

## Roles of saltcedar (*Tamarix* spp.) and capillary rise in salinizing a non-flooding terrace on a flow-regulated desert river<sup>☆</sup>

E.P. Glenn<sup>a,\*</sup>, K. Morino<sup>b</sup>, P.L. Nagler<sup>c</sup>, R.S. Murray<sup>a</sup>, S. Pearlstein<sup>a</sup>, K.R. Hultine<sup>d</sup>

<sup>a</sup> Environmental Research Laboratory of the University of Arizona, 1601 East Airport Drive, Tucson, AZ 85706, USA

<sup>b</sup> Laboratory of Tree Ring Research, University of Arizona, Tucson, USA

<sup>c</sup> U.S. Geological Survey, Southwest Biological Science Center, Sonoran Desert Research Station, Tucson, AZ, USA

<sup>d</sup> Desert Botanical Garden, Phoenix, AZ, USA

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### ABSTRACT

*Tamarix* spp. (saltcedar) secretes salts and has been considered to be a major factor contributing to the salinization of river terraces in western US riparian zones. However, salinization can also occur from the capillary rise of salts from the aquifer into the vadose zone. We investigated the roles of saltcedar and physical factors in salinizing the soil profile of a non-flooding terrace at sites on the Cibola National Wildlife Refuge on the Lower Colorado River, USA. We placed salt traps under and between saltcedar shrubs and estimated the annual deposition rate of salts from saltcedar. These were then compared to the quantities and distribution of salts in the soil profile. Dense stands of saltcedar deposited  $0.159 \text{ kg m}^{-2} \text{ yr}^{-1}$  of salts to the soil surface. If this rate was constant since seasonal flooding ceased in 1938 and all of the salts were retained in the soil profile, they could account for  $11.4 \text{ kg m}^{-2}$  of salt, about 30% of total salts in the profile today. Eliminating saltcedar would not necessarily reduce salts, because vegetation reduces the upward migration of salts in bulk flow from the aquifer. The densest saltcedar stand had the lowest salt levels in the vadose zone in this study.

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### 1. Introduction

In their natural state, many arid zone rivers have a pulse-flood flow regime, which produces seasonal overbank flooding. Floods refresh the alluvial aquifer, leach salts from the soil and allow mesic vegetation to establish on the terraces (Poff et al., 1997). Many arid zone rivers are now flow-regulated to prevent overbank flooding. No longer do floods or high water tables flush out salts periodically, and salts tend to accumulate in the aquifer and soil profile over time on these rivers (Poff et al., 1997; Jolly et al., 2008).

In the southwestern U.S., many flow-regulated (non-flooding) rivers have come to be dominated by saltcedar (*Tamarix ramosissima* and related species and hybrids) (Gaskin and Schaal, 2002), an imported halophyte shrub from Eurasia (Ungar, 1991; Glenn and Nagler, 2005; Pataki et al., 2005; Nagler et al., 2010). Saltcedar typically extracts water from aquifers as deep as 8–12 m below the soil

surface (Bruehlheide et al., 2010). However, it can also utilize water from the unsaturated zone if the connection with the aquifer is broken (Nippert et al., 2010). Saltcedar actively excretes salts from its leaves via salt glands (Storey and Thomson, 1994), and it has been commonly assumed that saltcedar is a major contributor to the salinization of riparian soils through deposition of exudates and salt-laden leaf litter, resulting in the competitive exclusion of mesic native trees such as cottonwood (*Populus* spp.) and willow (*Salix* spp.) from saltcedar-dominated river terraces (Brotherson and Field, 1987; Di Tomaso, 1998; Zavaleta, 2000). As a corollary, it has sometimes been assumed that removing saltcedar can allow the return of mesic native vegetation to floodplains (e.g., DeLoach et al., 2000).

Only a few previous studies have attempted to quantify the role of saltcedar in salt deposition, however, and these demonstrate complexities in the relationships among saltcedar, salty soil and native plant regeneration (Morris et al., 2009). Ladenburger et al. (2006) reported that soil salinity levels were higher under saltcedar shrubs compared to between shrubs, but that salt levels were not high enough to disrupt the germination of native shrubs and trees at riparian sites in Wyoming, USA. Lesica and DeLuca (2004) also reported elevated salt levels under saltcedar shrubs, but germination of a native grass was actually higher on these soils than on control soils, and they concluded that saltcedar is not allelopathic. Ohrtman (2009)

<sup>☆</sup> Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

\* Corresponding author. Tel.: +1 520 626 2664; fax: +1 520 573 0852.

E-mail addresses: [eglenn@ag.arizona.edu](mailto:eglenn@ag.arizona.edu) (E.P. Glenn), [kmorino@lrr.arizona.edu](mailto:kmorino@lrr.arizona.edu) (K. Morino), [pnagler@usgs.gov](mailto:pnagler@usgs.gov) (P.L. Nagler), [khultine@dbg.org](mailto:khultine@dbg.org) (K.R. Hultine).

reported elevated salt levels under saltcedar plants in both flooding and non-flooding sites on the Middle Rio Grande, New Mexico, USA, but salt levels were paradoxically lowest under the oldest saltcedar plants, and were not so high that restoration with native vegetation would be inhibited by salinity. Yin et al. (in press), working in the native range of saltcedar in Asia, reported that salt levels were higher in saltcedar mounds than between mounds, but that fertility factors such as potassium, organic matter and phosphorous were also higher, and that saltcedar mounds were overall a net positive factor in rehabilitating soils for restoration.

The above studies looked only at the role of saltcedar leaf exudates and litter in salinizing riparian areas. However, other processes can also contribute to salinization. Hydraulic lift of saline groundwater by salt-tolerant vegetation can deposit salts in the vadose zone, for example (Armas et al., 2009). Another source of salts in riparian soils is capillary rise of water and salts from the aquifer into the vadose zone (Gillham, 1984; Silliman et al., 2002). The capillary fringe can extend to 1–3 m above the water table, with fine textured soils supporting greater rise than sands (Gillham, 1984; Gerla, 1992; Kuo, 1999). Salts can accumulate to high levels in this zone due to the evaporation of water from the top of the capillary fringe or soil surface and deposition of salts in the vadose zone (Doering et al., 1964; Gerla, 1992; Costelloe et al., 2009; Grunberger et al., 2008). In fact, this is one mechanism by which salt flats are formed (Kinsman, 1969). Evapotranspiration (ET) by deeply rooted vegetation can counter this effect, by limiting the rise of water and salts to the soil surface through evapotranspiration (ET) (e.g., Nosetto et al., 2007).

The present study examined the distribution of salts and soil moisture as a function of soil depth, texture and distance from the river on two saltcedar-dominated river terraces at Cibola National Wildlife Refuge (CNWR), California, on the Lower Colorado River (Fig. 1). The terraces at CNWR were formerly flooded via a series of overflow arms and backwaters of the river, but overbank flooding was markedly curtailed after 1938, when Hoover Dam was completed (Olmsted and McDonald, 1967). The possibility of floods was further curtailed after 1964 when the main river flow at CNWR was diverted from the natural river channel into an engineered channel surrounded by flood control levees (Fig. 1) (United States

Bureau of Reclamation, 1996; 2008). The new channel supports flows of ca. 200–400 m<sup>3</sup> s<sup>-1</sup>, whereas the old channel now carries only 8–15 m<sup>3</sup> s<sup>-1</sup> to support wildlife in the refuge (U.S. Bureau of Reclamation, 2008). Since 1938 the vegetation on the terraces has converted from a mix of mesic and salt-tolerant plants to saltcedar-dominated stands growing with native salt-tolerant plants such as quailbush (*Atriplex lentiformis*), arrowweed (*Pluchea sericea*) and stunted mesquites (*Prosopis* spp.) (Nagler et al., 2009).

