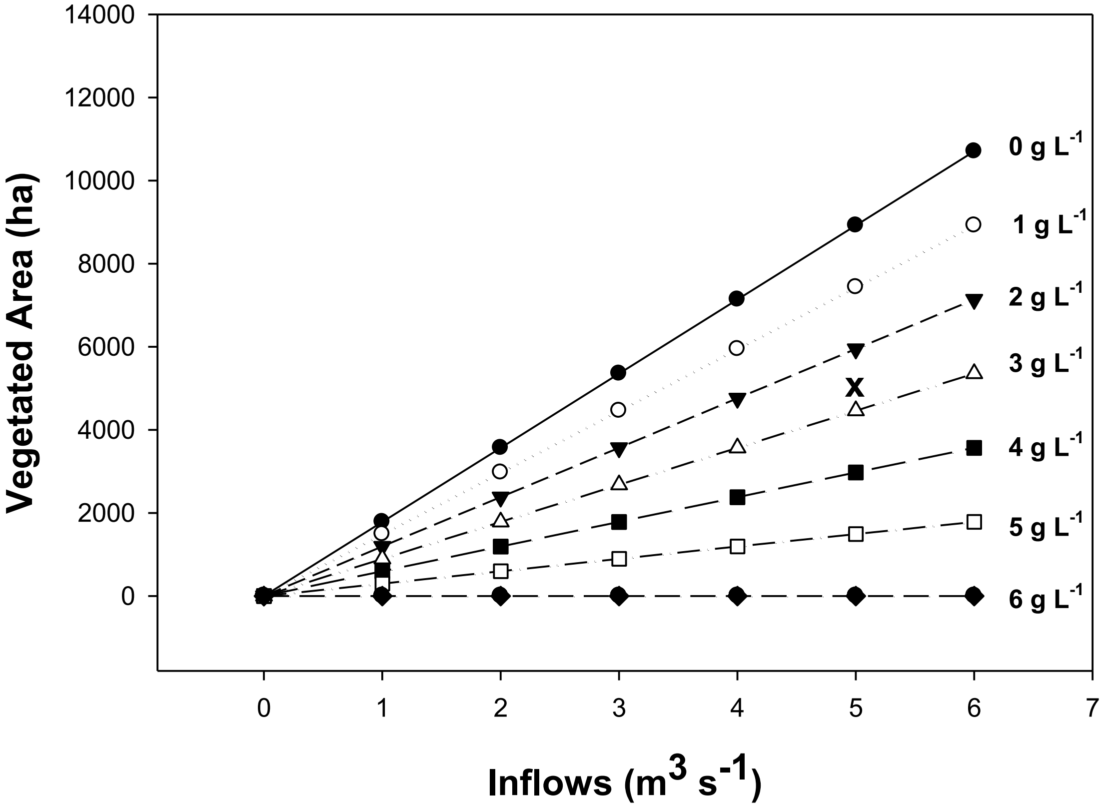


Fig. 5. Nomogram showing the projected area of *T. domingensis* in the Cienega as functions of volume and salinity of inflows. X indicates the present conditions in the Cienega.

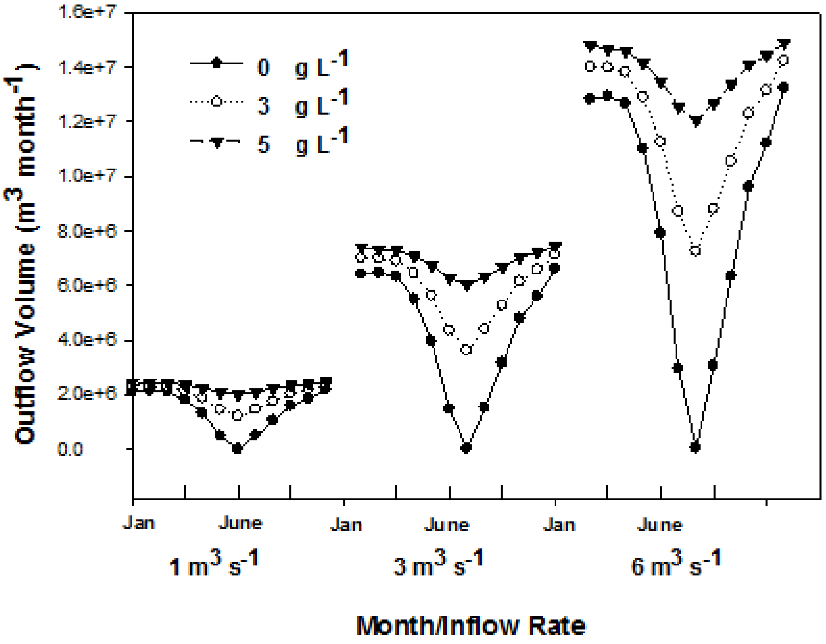


Fig. 6. Monthly projections of outflow volumes as functions of inflow volumes and salinities in the Cienega.

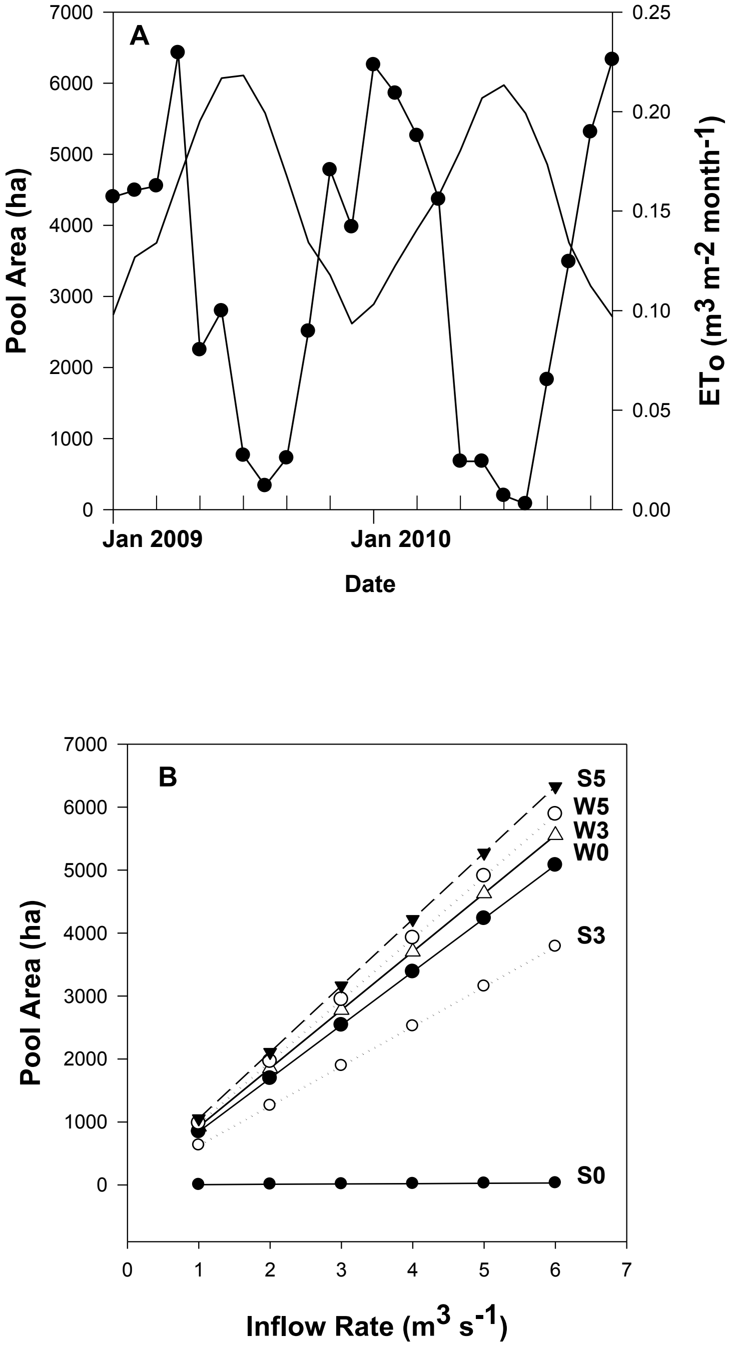


Fig. 7. (A) Projected pool area of the Santa Clara Slough based on outflows for the reference period, 2009-2010 (closed circles), compared to potential evapotranspiration (solid line). (B) Projections of summer (S) and winter (W) outflow pool area as functions of inflow volumes and salinities (0 = 0 g L^{-1}; 3 = 3 g L^{-1}; 5 = 5 g L^{-1}).

