REMOTE SENSING METHODS TO CLASSIFY A DESERT WETLAND

by

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This dissertation is dedicated to those that gave the courage to do it,

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ABSTRACT

The Cienega de Santa Clara is a 5600 ha, anthropogenic wetland in the delta of the Colorado River in Mexico. It is the inadvertent creation of the disposal of brackish agricultural waste water from the U.S. into the intertidal zone of the river delta in Mexico, but has become an internationally important wetland for resident and migratory water birds. The marsh is dominated by *Typha domengensis* with *Phragmites australis* as a sub-dominant species in shallower marsh areas. The most important factor controlling vegetation density was fire. The second significant ($P < 0.01$) factor controlling NDVI was flow rate of agricultural drain water from the U.S. into the marsh. Reduced summer flows in 2001 due to canal repairs, and in 2010 during the YDP test run, produced the two lowest NDVI values of the time series from 2000 to 2011 ($P < 0.05$). Salinity is a further determinant of vegetation dynamics as determined by greenhouse experiments, but was nearly constant over the period 2000 to 2011, so it was not a significant variable in regression analyses. Evapotranspiration (ET) and other water balance components were measured in Cienega de Santa Clara; we used a remote sensing algorithm to estimate ET from meteorological data and Enhanced Vegetation Index values from the Moderate Resolution Imaging Spectrometer (MODIS) sensors on the Terra satellite. We used Landsat NDVI imagery from 1978-2011 to determine the area and intensity of vegetation and to estimate evapotranspiration (ET) to construct a water balance. Remote sensing data was supplemented with hydrological data, site surveys and literature citations. The vegetated area increased from 1978 to 1995 and has been constant at about

4200 ha since then. The dominant vegetation type is *Typha domingensis* (southern cattail), and peak summer NDVI since 1995 has been stable at 0.379 (SD = 0.016), about half of $NDVI_{max}$. About 30% of the inflow water is consumed in ET, with the remainder exiting the Cienega as outflow water, mainly during winter months when *T. domingensis* is dormant.

CHAPTER 1

INTRODUCTION

1.1 Importance of Ciénega de Santa Clara

The Colorado River delta in Mexico historically supported several hundred thousand hectares of wetland and riparian habitat (Glenn et al., 2001). Due to construction of upstream dams and the diversion of water for agriculture and urban uses in the U.S. and Mexico, river flows to the delta are much diminished (Nagler et al., 2009). A bright spot in this picture is Cienega de Santa Clara, a large (ca. 5,600 ha) emergent marsh in the eastern portion of the delta (Figure 1) (Glenn et al., 1992; Zengel et al., 1995). This anthropogenic wetland was created starting in 1977 by the discharge of saline agricultural drain water from the Wellton-Mohawk Irrigation District in the U.S. to the north end of the Santa Clara Slough in Mexico, an area of former mudflats in the intertidal zone of the river. The primary plant species in the Cienega is southern cattail (*Typha domengensis* Pers.); with common reed (*Phragmites australis* (Cav.) Trin. ex Steud.) colonizing shallower areas within the *Typha* stands (Figure 2) (Zengel et al., 1995). The Cienega is included in the core area of the Biosphere Reserve of the Upper Gulf of California and Delta of the Colorado River, and supports numerous species of resident and migratory water birds, as well as other wildlife (Zengel et al., 1995, 1996; Hinojosa-Huerta et al., 2001, 2002). It supports 80% of the remaining Yuma clapper rails, a listed endangered species in both the U.S. and Mexico (Hinojosa-Huerta, 2001, 2002).

Water flows to the Cienega are not guaranteed (Gabriel and Keilli, 2010); in fact, the Cienega was the inadvertent creation of the 1974 Colorado River Basin Salinity Control Act in the U.S. (Glenn et al., 1992). Before this act was passed, Wellton-Mohawk drain water was delivered to Mexico in the Colorado River as part of their allotment of Colorado River water. However, the brackish water caused salinity problems in Mexican agricultural fields, and the U.S. pledged to replace Wellton-Mohawk drainage with higher quality water, and to build the Yuma Desalting Plant to ultimately desalinize the Wellton-Mohawk drainage water for delivery to Mexico. The Main Outlet Drain Extention (MODE) canal was built to convey drainage water to the intertidal zone of the Gulf of California while the plant was under construction, and ultimately to receive reverse-osmosis effluent brine from the YDP. However, due to delays and lack of funding, the YDP has only operated during brief test runs: at 33% capacity for 6 months in 1993; at 10% capacity for 3 months in 2007; and at 30% capacity for12 months in 2010-2011. Except during test runs of the YDP, flows to the Cienega have averaged about 4 m s^{-1} at a salinity of 2.8 g L^{-1} total dissolved solids since the MODE became operational (Garcia-Hernandez et al., 2000; Huckelbridge et al., 2010). As a result of these discharges, an internationally important wetland has been created.

1.2 Dissertation format

This dissertation is presented as three research drafts focused in the effects of the

YDP on the Cienega of Santa Clara imagery to assess the Cienega vegetation development during the test run in 2010-2011. The overall goal was to contribute to the development of management tools for those agencies and stakeholders charged with maintaining the environmental values of the Cienega while meeting treaty obligations to provide water to Mexico. The Cienega was formed on the principle of self-design. It was the water from agricultural drain water from the Wellton-Mohawk Irrigation District in the U.S, which is discharged onto the mudflats of the Santa Clara Slough. Since 1978 water has entered the Cienega in the Main Outlet Drain Extension (MODE) canal. Also, we measured evapotranspiration (ET); and we studied the development of the Cienega de Santa Clara from its creation in 1977 to the present to determine if it is on a sustainable trajectory.

For the first draft we used ground data with satellite imagery, high resolution images (Quickbird and WorldView 2), Moderate Resolution Imaging Spectrometer (MODIS) and Landsat imagery to assess the Cienega vegetation development. Also, we used a vegetation-index-based remote sensing method tested against a salt-and-water balance approach to measure ET and to construct a water balance for the Cienega de Santa Clara, a brackish *Typha*/*Phragmites* marsh in the delta of the Colorado River.

For the second draft we measured evapotranspiration (ET). This study was conducted from 2009-2011, before, during and after a trial run of the Yuma Desalting Plant in the USA, which will divert water from the wetland and replace it with brine from the desalting operation. We used a remote sensing algorithm to estimate ET from meteorological data and Enhanced Vegetation Index values from the Moderate Resolution Imaging Spectrometer (MODIS) sensors on the Terra satellite. ET estimates from the MODIS method were compared to results from a mass balance of inflows and outflows over the study period

For the third draft we studied the development of the Cienega de Santa Clara from its creation in 1977 to the present to determine if it is on a sustainable trajectory in terms of vegetation, hydrology and habitat value. We used Landsat NDVI imagery from 1978-2011 to determine the area and intensity of vegetation and to estimate evapotranspiration (ET) to construct a water balance. Remote sensing data was supplemented with hydrological data, site surveys and literature citations.

Appendix A

The objectives of this study were: 1) map vegetation in the Cienega before, during and after the test run of the YDP to determine effect of plant operation on the marsh vegetation; 2) determine longer term trends in vegetation dynamics in response to inflows, salinity and fire events; and, 3) identify the main factors controlling vegetation extent and green foliage density in the Cienega. In this study combined ground data with satellite imagery, including high resolution images for detailed vegetation mapping (Quickbird and WorldView 2) and high-frequency, moderate resolution images for detecting vegetation dynamics over time, using the Moderate Resolution Imaging

Spectrometer (MODIS) sensors on the Terra satellite. It determined the current status of the Cienega, its water budget and its sensitivity to environmental factors such as flows, salinities and fire.

Appendix B in this study used a vegetation-index-based remote sensing method tested against a salt-and-water balance approach to measure ET and to construct a water balance for the Cienega de Santa Clara, a brackish Typha/Phragmites marsh in the delta of the Colorado River, Mexico (Glenn et al., 1992; Zengel et al., 1995). At approximately 5,000 ha, this is perhaps the largest emergent marsh in the Sonoran Desert. It is supported mainly by flows of agricultural drainage water from the U.S., which discharge into the intertidal zone of the delta. Water has flowed at about 4.0 m3 s-1 at a salinity of 2.8 g L-1 TDS since 1976, creating a stable marsh which is 89% vegetated and 11% open water. T. domingensis is the dominant species, making up over 90% of the plant cover, while P. australis makes up 7% of the plant cover and 20 other species grow at lower densities along the edges of the marsh (Glenn et al., 1995; Zengel et al., 1995; Mexicano, 2012). It supports numerous species of water birds which use it as a nesting area and as a stopover site during their migration on the Pacific Flyway Glenn et al., 2001; Gomez-Sapiens et al., 2012; Hinojosa-Huert et al., 2012). 80% of the remaining endangered Yuma Clapper Rails nest in the Cienega (Hinojosa-Huerta et al., 2001. 2002). It provides a good case study for conducting a wetland water budget, because inflows are measured, and it is isolated from adjacent ecosystems (Huckelbridge et al., 2010).

Flow gauges monitored the inflows of water to the marsh, and salinity was measured in the inflow water and at numerous recording stations throughout the vegetated area of the marsh (Garcia-Hernandez et al., 2012). This allowed us to check the accuracy of the satellite-derived ET estimates by a mass balance approach (Jia et al., 2011), and to construct a complete water balance for the Cienega at monthly time steps during operation of the YDP. Our goal was to develop monitoring protocols to estimate changes in green foliage density and ET in response to inflow volumes and salinities.

Appendix C in this study examines the long-term stability of Cienega de Santa Clara, an anthropogenic marsh in the delta of the Colorado River in Mexico (Glenn et al., 1992; Flessa et al., 2012, this volume). The Cienega was formed on the principle of self-design. It was the inadvertent creation of disposal of agricultural drain water from the Wellton-Mohawk Irrigation Distict in the U.S, which is discharged onto the mudflats of the Santa Clara Slough in the intertidal zone of the Colorado River (Figure 1). Since 1978 water has entered the Cienega in the Main Outlet Drain Extension (MODE) canal at a rate of approximately 4-5 m3 s-1 at a salinity of 2-3 g L-1 TDS. The discharge point was originally a small (50 ha) marsh supported by local agricultural drain water from the Mexicali Valley Irrigation District. It has developed into a 5,635 ha marsh dominated by Typha domengensis (southern cattail), with smaller patches of Phragmites australis (common reed), Schoenoplectus americanus (three-square bulrush) and 21 other hydrophytes and with 13% open water lagoons within the marsh (Mexicano et al., 2012, this volume). It is a major wintering and stopover site for waterbirds on the Pacific

Flyway (Hinojosa-Huerta et al., 2008), and supports the largest remaining breeding population of the endangered Yuma clapper rail on the Lower Colorado River (Hinojosa-Huerta et al., 2001. 2002). Note that estimates of the size of the Cienega vary depending on how the perimeter is defined (Glenn et al., 2012, this volume).

The only regular maintenance activity is periodic dredging to keep the entry point for water from being block by silt. Although the source water from agricultural wells in the Wellton-Mohawk valley is relatively free of silt, the MODE canal passes beside the Gran Desierto de Altar on its way to the Cienega, and this unstabilized dune field adds sand to the canal, which is then deposited at the mouth of the Cienega. So far the discharge point has been extended 500 m from the original end of the canal into the marsh area by dredging (authors' personal observations). The Cienega also experiences fires that burn over most of the vegetation at irregular intervals (Yamillet-Carrillo et al., 2012, this volume). Fires occur in winter or spring when the dominant vegetation is dormant and flammable. Fires are either deliberately set by local residents to improve access, or by lightning strikes. Other than this, the Cienega has been left to develop without intervention.

CHAPTER 2

PRESENT STUDY

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

The Cienega de Santa Clara is a 5600 ha, anthropogenic wetland in the delta of the Colorado River in Mexico. It is the inadvertent creation of the disposal of brackish agricultural waste water from the U.S. into the intertidal zone of the river delta in Mexico, but has become an internationally important wetland for resident and migratory water birds. This study was conducted from 2009-2011, before, during and after a trial run of the Yuma Desalting Plant in the USA, which will divert water from the wetland and replace it with brine from the desalting operation. It used high-resolution Quickbird and WorldView 2 satellite imagery validated by ground surveys to prepare Normalized Difference Vegetation Index (NDVI) vegetation maps at approximately four month intervals over the study period. The marsh is dominated by Typha domengensis with Phragmites australis as a sub-dominant species in shallower marsh areas, and with open water lagoons covering 15% of the marsh area. Vegetation density was sensitive to inflow volumes and salinities, and it was concluded that any reduction in inflow volumes will result in a linear decrease in green foliage density in the marsh.

These results were confirmed by an analysis of evapotranspiration (ET) over the years 2000 to 2011. ET was estimated with the Enhanced Vegetation Index from the MODIS sensors on the Terra satellite, and validated by a mass balance approach based on inflow and outflow volumes and salinities. ET was proportional to the volume of inflows, but was also markedly stimulated by fires. Spring fires in 2006 and 2011 burned off accumulated thatch, resulting in vigorous growth of new leaves and a 30% increase in annual ET compared to non-fire years. Following fires, peak summer ET rates were equal to ETo, while in non-fire years peak ET was equal to only one-half to two-thirds of ETo. Over annual cycles, ET was always lower than ETo, because *T. domengensis* is dormant in winter and shades the water surface, reducing direct evaporation. Thus, ET of a Typha marsh is likely to be less than an open water surface under most conditions.

A further study used archival Landsat imagery to investigate the long-term stability of the vegetation in the Cienega from its creation in 1977 to the present to determine if it is on a sustainable trajectory in terms of vegetation, hydrology and habitat value. We used Landsat NDVI imagery from 1975-2011 to determine the area and intensity of vegetation and to estimate evapotranspiration (ET) to construct a water balance. Remote sensing data was supplemented with hydrological data, site surveys and literature citations. Flows into the marsh have been stable both month-to-month and year-to-year, with a mean annual value of 4.74 m3 s-1 (SD = 1.03). Salinity has been stable with a mean value of 2.09 g L-1 (SD = 0.13). The vegetated area increased from 1978 to 1995 and has been constant at about 4200 ha since then. Peak summer NDVI since 1995 has been stable at 0.379 (SD = 0.016), about half of NDVImax. About 30% of the inflow water is consumed in ET, with the remainder exiting the Cienega as outflow water, mainly during winter months when *T. domingensis* is dormant. Periodic monitoring of selenium, trace metals and pesticide residues in water, sediments and biota have not detected levels of concern, and nesting populations of the endangered Yuma clapper rails (*Rallus longirostris yumanensis*) have remained large since they were first monitored in 1989. The stability of the Cienega is attributed to high flow rates that flush out excess salts and contaminants, and to tidal flushing of outflows to the sea at the southern end of the Cienega.

REFERENCES

Abarca, F.U, Ingraldi, M.F., Varela-Romero, A., 1993. Observations on the Desert Pupfish (Cyprinodon macularius, Yuma Clapper Rail Rallus longirostris yumanensis and Shorebird Communities in the Cienega de Santa Clara, Sonora, Mexico. September 1993. Nongame and Endangered Wildlife Technical Report, Arizona Game and Fish Department, Phoenix, Arizona.

Ayers, R.S., Westcott, D.W., 1985. Water Quality for Agriculture. FAO Irrigation and Drainage Paper No. 29, Food and Agriculture Organization of the United Nations, Rome.

AZMET. The Arizona Meteorological Network; University of Arizona: Tucson, AZ, USA, 2012.
Available online: http://cals.arizona.edu/azmet/ .

Baldi, G., Nosetto, M.D., Aragon, R., Aversa, F., Paruelo, J.M., Jobbagy, E.T., 2008. Long-term satellite NDVI data sets: Evaluating their ability to detect ecosystem functional changes in South America. Sensors 8, 5397-5425.

Battin, J., 2004. When good animals love bad habitats: Ecological traps and the conservation of animal populations. Conserv. Biol. 18, 1482-1491.

Baugh, W.M., Groeneveld, D.P., 2006, Broadband vegetation index performance evaluated for a low-cover environment. Int. J. Remote Sens. 27, 4715-4730.

Beare, P.A., Zedler, J.B., 1987. Cattail invasion and persistence in a coastal salt marsh: The role of salinity reduction. Estuaries 10, 165-170.

Bedford, B.L., DeGaetano, A., Siegel, D.I., 1999. Quantification of the water budget and nutrient loading in a small peatland. J. Am. Water Resour. Assoc. 35, 753-769.

Bijoor, N.S., Pataki, D.E., Rocha, A.V., Goulden, M.L., 2011. The application of delta ^{18}O and delta D for understanding water pools and fluxes in a Typha marsh. Plant Cell Environ. 34, 1761-1775.

Bridgham, S.D., Megonigal, J.P., Keller, J.K., Bliss, N.B., Trettin, C., 2006. The carbon balance of North American wetlands. Wetlands 26, 889-916.

Brouwer, C.; Heibloem, M. Irrigation Water Management Training Manual No. 3; FAO: Rome,
Italy, 1986.

Bullock, A., Acreman, M., 2003. The role of wetlands in the hydrological cycle. Hydrol. Earth Sys. Sci. 7, 359-389.

Cohen M.J., Morrison J.I., Glenn E.P., 1999. Haven or Hazard: The Ecology and Future of the Salton Sea. Pacific Institute, Oakland, CA.

Conway, C.J., Nadeau, C.P., Priest, L., 2010. Fire helps restore natural disturbance regime to benefit rate and endangered marsh birds endemic to the Colorado River. Ecol. Appl. 20, 2024-2035.

Drexler, J.Z., Anderson, F.E., Snyder, R.L., 2008. Evapotranspiration rates and crop coefficients for a restored marsh in the Sacramento-San Joaquin Delta, California, USA. Hydrological Processes 22, 725-735.

Drexler, J.Z., Snyder, R.L., Spano, D., Paw, K.T.U., 2004. A review of models and micrometeorological methods used to estimate wetland evapotranspiration. Hydrol. Processes 18, 2071-2101.

Eddleman, W.R., 1989. Biology of the Yuma clapper rail in the southwestern U.S. and northwestern Mexico. Final Rep. Intra-Agency Agreement No. 4-AA-30-02060, Reclamation, Yuma Area Office, Yuma, Ariz. 127 pp.

Farnsworth, E.J., Meyerson, L.A., 2003. Comparative ecophysiology of four wetland plant species along a continuum of invasiveness. Wetlands 23, 750-762.

Flores, C., Bounds, D.L., Ruby, D.E., 2011. Does prescribed fire benefit wetland vegetation? Wetlands 31, 35-44.

Gabriel, J., Kelli, L., 2010. The Yuma desalting plant and Cienega de Santa Clara dispute: a case study review of a workgroup process. Water Policy 12, 401-415.

Garcia-Hernandez, J., Glenn, E.P., Artiola, J., 2000. Bioaccumulation of selenium)Se) in the Cienega de Santa Clara wetland, Sonora, Mexico. Ecotox. Environ. Safety 46, 298-304.

García-Hernández, J., King, K.A., Velasco, A.L., Shumilin, E., Mora, M.A., Glenn, E.P., 2001. Selenium, selected inorganic elements, and organochlorine pesticides in bottom material and biota from the Colorado River delta. J. Arid Environ. 49, 65-89.

García-Hernández, J., Sapozhnikova, Y.V., Schlenk, D., Mason, A.Z., Hinojosa-Huerta, O., Rivera-Díaz, J.J., Ramos-Delgado, N.A., Sánchez-Bon, G., 2006. Concentration of contaminants in breeding bird eggs from the Colorado River Delta, Mexico. Environ. Toxicol. Chem. 25, 164-1647.

Garde, L.M., Nicol, J.M., Conran, J.G., 2004. Changes in vegetation patterns on the margins of a constructed wetland after 10 years. Ecol. Manage. Restor. 5, 111-117.

Glenn, E.P., Felger, R.S., Burquez, A., Turner, D.S., 1992. Cienega de Santa Clara – Endangered wetland in the Colorado River Delta, Sonora, Mexico. Natur. Res. J. 32, 817-824.

Glenn, E.P., Huete, A.R., Nagler, P.L., Nelson, S.G., 2008. Relationship between remotely-sensed vegetation indices, canopy attributes and plant physiological processes: What vegetation indices can and cannot tell us about the landscape. Sensors 8, 2136-2160.

Glenn, E.P., Lee, C., Felger, R., Zengel, S., 1996. Effects of water management on the wetlands of the Colorado River delta, Mexico. Conserv. Biol. 10, 1175-1186.

Glenn, E., Thompson, T.L., Frye, R., Riley, J., Baumgartner, D., 1995. Effects of salinity on growth and evapotranspiration of *Typha domingensis* Pers. Aquatic Bot. 52, 75-91.

Glenn, E.P., Zamora-Arroyo, F., Nagler, P.L., 2001. Ecology and conservation biology of the Colorado River delta, Mexico. J. Arid Environ. 49, 5-15.

Goulden, M.L., Livak, M., Miller, S.D., 2007. Factors that control Typha marsh evapotranspiration. Aquatic Bot. 86, 97-106.

Grace, J.B., 1989. Effects of water depth on Typha latifolia and Typha domingensis. Am. J. Bot. 76, 527-535.

Gregoire, C., Elsaesser, D., Huguenot, D., Lange, J., Lebeau, T., Merli, A., Mose, R., Passeport, E., Payraudeau, S., Schutz, T., Schulz, R., Tapia-Padilla, G., Tournebize, J., Trevisan, M., Wanko, A., 2009. Mitigation of agricultural nonpoint-source pesticide pollution in artificial wetland ecosystems. Environ. Chem. Lett. 7, 205-231.

Groeneveld, D.P., Baugh, W. M., 2007. Correcting satellite data to detect vegetation signal for eco-hydrologic analyses. J. Hydrol. 135-145.

Groeneveld, D.P.; Baugh, W.M.; Sanderson, J.S.; Cooper, D.J. Annual groundwater evapotranspiration mapped from single satellite scenes. J. Hydrol. 2007, 344, 146-156.

Hellings, S.E., Gallagher, J.L., 1992. The effects of salinity and flooding on Phragmites australis. J. Appl. Ecol. 29, 41-49.

Hinojosa-Huerta, O., DeStefano, S., Shaw, W.W., 2001. Distribution and abundance of the Yuma clapper rail (Rallus longirostris yumanensis) in the Colorado River delta, Mexico. J. Arid Environ. 49, 171–182.

Hinojosa-Huerta O., DeStefano, S., Shaw, W.W., 2002. Evaluation of call-response surveys for monitoring breeding Yuma Clapper Rails (Rallus longirostris yumanensis). J. Field Ornithol. 73, 151-155.

Hinojosa-Huerta, O., Rivera0Diaz, J.J., Iturribarria-Rojas, H., Calvo-Fonseca, A., 2008. Population trends of Yuma clapper rails in the Colorado River delta, Mexico. Stud. Avian. Biol. 37, 69-73.

Hinojosa-Huerta, O., Iturrabarria-Rojas, H., Zamora-Hernandex, E., Calvo-Fonseca, A., 2008. Densities, species richness and habitat relationships of avian species in the Colorado River, Mexico. Stud. Avian Biol. 37, 74-82.

Huckelbridge, K.H., Stacey, M.T., Glenn, E.P., Dracup, J.A., 2010. An integrated model for evaluation hydrology, hydrodynamics, salinity and vegetation cover in a coastal desert wetland. Ecol. Eng. 36, 850-861.

Huete, A., Didan, K., van Leeuwen, W., Miura, T., Glenn, E., 2011. MODIS vegetation indices. Land Rem. Sens. Environ. Change 11, 579-602.

Jolly, I.D., McEwan, K.L., Holland, K.L., 2008. A review of groundwater-surface water interactions in arid/semi-arid wetlands and the consequences of salinity for wetland ecology. Ecohydrol. 1, 43-58.

Kaplan, D., 2012. Instability in newly established wetlands? Trajectories of floristic change in the re-flooded Hula poeatland, northern Israel. Mires Peat 9, Article 05, 1-10,

Kerr, J., Ostrovsky, M., 2003. From space to species: ecological applications for remote sensing. Trends in Ecol. Evol. 18, 299-305.

Leitz, F.B., Ewoldsen, E.I., 1977. Design criteria for the Yuma desalting plant. Desalination 24, 321-340.

Lemly, A.D., 1994. Irrigated agriculture and freshwater wetlands: A struggle for coexistence in the western United States. Wetlands Ecol. Manage. 3, 3-15.

Lenters, J.D., Cutrell, G.J., Istanbullouglu, E., Scott, D.T., Herrman, K.S., Irmak, A., Eisenhauer, D.E.2011. Seasonal energy and water balance of a *Phragmites australis*-dominated wetland in the Republican River basin of south-central Nebraska (USA). J Hydrol. 408, 19-34.

Letey, J., 2000. Soil salinity poses challenges for sustainable agriculture and wildlife. Calif. Agric. 54, 43-48.

Lin, Z.Q., Terry, N., Gao, S., Mohamed, S., Ye, Z.H., 2010. Vegetation changes and partitioning of selenium in 4-year-old constructed wetlands treating agricultural drainage. J. Phytorem. 3, 255-267.