We used this site to study the effects of saltcedar and capillary rise on the salt balance of a non-flooding river terrace. We placed salt traps under saltcedar bushes to measure the rate of salt deposition from saltcedar leaf exudates and litter-fall, and took soil cores and aquifer samples to quantify the amount of salts present in the aquifer and vadose zone as a function of soil depth. Due to scarcity of floods and very low precipitation, our hypothesis was that most of the salts originating from saltcedar, capillary rise and other processes since 1938 would still be present in the profile, allowing us to elucidate the biological and physical processes responsible for salt deposition, and to create an approximate mass balance of sources of salt deposition on a non-flooding river terrace.

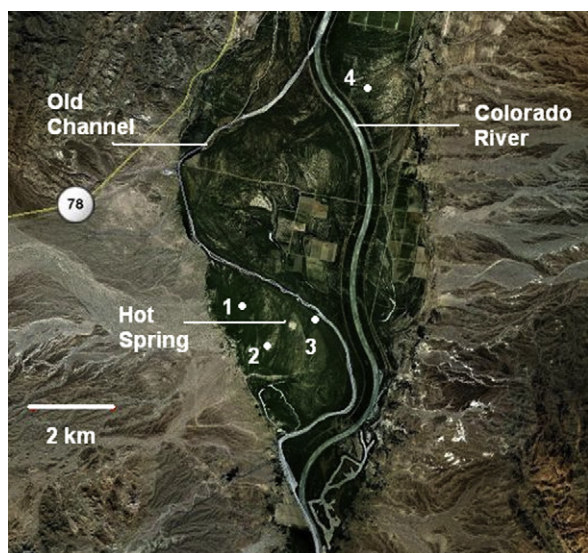
## 2. Methods

### 2.1. Study sites

CNWR is in an extreme arid environment in the Sonoran Desert. Mean annual precipitation is 100 mm yr<sup>-1</sup>, and temperatures range from lows of 4 °C in January to highs of 45 °C in June and July. Weather data from 1938 to 2010 were obtained from the NOAA Cooperative Station at the Blythe Airport for the years 1935–2006 (Station 040924).

(<http://www.wrcc.dri.edu/cgi-bin/cliGCStPl?ca0927> (040924) [http://weather-warehouse.com/WeatherHistory/PastWeatherData\\_BlytheRiversideCntyArpt\\_Blythe\\_CA\\_August.html](http://weather-warehouse.com/WeatherHistory/PastWeatherData_BlytheRiversideCntyArpt_Blythe_CA_August.html)). Data after 2006 were from the Parker, Arizona AZMET station (AZMET, 2011). Potential evapotranspiration were calculated from annual mean temperature and latitude using the Blaney–Criddle formula (Brouwer and Heibloem, 1986).

The primary sampling sites (Fig. 1) are three locations where saltcedar ET was measured by sap flux sensors and Bowen ratio moisture flux towers which were used to measure ET in previous studies (Nagler et al., 2008; 2009; Chatterjee, 2010; Zhu et al., 2011; Taghvaeian, 2011). The sites were chosen to represent a gradient of saltcedar stand density and distance from the active river channel. They were given fanciful names based on summer working conditions. Slitherin was located in a very dense saltcedar stand approximately 750 m from the river; Swamp was in a medium dense stand 200 m from the river; and Diablo was in a less dense stand 1500 m from the river. Leaf area index (LAI) values were 4.6 at Slitherin, and 3.4–3.5 at Swamp and Diablo (Nagler et al., 2008; 2009). At each site five observation wells were drilled from the soil surface to the aquifer and fitted with a PVC casing to measure depth and salinity of the aquifer at monthly intervals and soil moisture in the vadose zone using a neutron hydroprobe (methods described in detail in Nagler et al., 2008; 2009). One well was located near the moisture flux tower and the other four were located at the corners of a 100 m × 100 m square around the central well. Soil moisture data were collected monthly, June to August, 2006, as reported in Nagler et al. (2008) and they are repeated here to put the salinity data in context. Additional soil moisture measurements were conducted at Swamp on June 8 and 9 to test for diurnal variations in vadose zone moisture content characteristic of hydraulic lift (Caldwell et al., 1998). Moisture readings were taken at each of the 5 observation wells at Swamp at 9 am and 3 pm on June 8 and at



**Fig. 1.** Locator map of sites for soil salinity measurements: 1) Slitherin; 2) Diablo; 3) Swamp; 4) Burro. Also shown is an area where geothermal water approaches the surface (Hot Spring); the old channel of the river; and the new channel of the Colorado River carrying most of the flow since 1964.

12 noon and 3 pm, June 9 (plus or minus 30 min due to the time needed to travel between wells and take measurements).

A secondary site named Burro was established on a different terrace of the river (Fig. 1), in a former agricultural field that had been colonized by saltcedar. This was a medium dense stand (LAI = 3.0) approximately 500 m from the river.

## 2.2. Comparison of floodplain vegetation and river flows, 1938–2008

We tested our assumption that this has become a non-flooding river terrace by comparing archival aerial photographs (Norman et al., 2006) acquired in 1938, the year Hoover Dam was completed, with Quickbird high-resolution (30 cm) satellite imagery acquired in 2008 (Digital Globe, Inc.). Soil, water and biotic characteristics of each site are in Table 1. The black-and-white 1938 aerial photographs had approximately 2 m resolution, allowing individual tree canopies and geomorphological features on the floodplain to be distinguished. Flow rates for the Colorado River at Parker, Arizona (above CNWR) were obtained from the USGS Water Information System website (<http://waterdata.usgs.gov/nwis>). The history of levee and channel construction at CNWR are briefly described in United States Bureau of Reclamation (1996). We also compared rates of precipitation and potential evapotranspiration ( $ET_0$ ) from 1935 to 2010 to determine the extent to which flushing due to rain events had occurred on the terrace.

## 2.3. Soil sampling and analyses

Soil cores were augered near each observation well at Diablo, Swamp and Slitherin in 2007. Soil samples were collected for analysis at each soil layer encountered during drilling. Soil layers were identified by color changes and differences in soil texture between layers. Soil texture was determined using a field method based on physical characteristics (e.g., ability to form ribbons) (<http://www.osha.gov/doc/outreachtraining/htmlfiles/soiltex.html>). Samples from each layer were tested in the laboratory for electrical conductivity (EC) and pH of a saturated extract using methods in U.S. Salinity Laboratory Staff (1954). Soils were first oven-dried, then placed in a beaker and water was stirred in small increments to bring the soil to the saturation point, defined as having a glistening appearance on standing but with no free water on the surface. The amount of water required to achieve saturation ( $\text{g g}^{-1}$ ) was recorded for each sample. The saturated extract was collected by vacuum filtration and tested for EC and pH, with KCl used as a salinity

standard. For this study, reported EC values were expressed in units of g NaCl per kg soil, with EC ( $\text{dS m}^{-1}$ ) converted to NaCl equivalents based on the saturation percentage of water in the extract, and converting EC from KCl to NaCl equivalents using an empirically determined relationship between EC and NaCl determined with KCl and NaCl standard solutions in the laboratory (e.g.,  $1 \text{ dS m}^{-1} = 0.68 \text{ kg L}^{-1} \text{ KCl}$ ,  $0.79 \text{ g L}^{-1} \text{ NaCl}$ ). Percent sand, silt and clay were determined by the hydrometer method (Gee and Bauder, 1986). Soil sampling and laboratory testing were carried out by Earl Burnett (U.S. Bureau of Reclamation, Denver, CO).