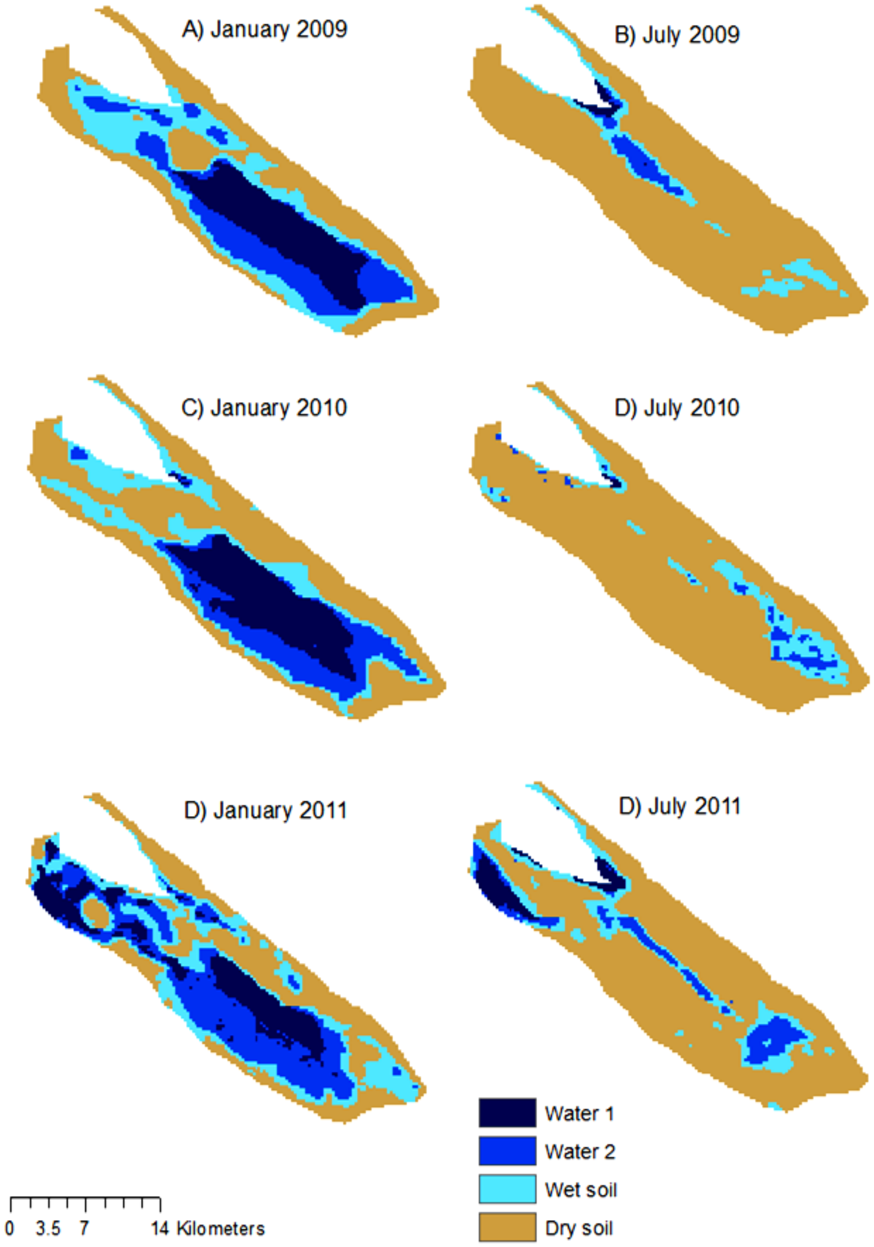


Fig. 8. January and July pool area in the Santa Clara Slough determined from MODIS imagery; images were obtained in neap tide conditions representing periods when overbank tidal flooding had not occurred for a month or longer.

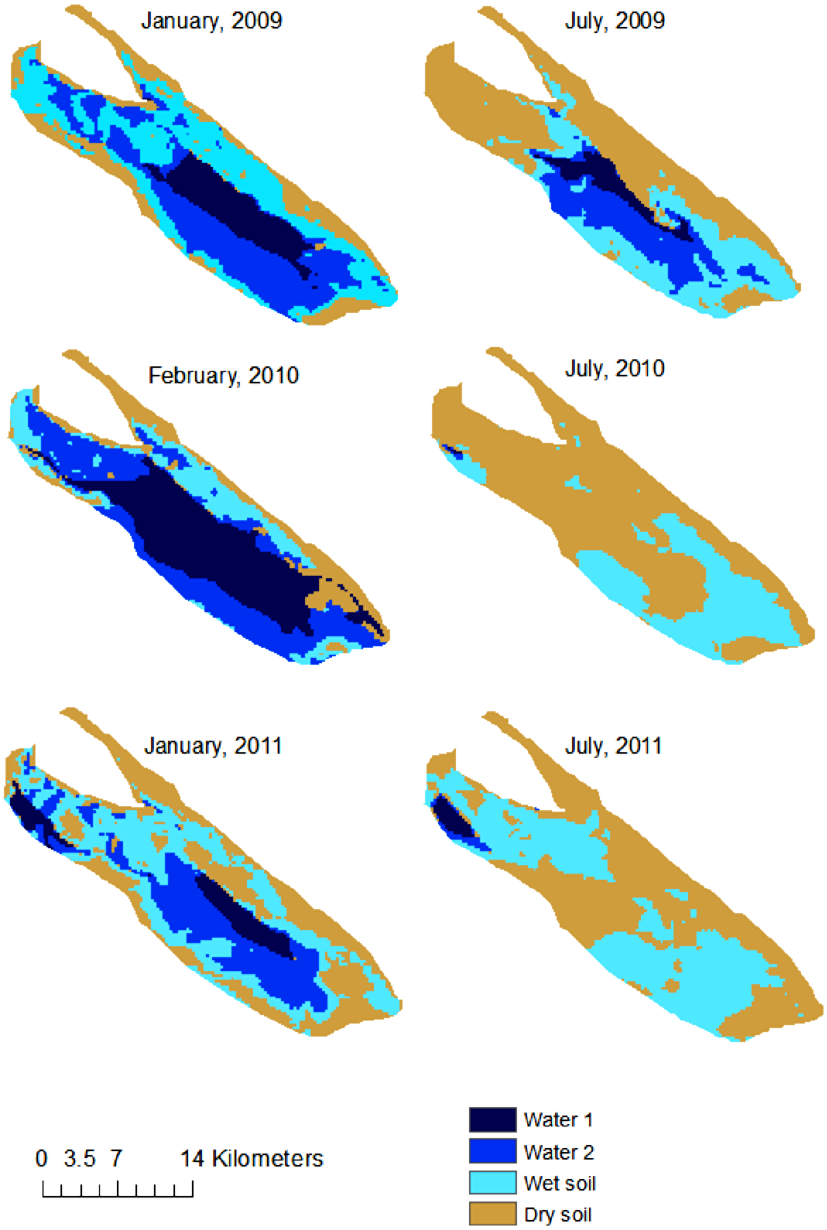


Fig. 9. January/February and July pool area in the Santa Clara Slough determined from MODIS imagery; images were obtained for dates following 5.2 m or higher tides that produced overbank flooding into the slough.

APPENDIX C

Spatiotemporal variation of NDVI and EVI and its suitability to model Yuma Clapper Rail detections in the Cienega de Santa Clara, Sonora, Mexico

Manuscript prepared for submission to Wetlands

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Abstract

The determination of habitat features that have a key role in the species habitat preferences and fitness is crucial to generate habitat models. These models can be used to predict species occupancy, abundance and help to explain habitat use pattern in response to changing variables. We developed empirical relationships between Vegetation Indices (NDVI and EVI) from the Moderate Resolution Imaging Spectroradiometer (MODIS) and the number of Yuma Clapper Rails (*Rallus longisrostris yumanensis*) counted along transects from 2000 to 2011in a brackish water marsh land to determine the feasibility of predicting Yuma Clapper Rails detected from VIs. Our approach was first to determine the strength of the relationship, then to analyze the spatiotemporal variation of VIs and detections. Lastly, we explored the effects of adding other habitat variables and the presence of fire events in the performance of linear models based on NDVI. We found both NDVI and EVI were positively related to the Yuma Clapper Rail detections, the relationship was weak to moderate but significant ($P < 0.001$) which suggests other factors besides the vegetation condition play an important role. We found NDVI was a better predictor than EVI. The analysis of the spatiotemporal variation in NDVI and detections showed that the wetland presents clusters of high NDVI values in the central lagoons in the north part of the wetland, and they are relatively stable over the years. In contrast, the edges of the wetland, and particularly the eastern edge tend to have more dynamic alternating clusters of high and low NDVI in response to an edge with higher fluctuations in water level due topographic effects. The clusters of high and low Yuma Clapper detections did not follow the same temporal trend that the NDVI; however, the effects of changes in the vegetation condition in the wetland were observed in subsequent

years. Fire events were followed by increases in the maximum NDVI value and a high numbers of clapper rails were detected during the fire year and in the following years as a result of vegetation renewal and habitat improvement. A model including all the variability among years was a better predictor of the rails detected per transect than models for fire and non-fire years. We did not find a significant effect from adding habitat features (water % or vegetation %). We recommend to include variables at both microhabitat level and landscape level relevant both before and during the breeding season in order to increase the explanatory power of models.

1. Introduction

At a global scale it is estimated that 44% of bird populations that depend on wetlands are currently of conservation concern (Eddleman et al., 1988; Morrison, et al., 2006; Wetlands International, 2006). Secretive marsh birds that strongly depend on wetlands for breeding (e.g. rails and bitterns) have showed declining population trends in North America. These declines have led to proposals for monitoring programs to detect population trends, integrate biological monitoring into wetlands adaptive management and to investigate effects of climate change (Conroy, et al., 2010; Conway, 2011).

Secretive marsh birds constitute useful indicators of the ecosystem health in wetlands since they are top predators and their presence can been associated with a variety of habitat features (Gottschalk et al., 2005). The determination of habitat features that play a key role in species habitat preferences and fitness is crucial to generating habitat models on which conservation actions can be based. In wetlands habitat models can be used to predict species probability of occupancy, to explain population trends and species habitat use patterns in response to changing variables (Naugle et al., 2001; Valente et al., 2011; Harms and Dinsmore, 2013).

Habitat features that have been used to explain parameters derived from species-habitat associations (probability of occupancy, habitat use, abundance) of marsh birds include wetland size, cover type at a landscape level and water depth, water-vegetation interspersion, percent of cover, emergent plant species present, presence of invasive species and vegetation structure derived from variables at a local scale (Conway et al., 1993; Benoit and Askins, 1999; Rehm and Baldassarre, 2007; Conway, 2011; Valente et al., 2011; Harms and Dinsmore, 2013).

The availability of remote sensing data and aerial imagery have made it possible to derive landscape metrics to analyze the habitat-species associations in a spatial context, thus generating spatial habitat models. These spatially comprehensive models can help to predict occurrence or abundance at unsurveyed locations and explain species distribution patterns within a site. Stralberg et al. (2010) and Pickens and King (2012) exemplify the use of remote sensing data analysis to establish spatial habitat models for secretive marsh bird species.

