SALTWATER INCURSION INTO MICRO TIDAL WETLANDS:

CASE STUDIES FROM MATAGORDA, TEXAS AND HUMACAO,

PUERTO RICO

A Dissertation

by

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ABSTRACT

Global climate change threatens the survival of microtidal wetlands by altering fundamental hydrological aspects such as precipitation patterns and tidal exchange. The combination of these stressors results in increased flooding period and soil salinity in coastal wetlands. In this study, we combined the use of detailed hydrological measurements (wetland water level and salinity), LIDAR elevation models, and water stable isotopes tracers (δD, $\delta^{18}O$) to study the balance between freshwater and saltwater inputs on two microtidal wetlands: a saltmarsh in Matagorda, Texas and a freshwater-forested wetland in Humacao, Puerto Rico.

In Matagorda, Texas, we described the process of connectivity between different hydrologic units (isolated and connected ponds) within the saltmarsh. Pond connectivity only occurred when water levels in major water bodies adjacent to our study site reached a threshold elevation of 0.39 m. Connections events were correlated to rainfall and— to a lesser extent— wind speed and direction. We conclude that connectivity within the saltmarsh is driven by the combined effect of tidal influence and rainfall inputs, factors that will be altered by sea level rise and climate change-related changes in long term weather patterns.

In Humacao, Puerto Rico, we gathered a detailed dataset of changes in salinity and water level in a freshwater forested wetland dominated by the endangered salt intolerant species *Pterocarpus officinalis*. In addition, we studied tree water use and

identified important water sources to the wetland using stable isotope tracers. Firstly, we provide evidence that recent hydrological alterations have effectively transformed the system from mostly freshwater, to a saltwater wedge estuary. Salinity inputs travel via a tidal creek channel that allows the progression of a saltwater wedge to the inland parts of the forest. Our results suggest that inland progression of the saltwater wedge is influenced by amplitude of tidal exchange in the middle portions of the tidal creek and by extended dry periods in the headmost part of the tidal creek. Isotope data showed that surface standing water was influenced by tidal water sources during the dry season, although the spatial extent of this influence was constrained to areas of the forest that had been previously deforested. The isotopic content of groundwater samples taken at increasing distances from the tidal creek revealed that— although surface waters are dominated by freshwater inputs (rainfall and runoff) during the wet season— the influence of tidal water sources at soil depths greater than 60 cm persists throughout the year. Nonetheless, isotopic content of *Pterocapus officinalis* stem water samples suggest that tree water uptake is constrained to very shallow, unsaturated parts of the soil.

We conclude from both case studies that the long term vulnerability of microtidal wetlands to climate change is determined by the interaction of increased annual variability of freshwater inputs along with a steady increase in mean sea levels, and aggravated by extreme climatic events.

DEDICATION

Para Violeta, nuestro sol brillante.

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Finally, I must thank my family for making everything possible. My parents, Raúl Colón, who me taught me to constantly seek knowledge on my own, and Lucía Rivera, who taught me the incredible value of persistence. Their values, love and support made me the person I am today. Equally important has been the unconditional love of my brothers, Carlos M. Colón and Juan R. Colón, who were my first mentors and believed in me more than anyone. My wife, Dorimar Ortiz, had an integral role during all the stages of my PhD preparation. She contributed in the field and in the revision of manuscripts, patiently supported me through the vicissitudes of graduate school, and took more than her share of household and parenting chores, all so I could complete my work. For everything, I am deeply grateful.

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1. INTRODUCTION

Human-induced climate change is considered one of the most serious environmental threats today (FitzGerald et al. 2008). The steady increase of the mean temperature of the planet causes the oceans to expand, provides more moisture to the atmosphere, and releases vast amounts of freshwater previously locked in polar ice caps (IPCC 2007). The result is global sea level rise, increased interannual and seasonal variability of weather patterns, and increased frequency of extreme weather events (Tompkins and Adger 2004). Although these climate changes occur worldwide, coastal areas are considered among the most vulnerable to climate change because of the convergence of threats in a single location (IPCC 2007). Coastal ecosystems are well located to provide ecosystem services that can enhance the resilience of both natural and human systems to climate change (Hopkinson et al. 2008, Craft et al. 2009). However, this location also puts them in the way of human development (Bertness et al. 2002, Feagin et al. 2010). Tidal wetlands, for example, occupy the narrow interface between the land and sea, and protect the coast from erosion and extreme weather events while providing habitat for wildlife as well as cultural and aesthetic services (Mitsch 2000, Mitsch and Gosselink 2007). Tidal wetlands are also very productive systems (Simpson et al. 1983), and contribute to reducing the effects of climate change by sequestering CO_2 from the atmosphere (Ross and Adam 2013).

Sea level rise affects tidal wetlands by increasing the hydroperiod (duration of inundation in a year) and overall salinity of the system (Mcleod et al. 2010, Crase et al.

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2013). Tidal wetlands can respond to the change in sea level by migrating inland

(Brinson et al. 1995), although their capacity to this is limited by human coastal

developments (Feagin et al. 2010). On the other hand, if sediment accumulation

(accretion) occurs at a higher rate than sea level rise, tidal wetlands can survive by

increasing the relative elevation to the sea (McKee 2011). In this context, microtidal

wetlands (tidal range <2m) are more vulnerable than macrotidal (>4 m) or mesotidal (2-4

m) wetlands because of the limited contribution of sediment by the tide (Craft et al.

2009). Instead, microtidal wetlands rely more on the accumulation of organic matter to

gain elevation (Simas et al. 2001) or prevent erosion by stabilizing the soil (Feagin et al.

2009). Freshwater microtidal wetlands, however, may be affected by sea level rise

regardless of their capacity to accrete sediments; a consistent increase in salinity may

shift the composition of the plant community towards salt-tolerant species (Sharpe and

Baldwin 2012). In both saltwater and freshwater microtidal wetlands, the dynamic

between freshwater inputs and tidal exchange is the most important factor affecting the

resilience of the system and the delivery of ecosystem services (Fig. 1-1).

In this dissertation, we aimed to increase our understanding of this dynamic

exchange between freshwater and saltwater inputs by studying two microtidal wetlands

with contrasting plant communities. The first wetland is a saltmarsh dominated by *Spartina alterniflora*, located at Matagorda, Texas. In this location, we studied how individual ponds connected to each other and the main factors driving the hydrological connectivity within the marsh itself. The second location is a freshwater forested wetland dominated by the tree *Pterocarpus officinalis* in Humacao, Puerto Rico. *Pterocarpus* trees have a low tolerance of salinity, thus they depend on freshwater inputs to survive. Hydrological modifications at this site have increased tidal exchange in the wetland but there was a lack of information about the extent and magnitude of saltwater intrusion into the *Pterocarpus* swamp. In both Matagorda, Texas and Humacao, Puerto Rico we combined the use of detailed tidal measurements (water level and salinity) with water stable isotopes tracers as our main methodology. By combining these methods, we aimed to provide new insights to our understanding of the hydrology of microtidal wetlands, and how it is affected by climate variability.

Figure 1 - 1. Conceptual model of microtidal wetland resilience to climate change
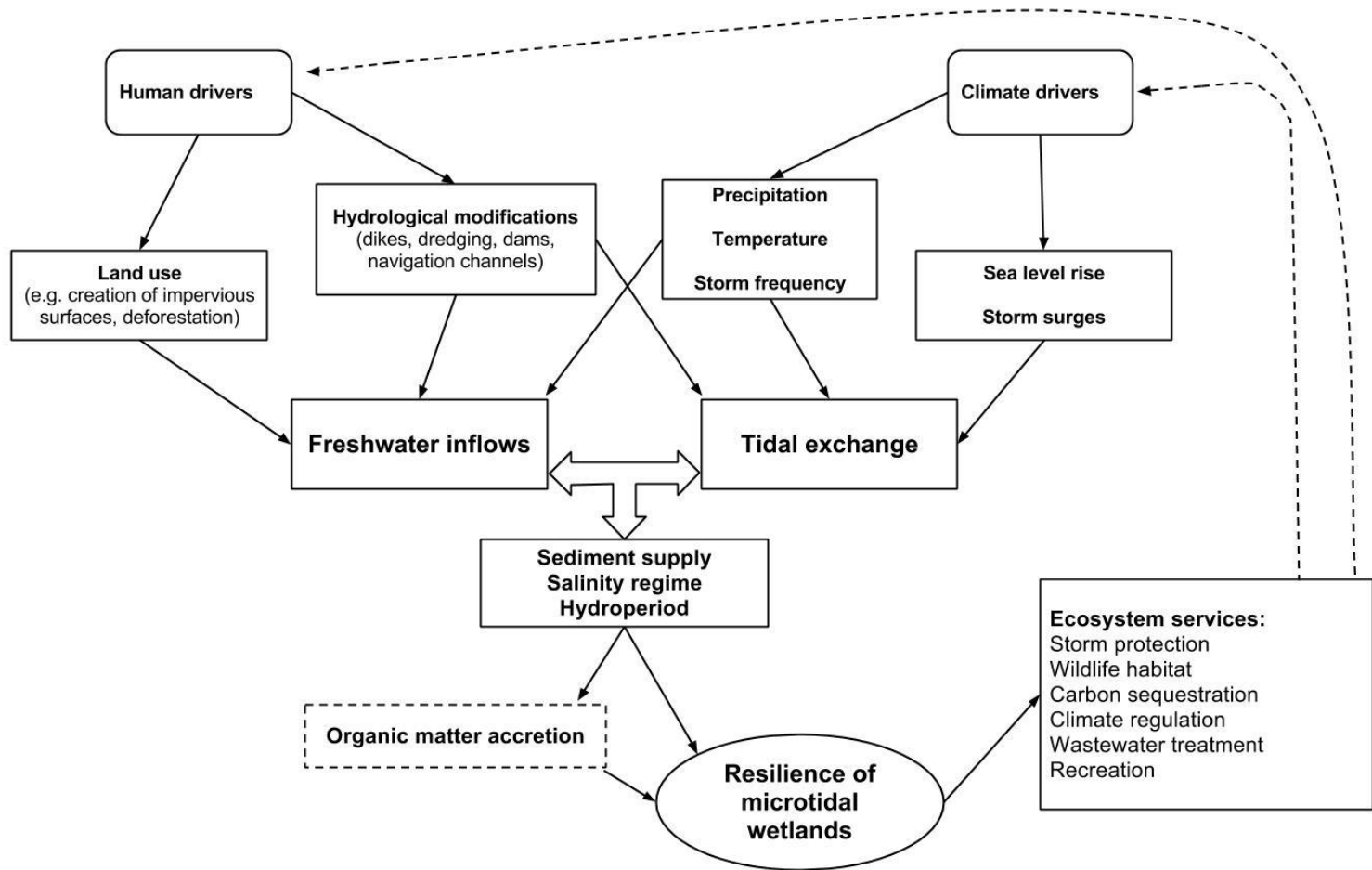
2. SALT MARSH CONNECTIVITY AND FRESHWATER VERSUS SALTWATER INFLOW: MULTIPLE METHODS INCLUDING TIDAL GAUGES, WATER ISOTOPES, AND LIDAR ELEVATION MODELS[*]

2.1 Overview

The hydrologic connectivity between different components of a coastal marsh is a key wetland structural element, and is essential in determining habitat value. Salt marshes are often thought of as being hydrologically connected via tidal creeks, though water bodies within a salt marsh may also be semi-permanently disconnected. At a salt marsh in Matagorda, Texas, USA, there are many 'isolated' ponds, located at varying elevations. Our objective was to quantify the hydrologic connectivity of spatially isolated ponds at this site. We sampled water for salinity and stable isotopes (δ^2H, δ^{18}O) to determine the relative contribution of tidal water and rainfall within each pond. We also quantified the water level at which each pond floods its banks and connects to other ponds, using a light detection and ranging (LIDAR)-based elevation model. We found that pond connectivity was driven by multiple factors, of which wind and rainfall were the most important. Salinity and isotopic values between any two sampled ponds were

correlated with the water level elevation at which the ponds connected. We conclude that the connectivity of the network, and the similarity of water samples within it, profoundly changes when specific water level thresholds are exceeded by both freshwater and saltwater.

2.2   Introduction

The ability of a wetland to perform important ecological functions is directly related to its hydrology (Odum et al. 1995, Mitsch and Gosselink 2007). In the case of tidal wetlands, like salt marshes, the energy and materials transferred by a pulsating hydroperiod drives many aspects of wetland structure and function (Odum et al. 1995). Through these hydrological processes, salt marshes respond to and interact with the estuaries in which they reside (Montalto and Steenhuis 2004). As for other wetlands, hydrological connectivity plays a significant role in determining the extent of these interactions (Gelwick et al. 2001). Traditionally, wetlands have been classified as 'connected' or 'isolated' depending on their geographical setting. However, the interactions between surface flows and the consideration of temporal variation in the hydrological connections have proven that connectivity among wetlands is a complex event and that few, if any, wetlands can be considered truly isolated (Leibowitz and Vining 2003, Wilcox et al. 2011). Wetlands may be connected by ecological, hydrological or geographical features, all of which have different effects on wetland structure and function.

Hydrological connectivity also occurs within the wetland ecosystem. Salt

marshes are often thought of as being hydrologically connected to the ocean via tidal

creeks, though water bodies within the marsh may also be semi-permanently

disconnected. A salt marsh is comprised of areas with different hydrological interactions

(e.g., tidal creeks, connected and isolated ponds, high marsh and low marsh vegetated

areas). The hydrologic connectivity within and between these different areas of the

marsh is a key wetland structural element and is essential in determining species

diversity and assemblage structure throughout the marsh (Gelwick et al. 2001). Surface

connectivity between ponds will determine the accessibility of safe foraging sites for

fishes, their energetic costs to access the marsh surface, as well as the risk level of

being stranded at low tide (West and Zedler 2000). Ponds that are regularly

disconnected from surface water inflows can lose water to evaporation and concentrate

fishes in a reduced habitat, increasing competition for resources and facilitating

predation by wading birds (Kushlan 1976, Winemiller 1989). The intermittent nature of

connectivity events among the different hydrological units of the marsh can isolate areas

of the marsh and create conditions where the extinction of populations is possible.

These populations are dependent on connection events to recolonize areas and interact

with other populations, creating excellent conditions for the study of metapopulation

dynamics (groups of populations that interact with each other to support the regional

persistence of the species (Sheaves and Johnston 2008)). Given the importance of

surface water connections for the structure and function of coastal wetlands like salt

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marshes, the dynamics of hydrological connectivity need to be studied in order to create working knowledge of coastal systems.

Traditionally, methods to study wetland hydrology include the deployment of a series of instruments in the field to gather basic information about the hydrodynamics (e.g., water level recorders, rain gauges, stream flow meters, and piezometers). The use of water level recorders is perhaps the most basic and useful tool to study wetland hydrology. Water level records can help quantify a number of hydrologic parameters, including hydroperiod, frequency of flooding, duration of flooding and water depth (Mitsch and Gosselink 2007). For example, Negishi et al. (2010) successfully integrated water level data into quantitative models to study connectivity of river floodplains in Japan. Tidal gauges can also be used to study long-term trends in sea level rise and subsidence in coastal ecosystems (Church and White 2006, González and Tornqvist 2006, Prandi et al. 2009). However, the data provided by these instruments alone does not provide enough information to study the complex connectivity of ponds in a salt marsh.

Stable isotopes (δ^{18}O and δ^2H) can act as tracers to study many aspects of an aquatic system's hydrology. This is because water can either be 'enriched' or 'depleted' in the heavy isotopes ^{18}O and ^{2}H, depending on how different factors affect isotopic fractionation during the phase changes of the water cycle. Water undergoes two important processes of fractionation, equilibrium fractionation and transport isotope fractionation (kinetic fractionation). During equilibrium fractionation there is no change

in the amount or concentration of the chemicals compounds, but the resulting products have different isotopic compositions due to differences in bond strength of the isotope species (Gat 2010). This type of fractionation occurs when precipitation forms in the atmosphere and is responsible for the predictable patterns of $\delta^{18}O$ and δ^2H described by the Global Meteoric Water Line (GMWL) (Craig 1961). Transport isotope fractionation occurs when water is evaporating from a surface and leads to the enrichment of the evaporating water body because lighter molecules react faster and become concentrated in the products. Research using stable isotopes to address hydrological problems includes the study of groundwater inflows and outflows (Stadnyk et al. 2005) surface water studies (e.g., springs; Criss et al. 2001, oceans and rivers; Lachniet and Patterson 2002, estuaries; Surge and Lohmann 2002 and water usage by plants (West et al. 2006, Greaver and Sternberg 2006) among many other applications. In addition, stable isotope values can be mapped across the landscape to create spatially-explicit predictions of isotopic values at different scales (West et al. 2010). However, there have been few studies that use water stable isotopes to study wetland hydrology, and those that did were primarily focused on the study of groundwater (Huddart et al. 1999, Wilcox et al. 2004, Clay et al. 2004). To date, there have been no studies using water stable isotopes as a tool to study hydrological connectivity in a salt marsh.