Additional soil samples were taken at each primary site as well as Burro in July, 2009, for analysis of individual cations and anions. Two locations at Diablo were sampled, designated Diablo Tower and Diablo Southwest (SW). Five samples each were collected directly underneath saltcedar shrubs and in bare soil between shrubs at 5 cm and 50 cm soil depth at each site, then pooled across samples for analysis of  $\text{Na}^+$ ,  $\text{K}^+$ ,  $\text{Mg}^{+2}$ ,  $\text{Ca}^{+2}$ ,  $\text{Cl}^-$ , and  $\text{SO}_4\text{--S}^{-2}$  by an analytical lab (IAS Laboratories, Inc., Tempe, AZ) using ICP for cations, an ion-specific electrode for  $\text{Cl}^-$ , and the turbidimetric method for  $\text{SO}_4\text{--S}^{-2}$  (Clesceri et al., 1998). At the same sites that soil samples were taken under shrubs, five samples of surface leaf litter and fresh saltcedar leaves were collected, dried to constant weight in a solar drier, then pooled and analyzed the same as the soil samples. Litter samples were from  $0.0675 \text{ m}^2$  of ground area and fresh leaf samples were grab samples from trees. Water samples were collected from the central observation well at each site and analyzed for the same cations and anions as the other samples.

## 2.4. Salt traps

Saltcedar plants were observed to guttate saline water during the night, which then coated leaves with salt when the water dried during the day. These salts were easily dislodged from the leaves by wind and were observed to accumulate on the soil surface under shrubs. We quantified salt exudate using salt traps, in a method modified from those used to estimate leaf area index (LAI) from deciduous trees based on leaf traps placed under canopies (e.g., Ishihara and Hiura, 2011). This method assumes that bucket contents represent a fair sample of salts dislodged from the same area of canopy over the buckets. Salt traps were placed out at Burro, Diablo Tower, and Diablo SW. Traps were 30.5 diameter white 20 L plastic buckets, with a thin layer of sand placed in the bottom to prevent salts from being blown from buckets between collection intervals. Bucket contents (sand plus salts) were collected at monthly intervals from July 17 to October 17, 2009, and quantified in the laboratory by measuring EC of a 1:2 (weight:weight) extract using a laboratory EC m calibrated with NaCl standards. Control measurements (sand only) had very low EC readings compared to field samples, and sand EC values were subtracted from sand plus salts measurements to account for the small contribution of sand to EC measurements. At each site, five buckets were placed under saltcedar canopies and five were placed in open areas between plants. At Burro, five buckets were also placed under mesquite trees that were present at the site.

## 2.5. Calculations and statistical analyses

Total salts in the soil cores were calculated based on analyses of EC at each soil depth, by expressing salt in  $\text{g kg}^{-1}$  based on NaCl calibration of the EC readings; results were then converted to  $\text{kg m}^{-3}$  soil, using 1.4 as the bulk density of soil (Nagler et al., 2008). Salt content of each layer in  $\text{kg m}^{-3}$  was converted to  $\text{kg m}^{-2}$  by dividing by the depth of the soil layer, which varied among the soil layers at each site. Then total salts on a ground area basis ( $\text{kg m}^{-2}$ )

**Table 1**

Soil and aquifer properties within stands of *Tamarix ramosissima* at three sites in Cibola National Wildlife Refuge. Soil properties are means of samples (and the numbers in parentheses represent standard error of the means) taken in different soil layers in the vadose zone above the aquifer ( $n = 5$  auger holes per site). pH was measured in saturated paste extracts. NaCl was calculated from EC in saturated paste extracts based on the depth of each strata and using a bulk density of 1.4 (Nagler et al., 2008) to convert NaCl per gram of soil to NaCl to  $\text{m}^2$  of ground surface summed over all strata. Also shown are the leaf area index (LAI) of saltcedar at each site.

Parameter	Swamp	Slitherin	Diablo
Soil pH	8.4 (0.04)	8.0 (0.07)	8.1 (0.04)
Soil NaCl ( $\text{kg m}^{-2}$ )	33.6	14.3	23.4
% Sand	48.3 (5.7)	45.2 (5.0)	54.2 (6.2)
% Silt	38.6 (3.5)	39.2 (3.4)	31.8 (4.3)
% Clay	13.1 (1.7)	15.6 (1.8)	14.0 (4.3)
Distance from River (m)	200	750	1500
Aquifer Salinity ( $\text{g L}^{-1}$ )	1.95 (0.09)	2.36 (0.24)	7.75 (0.09)
Depth to Aquifer (m)	2.7–3.4	3.7–4.0	3.4–3.7
LAI Saltcedar	3.48 (0.19)	4.59 (0.16)	3.40 (0.18)



were calculated by summing up the salts present in each layer from the surface to the top of the aquifer at each site.

Salts in exudates were also expressed as  $\text{kg m}^{-2}$  ground area based on the diameter of the collecting buckets. Total exudate over the 2008 growing season was calculated by summing the monthly values, July–October, then multiplying by 1.5 to account for the missing May and June periods (plants leaf out in April). Salts in leaf samples were expressed on an area basis by totaling up individual cations and anions, then dividing by areal coverage of the leaf sample, calculated by using LAI and specific leaf area values determined in a previous study (Nagler et al., 2009). Salts in litter were expressed as a rate of annual deposition based on their cation and anion contents and assuming that the annual crop of leaves in  $\text{g m}^{-2}$  was shed as litter each year. Finally, results were projected over wider areas by multiplying values measured under shrubs and between shrubs by the fractional cover of shrubs (0.78) and soil (0.22) determined for Diablo in a previous study (Nagler et al., 2009).

Distribution of cation and anion contents of soil samples were tested for normality of distribution by the Shapiro–Wilk's test (Shapiro and Wilk, 1965). Both raw and log-transformed values were not normally distributed ( $P < 0.05$ ), so the non-parametric Kruskal–Wallis test was used to analyze treatment effects (Kruskal and Wallis, 1952). Three one-way Kruskal–Wallis analyses of variances (ANOVA) were conducted, in which sample locations (Under or Between Shrubs), soil depth (5 cm or 50 cm) and sample site (Swamp, Slitherin, Diablo Tower, Diablo SW and Burro) were categorical variables.

### 3. Results and discussion

#### 3.1. River flows, vegetation and geomorphology of the floodplain, 1938–2010

River flows showed just a few large peaks that would indicate flood conditions from 1938 to 2010 (Fig. 2A). Flows were more variable from 1938 to 1964 than in the period after that, due to completion of Glenn Canyon Dam (upriver from Hoover Dam) in 1964, which allowed more precise control of river flows by river managers. The only large flood event on the river occurred 1983–1988 due to large releases from Lake Mead following an El Niño cycle. These flows caused damage at several points along the river, but no overbank flooding was reported in the CNWR river reach, which was protected by levees after 1964 (United States Bureau of Reclamation, 1996; 2008).