Vegetation indices (VIs) as the Normalized Difference Vegetation Index (NDVI) and the Enhanced Vegetation Index (EVI) can provide information about the photosynthetic activity and the canopy structure variations respectively (Huete et al., 2002). VIs are valuable metrics in species-environment relationships (Pettorelli et al., 2005; 2011). NDVI is a ratio calculated as:

$$NDVI = (\rho NIR - \rho Red)/(\rho NIR + \rho Red) \qquad (1)$$

where ρNIR and ρRed are atmospherically-corrected reflectance values received by satellite sensors in the near infrared and red bands, respectively. EVI is calculated as:

$$EVI = G((\rho_{NIR}-\rho_{red})/(\rho_{NIR} + C_1 \times \rho_{red} - C_2 \times \rho_{blue} + L)),$$

where ρ are atmospherically corrected surface reflectances, C1 and C2 are coefficient of the aerosol resistance term, L represents the canopy background and G is the gain factor usually set at 2.5 (Huete et al., 1994).

NDVI and other VIs such as EVI can be good indicators of the primary productivity of a system, phenology and quality of the vegetation, allowing relationships to be established between vegetation density and avian parameters such as species richness, distribution patterns, abundance and life-history traits.

The strength of the relationships between VIs and habitat value for birds are site, species and scale dependent. For example Mcfarland et al., (2012) found that NDVI and NDVI-related variables can be suitable to model bird species abundance and richness in a riparian area, and concluded that NDVI was a better predictor at the individual species level than the community level. In contrast Hurlbert and Haskell, (2003) found that NDVI was good predictor of bird species richness across North America.

Regarding marsh lands, Stralberg et al. (2010) modeled habitat potential for three bird species in a tidal marsh and reported that NDVI along with other landscape metrics was a good predictor of avian species abundance using high resolution data, and Klassen et al. (2010) used NDVI maps to explain the role of habitat availability in the marsh harriers migration route, finding that NDVI might play an indirect role.

In the Colorado River Delta wetlands in Mexico, the restoration of wetlands is being based on a science-based adaptive management approach (Glenn et al., 2013a). A comprehensive monitoring effort was implemented from the years 2009 to 2011 in the Cienega de Santa Clara (Fig 1), the largest marshland of the Sonoran Desert and habitat for 75% of the endemic Yuma Clapper Rail population (*Rallus longirostris yumanensis*) subspecies listed as endangered in the U.S. and in a threatened category in Mexico (Hinojosa-Huerta et al., 2001a; Diario official de la Federación, 2010; Hinojosa-Huerta et al., 2013). The Cienega monitoring effort included the monitoring of the hydrology, water quality, vegetation dynamics, weather parameters, effects of natural events, and marsh bird populations (Flessa et al., 2012).

Satellite derived data analysis has been a critical component of the monitoring efforts in the Colorado River Delta wetlands and moreover in the Cienega (Nagler et al., 2009; Glenn et al., 2001; Glenn et al., 2013b; Mexicano et al., 2012; 2013). In this ecosystem NDVI was positively correlated with the amount of water entering the wetland and increased markedly in years following fires events (Glenn et al., 2013b; Mexicano et al., 2012). Fires cleared out senescent thatch and returned nutrients to the water, enhancing the growth of new emergent plants in the following growing season.

The feasibility of using VIs as an input variables to explain marsh bird spatiotemporal abundance patterns in wetlands has not previously been explored at the micro-habitat level, as far as we know. Our hypothesis is that this type of analysis is the first step in the establishment of a predictive habitat model to couple remote sensed data and field data for an indicator species, in this case the Yuma Clapper Rail.

In the present study, we developed empirical relationships between VIs from the Moderate Resolution Imaging Spectrometer (MODIS) sensors on the Terra satellite and the number of Yuma Clapper Rails counted along transects from 2000 to 2011. Our working hypothesis was that VIs could be used to model Yuma Clapper Rail abundance both spatially within the Cienega de Santa Clara, and temporally from year to year. In other studies, areas with low vegetation density, water coverage and less presence of residual vegetation have been detected as the most suitable habitat features for Yuma Clapper Rails (Conway et al., 1993). On the other hand the presence of dense, old vegetation has been related to a higher risk to predation since the birds have to walk over the top of dense vegetation mats (Conway et al., 2010). Also old vegetation could lead to less food availability and a decrease in foraging activity (Conway, et al., 2010).Therefore, marsh bird populations can increase after major disturbances such as fires in the Cienega, which removes old growth and allows new growth to develop (Hinojosa-Huerta et al., 2013).

In the light of this we expected that NDVI and EVI could be used as proxies for the vegetation condition of the dominant emergent species (*Typha domingensis*) to explain the variance in the Yuma Clapper Rail detected spatially and temporally.

The objectives of this analysis were: 1) To determine the strength of the relationship between vegetation indices and Yuma Clapper Rail detections, 2) To conduct an exploratory spatial analysis to detect patterns of spatial association (clusters) in the number of Yuma Clapper Rails and in the vegetation indices (NDVI and EVI) across years, 3) To determine if NDVI and additional habitat variables can be used to predict distribution patterns of clapper rails in the Cienega and 4) To explore the effect of fire in the relationship between NDVI and VIs.

2. Methods

2.1. Study area

The Cienega de Santa Clara is 5500 ha wetland that depends on brackish water delivered from the Wellton-Mohawk Irrigation District in the United States to Mexico through the Main Outlet Drain Extension (M.O.D.E.) canal and to a lesser extent from agricultural runoff from the San Luis Río Colorado (Fig 1) (Glenn et al., 1992). The Cienega is a critical component of the Colorado River Delta in Mexico, since it provides habitat for 71% of the bird species listed for the Colorado River Delta region (Hinojosa-

Huerta et al., 2013). The Cienega avian community is composed of migratory and resident birds including nine species of marsh birds, 25 waterfowl species, 29 species of shorebirds, and 81 species of landbirds as well as seabirds (Hinojosa-Huerta et al., 2013).

The vegetated fraction of the Cienega is estimated in 89% (4,895 ha) and the remaining 11% represented by open water areas consisting of lagoons and channels (Glenn et al., 1992; Mexicano et al., 2012). *Typha domingenesis* (southern cattail) is the dominant vegetation type, with patches of *Phragmites australis* (common reed) and *Schoenoplectus* spp. (bulrush) interspersed within the cattail stands.

Water depth is in the range of 0.20 to 1.50 m (Flessa et al., 2013). Salinity varies from about 3 g L^{-1} total dissolved solids at the northern entry point of water to 6 g L^{-1} TDS at the southern end where water discharges into a shallow, unvegetated tidal basin (Glenn et al., 2013b; García-Hernandez et al., 2013). The inflow to the Cienega has averaged 4.28 m^3/s but with flow reductions in 2001, 2004, 2010. Fires have been common events in the Cienega, in the time frame considered for this study fires burned over 70% of the vegetation in 2006 and 2011 (Glenn et al., 2013b).

2.2. Bird surveys

Yuma Clapper Rail data were collected during the monitoring program for marsh birds in the Colorado River delta (Hinojosa-Huerta et al., 2001a). The monitoring protocol follows the procedures established by the North American Marsh Birds Monitoring Protocols (Conway, 2002). The protocol uses a call-response survey using pre-recorded vocalizations for six marsh bird species. Vocalization are played after five minutes of silence, using 30 seconds to play the vocalization and 30 seconds of silence.

Marsh bird individuals heard or observed are registered to obtain number of detections per sampling point.

From 2000-2010 marsh bird detection numbers were surveyed along15 transects randomly located in the upper half of the wetland and 11 additional transects located in the lower half during the 2010-2011 monitoring period. Transects encompassed five points separated by 200 meters and with a radius of 200 meters. Marsh bird surveys were conducted twice during the breeding season from March to May. Yuma Clapper Rails detected were averaged first by transect, dividing the total by the five points and then the average from the two survey dates was obtained. Therefore, our sampling unit was the transect (Fig. 2) that covers an area of 80 ha.

2.3. Vegetation Indices sampling and habitat features.

NDVI and EVI datasets were obtained from the Moderate Resolution Imaging Spectroradiometer (MODIS) land subsets products (MODIS/Aqua Vegetation Indices (NDVI/EVI) 16-Day L3 Global 1km SIN Grid (http://daac.ornl.gov/MODIS/MODIS-menu/products.html). We used earthexplorer.usgs.gov to request datasets. Vegetation Indices (NDVI and EVI) from this product use blue, red and near-infrared reflectance, at a spatial resolution of 250 m and temporal resolution of 16 days.

Individual pixels were sampled adjacent to the first, third and five point within the transect with the percentage of vegetation ≥ 90%, from MODIS images using the two dates per month and then obtaining an average for each month (Fig. 2). The pixel distance to the marsh bird survey point was variable (average 200 m) since we tried to avoid the error caused from the water signature in the NDVI/EVI pixel values. We excluded from the database the cattail dormancy period (October-March), therefore, the final average value per year for the vegetation indices was calculated using seven months of data per year.

A buffer of 125 meters was created around the transect lines to obtain a shapefile with polygons in order to associate NDVI and EVI values with rail detections. We created two spatial datasets, one including the 12 years of VI and rails detections data including 15 transects distributed in the upper half of the wetland and a second dataset including years 2010 and 2011, for which 26 transects were available. GIS procedures were carried out with ArcGIS 10.0 software (ESRI, Inc., Redland, CA) A Quickbird image from August 2008 was used to create a raster of the water features of the wetland at 1m pixel resolution, and a layer of the boundary of the vegetation was created to delimit the extent of vegetation cover. This layer was overlaid with the polygons layer and raster of water features to estimate the percentage of water and vegetation inside of each polygon using ArcMap 10.0. Projected Coordinate system used was WGS_1984_UTM_Zone_11N, Projection Transverse Mercator, Geographic Coordinate System GCS_WGS_1984 and Datum D_WGS1984.