Digital Elevation Models (DEM), as derived from LIDAR (Light Detection and Ranging) data, are high resolution representations of landscape topography. The increasing use of LIDAR DEMs in wetland hydrology studies (Martinez Romero et al.

2009, Negishi et al. 2010), as well as the recently developed method to use the intensity (amplitude) of the laser return to remotely monitor inundation in forested wetlands (Lang and McCarty 2009), are examples of the potential of LIDAR data to study wetland hydrology. LIDAR-based connectivity models, in combination with *in situ* water level data, have been used to predict potential habitat distributions in a floodplain in Japan (Negishi et al. 2010), and a mangrove forest in Australia (Knight et al. 2009). Terrestrial Laser Scanning (TLS) LIDAR models (Guarnieri et al. 2009) provide even finer scaled products than those of aerial LIDAR models, though they typically do not make it possible to study coarser scale tidal networks at the landscape scale (Lohani and Mason 2001). Regardless of the specific method of data collection, LIDAR DEMs are an ideal tool to study pond connectivity in salt marshes due to their ability to map elevation and predict the presence of water in ponds, as well as their increasing availability of coverage for coastal ecosystems.

The connectivity aspect of wetland hydrology, particularly in coastal systems, drives many ecological processes. Basic science and applied coastal management can benefit from studies aimed at understanding connectivity at finer scales. Each of the described methods provides valuable information regarding the hydrology of an ecosystem, however to study connectivity more fully, it is to necessary integrate traditional instrumentation (tidal and rain gauges) with new tools such as spatial modeling (LIDAR DEMs) and stable isotope analysis ($\delta^{18}O$ and δ^2H). Here, we demonstrate the power of combining these methods at a back-barrier salt marsh in

Matagorda, Texas, USA. At this location, there are many 'isolated' ponds and other more 'connected' ponds located at varying elevations. We have visually noted that water levels often appear to be different from one portion of the study site to the other. In particular, two adjacent and large bodies of water appear to have different water levels at times. We hypothesized that this water level difference was due to the hydrological isolation of an isolated pond from a more connected pond. We hypothesized that water level in the connected pond would appear somewhat astronomical and similar to the open bay, while the isolated pond would be rainfall driven. Our overall objective was to quantify the hydrologic connectivity of spatially isolated ponds at this site. By definition, we consider ponds to be hydrologically connected when there is a continuous, surface water interaction between the pond and another water body in the marsh initiated either by recurrent events (tides, etc) or by pulse events (storms, rain etc). Specific objectives included:

1. Determine the predominant mechanism that caused ponds to flood and connect to each other. Was it astronomical tides, wind driven tides, or rainfall?

2. Determine the relative contributions of tidal water and rainfall to individual ponds.

3. Determine the connectivity among individual ponds.

The nature of these objectives required us to apply a combination of methods to address them. Specifically, we will address the first objective by studying the relation

between water levels in the ponds and the likely drivers of connectivity (rainfall and tides). The second objective will be addressed by analyzing stable isotope composition of water samples from the ponds, and finally, the third objective will be addressed by applying a connectivity model based on LIDAR derived elevation data.

2.3   Methods

2.3.1   *Study site*

Our study area is a back-barrier wetland located at the Matagorda Bay Nature Park, on East Matagorda Peninsula, near Matagorda, Texas, USA (Fig. 2-1). The East Matagorda Peninsula is a transgressive barrier island that was connected to the mainland in the early 1920's as a result of human modification. The wetland is composed of typical salt marsh plant cover for this region, with zonation strongly dependent upon elevation: open water with some *Ruppia maritima* and *Halodule wrightii*, low marsh dominated by *Spartina alterniflora*, mid marsh dominated by *Salicornia virginica* and *Batis maritima*, unvegetated salt flat with some *Monanthocloe littoralis* around the edges, and high marsh dominated by *Spartina spartinae*.

As described by Mitsch et al. (2009), the ecological structure and function of salt marshes is similar around the world, with different plant associations occurring over different coastlines. The salt marsh at Matagorda is representative of coastal plain salt marshes in Eastern North America where *S. alteniflora, S. patens* and *Juncus roemerianus* are dominant species and tidal range is small in relation to similar

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ecosystems on the northern regions of North America and Europe. In contrast, European salt marshes are distinguished by the lack of vegetation in the high intertidal zone, an area dominated by *S. alterniflora* in Eastern North America. As with other salt marshes around the world, a complex hydrogeomorphology produces networks of tidal creeks and water-filled depressions (ponds) that are used extensively by waterfowl. The particularly low inflows of freshwater to the coastal areas in the Texas Gulf Coast can allow these smaller water bodies in the marsh to concentrate extreme salinities in relation to seawater due to evapotranspiration. Recently, the geographic coverage of S. *alterniflora* has changed because of invasive expansion to the west coast of the U.S. and coastal regions of China. On the other hand, more than 100,000 ha of *S. alterniflora* salt marshes were lost in the Gulf Coast of the United States during major die-off events in 2000-2004 (Silliman et al. 2005).

Water levels within East Matagorda Bay can be irregular, and dependent upon spatial location (Kraus and Militello 1999). The bay is connected to the Gulf of Mexico at Brown Cedar Cut, which is a small pass on the east flank of the bay (Fig. 2-1a). Astronomical tide levels in the open bay are approximately 0.1 m between daily high and low, and are semi-diurnal (as compared to approximately 0.4 m semi-diurnal tides in the open Gulf). However, in the bay, wind driven tides may affect water levels up to approximately 0.6 m. Water level differences between the east and west portions of the bay are pronounced at times, as water may be driven from one side of the bay to the other via wind set-up, particularly by northeasterly winds (Kraus and Militello 1999).

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Figure 2 - 1. Study area. (a) Location of back-barrier wetland site in Matagorda, Texas, USA. Dashed circle marks the location of Brown Cedar Cut. (b) Tides flow from East Matagorda Bay to the study area along the direction of black arrow. Water sample points are denoted by small circles. (c) Primary ponds are denoted by black stars. Tidal flow connects the connected pond, but not the isolated pond. Water sample points are denoted by small circles. Black dashed line marks the location of a known growth fault.
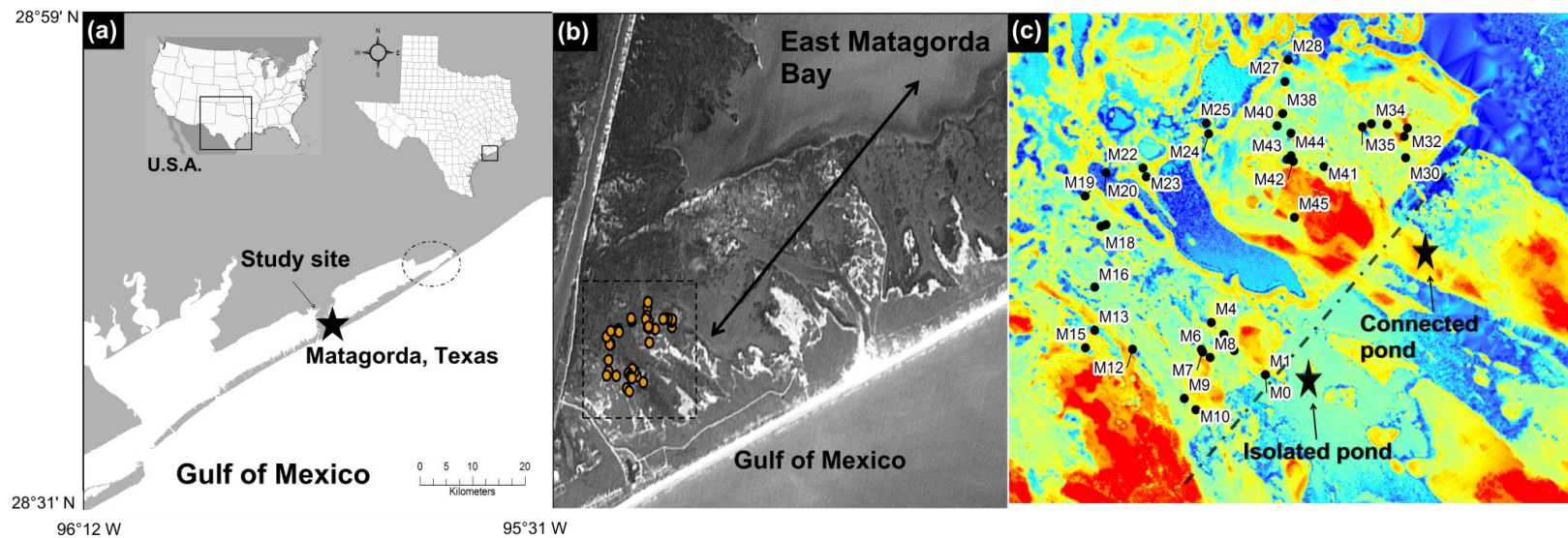
Our wetland study site lies in the southwestern corner of the bay (Fig. 2-1b). In general, tidal flow is uninhibited through this low-lying wetland, and is facilitated by the existence of an active growth fault that has funneled flow to much of the wetland interior (Cline et al. 2011, Feagin et al. 2009).

2.3.2 Tidal gauges

Water levels were recorded hourly over a period of 6 months (February – August 2010). This period of time included two relatively dry months (March and April with 64.5 and 66.8 mm respectively) and one relatively wet month (July, 315.9 mm) for Matagorda according to annual precipitation averages for the site (NCDC Station 416750). Water level measurements took place in two primary water bodies, an isolated pond and a connected pond. Within each pond, a small hole was made in the sediment surface beneath open water. A 2 m long, 0.10 m diameter PVC pipe that had been cut linearly along its length to facilitated water movement through the pipe was sunk into this hole. A titanium pressure gauge (HOBO U20-Ti, Onset Corp.) was suspended in the water within the pipe with wire, and set at approximately 10 cm below the original sediment surface. Gauges were suspended at this depth to avoid losing data if the ponds dried up. The elevation of the pond sediment surface, water level at time of installation, and water level at the end of the experiment were recorded. A survey grade Global Positioning System (GPS) point was recorded at the top of each PVC pipe to within 0.002 m vertical and 0.001 m horizontal accuracy (Trimble R7, Trimble Corp.), and water levels were referenced to absolute elevation in NAVD88 units. A third gauge was

installed several meters above water level, to enable atmospheric compensation of the water level data. Hourly rainfall was recorded at this site (HOBO Rain Gauge RG3, Onset Corp.).

We also acquired rainfall, wind direction, wind speed, atmospheric pressure, temperature, and relative humidity data sets over the same period from the Palacios weather station (NCDC Station 416750), located 27 km to the west at similar elevation and on the same bay complex, in this topographically flat portion of Texas. Rainfall and atmospheric pressure data collected from the Palacios station closely matched that collected on site. Therefore, it was assumed that data for the other parameters from the Palacios station could be extrapolated to the site.

Hourly water level data were then plotted and linearly regressed against the other parameters using both on-site and Palacios station data. In order to account for the effect of evapotranspiration (ET) on the change in water level in the ponds, we estimated ET using the Hargreaves Equation (Hargreaves and Samani 1985). The Hargreaves equation is recommended as one of the few valid temperature-based estimates for ET. It provides a simple calculation of ET estimates where there is lack of climatologic data necessary to calculate more complex equations (Shuttleworth 1993, Temesgen et al. 2005).

The Hargreaves equation is expressed as:

$$ET_0 = .0025(T_c + 17.8)\left(\sqrt{T_{max} - T_{min}}\right)R_a$$

where T_c is daily mean air temperature (°C), T_{max} is daily maximum air temperature (°C),

T_{min} is daily minimum air temperature (°C), and R_a is extraterrestrial radiation (MJ m^{-2}

day^{-1}). Since the mean air temperature T_c is calculated as an average of T_{max} and T_{min} and

R_a is computed from information on location of the site and time of the year, temperature

is the only parameter needed to calculate this equation (Temesgen 2005).

2.3.3   *Stable isotopes and water salinity*

Water samples were collected from the isolated and the connected ponds, and

from 45 smaller ponds arrayed across a hypothesized path of connection between them.

Sampling took place on March 1, 2010 at the end of a wet period. In the 7-day period

before the experiment, there had been 3.05 mm of rainfall. In a total of 28 days prior to

the experiment, there had been 115.06 mm. Recorded rainfall for the two years prior to

the study is shown in Fig. 2-2. At each of 45 individual ponds, water samples were

collected by filling and capping airtight, small bottles (20 mL, Nalgene) with pond water

collected 5 cm below the surface. Salinity was also measured in the field using a

refractometer. The spatial position of each water sample and pond location was

recorded using a Global Positioning System (GPS) to at least 0.5 m horizontal accuracy

per point (XRS Trimble, Trimble Corp.).

Water samples were analyzed for stable isotopes ratios (δ^2H, δ^{18}O). Samples

were refrigerated (3°C) between the date of initial sampling and the date of analysis.

Subsequently in the laboratory, all samples were analyzed using a continuous-flow

pyrolysis isotope ratio mass spectrometry technique (TC/EA-IRMS; Delta V Isotope

Ratio Mass Spectrometer; Thermo Scientific, Waltham, MA). Each sample was

analyzed for both hydrogen and oxygen isotope ratios. Stable isotope ratios are

expressed as follows:

$$\delta(\%_{00}) = \left[\frac{R_{sample}}{R_{standard}} - 1 \right] \times 1000$$

where R_{sample} represents the molar ratio of heavy to light isotope (here either $^2H/^1H$ or

$^{18}O/^{16}O$) and $R_{standard}$ is the molar ratio of (in this case) Standard Mean Ocean Water

(SMOW). Laboratory reference waters that had been calibrated to SMOW were used in

each run for normalization and quality control. δ^2H and $\delta^{18}O$ ratio values were then

plotted, along with the GMWL. Additionally, $\delta^{18}O$ ratios of the ponds were plotted

against salinity. Maps of δ^2H ratios across the landscape were also created within a

Geographic Information System (ArcGIS 10, ESRI, USA).

To sample groundwater from several of the smaller ponds, 1 m long, 0.05 m

diameter PVC pipes were sunk to 0.5 m depth below the sediment surface. After

insertion, water was pumped out of these pipes. These pipes did not refill within a 24-

hour period, due to a hard clay layer at the bottom of each of the smaller ponds. While

much of the study area is composed of silty sands, the bottom of the smaller ponds

appeared to be well sealed by a clay layer. Based upon this evidence, groundwater

exchange was considered minimal between all ponds.

2.3.4 *LIDAR elevation model*

As stated above, we consider ponds to be hydrologically connected when there is

a continuous, surface water interaction between the pond and another water body in the

marsh. A LIDAR-based technique was developed to assess the water levels at which

individual ponds would flood their banks and connect to other ponds. First, a 2006

LIDAR dataset was converted into a standard Digital Elevation Model (DEM) at 1 m^2

horizontal and 1 cm vertical resolution. Vertical bias in the DEM was up to 10 cm,

though consistent across the study area, as is typical of LIDAR imagery. In other

research, this same DEM had been registered with ground survey

Figure 2 - 2. On-site rainfall for the two years before the study.
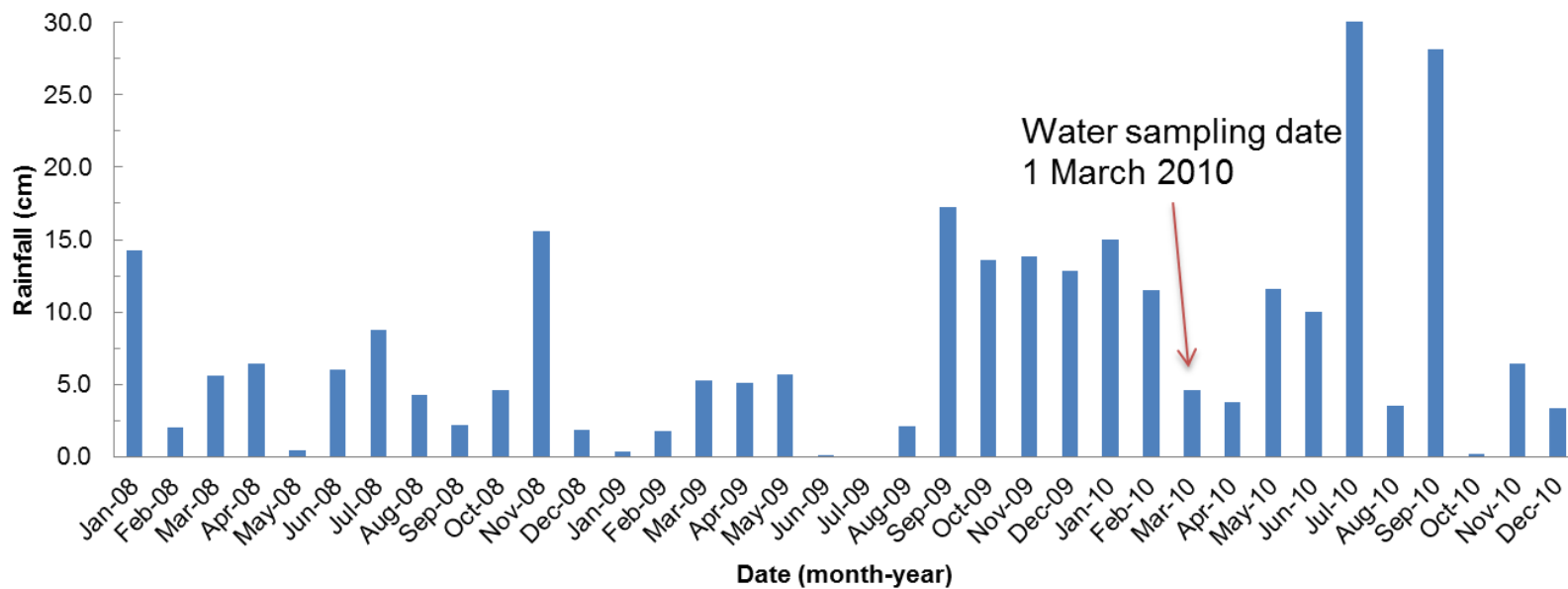
data and the relative point-to-point DEM precision within the scene was found to be within a few centimeters (Cline et al. 2011). We identified areas that would be above or below water at different water depths by producing vertical slices of the DEM at 1 cm intervals. We then identified the minimum water level for which adjacent ponds would be connected by water. All combinations of pond-to-pond connections, as dependent upon elevation, were then recorded into a matrix. Next, the minimum elevation at which any two ponds connected to each other was graphed, producing a hierarchical lineage of connection events for all ponds. For example, we first found the elevation at which a pond connects to another pond, and then found the elevation at which those two ponds then connected to a third, and so on, building a graphical representation. Salinity and stable isotope values for each small pond were then linearly regressed against the minimum elevation at which the pond connected to the primary connected pond and the open bay.