Rainfall and  $\text{ET}_0$  from 1935 to 2010 are compared in Fig. 2B. Annual rainfall never exceeded 230 mm, whereas annual  $\text{ET}_0$  ranged from 1600 to 2100 mm, 8–12 times greater than precipitation. Based on sap flux measurements in 2008–2009, actual ET ( $\text{ET}_a$ ) on the terrace is equal to 0.47  $\text{ET}_0$  (Nagler et al., 2009). The maximum recorded precipitation over 24 h was 7.65 cm (August 27, 1951) and the maximum monthly precipitation was 15 cm in the same month and year. These precipitation events could have saturated the soil to a maximum depth of approximately 26 cm and 50 cm, respectively, based on a saturation percentage of 30% as determined in soil analyses. Even these extreme wetting events would not be expected to leach salts through the entire profile, which is approximately 3 m deep, the top 50 cm of which is normally dry (see Fig. 4). Water arriving as precipitation would be expected to rapidly evaporate or be used to support plant transpiration based on  $\text{ET}_0$  and  $\text{ET}_a$  values. Vegetated desert landscapes receiving 100–125  $\text{mm yr}^{-1}$  rainfall generally have rates of downward water or salt flux under 1  $\text{mm yr}^{-1}$  (Scanlon and Goldsmith, 1997).

Geomorphology and vegetation patterns on the floodplain are compared in Fig. 3. The same ephemeral channels and vegetation

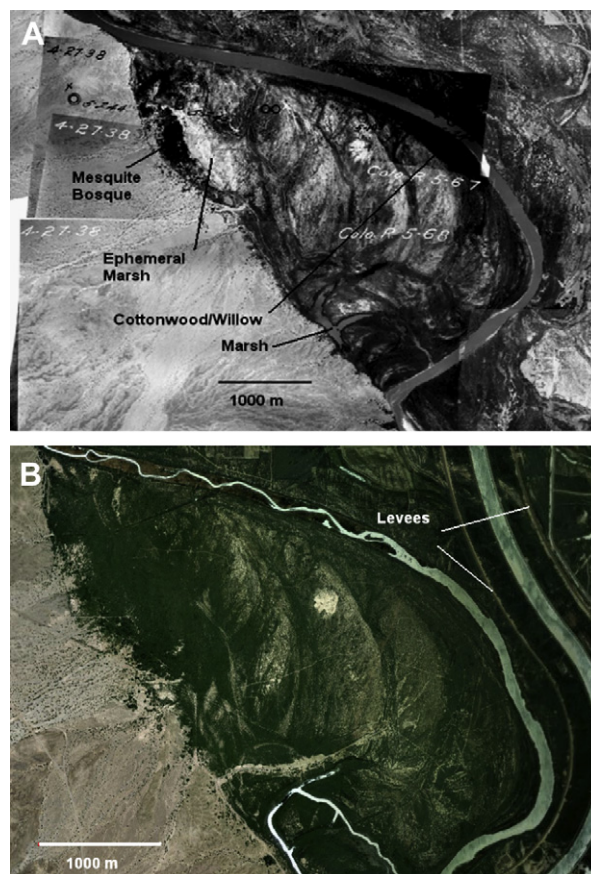


Fig. 2. Comparison of the floodplain at Cibola National Wildlife Refuge in 1938 (Top Panel) and 2008 (Bottom Panel).

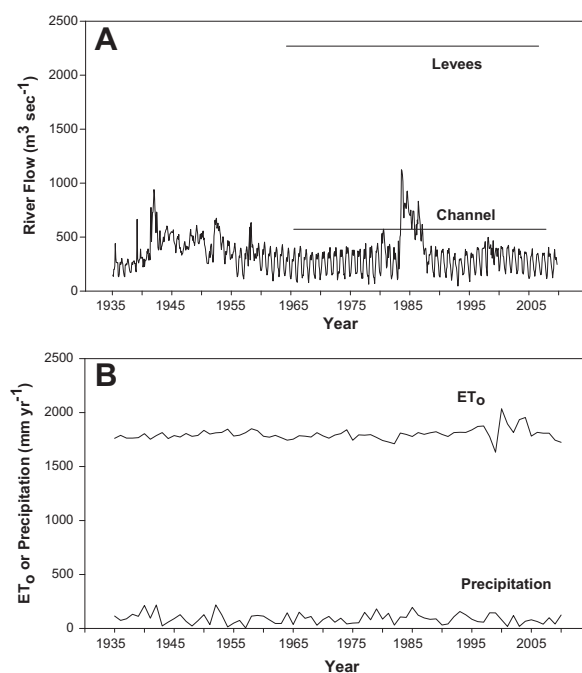
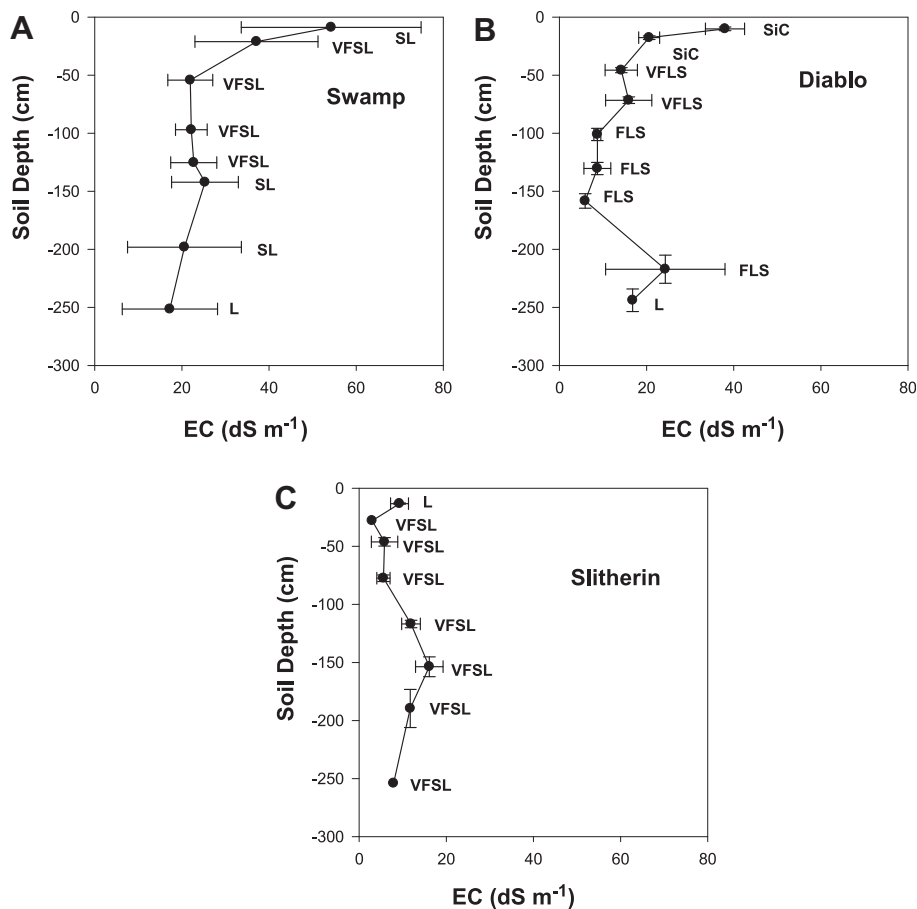


Fig. 3. (A) Flows at Cibola based on Parker USGS gauging stations, 1935–2010. Horizontal lines show the carrying capacity of the engineered main channel and levees completed in 1964. (B) Annual precipitation and potential evapotranspiration at Cibola 1935–2010.