2.4. Analysis

The two datasets (15 transects surveyed during 12 years, and 26 transect surveyed on 2010 and 2011) were analyzed separately. Preliminary analyses showed that data were normally distributed or close to normal, so parametric statistical tests were used. To determine the differences among years in the 15 transects dataset we used Analyses of Variance (ANOVA), and paired t-test to determine differences between the years 2010 and 2011in the 26 transects dataset. To analyze the relationship between VIs and Yuma Clapper Rail detections we used linear regression including the NDVI and detections values per transect from each of 12 years (N = 180). Statistical analyses were carried out with Statistics Package for the Social Sciences (SPSS v.13.0, SAS, Inc., Cary, NC, USA) software.

We used Global Moran´s I to determine the presence of spatial autocorrelation (clusters) in VIs and Yuma Clapper Rail detections. Local Indicators of Spatial Association (LISA) were used to determine the type of spatial correlation and location of different type of clusters (High-High; high values surrounded by high values, Low-Low; low values surrounded by low values; High-Low, high values surrounded by low values and Low-High, low values surrounded by high values) (Anselin, 1995; Anselin, 2005).

A weights matrix was created using the nearest neighbor method selecting four neighbors. A Univariate LISA analysis was conducted for each year and each variable to obtain the cluster map. A randomization process was used to obtain a pseudo p-value and Z-score after 999 permutations and compared.

Multiple regression analysis (Ordinary Least Squares, OLS) was used to analyze the dependence of the detections of clapper rails from VIs (NDVI, EVI), the percentage of water and percentage of vegetation in the 15 transects dataset. For this analysis we use an average per transect: a) from the twelve years period (2000-2011), b) from non-fire years and c) from fire years (2006 and 2011) to test the effect of fire in the relationship. The linear regression model was tested on the additional 11 transects, we use the 2011 NDVI values and compare this with the same year clapper rails data. We used OLS regression method with spatial dependence diagnostic tests to identify effects of spatial autocorrelation. The K-Nearest Neighbors matrix was set up to four neighbors. The spatial dependence diagnostic tests indicated there were not problems of multicollinearity (multicollinearity condition number <30), non-normality (Jarque-Bera test of normality of errors), heteroskedasticity (Breusch-Pagan, Koenke-Bassett and White test), spatial autocorrelation of the residuals (Moran's I) and that the classic model (OLS) was appropriate (Lagrange Multiplier test for spatial dependence was not significant). Spatial analysis were conducted on Open Geoda version 1.4.5. (Anselin, 2013).

3. Results

3.1. VIs and Yuma Clapper Rail detections relationship

VI values showed significant differences in the mean values among years (*p*-value <0.01, from one-way ANOVA in the 12 years, 15 transects dataset and in the two years 26 transects dataset (*p*-value < 0.01 from a *t*-paired test). Means per year increased about 20% during fires year in 2006 and 2011 (mean = 0.542, sd = 0.131 in 2006 and mean = 0.550, sd = 0.178 in 2011, N = 15 transects) (Fig. 3).

We found a positive relationship between VIs and the number of clapper rails ($R^2 = 0.127$, $F_{1,177} = 25.69$ *p*-value < 0.001 for NDVI and $R^2 = 0.082$, $F_{1,177} = 15.73$ *p*-value < 0.001 for EVI from a linear regression) (Fig. 4). Based on the R^2 the relationship was stronger for NDVI, therefore for the rest of the analysis we used the NDVI.

3.2. Temporal and spatial variation in NDVI and Yuma Clapper Rail detections from a clusters analysis.

NDVI showed significant positive spatial autocorrelation in all the years analyzed (Moran's I ranged from 0.2432 to 0.4797, pseudo *p*-values < 0.05 and Z-scores >2.0) and significant temporal changes in the clusters location (Fig. 5). The clusters of high-high NDVI areas (transects with high values of NDVI associated surrounded by transects of high NDVI values) were located close to the largest open water areas in the central north portion of the wetland, and this spatial pattern was relatively consistent across years (Fig. 5). Transects with high NDVI values associated were also found in the north east edge of the Cienega, on transects 11 and 12, in seven out of the 12 years. In 2008, a pattern of positive autocorrelation (high-high NDVI values) in the east edge and low-low values in the western edge, was detected. The later corresponded with an area that was affected by siltation from incoming canal water, which promoted a change in the cover from emergent vegetation to shrubs such as *Tamarix ramossisima* on the newly emerged surfaces. The cluster map of 2011 shows clusters of high NDVI in the east side as well in 2008.

Yuma Clapper Rail detections showed low-low clusters and low-high outliers more frequently than NDVI (Fig. 6). The location of the low-low clusters varied from year to year. Low-high pattern is interpreted as spatial outliers and tended to be present in transects located on the eastern edge of the wetland. High-high detected rails clusters were found on the two fire years and during 2007. These clusters were located in the central open water areas, as in the case of the NDVI for the same years. During the siltation year (2008) the number of rails was randomly distributed, however, the next year a transect in the area affected by siltation showed low-low values (Fig. 6).

3.3. Model predictions using NDVI

The best model in terms of prediction accuracy included all the years ($R^2 = 0.2566$, $F_{1,12} = 4.48$, $p = 0.053$) (Table 1). During fire years clapper rail detections showed a stronger relationship to NDVI, the relationship during non-fire years was not significant. Also the terms from the habitat variables (percentage of water and percentage of vegetation) were not significant.

The performance of the linear model using survey data from 2011 in a set of 11 transects confirmed the prediction is improved when all years are included, therefore a model based on the NDVI signature from fire years is not recommended to predict clapper rails in general (Fig. 7).

4. Discussion

4.1. Spatiotemporal variation in NDVI Yuma Clapper Rails

Th**e** positive relationship between NDVI and detections across transects and years was complemented with a more detailed spatiotemporal examination of the role of vegetation via NDVI in the number of clapper rail detected (LISA analysis).

The results of the LISA analysis indicate the spatial distribution of vegetation indices values and clapper rail detections differs from random. Clusters of high NDVI/EVI in the areas close to the lagoons in the north part of the wetland were relatively stable, our analysis indicates clapper rails usage of central lagoon areas was intensified during and after fires years. Hinojosa-Huerta et al. (2013) reported the same pattern of spatial variation for the density of clapper rails in the Cienega. In contrast, in the edges of the wetland the distribution of clusters of high and low values for the VIs was more dynamic, and in that area our analysis detected a pattern of spatial outliers, a transect with low detections with neighbors transects of high numbers of clapper rails. This can be explained as a result of the bathymetry of the wetland and the differences between zones more or less susceptible to fluctuations in water depth (Glenn et al., 2013b). The open water lagoons have an approximate depth of 1 m versus 0.20-0.30 cm in the edges, as a result the lagoons are less susceptible to changes in the water inflow volume, a variation of 0.15 m was registered in the open lagoons in contrast variation in the edges can be as large as 0.20-0.25 m (Flessa et al. 2012).

The differences in bathymetry are important drivers in the flow direction and salinity variations in the wetland. The north half of the Cienega is an upper basin deeper than the lower half, and this is reflected in the salinity distribution since the north half has lower salinity than the lower half (García-Hernández, 2013). The flow direction is affected by a fault that promotes the flow from north to south with lower salinity along this middle line (Glenn et al., 2013). Thus, the salinity at the center of the wetland is highly correlated with the salinity of the inflow and the flow volume from the M.O.D.E. canal, which remains relatively stable (about 2.6 g L^{-1} TDS at an inflow rate of 4 m^3s^{-1}) and maintains the cattail growth currently considered under salt stress conditions (Baeza et al., 2013), along the edges the salinity increases and variations are less correlated with salinity from the water source (Hernández-García, 2013). This creates a zone where vegetation is more susceptible to present salinity stress and produce variation in the availability of vegetation patches suitable for marsh bird use. This pattern of high variation was detected as the presence of transects with low detections of rails next to a transect with high detections.

Based on the framework proposed by Wiens et al (1985) boundaries represent discontinuities in landscape features where the rates or magnitudes of ecological transfers change abruptly in relation to those within patches. In the case of the Cienega the transitional area of standing water between the emergent vegetation and the dry soil can be seen as the boundary. The vegetation density and species composition is expected to be less stable at the boundaries where plant species tolerances to salinity, water depth and soil oxygen interact to promote changes in the biomass and composition of vegetation (Baeza et al., 2013). The extent of the boundary will reflect these differences. Due to topographic differences on the west side of the Cienega the boundary or transitional zone is more abrupt than on the east side. This difference in boundary extent might explain the presence of high and low clusters of NDVI/EVI values on the east side, and therefore the observed variations in Yuma Clapper Rail detections.

4.2. NDVI as a predictor of clapper rails

Our study indicates NDVI is a better predictor of clapper rails distribution than EVI. The global regression (Fig. 4a) showed that a high variation in the number of Yuma Clapper Rails detected in the Cienega can be found in a range of 0.35-0.65 NDVI values. That range of NDVI values was classified as low density vegetation to high density by Mexicano et al., (2013) in the Cienega. This suggests there is an optimum of NDVI values that interact with other factors besides vegetation to determine the bird’s abundance spatial patterns. The former is reflected in the weak to moderate correlation coefficient ($R^2 = 0.127$).

Yuma Clapper Rail detections did not follow the same temporal trends as the vegetation. This is expected since there is a time lag between the disturbance events (fire, flow variation and desiccation) and the expected effect in the clapper rail population. In the case of fire Conway et al., (2010) found the increase in the number of Yuma Clapper Rails in response to fire was observed after six month to 2.5 years. In addition they do not necessarily use areas of suitable vegetation in proportion to their availability (Conway et al., 1993).