2.4   Results

2.4.1   Tidal gauges

For the majority of the tidal gauge study period, the water level in the isolated pond decreased (Fig. 2-3). During these same periods, the water level in the connected pond was much more variable. There was an apparent, slight degree of semi-diurnal astronomical forcing within this variability, as evidenced by cyclic water level change at the daily time scale, but the wind-driven tidal influence of the open bay was predominant. Water level records (Fig. 2-3) suggests that when water levels rose above 0.39 m in elevation, the two ponds connected and began to follow the same tidal beat. The water level of the connected pond typically rose first, and when it exceeded the 0.39 m threshold, water spilled over into the isolated pond, and the water levels in the two ponds then equilibrated.

Figure 2 - 3. Water levels in the connected pond and the isolated pond, on-site rainfall, and wind strength at the site. Dashed line represents 0.39 m elevation (connection threshold). All dates are in 2010.
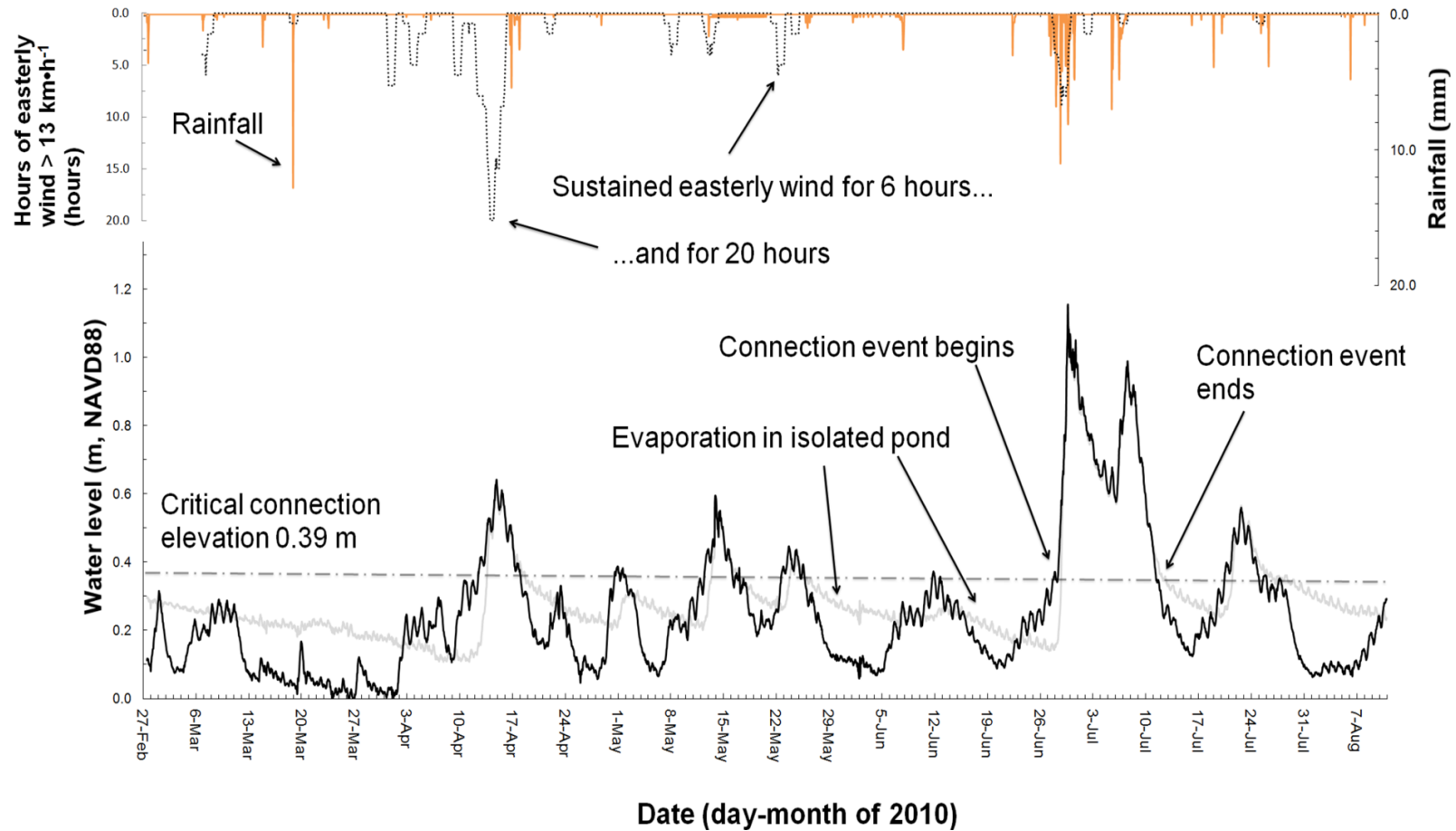
Figure 2 - 4. Scatter plot of change in water level in the isolated pond versus water level in the connected pond.
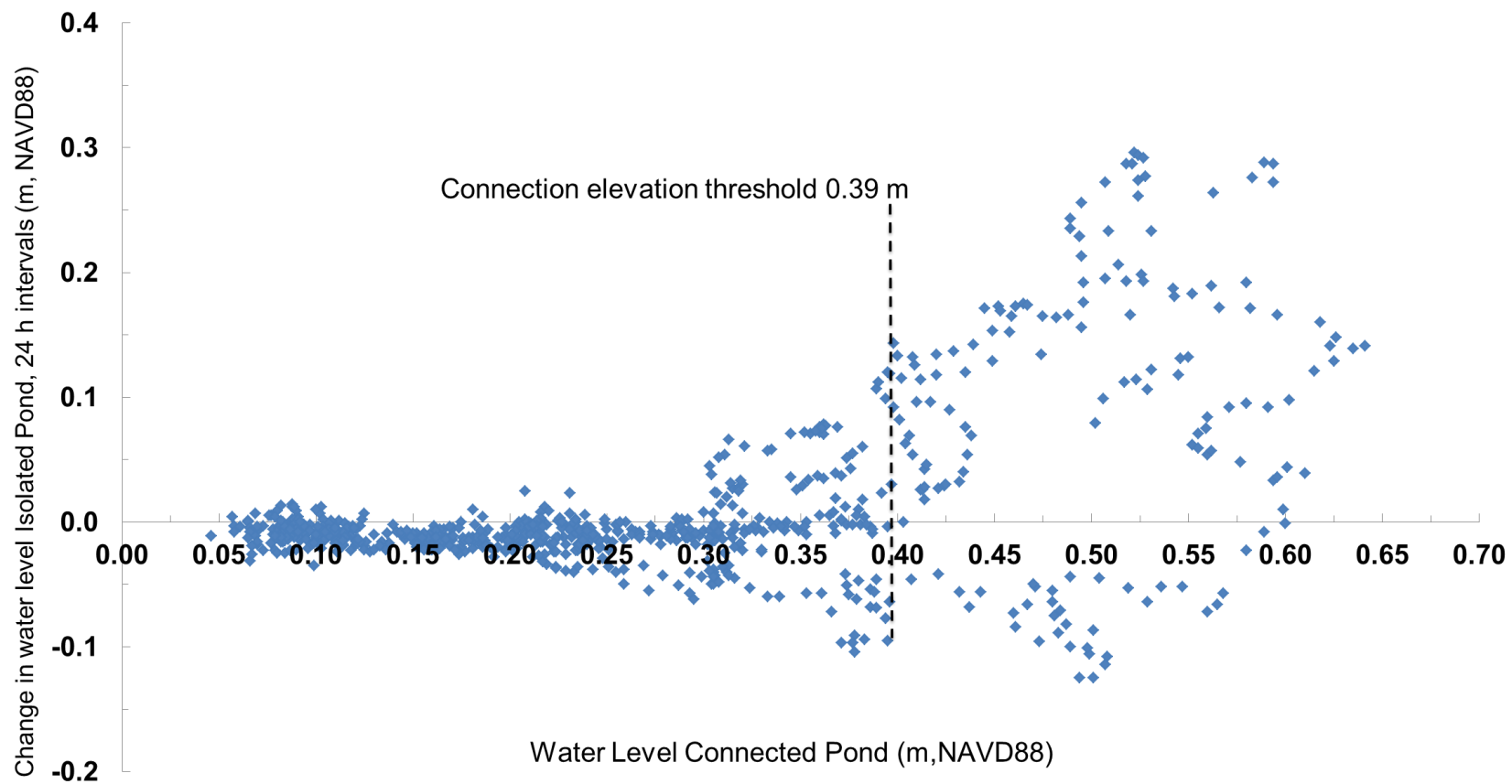
Once the water level in the connected pond dropped below 0.39 m, the ponds became disconnected, and the water level in the isolated pond returned to a linear and decreasing trend. The relation between change in water level in the isolated pond and water level in the connected pond is shown more clearly in Fig. 2-4. Slightly negative values of water level change in the isolated pond are clustered for connected pond water levels < 0.39 m, while relatively larger positive or negative values exist for connected pond water levels > 0.39 m (Fig. 2-4), showing the threshold effect of water levels in the connected pond and the isolated pond.

Rising water in the isolated pond was temporally related to rainfall events (Fig. 2-3). Still, peak water level in this pond often was attained before rainfall, regardless of whether rainfall data came from on-site or from the Palacios station. Fig. 2-3 depicts rainfall from the on-site rainfall gauge. If rainfall was summed for the 24 hours immediately preceding each instance when water level exceeded 0.39 m in the isolated pond, the sum was always $\geq$ 15 mm during that period (Fig. 2-5a). When water level readings at 24 hour intervals were regressed against rainfall totals at the same intervals, water level rise was linearly correlated with rainfall ($R^2 = 0.37$). Much of the residual error was due to water level values between 0.39 to 0.7 m, yet with 0 mm rainfall in the previous 24 hours. When water level readings at 7 day intervals were regressed against rainfall totals at the same intervals, rainfall was even more correlated with water level rise ($R^2 = 0.64$).

Figure 2 - 5. Water level in the isolated pond as a function of (a) rainfall at the site, and (b) hours of easterly winds > 13 km per hour, and in the connected pond as a function of (c) rainfall, and (d) hours of easterly winds > 13 km per hour. Dashed line is representative of the critical connection elevation (0.39 m).
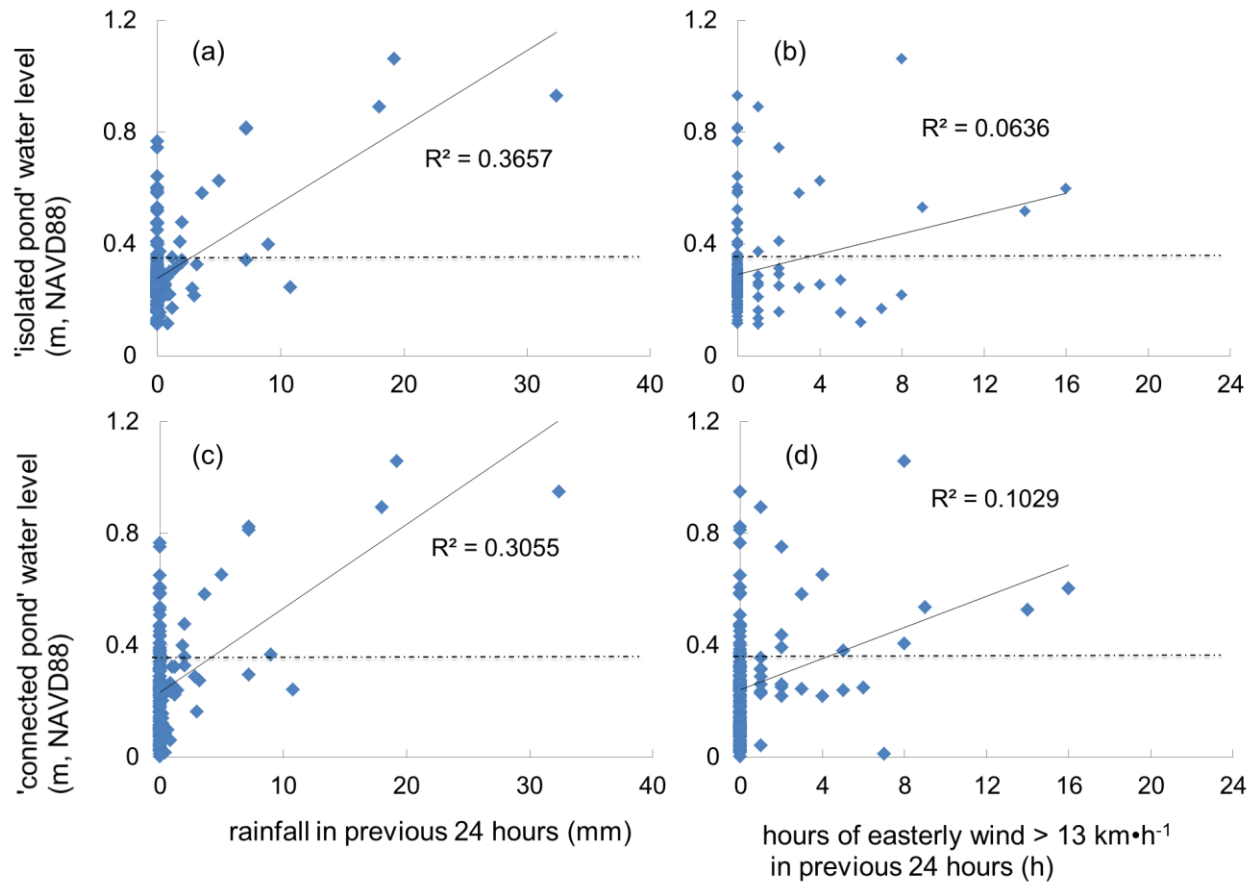
Rising water events in the isolated pond were also temporally related to sustained easterly winds (from 45 to 135 degrees) greater than 13 km per hour (Fig. 2-5). The water level always exceeded 0.39 m in the isolated pond when sustained easterly winds blew for ≥ 9 hours during the previous 24-hour period (Fig. 2-5b). Still, when water level readings at 24 hour intervals were regressed against hours of previously sustained wind at the same intervals, wind was only weakly correlated with rising water ($R^2 = 0.06$). Residual error was high because many periods had a water level well above this threshold, but no strong or sustained easterly wind. All other wind direction and speed combinations were less correlated. Interestingly, water level in the connected pond was less correlated with rainfall ($R^2 = 0.31$) than the isolated pond (Fig. 2-5c), but more correlated with sustained easterly winds (Fig. 5d) than the isolated pond ($R^2 = 0.10$). The connected pond appeared to be somewhat more subject to wind tidal forcing similar to that seen in the larger East Matagorda Bay, as expected (Kraus and Militello 1999).

For the month of March, the total water loss in the isolated pond was 0.112 m (3.61 mm d^{-1}) (Fig. 2-3). Estimation of ET using the Hargreaves equation resulted in a potential ET water loss of 3.59 mm d^{-1} for our site during this month, closely matching the observed trend of 3.61 mm d^{-1}. Thus, we assume that ET drives water level in the isolated pond, during periods with an absence of both rainfall and connection events.

