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Evapotranspiration and water balance of an anthropogenic coastal desert wetland: Responses to fire, inflows and salinities

Edward P. Glenn[a,*], Lourdes Mexicano[a], Jaqueline Garcia-Hernandez[b], Pamela L. Nagler[c], Martha M. Gomez-Sapiens[a], Dawei Tang[a], Marcelo A. Lomeli[d], Jorge Ramirez-Hernandez[d], Francisco Zamora-Arroyo[e]

[a] *Department of Soil, Water and Environmental Science, University of Arizona, Tucson, AZ, USA*
[b] *Centro de Investigacion en Alimentacion y Desarrollo A.C., Hermosillo, Mexico*
[c] *U.S. Geological Survey, Sonoran Desert Research Station, University of Arizona, Tucson, AZ, USA*
[d] *Istituto Ingenieria, Universidad de Baja Calfiornia, Ensenada, Mexico*
[e] *Sonoran Institute, Tucson, AZ, USA*

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ABSTRACT

Evapotranspiration (ET) and other water balance components were estimated for 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 to 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. The goal was to estimate the main components in the water budget to be used in creating management scenarios for this marsh. 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 then compared to results from a mass balance of water and salt inflows and outflows over the study period. By both methods, mean annual ET estimates ranged from 2.6 to 3.0 mm d^{-1}, or 50 to 60% of reference ET (ET_o). Water entered at a mean salinity of 2.6 g L^{-1} TDS and mean salinity in the wetland was 3.73 g L^{-1} TDS over the 33 month study period. 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 estimates over the years 2000–2011 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 peak summer ET compared to non-fire years. Following fires, peak summer ET estimates 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, estimated ET was always lower than ET_o, because *T. domingensis* 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.

1. Introduction

1.1. Wetland water budgets and the role of evapotranspiration

The water budget of a wetland determines its aerial 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 can be especially difficult to estimate (Drexler et al., 2004). For wetlands dominated by emergent vegetation, there is a debate about the magnitude of wetland ET in relation to reference crop ET (ET_o) (Allen et al., 1998) 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

* Corresponding author. Tel.: +1 520 626 2664; fax: +1 520 573 0852.
E-mail addresses: eglenn@ag.arizona.edu, eglkenn@ag.arizona.edu (E.P. Glenn).

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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 and Meyerson, 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).

Emergent wetlands can differ markedly in their ecohydrology, depending on dominant species, salinity, water depth and other environmental factors (Bullock and Acreman, 2003; Jolly et al., 2008), explaining much of the discrepancy in ET among studies. However, part of the controversy is also due to differences in methods to estimate wetland ET (reviewed in Drexler et al., 2004).

1.2. Methods for estimating wetland ET

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 (Environmental Protection Agency, 1998; Mitsch and Gosslink, 2000; Jia et al., 2011). Under ideal conditions the water balance approach can provide an accurate estimate of ET over annual or long time periods, when seasonal changes in storage become negligible, especially for 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), scintillometry (Lenters et al., 2011), surface renewal methods based on heat fluxes over the canopy (Drexler et al., 2004, 2008), diurnal fluctuations in ground water 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 can have a potential error or uncertainty of 20–30%, and often no alternative method for independently validating ET estimates is available (Drexler et al., 2004).

1.3. Remote sensing and water budget approaches to estimating wetland ET in the present study

This study used a vegetation-index-based remote sensing method based on the Enhanced Vegeation Index from the Moderate Resolution Imaging Spectrometer (MODIS) on the Terra satellite (Huete et al., 2002, 2011), tested against a salt-and-water balance approach (Jia et al., 2011) to estimate 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 5000 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. 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). 80% of the remaining endangered Yuma Clapper Rails (an endangered marsh bird) 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).

1.4. Objectives of the study

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. 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 water balance for the Cienega during operation of the YDP. The objectives of the study were to estimate the seasonal and inter-annual variations in ET from the marsh in response to inflows and environmental factors, and to create a water budget for the Cienega, which could ultimately be used as a management tool for predicting the ecosystem response to different operating scenarios of the YDP.

2. Materials and methods

2.1. Description of Cienega vegetation and hydrology

The Cienega is located at the north end of the Santa Clara Slough, a tidal basin formed where the Cerro Prieto fault line enters the Gulf of California (Fig. 1A). The Cienega is in a hot desert environment, with mean annual temperature of 23.3 °C and annual rainfall of 80 mm based on data from Yuma in the U.S., approximately 100 km to the north (AZMET, 2012). The southern end of the Santa Clara Slough is flushed approximately monthly by high tide events but tide water does not enter the Cienega (Flessa et al., 2012). The main source of water in the Cienega is the MODE canal, which discharges at the northern end of the marsh, along with a smaller volume of local drain water entering in the Riito canal (Fig. 1B). Water in the MODE canal has entered the wetland at a rate of approximately 4 $m^3\ s^{-1}$ and 2–3 $g\ L^{-1}$ Total dissolved solids (TDS) since 1977, and the footprint of the marsh has been stable at about 5000 ha since 1995 (Zengel et al., 1995). *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). Vegetation covers about 85% of the marsh area and open water lagoons cover and additional 15%; the lagoons are in areas of deeper water where emergent vegetation does not grow, and they have been stable features in the marsh since 1995 (Zengel et al., 1995). Fig. 1B shows the sampling stations for salinity measurements and locations where MODIS pixels were acquired to estimate ET.