Liu, G.D., Ga, B., Miao, S.L., Li, Y.C., Migliaccio, K.W., Qian, Y., 2010. Phosphorous release from ash and remaining tissues of two wetland species after a prescribed fire. J. Environ. Qual. 39, 1585-1593.

Masoner, J.R., Stannard, D.I.,, 2010. A comparison of methods for estimating open-water evaporation in small wetlands. Wetlands 30, 513-524.

Mitsch, W.J. 1992. Landscape design and the role of created, restored, and natural riparian wetlands in controlling nonpoint source pollution. Ecol. Eng. 1, 27-47.

Mitsch, W.J., Gosslink, J.G., 2000. Wetlands, 3rd Edition. Van Norstrand Reinhold, New York.

Mitsch, W.J., Wilson, R.F., 1996. Improving the success of wetland creation and restoration with know-how, time, and self-design. Ecol. Appl. 6, 77-83.

Mitsch, W.J., Zhang, L., Anderson, C.J., Altor, A.E., Hernandez, M.E., 2005. Creating riverine wetlands: Ecological succession, nutrient retention, and pulsing effects. Ecol. Eng. 25, 510-527.

Moreno-Mateos, D., Comin, F.A., 2010. Integrating objectives and scales for planning and implementing wetland restoration and creation in agricultural landscapes. J. Environ. Manage. 11, 2087-2095.

Mould, D.J., Frahm, E., Salzmann, T., 2011. Evaluating the use of diurnal groundwater fluctuations for estimating evapotranspiration in wetland environments: case studies in England and northeast Germany. Ecohydrol. 3, 294-305.

Muller, E., 1997. Mapping riparian vegetation along rivers: Old concepts and new methods. Aquatic Bot. 58, 411-437.

Nagler, P.L.; Cleverly, J.; Glenn, E.; Pampkin, D.; Huete, A.; Wan, Z.M. Predicting riparian evapotranspiration from MODIS vegetation indices and meteorological data. Remote Sens. Environ. 2005, 94, 17-30.

Nagler, P.L., Glenn, E.P., Hursh, K., Curtis, C., Huete, A., 2005. Vegetation mapping for change detection on an arid-zone river. Environ. Monitor. Assess. 109, 255-274.

Nagler, P.L., Glenn, E.P., Hinojosa-Huerta, O., 2009. Synthesis of ground and remote sensing data for monitoring ecosystem functions in the Colorado River Delta, Mexico. Rem, Sens. Environ. 113, 1473-1485.

Nagler, P.L.; Morino, K.; Murray, R.S.; Osterberg, J.; Glenn, E.P. An Empirical Algorithm for Estimating Agricultural and Riparian Evapotranspiration Using MODIS Enhanced Vegetation Index and Ground Measurements of ET. I. Description of Method. Remote Sens. 2009, 1, 1273-1297.

Nagler, P.; Scott, R.; Westenburg, C.; Cleverly, J.; Glenn, E.; Huete, A. Evapotranspiration on
western US rivers estimated using the Enhanced Vegetation Index from MODIS and data from
eddy covariance and Bowen ratio flux towers. Remote Sens. Environ. 2005, 97, 337-351.

Oak Ridge National Laboratory Distributed Active Archive Center (ORNL DAAC). MODIS
Subsetted Land Products, Collection 5; ORNL DAAC: Oak Ridge, TN, USA. Available online:
www.daac.ornl/MODIS/modis.html.

Piest, L. and J. Campoy, 1998. Report of Yuma Clapper Rail Surveys at Cienega de Santa Clara, Sonora. Arizona Game and Fish Department, Yuma Regional Office, Yuma Arizona.

Sun, Z.P., Wei, B., Su, W., Shen, W.M., Wang, C.A., You, D.A., Liu, Z., 2010. Evapotranspiration estimation based on the SEBAL model in the Nansi Lake Wetland of China. Math. Computer Model. 54, 1086-1092.

Tanner, R., Glenn, E.P., Moore, D., 1999. Food chain organisms in hypersaline, industrial evaporation ponds. Water Environ. Res. 71, 494-505.

Taylor, J.R., 1997. An Introduction to Error Analysis, 2nd Edition. University Science Books, Sausalito, CA.

Tarver, D.P., Rogers, J.A., Mahler, M.J., Lazor, R.L., 1986. Aquatic and Wetland Plants of Florida. Third Edition. Florida Department of Natural Resources, Tallahassee, Florida.

Tian, H.Q., Xu, X.F., Miao, S.L., Sindhoj, E., Beltran, B.J., Pan, Z.J., 2010. Modeling ecosystem responses to prescribed fires in a phosphorous-enriched Everglades wetland: 1. Phosphorus dynamics and cattail recovery. Ecol. Model. 221, 1252-1266.

Towler, B.W., Cahoon, J.E., Stein, O.R., 2004. Evapotranspiration crop coeffients for cattail and bulrush. J. Hyrdol. Eng. 9, 235-239.

U.S. Department of Interior, 2004. Reverse Osmosis Treatment of Central Arizona Project Water for the City of Tucson. Desalination Research and Development Report No. 36, Bureau of Reclamation, Denver, Colorado.

Urban, N.H., Davis, S.M., Aumen, N.G., 1993. Flucuations in sawgrass and cattail densities in Everglades-Water-Conservation-Area-2A under varying nutrient, hydrologic and fire regimes. Aquatic. Bot. 46, 203-223.

Verbesselt, J., Hyndman, R., Newnham, G., Culvenor, D., 2010. Detecting trend and seasonal change in satellite time series. Rem. Sens. Environ. 114, 106-115.

Wijt, H.B., Gallagher, J.L., 2006. Effect of oxygen availability and salinity on early life history stages of salt marsh plants. I. Different germination strategies of Spartina alterniflora and Phragmites australis (Poaceae). Am. J. Bot. 83, 1337-1342.

Wu, L., 2004. Review of 15 years of research on ecotoxicology and remediation of land contaminated by agricultural drainage sediment rich in selenium. Ecotoxic. Environ. Safe. 57, 257-269.

Zedler, J.B., Callaway, J.C., 1999. Tracking wetland restoration: Do mitigation sites follow desired trajectories? Restor. Ecol. 7, 69-73.

Zengel, S.A., Glenn, E.P., 1996. Presence of the endangered desert pupfish (*Cyprinodon macularius*, Cyprinidontidae) in Cienega de Santa Clara, Mexico, following an extensive marsh dry-down. Southwest. Natur. 41, 73-78.

Zengel, S.A., Meretsky, V.J., Glenn, E.P., Felger, R.S., Ortiz, D., 1995. Cienega de Santa Clara, a remnant wetland in the Rio Colorado delta (Mexico) – Vegetation distribution and the effects of water flow reduction. Ecol. Eng. 4, 19-36.

Zhoui, L., Zhou, G.S., Liu, S.H., Sui, X.H., 2010. Seasonal contribution and interannual variation of evapotranspiration over a reed marsh (*Phragmites australis*) in Northeast China from 3-year eddy covariance data. Hydrol. Processes 24, 1039-1047.

APPENDIX A

VEGETATION DYNAMICS IN RESPONSE TO WATER INFLOW RATES AND FIRE IN A BRACKISH *TYPHA DOMINGENSIS* (PERS) MARSH IN THE DELTA OF THE COLORADO RIVER, MEXICO

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Abstract

The Cienega de Santa Clara is a 5600 ha, anthropogenic wetland in the delta of the Colorado River in Mexico. It is the inadvertent creation of the disposal of brackish agricultural waste water from the U.S. into the intertidal zone of the river delta in Mexico, but has become an internationally important wetland for resident and migratory water birds. We used high resolution Quickbird and WorldView2 images to produce seasonal vegetation maps of the Cienega before, during and after a test run of the Yuma Desalting Plant, which will remove water from the inflow stream and replace it with brine. We also used moderate resolution, 16-day composite NDVI imagery from the Moderate Resolution Imaging Spectrometer (MODIS) sensors on the Terra satellite to determine the main factors controlling green vegetation density over the years 2000 to 2011. The marsh is dominated by *Typha domengensis* with *Phragmites australis* as a sub-dominant species in shallower marsh areas. The most important factor controlling vegetation density was fire. Spring fires in 2006 and 2011 were followed by much more rapid green-up of *T. domingensis* in late spring and 30% higher peak summer NDVI values compared to non-fire years ($P < 0.001$). Fires removed thatch and returned nutrients to the water, resulting in more vigorous vegetation growth compared to non-fire years. The second significant ($P < 0.01$) factor controlling NDVI was flow rate of agricultural drain water from the U.S. into the marsh. Reduced summer flows in 2001 due to canal repairs, and in 2010 during the YDP test run, produced the two lowest NDVI values of the time series from 2000 to 2011 ($P < 0.05$). Salinity is a further

determinant of vegetation dynamics as determined by greenhouse experiments, but was nearly constant over the period 2000 to 2011, so it was not a significant variable in regression analyses. Any reduction in inflow volumes will result in a linear decrease in green foliage density in the marsh.

Keywords: emergent wetland; cattail marsh; brackish; fire effects; Quickbird; MODIS; remote sensing

A.**1. Introduction**

The Colorado River delta in Mexico historically supported several hundred thousand hectares of wetland and riparian habitat (Glenn et al., 2001). Due to construction of upstream dams and the diversion of water for agriculture and urban uses in the U.S. and Mexico, river flows to the delta are much diminished (Nagler et al., 2009). A bright spot in this picture is Cienega de Santa Clara, a large (ca. 5,600 ha) emergent marsh in the eastern portion of the delta (Figure 1) (Glenn et al., 1992; Zengel et al., 1995). Starting in 1977 this anthropogenic wetland was created by the discharge of saline agricultural drain water from the Wellton-Mohawk Irrigation District in the U.S. to the north end of the Santa Clara Slough in Mexico, an area of former mudflats in the intertidal zone of the river. The primary plant species in the Cienega is southern cattail (*Typha domengensis* Pers.); with common reed (*Phragmites australis* (Cav.) Trin. ex Steud.) colonizing shallower areas within the *Typha* stands (Figure 2) (Zengel et al., 1995). The Cienega is included in the core area of the Biosphere Reserve of the Upper Gulf of California and Delta of the Colorado River, and supports numerous species of resident and migratory water birds, as well as other wildlife (Zengel et al., 1995, 1996; Hinojosa-Huerta et al., 2001, 2002). It supports 80% of the remaining Yuma clapper rails, a listed endangered species in both the U.S. and Mexico (Hinojosa-Huerta, 2001, 2002).

Water flows to the Cienega are not guaranteed (Gabriel and Kelli, 2010); in fact, the Cienega was the inadvertent creation of the 1974 Colorado River Basin Salinity

Control Act in the U.S. (Glenn et al., 1992). Before this act was passed, Wellton-Mohawk drain water was delivered to Mexico in the Colorado River as part of their allotment of Colorado River water. However, the brackish water caused salinity problems in Mexican agricultural fields, and the U.S. pledged to replace Wellton-Mohawk drainage with higher quality water, and to build the Yuma Desalting Plant (YDP) to ultimately desalinize the Wellton-Mohawk drainage water for delivery to Mexico. The Main Outlet Drain Extension (MODE) canal was built to convey drainage water to the intertidal zone of the Gulf of California while the plant was under construction, and ultimately to receive reverse-osmosis effluent brine from the YDP. However, due to delays and lack of funding, the YDP has only operated during brief test runs: at 33% capacity for 6 months in 1993; at 10% capacity for 3 months in 2007; and at 30% capacity for12 months in 2010-2011. Except during test runs of the YDP, flows to the Cienega have averaged about 4 m s^{-1} at a salinity of 2.8 g L^{-1} total dissolved solids since the MODE became operational (Garcia-Hernandez et al., 2000; Huckelbridge et al., 2010). As a result of these discharges, an internationally important wetland has been created.

The present study was part of a monitoring program designed to detect effects of the YDP on the Cienega during the test run in 2010-2011. The objectives of this study were: 1) map vegetation in the Cienega before, during and after the test run of the YDP to determine effect of plant operation on the marsh vegetation; 2) determine longer term trends in vegetation dynamics in response to inflows, salinity and fire events; and, 3) identify the main factors controlling vegetation extent and green foliage density in the

Cienega. The overall goal was to contribute to the development of management tools for those agencies and stakeholders charged with maintaining the environmental values of the Cienega while meeting treaty obligations to provide water to Mexico. The research combined ground data with satellite imagery, including high resolution images for detailed vegetation mapping (Quickbird and Worldview 2) and high-frequency, moderate resolution images for detecting vegetation dynamics over time, using the Moderate Resolution Imaging Spectrometer (MODIS) sensors on the Terra satellite.

A.2. Materials and Methods

A.2.1 Vegetation mapping using high resolution Quickbird and Worldview 2 imagery

Several approaches to vegetation mapping with satellite imagery have been developed (Muller, 1991; Nagler et al., 2005). For example, vegetation types can sometimes be differentiated based on spectral properties, using either supervised or unsupervised classification programs in which satellite bands are combined to produce unique signatures for each vegetation type. Another approach is for trained interpreters to divide the image into polygons representing different vegetation types based on expert opinion. In the present study, there were only two major vegetation types and their locations were stable, and our main interest was in detecting changes in green vegetation density over time. Therefore, we developed an approach based on the Normalized

Difference Vegetation Index (NDVI) (Pettorelli et al., 2005). NDVI is calculated from Red and NIR (Near Infra Red) bands as:

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

NDVI reduces the image to a single layer with NDVI values from -1.0 to +1.0, with water having strongly negative values, soils slightly negative to slightly positive, and vegetation having positive values.

NDVI of vegetation is strongly sensitive to chlorophyll absorption of Red (wavelength = 0.75-1.4 μm) and scattering and reflection of NIR by cell walls and stacked layers of cells in leaves, and provides a measure of canopy "greenness" (Glenn et al., 2008). Vegetation indices have been highly successful in assessing vegetation condition, foliage, cover, phenology, and processes such as evapotranspiration (ET) and primary productivity, related to the fraction of photo synthetically active radiation absorbed by a canopy (Glenn et al., 2008; Kerr and Ostrovsky, 2003; Pettorelli et al., 2005). Vegetation indices are robust satellite data products computed the same way across all pixels in time and space, regardless of surface conditions. As ratios, they can be easily cross-calibrated across sensor systems, ensuring continuity of data sets for long-term monitoring of ecosystems by different satellites and sensor systems (Baldi et al., 2008; Verbesselt et al., 2010).

Vegetation maps were prepared from high-resolution Quickbird or Worldview 2 images at approximately quarterly intervals, 2008 – 2011. Quickbird images were acquired for the Cienega for September 2008; February 2009; April 2009; August 2009; January 2010; and October 2011; and, Worldview 2 images from April 2010; July 2010; and April 2011. Both Quickbird and WorldView2 satellites are owned by Digital Globe, Inc. (Longmont, CO), and sensor systems are inter-calibrated so satellite data can be used interchangeably. Resolution is approximately 0.6 m in the panchromatic (black-and-white, 445-900 nm) bands and 2.4 m in the multispectral (blue, green, red, NIR) bands. The radiometric corrections applied to these products include relative radiometric response between detectors, non-responsive detector fill, and a conversion for absolute radiometry. The sensor corrections account for internal detector geometry, optical distortion, scan distortion, any line-rate variations, and miss-registration of the multi-spectral bands. Hence, digital number (DN) values are accurately related to at-satellite reflectance values, but do not account for effects of atmospheric conditions on band values. In the absence of atmospheric data, bands can be inter-calibrated among images using pseudo-invariant objects on images, such as clear deep water, bright sand, rock or dense vegetation (Song et al., 2001). In the present study, minimum (water), mean and maximum (dense vegetation) NDVI values were treated as pseudo-invariant features, and were similar for spring and summer images, with standard errors less than 5% for minimum and maximum values among images (Figure 3). Hence, further corrections were not attempted.

An Area of Interest (AOI) file was created in ERDAS Imagine software (ERDAS Imagine, Inc., Atlanta, GE) around the perimeter of the Cienega. It had an area of 5635 ha that encompassed the permanently vegetated portion of the marsh plus a buffer zone of areas that were vegetated on some images but not others. This AOI file was applied to each image for analyses of land cover types. NDVI ranges defining land cover classes were determined by comparing known land cover types visible on panchromatic Quickbird images, aerial photographs and in ground surveys with NDVI values of the same features on NDVI images. In an iterative procedure, we used supervised and unsupervised classification programs in ERDAS to divide the September 2009 image into as many as 12 classes, then we inspected the panchromatic image and aerial photographs to attempt to produce a final set of classes that corresponded to identifiable, stable features on the whole set of images. We ultimately defined six classes corresponding to: 1) water; 2) soil or dry *T. domengensis* (when dead or dormant); 3) low-density vegetation; 4) medium density vegetation; 5) high density vegetation; and 6) highest intensity vegetation. The stability of the classes was tested by selecting 60-100 sample points in each class among the panchromatic images, based on visual interpretation of land cover types, in comparison with NDVI ranges selected to define classes (Figure 4). There was minimal overlap between soil, water and vegetation NDVI values. Different vegetation classes tended to overlap as they did not represent different vegetation types, but represented the continuum of green vegetation density across species and seasons. During winter dormancy, dry *T. domengensis* NDVI overlapped soil values and they could not be unambiguously distinguished from each other. However, areas which remained in the soil/dry vegetation class year around were assumed to be soil, whereas areas that moved into the vegetation classes in summer were assumed to be winter-dormant *T. domengensis*. Water NDVI varied over a wide range apparently corresponding to different water depths, but its range of NDVI values did not overlap with the soil/dry vegetation class.

Vegetation maps and statistical summaries of classes were prepared by distributing pixels into each of the six land cover classes. ERDAS Imagine software was used for image processing, production of NDVI images, vegetation mapping and statistical analyses of data.

A.2.2 NDVI from MODIS

MODIS has near-daily coverage of most of the earth with about 250 m resolution (Huete et al., 2011). We used the MOD13Q1 NDVI product, which is a composite of 3-5 high quality, cloud-free images selected for each 16 day collection period. Band values are corrected to at-surface reflectance values and vegetation index products are supplied to end-users as pre-processed, validated data sets. MODIS data in this study was obtained from the Oak Ridge National Laboratory DAAC website (http://daac.ornl.gov/cgi-bin/MODIS/GLBVIZ_1_Glb/modis_subset_order_global_col5.pl). We defined a 1,600 ha study area (256 MODIS pixels) in the center of the Cienega to study changes in NDVI over time (Figure 5). This polygon excludes major open water areas in the north and south ends of the Cienega, as water interferes with the NDVI vegetation signal. It also excludes the western edge of the Cienega, which is prone to periodic drying due to buildup of silt in the entry channel; our goal was to pick a stable area of vegetation to detect changes due to variable inflow rates, fires and salinity changes.

A.2.3 Ground data sources and regression analyses of NDVI on flow volumes and salinities

Flow data were daily mean values recorded in the MODE canal at the Southerly International Boundary between the U.S. and Mexico by the International Boundary and Water Commission (http://www.ibwc.gov/wad/DDQWMSIB.HTM). Salinity of inflow water was monthly values provided by the same source. The occurrence of major fire events in 2006 and 2011 were based on local observations and verified by obtaining archival Landsat 5 images from the U.S. Geological Survey Earth Explorer website (http://earthexplorer.usgs.gov/). Statistical analyses were performed using Systat, Inc. software (Chicago, IL). Ground observations were based on monthly field observations made before, during and after the period of YDP operation in 2010.

MODIS NDVI data was regressed against MODE inflow volume and salinity data for each year from 2000 to 2011. Although the YDP operated for 12 months in 2010-2011, for most of the year replacement water is from other sources in the U.S. and Mexico were discharged into the canal to minimize disturbance to vegetation and wildlife in the Cienega (see Hinojosa-Huerta et al., 2002). No replacement water was available for part of the run; flows were markedly reduced and salinities increased from May 9 to

September 14, 2010. Therefore, we used this time period each year to compare NDVI among years. Analyses were conducted using Systat software (Longmont, CO).

A.3. Results

A.3.1 Vegetation maps and distribution of land cover classes

T. domengensis is winter dormant, greening up in late April and becoming dormant in November at this location. On the other hand, the sub-dominant species, *P. australis,* is green all year in the Cienega. This offers a means of distinguishing between the two species based on phenology, as *P. australis* appears as green islands amidst dormant *T. domengensis* stands in winter (see Figure 2).

Figure 6 illustrates key changes in the vegetation maps over time and Table 1 displays the number of hectares in each cover class on each image. In September 2008 (Figure 6A) vegetation cover was near its maximum area extent, but the western edge of the Cienega fell into the low-density vegetation class. This was due to a dry-down of that portion of the marsh due to the deposition of silt in the north end of the marsh. As it crosses Gran Desierto, wind-blown sand enters the canal and it is deposited in the Cienega, requiring periodic dredging to keep the entry point for water and silt into the Cienega open. Between dredging events, the western edge can be cut off from inflows.

Figure 6B shows the Cienega during winter (February 2009). *T. domengensis* was mostly dormant, and fell mainly into the soil/dry vegetation or low vegetation density classes. The green areas denote *P. australis* stands, which are found in shallower areas around the periphery of the marsh and in islands near the entry point of water, where silt accumulates. Note that more open water is visible in winter than summer, as evaporative demand is low in winter. By August 2009 (Figure 6C), the marsh had re-greened. Dredging had restored part of the western edge of the marsh. Figure 6D shows the marsh in July, 2010, during the test run of the YDP. Although the marsh had greened up normally, most of the vegetation was in the medium-density class, with less high and very-high-density vegetation present compared to September 2008 or August 2009. Figure 6E shows the Cienega in April 2011; just three weeks after a fire had burned most of the dry vegetation. *T. domengensis* responded by rapidly sending up new green shoots, resulting in early greening compared to previous years. Figure 6F shows the Cienega in October 2011. Although the image was captured after peak greenness, most of the vegetation fell in the high-density class. Periodic fire burns off thatch in *Typha* marshes, returning nutrients to the water and allowing additional light penetration to support new growth.

Table 1 reports the actual acreage of each cover class for each date. Across images, open water occupied 727 ha of the marsh; from the spring and summer images, bare soil occupied 1235 ha, mainly around the periphery of the vegetated area, as the AOI file encompassing the study are was purposely drawn with a buffer zone to capture areas that were only occasionally vegetated. The area occupied by *P. australis* was inferred from the amount of vegetation in the medium density or higher vegetation classes in winter, and was about 350 ha. Table 1 also gives mean values and coefficients of variation (CV = standard deviation/mean) for each class across images. CVs for water, soil/dry vegetation and total vegetated area were relative low (0.29-0.37) compared to CVs for individual vegetation classes (0.47-0.91). The classification procedure was able to capture the seasonal and inter-annual vegetation dynamics inherent in this type of marsh. Table 1 supports the visual interpretation of the vegetation maps, showing a decrease in the amount of high-density and highest-density vegetation classes during the run of the YDP compared to August 2009, although the differences were not large. Table 1 shows a marked increase in the high-density vegetation classes in 2011 compared to previous years, due to the fire in the spring of that year.

A.3.2. Flows, salinities and NDVI values from 2000 to 2011

MODIS NDVI values are in Figure 7A and mean flows and salinities during the May 9 - September 14 period for each year are in Figure 7B. Salinities were nearly constant from 2000 to 2011 (mean = 2.66 g L^{-1}, CV = 8.7%), but showed a small increase during the YDP test run in 2010 (Garcia-Hernandez, 2012). Inflows were more variable (mean = 4.03 $m^3 s^{-1}$, CV = 16.8%), with distinct interruptions in summer flows noted for 2001 and 2010. In 2001, flows decreased from 4.0 m s^{-1} in May and June, to 1.2 m s^{-1} in July and August, with normal flows restored in September. Flows in 2010 did not decrease to the same low values as in 2010, but the period of flow interruption was longer, with a mean flow of 2.8 m s^{-1} from May 9 - September 14. NDVI values (Figure 7B) showed the expected annual pattern for vegetation that undergoes winter dormancy, with green-up starting in April, peak greenness occurring in August and September, and senescence of leaves starting in October and November. Peak summer values varied considerably among years. Fires in the spring of 2006 and 2011 burned the dormant *T. domengensis* biomass throughout the marsh, and lead to an early and vigorous green-up in April in both years, with peak NDVI values 30% higher than in non-fire years (Figure 8). Lowest NDVI values were in the two years of reduced flow (2001 and 2010). The mid-summer interruption in flows in 2001 is clearly evident in the NDVI response, which showed a mid-summer dip in NDVI, followed by a recovery in September following restoration of flows (Figure 7A). The pattern in 2010 was different, with peak NDVI reduced over a longer period but not as deeply as in 2001 (Figure 7A).