2.4.2   *Stable isotopes and water salinity*

Salinity and isotope sampling of individual ponds revealed an arcing, east-to-west spatial gradient, stretching from the more saline connected pond to the fresher

isolated pond (Fig. 2-6). All water samples fell below the GMWL, reflecting the importance of evaporation in determining the isotopic composition of water in these ponds (Fig. 2-7). We then split all water samples into two clearly discernible categories: those located south and west of sample M20 (average group salinity = 16.8 ppt, Fig. 2-1c) and those located to its north and east (average group salinity = 27.1 ppt, Fig. 2-1c. Sample point M20 fell near the geographic center of the arc of points in Fig. 2- 6. We then graphed the two categories as separate isotopic enrichment lines (Fig. 2-7b). The more saline set of samples was on the connected pond side (M21-M45), while the more fresh set was on the isolated pond side (M1-M20). An analysis of the $\delta^{18}O$ versus salinity plots (Fig. 2-8) reveals different interactions for each group (isolated and connected). On the M1-M20 side, the data shows a mixing trend between freshwater inputs for the site (rainfall, $\delta^{18}O$ =-3.6‰ and $\delta^{2}H$ =-27‰, (Bowen and Wilkinson 2002)) and oceanic water (35 ppt, $\delta^{18}O$= 0), influenced by evaporation of both water sources (Fig. 2-8).

Figure 2 - 6. Map of water sample results. (a) Salinity and (b) δ2H values from sampled ponds
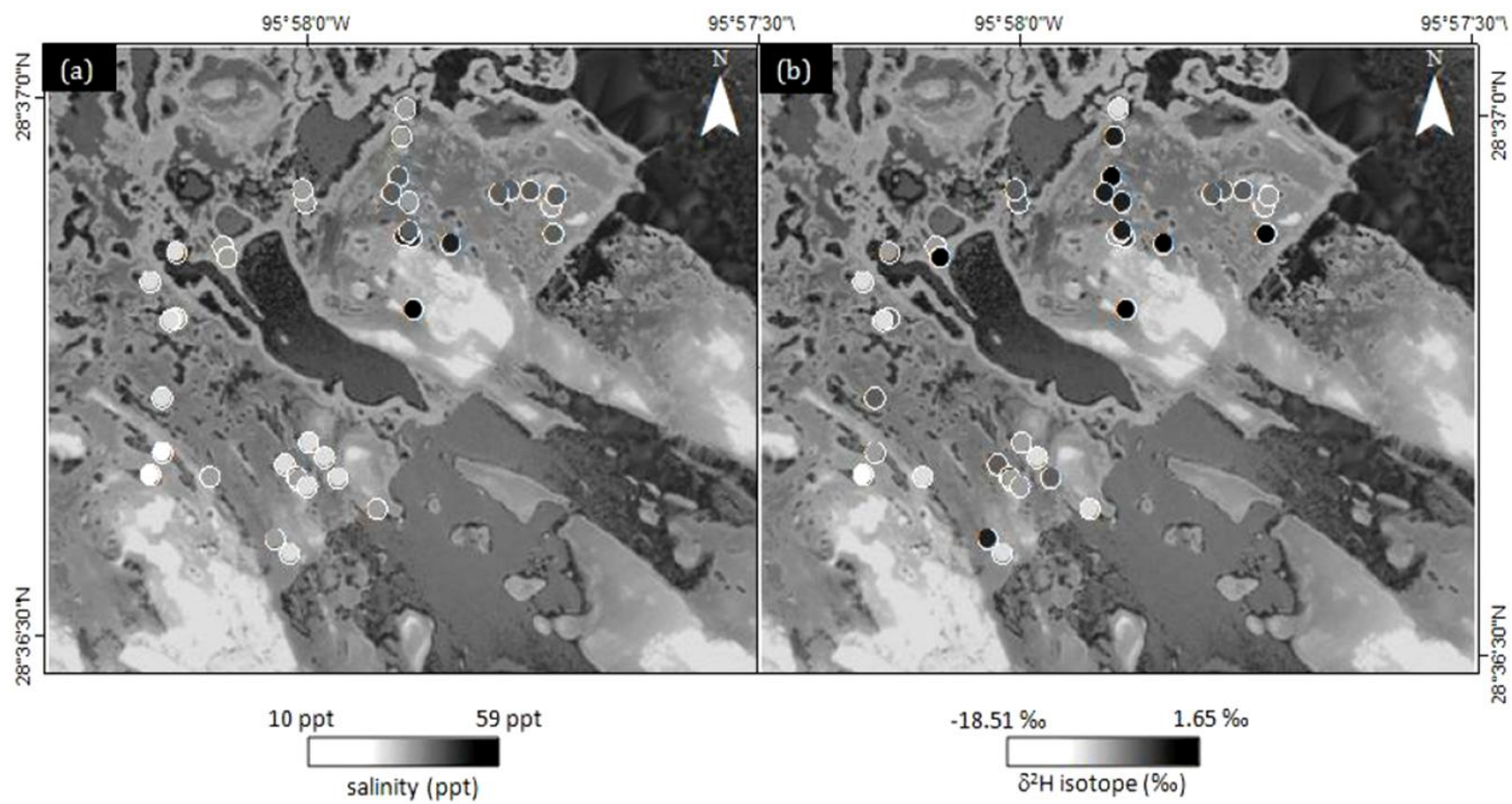
This relation is consistent with other δ^{18}O-salinity relations in open estuaries in the Gulf of Mexico (Surge and Lohmann 2002, Hyeong and Lawrence 2004) and gives validity to the connectivity patterns proposed by our LIDAR elevation model (see below). However, it is evident that there is no relation between δ^{18}O and salinity on the M21-M45 side (Fig. 2-8b).

2.4.3 LIDAR elevation model

When water levels were low, ponds were isolated (Fig. 2-9a). As water levels rose, ponds connected to each other first through a network of small tidal creeks and connections from one pond to the next, and then finally through overland flow as water continued to rise (Fig. 2-9b). The two large, primary ponds connected across sample locations 20-23 (Fig. 2-9c). Based on the LIDAR model analysis, this location appears to be the last section of the smaller pond network to be overtopped by rising water levels, allowing the isolated and connected ponds to connect.

Figure 2 - 7. Isotopic enrichment lines for (a) the entire site, GMWL, and rainfall; and (b) sample ponds M1 through M20 (more fresh, isolated side) and M21 through M45 (more salty, connected side).
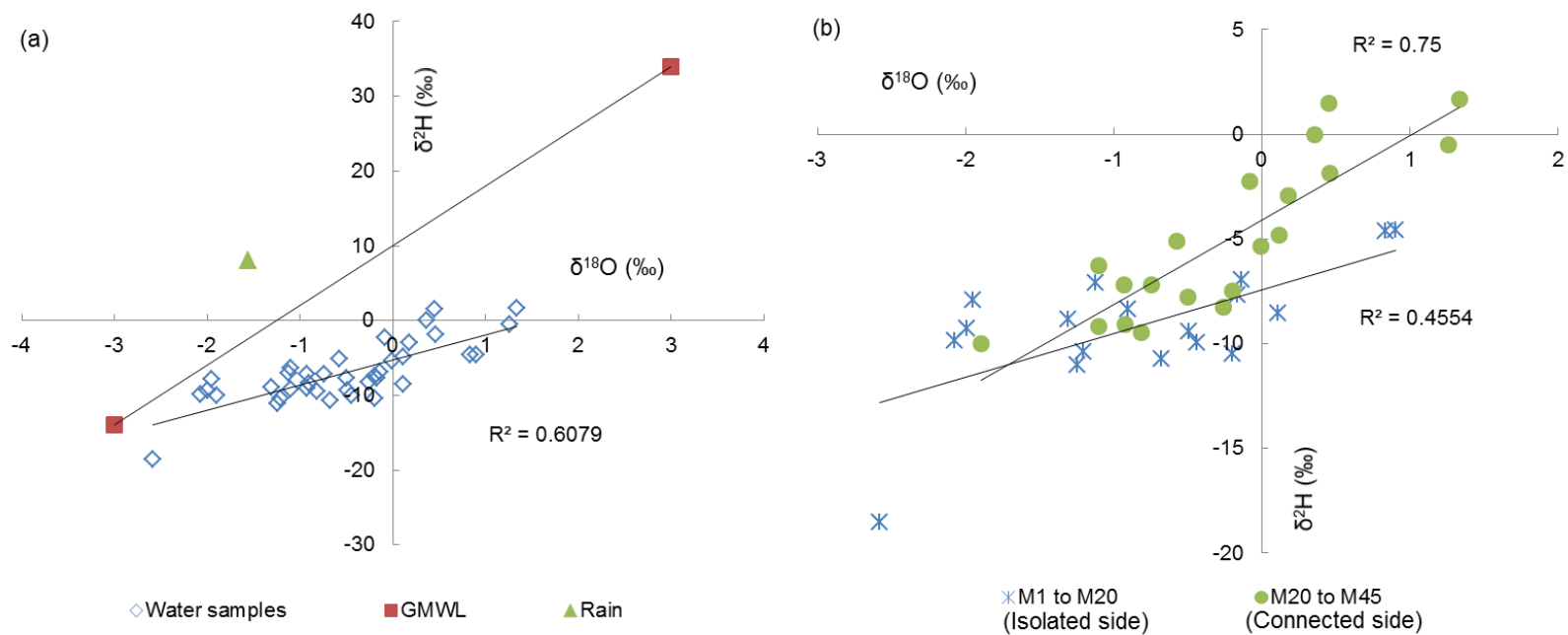
Figure 2 - 8. $\delta^{18}O$ versus salinity scatter plot for ponds (a) M1-M20 and (b) M21-M45
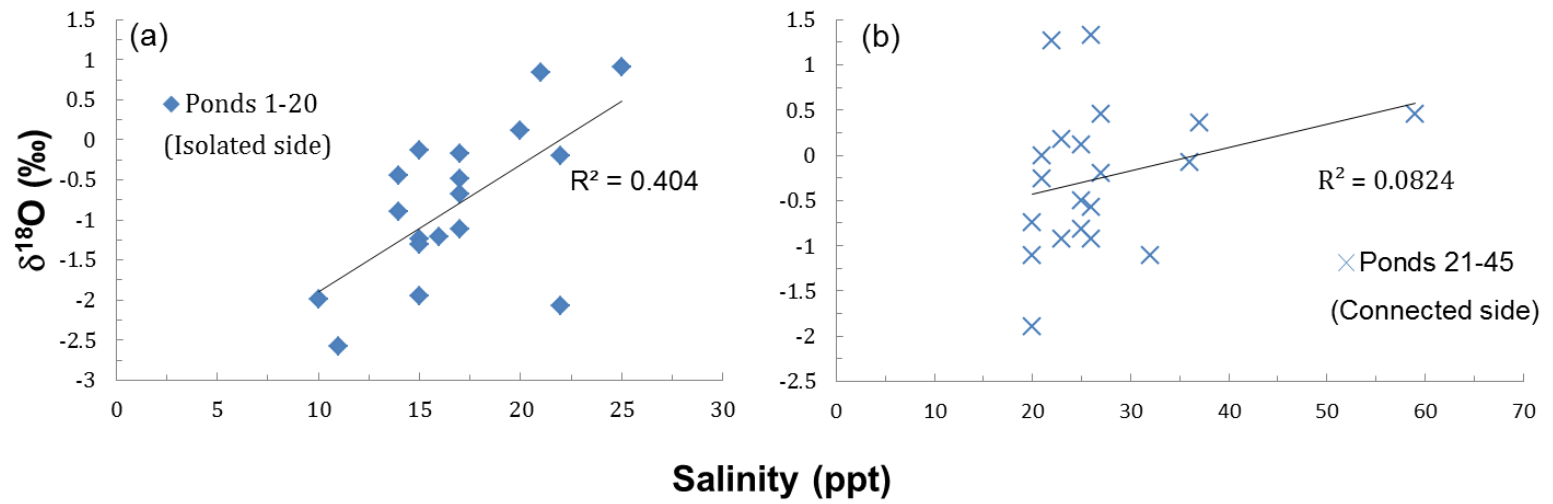
The threshold water elevation at which this connection occurred was 0.39 m (Fig. 2-9d) for all ponds except ponds M13 and M15, which remained disconnected after water level reached this elevation. The salinity and δ^2H isotopic value of each individual pond was correlated with the threshold water elevation at which they connected to the rest of the hydrological network ($R^2 = 0.42$ and 0.37 respectively, Fig. 2-10).

2.5   Discussion

Hydrological connectivity among wetlands is extremely important in the context of policy and management, where wetlands that do not show a significant nexus to navigable waters are not protected by federal law. However, surface water flow across landscapes is not always uniform, and rainfall induced subsurface flows can 'spill' on different parts of a watershed depending on spatial patterns of transient saturation and geological features (Meerveld and McDonnell 2006). Furthermore, by analyzing a four-year data set of surface water runoff, Wilcox et al. (2011) demonstrated that wetland depressions in the Gulf Coast, previously thought to be isolated, have a measurable connectivity to adjacent waterways. Their results demonstrate that wetlands in the Gulf Coast have a particularly complex relation with other water bodies. Nonetheless, our rests seek to understand connectivity at a finer scale, within the salt marsh itself. At this scale, knowledge about important ecological processes can benefit from studying hydrologic connectivity.

Figure 2 - 9. Connectivity of ponds at the study site at (a) 0.20 m, and (b) 0.45 m water elevation. Gray is water; white is land. Stars denote primary pond locations, water sample points are labeled alpha numerically. (c) In the LIDAR elevation analysis, the path of connectivity from the connected pond to the isolated pond appears to flow in an arc-shaped pattern. The connectivity elevation threshold occurs spatially at sample points M20-M23. Arrow denotes hypothesized connection path. Large ellipse denotes elevation threshold location. (d) Elevation at which each pond connected to the rest of the hydrological network of the salt marsh, 0.39 m was the threshold water elevation at which all ponds were connected (except M15 and M13). Large ellipse corresponds with pond sample points M20-M23. Dotted line highlights the 0.39 m water elevation threshold.

(a) water level = 0.20 m
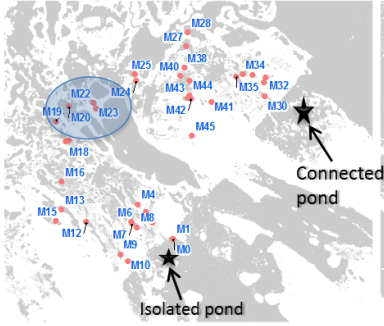
(b) water level = 0.45 m
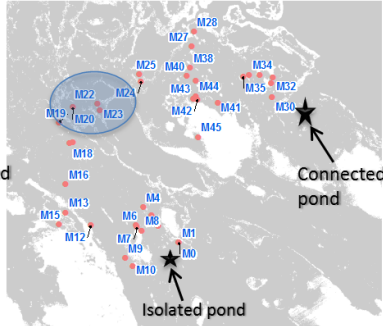
(c) hypothesized connection route

Connected pond

Isolated pond

(d)

critical connection location

critical connection elevation (.39

more fresh, 'isolated' ponds on the southwest

more salty, 'connected' ponds on the northeast

elevation (m NAVD88)

individual pond (sample number)

M15 M13 M4 M5 M6 M7 M3 M16 M9 M12 M2 M17 M18 M19 M1 M8 M10 M20 M22 M23 M35 M31 M43 M32 M25 M30 M42 M41 M24 M33 M34 M45 M41 M39 M28 M27 M40 M38

Figure 2 - 10. Salinity and δ²H isotopic value of each pond versus the threshold water elevation at which they connected to the rest of the hydrological network.
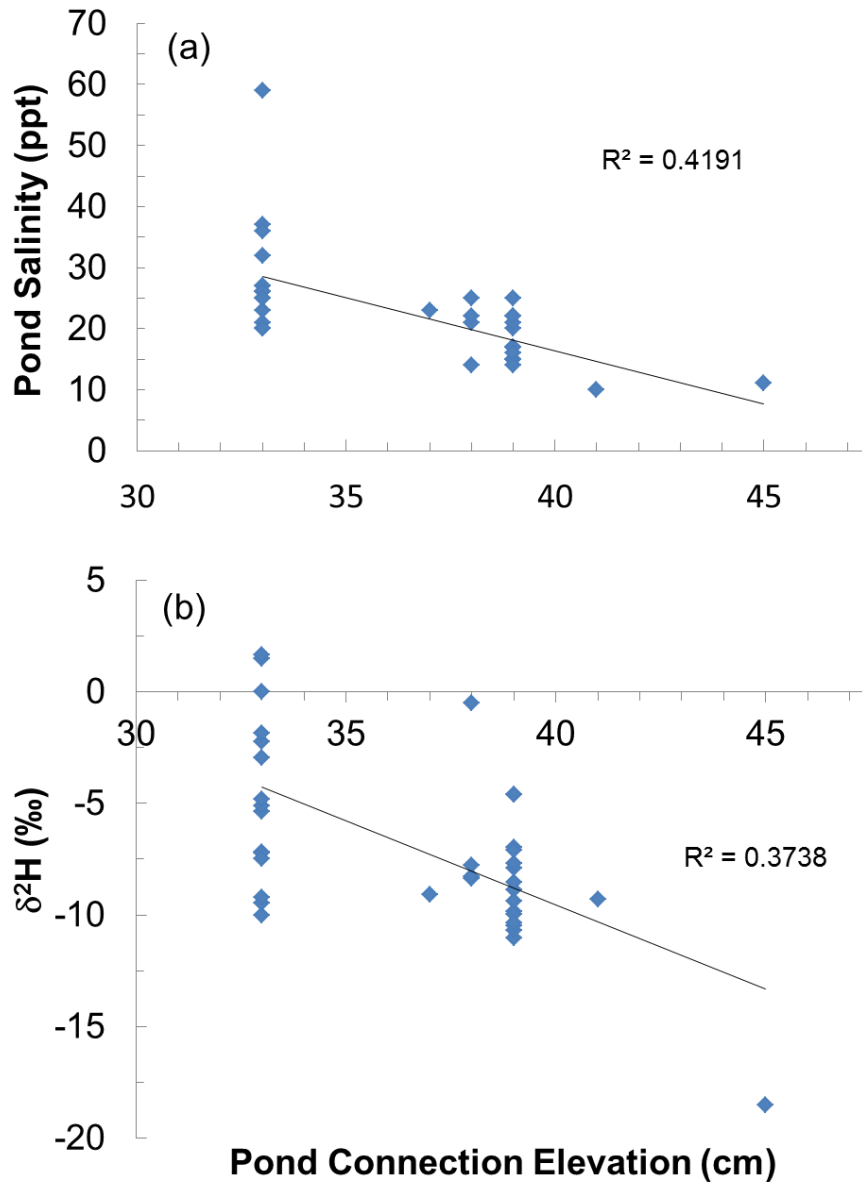
For example, it is known that tidal creeks in salt marshes serve as important nurseries for blue crabs (*Callinectes sapidus*). Inundation of these tidal creeks varies across the landscape depending on its topographical and hydrological features. Therefore, surface water connectivity has the potential to affect the availability of tidal creeks for nursery and dispersal of blue crabs across the salt marsh landscape. Likewise, the absence of blue crabs could translate in the loss of marsh vegetation cover due to the proliferation of the marsh periwinkle (*Littoria irrorata*), an herbivorous snail whose populations are controlled by blue crab predation (Silliman and Bertness 2002). The presence of surface connectivity in salt marshes is also relevant for metapopulation dynamics studies. Transport through surface water connections is considered a mechanism by which wetlands can support metapopulations at regional scales (Leibowitz and Vining 2003). In addition, Kudoh and Whigham (2001) showed that plant populations adjacent to tidal streams in a salt marsh were part of a metapopulation that served as a reserve of genetic variation in the system. Again, the ability of a salt marsh to maintain a functional ecosystem across its structural landscape may be determined by the intermittent nature of the hydrologic connections between ponds, tidal creeks and adjacent water bodies.