The average depth of water in the Cienega is 0.32 m, but depths exceed 1 m in the open water lagoons (Flessa et al., 2012). At a mean inflow rate of 4 $m^3\ s^{-1}$ from the MODE canal, the nominal detention time for water is 46 d. The flow of water appears to be predominantly through the middle of the marsh following the deeper water channel formed by the Cerro Prieto fault line (Zengel et al., 1995). The main outlet for water is at the southern end of the Cienega where water flows into the Santa Clara Slough, but water also spills out of the wetland at several points along the western edge (Fig. 1B).

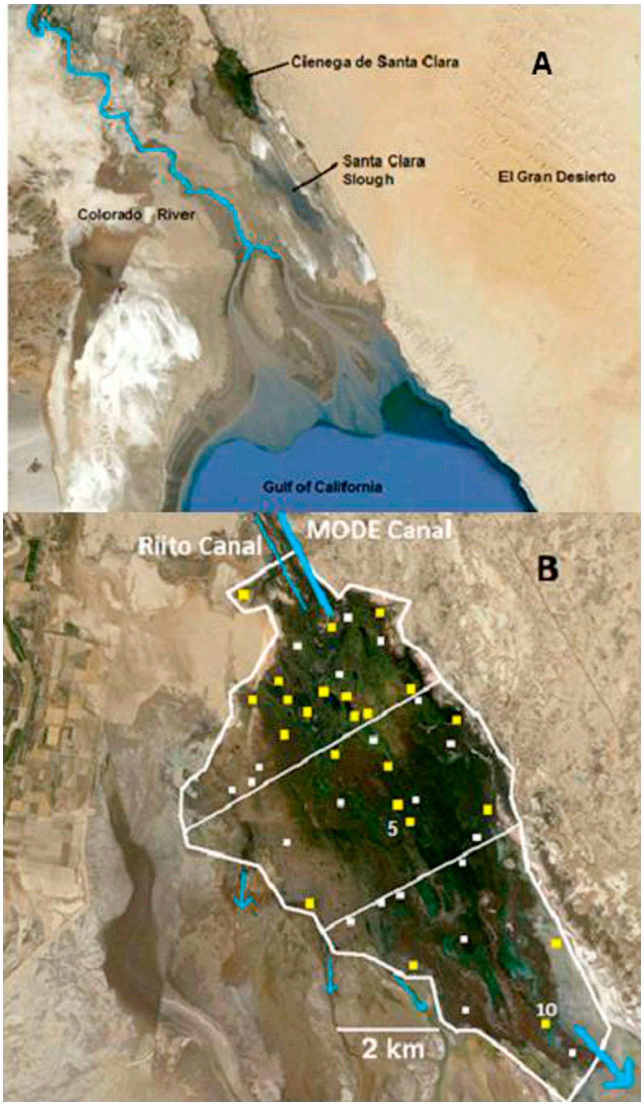

Fig. 1. (A) Location of Cienega de Santa Clara in the delta of the Colorado River. (B) Footprint of Cienega de Santa Clara and locations from which MODIS EVI data (white squares) and salinity data (yellow squares) were collected. Salinity stations 5 and 10 are numbered as they are mentioned in the text. Blue lines indicate entry points for water in the MODE and Riito canals; blue arrows indicate exit points of water into the Santa Clara Slough. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

2.2. Determining vegetation cover and open water areas

Separate methods were used to estimate ET for open water lagoons and vegetated areas within the Cienega, hence estimates were needed of the extent of vegetation and open water areas during the study period. The footprint area of the Cienega and extent of open water areas within the Cienega were determined on April 24, 2010 Quickbird satellite image (Digital Globe, Inc., Longmont, CO). Polygons delineating the perimeter of the Cienega and each of 543 open water areas within the Cienega were created using ArcMap 10 sofltware (ESRI, Inc., Redland, CA). The area of the whole Cienega was 4913 ha, of which 518 ha were open water areas (11% of the total area) and 89% was thickly vegetated. Analyses of other Quickbird images collected 4 times per year from 2009 to 2001 showed that the footprint area of the marsh and areas of open water were stable during the study, although the intensity of vegetation changed seasonally and inter-annually (Flessa et al., 2012), hence the proportion of vegetation and open water in the marsh were assumed to be constant over this study. Weighted mean ET was therefore estimated by:

$$ET_{RS} = 0.11E_{water} + 0.89E_{veg} \quad (1)$$

where ET_{RS} is ET estimated by remote sensing.