A multiple linear regression analysis showed that inflow volumes and presence or absence of fire could explain 95% of the variation in NDVI for the May 9 - September 14 period across years (Table 2) ($P < 0.001$), whereas salinity was not significant ($P= 0.702$). Figure 9 shows the linear regression of NDVI on flow rates in both fire years and non-fire years. The regression of NDVI on flows was significant ($P < 0.01$) when all years were included in the analysis and when fire years were excluded, but as expected the slope of the response changed. Figure 9 shows that 2001 and 2010 stand out as years of low flow and low NDVI. Two-way ANOVA of non-fire years shows that 2001 and 2010 differed from other years in both flows ($P < 0.001$) and NDVI ($P = 0.023$).

A.4. Discussion

A.4.1 Land covers classes in Cienega de Santa Clara

Vegetation maps from 2009 - 2011 show that the overall size of the Cienega is stable at about 6,500 ha. Open water accounts for about 15% of the surface area, mainly in the form of lagoons within the *T. domengensis* and *P. australis* stands, but also as transient drainage channels from the periphery onto the mudflats to the west and south of the Cienega. The open water lagoons are stable features and appear to be areas of deeper water where *T. domengensis* has not established. These lagoons are colonized by dense growth of the submerged aquatic species, *Najas marinas*, which might prevent the establishment of emergent plants, explaining the stability of the lagoons over time. The islands of *P. australis* within the *T. domengensis* stands also appear to be stable features. *P. australis* preferentially grows in shallow water areas, and is most abundant along the margins of the Cienega and near the entry point for water at the north end of the Cienega on silt bars that have built up over time. Silt entering the Cienega is accumulating near the entry point for water and, although most of the vegetation units are stable over time, the western edge is prone to periodic dry-downs due to silt build up in the distribution canal, and must be restored through periodic dredging, as captured in the Quickbird series of images in this study.

A.4.2 Factors controlling the extent and density of vegetation in the Cienega

Presence or absence of fire and water inflow rates were both significant factors controlling the density of green vegetation in spring and summer, as estimated by NDVI values on MODIS images. Fires in spring of 2006 and 2011 removed accumulated thatch and returned nutrients to the water (see Liu et al., 2010), resulting in a marked increase in NDVI values the following summer. Fires markedly improve the habitat value of *Typha* marshes for nesting birds, as noted in other studies (Conway et al., 2010). Yuma clapper rail densities were significantly correlated with NDVI values during the nesting period in Cienega de Santa Clara (Gomez-Sapiens et al., 2012. In preparation). Fire timing and frequency appears to be self-limiting in *Typha* marshes. Fires require high thatch levels as fuel; hence newly greened marshes or marshes in summer might not be prone to burning. Fires occur mainly in winter and early spring and are most intense when thatch levels are high. *Typha* grows by initiating new shoots from rhizomes in spring, but the previous year's senescent leaves are retained on the plant, leading to the accumulation of thatch over multiple years that are periodically removed by fire. In the Cienega, fires are sometimes deliberately set by local residents to allow fishermen and hunters easier access to areas within the marsh, and can also be started by lightning strikes.

Flows control vegetation density by determining the flooded area available for the growth of emergent vegetation, and the amount of water available for evapotranspiration and growth. Flows in the MODE tended to be fairly constant during most years from 2000 to 2011, but reduced summer flows in 2001 and 2010 led to reductions in marsh NDVI, and the regression of NDVI on flows was significant ($P < 0.01$) across all years.

Although not a significant factor in the multiple regression analysis, salinity is also a determinant of vegetation density in the Cienega (Huckelbridge et al., 2010). Both greenhouse studies (Glenn et al., 1995; Baeza et al., 2012) and field observations (Glenn et al., 1995) show that *T. domengensis* exhibits a linear decrease in growth with salinity over the range of 0 g L^{-1} total dissolved salts (TDS) (i.e., fresh water), and 6 g L^{-1} TDS, the apparent upper limit for growth. *T. domengensis* survives at salinities up to 8 g L^{-1}, but growth rates approach zero at 6 g L^{-1} in greenhouse experiments, which is also the apparent salinity limit for the distribution of *T. domengensis* in the Cienega de Santa Clara (Glenn et al., 1995).

A.4.3 Comparison to other studies

There is continuing debate about the controls on wetland productivity and whether marshes are profligate water users (Goulden et al., 2007). In some studies productivity of *Typha* spp. is high and evapotranspiration (ET) rates equal or exceed open water evaporation rates (Farnsworth et al., 2003; Towler et al., 2004; Drexler et al., 2008). However, other studies report more modest rates of ET and net primary production for *Typha* marshes (Goulden et al., 2007). The present study shows that the amount of thatch largely controls the NDVI and ET (see Glenn et al., 2012. In preparation) in Cienega de Santa Clara. In restored marshes (Drexler et al., 2008) or lysimeter studies (Towler et al., 2004) ET rates tend to be high, because plants are maintained for maximum productivity. On the other hand, in natural marsh, thatch will unavoidably build up over time, reducing light levels inside the canopy and sequestering nutrients in senescent biomass (Goulden et al., 2007). Figure 8 shows that total seasonal NDVI (areas under curves) was nearly twice as great after fires compared to non-fire years in the Cienega. We postulate that much of the variation in ET and NPP rates noted in the literature for *Typha* wetlands could be due to differences in experimental conditions, in particular to how much thatch is present during measurement periods.

A.4.4 Need for a mechanistic model of vegetation versus flows, fire and salinity levels in the Cienega

The multiple linear regression equation in Table 2 identifies the roles of fire and flow rates in controlling vegetation density, but does not constitute a model of vegetation dynamics in the Cienega. The equation is only predictive over the range of conditions occurring from 2000 to 2011. Thus, salinity does not appear as a significant factor in the equation because it did not vary greatly, but proposed operating scenarios for the YDP do include significant increases in inflow salinities. Furthermore, the dips in NDVI in response to flow reductions in 2001 and 2011 are rather small, due to the transient nature of the flow reductions. More prolonged reductions in flow would be expected to reduce NDVI or marsh area more substantially than noted in this study. The multiple linear regression equation treats fire and flows as separate variables, but in reality an interaction between flows, salinities and presence or absence of fire is expected. The preliminary model formulated by Huckelbridge et al. (2010) can now be refined using data gathered during the 2008-2011 monitoring program (Flessa et al., 2012), to provide a validated tool for predicting the vegetation and hydrological response of the Cienega to alterations in inflows and salinities.

References

Baldi, G., Nosetto, M.D., Aragon, R., Aversa, F., Paruelo, J.M., Jobbagy, E.T., 2008. Long-term satellite NDVI data sets: Evaluating their ability to detect ecosystem functional changes in South America. Sensors 8, 5397-5425.

Conway, C.J., Nadeau, C.P., Priest, L., 2010. Fire helps restore natural disturbance regime to benefit rate and endangered marsh birds endemic to the Colorado River. Ecol. Appl. 20, 2024-2035.

Drexler, J.Z., Anderson, F.E., Snyder, R.L., 2008. Evapotranspiration rates and crop coefficients for a restored marsh in the Sacramento-San Joaquin Delta, California, USA. Hydrological Processes 22, 725-735.

Farnsworth, E.J., Meyerson, L.A., 2003. Comparative ecophysiology of four wetland plant species along a continuum of invasiveness. Wetlands 23, 750-762.

Gabriel, J., Kelli, L., 2010. The Yuma desalting plant and Cienega de Santa Clara dispute: a case study review of a workgroup process. Water Policy 12, 401-415.

Garcia-Hernandez, J., Glenn, E.P., Artiola, J., 2000. Bioaccumulation of selenium)Se) in the Cienega de Santa Clara wetland, Sonora, Mexico. Ecotox. Environ. Safety 46, 298-304.

Glenn, E.P., Felger, R.S., Burquez, A., Turner, D.S., 1992. Cienega de Santa Clara – Endangered wetland in the Colorado River Delta, Sonora, Mexico. Natur. Res. J. 32, 817-824.

Glenn, E.P., Huete, A.R., Nagler, P.L., Nelson, S.G., 2008. Relationship between remotely-sensed vegetation indices, canopy attributes and plant physiological processes: What vegetation indices can and cannot tell us about the landscape. Sensors 8, 2136-2160.

Glenn, E., Thompson, T.L., Frye, R., Riley, J., Baumgartner, D., 1995. Effects of salinity on growth and evapotranspiration of *Typha domingensis* Pers. Aquatic Bot. 52, 75-91.

Glenn, E.P., Zamora-Arroyo, F., Nagler, P.L., 2001. Ecology and conservation biology of the Colorado River delta, Mexico. J. Arid Environ. 49, 5-15.

Goulden, M.L., Livak, M., Miller, S.D., 2007. Factors that control Typha marsh evapotranspiration. Aquatic Bot. 86, 97-106.

Hinojosa-Huerta, O., DeStefano, S., Shaw, W.W., 2002. Evaluation of call-response surveys for monitoring breeding Yuma Clapper Rails (*Rallus longirostris yumanensis*). J. Field Ornith. 73, 151-155.

Hinojosa-Huerta, O., DeStefano, S., Shaw, W.W., 2001. Distribution and abundance of the Yuma clapper rail (*Rallus longirostris yumanensis*) in the Colorado River delta, Mexico. J. Arid Environ. 49, 171-182.

Huckelbridge, K.H., Stacey, M.T., Glenn, E.P., Dracup, J.A., 2010. An integrated model for evaluation hydrology, hydrodynamics, salinity and vegetation cover in a coastal desert wetland. Ecol. Eng. 36, 850-861.

Huete, A., Didan, K., van Leeuwen, W., Miura, T., Glenn, E., 2011. MODIS vegetation indices. Land Rem. Sens. Environ. Change 11, 579-602.

Kerr, J., Ostrovsky, M., 2003. From space to species: ecological applications for remote sensing. Trends in Ecol. Evol. 18, 299-305.

Liu, G.D., Ga, B., Miao, S.L., Li, Y.C., Migliaccio, K.W., Qian, Y., 2010. Phosphorous release from ash and remaining tissues of two wetland species after a prescribed fire. J. Environ. Qual. 39, 1585-1593.

Muller, E., 1997. Mapping riparian vegetation along rivers: Old concepts and new methods. Aquatic Bot. 58, 411-437.

Nagler, P.L., Glenn, E.P., Hursh, K., Curtis, C., Huete, A., 2005. Vegetation mapping for change detection on an arid-zone river. Environ. Monitor. Assess. 109, 255-274.

Nagler, P.L., Glenn, E.P., Hinojosa-Huerta, O., 2009. Synthesis of ground and remote sensing data for monitoring ecosystem functions in the Colorado River Delta, Mexico. Rem, Sens. Environ. 113, 1473-1485.

Pettorelli, N., Vik, J., Mysterud, A., Caillard, J., Tucker, C., Stenseth, N., 2005. Using the satellite-derived NDVI to assess ecological responses to environmental change. Trends Ecol. Evol. 20, 503-510.

Song, C., Woodcock, C.E., Seto, K.C., Lenney, M.P., Macomber, S.A., 2001. Classification and change detection using Landsat TM : when and how to correct atmospheric effects? Rem. Sens. Environ. 75, 230-244.

Towler, B.W., Cahoon, J.E., Stein, O.R., 2004. Evapotranspiration crop coeffients for cattail and bulrush. J. Hyrdol. Eng. 9, 235-239.

Verbesselt, J., Hyndman, R., Newnham, G., Culvenor, D., 2010. Detecting trend and seasonal change in satellite time series. Rem. Sens. Environ. 114, 106-115.

Zengel, S.A., Glenn, E.P., 1996. Presence of the endangered desert pupfish (*Cyprinodon macularius*, Cyprinidontidae) in Cienega de Santa Clara, Mexico, following an extensive marsh dry-down. Southwest. Natur. 41, 73-78.

Zengel, S.A., Meretsky, V.J., Glenn, E.P., Felger, R.S., Ortiz, D., 1995. Cienega de Santa Clara, a remnant wetland in the Rio Colorado delta (Mexico) – Vegetation distribution and the effects of water flow reduction. Ecol. Eng. 4, 19-36.

Table 1. Number of hectares in each covers class in Quickbird or World View 2 images of Ciénega de Santa Clara. “Low”, “medium”, “high” and “highest” refer to intensity of “greenness” not elevation above the ground.

Class	**Sep-08**	**Feb-09**	**Apr-09**	**Aug-09**	**Jan-10**	**Apr-10**	**Jul-10**	**Apr-11**	**Oct-11**	**Mean (CV)**
Water	1007	958	869	543	759	569	465	478	588	723 (29)
Soil/Dry Veg	1736	2249	1581	1509	1483	474	1682	1087	1228	1448 (34)
Low Veg	1886	3027	3786	1335	3991	2569	1442	1952	940	2325 (47)
Medium Veg	726	313	257	1311	250	1949	1535	1380	836	951 (65)
High Veg	779	92	0	1421	0	820	1261	1296	2535	912 (91)
Highest Veg	406	0	45	419	50	154	150	337	407	219 (79)
All Veg	3797	3432	4088	4486	4291	5492	4388	4965	4718	4029 (37)

Table 2. Multiple linear regression of NDVI on inflows in the MODE canal and during the May 9 – September 14 period and presence or absence of fire for years 2000 – 2011 in Cienega de Santa Clara. The regression equation of best fit was: NDVI = 0.041 Flows (m s-1) + 0.143 Fire (0 or 1) + 0.369, $F = 85$, $P < 0.001$. The Standard Coefficient expresses the fraction of variability in the dependent variable that is explained by each dependent variable.

Variable	**Coefficient**	**Std. Coefficient**	**P**
Constant	0.369	0.000	< 0.001
Flows	0.041	0.370	0.002
Fire	0.143	0.745	< 0.001

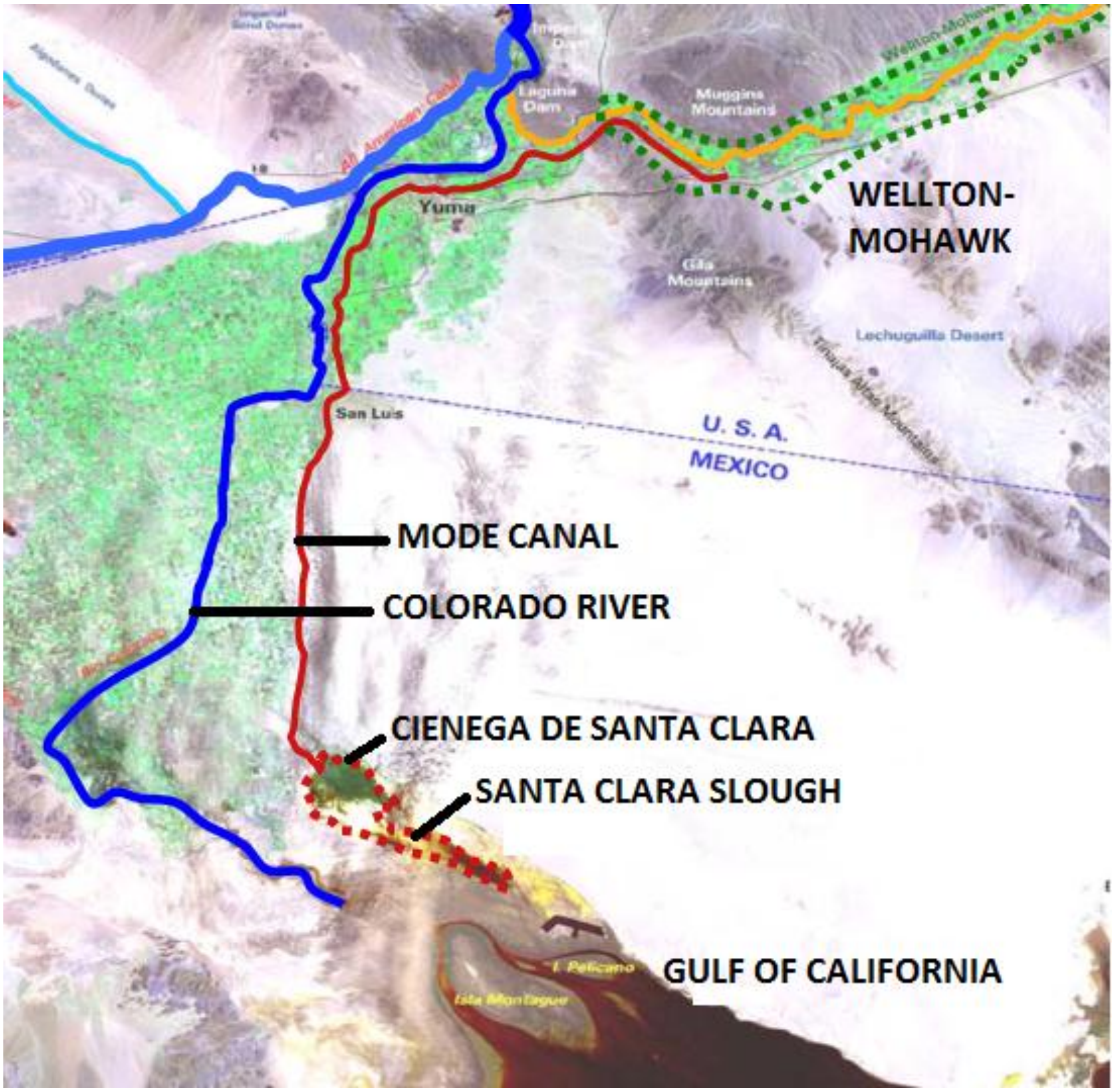

Figure 1. Locator map showing Wellton-Mohawk Irrigation District, the origin of water flowing to Cienega de Santa Clara in the MODE canal.

Figure 2. Aerial photograph taken by Francisco Zamora-Arroyo in January, 2010, showing green *Phragmites australis* amidst dormant *Typha domengensis* in the Ciénega de Santa Clara.

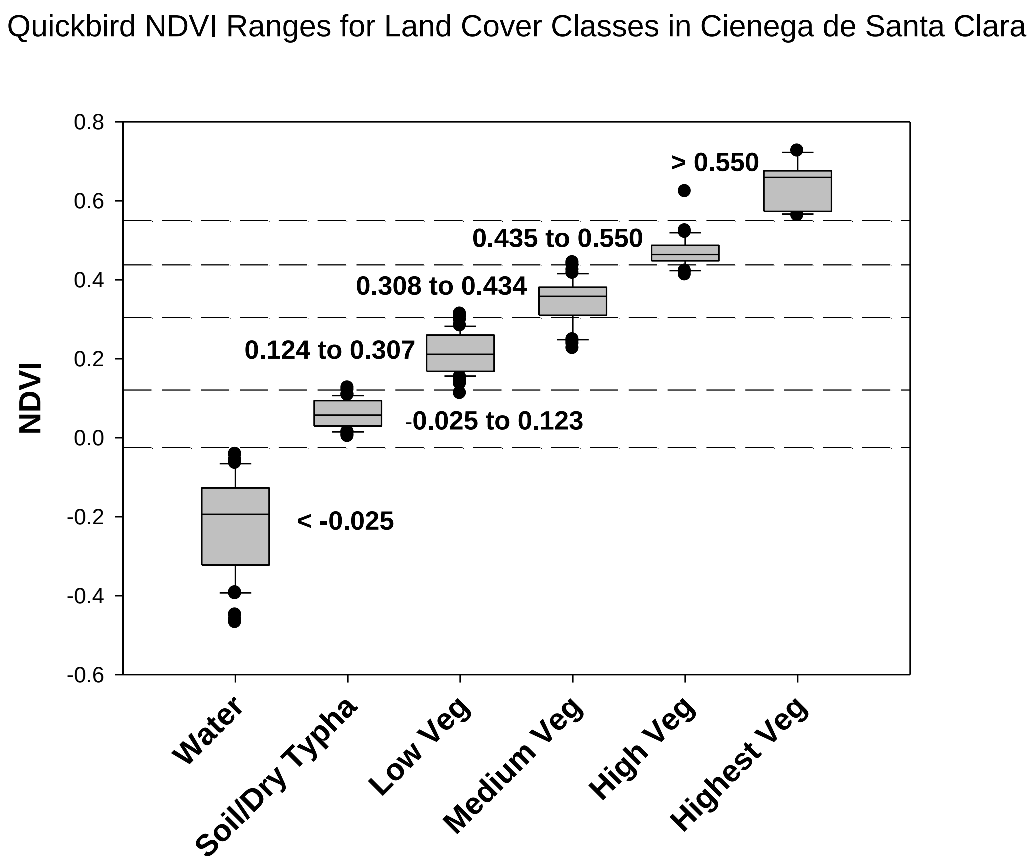

Figure 3 Stability of NDVI values among Quickbird (QB) and WorldView 2 (WV2) images used in producing the seasonal Ciénega vegetation maps. Values are minimum, mean and maximum values for all pixels on each image.

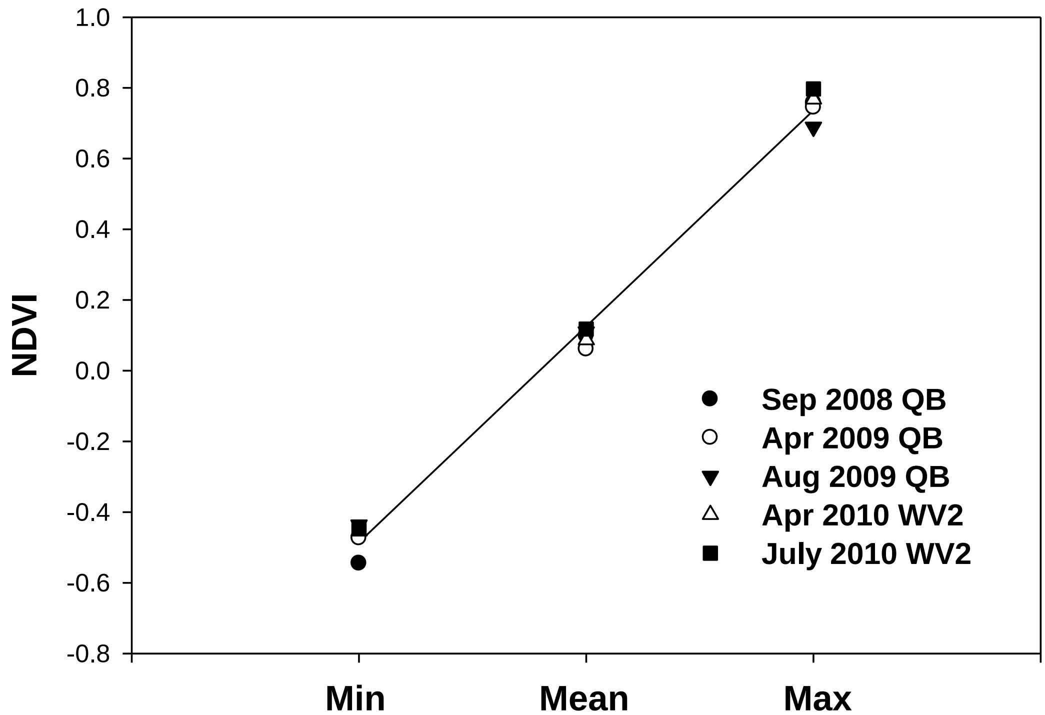

Figure 4. Distribution of NDVI values in each land cover class defined on Quickbird and WorldView 2 images. The August 2009 Quickbird image was converted to an NDVI layer using the Red and NIR bands. Six final classes were identified by comparing NDVI values of land cover classes across images. Stability of classes was tested by sampling additional 60-100 pixels on different images within each class identified visually on images. Box plots show the median, 25% quartiles for each class, and a description of what each class represents.

Figure 5. Footprint of a box of 256 MODIS pixels chosen to represent the dense vegetation area in the center of the Cienega for NDVI time-series analyses.