Fire has been determined to be one of the main drivers in vegetation condition in the study system along with topography, siltation, inflow volumes and salinity (Glenn et al., 2013b; García-Hernández et al., 2013). Our data suggests the clapper rail population response to fire can be detected in the same year that the fire occurred. However, it is necessary to analyze the response in the population in the following years from 2011 to determine the temporal extent of the effect.

According to Conway et al., (2010) the increase in Yuma Clapper Rail numbers after fires might be the result of the removal of decadent vegetation detected as low NDVI patches that may inhibit rail movement, reduce forage efficiency, and increase the rail's risk of predation since they might need to walk on top of mats of decadent vegetation. Lastly, fire could have an effect on food availability but there is a lack of information about the exact pathway of ecological mechanisms to validate this hypothesis.

NDVI regression models fit improved at the transect level, confirming the rails response to NDVI is affected by microsite differences not detected by this study. These differences increase the inaccuracy in the prediction of rails in some areas of the wetland.

The relevance of some microhabitat variables changes with the species reproductive stage. Percent of water and percent of vegetation are important variables before the breeding season (Conway et al., 1993). This might explain why our additional habitat variables did not improve the models. Building a model that includes micro and landscape habitat variables relevant before and during breeding season, might improve the prediction power along with NDVI. Landscape features such as distance to upland and water interspersion (interface between vegetation and water in m/ha) have proven to be relevant (Conway et al., 1993; Rehm and Baldasarre, 2007). Although microhabitat variables as a percent of residual vegetation, stem density, presence of plant species not preferred by clapper rails, presence of high ground within the marsh and NDVI at a higher resolution should be considered (Anderson and Ohmart, 1986; Conway et al., 1993).

4.3. Conclusions and management implications

-The relationship between VIs and Yuma Clapper Rail detections was highly significant with a weak to moderate explanatory power. This suggests other factors not considered in this study play a significant role in the clapper rail distribution pattern.

-NDVI spatial analysis reflects the differences between open water features that are relatively stable in terms of vegetation condition and the highly dynamic edges where vegetation patches change from high to low NDVI values from year to year.

-Clapper rail detections clusters did not follow the same exact trend than the NDVI over the years but the spatiotemporal distribution of clusters suggest they are responding to vegetation changes and disturbance events in the following years.

- Fire events were followed by increases in the maximum NDVI value and in the number of clapper rails detected during the fire and in the following years as a result of vegetation renewal and habitat improvement.

-A model including all the variability among years was a better predictor of the rails detected per transect than models for fire and non-fire years.

-We recommend to include variables at both microhabitat level and landscape level relevant at both before and during the breeding season in order to increase the explanatory power of models.

- Our analysis points out the need to maintain the current water inflow (4 m^3/s) in order to minimize effects on the water depth and the presence of standing water in sensitive areas as the eastern edge of the wetland, currently used by Yuma Clapper Rails.

-Prescribed fires have been proposed as a surrogate of flooding events that among other beneficial effects remove decadent vegetation and create marshes improving and creating marsh bird habitat. We think the development of a robust habitat model based on NDVI and other factors could be a very useful management tool to select the areas that will benefit most from active management in wetlands.

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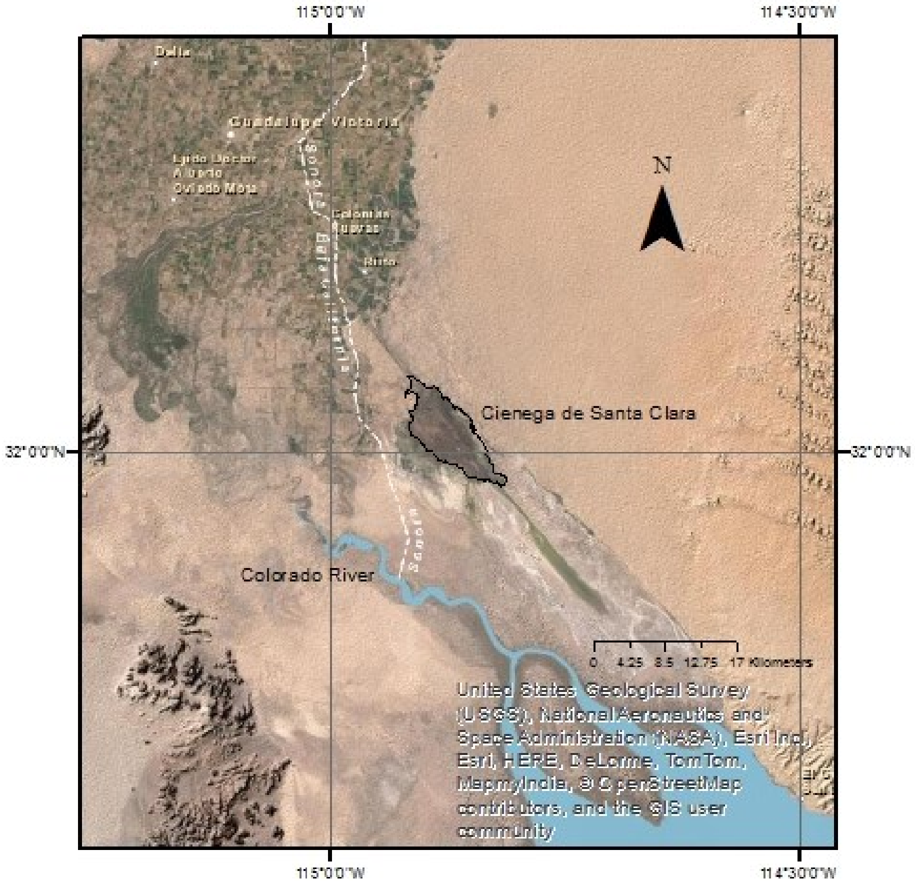


Fig. 1. Cienega de Santa Clara in the Colorado River Delta, Mexico.

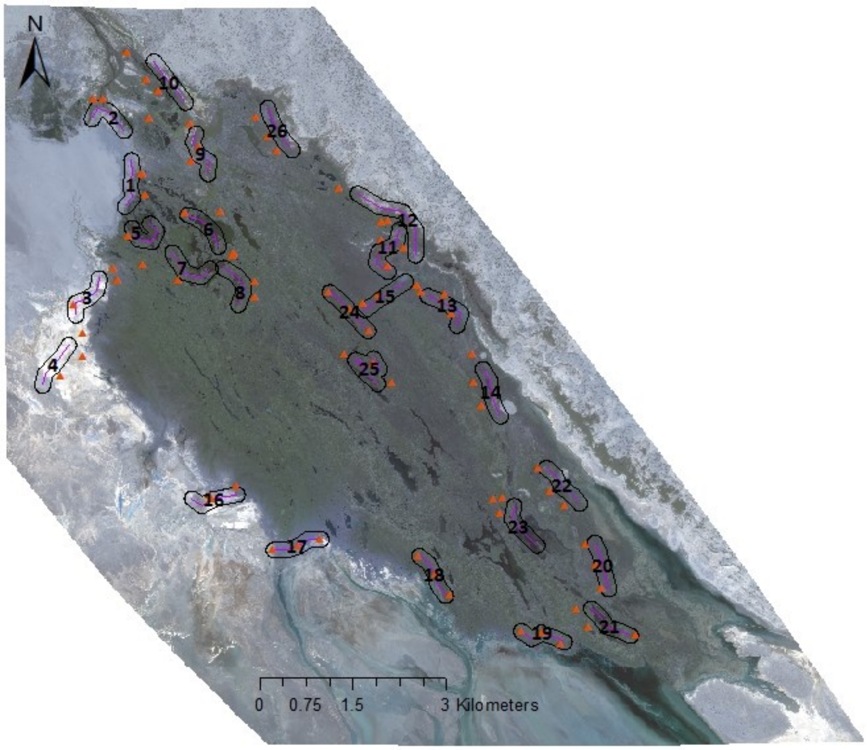


Fig. 2. Polygons enclose transects used for the marsh bird monitoring protocol. Triangles represent the pixel sampling points used to retrieve NDVI and EVI values from the Moderate Resolution Imaging Spectroradiometer (MODIS).