2.3. E_{water} and E_{veg} estimates

In this study, both E_{water} and E_{veg} depend on an estimate of reference evapotranspiration (ET_o) as determined from meteorological data. We used the Blaney–Criddle formula for ET_o ($ET_{o\text{-}BC}$), which requires mean monthly temperature (T_{mean}) and hours of potential sunlight based on latitude (p, obtained from a table) (Brouwer and Heibloem, 1986):

$$ET_{o\text{-}BC} = p(0.46T_{mean} + 8) \quad (2)$$

Mean monthly temperature data were obtained from the Yuma Valley AZMET station (AZMET, 2012). E_{water} was assumed to be equal to $ET_{o\text{-}BC}$ (Huckelbridge et al., 2010; Mould et al., 2011). As justification, annual evaporation from the Salton Sea (150 km from the Cienega), for which evaporation data are available, is 1798 mm yr^{-1} (Ponce, 2005), compared to our estimate of $ET_{o\text{-}BC}$ of 1820 mm yr^{-1} for the Cienega based on Yuma AZMET data (AZMET, 2012).

E_{veg} estimates were based on 16-d, composite EVI values from MODIS (MOD13Q1 product) (Nagler et al., 2009b). MODIS provides near-real-time imagery of most of the earth at daily intervals with 250 m resolution in the red and NIR bands and 500 m in the blue band. Images 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). EVI was used rather than the more familiar Normalized Difference Vegetation Index (NDVI) because in previous studies it was more highly correlated with ET ($r = 0.83$) than the MODIS NDVI product ($r = 0.72$) ($P < 0.05$) at 11 upland and riparian moisture flux tower sites in the southwestern U.S. (Nagler et al., 2005a; Glenn et al., 2010). EVI is calculated as:

$$EVI = \frac{G(\rho_{NIR} - \rho_{Red})}{\rho_{NIR} + C1 \times \rho_{Red} + C2 \times \rho_{Blue} + L} \quad (3)$$

where $C1$ and $C2$ are coefficients designed to correct for aerosol resistance, which uses the blue band to correct for aerosol influences in the red band. The coefficients, $C1$ and $C2$, are set at −6 and 7.5, respectively, based on empirical studies in which the coefficients were varied in magnitude and the signal-to-noise ratios were compared for canopies of different leaf area index (LAI) and different soil types, using band values corresponding to those in the MODIS sensors (Huete et al., 1994). G is a gain factor (set at 2.5), and L for this model is a canopy background adjustment (set to 1.0). EVI data were obtained from the Oak Ridge National Laboratory DAAC site (ORNL DAAC, 2012).

In estimating E_{veg} via EVI, it was important to avoid pixels that included open water areas, because the negative values for water can give artificially low estimates of vegetation density and ET. This is a particular problem with MODIS imagery due to the large pixel area (6.25 ha). Two methods were compared for obtaining spatially distributed EVI data over the Cienega. In the first, a shape file was created that encompassed most of the vegetated area in the Cienega, but excluded the open water areas identified on the April 24, 2010 Quickbird image. In the second method, individual pixels from 21 sites distributed throughout the Cienega (Fig. 1B) were obtained using the MODIS subset tool at the ORNL DAAC website (ORNL DAAC, 2012). This tool displays the footprint area of a selected pixel on a high-resolution Quickbird image. The

Cienega was divided into three sections representing upper, middle and lower sections of the marsh and seven pixels were randomly selected in each section (Fig. 1). If a selected pixel contained water on inspection of the Quickbird image, a new pixel location was randomly selected.

Algorithms from Nagler et al. (2009b) were used to estimate E_{veg}. This method is one of several remote sensing methods that use a crop-coefficient approach for estimating ET, derived from the relationship between crop ET and ET_o described in Allen et al. (1998):

$$ET = K_c\ ET_o \quad (4)$$

where K_c is a crop coefficient, determined empirically, that relates crop ET to that of a hypothetical reference crop based on meteorological data. In the remote sensing methods (ET_{RS}), K_c is replaced by a satellite-derived VI that relates to the status of the canopy at the time of measurement:

$$ET = k\ VI(ET_o) \quad (5)$$

where k is an empirically derived coefficient relating ET measured on the ground to $VI(ET_o)$ (Bausch and Neale, 1989; Choudhury et al., 1994). Numerous variations on Eq. (5) have been developed for particular crops, including non-linear equations (e.g., Nagler et al., 2005a,b), and the method has been extended to natural ecosystems as well, with corrections for soil moisture and other environmental and plant factors applied to the algorithms (reviewed in Glenn et al., 2010, 2011). While these methods often successfully reproduce ground measurements of ET, it needs to be noted that they are not direct measurements of ET, but are rather approximations with sources of error and uncertainty, and they should be tested against other methods of estimating ET whenever possible, and caution should be exercised in extending them beyond the biome for which they were developed (Glenn et al., 2007, 2011; Kalma et al., 2008).