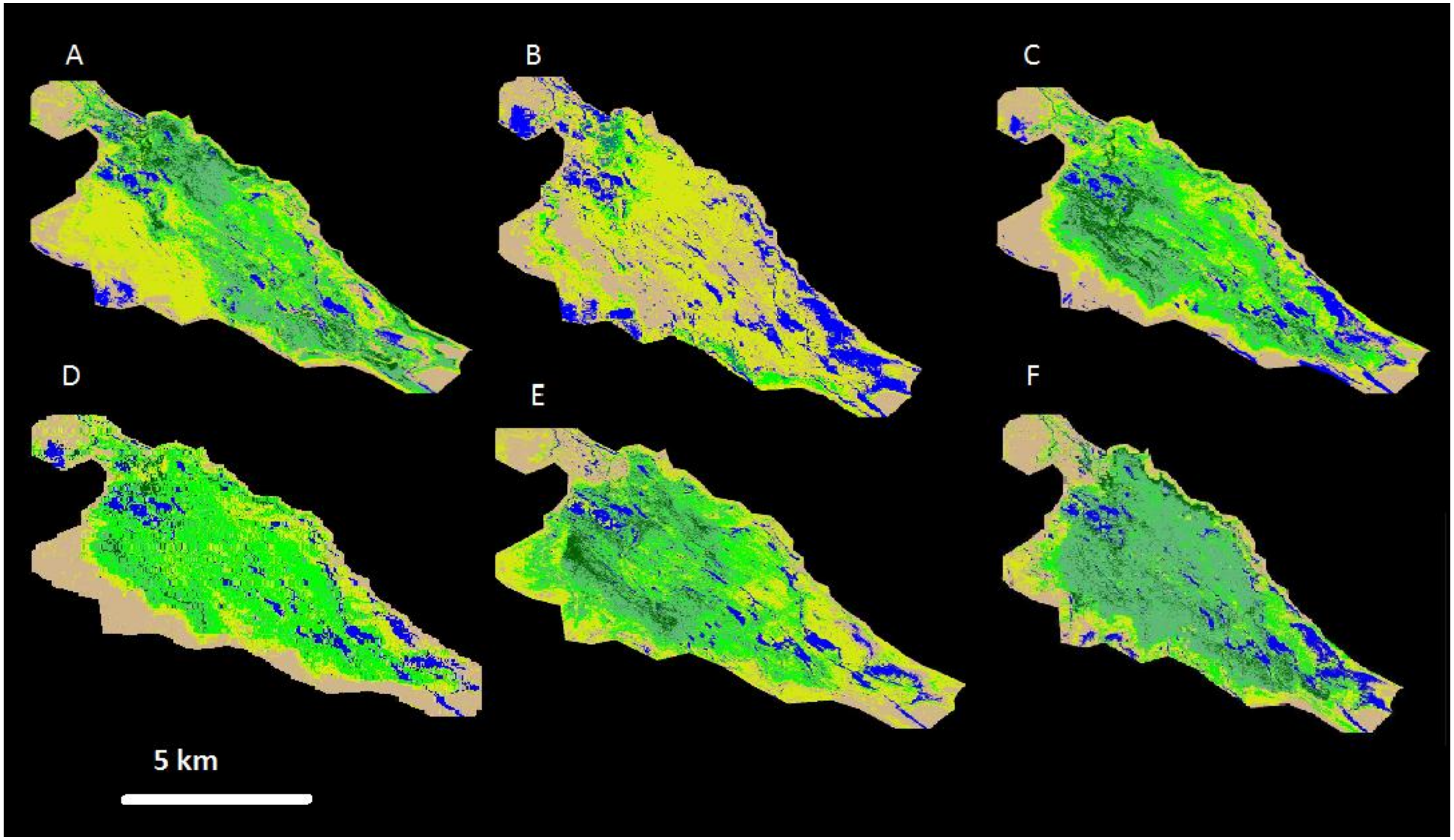

Figure 6. NDVI-based vegetation density maps of Cienega de Santa Clara on different dates. Images were from September 2008 (A); February 2009 (B); August 2009 (C); July 2010 (D); April 2011 (E); and October 2011 (C).

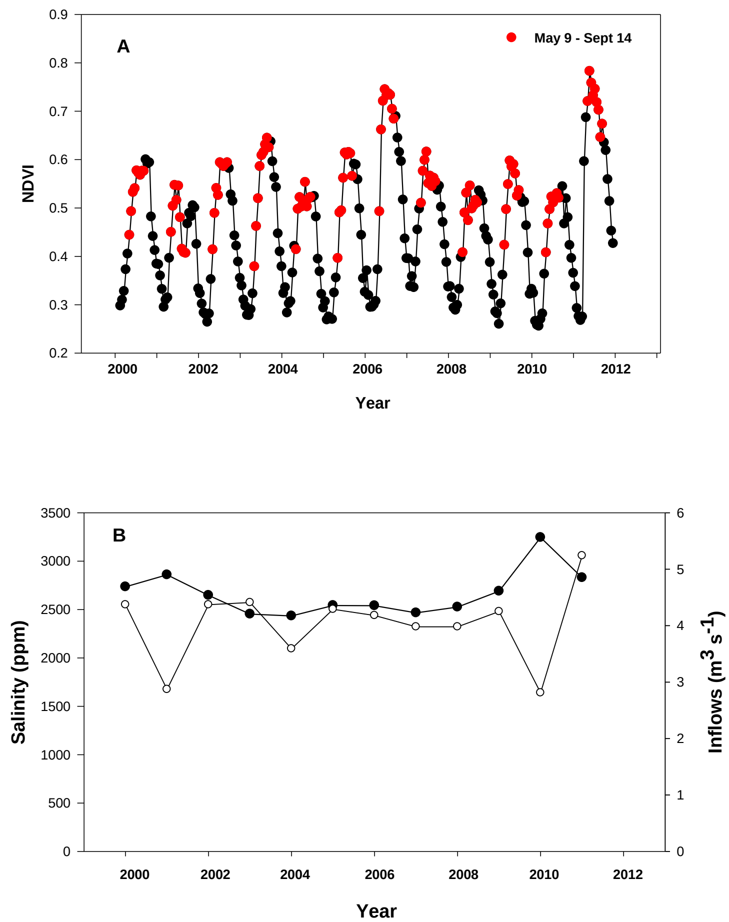

Figure 7. (A) MODIS NDVI values for vegetation in Cienega de Santa Clara at 16-day intervals, 2000 – 2011. Red data points show values for the period May 9 – September 14, which corresponds to the period of reduced flows to the Cienega during the 2010 test run of the Yuma Desalting Plant. (B) Salinity (closed circles) and flow volumes (open circles) in the MODE canal during the same period each year. Note that high NDVI values in 2006 and 2011 occurred following spring fires that removed thatch from *T. domengensis* stands.

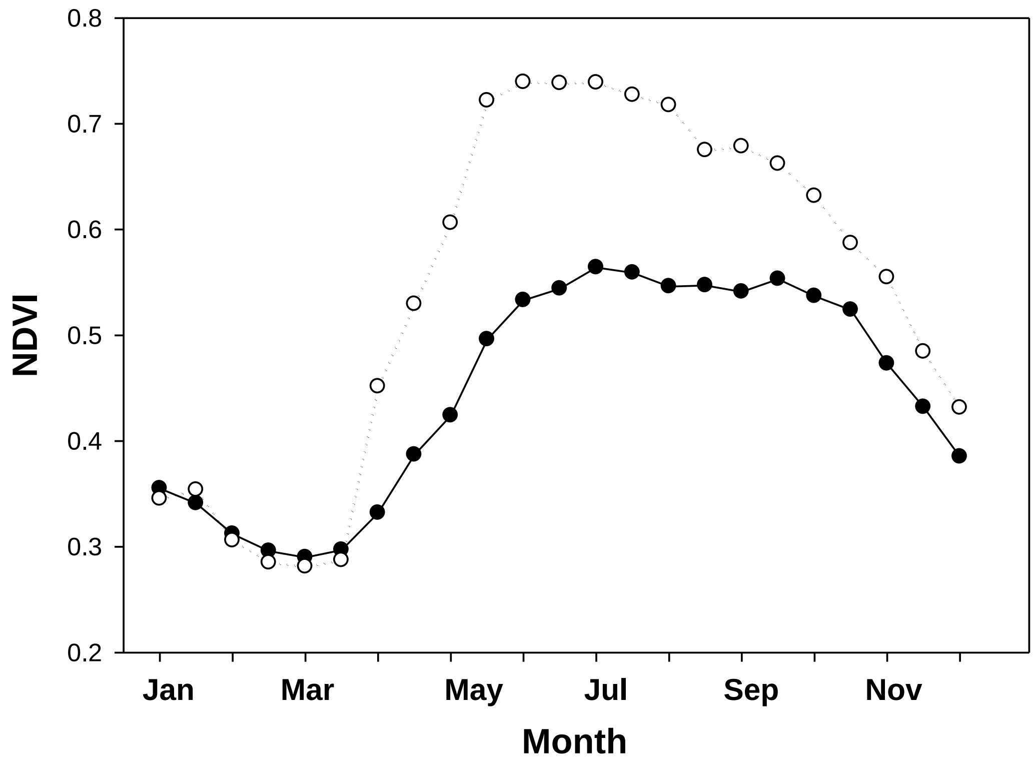

Figure 8. Phenology of *T. domengensis* growth in Cienega de Santa Clara, showing non-fire years (2000-2005, 2007-2010) (closed circles) and fire years (2006, 2011) (open circles) separately.

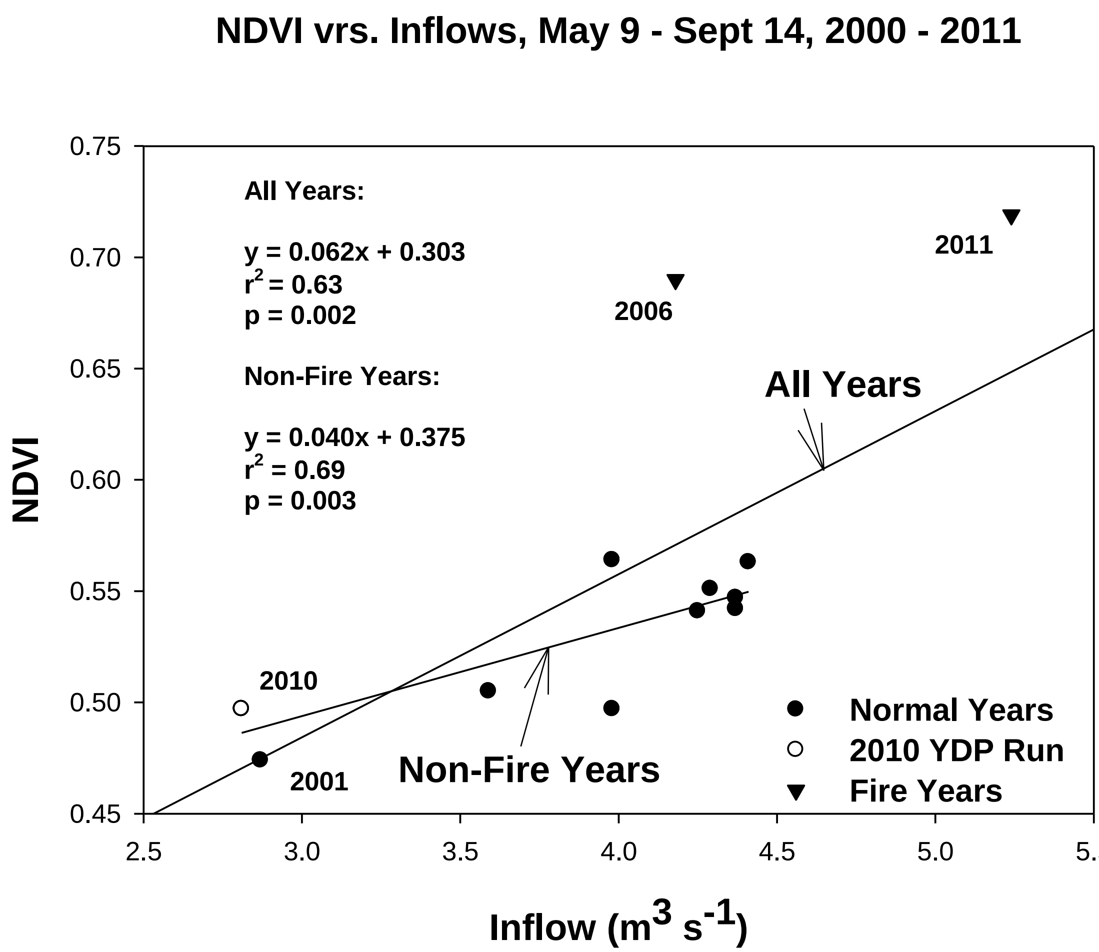

Figure 9. Linear regression analyses of NDVI values in Cienega de Santa Clara during the period May 9 – September 14, 20000 to 2011. Fire years are shown separately from non-fire years, and regression equations for all years and for non-fire years only are show separately.

APPENDIX B

EVAPOTRANSPIRATION AND WATER BALANCE OF AN ANTHROPOGENIC COASTAL DESERT WETLAND; RESPONSES TO FIRE, INFLOWS AND SALINITIES

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Abstract

Evapotranspiration (ET) and other water balance components were measured in Cienega de Santa Clara, an anthropogenic brackish wetland in the delta of the Colorado River in Mexico. The marsh is in the Biosphere Reserve of the Upper Gulf of California and Delta of the Colorado River, and supports a high abundance and diversity of wildlife. Over 95% of its water supply originates as agricultural drain water from the USA, sent for disposal in Mexico. This study was conducted from 2009-2011, before, during and after a trial run of the Yuma Desalting Plant in the USA, which will divert water from the wetland and replace it with brine from the desalting operation. We used a remote sensing algorithm to estimate ET from meteorological data and Enhanced Vegetation Index values from the Moderate Resolution Imaging Spectrometer (MODIS) sensors on the Terra satellite. ET estimates from the MODIS method were compared to results from a mass balance of inflows and outflows over the study period. By both methods, mean annual ET ranged from 3-3.5 mm d^{-1}, or 60-70% of potential ET (ET_o). Water entered at a salinity of 2.6 g L-1 TDS and exited at 6-7 g L^{-1} TDS, which corresponds to the salt tolerance limit of *T. domingensis* as determined in other studies. Over an annual cycle, 54% of inflows supported ET while the rest exited the marsh as outflows; however, in winter when ET was low, up to 90% of the inflows exited the marsh. An analysis of ET over the years 2000 to 2011 showed that annual ET was proportional to the volume of inflows, but was also markedly stimulated by fires. Spring fires in 2006 and 2011 burned

off accumulated thatch, resulting in vigorous growth of new leaves and a 30% increase in annual ET compared to non-fire years. Following fires, peak summer ET rates were equal to ET_o, while in non-fire years peak ET was equal to only one-half to two-thirds of ET_o. Over annual cycles, ET was always lower than ET_o, because *T. domengensis* is dormant in winter and shades the water surface, reducing direct evaporation. Thus, ET of a *Typha* marsh is likely to be less than an open water surface under most conditions.

Keywords: cattail, common reed, MODIS, remote sensing, wetland water budget, salinity, Colorado River delta

B.1. Introduction

The water budget of a wetland determines its extent, ecology, water quality, carbon storage, rates of ground water recharge or discharge and outflows into adjacent ecosystems (Bijoor et al., 2011; Bridgham et al., 2006; Mitsch and Gosslink, 2000). However, constructing accurate water budgets for wetlands can be difficult, and generalizations about wetland hydrology are prone to error (Bullock and Acreman, 2003). Evapotranspiration (ET) is often the largest discharge term in a wetland water budget, and is especially difficult to measure (Drexler et al., 2004). There is a debate about the magnitude of wetland ET in relation to potential ET (ET_o) and the relative importance of plant transpiration and open-water evaporation in contributing to wetland ET (Drexler et al., 2004, 2008; Bijoor et al., 2011). Some studies show that the presence of wetland reed vegetation elevates ET above ET_o from an open water surface, by increasing the evaporative surface area through high leaf area index (LAI) (e.g., Towler et al., 2004), while others report that wetland ET is approximately equal to open-water evaporation or ET_o (e.g., Drexler et al., 2008; Farnsworth et al., 2003; Sun et al., 2010), and yet others report that wetland ET is generally lower than ET_o due to constraints on stomatal conductance of leaves and shading of the water surface by the canopy (e.g., Bijoor et al., 2011; Goulden et al., 2007; Lenters et al., 2011; .

Different types of wetlands differ markedly in their ecohydrology (Bullock and Acreman, 2003; Jolly et al., 2008), explaining much of the discrepancy among studies.

However, part of the controversy is also due to differences in methods to measure wetland ET (reviewed in Drexler et al., 2004). Traditionally, ET of wetlands has been estimated as a residual in water balance equations, when inflows, outflows, precipitation, change in storage and ground water discharge or recharge rates can be estimated. Under ideal conditions the water balance approach can provide an accurate estimate of ET over annual or longer time periods, when seasonal changes in storage become negligible, especially for small wetlands with well-defined inflows and outflows (e.g., Bedford et al., 1999). More recent advances in measuring wetland ET include moisture flux towers (Goulden et al., 2007; Zhou et al., 2010), scintillometery (Lenters et al., 2011), surface renewal methods based on heat fluxes over the canopy (Drexler et al., 2004, 2008), diurnal fluctuations in groundwater levels (Mould et al., 2011), remote sensing (Sun et al., 2010) and micrometeorological models (Drexler et al., 2004). These methods give real-time or near-real-time estimates that can reveal seasonal trends and environmental controls on ET. However, a review of these methods concluded that no one method is suited for all wetlands. Further, each method has a potential error or uncertainty of 20-30%, and often no alternative method for independently validating ET estimates is available (Drexler et al., 2004).

This study used a vegetation-index-based remote sensing method tested against a salt-and-water balance approach to measure ET and to construct a water balance for the Cienega de Santa Clara, a brackish *Typha*/*Phragmites* marsh in the delta of the Colorado River, Mexico (Glenn et al., 1992; Zengel et al., 1995). At approximately 6,500 ha, this is perhaps the largest emergent marsh in the Sonoran Desert. It is supported mainly by flows of agricultural drainage water from the U.S., which discharge into the intertidal zone of the delta. Water has flowed at about 4.0 $m^3 s^{-1}$ at a salinity of 2.8 g L^{-1} TDS since 1976, creating a stable marsh which is 87% vegetated and 13% open water (Mexicano et al., 2012; in preparation). *T. domingensis* is the dominant species, making up over 90% of the plant cover, while *P. australis* makes up 7% of the plant cover and 20 other species grow at lower densities along the edges of the marsh (Glenn et al., 1995; Zengel et al., 1995; Mexicano, 2012; in preparation). It supports numerous species of water birds which use it as a nesting area and as a stopover site during their migration on the Pacific Flyway (Glenn et al., 2001; Gomez-Sapiens et al., 2012; in preparation; Hinojosa-Huerta et al., 2012; in preparation). 80% of the remaining endangered Yuma Clapper Rails nest in the Cienega (Hinojosa-Huerta et al., 2001. 2002). It provides a good case study for conducting a wetland water budget, because inflows are measured, and it is isolated from adjacent ecosystems (Huckelbridge et al., 2010).

The study was prompted by a test run of the Yuma Desalting Plant (YDP), which is expected to reduce inflows and increase the salinity of the water in the Cienega (Gabriel and Kelli, 2010). The test run was conducted in 2010, and an intensive monitoring program was conducted from 2009 to 2011 before, during and after the operation of the YDP to document biological and hydrological responses of the ecosystem to reduced flows and altered salinities (Flessa et al., 2012). The Cienega Monitoring Program conducted ground, aerial and satellite surveillance of the Cienega to document changes in the extent and vigor of the vegetation and to construct a water budget and a predictive model of the vegetation in response to changes in inflows and salinities. Flow gauges monitored the inflows of water to the marsh, and salinity was measured in the inflow water and at numerous recording stations throughout the vegetated area of the marsh (Garcia-Hernandez et al., 2012). This allowed us to check the accuracy of the satellite-derived ET estimates by a mass balance approach, and to construct a complete water balance for the Cienega at monthly time steps during operation of the YDP. Our goal was to develop monitoring protocols to estimate changes in green foliage density and ET in response to inflow volumes and salinities.

B.2. Materials and Methods

B.2.1 ET estimation by MODIS imagery

Vegetation index values from satellite images are valuable tools in evaluating biophysical processes over wide areas and for change-detection over long time spans (Pettorelli et al., 2005; Baldi et al., 2008; Verbesselt et al., 2010; Song et al., 2001). In this study, ET estimates were based on Enhanced Vegetation Index (EVI) values from the Moderate Resolution Imaging Spectrometer (MODIS) sensors on the Terra satellite (Nagler et al., 2005a,b, 2009). MODIS provides near-real-time imagery of most of the earth at daily intervals. Images with 250 m resolution in the Red and NIR bands and 500 m in the Blue band are georectified and radiometrically and atmospherically corrected before being released to end users as vegetation index (VI) and other products (Huete et al., 2002, 2011). MODIS products are valuable in phenological and change-detection studies, which require stable vegetation index values over time (Huete et al., 2011).

The EVI is similar to the more familiar Normalized Difference Vegetation Index (NDVI) in combining Red and NIR bands in a ratio that captures the distinct spectral signature of green vegetation, and allows the landscape to be divided into water, soil and vegetation components (Glenn et al., 2008). EVI also uses the Blue band on MODIS and is less sensitive to soil effects than NDVI, and saturates at higher levels of leaf area index (LAI) than NDVI (Huete et al., 2002, 2011). EVI is calculated as:

$$EVI = G(\rho NIR - \rho Red)/(\rho NIR + C_1 \times \rho Red + C_2 \times \rho Blue + L) \quad (1)$$

where C_1 and C_2 are coefficients designed to correct for aerosol resistance, which uses the blue band to correct for aerosol influences in the red band. The coefficients, C_1 and C_2, are set at -6 and 7.5, respectively, G is a gain factor (set at 2.5), and L for this model is a canopy background adjustment (set to 1.0). Like NDVI, green vegetation has high EVI values (up to 0.85), while bare soil or dormant vegetation has slightly positive values (about 0.10) and water generally has negative values (-0.35 to 0.0). EVI data were obtained from the Oak Ridge National Laboratory DAAC site (ORNL DAAC, 2012). The MOD13Q1 products was used, which is a 16-day composite of 3-5 high quality images (as close to cloud-free as possible) collected during each measurement period. Composite images are constructed on a pixel-by-pixel basis over the collection period to increase the likelihood that a cloud-free image can be constructed (Huete et al., 2011).

In estimating ET via EVI, it was important to avoid pixels that included open water areas, because the negative values for water can artificially lower estimates of vegetation density and ET. Two methods were compared for obtaining spatially-distributed EVI data over the Cienega. In the first, an AOI file was created that encompassed the vegetated footprint of the Cienega (Figure 1) based on a August, 2009 Quickbird Image (Mexicano et al., 2012; in preparation). The AOI file encompassed 5635 ha, a somewhat smaller area than the 6500 ha area defined for the Cienega in Mexicano et al. (2012; in preparation), because that study included a buffer zone around the periphery of the Cienega to capture areas that were only occasionally vegetated, and the present study was focused on the area of permanent marsh. Then a new AOI was prepared that encompassed most of the vegetated area in the Cienega, but excluded the main open water areas. These were identified on high-resolution Quickbird (Digital Globe, Inc., Longmont, CO) images (see Mexicano et al., 2012; in preparation). In the second method, individual pixels from 21 sites distributed throughout the Cienega (Figure 1) were obtained using the Oak Ridge National Laboratory MODIS subset tool (ORNL DAAC, 2012). This tool displays the footprint of a selected pixel on a high-resolution Quickbird image. The Cienega was divided into three sections of approximately equal area and seven pixels were randomly selected in each quadrant. If a selected pixel contained water on inspection of the Quickbird image, a new pixel location was randomly selected.

B.2.2 Determining vegetation cover and open water areas

EVI values were converted to scaled values (EVI*) between bare soil and full vegetation cover by the formula:

$$EVI^* = 1 - (EVI_{max} - EVI)/(EVI_{max} - EVI_{min}) \qquad (2)$$

where EVI_{max} and EVI_{min} were set at 0.542 and 0.091, respectively, based on a large data base of wetland and riparian values from a previous study (Nagler et al., 2005a,b). The advantage of this transformation is that it allows regressions of ET versus EVI* to pass through the origin, where at 0 ET (bare dry soil) EVI* = 0. Open water areas within the Cienega were estimated on seven Quickbird and WorldView 2 images acquired from September 2008 to July 2011 (Mexicano et al., 2012; in preparation). Open water areas were fairly stable, with a mean value of 738 ha (Std. Error = 82 ha), representing 13.1% of the surface area.

B.2.3 Procedure for calculating ET.

ET in this wetland is a combination of transpiration by emergent vegetation (E_{veg}) and evaporation from open water areas (E_{water}). The two parameters were estimated separately and added together to estimate total ET:

$$ET = 0.869E_{veg} + 0.131E_{water} \qquad (3)$$

Both estimates depended on an estimate of potential evapotranspiration (ET_o) as determined from meteorological data. We used the Blaney-Criddle formula for ET_o, which requires mean monthly temperature (T_{mean}) and hours of potential sunlight based on latitude (p, obtained from a table)(Brouwer and Heibloem, 1986):

$$ET_o = p(0.46Tmean+8) \tag{4}$$

The Blaney-Criddle formula for ET_o was used because in previous studies it was more highly correlated with riparian ET than the more complete Penman-Monteith formula (Nagler et al., 2005a,b, 2009). This is because the riparian and marsh vegetation in southwestern U.S. river systems are largely deciduous, becoming dormant in winter, and their ET curve more closely tracks the temperature curve, which dominates the Blaney-Criddle method, rather than the radiation curve, which dominates reference-crop ET_o calculated by the Penman-Monteith formula. E_{water} was assumed to be equal to ET_o (Huckelbridge et al., 2010; Mould et al., 2011). E_{veg} was calculated by an algorithm relating EVI to ET_o, developed for crop and riparian vegetation on the Lower Colorado River (Nagler et al., 2009):

$$E_{veg} = 1.22(EVI^*)ET_o \tag{5}$$

Equation (2) was developed by regressing ET measured on the ground for alfalfa and riparian plants on the Lower Colorado River with EVI* and meteorological data obtained from AZMET stations. It has an error or uncertainty of about 20%, within the range of errors of the ground methods used to measure ET (Glenn et al., 2010). The equation is similar to the simple equation developed by Groeneveld et al. (2007) for riparian and desert phreatophytes in the western US, in which ET is scaled between 0 (bare soil) and 1 (a fully transpiring crop such as alfalfa, with E_{veg} assumed to be equal to ET_o) based on NDVI values from Landsat images. The factor 1.22 in Equation (5) was derived from the regression line of best fit between EVI* and measured ET in the study of Nagler et al. (2009). The validity of using vegetation indices for estimating ET, and underlying assumptions and sources of error inherent in the methods, are discussed in Glenn et al. (2010, 2012). T_{mean} and precipitation data were obtained from the Yuma Valley AZMET station for the period 2000 – 2011 (AZMET, 2011).