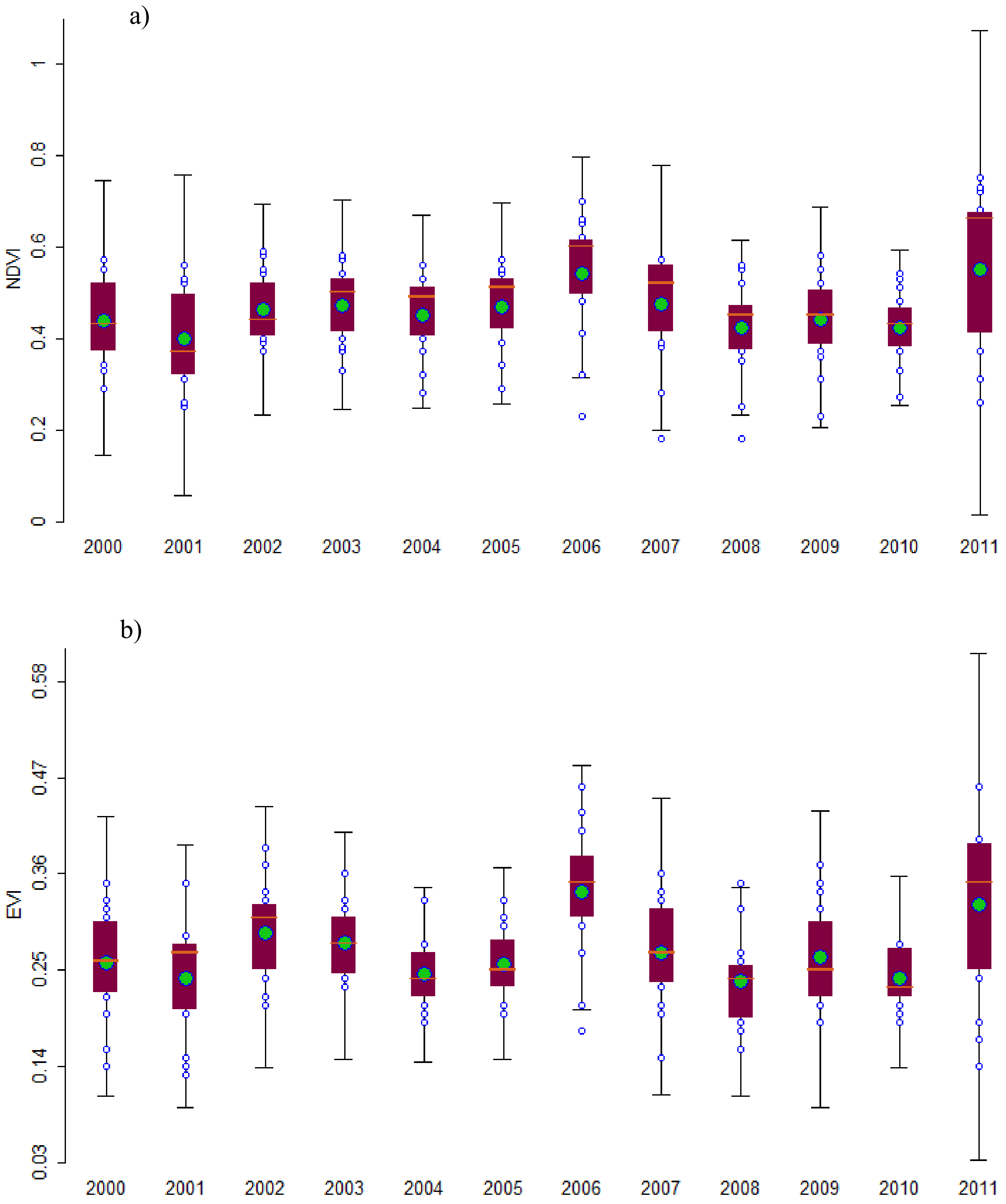


Fig. 3. NDVI (a) and EVI (b) box plot (25th and 75th percentile of distribution), mean per transect (blue circles) median (red bar), mean per year (green dots), 1.5 hinge (black line).

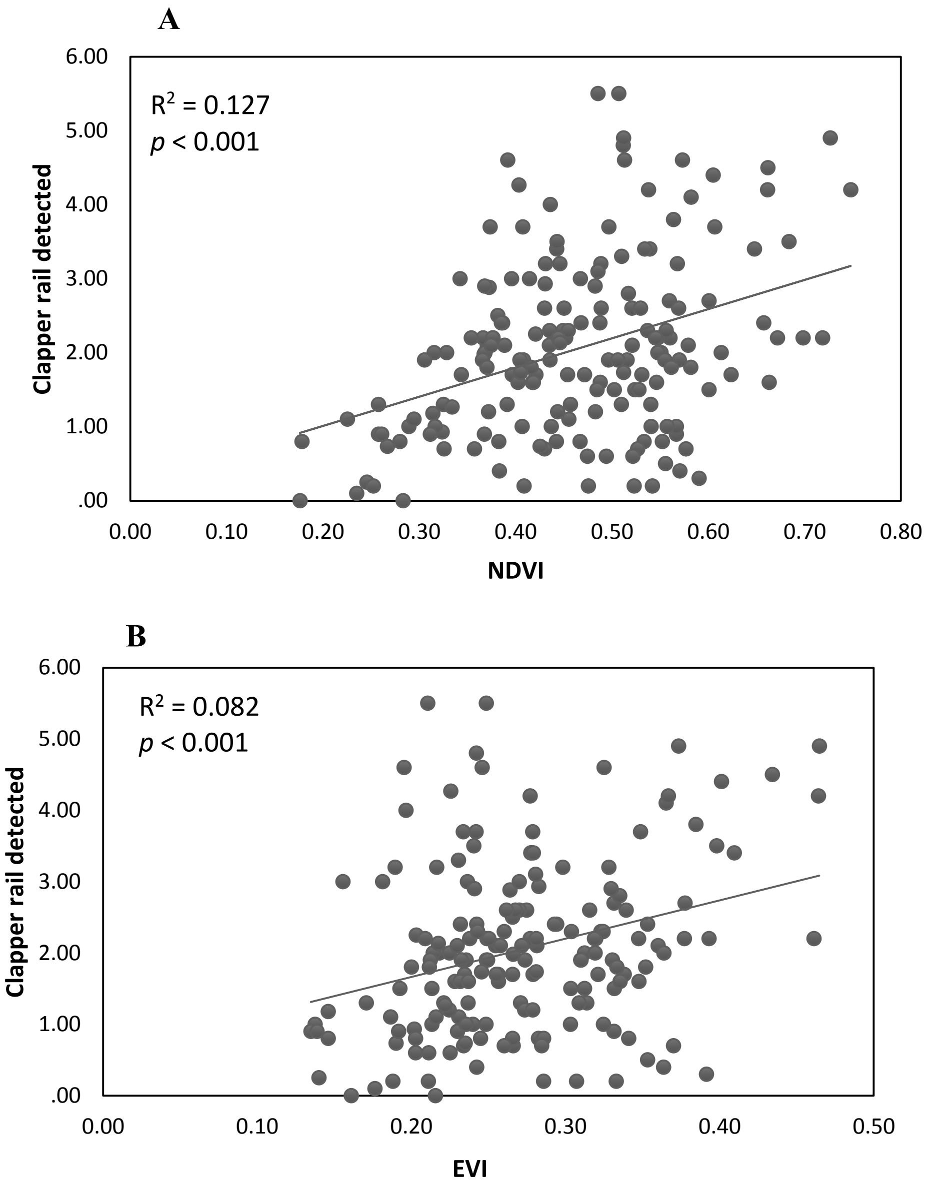


Fig. 4. Relationship between VIs and Yuma Clapper Rail detections, in the 15 transects survey from 2000 to 2011.

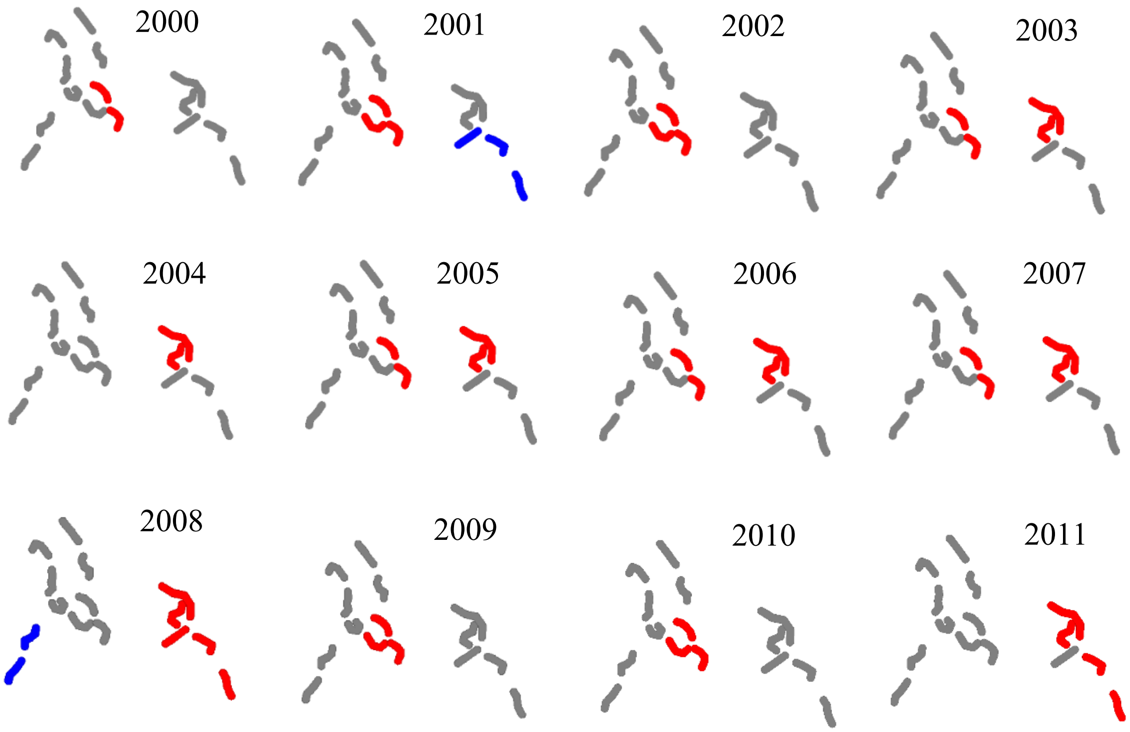


Fig. 5. LISA cluster maps of NDVI, high-high autocorrelation in red and low-low in blue (*p*-values <0.05 from 999 permutations), autocorrelation was not significant for transects in grey. Years 2000-2005, 2007 represent normal conditions, fire events occurred on 2006 and 2011, siltation conditions on the west of the wetland were observed in 2008 and a reduction in the water inflow was registered in 2010.