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

$$EVI^* = 1 - \frac{EVI_{max} - EVI}{EVI_{max} - EVI_{min}} \quad (6)$$

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). Although VI values can be used directly to estimate ET (Choudhury et al., 1994), the scaling procedure allows regressions of ET versus EVI* to pass through the origin, where at 0 ET (bare dry soil) EVI* = 0. Furthermore, the scaling procedure helps in normalizing VI values across different sensor systems and VI algorithms (Baugh and Groeneveld, 2006).

E_{veg} was then calculated as:

$$E_{veg} = 1.22(EVI^*)ET_{o\text{-}BC} \quad (7)$$

Eq. (7) was developed for crops and riparian plants on the Lower Colorado River by regressing ET measured on the ground for alfalfa, cottonwood, arrowweed, saltcedar and mixed plant stands at 10 sites on the river with EVI* and meteorological data obtained from AZMET stations. The standard error of estimate was 20% of the mean value across sites and plant types (Nagler et al., 2009b). It was subsequently used to estimate riparian and agricultural ET in the wildlife refuges and irrigation districts on the Lower Colorado River (Murray et al., 2009), where it was compared to crop-coefficient estimates; ground water discharge by phreatophytes in the basin and range province of Arizona (Tillman et al., 2012), where it was compared to catchment water balance estimates; and ET in riparian sites on six western rivers (Nagler et al., 2012), where it adequately reproduced saltcedar ET rates measured by sap flow sensors at two sites, as well as Landsat estimates of ET using a method developed by Groeneveld et al. (2007), with different methods producing ET estimates within 15–20% of each other across the studies. However, the present study was the first application of the method to emergent wetland vegetation.

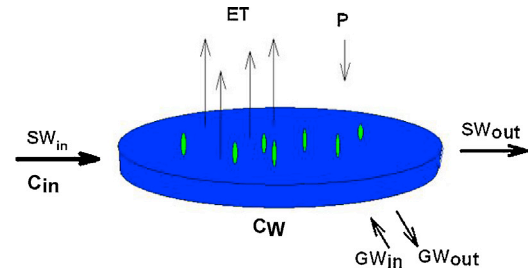

Fig. 2. Schematic of Cienega de Santa Clara showing water budget terms used in calculating evapotranspiration by a mass balance approach. SW_{in} = surface water in; C_{in} = salinity of inflow water; ET = evapotranspiration; P = precipitation; C_w = salinity in the wetland; GW_{in} = ground water in; GW_{out} = ground water out; SW_{out} = surface water out. The schematic assumes that water volume in the wetland is constant.

2.4. ET by a mass balance approach

Over long time periods (one year or greater) mass balance approaches can provide accurate estimates of ET for well-defined catchment areas (e.g., King et al., 2011; Senay et al., 2011). Long time periods are required so that inflows and outflows, which can be measured, are large in relation to changes in storage terms (surface water and ground water), which are more difficult to measure. A mass balance estimate of ET (ET_{MB}) was developed for the Cienega using inflow and salinity data collected in monthly time steps over 33 months (January 2009–September 2011). ET_{MB} was then compared to ET_{RS} for the same time period. ET_{MB} of a wetland can be calculated as (Environmental Protection Agency, 1998; Mitsch and Gosslink, 2000):

$$ET_{MB} = SW_{in} + P + GW_{in} - SW_{out} - GW_{out} - \Delta S \quad (8)$$

where SW_{in} is surface water inflow, P is precipitation, GW_{in} is ground water in, SW_{out} is surface water outflow, GW_{out} is ground water outflow and ΔS is change in storage in the wetland (shown schematically in Fig. 2). SW_{in} was measured by flow gages in the MODE canal and in the Riito canal (Flessa et al., 2012). P data were from the Yuma Valley AZMET station (AZMET, 2012). SW_{out}, GW_{in} and GW_{out} were not measured in this study. However, combined outflows can be calculated from the difference in salinity of inflow water compared to mean salinity in the wetland, by conservation of mass (Jia et al., 2011). At equilibrium, salts out of the wetland must be equal to salts in, to maintain a constant salinity in the wetland. Furthermore, ET removes pure water from the wetland, hence the salinity of the wetland increases in proportion to ET losses. Jia et al. (2011) defined an equilibrium wetland outflow ratio, β as:

$$\beta = \frac{W_{out}}{W_{in}} = \frac{C_{in}}{C_w} \quad (9)$$

where W_{in} are total water inflows, W_{out} are total water outflows, C_w is the mean salinity in the wetland, and C_{in} is the salinity of the inflow water. As an example, if half the inflow water exits the marsh as surface flows or net ground water discharge, β = 0.5, and if the inflow salinity in 2 g L^{-1}, the mean wetland salinity is 4 g L^{-1}. In the Jia et al. (2011) model C_w is corrected for the uptake of salt into vegetation, which is harvested in their model system, but in the Cienega salts in vegetation are recycled within the wetland as litter decays or burns, hence this factor is not included in Eq. (9).