B.2.4 ET by a mass balance approach

According to the water balance equation for a wetland, ET can be calculated as (Mitsch and Gosslink, 2000):

$$ET = P + SWI + GWI - SWO - GWO - \Delta S \tag{6}$$

P is precipitation, SWI is surface water inflows, GWI is groundwater inflows, GWO is groundwater outflows and ΔS is change in water storage in the wetland. SWI was the sum of MODE water and local agricultural drainage entering the Cienega from the Riito Canal, which was gaged from 2009-2011 (Garcia-Hernandez et al., 2012). GWI and GWO were assumed to be small and their sum was set at zero. SWO is not measured in the Cienega as there is not a single exit point for water. However, salts are assumed to be conserved in the system because pure water is evaporated to the atmosphere. Hence, under equilibrium conditions SWO can be estimated by the salinity in the outflow water compared to the inflow salinity:

$$SWO = SWI\ (1 - TDS\ In\ /\ TDS\ Out) \qquad (7)$$

(Ayers and Westcott, 1985). Outflow salinity is not measured due to lack of a single exit point, but Inflow Salinity is measured, and Mean Salinity in the Cienega can be calculated from spatially distributed salinity measurements made throughout the water body. Outflow Salinity is related to Mean Salinity and Inflow Salinity by:

$$\text{Outflow Salinity} = 2(\text{Mean Salinity}) - \text{Inflow Salinity} \qquad (8)$$

and SWO can be calculated by Equations (7).

Equation (6) cannot accurately model ET over short time steps, because over short time steps ΔS is unknown. However, over longer time periods, SWI and SWO become large compared to ΔS, increasing the accuracy of the estimate. We computed ET by mass balance over the period January 1, 2009 through June, 2011, during which period accurate salinity data within the Cienega were available. During this period, flows into the Cienega were approximately 5 x 10^8 m^3, 13 times higher than the volume of the Cienega (approximately 3.8 x 10^7 m^3), justifying the assumption that net ΔS over the study period could be ignored in the ET estimation.

B.2.5 Other data sources and statistical analyses

Salinity data was collected monthly from nine stations in 2009 and from 23 stations in 2010 and through June, 2011, by the Cienega Monitoring Team (described in Garcia-Hernandez et al., 2012) (Figure 1). An attempt was made to collect salinity data throughout the marsh, but monitoring points were not randomly selected but were located where access routes were available. Monthly inflow salinity was from Station 8, where the MODE canal enters the Cienega, and monthly mean salinity was calculated as the mean of all reporting stations. Inflow volume and salinity data were from the IBWC gage station at the Southerly International Boundary, covering 2009 and through June, 2011 (data supplied by International Boundary and Water Commission, El Paso, TX).

Multiple linear regression analyses were carried out with Systat software (Systat, Inc., Chicago, IL). The standard error for the ET estimate based on MODIS was calculated from the variance among the EVI point samples in the marsh (n = 21). The standard error for the ET estimate based on the mass balance analysis was calculated from the standard errors of the monthly salinity and flow measurements (n = 58) used in the ET calculation, using the propagation of errors method in Taylor (1997).

B.3. Results

B.3.1 Comparison of methods for determining spatially distributed EVI values

The AOI and pixel-sampling methods were highly correlated ($r^2 = 0.98$) (Figure 2), but the AOI method produced EVI values 7.5% lower than the pixel-sampling method. Because the AOI method was unable to exclude all pixels containing mixtures of open water and vegetation, we chose to use the pixel-sampling method to represent foliage density and ET in the vegetated fraction of the Cienega.

B.3.2 ET in the Cienega, 2000-2011

ET in the Cienega was variable among years, but was markedly stimulated by early spring fires in 2006 and 2011 (Figure 3). These fires burned over 70% of the marsh area and removed accumulated thatch from previous years' growth, allowing more light penetration into the canopy (Conway et al., 2010), and presumably returned nutrients to the water (Liu et al., 2010). Following fires in 2006 and 2010, peak summer ET rates equaled ET_o; however, in non-fire years peak rates were one-half to two-thirds of ET_o. Mean monthly values for ET over fire and non-fire years are in Figure 4, showing that ET increased by 30% following spring fires. Mean annual ET in non-fire years was 1081 mm yr^{-1}, 58.8% of ET_o (1840 mm yr^{-1}), while ET in fire years was 1299 mm yr^{-1}, 70.6% of ET_o. Open water evaporation averaged 241 mm yr^{-1}, 22% of total ET in non-fire years.

Analysis of NDVI values in the Cienega (Mexicano et al., 2012; in preparation) showed that besides fire, summer flow rates in the MODE canal were also a determinant of foliage density. As expected, a similar relationship was found for ET calculated from EVI. A multiple linear regression analysis for the period May 15 to September 15 of each year showed that both flow rates and presence or absence of fire were significant predictors of ET ($P < 0.05$). Inflow salinity did not vary widely from 2000 to 2011 and it was not a significant factor in the multiple regression analysis ($P = 0.635$), and the constant term was also non-significant ($P = 0.592$) so the final equation was passed through the origin and included only flows and fire effects as categorical variables (Table 1). A response surface for the relationship is in Figure 5.

B.3.3 Monthly inflow volumes and salinities and projections of outflow from MODIS ET data

Monthly values of inflow salinity, mean salinity in the Cienega, and calculated outflow salinity are in Figure 6A. Inflow salinity was steady at about 2.2 g L^{-1} in 2009, but increased to a high of 3.6 in July 2010, during operation of the YDP. Mean salinity tended to be variable from month to month in 2009, perhaps due to limited number of recording stations (nine), but it closely tracked inflow salinities in 2010, with 23 stations in operation (see Garcia-Hernandez et al., 2012). During summers calculated outflow salinities were in the range of 6-7 g L^{-1}, which corresponds to the salt tolerance limit of *T. domingensis* for growth and transpiration determined in greenhouse studies (Glenn et al., 1995; Baeza et al., 2012).

Figure 6B shows inflow data for January 2009-June 2011, as well as ET and outflows estimated from MODIS data. Outflows were calculated as inflows (MODE flows plus Riito Drain flows plus precipitation) minus ET, assuming no change in storage within the Cienega, and assuming negligible losses to ground water infiltration. Inflows tended to be variable on a monthly basis, with brief dips in September 2009 and January 2010, and a more prolonged dip in May to August 2010, during flow interruptions associated with operation of the YDP, followed by recovery of flows in September 2010. ET had a sharp peak in September 2009, but the summer peak was truncated in 2010. Outflows were near zero during September 2009, due to high ET rates. The calculated values became negative in August 2010, as ET exceeded inflows, presumably leading to a decrease in the volume of water in the Cienega. Calculated outflows were highest in winter in both 2009 and 2010. Over the whole study period, 54% of inflow water exited the Cienega, but during winter as much as 90% of inflows exited the Cienega due to low ET rates.

B.3.4 Comparison of MODIS ET estimates to a mass balance estimate

Parameters needed to calculate ET by mass balance are in Table 2. Inflows over the 912 day study period averaged 429,408 $m^3\ d^{-1}$, while outflows estimated from the salt balance averaged 233,280 $m^3\ d^{-1}$, so by difference 196,128 $m^3\ d^{-1}$ were lost to ET. Dividing by the area of the Cienega (5635 ha) gives an ET estimate of 3.48 mm $m^{-2}\ d^{-1}$ compared to an estimate of 2.97 mm $m^{-2}\ d^{-1}$ by the MODIS method, 15% lower. The difference between estimates was significant ($P < 0.05$) by unpaired t-test. Both methods are subject to error and uncertainty. Sources of possible error or uncertainty in the MODIS ET estimate are: the ET algorithm; the estimates of vegetated and open water areas in the Cienega; the representativeness of the sampled pixels to the whole Cienega; and, the estimate of ET_o from Yuma temperature data rather than on-site measurements. The main sources of uncertainty in the mass balance estimate are: variability in salinity measurements among stations and uncertainty about how closely the point estimates of salinity predict the true mean salinity in the Cienega; lack of actual outflow data; and, the estimate of the evaporating surface area of the Cienega.

B.4. Discussion

B.4.1 Factors controlling ET in the Cienega

The present results help explain some of the discrepancies among studies of other *Typha* marshes. Under optimal conditions after fires, *T. domengensis* ET was equal to ET_o, but in most years the canopy appeared to be light-limited due to the buildup of thatch from previous years of growth, and peak summer ET was about half to two-thirds of ET_o. In fire years, ET was equal to values reported for restored marshlands in the San Joaquin Valley of California, for which ET was equal to ET_o (Drexler et al., 2008), whereas, in non-fire years ET in the Cienega was similar to relatively low values measured in an unmanaged marsh with accumulated thatch in Irvine, California (Goulden et al., 2004; Bijoor et al., 2011). Annual rates of ET were always lower than ET_o in the Cienega, because plants are dormant in winter but shade the water surface, reducing E_{water}. Hence, the presence of vegetation appeared to reduce water losses compared to open water in this ecosystem, even in years of vigorous growth.

The role of fire in wetland ecology is controversial and appears to be site specific. Flores et al. (2011) studied fire effects in a coastal *Spartina*/*Schoenoplectus* marsh in Maryland, USA, and found that yearly or every-third-year prescribed fires increased overall green plant biomass and stem density of vegetation, but did not necessarily improve the habitat value for waterfowl by reducing the abundance of undesirable species as intended. Urban et al. (1993) studied fire effects on sawgrass and cattails in the Florida Everglades and reported that cattails regreened within one year of burning, but did show an increase in green biomass in post-fire years as noted in this study. On the other hand, Conway et al. (2010) reported that burning markedly increased green shoot growth the following year and improved the habitat quality of *Typha* marshes on the lower Colorado River for marsh birds, by removing thatch and creating new nesting sites. In the Cienega, fire enhances the habitat value of the marsh for Yuma clapper rails (Hinojosa-Huerta et al., 2012; in preparation). Periodic burning also returns nutrients, in particular phosphorous, to the water column (Tian et al., 2010).

Water inflow rate during summer was also a significant determinant of ET (Mexicano, 2012; in preparation) showed a linear decrease in canopy NDVI with decreased summer flow rates over the years 2000 to 2011. From 2009 to 2011, over 95% of inflows were from the USA via the MODE canal, while local flows and precipitation provided less than 5% of inflows. Thus, the Cienega is nearly completely dependent on summer flows in the MODE. However, inspection of Figure 5B shows that in winter water mostly passes through the marsh without supporting ET. Winter outflows pool in the Santa Clara Slough, an extension of the Cienega in the intertidal zone of the delta. This pooled water supports numerous shorebirds in winter (Gomez-Sapiens et al., 2012). In summer the Santa Clara Slough is dry except during extreme high tide events (Nelson et al., 2012; in preparation).

Salinity is the third factor impacting ET in the Cienega. Although it was not significant in the regression analysis (because it did not vary greatly over years), the upper limit for *T. domengensis* growth is about 6 g L^{-1} TDS based on greenhouse experiments (Glenn et al., 1995; Baeza et al., 2012). The present study supported that conclusion by showing that the outflow water was 6-7 g L^{-1} TDS. Over an annual cycle, however, only 54% of the inflows were actually consumed in ET, because winter flows were not completely utilized. The mean salinity in the Cienega is about 3.7 g L^{-1} TDS, suggesting that the vegetation is already salt-stressed to some extent. However, when thatch is removed by fire, the results show that *T. domingensis* stands are still capable of vigorous growth and high ET rates.

B.4.2 Accuracy of ET estimates

Although the 15% different between mass balance and MODIS estimates of ET was statistically significantly ($P < 0.05$), the difference was within the range of 10-30% reported for other studies in which remotely sensed ET estimates are compared to ground measurements (Glenn et al., 2010, 2012). The best agreement occurs when ground measurements have high accuracy. For example, NDVI models of wheat and corn ET agree within 5-10% with ET results obtained with high-precision weighing lysimeters (reviewed in Glenn et al., 2011), whereas remote sensing algorithms tested against moisture flux tower results have a wider variance, proportional to the errors inherent in the flux tower methodology (Glenn et al., 2010, 2012). We conclude that the simple algorithm used in this study has sufficient accuracy to determine effects of water management on ET in the Cienega, and to construct water budgets with sufficient accuracy to guide management decisions on allocations of water needed to sustain the marsh. Given the multiple sources of possible error in both mass balance and MODIS ET estimates, it is not possible to choose between ET estimates. Rather, it can be stated that annual ET in the Cienega averages about 3-3.5 mm/d^{-1} during non-fire years, and is about 30% higher following fires.

B.4.3 Implications for management

The Cienega is part of the Biosphere Reserve of the Upper Gulf of California and

Delta of the Colorado River. Although it contains valuable wildlife habitat, it is not presently managed explicitly to support wildlife. Hunting and fishing are allowed in the marsh. Fires are not prescribed burns but are either started by lightning or by local residents, either deliberately to improve access or accidently. The present results show that fires markedly increase vegetation vigor, while other studies show that at least Yuma clapper rails respond positively to post-fire conditions in *Typha* marshes (Conway et al., 2010). Therefore, a program of prescribed fires might be helpful in maintaining the Cienega as high-quality marsh bird habitat. The timing and frequency of fires required to produce habitat improvement need to be determined. Fortunately, fires so far have occurred in winter and early spring, when *Typha* is dormant and will burn, and fires of this type do not interfere with nesting of marsh birds elsewhere on the Colorado River (Conway et al., 2010).

The study also shows that the Cienega is nearly completely dependent on flows of water from the USA, and that reductions in flows or increases in salinity will have negative effects on the marsh. Based on a previous dry-down in 1993, the effect of reduced flows is to for water to remain in a deeper-water channel running through the middle of the marsh, resulting in drying out of the edges of the marsh away from the channel (Mexicano, 2012). The present results show that flows to the Cienega could probably be reduced during the dormant period of *T. domingensis* (November to March) without damage to the marsh. However, this would potentially negatively impact shorebird habitat in the Santa Clara Slough in winter when most of the visitations take place (Gomez-Sapiens, 2012). Based on the present results, a model of Cienega vegetation and habitat value considering both the Cienega and the Santa Clara Slough is under preparation with inflow volume and salinity as driving variables in monthly time steps.

References

Ayers, R.S., Westcott, D.W., 1985. Water Quality for Agriculture. FAO Irrigation and Drainage Paper No. 29, Food and Agriculture Organization of the United Nations, Rome.

AZMET. The Arizona Meteorological Network; University of Arizona: Tucson, AZ, USA, 2012.
Available online: http://cals.arizona.edu/azmet/ .

Baldi, G., Nosetto, M.D., Aragon, R., Aversa, F., Paruelo, J.M., Jobbagy, E.T., 2008. Long-term satellite NDVI data sets: Evaluating their ability to detect ecosystem functional changes in South America. Sensors 8, 5397-5425.

Bedford, B.L., DeGaetano, A., Siegel, D.I., 1999. Quantification of the water budget and nutrient loading in a small peatland. J. Am. Water Resour. Assoc. 35, 753-769.

Bijoor, N.S., Pataki, D.E., Rocha, A.V., Goulden, M.L., 2011. The application of delta ^{18}O and delta D for understanding water pools and fluxes in a Typha marsh. Plant Cell Environ. 34, 1761-1775.

Bridgham, S.D., Megonigal, J.P., Keller, J.K., Bliss, N.B., Trettin, C., 2006. The carbon balance of North American wetlands. Wetlands 26, 889-916.

Brouwer, C.; Heibloem, M. Irrigation Water Management Training Manual No. 3; FAO: Rome,
Italy, 1986.

Bullock, A., Acreman, M., 2003. The role of wetlands in the hydrological cycle. Hydrol. Earth Sys. Sci. 7, 359-389.

Conway, C.J., Nadeau, C.P., Priest, L., 2010. Fire helps restore natural disturbance regime to benefit rate and endangered marsh birds endemic to the Colorado River. Ecol. Appl. 20, 2024-2035.

Drexler, J.Z., Anderson, F.E., Snyder, R.L., 2008. Evapotranspiration rates and crop coefficients for a restored marsh in the Sacramento-San Joaquin Delta, California, USA. Hydrol. Processes 22, 725-735.

Drexler, J.Z., Snyder, R.L., Spano, D., Paw, K.T.U., 2004. A review of models and micrometeorological methods used to estimate wetland evapotranspiration. Hydrol. Processes 18, 2071-2101.

.

Farnsworth, E.J., Meyerson, L.A., 2003. Comparative ecophysiology of four wetland plant species along a continuum of invasiveness. Wetlands 23, 750-762.

Flores, C., Bounds, D.L., Ruby, D.E., 2011. Does prescribed fire benefit wetland vegetation? Wetlands 31, 35-44.

Gabriel, J., Kelli, L., 2010. The Yuma desalting plant and Cienega de Santa Clara dispute: a case study review of a workgroup process. Water Policy 12, 401-415.

Garcia-Hernandez, J., Glenn, E.P., Artiola, J., 2000. Bioaccumulation of selenium)Se) in the Cienega de Santa Clara wetland, Sonora, Mexico. Ecotox. Environ. Safety 46, 298-304.

Glenn, E.P., Felger, R.S., Burquez, A., Turner, D.S., 1992. Cienega de Santa Clara – Endangered wetland in the Colorado River Delta, Sonora, Mexico. Natur. Res. J. 32, 817-824.

Glenn, E.P., Huete, A.R., Nagler, P.L., Nelson, S.G., 2008. Relationship between remotely-sensed vegetation indices, canopy attributes and plant physiological processes: What vegetation indices can and cannot tell us about the landscape. Sensors 8, 2136-2160.

Glenn, E., Thompson, T.L., Frye, R., Riley, J., Baumgartner, D., 1995. Effects of salinity on growth and evapotranspiration of *Typha domingensis* Pers. Aquatic Bot. 52, 75-91.

Glenn, E.P., Zamora-Arroyo, F., Nagler, P.L., 2001. Ecology and conservation biology of the Colorado River delta, Mexico. J. Arid Environ. 49, 5-15.

Goulden, M.L., Livak, M., Miller, S.D., 2007. Factors that control *Typha* marsh evapotranspiration. Aquatic Bot. 86, 97-106.

Groeneveld, D.P.; Baugh, W.M.; Sanderson, J.S.; Cooper, D.J. Annual groundwater evapotranspiration mapped from single satellite scenes. J. Hydrol. 2007, 344, 146-156.

Hinojosa-Huerta, O., DeStefano, S., Shaw, W.W., 2002. Evaluation of call-response surveys for monitoring breeding Yuma Clapper Rails (*Rallus longirostris yumanensis*). J. Field Ornith. 73, 151-155.

Hinojosa-Huerta, O., DeStefano, S., Shaw, W.W., 2001. Distribution and abundance of the Yuma clapper rail (*Rallus longirostris yumanensis*) in the Colorado River delta, Mexico. J. Arid Environ. 49, 171-182.

Huckelbridge, K.H., Stacey, M.T., Glenn, E.P., Dracup, J.A., 2010. An integrated model for evaluation hydrology, hydrodynamics, salinity and vegetation cover in a coastal desert wetland. Ecol. Eng. 36, 850-861.

Huete, A.; Didan, K.; Miura, T.; Rodriquez, E.; Gao, X.; Ferreira, L. Overview of the radiometric and biophysical performance of the MODIS vegetation indices. Remote Sens. Environ. 2002, 83, 195-213.

Huete, A., Didan, K., van Leeuwen, W., Miura, T., Glenn, E., 2011. MODIS vegetation indices. Land Rem. Sens. Environ. Change 11, 579-602.

Jolly, I.D., McEwan, K.L., Holland, K.L., 2008. A review of groundwater-surface water interactions in arid/semi-arid wetlands and the consequences of salinity for wetland ecology. Ecohydrol. 1, 43-58.

Kerr, J., Ostrovsky, M., 2003. From space to species: ecological applications for remote sensing. Trends in Ecol. Evol. 18, 299-305.

Lenters, J.D., Cutrell, G.J., Istanbullouglu, E., Scott, D.T., Herrman, K.S., Irmak, A., Eisenhauer, D.E.2011. Seasonal energy and water balance of a *Phragmites australis*-dominated wetland in the Republican River basin of south-central Nebraska (USA). J Hydrol. 408, 19-34.

Liu, G.D., Ga, B., Miao, S.L., Li, Y.C., Migliaccio, K.W., Qian, Y., 2010. Phosphorous release from ash and remaining tissues of two wetland species after a prescribed fire. J. Environ. Qual. 39, 1585-1593.

Masoner, J.R., Stannard, D.I.,, 2010. A comparison of methods for estimating open-water evaporation in small wetlands. Wetlands 30, 513-524.

Mitsch, W.J., Gosslink, J.G., 2000. Wetlands, 3rd Edition. Van Norstrand Reinhold, New York.

Mould, D.J., Frahm, E., Salzmann, T., 2011. Evaluating the use of diurnal groundwater fluctuations for estimating evapotranspiration in wetland environments: case studies in England and northeast Germany. Ecohydrol. 3, 294-305.

Nagler, P.L.; Cleverly, J.; Glenn, E.; Pampkin, D.; Huete, A.; Wan, Z.M. Predicting riparian
evapotranspiration from MODIS vegetation indices and meteorological data. Remote

Sens.

Environ. 2005, 94, 17-30.

Nagler, P.L.; Morino, K.; Murray, R.S.; Osterberg, J.; Glenn, E.P. An Empirical Algorithm for Estimating Agricultural and Riparian Evapotranspiration Using MODIS Enhanced Vegetation Index and Ground Measurements of ET. I. Description of Method. Remote Sens. 2009, 1, 1273-1297.

Nagler, P.; Scott, R.; Westenburg, C.; Cleverly, J.; Glenn, E.; Huete, A. Evapotranspiration on

western US rivers estimated using the Enhanced Vegetation Index from MODIS and data from

eddy covariance and Bowen ratio flux towers. Remote Sens. Environ. 2005, 97, 337-351.

Oak Ridge National Laboratory Distributed Active Archive Center (ORNL DAAC). MODIS

Subsetted Land Products, Collection 5; ORNL DAAC: Oak Ridge, TN, USA. Available online:

www.daac.ornl/MODIS/modis.html.

Pettorelli, N., Vik, J., Mysterud, A., Caillard, J., Tucker, C., Stenseth, N., 2005. Using the satellite-derived NDVI to assess ecological responses to environmental change. Trends Ecol. Evol. 20, 503-510.

Song, C., Woodcock, C.E., Seto, K.C., Lenney, M.P., Macomber, S.A., 2001. Classification and change detection using Landsat TM : when and how to correct atmospheric effects? Rem. Sens. Environ. 75, 230-244.

Sun, Z.P., Wei, B., Su, W., Shen, W.M., Wang, C.A., You, D.A., Liu, Z., 2010. Evapotranspiration estimation based on the SEBAL model in the Nansi Lake Wetland of China. Math. Computer Model. 54, 1086-1092.

Taylor, J.R., 1997. An Introduction to Error Analysis, 2nd Edition. University Science Books, Sausalito, CA.

Tian, H.Q., Xu, X.F., Miao, S.L., Sindhoj, E., Beltran, B.J., Pan, Z.J., 2010. Modeling ecosystem responses to prescribed fires in a phosphorous-enriched Everglades wetland: 1. Phosphorus dynamics and cattail recovery. Ecol. Model. 221, 1252-1266.

Towler, B.W., Cahoon, J.E., Stein, O.R., 2004. Evapotranspiration crop coeffients for cattail and bulrush. J. Hyrdol. Eng. 9, 235-239.

Urban, N.H., Davis, S.M., Aumen, N.G., 1993. Flucuations in sawgrass and cattail densities in Everglades-Water-Conservation-Area-2A under varying nutrient, hydrologic and fire regimes. Aquatic. Bot. 46, 203-223.

Verbesselt, J., Hyndman, R., Newnham, G., Culvenor, D., 2010. Detecting trend and seasonal change in satellite time series. Rem. Sens. Environ. 114, 106-115.

Zengel, S.A., Glenn, E.P., 1996. Presence of the endangered desert pupfish (*Cyprinodon macularius*, Cyprinidontidae) in Cienega de Santa Clara, Mexico, following an extensive marsh dry-down. Southwest. Natur. 41, 73-78.

Zengel, S.A., Meretsky, V.J., Glenn, E.P., Felger, R.S., Ortiz, D., 1995. Cienega de Santa Clara, a remnant wetland in the Rio Colorado delta (Mexico) – Vegetation distribution and the effects of water flow reduction. Ecol. Eng. 4, 19-36.