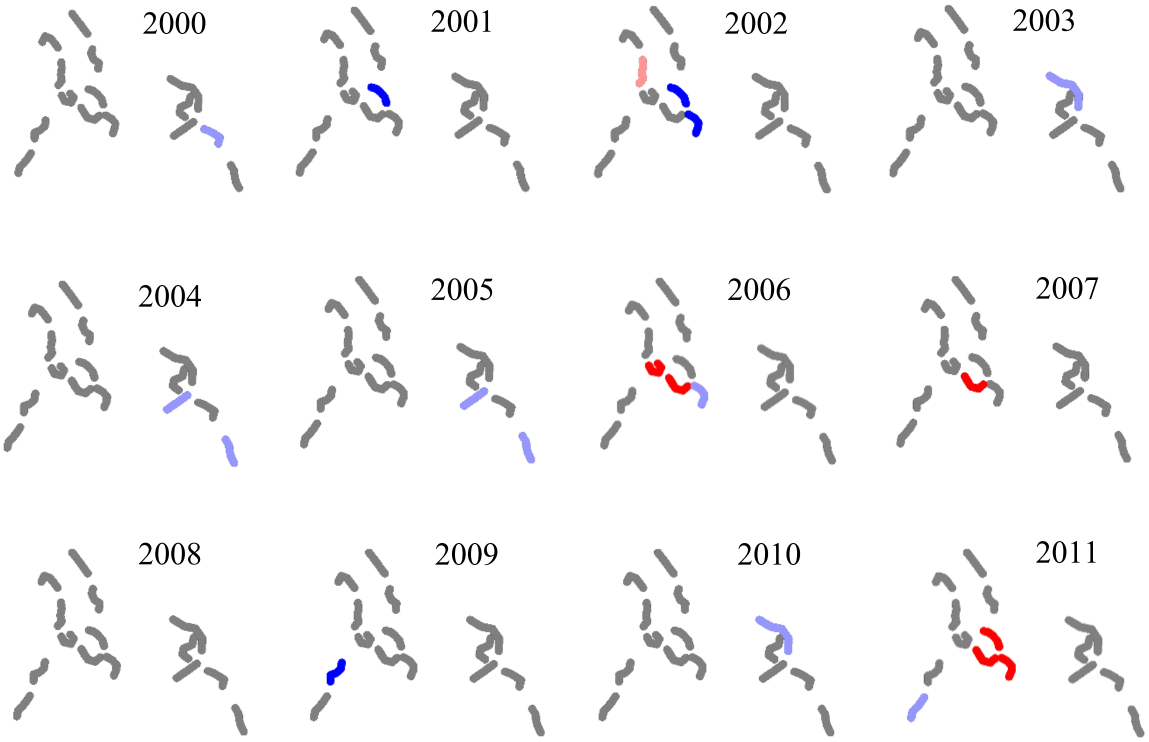
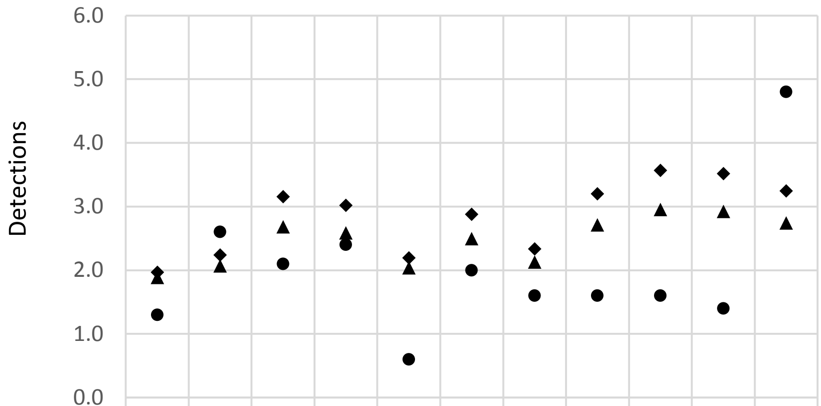


Fig. 6. LISA cluster maps of the number of Yuma Clapper Rails detected, high-high autocorrelation in red, low-low in blue, high-low in light red and low-high in light blue (*p*-values <0.05 from 999 permutations), autocorrelation was not significant for transects in grey. Years 2000-2005, 2007 represent normal conditions, fire events occurred on 2006 and 2011, siltation conditions on the west of the wetland were observed in 2008 and a reduction in the water inflow was registered in 2010.

Table 1. Summary of regression models to predict the Yuma Clapper Rail detections from NDVI including all years, non-fire years, fire years and habitat variables.

Model	R^2	Probability (*F*-statistics)	Log likelihood	AIC (Akaike information criterion)
NDVI (All years)	0.2566	0.053	-8.82	21.64
NDVI (Non-fire years)	0.0800	NS	-9.55	23.10
NDVI (Fire years)	0.3390	0.022	-19.97	43.94
NDVI (All years), Water %	0.2570	NS	-8.82	23.64
NDVI (All years), Vegetation %	0.2997	NS	-8.37	24.87



Transect	16	17	18	19	20	21	22	23	24	25	26
◆ Fire model	2.0	2.2	3.2	3.0	2.2	2.9	2.3	3.2	3.6	3.5	3.2
● Detections 2011	1.3	2.6	2.1	2.4	0.6	2.0	1.6	1.6	1.6	1.4	4.8
▲ All years model	1.9	2.1	2.7	2.6	2.0	2.5	2.1	2.7	3.0	2.9	2.7

Fig. 7. Number of Yuma Clapper Rail predicted by the fire-years regression model and all-years regression model. NDVI and detections (circles) correspond to year 2011, transects 11 to 26. Equation from linear regression as clapper rail detected (fire years model) = 4.5661NDVI (TRANSECT) + 0.0949 and clapper rail detected (all years model) = 3.0539NDVI (TRANSECT) + 0.6335.

APPENDIX D

An Update of Shorebird Aerial Surveys in the Colorado River Delta and Upper Gulf of California.

By

Martha M. Gomez Sapiens

Supplementary Information prepared as a Note for Western Birds

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1. Introduction

The first aerial surveys in the Colorado River Delta were conducted by Morrison (1993) as a part of the project "Aerial surveys of Neartic shorebirds wintering in México". During this effort 160,000 shorebirds were registered using the wetlands near the mouth of the Colorado River. This allowed us to include the Colorado River delta wetlands as part of the Shorebird Hemispheric Reserve Network, in the category of International Importance Site.

Later in 1992, 1993 and 1994, Harrington overflew part of the delta, however his surveys were focused on *Branta canadensis* census in Sonora, Sinaloa and Nayarit, therefore shorebird numbers detected by this survey were very low (< 100 individuals).

Mellink et al. (1997) conducted an aerial survey during the winter of 1994 to estimate non-breeding waterbird species abundance in the Colorado River Delta. They reported about 65,000 shorebirds, 75% of which corresponded to *Calidris mauri* (Western sandpiper).

I determined shorebirds abundance at a regional scale using the data from nine aerial surveys from winter and spring in 2006, 2007, 2009, 2010 and 2011 in order to complement information from ground surveys and to evaluate the importance of the Cienega and Santa Clara Slough habitats in the Colorado River Delta Wetlands. The objectives were 1) to census the shorebirds using the shoreline, estuaries, and the Cienega de Santa Clara y Santa Clara Slough wetlands, 2) To detect high aggregation areas.

2. Methods

Shorebirds were counted by two observers and grouped into three category sizes (small, medium and large) since it was very difficult to identify at the species level in most cases from the plane. The area was divided into six survey sites (Bahia San Jorge, Bahia Adair, Sonoran Coast, Coast of Baja, Islands and South Cienega). The tracks generated during the flights were used to determine the survey time in minutes by splitting the track in the six survey sites and eliminating areas of the track where we were not counting shorebirds. Tracks were recorded using MapSource Garmin v. 6.13.7.

A linear regression was obtained using time as a dependent variable and the transect length as the independent variable on SPSS version 13.0. The resulted equation (time = transect length * 0.624 + 0.286) was used to estimate the time in the cases where the track was not available (three cases). Abundance data were divided by the number of survey minutes. A one-way ANOVA was performed in order to determine differences among category sizes on each site on SPSS v. 13.0.

The abundance data collected during aerial surveys was integrated in a density raster map using ArcGIS Desktop 9.3. We used a density point function with a search radius of 1000 m and a cell size of 500 m to create the density layer. Shorebird abundance was divided in seven classes in order to determine high aggregation areas (> 25,000 shorebirds/km^2).

3. Results and discussion

The highest abundance was detected during the first two years of surveys in December 2006 and April 2007 (Fig.1). The first year we detected the highest numbers from aerial surveys (238,664 shorebirds and over 2500 per min), 80% of which corresponded to big flocks of Western Sandpipers and the rest composed of medium size shorebirds (Dowitchers) and larger sized shorebirds such as American Avocets and Black-necked Stilts (Fig. 1 and 2). The following year, during April 2007, 129,575 shorebirds were observed using the Cienega habitats and 97% of them corresponded to small shorebirds (Western Sandpipers).

In December 2009, lower numbers of shorebirds were found in the entire region (less than 700 birds/min) including the wetlands along the coast. The same year in the Cienega and Santa Clara Slough 20,062 were recorded, from which 50% were small shorebirds. In 2010, shorebird numbers increased again to 67,673 in February from which 62% were small size shorebirds; however, the highest amount of birds per minute was detected in the Islands (2056 shorebirds/min). During 2010 spring migration (April survey) lower numbers were recorded for the entire region, 26,443 in total, less than 500 shorebirds/min and higher percentage of Dowitchers (Fig 1 and 2).

The April survey was conducted after the El Mayor-Cucapah Earthquake. The earthquake had some effects in the delta inundation patterns, diverting the overflows to the western side of the basin, and a slight change in the spatial distribution of shorebirds could be detected, as evidenced by a small flock was using the newly inundated area. At the end of the year abundances were low again in the region and in the Cienega we detected 10,742 shorebirds, less than 200 birds/min and 63% of them were small size (Fig 1 and 2).

In 2011, during January and February, shorebirds counted in the Cienega and Santa Clara Slough were 5,668 and 11,477 respectively, and about 63% of them were small shorebirds. During the April survey we recorded 37,334 shorebirds, mostly of the small size class. During January and February the Cienega and Santa Clara Slough presented the lowest number of birds detected during the survey in regard to the Islands and the Coast of Baja Sites (less than 300 birds/min). However, during spring migration the Cienega presented the highest rate among all sites (382 birds/min).