In the Cienega, salinity was measured monthly at a series of fixed stations throughout the marsh, by which C_w could be determined at monthly time steps (see Section 2.5), allowing W_{out} in Eq. (9) ($SW_{out} + GW_{out} - GW_{in}$ in equation) to be calculated, hence ET by mass balance could be estimated from Eqs. (8) and (9) as (Jia et al., 2011):

$$ET = (SW_{in} + P)(1 - \beta) \quad (10)$$

This calculation assumes that the wetland is in equilibrium with respect to water and salt inflows and outflows with negligible changes in ΔS; over time periods of a year or longer, these are considered to be reasonable assumptions for mature wetlands (Jia et al., 2011). Total flows into the Cienega were approximately 2.7×10^8 m^3 during the 33 month measurement period, 17 times greater than the volume of the Cienega (approximately 1.6×10^7 m^3), justifying the assumption that changes in storage volume over the study period could be ignored in the ET_{MB} estimation. Spatial and temporal variations in salinity during the study are presented in Section 3.3.

ET calculated by Eq. (10) represents the total volume of ET in m^3 from the wetland over the 33 month measurement period; it was converted to a daily mass-balance ET rate in mm d^{-1} by dividing by the area of the wetland as determined by Quickbird imagery and the number of days of measurement, times 1000 mm m^{-3}, for comparison with ET_{RS}.

2.5. Other data sources and statistical analyses

Salinity data were collected monthly from nine stations in 2009 and from 22 stations in the marsh from 2010 and through June 2011, by the Cienega Monitoring Team (described in Flessa et al., 2012) (location of measurement stations are shown in Fig. 1). Electrical conductivity (EC) was measured with a YSI 6600V2 sonde (YSI, Inc., Yellow Springs, OH). EC in dS m^{-1} was concerted to TDS in g L^{-1} using a factor of 0.635, determined by measuring the EC of water samples in the field then filtering the sample and reducing it to dryness in an oven in the laboratory to determine weight of dissolved solids. An attempt was made to collect salinity data throughout the marsh, but monitoring points were located where access routes were available. C_w in Eq. (8) was calculated as the mean of all reporting stations over the entire measurement period. A contour plot of salinity values over the whole measurement period was created using SigmaPlot software (Systat, Inc., Chicago, IL). Due to the lack of data along the center line for the southern portion of the Cienega, an additional salinity point was calculated at the point equidistant between stations 5 and 10 (Fig. 1B) based on the mean of those values (3.14 and 4.02 g L^{-1} TDS, respectively). This interpolated value was used in construction of the contour plot but was not included in the mass balance calculations.

Inflow volume and salinity data in the MODE canal 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). Inflow salinity was corrected by a small contribution of water at slightly higher salinity from the local Riito Drain in Mexico, which was sampled approximately monthly during the study, and inflow volume was corrected for the small contribution of rainfall and inflows from the Riito Drain, which was gaged during the study period (see Table 1) (Flessa et al., 2012).

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_{RS} estimate based on the mass balance analysis was calculated from the standard errors of the monthly salinity and flow measurements ($n = 33$) used in the ET calculation, using the propagation of errors method in Taylor (1997). The coefficient of variation (CV%) was calculated as mean/standard deviation × 100.

Table 1
Mean and standard errors of hydrological parameters used in estimating ET by a water budget equation, compared to ET estimating by MODIS satellite imagery.

Parameter	Mean (std. error)
Flows in (m^3 s^{-1})	
MODE	4.75 (0.18)
Riito	0.21 (0.08)
Precipitation	0.013
Total in (m^3 s^{-1})	4.97 (0.19)
TDS in (g L^{-1})	
MODE	2.59 (0.06)
Riito	3.46 (0.15)
Precipitation	0.0
Weighted mean TDS in (g L^{-1})	2.62 (0.07)
TDS mean Cienega (g L^{-1})	3.73 (0.08)
ET by mass balance (mm d^{-1})	2.62 (0.11)
ET by MODIS (mm d^{-1})	2.97 (0.16)

3. Results

3.1. Comparison of methods for determining spatially distributed EVI values

The AOI and pixel-sampling methods were highly correlated ($r^2 = 0.98$) (Fig. 3), 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. The variance of EVI values across sample points was small (coefficient of variation = 6.4% of the mean value) over all images.

3.2. ET_{RS} in the Cienega, 2000–2011

Peak summer EVI values were variable among years, but were markedly stimulated by early spring fires in 2006 and 2011; as a result, ET_{RS} estimated from EVI and ET_{o-BC} followed the same trend (Fig. 4). These fires burned over 70% of the marsh area and removed accumulated thatch from previous years' growth,