Zhoui, L., Zhou, G.S., Liu, S.H., Sui, X.H., 2010. Seasonal contribution and interannual variation of evapotranspiration over a reed marsh (*Phragmites australis*) in Northeast China from 3-year eddy covariance data. Hydrol. Processes 24, 1039-1047.

Table 1. Multiple regression of summer ET in the Cienega de Santa Clara on presence (1.0) or absence (0.0) of fire and water inflow rates, 2000 to 2011. The constant term was not significant ($P = 0.752$) so for the final equation the regression was passed through the origin. The final regression equation was: ET (mm d-1) = 1.07 (Fire) + 1.349 (Flows), $r^2 = 0.986$, $P = 0.002$.

Effect	Coefficient	Std. Coefficient	P-Value
Fire	1.007	0.929	< 0.001
Flows	1.349	0.438	0.012

Table 2. Mean and standard errors of hydrological parameters used in estimating ET by a water budget equation, compared to ET estimating by MODIS satellite imagery. Standard errors of flows and salinities were calculated from monthly values (n = 58). The standard error of ET by Mass Balance was calculated as the propagation error based on the standard errors of the flows and salinities by which it was calculated. The standard error of ET by MODIS was calculated from the variance among the individual EVI sample points (n = 21) in the Cienega. ET means by Mass Balance and MODIS methods were significantly different at $P < 0.05$ by unpaired t-test.

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 In (g L^{-1}):	2.62 (0.07)
TDS Mean Cienega (g L^{-1})	3.73 (0.08)
TDS Out (g L^{-1})	4.83 (0.15)
Flows Out (m^3 s^{-1})	2.27 (0.07)
ET by Mass Balance (mm d^{-1})	3.48 (0.14)
ET by MODIS (mm d^{-1})	2.97 (0.16)

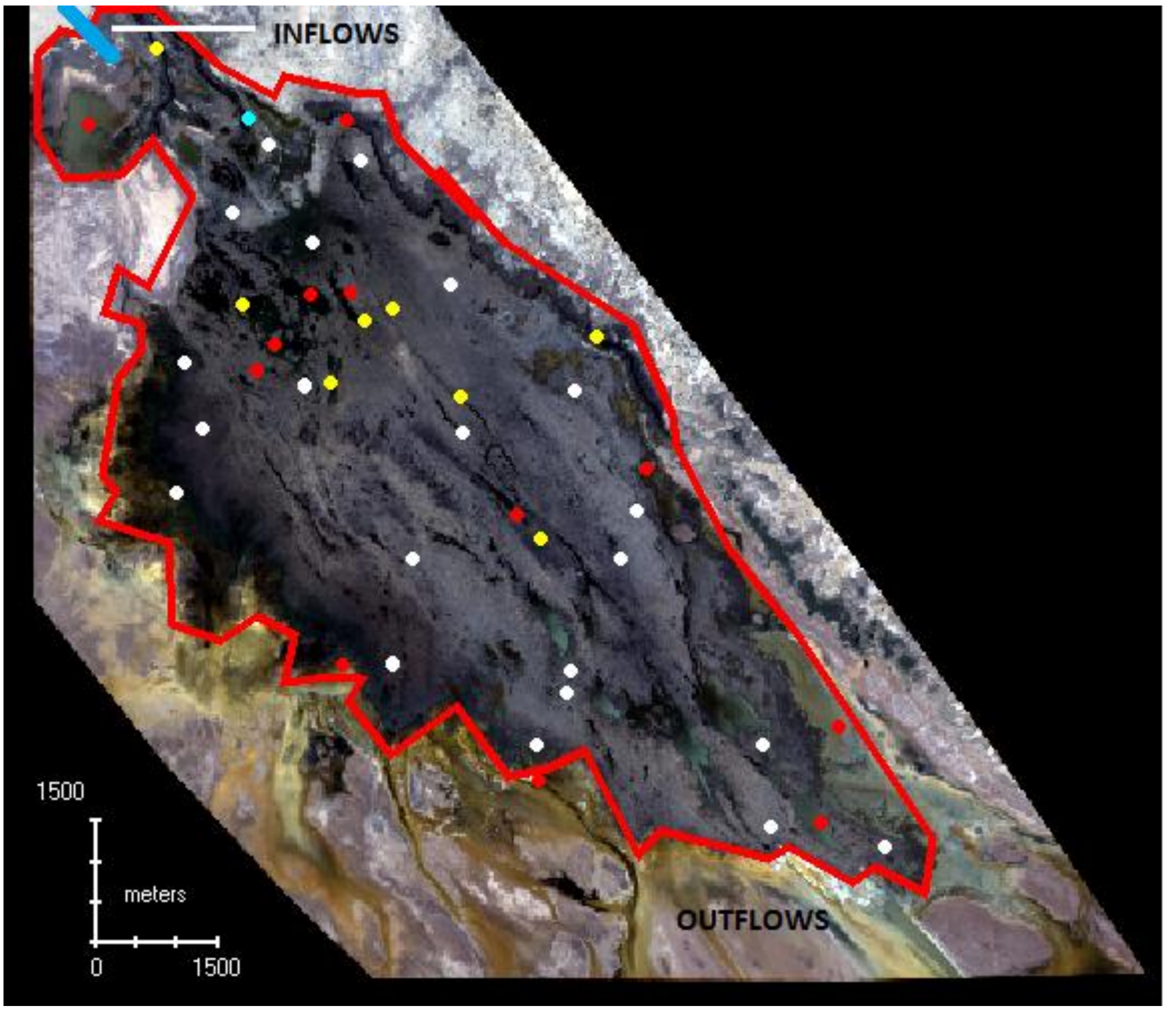

Figure 1. Footprint of Cienega de Santa Clara and locations from which MODIS EVI data (white circles) and salinity data (other circles) were acquired. The blue circle indicates the salinity station at the entry point for water from the MODE canal; yellow circles are additional salinity stations monitored in 2009, and red circles are stations added in 2010.

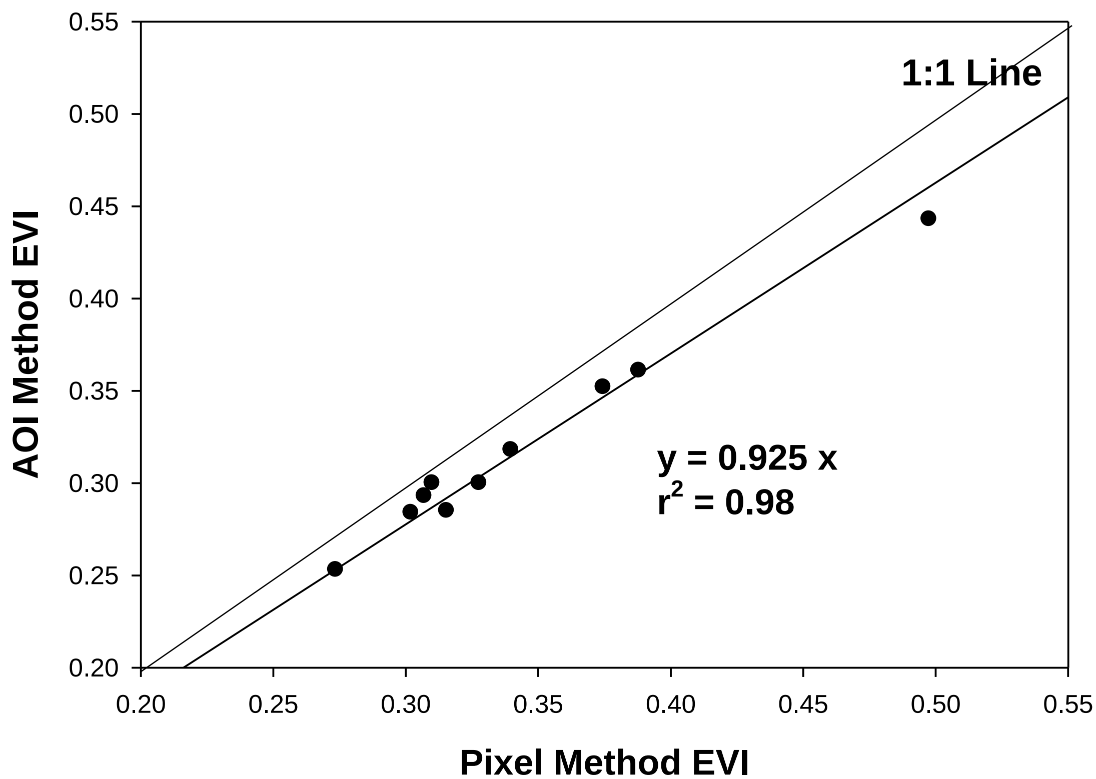

Figure 2. Comparison of two methods to determine mean MODIS EVI over the vegetated portion of the Cienega. The Area of Interest (AOI) method used a shape file that included vegetation but excluded major open water areas; the pixel method sampled 20 individual pixels in which standing water was not present based on inspection of high-resolution Quickbird imagery.

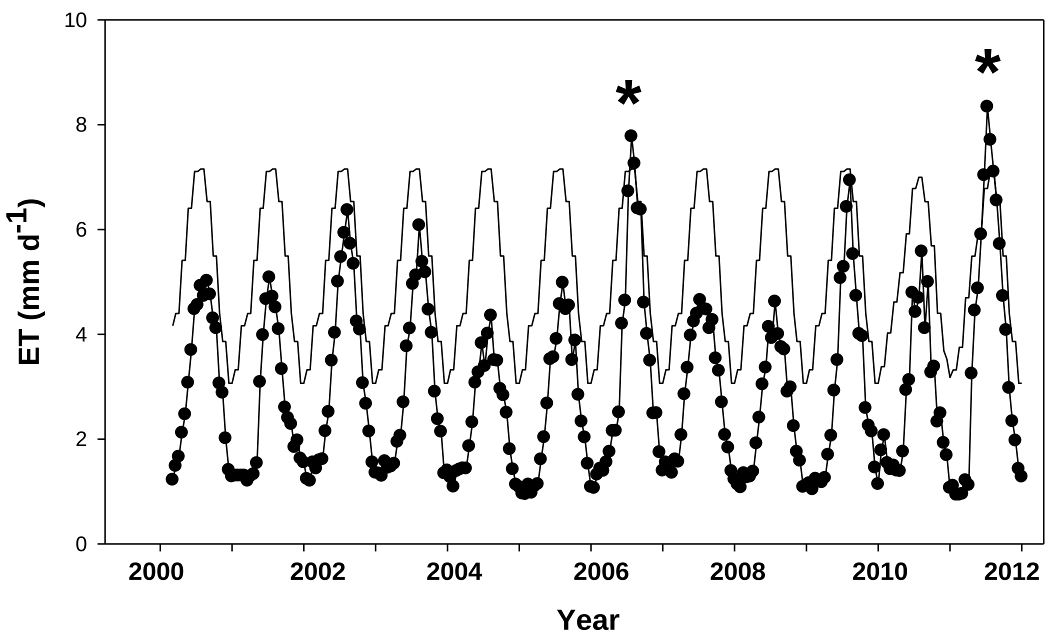

Figure 3. ET values in Cienega de Santa Clara, 2000-2011 based on MODIS EVI. Asterisks rows show when major fires occurred. Solid line without symbols shows potential ET.

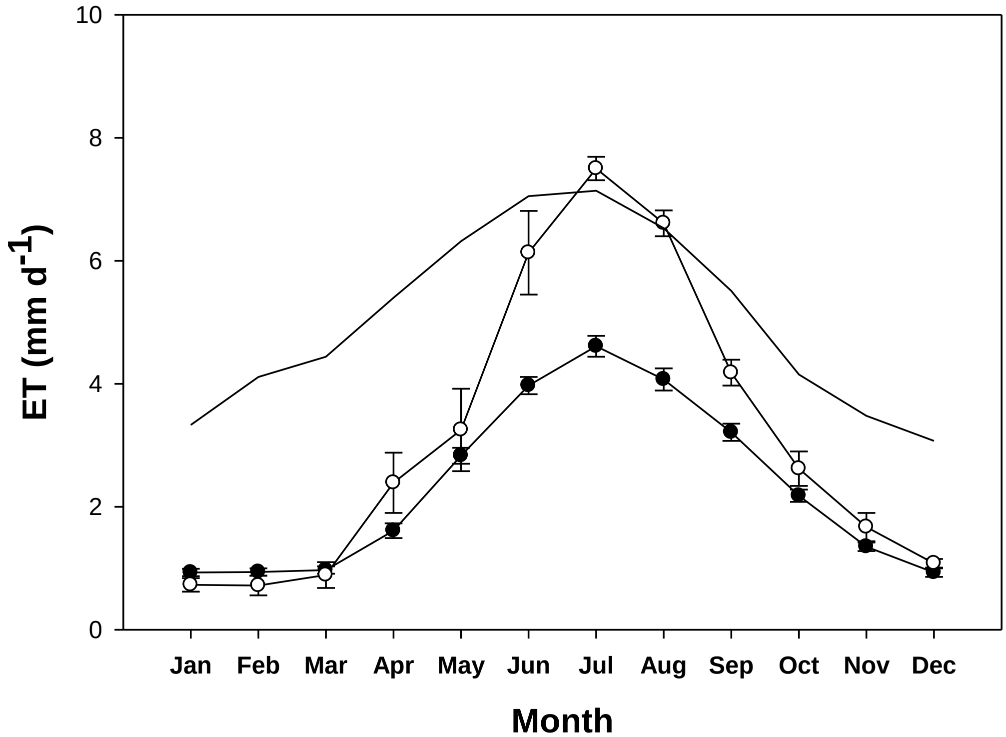

Figure 4. Mean monthly ET in the vegetated portion of the Cienega, 2000 - 2011, with fire years (2006 and 2011) (open circles) shown separately from non-fire years (closed circles). Error bars are standard errors of means. Solid line shows potential ET.

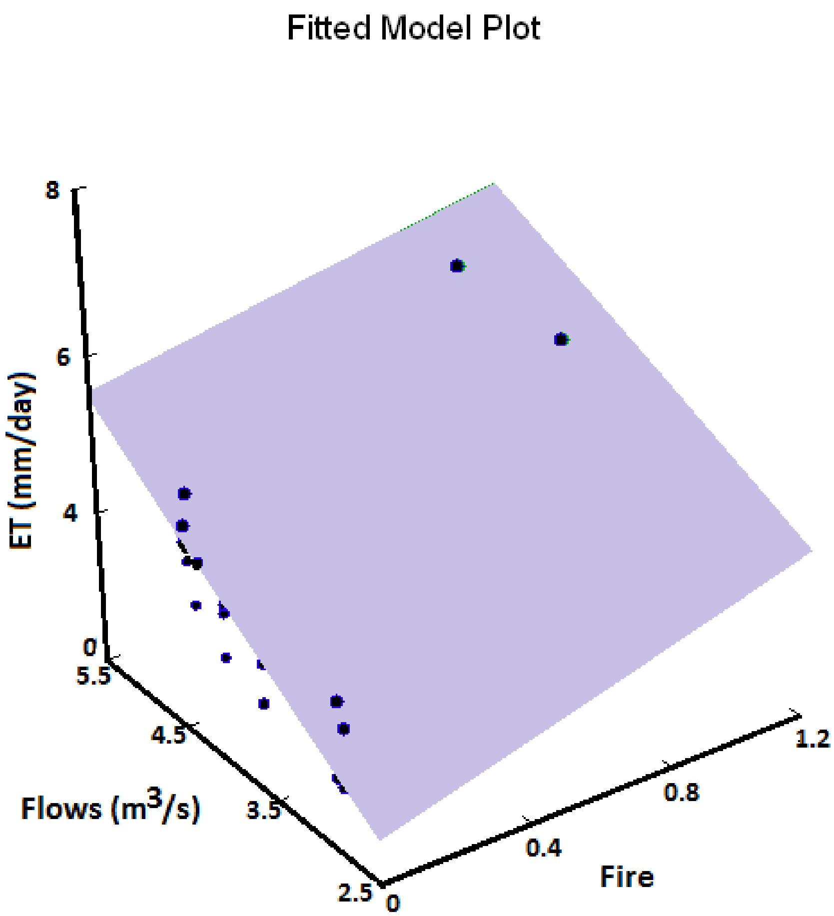

Figure 5. Response surface of ET as affected by water inflows and presence or absence of fire based on a multiple linear regression analysis.

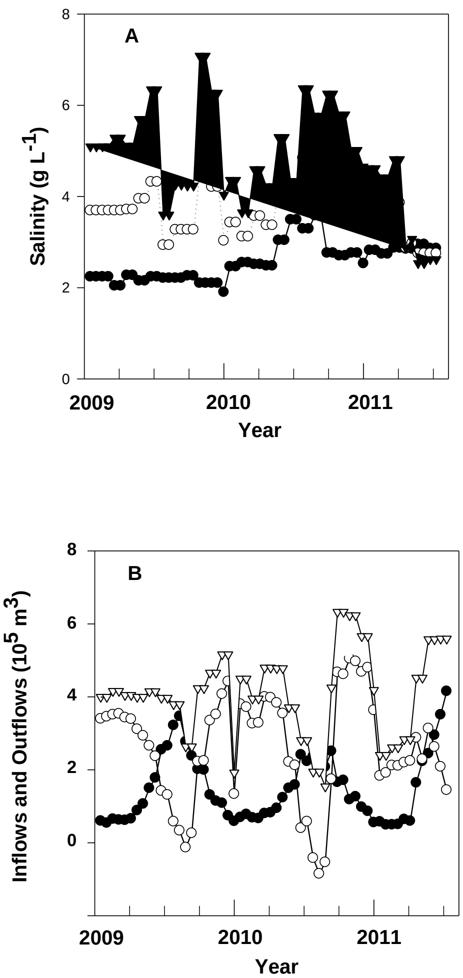

Figure 6. (A) Measured inflow salinity (closed circles), mean salinity (open circles) and outflow salinity (closed triangles) of water in Cienega, January 2009 through June, 2011. (B) Measured inflows (open triangles), ET by MODIS EVI (closed circles) and calculated outflows for the same period.

APPENDIX C

LONG-TERM SUSTAINABILITY OF CIENEGA DE SANTA CLARA, AN ANTHROPOGENIC WETLAND CREATED BY DISPOSAL OF AGRICULTURAL DRAIN WATER IN THE DELTA OF THE COLORADO RIVER, MEXICO

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Abstract

The Ciénega of Santa Clara is a valuable coastal wetland sustained almost entirely by discharge of brackish agricultural drain water from the U.S. and Mexico. In other locations, agricultural drain water has been problematic in supporting wetlands due to problems of salinity buildup, toxic substances and undesirable plant succession processes. We studied the development of the Cienega de Santa Clara from its creation in 1978 to the present to determine if it is on a sustainable trajectory in terms of vegetation, hydrology and habitat value. We used Landsat NDVI imagery from 1978-2011 to determine the area and intensity of vegetation and to estimate evapotranspiration (ET) to construct a water balance. Remote sensing data was supplemented with hydrological data, site surveys and literature citations. Flows into the marsh have been stable both month-to-month and year-to-year, with a mean annual value of 4.74 $m^3 s^{-1}$ (SD = 1.03). Salinity has been stable with a mean value of 2.09 g L^{-1} (SD = 0.13). The vegetated area increased from 1978 to 1995 and has been constant at about 4200 ha since then. The dominant vegetation type is *Typha domingensis* (southern cattail), and peak summer NDVI since 1995 has been stable at 0.379 (SD = 0.016), about half of $NDVI_{max}$. About 30% of the inflow water is consumed in ET, with the remainder exiting the Cienega as outflow water, mainly during winter months when *T. domingensis* is dormant. Periodic monitoring of selenium, trace metals and pesticide residues in water, sediments and biota have not detected levels of concern, and nesting populations of the endangered Yuma clapper rails (*Rallus longirostris yumanensis*) have remained large since they were first

monitored in 1989. The stability of the Cienega is attributed to high flow rates that flush out excess salts and contaminants, and to tidal flushing of outflows to the sea at the southern end of the Cienega.

Keywords: oligohaline marsh; *Typha*; *Phragmites*; constructed wetland; agriculture wastewater

C.1. Introduction

C.1.1 Use of agricultural wastewater to support created wetlands

Created wetlands can increase the amount of aquatic habitat available for wildlife and provide replacement habitat when natural wetlands are lost. Attractive water sources for artificial wetlands are agricultural return flows, which are generated in very large quantities in arid-zone irrigation districts around the world (Gregoire et al., 2009). Often this water is discharged into rivers, reservoirs or the ocean; prior use of the water to support wetlands not only creates wildlife habitat but can reduce pesticide and nutrient levels entering receiving water bodies (Gregiore et al., 2009).

However, agricultural return flows are saline and can contain a wide variety of potentially toxic chemicals (Lemly et al., 1993; Lemly, 1994). Some large-scale failures have occurred when agricultural wastewater has been used to create aquatic habitats. For example, the Kesterson National Wildlife Refuge, created by disposal of drain water from the San Joaquin Valley irrigation districts, produced wildlife deformities due to excess selenium in the inflow water, and the wetland was ultimately capped and closed (Wu 2004). Subsequent investigations showed that selenium hazards were present in over half of western U.S. wildlife refuges receiving agricultural drain water, and that elevated salinities and other toxicity problems were also present in these wetlands (Lemly, 1994).

Other created aquatic habitats have problems of sustainability over time; for example, the Salton Sea, created by disposal of drain water from the Imperial Valley

Irrigation District, provides important avian habitat but has become hyper saline and is experiencing an ecosystem turnover that threatens avian populations (Cohen et al., 1999). Evaporation ponds sometimes provide good wildlife habitat when they are new but they too eventually become hyper saline and can become "ecological traps" (Battin, 2004) that attract wildlife yet are either toxic or do not provide essential habitat requirements (Tanner et al., 1999). In reviewing the effects of irrigation drainage on wetlands, Lemly (1994) advised caution in using these water supplies, noting the negative effects of salinity and elemental toxicity that have occurred throughout the western U.S.

Even well-designed created wetlands that are hydraulically sustainable can experience plant species successions, such that desired species for maintaining wildlife are replaced by less desirable, invasive species over time (Garde et al., 2004; Lin et al., 2010; Kaplan, 2012). A review of the performance of mitigation wetlands that were designed to provide specific habitat types for wildlife showed that they frequently do not follow the intended trajectory of vegetation development (Zedler and Callaway, 1999). An alternative approach is to follow the principle of “self-design”, allowing a natural succession of plant species to develop in a constructed wetland (Mitsch, 1992; Mitsch and Wilson, 1996; Mitsch et al., 2005). However, the final ecosystem might not fulfill the original goals set for the restoration project (Moreno-Mateos et al., 2010).

C.1.2 Development of Cienega de Santa Clara in Mexico

This study examines the long-term stability of Cienega de Santa Clara, an anthropogenic marsh in the delta of the Colorado River in Mexico (Glenn et al., 1992; Flessa et al., 2012; in preparation). The Cienega was formed on the principle of self-design. It was the inadvertent creation of disposal of agricultural drain water from the Wellton-Mohawk Irrigation District in the U.S, which is discharged onto the mudflats of the Santa Clara Slough in the intertidal zone of the Colorado River (Figure 1). Since 1978 water has entered the Cienega in the Main Outlet Drain Extension (MODE) canal at a rate of approximately 4-5 $m^3 s^{-1}$ at a salinity of 2-3 $g L^{-1}$ TDS. The discharge point was originally a small (50 ha) marsh supported by local agricultural drain water from the San Luis Irrigation District. It has developed into a 6,500 ha marsh dominated by *Typha domengensis* (southern cattail), with smaller patches of *Phragmites australis*, *Schoenoplectus americanus* and 21 other hydrophytes and with 15% open water lagoons within the marsh (Mexicano et al., 2012; in preparation). It is a major nesting and stopover station for water birds on the Pacific Flyway (Hinojosa-Huerta et al., 2008), and supports the largest remaining breeding population of the endangered Yuma clapper rail on the Lower Colorado River (Hinojosa-Huerta et al., 2001. 2002).

The only regular maintenance activity is periodic dredging to keep the entry point for water from being block by silt. Although the source water from agricultural wells in the Wellton-Mohawk valley is relatively free of silt, the MODE canal passes beside the Gran Desierto del Altar on its way to the Cienega, and this unstabilized dune field adds sand to the canal, which is deposited at the mouth of the Cienega. So far the discharge point has been extended 500 m from the original end of the canal into the marsh area by dredging (authors' personal observations). The Cienega also experiences fires that burn over most of the vegetation at irregular intervals (Yamillet-Carrillo et al., 2012; in preparation). Fires occur in winter or spring when the dominant vegetation is dormant and flammable. Fires are either deliberately set by local residents to improve access, or by lightning strikes. Other than this, the Cienega has been left to develop without intervention.