Density of shorebirds in the edges of the Cienega was mostly covered by cells that represented abundances of 100-1000 and 1000-10,000 shorebirds per km^2 (Fig. 3). The southwestern areas from the vegetated portion showed the highest number of cells with highest densities (1000-10,000 shorebirds per km^2) occurring in the central areas of the Santa Clara Slough.

In the Delta we detected high density cell areas with more than 10,000 shorebirds per km^2 in the Cienega outflow area (red and dark orange color), in the central portion of the Santa Clara Slough. High aggregation areas (<10,000 shorebird/km^2) were detected also in the Islands (Gore Island) and in the Coast of Baja. Bahia Adair and San Jorge abundances were lower (Fig. 3).

Even though aerial surveys provide a snapshot of the habitat use, it is unlikely that in subsequent years the survey dates were inappropriate since the rest of the monitored areas do not show high variation in abundance. Therefore, habitat use in the Santa Clara Slough seems to be more variable than the intertidal wetlands.

Despite this high variability in shorebird habitat use (5,600-238,000), in five out of the eight surveys conducted the number of shorebirds detected was high enough in the Cienega and Santa Clara Slough to be considered a wetland of regional importance (> 100,000 shorebirds per year) according to the Western Hemisphere Shorebird Reserve Network criteria. The spatial distribution of shorebirds in the Santa Clara Slough was affected by the water level, the size of the inundation zone and possibly by food availability.

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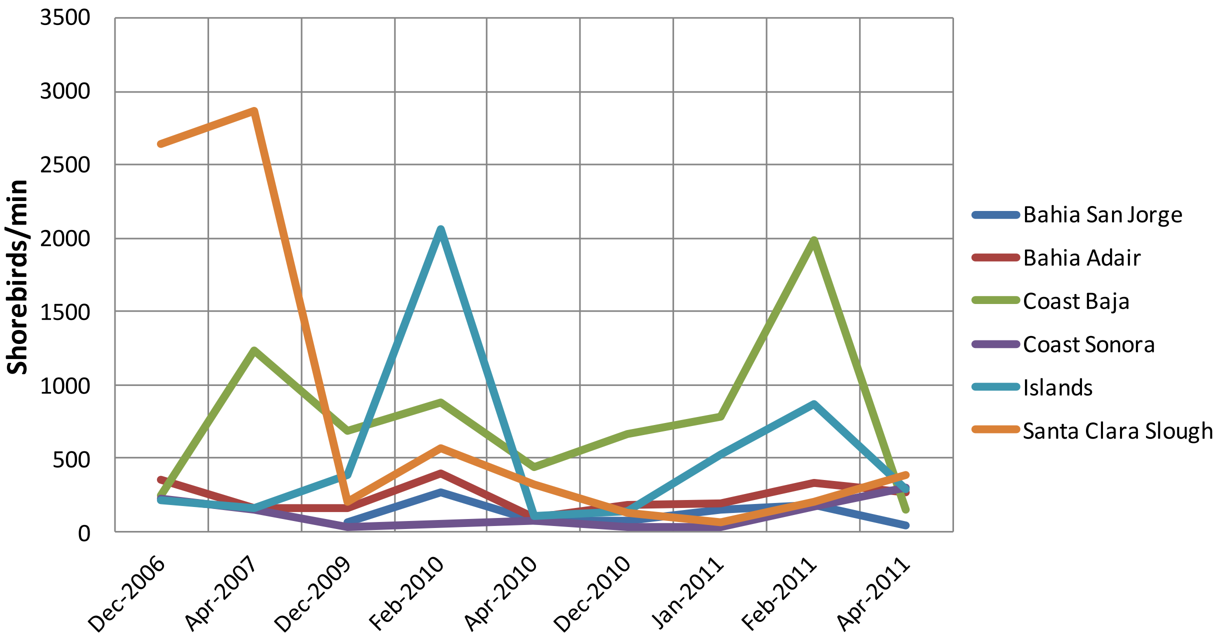

Fig. 1. Shorebirds per minute of aerial survey in the Upper Gulf and Colorado River Delta Region.

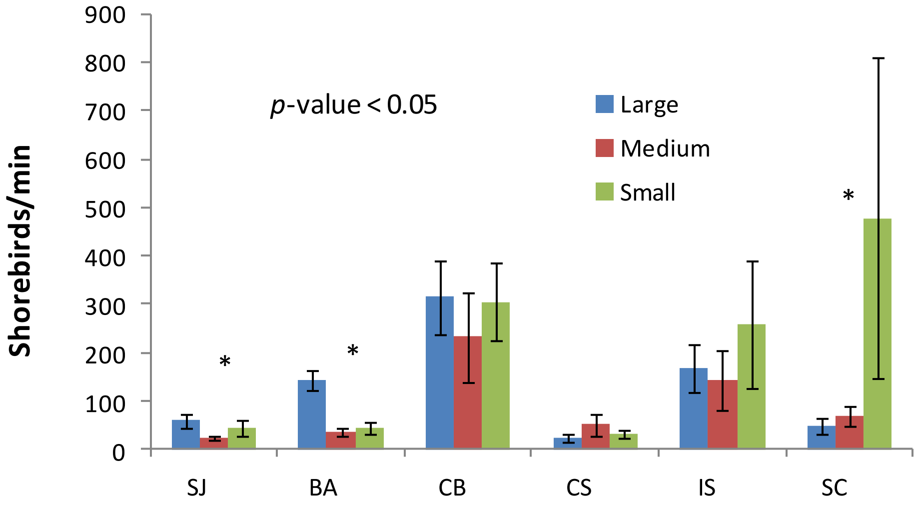

Fig. 2. Mean shorebirds/minute of survey and standard error for each size category in SJ, San Jorge; BA, Bahia Adair; CB, Coast Baja; CS, Coast Sonora, IS, Islands and SC, Cienega and Santa Clara Slough, * = p-value < 0.05.

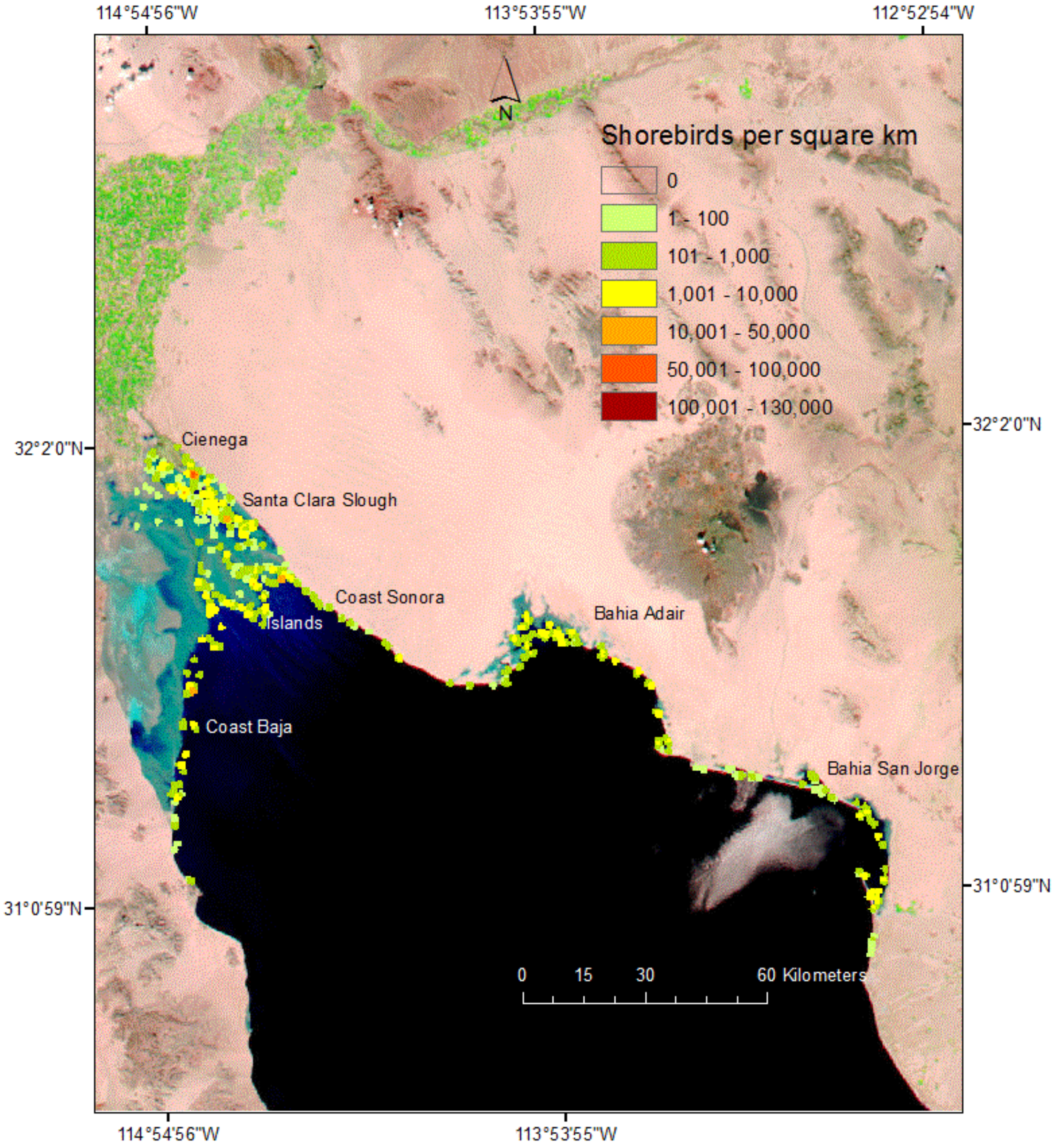

Fig. 3. Shorebirds density (birds/ km^2) determined using the data from nine aerial surveys from 2006 to 2011 conducted during winter and spring migration in the Colorado River Delta and Upper Gulf of California. Cell size 500 m.