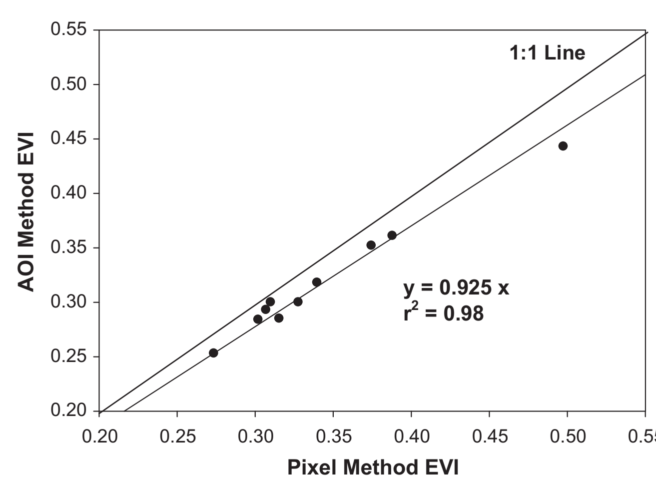

Fig. 3. Comparison of two methods to determine mean MODIS EVI over the vegetated portion of the Cienega. The Area of Interest (AOI) method used a shape file that included vegetation but excluded major open water areas; the pixel method sampled 21 individual pixels in which standing water was not present based on inspection of high-resolution Quickbird imagery.

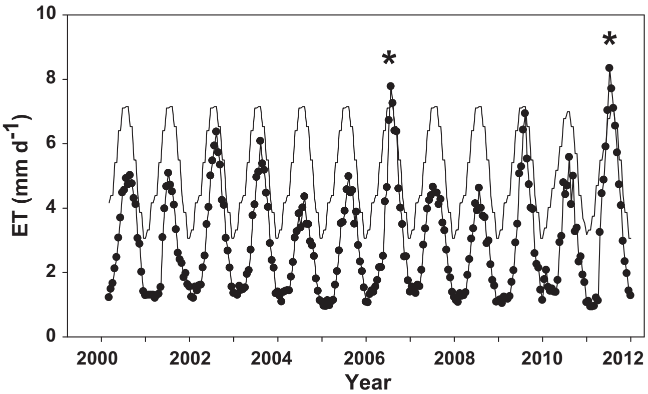

Fig. 4. ET_{RS} estimates for Cienega de Santa Clara, 2000–2011 based on MODIS EVI. Asterisks show when major fires occurred. Solid line without symbols shows potential ET.

allowing more light penetration into the canopy in spring (Conway et al., 2010), and presumably returned nutrients to the water (Liu et al., 2010). Following fires in 2006 and 2010, peak summer ET_{RS} estimates equaled $ET_{o\text{-}BC}$; however, in non-fire years peak rates were one-half to two-thirds of $ET_{o\text{-}BC}$. Mean monthly ET_{RS} estimates over fire and non-fire years are in Fig. 5, showing that peak summer ET_{RS} increased by 30% following spring fires. Mean annual ET_{RS} in non-fire years was 1081 mm yr^{-1}, 59% of $ET_{o\text{-}BC}$ (1820 mm yr^{-1}), while ET_{RS} in fire years was 1299 mm yr^{-1}, 71% of $ET_{o\text{-}BC}$.

Summer flow rates in the MODE canal were also a determinant of ET_{RS}. A multiple linear regression analysis for the period May 15 to September 15 of each year showed that both flow rates ($P = 0.012$) and presence or absence of fire ($P < 0.001$) were significant predictors of ET_{RS}. 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 ($P = 0.752$). The equation of best fit was:

$$ET_{RS}\,(\mathrm{mm\ d^{-1}}) = 1.07\,(\mathrm{Fire}) + 1.349\,(\mathrm{Flows}) \qquad (11)$$

where Fire was either 0 (non-fire year) or 1 (fire year) and flows were in $m^3\ s^{-1}$ in the MODE canal (Table 1).

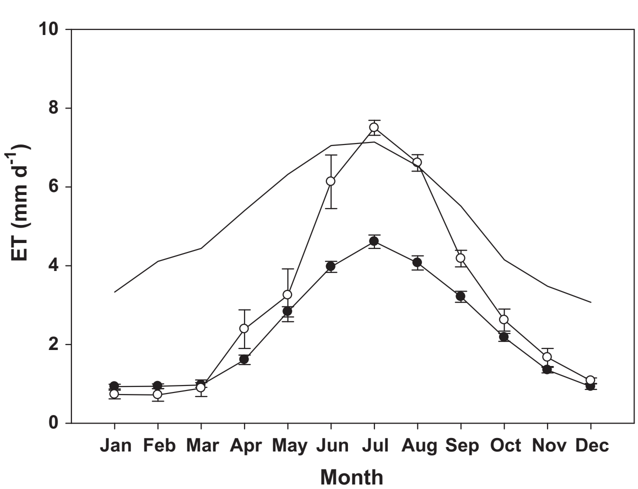

Fig. 5. Mean monthly ET_{RS} in the vegetated portion of the Cienega, 2000–2011, with fire years (2006 and 2011) (open circles) shown separately from non-fire years (closed circles). Error bars are standard errors of means. Solid line shows potential ET.