C.1.3 Goals and objectives of this study

A common problem in evaluating the performance of created wetlands is lack of long-term monitoring. In many cases after construction and an initial short period of evaluation, created wetlands are not subjected to further monitoring, yet plant succession and toxicity processes can take many years to play out. In the present study, we evaluated the stability over time of Cienega de Santa over its 34 year lifetime. We combined satellite imagery with flow volume and salinity data to compile an annual record of record of vegetation composition, density, evapotranspiration (ET) and marsh area from the creation of the Cienega in 1978 to 2011. The overall goal was to track the long term development of the Cienega and to determine if it has reached a sustainable equilibrium state in terms of size, vegetation, habitat quality and water balance parameters. The factors contributing to the stability of the marsh are discussed in comparison to other created aquatic ecosystems that receive agricultural return flows in arid environments. Implications for future management of the Cienega are also discussed.

C.2. Materials and Methods

C.2.1 Landsat satellite imagery

Landsat represents the world's longest continuously acquired collection of space-based moderate-resolution land remote sensing data. We acquired a July or August image for each year from 1978 to 2011 (Table 1). 1978-1983 images were Landsat 2, 3 or 4; subsequent images were Landsat 5, except for 1990 when a Landsat 4 was used. Images were acquired from the USGS Earth Explorer website (http://earthexplorer.usgs.gov/). Level 1T images for cloud-free scenes with a quality score of 9 (out of 10) were selected for analysis; these images are systematically corrected for radiometric and geometric accuracy and no atmospheric correction DN values were attempted.

Green vegetation density was quantified by the Normalized Difference Vegetation Index (NDVI) (Pettorelli et al., 2005):

$$\mathrm{NDVI} = (\mathrm{NIR} - \mathrm{Red}) / (\mathrm{NIR} + \mathrm{Red}) \qquad (1)$$

NDVI reduces the image to a single layer with NDVI values from -1.0 to +1.0, with water having strongly negative values, soils slightly negative to slightly positive, and vegetation having positive values.

For each image, an area of interest file was created, encompassing the flood area of the Cienega. NDVI ranges for soil, water and vegetation were determined based on training sites determined on high-resolution Quickbird images (Mexicano et al., 2012) and by sampling pixels of each cover class on each image. Differences among images were slight, and pixels in the AOI file were classified as vegetated if the NDVI value was 0.10 or greater.

C.2.2 Evapotranspiration (ET) estimation by Landsat imagery

Annual estimates of ET from the vegetated portion of the Cienega were determined from Landsat images using methods described in Baugh and Groeneveld (2006), Groeneveld and Baugh (2007) and Groeneveld et al. (2007). NDVI values were scaled (NDVI*) between bare soil ($NDVI_{Soil}$) and maximum vegetation response ($NDVI_{Max}$) on each image using the relationship:

$$NDVI^* = 1 - (NDVI_{Max} - NDVI)/(NDVI_{Max} - NDVI_{Soil}) \quad (2)$$

where $NDVI_{Max}$ represents fully transpiring vegetation and $NDVI_{Soil}$ is the NDVI of bare soil where ET is assumed to be zero. NDVImax was selected from histograms of Cienega NDVI values in 1994, 2006 and 2011 when vegetation was at maximal greenness due to fires that removed that during the dormant season ($NDVI_{Max}$ = 0.720

over these years). $NDVI_{Soil}$ was set at 0.078, the mean value of bare soil determined on images.

Following Groeneveld et al. (2007), ET was calculated as:

$$ET = NDVI*(ET_o) \tag{3}$$

where ET_o is potential ET from a fully transpiring plant canopy determined from meteorological data. While this method of estimating ET is only an approximation, Groeneveld et al. (2007) reported an r^2 of 0.94 for ET determined from single summer Landsat images and annual ET measured at 15 moisture flux tower sites set in western U.S. plant communities.

ET_o was calculated by the Blaney-Criddle formula (Brouwer and Heibloem, 1986).

$$ET_o = p(0.46T_{mean} + 8) \tag{4}$$

where p is day light hours determined from a table by month and latitude and T_{mean} is mean monthly air temperature, obtained from the Yuma, Arizona AZMET station (AZMET, 2012). Evaporation from the open water portion of the marsh was assumed to be equal to ET_o. ET estimates for 2009-2011 were validated by comparison to ET determined by a salt and water balance approach for those years (Glenn et al., 2012).

C.2.3 Other data sources

Daily flow rates of MODE water were determined from the International Boundary and Water Commission (IBWC) gaging station at the Southerly International Boundary (http://www.ibwc.state.gov/wad/DDQWMSIB.HTM). Monthly salinity values of MODE water were obtained on request from the IBWC U..S. Section (El Paso, TX).

C.3. Results

C.3.1 Time course of marsh development

Figures 2 and 3 illustrate the time course of marsh development from 1975 to 2011 at approximately five year intervals, based on Landsat imagery. In 1975 the MODE canal had not yet been constructed and the future discharge point was a mudflat at the north end of the Santa Clara Slough in the intertidal zone of the Colorado River (Figure 2A). The point at which the Santa Anna Fault line enters the Gulf of California is visible as a low area partially filled with local agricultural drain water. Also visible is a fringe of natural vegetation supported by artesian springs along the escarpment between the tidal zone and the Gran Desierto dunes on the eastern edge of the future Cienega. By 1980 discharge of MODE water had created a large brackish pool of water in the northern part of the Santa Clara Slough, with marsh vegetation starting to develop at the entry point of the MODE canal and in high spots within the flooded area (Figure 2B). Marsh vegetation occupied more area than open water by 1985 (Figure 2C) and continued to expand through 1995 (Figures 2D, 3A), by which time the final vegetated footprint of the Cienega was established.

A series of permanent, open-water lagoons also developed at apparent deeper-water locations in the Cienega (compare locations of open water lagoons in Figure 3A with initial pattern of flooding, Figure 2B). The overall area of the Cienega and the vegetation footprint were stable between 1995 and 2011 (Figures 3A-D). The 2011 image (Figure 3D) shows the effect that a spring fire had on the vegetation; the fire cleared out accumulated thatch and resulted in a flush of new vegetative growth over much of the marsh by July, 2011 (see also *3.3*).

The main plant species in the Cienega is *Typha domengensis* (southern cattail) (dominant and subdominant species are in Table 2) (Zengel et al., 1995). However, colonies of *Phragmites australis* (common reed) have developed near the entry point for water, growing in shallower areas created by silt deposited by inflow water (Figure 4). These patches of *P. australis* have been stable in size and location from 1995 to 2011.

C.3.2 Stability of vegetated area, vegetation vigor, ET and inflow and outflow volumes and salinities

The vegetated area of the marsh and vegetation vigor were determined by NDVI values on Landsat imagery. The vegetated area of marsh increased up to 1995, when it stabilized at about 4000 ha (Figure 4A). Flows in the MODE canal decreased from an initial value of about 7 $m^3 s^{-1}$ in 1978 to 4-5 $m^3 s^{-1}$ by 1995-1997, and remained in that range through 2010 (Figure 4B). A notable reduction in flows took place in 1993 due to floods on the Gila River in the Wellton-Mohawk Irrigation District, which required the canal to be closed for repairs for an extended period. Salinity decreased slightly from 3.5 g L^{-1} in 1978 to 2-3 g L^{-1} from 1995 to 2011, with dips recorded in 1983, 1993 and 1997 when flood waters were conveyed to the Cienega in the MODE canal. Mean NDVI values of the vegetated portion of the marsh rose from 1978 to 1984 but have been remarkably stable since then (Figure 4C). Flows have been relatively stable on a monthly basis, ranging from a low of 3.5 $m^3 s^{-1}$ in August to 5.2 $m^3 s^{-1}$ in November (Figure 5).

ET was estimated from summer NDVI values by assuming a fixed relationship between NDVI and ET_o. Although only an approximation of actual ET, these estimates allow a calculation of the annual water balance in the Cienega. ET losses by vegetation in the Cienega have been steady at about 1000 mm yr^{-1} since the 1980s, about half of ET_o (Figure 6A). Total ET was 23% higher when open water evaporation is added in. Landsat total ET estimates for the period 2009-2011 were 2000-2011 1,263 mm yr^{-1} (SE = 34), very close to the value of 2,370 mm yr^{-1} estimated by a salt and water mass balance approach (Glenn et al., 2012). However, both methods are subject to errors on the order of 20% (Glenn et al., 2012).

Total vegetation water use was calculated by multiplying the annual vegetation ET rate by the area of vegetation. Vegetation water use increased up to 1995 as the marsh grew (Figure 6B), but since then has consumed about 30% of annual inflows (Figure 6C), with the rest lost as open water evaporation (7%) within the marsh and outflows into the Santa Clara Slough to the south (73%).

C.3.3 Effects of salinity and fire on NDVI

Mean annual NDVI was significantly correlated with salinity ($P < 0.05$), although the correlation coefficient was low ($r = 0.37$). The correlation was mainly due to the reduction in salinity and increase in NDVI during the period 1978-1995. At least three major winters or spring fires have been documented in the Cienega, with over 70% of all vegetation burned over in 1994, 2006 and 2011. Histograms of NDVI values for summer Landsat scenes in the year before a fire and in the year of the fire shows a marked upward shift in peak NDVI (Figure 7). Fires burned out accumulated thatch, allowing more light to penetrate into the canopy to stimulate new shoot growth from rhizomes.

C.4. Discussion

C.4.1 Stability of inflows and vegetation over time

Although there have been several brief interruptions in water delivery, inflow volumes and salinities have been remarkably stable over the 34 year life of the Cienega. The salinity is beyond the optimal salinity for growth of *T. domingensis* (Glenn et al., 1995; Baeza et al., 2012) the dominant species in the marsh. However, 2-3 g L^{-1} TDS is sufficiently low to maintain vigorous stands of this species in the Cienega as also reported for California coastal wetlands (Beare and Zedler, 1987). Only 30% of the inflow water is actually consumed in transpiration by the vegetation. Most of the outflow occurs during the dormant period for *T. domingensis* (November to March), due to low ET_o, and shading of the water surface by the dormant vegetation (Glenn et al., 2012). Outflows fall to near-zero during summer when ET is at its maximum. Stable open water lagoons support near-monocultures of the submerged aquatic plant, *Najas marinas* (Zengel et al.,1995), which is considered an excellent food source for waterfowl (Tarver et al., 1986). As a consequence of the steady inflows, the Cienega is flushed about five times per year, and has avoided a gradual increase in salinity as has occurred in other wetlands that tend to receive flushes of agricultural drain water on a seasonal basis, followed by dry periods (Lemly, 1994). During the period 2009-2011, mean salinity in the marsh was 3.73 g L^{-1} TDS, well within the tolerance range of *T. domingensis* for vigorous growth and reproduction (Beare and Zedler, 1987; Glenn et al., 1995, Baeza et al., 2012; in preparation).

C.4.2 Selenium, pesticides and other toxicity issues

Although not the topic of this study, selenium, trace metals and pesticides have been periodically monitored in water, sediments and biota in the Cienega (Garcia-Hernandez et al.. 2000, 2001, 2006). Pesticide levels are low, as is typical of subsurface drain water from agricultural areas after the water passes through the soil profile (Lemly, 1994). Selenium was identified as a possible chemical of concern in 2000, (Garcia-Hernandez et al., 2000) but has not bioaccumulated to levels of concern in sediments, fish or birds in the marsh (Garcia-Hernandez. et al., 2001, 2006). Levels in 2012 were similar to levels measured in 2000 (Garcia-Hernandex et al., 2012), suggesting the levels have reached a non-toxic equilibrium level due to the flushing action of the inflows.

C.4.3 Habitat value of the Cienega

The Cienega was first noted as a valuable aquatic ecosystem in 1989 (Eddleman, 1989), during a brief study of the food chain organisms including invertebrates, fish and birds. It was subsequently found to provide nesting habitat to the largest remaining population of the endangered Yuma clapper rails (Abarca et al., 1993; Piest et al., 1998; Hinojosa-Huerta et al., 2001, 2002). Periodic monitoring of Yuma clapper rail abundance from 1992 to 2012 showed no overall decline in nesting populations over time (Hinojosa-Huerta et al., 2008, 2012; in preparation). The outflow from the Cienega also provides habitat for migratory shorebirds in winter, with several hundred thousand individuals feeding in the Santa Clara Slough to the south of the Cienega (Gomez-Sapiens et al., 2012). The Cienega also supports the endangered pupfish (*Cyprinodon macularius*) in shallow areas on the periphery of the marsh (Zengel and Glenn, 1996). Vegetation vigor and marsh bird habitat value are markedly stimulated by occasional fires, which release nutrients back to the water column and remove thatch, improving the habitat value for nesting rails (Conway et al., 2010).

C.4.4 Factors contributing to the stability and habitat value of the Cienega

The Cienega appears to be an exception to the generalization that agricultural drain water degrades the quality of wetlands, leads to toxicity effects and cannot support high quality wildlife habitat in the long run (Lemly et al., 1993; Lemly, 1994; Letey, 2000). Several factors appear to be responsible for its stability over time. First, the MODE water inflow rates and salinities are remarkably stable over seasonal cycles and over years. The Wellton-Mohawk Irrigation District has a permanently high perched aquifer, recharged by surface flows and underflows from the Gila River, and from application of Colorado River water to the fields (Leitz and Ewoldsen, 1977) . From about 1995 to the present the flows have been remarkably steady at 4-5 $m^3 s^{-1}$ and 2-3 g L^{-1} TDS, with flows varying by less than 25% from month to month (International Boundary and Water Commission, 2012).

The winter flows might be especially important in preventing the accumulation of salts and chemicals of concern, as they flush the Cienega several times during the dormant period of the vegetation in winter. Furthermore, the Santa Clara Slough, which receives the outflow water, is flushed by tides on an approximately monthly basis (Nelson et al., 2012; in preparation). Hence, key factors contributing to the stability of the Cienega are its stable hydro period, flushing flows, and the fact that it is an open system connected to the Gulf of California.

The stability of the main vegetation units is due to the fact that *T. domingensis*, *N. marinas* and *P. australis* frequently form climax communities in marshes. In fact all three are considered invasive species in some wetland systems (Kaplan, 2012). In the Cienega they are distributed according to depth of water. *P. australis* requires shallow water and aerated soils (Hellings and Gallagher, 1992; Wijt and Gallagher, 2006), which occurs on silt deposits near the MODE entry point and along the periphery of the marsh. *T. domingensis* grows under anaerobic conditions in water up to 1.15 m deep (Grace, 1989), while *N. marinas* dominates the open lagoon areas where water is apparently too deep for *T. domingensis*. The combination of dense *T. domingensis* stands and open water lagoons with submerged aquatic vegetation has created valuable habitat for a wide variety of marsh birds, shorebirds, and other waterfowl (Abarca et al., 1993).

While the Cienega has reached a stable condition over the first 34 years, some longer term processes can impact the future development of the marsh. The continual entry of silt can eventually lead to more shallow water areas, favoring the growth of *P. australis* over *T. domingensis*, with a reduction in marsh bird habitat (Kaplan, 2012). Eventually some of the marsh could fill in enough to support the growth of *Tamarix* spp. and other halophyte shrubs, changing portions of the marsh to shrublands. Siltation can be controlled through **co**ntinued dredging and silt removal near the entry point of the MODE canal. However, excessive dredging can release soluble forms of selenium from the sediments into the water column, increasing the selenium hazard (Garcia-Hernandez et al., 2000, 2001, 2006). A continued flow of water from the Wellton-Mohawk Irrigation District is not guaranteed, and if farming practices in the district become more waterefficient, flows to the Cienega would be diminished. Operation of the Yuma Desalting Plant would also reduce inflow volumes and increase salinities (Leitz and Ewolden, 1977).

C.4.5 Implications for management

The example of the Cienega de Santa Clara shows that under the proper set of conditions, agricultural drainage water can be used to support valuable wetland habitat. An outlet to the ocean and flushing flows are important elements in creating a flow-through system in which salts and contaminants do not accumulate. The extreme tidal amplitude in the northern Gulf of California contributes to flushing and mixing the outflows from the Cienega into the ocean. Not only agricultural brines, but effluents from desalination plants could conceivably be used to support expanded wetlands in the delta of the Colorado River. For example, a collector drain has been proposed to carry desalination plant brine from reverse osmosis plants in southern Arizona for disposal in the vicinity of the Cienega (U.S. Department of Interior, 2004). This water would have salinities similar to MODE water (2-3 g L^{-1} TDS) and, based on the 34 year record of the Cienega, could result in enhanced and sustainable coastal wetland systems that contribute to restoration of wetlands lost through water diversions in the U.S. and Mexico (Glenn et al., 1996).

Table 1. List of Landsat images used in this study.

3-Aug-78	Landsat-2
11-Jul-79	Landsat-2
5-Jul-80	Landsat-2
18-Jul-81	Landsat-2
4-Jul-82	Landsat-3
17-Jul-83	Landsat-4
27-Jul-84	Landsat-5
30-Jul-85	Landsat-5
1-Jul-86	Landsat-5
17-Jul-86	Landsat-5
18-Aug-86	Landsat-5
3-Sep-86	Landsat-5
4-Jul-87	Landsat-5
22-Jul-88	Landsat-5
25-Jul-89	Landsat-5
20-Jul-90	Landsat-4
15-Jul-91	Landsat-5
1-Jul-92	Landsat-5
4-Jul-93	Landsat-5
7-Jul-94	Landsat-5
26-Jul-95	Landsat-5
16-Jul-96	Landsat-5
15-Jul-97	Landsat-5
18-Jul-98	Landsat-5
21-Jul-99	Landsat-5
7-Jul-00	Landsat-5
10-Jul-01	Landsat-5
29-Jul-02	Landsat-5
16-Jul-03	Landsat-5
2-Jul-04	Landsat-5
5-Jul-05	Landsat-5
8-Jul-06	Landsat-5
11-Jul-07	Landsat-5
13-Jul-08	Landsat-5
30-Jun-09	Landsat-5
3-Jul-10	Landsat-5
22-Jul-11	Landsat-5

Table 2. Dominant and subdominant plant species in the Cienega de Santa Clara.

Species	Notes
Typha domingensis (southern cattail)	Dominant emergent species; present throughout marsh
Shoenoplectus americanus, S. maritimus (three square, alkali bulrush)	Subdominant species present throughout marsh
Phragmites australis (common reed)	Forms dense monocultures on silt mounds near MODE entry point
Najas marinas (spiney niad)	Most common submerged aquatic species, dominant in open-water lagoons
Distichlis spicata, D. palmeri (salt grass, Palmer's salt grass)	Saltgrasses in shallow water around periphery of marsh
Allenrolfia occidentalis, Tamarix ramosissima (iodine bush, saltcedar)	Halophyte shrubs found on high ground within marsh
Juncus cooperi (Spiney rush)	Emergent species in shallow water near MODE entry point
Ruppia maritima (widgeon grass)	Submerged aquatic species found in saline outflow at south end of marsh

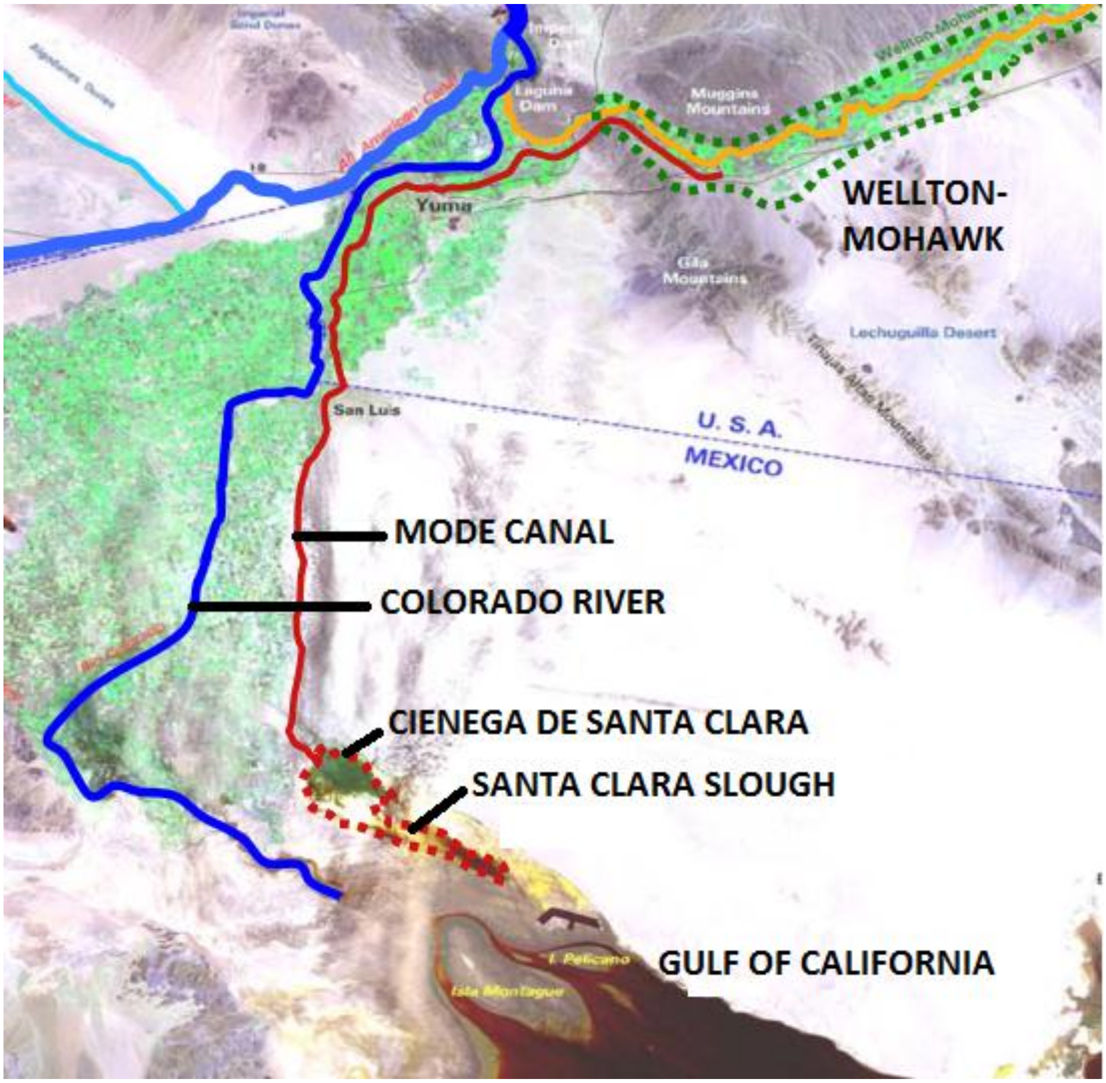

Figure 1. Locator map for Cienega de Santa Clara and the MODE canal water sources.

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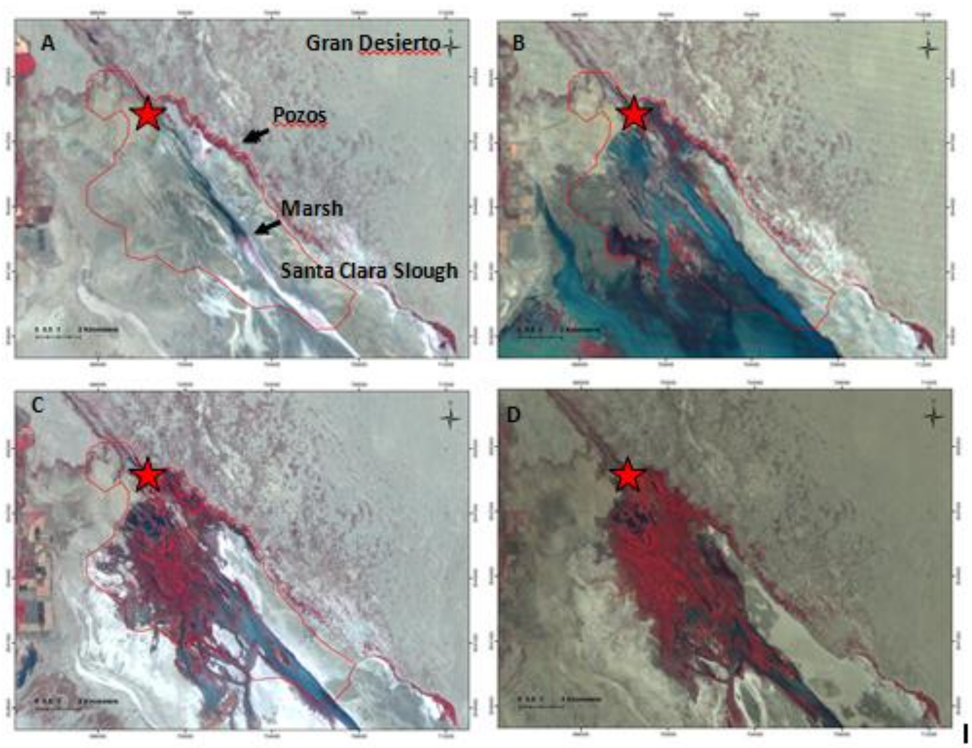

Figure 2. Development of the Cienega de Santa Clara showing Landsat images for the years 1975 (A), 1985 (B), 1990 (C) and 1995 (D). Red star shows the entry point for MODE canal water. Vegetation is shown as false-color red for reflectance by the NIR band.