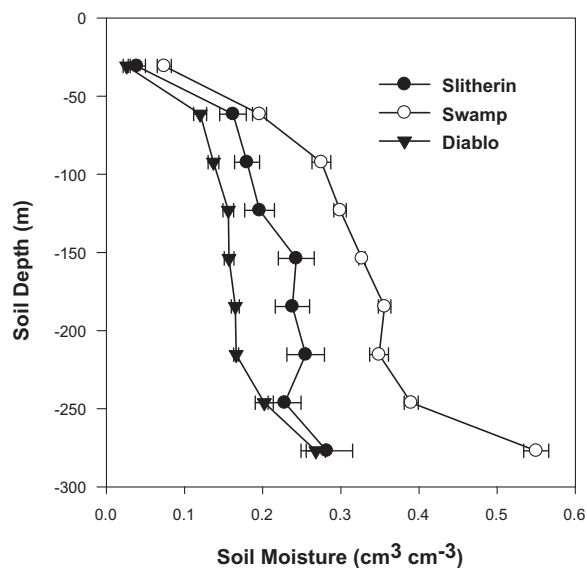
**Fig. 4.** Electrical conductivity (A–C) of saturated soil extracts at different soil depths at three sites (Swamp, Diablo and Slitherin) in Cibola National Wildlife Refuge. Error bars are standard errors. Letters denote soil texture classes at each depth (L = Loam; Si = Silt; S = Sand; C = Clay; F = Fine; VF = Very Fine).

patterns evident in 1938 were still evident in 2008, indicating a lack of major flood flows that would have reworked the alluvial soils and channels. A bare spot caused by seepage of geothermal water onto the floodplain was nearly identical in appearance on 1938 and 2008 images, and large dead cottonwood and mesquite tree trunks left over from the pre-1938 mesic period were still present on parts of the floodplain in 2008. A notable change in vegetation types took place, however. Nagler et al. (2009) showed that a fringe of cottonwood and willow trees up to 200 m wide occurred along the river channel in the area now occupied by saltcedar (as at Swamp). Further back, ephemeral channels were lined with native trees and large individual trees (probably mesquites) dotted the floodplain between channels. This area is now occupied mainly by saltcedar and arrowweed, interspersed in places with scattered, stunted mesquites and saltbushes (Nagler et al., 2009).

The area around the Slitherin site was apparently an ephemeral marsh area in 1938 but became occupied by saltcedar by 2008, due to lack of seasonal flows to fill the marsh. Marsh areas in the southern portion of the floodplain are still present, as they are maintained by managed flows from the old channel into the marsh areas. Although salt-tolerant vegetation undoubtedly occurred on this terrace in 1938, saltcedar was not yet widely present on the Lower Colorado River (Clover and Jotter, 1944) and the saline sites were probably dominated by native halophytes.

We concluded that the floodplain has not been reworked by floods in any major way since 1938, and that salts present in the soil might reflect the history of salt deposition by saltcedar and capillary rise from the aquifer since 1938. We recognized that precipitation, although low, will tend to leach surface salts downward, and salts

from the aquifer can reach the soil surface under some conditions, complicating observed patterns of salt distribution in the soil. Hence, we recognized that even though salts from capillary rise were deposited originally in the lower part of the profile, and salts from



**Fig. 5.** Soil moisture content measured by neutron hydroprobe at three sites in Cibola National Wildlife Refuge. Values are the means and standard errors of monthly measurements at five stations at each site, June–August, 2006.

saltcedar were deposited on the soil surface, significant mixing undoubtedly occurred over the years.

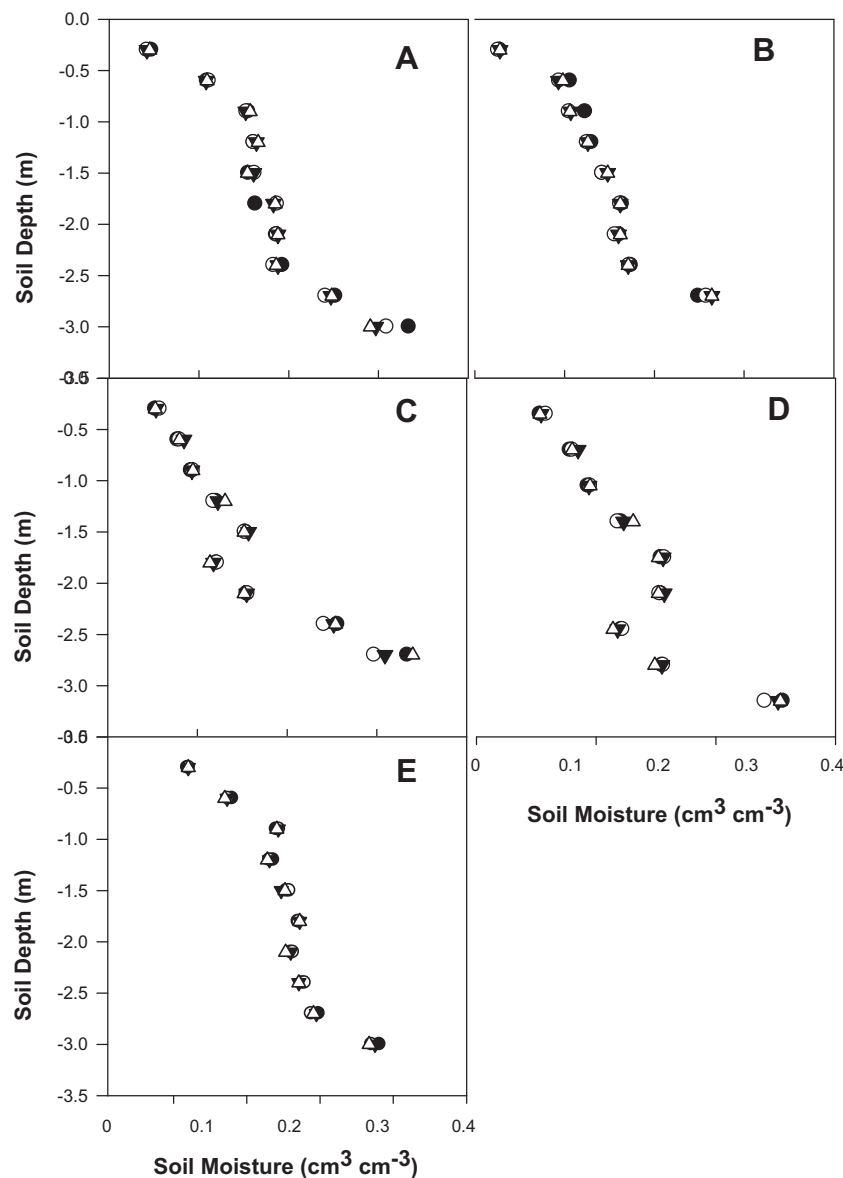
### 3.2. Soil salt levels at different soil depths

Mean soil and aquifer properties at the three main sites are in Table 1. Vadose zone soils pooled across depths were all classified as silt loams. Swamp, nearest the river, had higher soil salt levels as measured by EC than Slitherin or Diablo, but the aquifer was most saline at Diablo. EC was negatively correlated with % sand ( $r = -0.513$ ,  $P < 0.001$ ) and positively correlated with % silt ( $r = 0.592$ ,  $P < 0.001$ ) and % clay ( $r = 0.260$ ,  $P < 0.05$ ) across sites. As noted in previous studies (Nagler et al., 2008; 2009), aquifer salinity increased with distance from the river channel across sites.