The objectives in our study were aimed at understanding surface connectivity of ponds within a salt marsh by answering three basic questions regarding this issue: (1) What is the main driver of connection events between the two primary ponds?, (2) How much saltwater or freshwater influence can we expect in different areas of the marsh?,

and (3) How do smaller, individual ponds connect among themselves and the primary ponds? Regarding the first question, water level data showed the behavior of the two primary ponds once they were connected, an event that could also be inferred from the spatial distribution of isotopic values of water among the network of ponds. However, none of the variables analyzed as potential drivers of connection events (wind and rainfall), stand out as the likely unique driver. It is likely that water level in the connected pond is also affected by additional drivers not considered in this study, an aspect of salt marsh connectivity that could benefit from further research (for example oceanic eddy-driven water level changes at a regional scale, water level set up or surge from offshore storms, growth fault movement of the landscape).

According to the isotope data, the influence of saltwater is stronger in the ponds near the connected pond, reflecting its marine source, whereas rainfall appeared to have a stronger influence in the ponds near the isolated side, where the less saline samples were also taken. However, isotope data suggests that freshwater sources (rainfall) undergoes a mixing process with the waters already present in all the ponds (regardless of their location), possibly diluting the hypersaline ponds in the process. Analysis of isotopic data proved particularly useful in the identification of freshwater sources in ponds where salinity data was at an intermediate scale and did not provide sufficient evidence to infer a direct influence of freshwater. Water isotope analyses can benefit connectivity studies in coastal systems that usually rely on salinity measurements to

study the extent of freshwater influences by providing detailed information on freshwater sources and it interactions with saltwater.

The DEM connectivity model provided valuable information on the connection process of the individual ponds. The integration of the DEM model allowed us to determine that the elevation threshold at which all the individual ponds connected was 0.39 m, a result that could only be inferred by the water level data gathered for the first objective, and that was unattainable by just obtaining water samples. By analyzing the subtle elevation gradients across different areas of a wetland, connectivity studies in coastal systems can create detailed quantifications of hydrological connectivity across several scales.

The interaction between the isolated pond and the connected pond, and their relation to the hydrological network connecting the smaller ponds between them, was quite complex. A fine balance appears to exist between the freshwater influences across the landscape and the saltwater influences from the connected pond. When the two primary ponds were disconnected, the isolated pond water level was primarily influenced by evaporation, and to a lesser degree, rainfall. Though there was a correlation between water level and rainfall ($R^2 = 0.37$ for the 24 hour interval), a detailed inspection of the dataset shows that this was not necessarily a causal relationship. For example, considering the most intense daily rainfall event, during which water levels did not exceed the 0.39 m water elevation (on a single day in March),

the water level increase after rain was approximately 0.02 m for the isolated pond. We interpret this as a response of the isolated pond to rainfall.

The connected pond water levels were likely primarily driven by winds and astronomical tides. Given a detailed inspection of the dataset, rainfall never caused water to rise in the connected pond. Though the R^2 values were relatively lower for the easterly wind as a driving factor of water rise, a detailed inspection showed that several of these wind events appeared causal for this pond. Still, there were many instances when water rose in this pond, but there was no strong sustained easterly wind. It is clear that when the connected pond water level exceeded the 0.39 m threshold value, water poured into the isolated pond. This appeared to be a one-way relationship driven by the connected pond, as water level increases and decreases always preceded the isolated pond by a several hours. It is likely that saltwater, and the species and materials associated with it, are introduced to the isolated pond exclusively from the connected pond.

Still, there were some instances when the two ponds became connected, yet no rain fell and sustained, strong easterly winds were absent. Astronomical tides, in combination with a weaker easterly wind tidal effect, may have increased or dampened the ability of the connected pond to overtop its banks and spill into the isolated pond. Yet, the astronomical influence appears to be small, around 0.05 m in amplitude. A possible explanation for these events is that connection between the connected pond and the isolated pond is facilitated by the water level in the isolated pond. Water level in the

isolated pond only starts to rise when the water level in the connected pond reaches the 0.39 m threshold. This is because when water level reaches this elevation, it starts filling the isolated pond until they are both at the same water level and start to behave as one single pond. Hence, if the water level in the isolated pond is still relatively high (from previous connection events), it fills faster and the connection event occurs more quickly, even with less intense conditions of wind and rain. In all of the connection events that could not be directly attributed to rainfall or easterly winds, water levels in the isolated pond were approximately 0.10 m higher than water levels preceding connection events directly attributed to the mentioned factors.

Stable isotopes contributed to our understanding of connectivity by revealing the different relations between $\delta^{18}O$ and salinity of the two sets of ponds and the manner in which δ^2H is distributed across the landscape. The different relation between $\delta^{18}O$ and salinity of the two sets of ponds is an indication of different interaction among its water sources. Analysis of the $\delta^{18}O$-salinity scatter plot of ponds M21-M45 reveal a more complicated scenario of freshwater and saltwater mixing that needs further research. The spatial distribution of δ^2H across the salt marsh suggests that individual ponds near the isolated pond were primarily influenced by rainfall at the time of sampling. Although water level data clearly demonstrates that water levels in the isolated pond are influenced either by evaporation or by spillage from the connected pond, it is important to view this data in the climatic context of the water sampling date (March 1, 2010). Measured rainfall for the site during the three months before the sampling date

(December 2009-February 2010) was almost double than their monthly averages. This would explain the lower δ^2H values in individual ponds near the isolated pond and the absence of hypersaline conditions at the isolated pond during the sampling period. It is likely that isotope and salinity values in the isolated pond and its associated individual ponds would have been higher if the sampling had taken place during the final days of June 2010, after a long period of evaporation and lack of rainfall that was preceded by a connection event near the end of May. Longer-term studies in combination with frequent water samplings should be able to provide more information regarding this relation between rainfall, water level and salinity in the isolated pond.

It is noteworthy, however, that stable isotope studies of saline surface waters cannot ignore the important role of salinity in isotope fractionation. Salinity effects on isotope fractionation vary with temperature, salt concentration and salt composition of the mixture, affecting the process of isotopic enrichment during evaporation (Gat 2010). When salt concentration exceeds that of seawater appreciably, the enrichment process is reduced relative to a freshwater body evaporating under the same conditions. This is primarily because atmospheric vapor pressure is reduced by an increase in salinity. However, given the spatial scale of our study, we can assume that salinity effects on fractionation were similar throughout our studied landscape and that they could account only for variability on samples that had considerable salinity (less than two samples had > 35 ppt). Consequently, the arcing, east-to-west spatial gradient of the water isotope data was consistent with the connection path proposed by the LIDAR elevation model

and our conclusion that freshwater inputs are more influential on the west side of the pond network. The fact that individual ponds M13 and M15 (the two ponds with the lowest salinity and the most depleted δ^2H values) sat above even the 0.39 m connectivity threshold according to the LIDAR elevation model, suggests that elevation is a determining factor for pond connectivity in a salt marsh. In the case of ponds M13 and M15 we can infer that ponds that connect at higher elevations are less influenced by ocean water.

It is important to note the possible influence of groundwater recharge as an explanation for the water loss in the isolated pond. The influence of groundwater in a salt marsh can vary spatially or temporally, depending on the location and specific hydrologic conditions at the site (Wilson et al. 2011). However, in addition to our observation of no apparent groundwater exchange through the use of PVC piping, the observed rate of water loss closely matched the estimated rates of ET for Matagorda, suggesting that no water was lost to groundwater recharge.

2.6   Conclusion

The findings presented in this paper demonstrate that the hydrology of coastal ecosystems can be better understood by applying integrated methods. An integration of *in situ* hydrological data and spatial representation of water characteristics (salinity and isotopes ratios), along with the fact that each individual pond was analyzed through a hierarchical representation of the connectivity process, allowed us to inspect pond connectivity in detail. For instance, the DEM model allowed us to obtain the 0.39 m

threshold value; however, it did not provide information about other hydrological aspects of the marsh, like the evaporative nature of the isolated pond or the intermittence of the connectivity events. Salt marshes in dry subtropical areas represent excellent research models to study connectivity and its effects on ecosystem structure and function (Sheaves and Johnston 2008). Our results suggest that the methods we integrated to study connectivity in a salt marsh can be applied to most ecosystems across different spatial scales.

3. IMPLICATIONS OF SALTWATER INTRUSION FOR THE HYDROLOGIC MANAGEMENT OF *PTEROCARPUS OFFICINALIS* FORESTED WETLANDS

3.1   Overview

Freshwater forested wetlands in the Caribbean are often dominated by the leguminous tree *Pterocarpus officinalis*. Deforestation and changes in land use have limited the extension of *Pterocarpus officinalis* forests to genetically isolated populations, primarily restricted to the coast. Although it is well known that increased salinity affects important ecological functions of these wetlands, little is known about the hydrologic components that govern saltwater intrusion in coastal *Pterocarpus officinalis* forests. We examined the extent of tidal forcing and saltwater influence in the *Pterocarpus officinalis* swamp of the Humacao Natural Reserve, Puerto Rico. Automated water level and conductivity recorders were installed across a tidal creek transect within the *Pterocarpus officinalis* forest at four different distances from the ocean from January 2011 to May 2012. Across the transect, water level responded similarly to high rainfall events. Still, for moderate rainfall events, water level increased at upstream locations only. The influence of a saltwater wedge was observed at all instrument locations. However, salinity fluctuated on a 24-48 hour period in response to rainfall at the downstream locations, whereas it did not for the deeper parts of the forest further upstream. Seasonal increases in salinity were evident at all locations, with high

salinity related to the dry season when both rainfall and mean sea level were low. During this season, a sand barrier formed at the mouth of the tidal creek, and though there was no overland connection to the ocean, salinity increased. Subsequently, rainfall and increased outflows re-established the connection to the ocean and the saltwater wedge receded from the depths of the forest. Our work demonstrates that for preventing saltwater intrusion into the Humacao Natural Reserve, the amount of rainfall is more important than the sea level or the existence of an overland hydrological connection to the ocean. Our work suggests that preventing saltwater intrusion with barriers is unlikely to be successful for *Pterocarpus officinalis* forest conservation in the Caribbean. Rather, upstream freshwater drainages will need to be restored to their former state and remnant canal networks will need to be re-graded to allow sheet flow to once again reach these forests.

3.2   Introduction

Forested wetlands are a regular feature of low-lying coastal areas in the Caribbean (Bacon 1990). Mangroves dominate most of these wetlands except for forested areas influenced by freshwater, where the leguminous tree *Pterocarpus officinalis* (hereby *Pterocarpus*) is the main species. Adapted to flooded ecosystems, *Pterocarpus* inhabits river floodplains, coastal basins and subtropical rain forests (Alvarez-López 1990). *Pterocarpus* wetlands are now limited to small genetically isolated patches that are scattered throughout the Caribbean region, due to human disturbance and clearing (Rivera-Ocasio et al. 2006, Dulormne et al. 2009). Although

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the floristic composition of these wetlands has been well described by Alvarez-López (1990) and Migeot and Imbert (2011), these ecosystems have not received the same research attention as mangroves or upland rainforests (Migeot and Imbert 2011).

In the last century, the island of Puerto Rico lost nearly all of its *Pterocarpus* forested wetland cover (Helmer 2004). Today, the total area of *Pterocarpus* cover is only 261 hectares based on 2000 Landsat imagery classification (Gould 2007). Furthermore, remnant *Pterocarpus* wetlands in Puerto Rico are restricted to the coast, abutting mangrove ecosystems (Cintrón 1983). In Puerto Rico, most of the *Pterocarpus* cover is classified as estuarine according the National Wetlands Inventory (NWI) (Table 3-1). The remaining stands now occur near their ecological limits in term of salinity (Rivera-Ocasio et al. 2007). Currently, sea level rise and associated saltwater intrusion are the most serious threats affecting the *Pterocarpus* ecosystem. Elevated soil and water salinity is known to affect litter, flower and fruit production (Eusse and Aide 1999). Salinity also affects recruitment and growth in adults and seedlings, and even small changes in salinity (>1‰) reduced nodulation in the roots of seedlings (Saur et al. 1998a, Fougnies et al. 2007, Rivera-Ocasio et al. 2007, Dulormne et al. 2009). Still, *Pterocarpus* has some adaptations that allow the tree to tolerate and avoid lower level salinities (Eusse and Aide 1999). Buttress roots create mounds of sediment that modify the micro-topography of the forest (Kaponen et al. 2004) and these trees accumulate Na in the rachis of the leaf, away from photosynthetic tissue (Medina et al. 2007).

Moreover, they have a shallow root system that is likely driven by a strategy to avoid saltwater intrusion into the lower depths of the water table (Medina et al. 2007).

At the current rate of eustatic sea level rise and the current frequency of hurricanes, losses of tropical forested wetlands (Bacon 1990, Ross et al. 2009) and resultant impacts to the built environment (Lewsey et al. 2004) are already a major problem in Puerto Rico and island nations in the Caribbean. It has been projected that the rate of eustatic sea level rise may further

Table 3 - 1. NWI classification of *Pterocarpus* cover in Puerto Rico.

NWI wetland classification	Area (ha)	*Pterocarpus* cover (%)
Estuarine deep water	0.00537	0.002097
Estuarine and Marine wetland	255.4027	99.71783
Freshwater emergent wetland	0.510524	0.199326
Freshwater forested/shrub	0.206803	0.080743

accelerate under continued global warming, due primarily to the thermal expansion of the ocean (IPCC 2007, Church et al. 2011).

We do know that sea level rise will bring more salt water into *Pterocarpus* dominated ecosystems, yet we have a poor understanding of how these important wetlands will respond other than the prediction that they will either migrate landward or be lost (Brinson et al. 1995). Seasonal precipitation and freshwater inflow sources have

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not yet been discussed as a potential mediator of saltwater intrusion in *Pterocarpus* forests, though they have for coastal dune plants (Greaver and Sternberg 2006, 2007, 2010) and hardwood hammock tree species (Sternberg and Swart 1987, Ish-Shalom et al. 1992). An understanding of the primary forces that control the sustainability of these environments is critically needed.

The primary objective of this study was to quantify the spatial and temporal variations of saltwater intrusion into the *Pterocarpus* forest of the Humacao Natural Reserve (HNR). Specific research questions included: What is the seasonal nature of saltwater intrusion into the HNR? What is the relation between observed patterns of salinity and other hydrological factors? Are there potential management strategies that can be used to ameliorate an increasing likelihood of saltwater intrusion due to sea level rise?