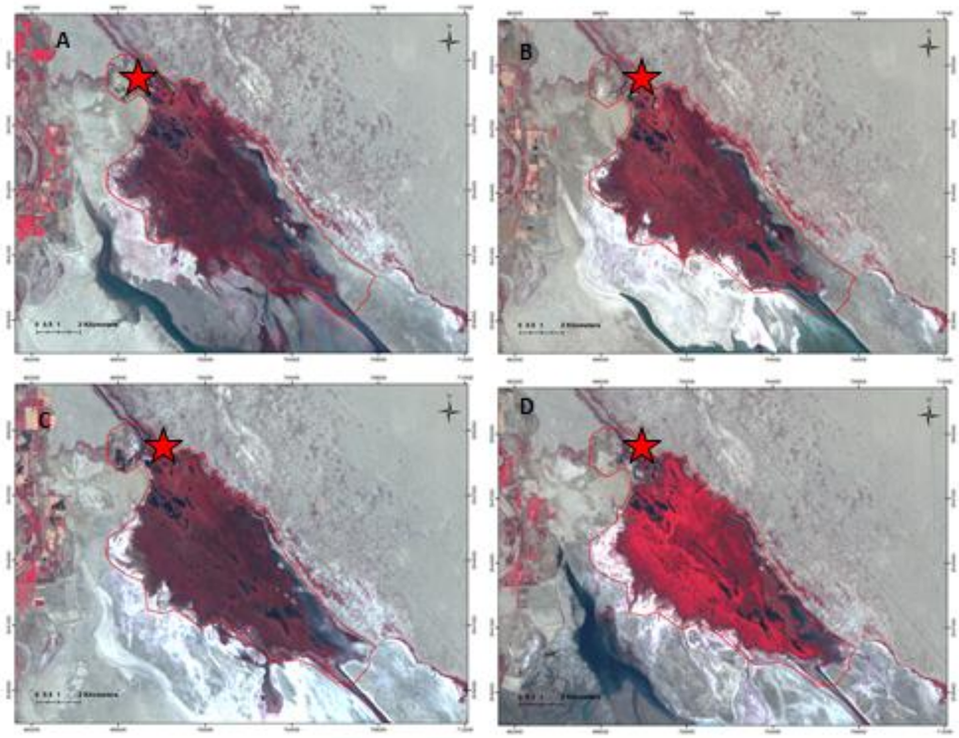

Figure 3. Development of the Cienega de Santa Clara showing Landsat images for the years 1975 (A), 1985 (B), 1990 (C) and 1995 (D). Red star shows the entry point for MODE canal water. Vegetation is shown as false-color red for reflectance by the NIR band.

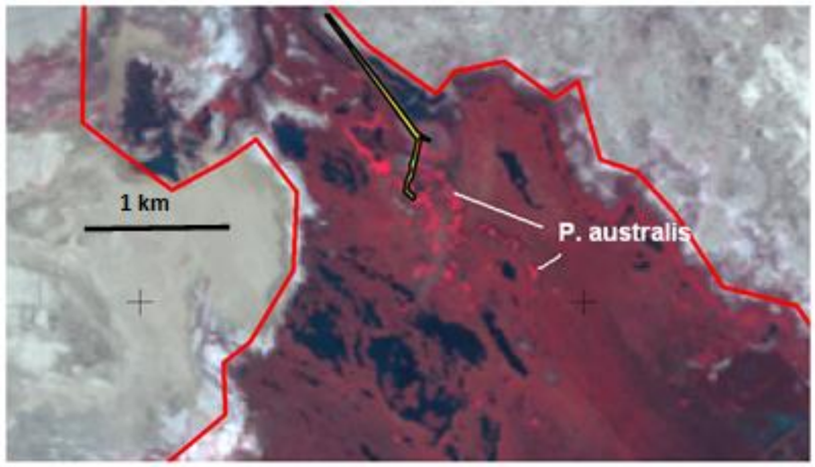

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Figure 4. Close-up 1995 image of the Cienega de Santa Clara at the entry point for MODE canal water (yellow line), showing the development of *Phragmites australis* (brighter red spots) on silt mounds around the mouth of the canal.

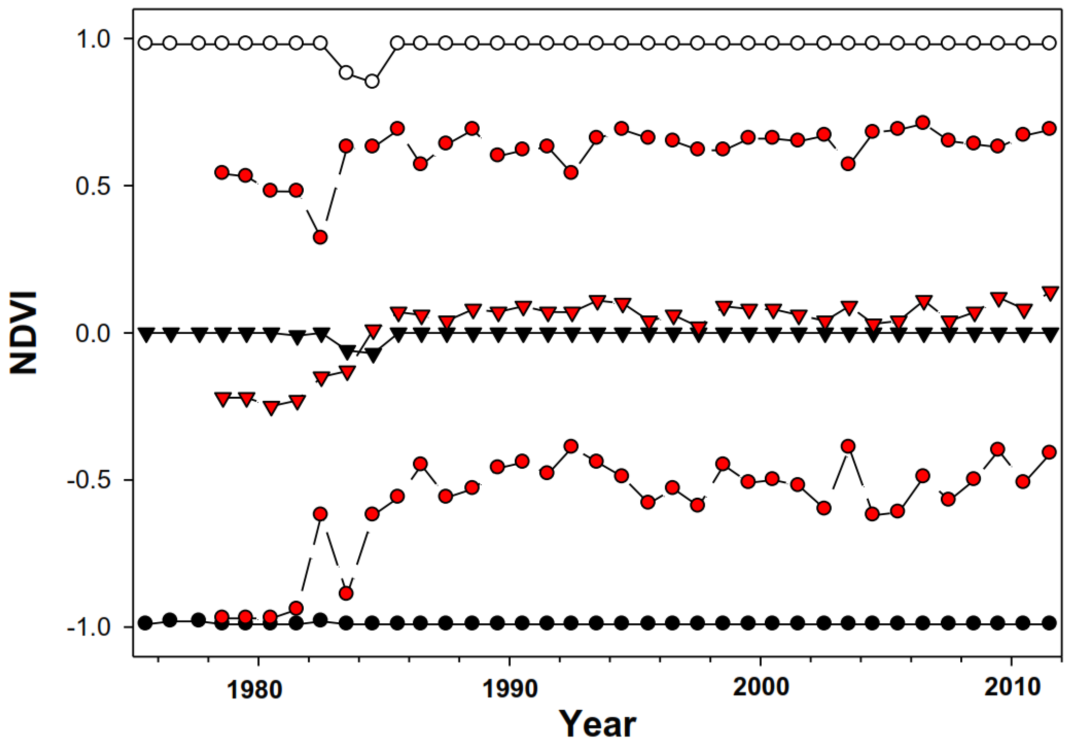

Figure 5. Minimum (closed circles), mean (black triangles) and maximum (open circles) NDVI values for whole Landsat images, 1975-2011, and for values within the footprint area of the Cienega de Santa Clara (red symbols). A separate area of interest file was prepared for the Cienega each year as it increased in area.

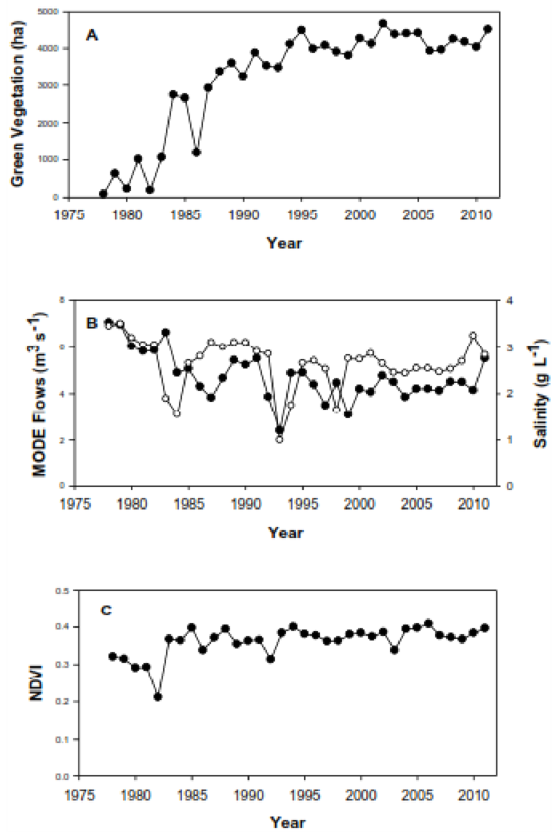

Figure 6. (A) Vegetation development; (B) MODE inflow rates (closed circles) and salinities (open circles); (C) mean NDVI for the Cienega de Santa Clara, 1978-2011

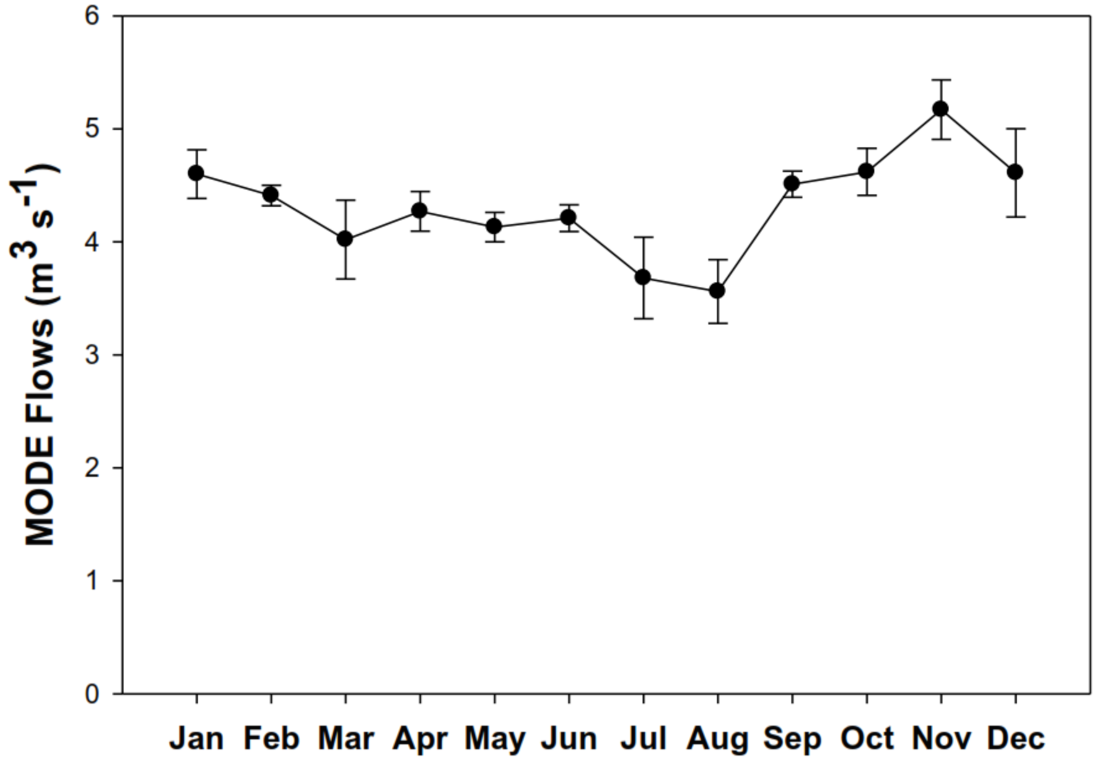

Figure 7. Mean monthly inflow rates of MODE water, 1978-2011. Error bars are standard errors of means.

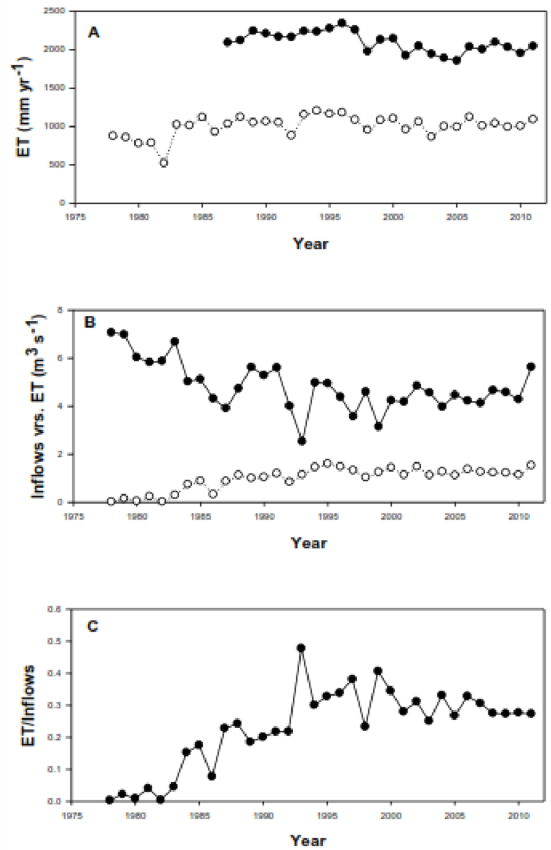

Figure 8. (A) Potential ET (closed circles) and ET by vegetation in Cienega de Santa Clara, 1978-2011; (B) inflow rates (MODE water plus local sources and precipitation) (closed circles) compared to ET by vegetation, 1978-2011; (C) fraction of inflow water consumed by vegetation in transpiration.

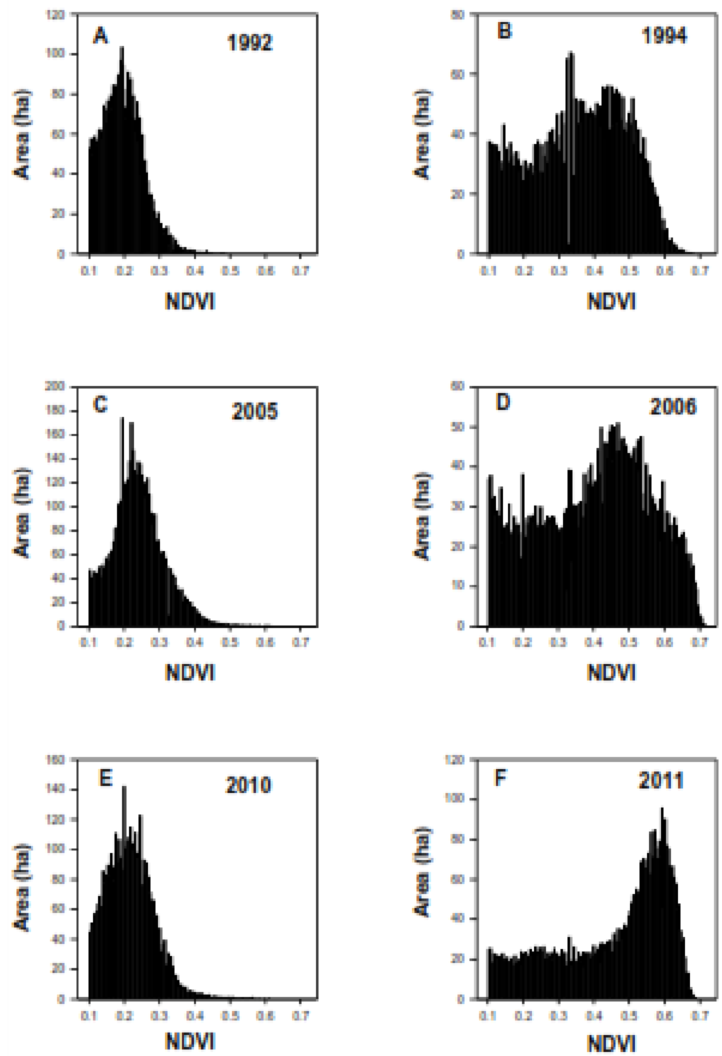

Figure 9. Effect of fire on the distribution of summer NDVI values in Cienega de Santa Clara, comparing non-fire years (A,C,E) with years in which winter/spring fires removed thatch (B, D, F).

References

Abarca, F.U, Ingraldi, M.F., Varela-Romero, A., 1993. Observations on the Desert Pupfish (Cyprinodon macularius, Yuma Clapper Rail Rallus longirostris yumanensis and Shorebird Communities in the Cienega de Santa Clara, Sonora, Mexico. September 1993. Nongame and Endangered Wildlife Technical Report, Arizona Game and Fish Department, Phoenix, Arizona.

Battin, J., 2004. When good animals love bad habitats: Ecological traps and the conservation of animal populations. Conserv. Biol. 18, 1482-1491.

Baugh, W.M., Groeneveld, D.P., 2006, Broadband vegetation index performance evaluated for a low-cover environment. Int. J. Remote Sens. 27, 4715-4730.

Beare, P.A., Zedler, J.B., 1987. Cattail invasion and persistence in a coastal salt marsh: The role of salinity reduction. Estuaries 10, 165-170.

Brouwer, C.; Heibloem, M. Irrigation Water Management Training Manual No. 3; FAO: Rome,
Italy, 1986.

Cohen M.J., Morrison J.I., Glenn E.P., 1999. Haven or Hazard: The Ecology and Future of the Salton Sea. Pacific Institute, Oakland, CA.

Conway, C.J., Nadeau, C.P., Priest, L., 2010. Fire helps restore natural disturbance regime to benefit rate and endangered marsh birds endemic to the Colorado River. Ecol. Appl. 20, 2024-2035.

Eddleman, W.R., 1989. Biology of the Yuma clapper rail in the southwestern U.S. and northwestern Mexico. Final Rep. Intra-Agency Agreement No. 4-AA-30-02060, Reclamation, Yuma Area Office, Yuma, Ariz. 127 pp.

García-Hernández J., Glenn, E.P., Artiola, J., Baumgartner, D.J., 2000. Bioaccumulation of selenium (Se) in the Cienega de Santa Clara wetland, Sonora, Mexico. Ecotox. Environ. Safe. 46, 298-304.

García-Hernández, J., King, K.A., Velasco, A.L., Shumilin, E., Mora, M.A., Glenn, E.P., 2001. Selenium, selected inorganic elements, and organochlorine pesticides in bottom material and biota from the Colorado River delta. J. Arid Environ. 49, 65-89.

García-Hernández, J., Sapozhnikova, Y.V., Schlenk, D., Mason, A.Z., Hinojosa-Huerta, O., Rivera-Díaz, J.J., Ramos-Delgado, N.A., Sánchez-Bon, G., 2006. Concentration of contaminants in breeding bird eggs from the Colorado River Delta, Mexico. Environ. Toxicol. Chem. 25, 164-1647.

Garde, L.M., Nicol, J.M., Conran, J.G., 2004. Changes in vegetation patterns on the margins of a constructed wetland after 10 years. Ecol. Manage. Restor. 5, 111-117.

Glenn E.P., Felger, R.S., Burquez, A., 1992. Cienega de Santa Clara - endangered wetland in the Colorado River Delta, Sonora, Mexico. Nat. Resour. J. 32, 817-824.

Glenn, E.P., Lee, C., Felger, R., Zengel, S., 1996. Effects of water management on the wetlands of the Colorado River delta, Mexico. Conserv. Biol. 10, 1175-1186.

Glenn E.P., Thompson, T.L., Frye, R., Riley, J., Baumgartner, D., 1995. Effects of salinity on growth and evapotranspiration of Typha domingensis Pers. Aquatic Bot. 52, 75-91.

Grace, J.B., 1989. Effects of water depth on Typha latifolia and Typha domingensis. Am. J. Bot. 76, 527-535.

Groeneveld, D.P., Baugh, W. M., 2007. Correcting satellite data to detect vegetation signal for eco-hydrologic analyses. J. Hydrol. 135-145.

Groeneveld, D. P., Baugh, W.M., Sanderson, J., Cooper, D.J., 2007. Annual groundwater evapotranspiration mapped from single satellite scenes. J.Hydrol., 146-
156.

Hellings, S.E., Gallagher, J.L., 1992. The effects of salinity and flooding on Phragmites australis. J. Appl. Ecol. 29, 41-49.

Gregoire, C., Elsaesser, D., Huguenot, D., Lange, J., Lebeau, T., Merli, A., Mose, R., Passeport, E., Payraudeau, S., Schutz, T., Schulz, R., Tapia-Padilla, G., Tournebize, J., Trevisan, M., Wanko, A., 2009. Mitigation of agricultural nonpoint-source pesticide pollution in artificial wetland ecosystems. Environ. Chem. Lett. 7, 205-231.

Hinojosa-Huerta, O., DeStefano, S., Shaw, W.W., 2001. Distribution and abundance of the Yuma clapper rail (Rallus longirostris yumanensis) in the Colorado River delta, Mexico. J. Arid Environ. 49, 171–182.

Hinojosa-Huerta O., DeStefano, S., Shaw, W.W., 2002. Evaluation of call-response surveys for monitoring breeding Yuma Clapper Rails (Rallus longirostris yumanensis). J. Field Ornithol. 73, 151-155.

Hinojosa-Huerta, O., Rivera0Diaz, J.J., Iturribarria-Rojas, H., Calvo-Fonseca, A., 2008. Population trends of Yuma clapper rails in the Colorado River delta, Mexico. Stud. Avian. Biol. 37, 69-73.

Hinojosa-Huerta, O., Iturrabarria-Rojas, H., Zamora-Hernandex, E., Calvo-Fonseca, A., 2008. Densities, species richness and habitat relationships of avian species in the Colorado River, Mexico. Stud. Avian Biol. 37, 74-82.

Kaplan, D., 2012. Instability in newly established wetlands? Trajectories of floristic change in the re-flooded Hula poeatland, northern Israel. Mires Peat 9, Article 05, 1-10,

Leitz, F.B., Ewoldsen, E.I., 1977. Design criteria for the Yuma desalting plant. Desalination 24, 321-340.

Lemly, A.D., 1994. Irrigated agriculture and freshwater wetlands: A struggle for coexistence in the western United States. Wetlands Ecol. Manage. 3, 3-15.

Lemly, A.D., Finger, S.E., Nelson, M.K., 1993. Sources and impacts of irrigation drainwater contaminants in arid wetlands. Environ. Toxicol. Chem. 12, 2265-2279.

Letey, J., 2000. Soil salinity poses challenges for sustainable agriculture and wildlife. Calif. Agric. 54, 43-48.

Lin, Z.Q., Terry, N., Gao, S., Mohamed, S., Ye, Z.H., 2010. Vegetation changes and partitioning of selenium in 4-year-old constructed wetlands treating agricultural drainage. J. Phytorem. 3, 255-267.

Mitsch, W.J. 1992. Landscape design and the role of created, restored, and natural riparian wetlands in controlling nonpoint source pollution. Ecol. Eng. 1, 27-47.

Mitsch, W.J., Wilson, R.F., 1996. Improving the success of wetland creation and restoration with know-how, time, and self-design. Ecol. Appl. 6, 77-83.

Mitsch, W.J., Zhang, L., Anderson, C.J., Altor, A.E., Hernandez, M.E., 2005. Creating riverine wetlands: Ecological succession, nutrient retention, and pulsing effects. Ecol. Eng. 25, 510-527.

Moreno-Mateos, D., Comin, F.A., 2010. Integrating objectives and scales for planning and implementing wetland restoration and creation in agricultural landscapes. J. Environ. Manage. 11, 2087-2095.

Piest, L. and J. Campoy, 1998. Report of Yuma Clapper Rail Surveys at Cienega de Santa Clara, Sonora. Arizona Game and Fish Department, Yuma Regional Office, Yuma Arizona.

Tanner, R., Glenn, E.P., Moore, D., 1999. Food chain organisms in hypersaline, industrial evaporation ponds. Water Environ. Res. 71, 494-505.

Tarver, D.P., Rogers, J.A., Mahler, M.J., Lazor, R.L., 1986. Aquatic and Wetland Plants of Florida. Third Edition. Florida Department of Natural Resources, Tallahassee, Florida.

U.S. Department of Interior, 2004. Reverse Osmosis Treatment of Central Arizona Project Water for the City of Tucson. Desalination Research and Development Report No. 36, Bureau of Reclamation, Denver, Colorado.

Wijt, H.B., Gallagher, J.L., 2006. Effect of oxygen availability and salinity on early life history stages of salt marsh plants. I. Different germination strategies of Spartina alterniflora and Phragmites australis (Poaceae). Am. J. Bot. 83, 1337-1342.

Wu, L., 2004. Review of 15 years of research on ecotoxicology and remediation of land contaminated by agricultural drainage sediment rich in selenium. Ecotoxic. Environ. Safe. 57, 257-269.

Zedler, J.B., Callaway, J.C., 1999. Tracking wetland restoration: Do mitigation sites follow desired trajectories? Restor. Ecol. 7, 69-73.

Zengel S.A., Glenn, E.P., 1996. Presence of the endangered desert pupfish (Cyprinodon macularius, Cyprinidontidae) in Cienega de Santa Clara, Mexico, following an extensive marsh dry-down. Southwest. Nat. 41, 73-78.

Zengel, S.A., Meretsky, V.J., Glenn, E.P., Felger, R.S., Ortiz, D., 1995. Cienega de Santa Clara, a remnant wetland in the Rio Colorado delta, Mexico. Vegetation distribution and the effects of water flow reduction. Ecol. Eng. 4, 19-36.