Soil texture classes, EC values at different soil depths based on auger samples and hydroprobe readings are in Fig. 4. Each site contained eight to nine distinct soil layers with different textural properties, apparently corresponding to past depositional events when

flooding still occurred on the floodplain. Soil profiles at Swamp (Fig. 4A) and Diablo (Fig. 4B) showed a marked increase in EC near the soil surface. This pattern is what is expected in soils over shallow, saline aquifers (Doering et al., 1964; Costelloe et al., 2009). Salts move to the soil surface by bulk transport in water and accumulate in the top 30 cm as water evaporates. Finer soils such as silts and clays support a greater height of capillary rise than sand, and are less likely to be leached by precipitation events, explaining why EC was negatively correlated with % sand and positively correlated with % silt and % clay. Eventually, an equilibrium can be reached due to the downward diffusion of salts along the steep concentration gradient between the surface and the deep soil layers (Doering et al., 1964).

Slitherin showed a different pattern (Fig. 4C), with relatively low ECs throughout the profile and no accumulation of salts at the soil surface. Saltcedar at Slitherin had the highest LAI and rates of ET (Nagler et al., 2009) of the sites. It is possible that saltcedar controlled the upward movement of salts and water by shading the soil surface and extracting water from the vadose zone for transpiration, as



**Fig. 6.** Soil moisture contents measured by neutron hydroprobe at Swamp at 9:00 am (closed circles) and 3 pm (open circles) on June 8 and 12 noon (closed triangles) and 3 pm (open triangles) on June 9, 2006. Panels show individual readings at each of 5 probe ports at the site, with ports located 100 m apart in a square pattern and an additional port in the center of the square (A = southwest corner; B = center; C = northwest corner; D = northeast corner and E = southeast corner).

Nosetto et al. (2007) noted for deep-rooted oak trees growing in naturally-saline soils in Hungary.

### 3.3. Soil moisture content at different soil depths

All sites had a layer of moist soil above the aquifer, extending to within 50 cm of the soil surface, but the surface soils were dry at all sites during our sampling events (Fig. 5). These profiles are also what are expected for soils over a shallow aquifer, as water rises by capillarity into the vadose zone then evaporates as it approaches the soil surface (Doering et al., 1964; Gerla, 1992; Costelloe et al., 2009). Gillham (1984) showed that silt loams typical of these sites can support a capillary rise of 2–3 m above the aquifer. Grunberger et al. (2008) showed a linear increase of soil moisture content with increasing depth in a fallow paddy field over a saline aquifer at 1.6 m depth, and a marked increase in salt content in the top 15 cm of soil depth, similar to the patterns in Figs. 4 and 5.

Other studies (Zhu et al., 2011; Taghvaeian, 2011) showed that Swamp, the site closest to the river, experiences a fluctuating depth to water matched to the river stage. Moisture content in the vadose zone was much higher at Swamp than at Slitherin or Diablo. Proximity to the river and lower depth to the water table contribute to the higher vadose zone moisture content at this site compared to Slitherin and Diablo.

We considered the possibility that vadose zone water could also have originated from hydraulic lift by saltcedar roots at these sites. Plants that exhibit hydraulic lift take up water from the aquifer or deep soil layers at night, and deposit it in shallower soil layers via a water potential gradient in the root zone (Naumburg et al., 2005); they then consume this water during the day to support transpiration (Caldwell et al., 1998). However, soil moisture profiles were static rather than dynamic at all sites, varying by less than 5% among monthly measurements over a summer (Nagler et al., 2008). Hydroprobe measurements of soil moisture content at different depths were also constant over the course of two days (illustrated for the five probe ports at Swamp in Fig. 6). By contrast, a pattern of depletion during the day and recharge during the night is expected from hydraulic lift (Caldwell et al., 1998). We concluded that the vadose zone water was most likely maintained by capillary rise rather than hydraulic lift.

The concentration of salts in the soil solution in the capillary fringe was calculated from the moisture content and salt content of

the soil from 50 to 250 cm soil depth (data in Fig. 4). Calculated salinities are 17 g L<sup>-1</sup> at Slitherin, 41 g L<sup>-1</sup> at Swamp, and 35 g L<sup>-1</sup> at Diablo. The value at Slitherin is low enough to support growth and ET, whereas the values at Swamp and Diablo are near the tolerance limit of saltcedar (Glenn et al., 1998). Based on ECs of saturated extracts, the vadose zone salinities would be classified as strongly saline (Slitherin) to very strongly saline (Diablo and Swamp) throughout their profiles (Abrol et al., 1988).

### 3.4. Cation and anion content of leaves, litter, the aquifer and soils

Cations and anions in aquifer, leaf and litter samples are in Table 2, and mean soil values across sites are in Table 3. Saltcedar leaves and litter were high in salts at all sites, ranging from 100 to 175 g kg<sup>-1</sup>. Na<sup>+</sup> and Cl<sup>-</sup> were the dominant cations and anions in aquifer, leaf and litter samples, accounting for over 75% of total on in all samples. Soil salts at the 5 cm depth were also dominated by Na<sup>+</sup> and Cl<sup>-</sup>, but Ca<sup>2+</sup> and SO<sub>4</sub>-S<sup>2-</sup> were the dominant ions at the 50 cm depth (Table 3). All cations and anions except Mg<sup>2+</sup> and SO<sub>4</sub><sup>2-</sup> were significantly higher at 5 cm than at 50 cm ( $P < 0.05$ ). On the other hand, cation and anion values were not significantly different under shrubs compared to between shrubs ( $P > 0.05$ ) and differences among sites were non-significant ( $P > 0.05$ ).

Results are displayed in the form of a Stiff diagram (Stiff, 1951) in Fig. 7. Leaf and litter samples had higher levels of K<sup>+</sup> than aquifer samples, indicating active uptake of this nutritional ion by saltcedar plants, but they otherwise were similar to aquifer samples. Soils at the 5 cm depth also had similar elemental ratios as aquifer samples, except that they were much lower in SO<sub>4</sub>-S<sup>2-</sup>. This was likely due to the formation of sparingly-soluble gypsum (CaSO<sub>4</sub>) in 1:1 soil:water extracts, since Ca<sup>2+</sup> was abundant in soils (Doering et al., 1964). On the other hand, soil samples at 50 cm had comparatively low levels of Na<sup>+</sup> and Cl<sup>-</sup>, and were dominated by Ca<sup>2+</sup> and Mg<sup>2+</sup>. Results for aquifers, leaf and litter samples, indicate a common origin of Na<sup>+</sup> and Cl<sup>-</sup> from the aquifer.

### 3.5. Salt deposition rates in leaf litter and exudates, and calculated salt content in the transpiration stream

Salt traps placed under saltcedar shrubs showed that 0.13–0.18 kg m<sup>-2</sup> of salts were exuded from leaves over the growing season, and an additional 0.03–0.04 kg m<sup>-2</sup> were deposited in leaf litter at the end of the growing season (Fig. 8). Salt traps placed in bare soil between shrubs or under mesquite trees had lower rates of deposition, averaging 0.06 and 0.04 kg m<sup>-2</sup>, respectively. The calculated annual salt deposition, based on 0.78 fractional cover (Nagler et al., 2009), was 0.159 kg m<sup>-2</sup> yr<sup>-1</sup>. The total deposition from 1938 to 2010 (72 years), assuming a constant rate of deposition, would be

**Table 2**

Cation and anion contents (mmol kg<sup>-1</sup>) of aquifer, leaf and litter samples, and total salts (kg kg<sup>-1</sup>) from saltcedar sites in Cibola National Wildlife Refuge. Total salts are the sum of the individual cations and anions.