3.3   Methods

3.3.1   *Study site*

The Humacao Natural Reserve (HNR) in Puerto Rico contains the largest and best preserved *Pterocarpus* forest in the United States, yet the forested stand is only 150 ha in size (this is still over half of all its coverage in Puerto Rico, (Alvarez-López 1990, Gould 2007)). Historical images of the *Pterocarpus* forest reveal land use changes in this portion of the HNR area since 1956 (Fig. 3-1). During the first decades of the 1900's, most of the *Pterocarpus* forest in the present location of the HNR was cleared.

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The only remaining stand is visible as dark canopy cover in the historical images. Also, during this period a series of ditches and channels were constructed to drain the valley and plant sugar cane crops. The forest was cut in the area immediately surrounding the main river meander (Fig. 3-1, 1951). Subsequently, the crops were abandoned following the industrialization of Puerto Rico in the 1950's and 1960's. The abandoned terrain was flooded periodically following intense rainfall events and some *Pterocarpus* re-colonization can be observed in the exposed meanders of the river in the subsequent image of 1978 (Fig. 3-1). In 1979, after Hurricane David and Tropical Storm Frederick flooded the abandoned fields, a series of permanent coastal lagoons formed (Fig. 3-1, 1997).

The HNR received its official reserve designation in 1986 after the development of these lagoons. However, the community of Punta Santiago was located immediately between these lagoons and the ocean. This residential area flooded repeatedly, every time enough runoff was accumulated in the lagoons. Subsequently in 2000, the US Army Corps of Engineers (USACE) modified the Antón Ruiz River and several other drainages, opening up a connection between the lagoons and the ocean (Fig. 3-1, 2010), and allowed flood waters to flow into the ocean (Schwartz 2004). Unfortunately, this has also increased salt water intrusion into the lagoons and the adjacent *Pterocarpus* forest (Ferrer 2007). The remnant *Pterocarpus* area is now threatened by salt water intrusion due to rising global sea levels, a problem that will be exacerbated by climate change over the next 100 years (Rivera-Ocasio et al. 2007).

3.3.2   Tidal gauge sampling

To investigate the seasonal extent of saline water intrusion into the forest we established a primary transect from the freshwater 'upstream' direction to the saltwater 'downstream' direction, that ranged from an area of 100% *Pterocarpus* coverage to an area of 100% mangrove coverage across the tidal creek of the Antón Ruiz River estuary (Fig. 3-2). We selected this area because it is where the freshwater outflows of the forest interact with the saltwater inflows from the ocean. Four permanent stations (TG1, TG2, TG3, TG4) were established along this transect at different distances from the entrance of the estuary. At each station, specific conductivity (μS/cm), water depth (cm), and temperature (^{0}C) were recorded hourly over the course of the study period at monitoring wells. The study period spanned from January 1st 2011 to May 22nd 2012 on all stations except for TG3, which started recording date on April 20th 2011 due to instrument malfunction and replacement. Monitoring wells consisted of a Schlumberger CDT-Diver sensor (Schlumberger 2012) suspended inside a 2" diameter PVC pipe with horizontal slits to allow adequate flow of water through the sensor. An atmospheric pressure gauge was installed above water level to compensate for barometric pressure and convert pressure data into water-column depth above the instrument. Specific conductivity readings were converted to practical salinity units (psu) according to Fofonoff and Millard (1983).

Figure 3 - 1. Historical development of the *Pterocarpus* forest in the HNR (1956, 1978, 1997 and 2010). Additional features identified are: sugar cane crops and *Pterocarpus* deforestation location (1951); abandoned fields and *Pterocarpus* recovery area (1978); Las Mulas creek, coastal lagoons, Antón Ruiz River estuary and Punta Santiago Community location (1997); mangrove encroachment, US CoE drainage channel and study area location (2010).
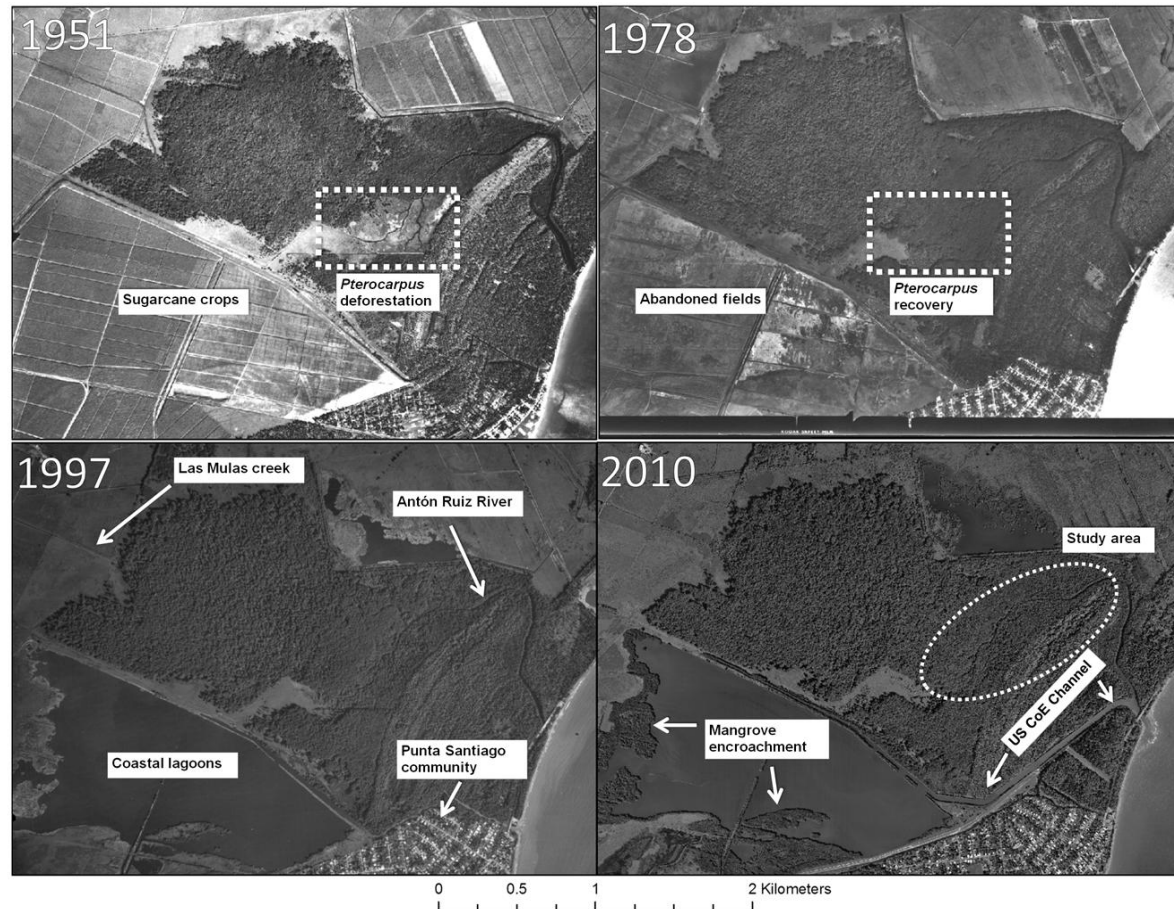
Figure 3 - 2. Tidal gauges location along the estuary of the Antón Ruiz River in the HNR
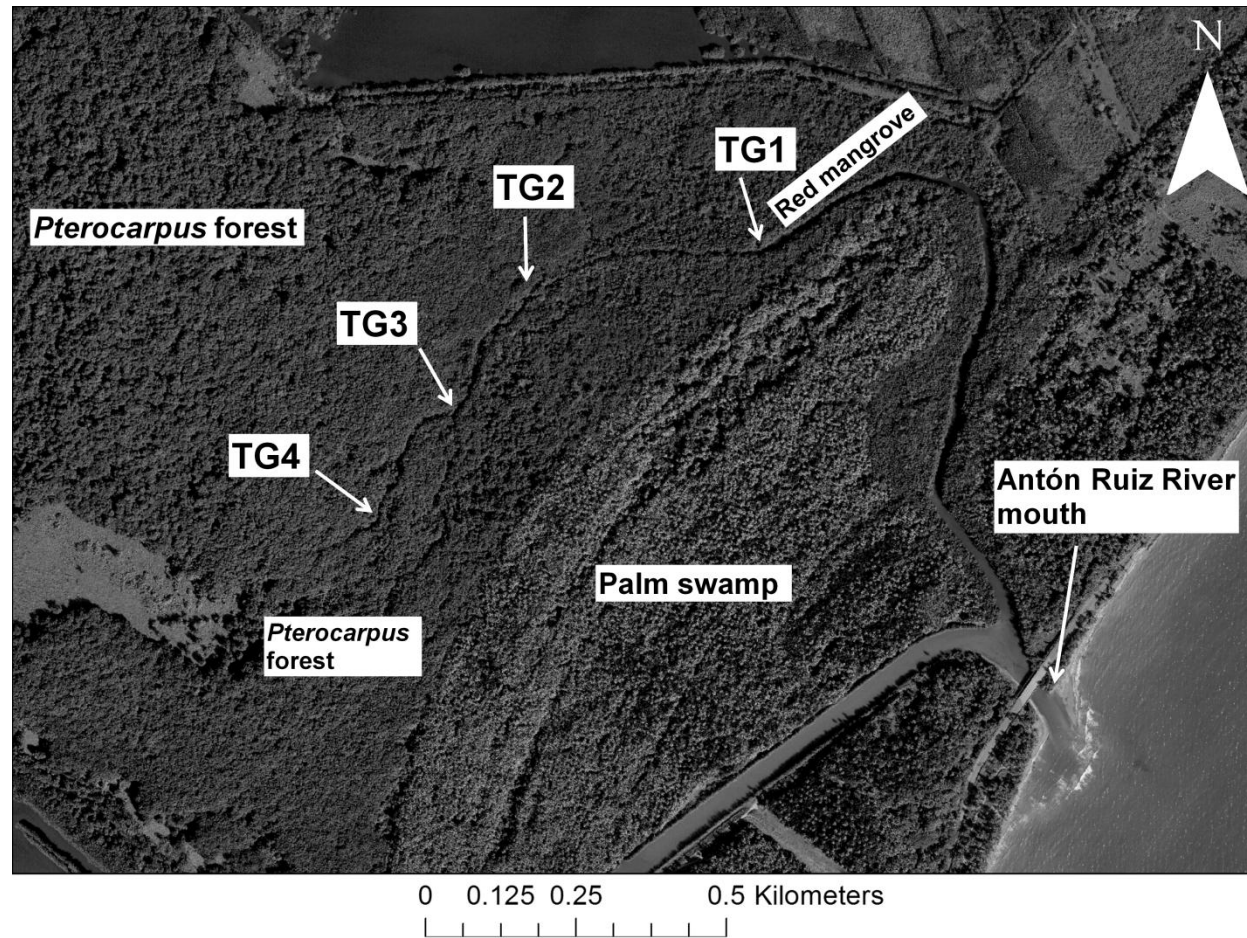
Hydrographs of the time series were constructed using Sigmaplot 12.0 analysis software. In order to complement our analysis we obtained daily precipitation data from the Juncos 1 SE National Weather Service station (NWS, station JNCP4). Monthly mean sea levels were obtained from a sea level monitoring buoy in Fajardo, PR (NOAA station FRDP4 – 9753216). Both of these stations are located within 30 miles of the HNR on the east coast of Puerto Rico. Daily rainfall and salinity data was analyzed by a Pearson correlation analysis for each instrument location (Minitab, Inc. 2013).

3.4   Results

Monthly averages of rainfall computed from the nearby Juncos station reveal the occurrence of the dry season during the late winter and early spring months (February-March-April, Fig. 3-8). Similarly, monthly averages of the Mean Sea Level (MSL) variations of nearby tidal stations suggest that during this time of the year MSL is at its lowest (Fig. 3-9). Also of note, the rainy season in 2011 had exceptionally high amounts of rainfall at the Juncos station, in particular during the summer months, and August being recorded as the wettest month on record (National Weather Service 2011). Sea levels generally increase during these months as well.

Wetland water level response to rainfall events was consistent for all instrument locations, showing peaks and valleys that often corresponded to rainfall events (Figs. 3-3 to 3-6). However, response to particular rainfall events varied between TG1 and the remaining three instruments during the study period. While TG1 showed peaks in water level corresponding only to rainfall events larger than 80 mm, TG2, TG3, and TG4

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showed water level peaks corresponding to all rainfall events larger than 40 mm (Figs. 3-3 to 3-6). Overall, water level response to rainfall appeared highest in downstream instruments (TG1 and TG2).

The overall tidal amplitude decreased with instrument distance to the mouth of the estuary (~30 cm (TG1) to ~5 cm (TG4)), and varied in a biweekly pattern that corresponds to extreme tides events (spring and ebb). Nevertheless, there is a loss of the tidal signal during the dry periods of the study (Feb-Mar-April for both 2011 and 2012). This loss in the tidal signal is an indication of the closure of the tidal creek's connection to the ocean due to the formation of a sand barrier at the outlet to the ocean. This barrier is fed by sand coming from the littoral drift along the adjacent beach, and accumulates initially in a bar. In the absence of sufficient estuarine outflow, this bar forms into a subaerial barrier (Fig. 3-7). Accordingly, periods of tidal signal loss are concluded by the onset of rainfall events that exceed 20 mm, for example on May 21st 2011 or March 26th 2012 (Fig 3-3 to 3-6). This occurs because the runoff provides the necessary force to remove the sandbar and re-establish the connection to the ocean.

Salinity fluctuations varied with distance to the ocean as well. TG1 and TG2 showed an average salinity near 25 psu throughout most of the study period. The pulsed decreases in salinity corresponded to peaks in water levels caused by rainfall events ($P = 0.012$ and $P = 0.039$, respectively). Once the water level peak of the corresponding rainfall event had passed, salinity of TG1 returned to near 25 psu within 24-48 hours

(Figure 3-3). Furthermore, the TG1 hydrograph shows no evident increase in salinity during the drier periods typical of the dry season (February-March-April for example).

In contrast, TG2 showed a similar decrease in salinity following rainfall events, but with a longer period of reestablishment to higher salinities (~ 7 days, Fig. 3- 4). In addition, the most evident increases in salinity in this location occurred during periods of low tidal influence and the drier times of the study period (rainfall < 20 mm, dry season). Salinity fluctuations of TG3 were influenced by bi-weekly occurrences of extreme tides and seasonality of tidal amplitude. In addition, rainfall events had minimal influence on changes in salinity ($P = 0.295$). Although some rainfall events were linked with a decrease in salinity, other events were followed by short periods of increased salinity.

Figure 3 - 3. Rainfall, water level, and salinity hydrograph for instrument location TG1.
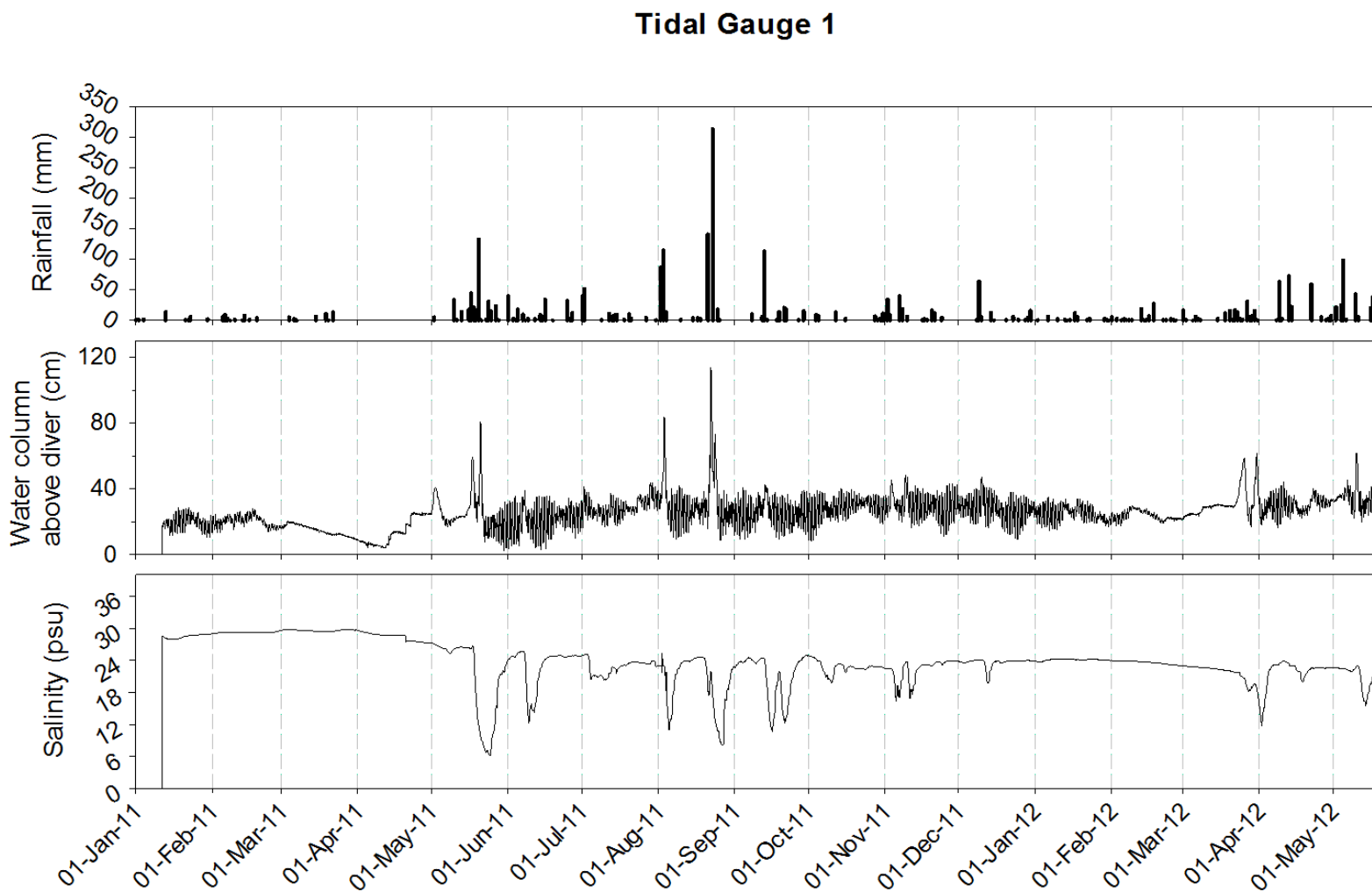
Tidal Gauge 1

Figure 3 - 4. Rainfall, water level and salinity hydrograph for instrument location TG2.