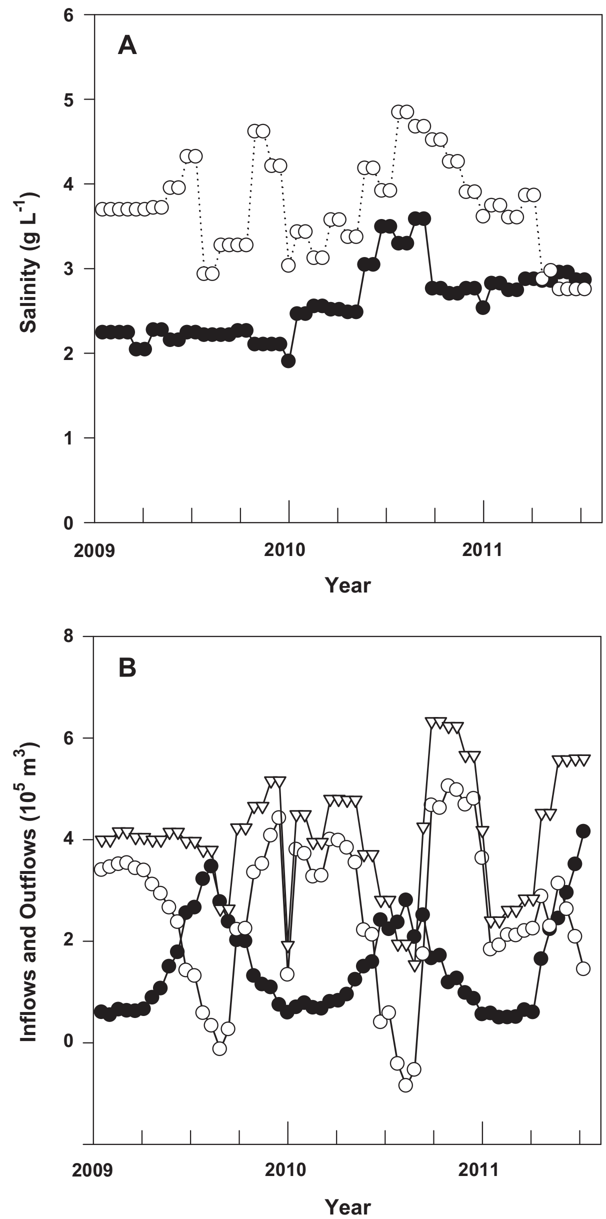

Fig. 6. (A) Measured inflow salinity (closed circles), measured mean salinity (open circles) of water in Cienega, January 2009 through June 2011. (B) Measured inflows (open triangles), ET_{RS} by MODIS EVI (closed circles) and calculated outflows for the same period. Outflows were calculated as inflows minus ET_{RS}.

3.3. Monthly inflow volumes and salinities and projections of outflows

Monthly values of inflow salinity and mean salinity in the Cienega are in Fig. 6A. Inflow salinity was steady at about 2.2 g L^{-1} in 2009, but increased to a high of 3.6 g L^{-1} 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 22 stations in operation.

Fig. 6B shows inflow data for January 2009–June 2011, as well as ET_{RS} and outflows estimated from MODIS data. Outflows were calculated as inflows (MODE flows plus Riito Drain flows plus precipitation) minus ET_{RS}, assuming no change in storage within the Cienega. Inflows tended to be variable on a monthly basis, with brief

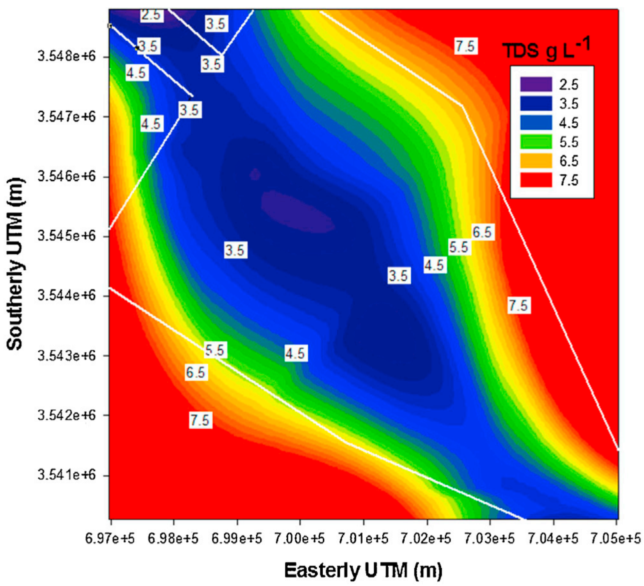

Fig. 7. Contour plot of salinity in the Cienega de Santa Clara based on monthly measurements at 22 stations in the marsh, 2009–2011. The white lines show the approximate area covered by the Cienega.

dips in September 2009 and January 2010, and a more prolonged dip in May–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.

The distribution of salinity within the Cienega, based on mean values at each salinity station over the study plus interpolated values between stations 5 and 10 (see Fig. 1B), is in Fig. 7. Salinity in the main body of the Cienega appeared to be well mixed, with nearly constant salinities between the inflow and outflow ends of the Cienega, but with higher salinities along the periphery of the Cienega. This pattern can be attributed to the presence of the deeper water channel (the Cierro Prieto fault line) tending to produce preferential flow through the middle of the Cienega.