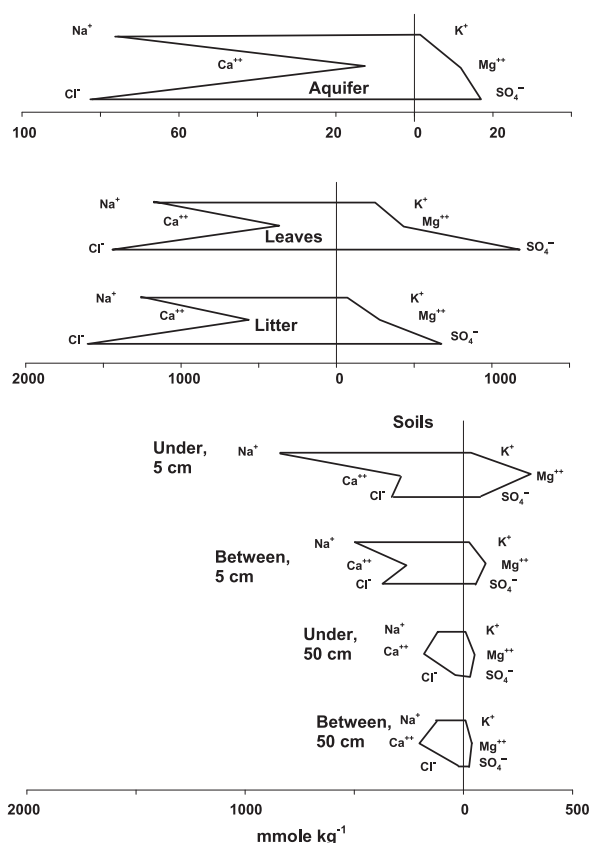
	Na <sup>+</sup>	K <sup>+</sup>	Ca <sup>++</sup>	Mg <sup>++</sup>	Cl <sup>-</sup>	SO <sub>4</sub> -S <sup>2-</sup>	Total Salts
<i>Slitherin</i>							
Aquifer	21.8	0.14	4.50	2.96	16.6	7.19	0.00154
Leaves	1409	256	450	458	1729	1434	0.175
Litter	1454	76.9	550	188	2414	684	0.167
<i>Swamp</i>							
Aquifer	30.0	0.31	4.00	3.29	19.1	8.75	0.00186
Leaves	773	231	275	458	1377	641	0.115
Litter	500	76.9	525	213	774	684	0.0889
<i>Diablo Tower</i>							
Aquifer	105	0.51	24.5	21.7	137	26.0	0.00943
Leaves	1045	205	325	541	2014	1444	0.173
Litter	—	—	—	—	—	—	—
<i>Diablo SW</i>							
Aquifer	100	0.48	20.3	17.9	129	21.2	0.00882
Leaves	818	231	375	408	1771	1050	0.147
Litter	1318	76.9	525	233	2600	994	0.181
<i>Burro</i>							
Aquifer	118	0.36	10.5	10.4	114	19.1	0.00788
Leaves	1818	282	400	321	311	1291	0.127
Litter	1772	51.3	650	500	586	341	0.110

**Table 3**

Mean values and standard errors (in parentheses) for cations and anions in soil samples taken under (U) or between (B) saltcedar bushes at 5 cm and 50 cm soil depths. Unites are mg kg<sup>-1</sup>.  $P$  values are probability of significant treatment differences comparing under with between, 5 cm and 50 cm, and by sites by the Kruskal–Wallis test. Significance levels are noted if  $P < 0.05$  (\*) or  $P < 0.01$  (\*\*).

Analyte	5 cm Under	5 cm Between	50 cm Under	50 cm Between	P U/B	P 5/50	P Site
Na <sup>+</sup>	843 (196)	499 (263)	124 (36)	118 (43)	0.914	0.039*	0.147
K <sup>+</sup>	32 (3.1)	20 (2.2)	9.0 (1.6)	7.4 (0.7)	0.871	0.002**	0.766
Ca <sup>2+</sup>	295 (18)	265 (39)	200 (18)	184 (18)	0.870	0.022*	0.420
Mg <sup>2+</sup>	106 (46)	97 (64)	34 (5.0)	52 (18)	0.785	0.140	0.139
Cl <sup>-</sup>	314 (22)	371 (154)	26 (5)	41 (15)	0.233	0.002**	0.593
SO <sub>4</sub> -S <sup>2-</sup>	74 (20)	51 (13)	40 (8)	25 (8)	0.623	0.140	0.285
Total Salts	95 (51)	39 (14)	14 (2)	14 (2)	0.664	0.009**	0.460



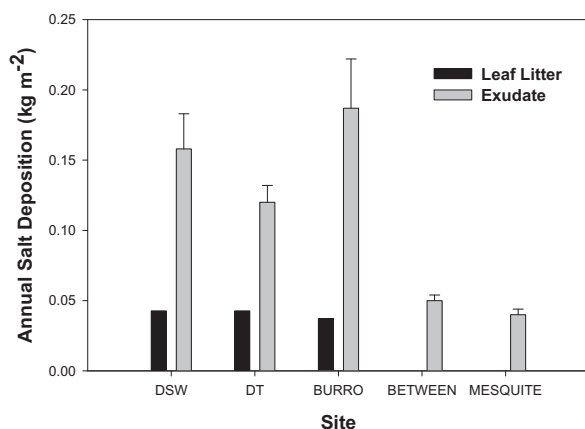


**Fig. 7.** Stiff Diagram showing ratios of cations and anions in leaves, litter and soils averaged over five sites at the Cibola National Wildlife Refuge on the Colorado River. Soil samples were taken under or between saltcedar shrubs at 5 cm and 50 cm soil depths. Units are  $\text{mg kg}^{-1}$ .

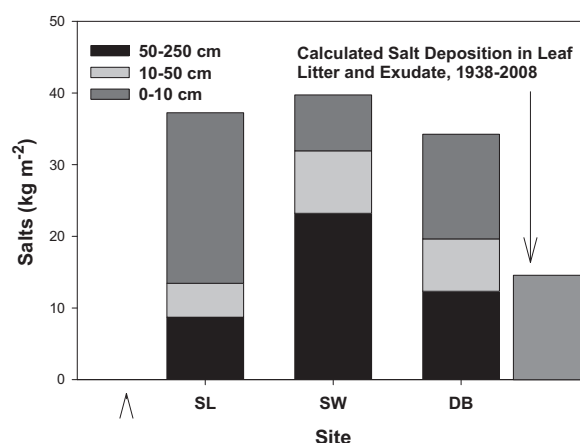
$11.4 \text{ kg m}^{-2}$ , compared to total salt loads of  $36\text{--}40 \text{ kg m}^{-2}$  in the soils today (Fig. 9).

### 3.6. Comparison of saltcedar to other halophytes

Likely replacement species for saltcedar on most of these terraces would be other halophytes, due to the high salinity of the soils and aquifers. Halophytes like saltcedar exclude most of the salts in the soil solution from their transpiration stream, but take up small amounts for osmotic adjustment of cell vacuoles in stem and leaves (Glenn and

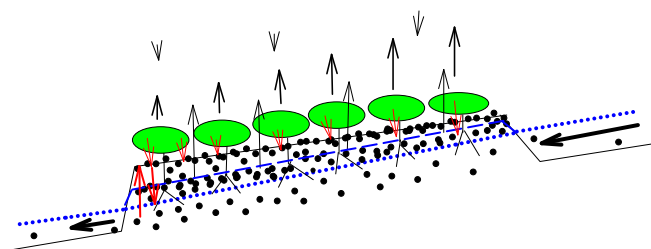


**Fig. 8.** Annual salt deposition from leaves and exudate of saltcedar plants at sites in Cibola National Wildlife Refuge. DSW = Diablo Southwest; DT = Diablo Tower site.