Tidal Gauge 2
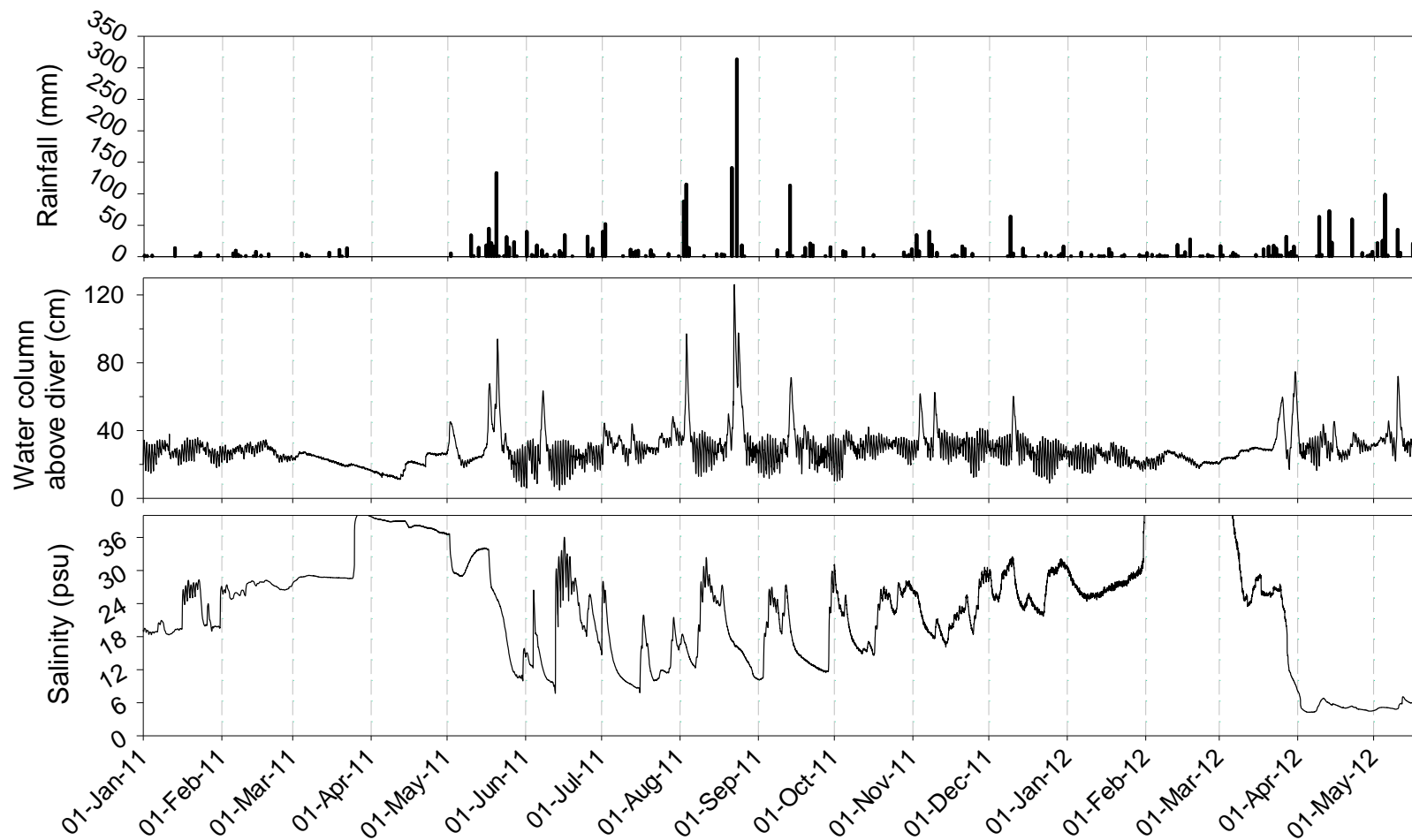
Figure 3 - 5. Rainfall, water level and salinity hydrograph for instrument location TG3.

Tidal Gauge 3
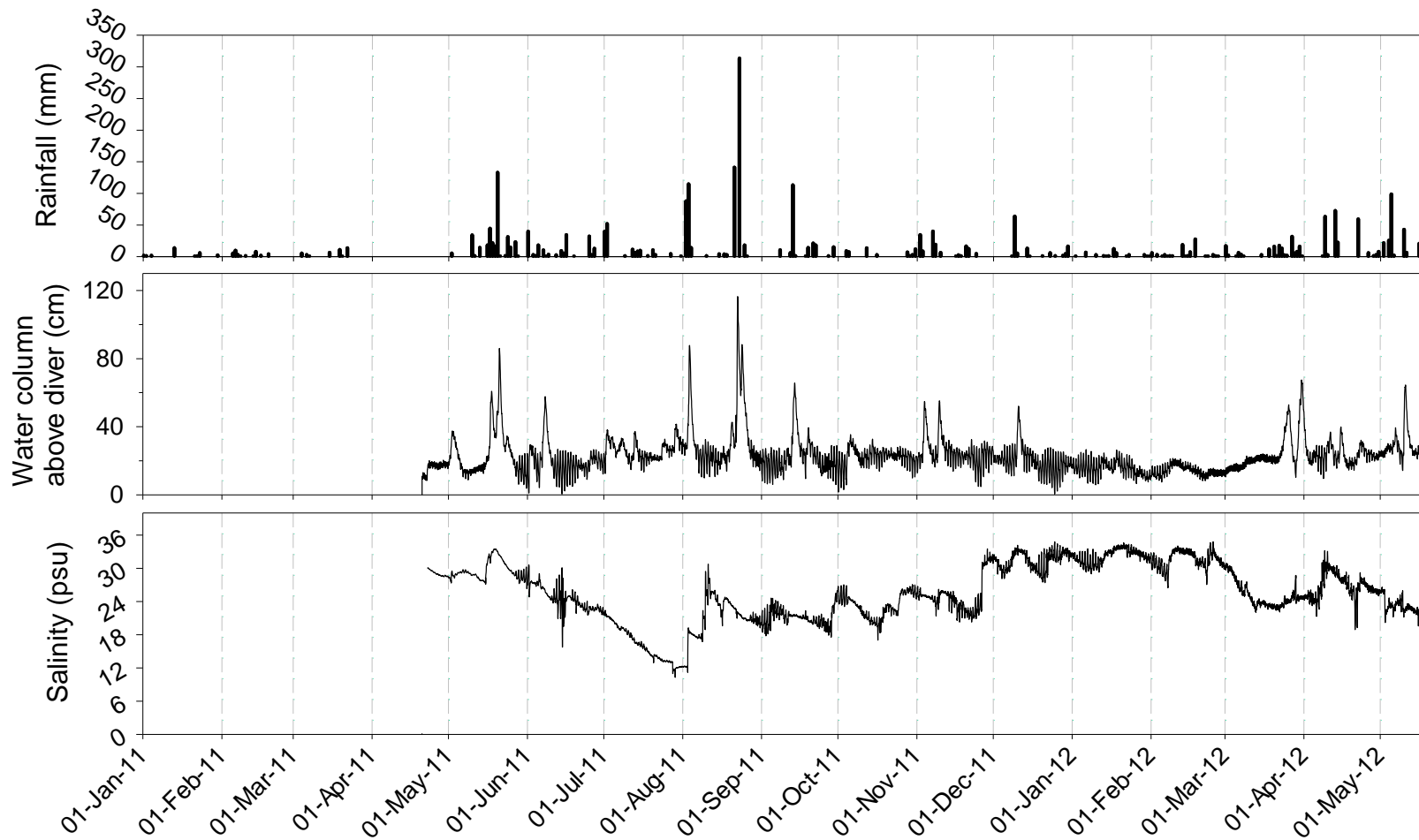
Figure 3 - 6. Rainfall, water level and salinity hydrograph for instrument location TG4

Tidal Gauge 4
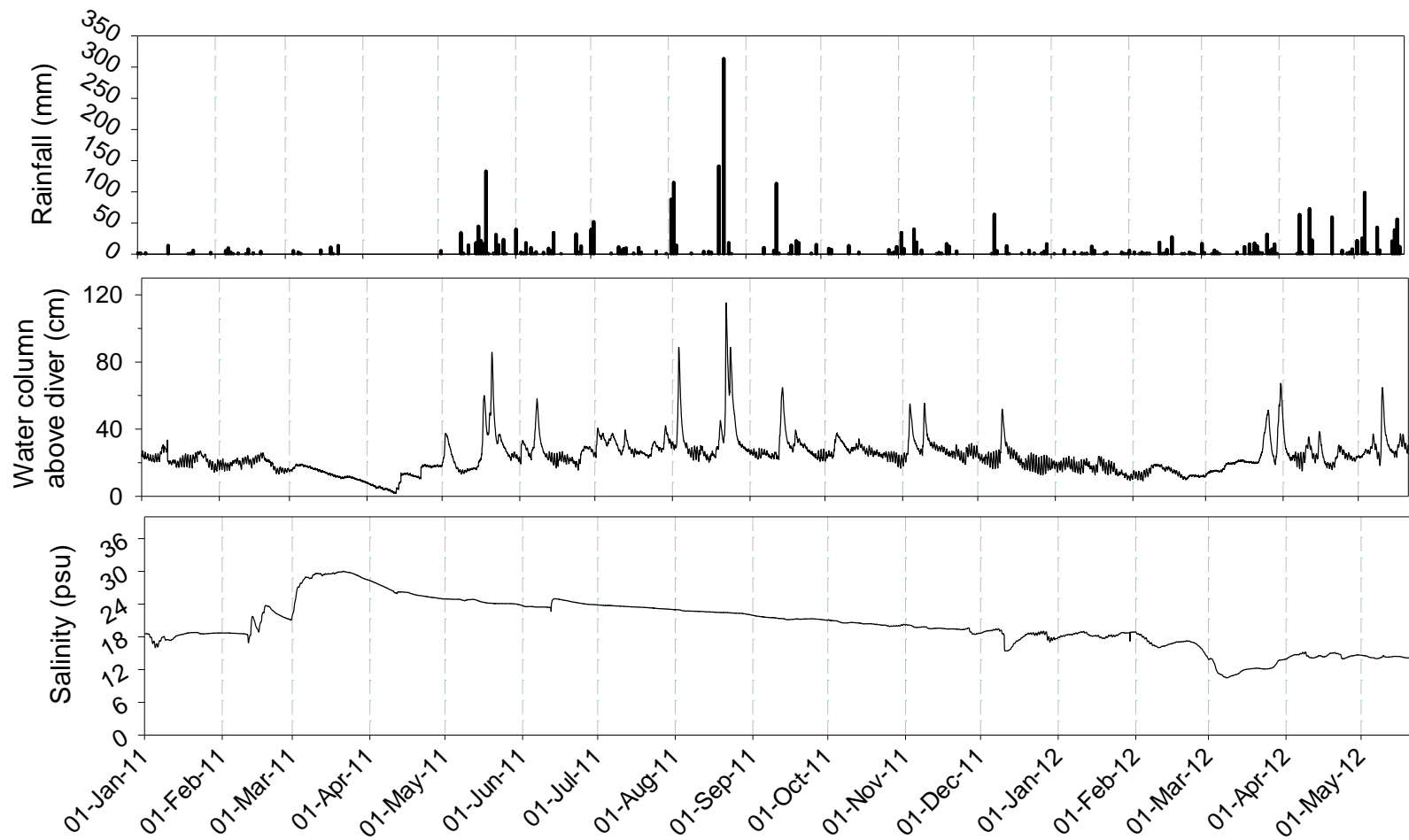
Figure 3 - 7. Subaerial sandbar forming at the mouth of the Antón Ruiz River. Image shows incomplete progress, as the bar begins to form and the connection is relatively shallow. Photo by R.J. Colón-Rivera.

Figure 3 - 8. Mean monthly rainfall for the Juncos NOAA station (1931-2011) and 2011, 2012 monthly averages.

Mean Monthly Rainfall (Juncos Station 1931-2012)
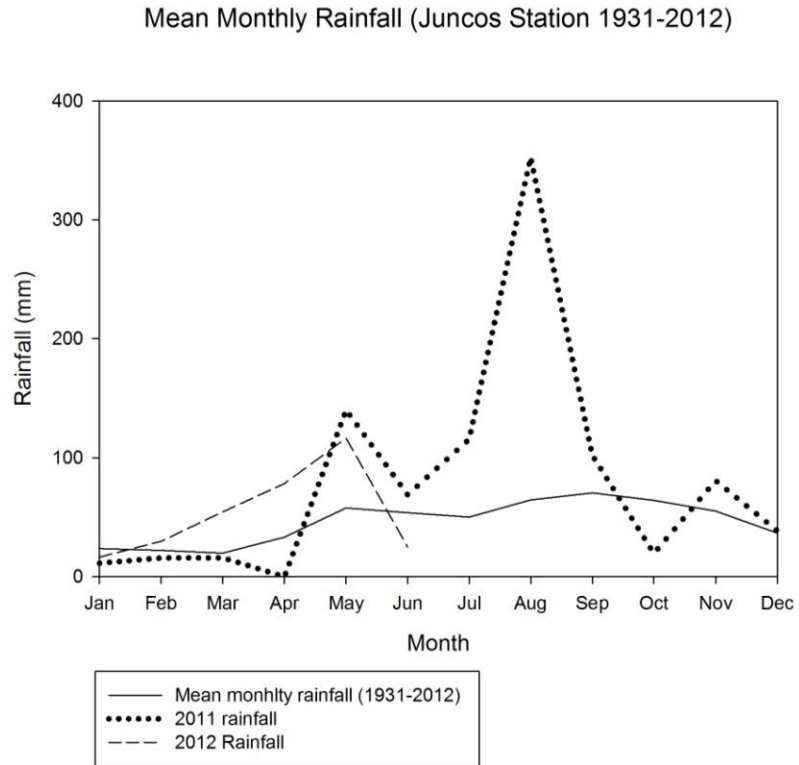
Figure 3 - 9. Monthly variations of Mean Sea Level on the Fajardo NOAA station.

Monthly Mean Sea Level East Coast of Puerto Rico
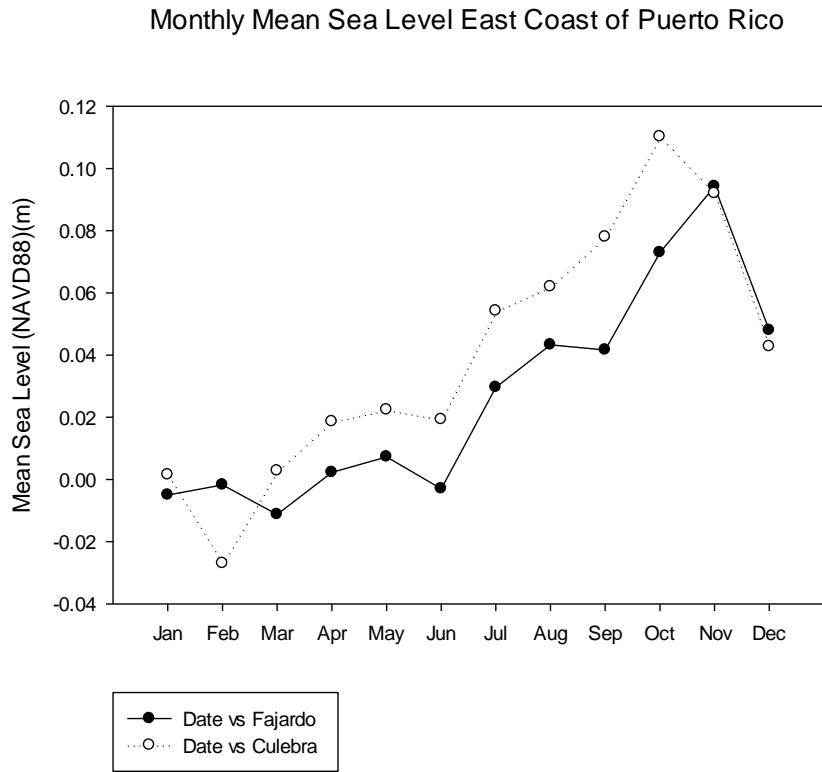
The most conspicuous change in TG3 salinity began after the removal of the sand barrier at the conclusion of the dry season 2011 was followed by a month (July 2011) of reduced tidal amplitude. At this time salinity began a seasonal decrease that lasted until the large rainfall event in August 2011 (Fig. 3-5). Salinity of TG4 showed no relation to rainfall as well (P = 0.921). It did, however exhibit a sudden increase in salinity in the dry season of 2011along with a longer-term trend decrease in salinity over the study period (Fig. 3-6).