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

Parameters needed to calculate ET by mass balance are in Table 1. Inflows over the 912 d study period averaged 429,408 $m^3\,d^{-1}$. Mean inflow salinity was 2.62 $g\,L^{-1}$ and mean salinity in the wetland was 3.73 $g\,L^{-1}$; hence, ET calculated by Eq. (8) was 2.62 $mm\,d^{-1}$, compared to 2.97 $mm\,d^{-1}$ by the MODIS method. The difference between estimates was not significant ($P > 0.05$) by unpaired *t*-test.

4. Discussion

4.1. *Sources of error and uncertainty in ET_{RS} and ET_{MB} estimates*

Both methods for estimating actual ET in this study are subject to error and uncertainty. Sources of possible error or uncertainty in the ET_{RS} estimate are: the ET algorithm, which is based on an empirical fit of EVI* with ET and meteorological data from other sites and different species of plants than encountered in the Cienega; the estimates of vegetated and open water areas in the Cienega, which can differ by as much as 10% depending on how the perimeter of the wetland is defined; the representativeness of the sampled pixels to the whole Cienega; and the estimate of ET_o from Yuma temperature data and the Blaney–Criddle formula for ET, rather than on-site measurements by the more complete FAO-56 method (Allen et al., 1998; Fisher et al., 2011), which were not collected over the entire study period. A particular problem with the ET_{RS} estimates is the unknown effect of salinity stress on plant transpiration in the Cienega. Eq. (7) assumes that stomatal conductance can be regarded as a constant in daily time step, which can be true for unstressed agricultural crops (Allen et al., 1998), but is not necessarily true for natural stands of plants (Nagler et al., 2009a).

The main sources of uncertainty in ET_{MB} 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; the estimate of the evaporating surface area of the Cienega; and assumptions inherent in the mass balance equations, which assume equilibrium conditions with respect to salts in and out of the Cienega and negligible changes in water storage in the wetland over the study period. Idealized wetland flow models might not capture the variances introduced by heterogeneous vegetation patterns, differences in water depth and areas of preferential flow within the wetland (Kadlec, 2000). The model of Jia et al. (2011) assumes that the water in the wetland is well-mixed. As is expected for large, shallow wetlands with a long residence time of water (Holland et al., 2004), this was true for the main path of water through the Cienega, with salinity at the lowest salinity station (station 10 = 4.02 $g\,L^{-1}$) nearly the same as the mean salinity throughout the marsh (3.73 $g\,L^{-1}$). However, salinity along the periphery of the Cienega was higher than in the path of main flow. The Cienega fit the model of a series of well-mixed zones in the main flow path, which interchange water with shallower side zones that are not in the main path (Kadlec, 2000; Holland et al., 2004). The estimate of wetland salinity as an average of all salinity measurements does not take into account differences in water depth and flow rates at the different stations, and as with ET_{RS}, ET_{MB} should be regarded as an approximation rather than an actual measurement of ET in this study. Hence, the agreement of 13% between ET_{RS} and ET_{MB} should be interpreted with caution, since these methods have inherent errors or uncertainties on the order of 10–30% (e.g., King et al., 2011; Kalma et al., 2008; Glenn et al., 2007, 2010, 2011).

4.2. *Factors controlling ET_{RS} in the Cienega*

Despite the uncertainties inherent in wide-area wetland ET studies (Drexler et al., 2004), the present results help explain some of the discrepancies among studies of other *Typha* marshes. Under optimal conditions after fires, *T. domingensis* ET_{RS} was equal to $ET_{o\text{-}BC}$, 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, CA (Goulden et al., 2007; Bijoor et al., 2011). Annual rates of ET_{RS} and ET_{MB} were lower than $ET_{o\text{-}BC}$ 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 losses that would occur from an open water surface of equal area, even in years of vigorous *Typha* growth.

Water inflow rate during summer was a significant determinant of ET. 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 Fig. 6B shows that in winter water mostly passes through the marsh without supporting ET, with only 54% of inflows normally consumed in ET on an annual basis. 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., in preparation). In summer the Santa Clara Slough is dry except during extreme high tide events.

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 (Glenn et al., 2001). 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 the reductions in flows or increases in salinity will have negative effects on the marsh. The present results show that flows to the Cienega could probably be reduced during the dormant period of *T. domingensis* (November–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 et al., in preparation). Furthermore, winter flows flush excess salts and selenium from the marsh, making it a sustainable ecosystem over time (Garcia-Hernandez et al., 2000). Other wetlands receiving agricultural brine need periodic flushes of fresh water to avoid eventual salinization (Jia et al., 2011). Hence, the sustainability of the Cienega depends on flushing flows that keep the wetland in salt balance, and its connection to the Gulf of Mexico, is the ultimate receiving body for saline outflows.

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