**Fig. 9.** Calculated salt content of soil profiles at three saltcedar sites at Cibola National Wildlife Refuge, based on electrical conductivity of saturated soil extracts, partitioned into three intervals of soil depth. The insert shows the amount of salt that would have been deposited in leaf litter and exudate from 1938 to 2010, assuming the rate measured in 2009 was the same for all years.

O'Leary, 1984; Glenn, 1987; Flowers and Colmer, 2008). Excess salts are then excreted from the leaves of many species, or stored in the leaves and deposited in litter-fall when leaves senesce (Rozema et al., 1981; Flowers and Colmer, 2008). The data for CNWR allow the mean concentration of salts in the plant transpiration stream to be calculated for saltcedar. Dividing annual canopy-level salt deposition ( $0.19 \text{ kg m}^{-2} \text{ yr}^{-1}$  from salt exudates plus leaf litter) by transpiration (ca.  $1 \text{ m}^3 \text{ m}^{-2} \text{ yr}^{-1}$ ) (Nagler et al., 2009), produces a value of  $0.19 \text{ g L}^{-1}$ , lower than aquifer or river salinities. This is similar to estimates from Rozema et al. (1981), which ranged from  $0.20$  to  $0.55 \text{ g L}^{-1}$  in studies with European salt marsh halophytes (Rozema et al., 1981). Rates of salt excretion by saltcedar are also similar to rates for other halophytes. For example, the salt marsh grass, *Spartina anglica*, excreted NaCl at a rate of  $2.3 \text{ } \mu\text{mol cm}^{-2}$  leaf area per 6 days in a greenhouse study (Rozema et al., 1981). Based on an annual growth cycle of 220 days and a LAI of 2.5 (Nagler et al., 2009), the leaf-level rate for saltcedar excretion in this study would be  $2.2 \text{ } \mu\text{mol cm}^{-2}$  per 6 days. However, rates of excretion depend on salinity of the growth media and vary widely among species (Rozema et al., 1981), reaching ten times higher than saltcedar in a desert xerohalophyte from the Tamaraceae (Ramadan, 1998).



**Fig. 10.** Simplified conceptual diagram of the terrace at Cibola National Wildlife Refuge, showing flows of water (arrows), accumulation of salts (black circles), and location of the alluvial aquifer (blue circles) and the capillary fringe above the aquifer (dashed blue line). Water enters the aquifer as underflow from the river, replacing pure water discharged by transpiration by saltcedar and evaporation from the capillary fringe and soil surface. Scant rainfall contributes to downward leaching of salts. Salt concentrations in the aquifer fringe increase with distance from the river as pure water is withdrawn in evapotranspiration. Red arrows show the upward movement of salts in the capillary fringe and the downward movement of salts due to diffusion and leaching, and deposition by saltcedar leaves and exudates. At the downstream end of the terrace water and salts are discharged back to the river. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)



Co-occurring halophytes at this site include arrowweed (*Pluchea sericea*), which has lower salt tolerance than saltcedar, and the euhalophytes iodine bush (*Allenrolfia occidentalis*), quailbush (*Atriplex lentiformis*) and fourwing saltbush (*Atriplex canescens*), which have higher salt tolerance than saltcedar (Glenn and O'Leary, 1984; Glenn et al., 1998; Vandersande et al., 2001). Salt content and excretion rates have not been measured in detail for these species, but *Atriplex* spp. excrete salts via salt bladders on leaves while iodine bush accumulates salts in its succulent stems, hence replacement vegetation would also contribute salts to surface soils.

#### 4. Conclusions

The river terrace studied here presented a picture of the salinization processes that can occur on saltcedar-dominated, flow-regulated arid zone rivers. Three linked processes led to high salt contents in the aquifer, at the soil surface, and in the vadose zone above the aquifer. A conceptual diagram illustrates these processes in Fig. 10. The salinity of the aquifer increased with distance from the river due to the progressive extraction of water from the aquifer by saltcedar and other halophytes, with most of most salts left behind in the aquifer (Zhu et al., 2011; Taghvaeian, 2011). Depth to water responded to both river stage and to diurnal changes in saltcedar transpiration close to the river (at Swamp), and to diurnal and seasonal changes in saltcedar transpiration rates but not to river stage away from the river (at Slitherin and Diablo) (Zhu et al., 2011; Taghvaeian, 2011). There was a net hydraulic gradient from the river into the terrace at all times of year, which was highest in summer and lowest in winter when saltcedar was dormant (Taghvaeian, 2011). Hence, the aquifer is part of a dynamic system, supported by underflow from the river into the aquifer that replaces water lost in transpiration (Zhu et al., 2011; Taghvaeian, 2011).

The second process leading to salinization of the terrace was accumulation of salts above the aquifer by capillary rise of water and salts into that soil layer. Salt concentrations in this layer were about 10 times higher than in the aquifer, indicating that evaporation of water from the soil surface and capillary fringe was taking place (e.g., Doering et al., 1964). This layer, too, should reach an equilibrium state, due to downward diffusion of salts back to the aquifer along the steep concentration gradient that develops in the profile, with downward diffusion and leaching eventually matching the rise of salts from the aquifer into the capillary fringe by bulk transport (Doering et al., 1964).

The third process leading to salinization was transport of aquifer salts to the surface by saltcedar through production of exudates and leaf litter. Over all sites, saltcedar exudates and leaf litter could account for 30% of the salts present in the profile, assuming a constant production rate since 1938. These salts will tend to move downward through periodic leaching events and diffusion, mixing with salts rising from the aquifer and becoming part of the overall equilibrium condition for distribution of salts through the profile.

Elimination of saltcedar from this terrace would not necessarily reduce the overall salt load in the soil, or allow the return of mesic native vegetation. Soil EC levels were lowest at Slitherin, which had the densest growth of saltcedar and the highest ET rates of all the sites (Nagler et al., 2008; 2009). Deep-rooted vegetation can reduce the upward movement of water and salts through the soil through extraction to support transpiration (e.g., Nosetto et al., 2007; Scanlon and Goldsmith, 1997). Furthermore, surface salts were not significantly different under saltcedar bushes than between bushes, even though salt deposition by saltcedar was highest under rather than between bushes.

Restoration of terraces like this one with mesic vegetation would be a daunting task. Under present conditions, the aquifer at Swamp

could marginally support cottonwood and willow trees (Glenn et al., 1998; Vandersande et al., 2001) if the aquifer could be accessed through pole plantings (e.g., Anderson and Ohmart, 1985; Anderson, 1995). Anderson (1995) considered  $2 \text{ g L}^{-1}$  as the upper limit for successful establishment of these species on the Lower Colorado River, but they considered 2.5 m as the threshold depth to water. The depth and salinity of the aquifer at sites away from the river are well beyond the tolerance limits of cottonwoods and willows (Anderson, 1995). Very large volumes of river water would be required to wash salts from the soils and refresh the aquifers to the point that native trees could return on most of the terrace.

Further water and a restored flood regime would be needed to maintain these conditions over time. Hence, this terrace is likely to remain a halophyte-dominated ecosystem into the future. Anderson (1995), reviewing efforts to restore native tree habitat to the lower Colorado River, concluded that removing saltcedar would result in a serious loss of wildlife habitat, since desirable replacement species would not be able to tolerate the high salinity and lower water table conditions on the floodplains today (see also van Riper et al., 2008; Sogge et al., 2008, for habitat value of saltcedar). The present results, showing the distribution of salts in the aquifer and soil profile of a non-flooding river terrace, support those conclusions.

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