3.5   Discussion

Prior to the construction of the USACE channel, the HNR estuarine system only connected to the ocean on occasions of substantial rainfall and the lagoon system was considered oligohaline (psu >5; Schwartz 2004). Our data reveals that the construction of the channel in the HNR not only caused recurrent saltwater intrusion events (Ferrer 2007); it permanently altered the hydrology of the estuary, which now shows the seasonal behavior of a saltwater wedge or "blind" estuary (Savenije 2012). Nonetheless, this type of estuary is commonly found in Puerto Rico and other tropical regions (Ewel 2010) with limited tidal range, particularly on low order stream outlets that drain small watersheds. In fact, because of the estuarine location of many of the remaining *Pterocarpus* stands in Puerto Rico, our study is concerned with the inland propagation of the saltwater wedge on micro-tidal estuaries.

Pulses of freshwater inputs to these estuaries usually promote the vertical stratification of the water column and ease the inland progression of a saltwater wedge (Van der Tuin 1993). Current understanding of this topic states that this progression is further enhanced by the seasonal limitation of freshwater inputs to the estuary (Van der Tuin 1993, Haralambidou et al. 2010). However, the extent of this propagation is determined by the geomorphology of the tidal channel and can vary from site to site.

The inland and seaward movements of the saltwater wedge in the *Pterocarpus* forest can be traced across the tidal creek transect (Fig. 3-10). TG1 shows the constant presence of the wedge with an average salinity near 25 psu. Rain or freshwater inputs, as observed in the water level peaks, force the wedge back towards the ocean and briefly decrease the salinity until it is reestablished around 25 psu. The wedge at TG2 shows a similar pattern except the time to reestablish the average salinity value is of longer duration. Moreover, TG2 also displays a reduction in salinity during the ebb tides, when tidal influence is at its lowest. Abrupt spikes in salinity during the dry season in both 2011 and 2012 indicate that the saltwater wedge moves well inland of this location, as expected. The wedge at TG3 seems to respond to the biweekly patterns of extreme tides, with spikes of increased salinity related to spring tides, where tidal influence is at its highest. Contrary to TG1 and TG2, salinity measurements at TG3 show longer-term variation unrelated to rainfall events. At TG4, the saltwater wedge appears during the 2011 dry season and stays relatively undiluted for the remaining of the study period.

In general, these results indicate that saltwater intrusion along the transect is dependent on the amount of freshwater input to the forest and is enhanced by tidal influence. At the HNR, the dry season is the most important time of the hydrological cycle due to the increased chance of saltwater intrusion into the adjacent subsurface of the *Pterocarpus* forest. Both the lowest average rainfall and lowest average sea levels occur during this time. As a result, the mouth of the estuary closes due to lack of freshwater runoff in combination with low ocean water level, causing the saltwater wedge to become trapped in the tidal creek channel. The higher density of the saltwater allows the wedge to penetrate inland when the pressure from the freshwater lens above the wedge decreases. Consequently, as freshwater inputs decrease in the dry season, the water level in the tidal creek also decreases and the trapped saltwater wedge travels further into the tidal creek channel. This is why the inland movement of the saltwater wedge appears to be at its maximum during the dry season. During the particularly low-rainfall dry season of 2011, the saltwater wedge was able to reach the deepest parts of the forest as observed by the increase in salinity recorded by TG4 during that period. Once it reached this location, it remained there, undiluted by the lack of rainfall or wind-driven turbulence in the tidal creek. This observation suggests that drought-induced saltwater intrusion can be expected in the inner parts of the forest.

Figure 3 - 10. Conceptual representation of saltwater wedge inland movement as measured at different locations of the tidal creek. Presence of the saltwater wedge is constant at instrument location TG1 and diminishes towards TG4. Main drivers of changes in salinity also vary across the transect, with daily rainfall dominating salinity of TG1 and to a lesser extent TG2, which is also influenced by tidal exchange. Salinity changes in TG3 respond to weekly and seasonal patterns of tidal influence while saltwater intrusion to TG4 appears to be drought induced.

Instrument location in tidal creek (not to scale)
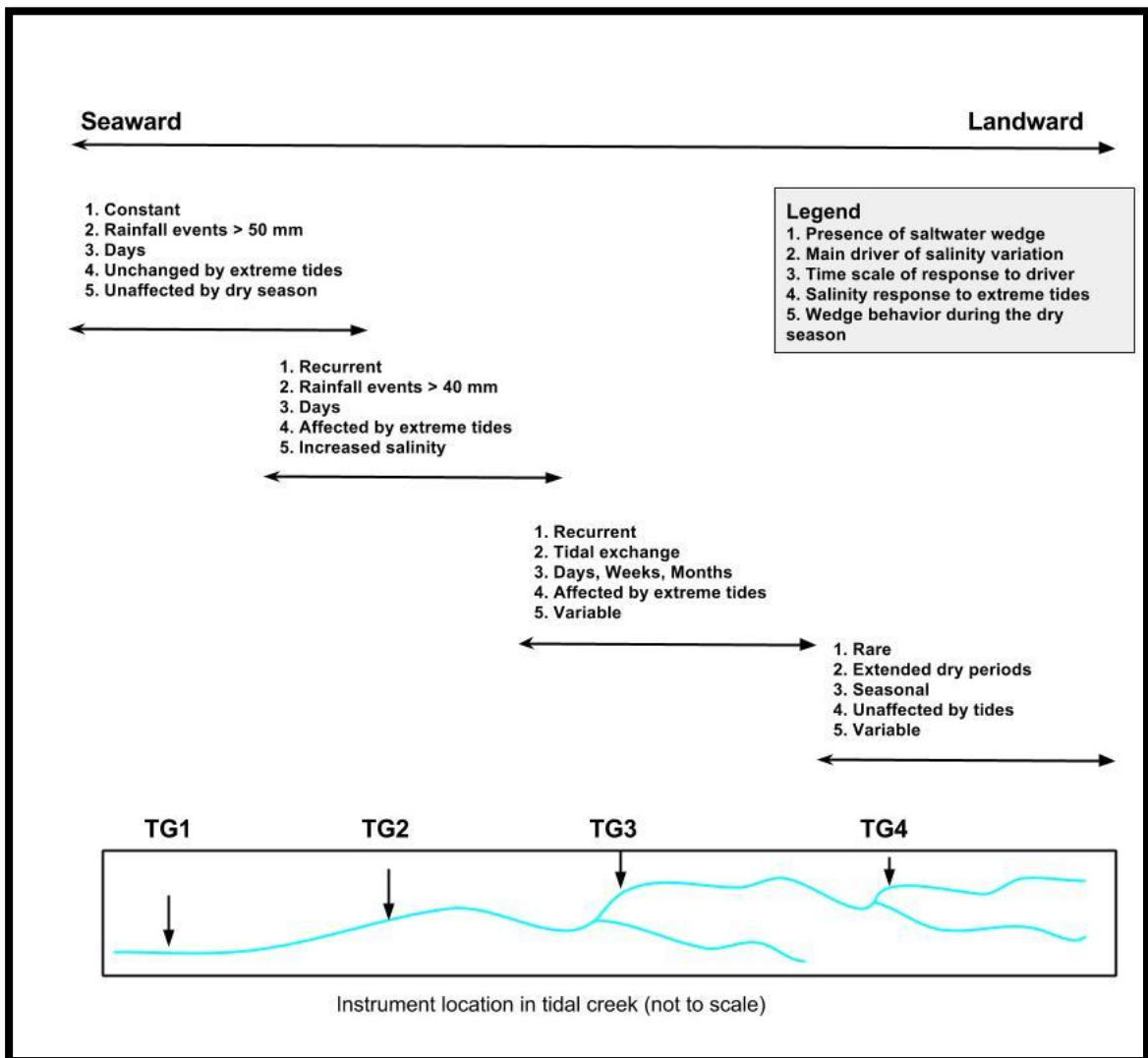
3.5.1   Restoration and hydrological modification in the HNR

In the case of *Pterocarpus* forests of the HNR, the USACE and local managers have tried reducing the inland migration of the salt wedge by emplacing a submerged barrier in the main channel. This barrier is constructed of sand bags, and the idea is to block the denser saltwater from penetrating upstream. However, the initial success of this approach has been limited by the destruction of these barriers by recreational boat transit and the direct disturbance of local anglers. In the case of the disturbance by local anglers it appears that the conversion of the estuary from freshwater to mixed freshwater-saltwater improved the diversity of commercial species like the blue crab (*Callinectes sapidus*) and the common snook (*Centropomus undecimalis*), which in turn increased the fishing pressure at the site.

The best and most ideal solution to prevent loss of more *Pterocarpus* trees would be to restore the historical flow of the Antón Ruiz River to the upper reaches of the forest, thereby increasing the freshwater discharge into the forest. Construction equipment could be used to re-trench the drainage connection from its current terminus over to the forest, a distance of ~1.5 km (Fig. 3-11a). This hydrological modification would be relatively expensive and could alter the private property of adjacent landowners such that it no longer floods. However, this outcome would likely be of interest to the property owners considering that sugar cane is no longer the preferred land use.

A second solution would be to remove the remnant network of small canals that still exists in the freshwater marshes and grazing fields immediately adjacent to the forest (Fig. 3-11b). These canals continue to drain these lands, interrupting the natural sheet flow of water moving towards the upper reaches of the forest. A more limited, yet complementary approach would be to manage upstream inputs of Las Mulas, a small freshwater drainage that currently enters the forest. This drainage runs down from the mountains and through an agricultural farm before arriving at the upper reaches of the forest. As the farm uses much water for the growth of grass and ornamental plants, accommodations could be made for runoff to more directly enter into this drainage.

In Puerto Rico in general, the re-establishment of *Pterocarpus* forests has been observed along limited stretches of riverine habitat that drain into estuaries, following the natural restoration of hydrological patterns after the abandonment of sugar cane plantations on the coastal zone (Lugo 2006). In certain locations, complex watersheds with urban components could also provide the constant flow necessary to avoid saltwater intrusion during extended dry periods (Brantley et al. 2008), although probably at the cost of reduced water quality (Ewel 2010).

3.5.2   Pterocarpus *forests and climate change in the Caribbean*

The main climate change-related stressors for Caribbean island nations will be altered rainfall patterns, increased frequency of storm events and sea level rise (IPCC 2007). These factors control the inputs and outputs of seawater into coastal wetlands. Since *Pterocarpus* forests are primarily established on coastal margins in Puerto Rico, their ability to persist in the future will be tied to the extent that freshwater inputs can counter-balance seawater inputs. The general notion is that as sea level continues to rise, it will bring more saltwater to coastal wetlands, particularly during the dry season. On coastal *Pterocarpus* forests that are non-riverine (e.g. coastal basin forests), sea level rise will first affect the wetland by increasing the water table elevation and the extent of storm surge events.

Figure 3 - 11. Potential hydrological modification and restoration sites.
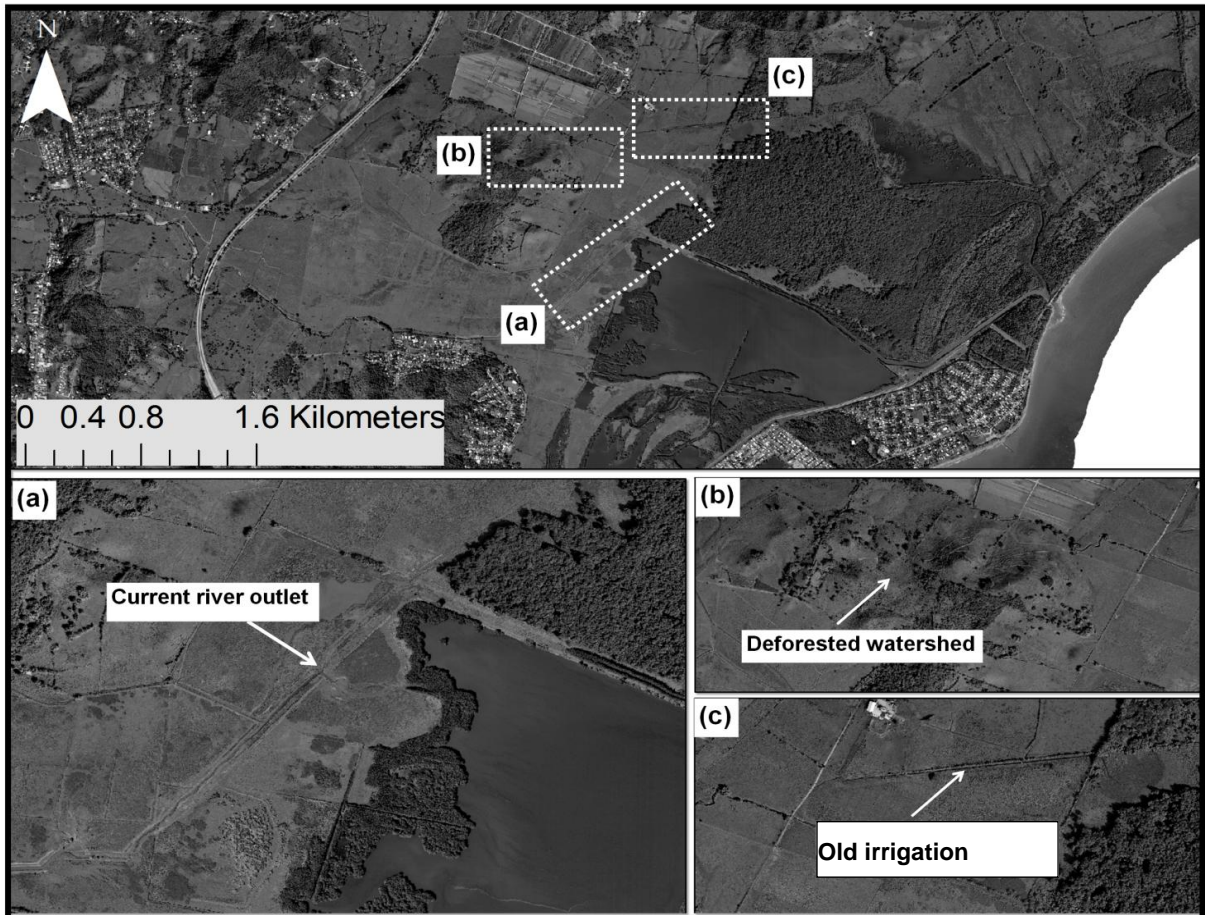
Sea level rise is already accelerating in the Caribbean basin (IPCC 2007), though the increase of sea level will not occur in a uniform manner. Physical factors like ocean density (determined by temperature and salinity), gravitational forces, and geomorphological features of the coast influence the way sea level rise is felt on different locations. Temporal variations reveal there are several annual and semiannual cycles, and spatial patterns show that these cycles vary through the Caribbean basin (Torres and Tsimplis 2012). Similarly, it is not clear exactly how the rainfall patterns will be altered in the Humacao area. Rainfall is locally variable and primarily orographic in nature for this basin (Scholl et al. 2009), but it can be affected by broad scale interactions with continental-scale climate conditions (Hopkinson et al. 2008). Recently, numerical models that study estuarine hydrology have evolved to incorporate freshwater-saltwater interactions (Langevin et al. 2005, Michot et al. 2011). However, direct hydrological observations of the saltwater wedge is a vital complement to the numerical model approach, necessary to account for local variability (Duarte et al. 2003).

The general assumption is that relative sea level rise will gradually increase salinity inputs, particularly in the area where tidal influence ends. Although this place in the estuary (the upper limit of tidal influence) is already dynamic, higher sea levels will mean a higher extent of the highest high tide. The case of the HNR that we present here suggests that the important feature is not whether tide elevation rises and directly flows deeper into the estuary on a particular day, but rather that it is the interaction between

monthly-to-yearly variations in average sea level and rainfall amount/seasonality that will determine the salt wedge migration, in particular for the uppermost limits of tidal influence. Thus, an increasing freshwater input, whether by rainfall or human modification of the hydrological regime, may counter-balance the saltwater intrusion into *Pterocarpus* forests by relative sea level rise.

4. SALTWATER INPUTS AND *PTEROCARPUS OFFICINALIS* WATER USE IN THE HUMACAO NATURAL RESERVE, PUERTO RICO

4.1 Overview

In the Caribbean, freshwater-forested wetlands dominated by *Pterocarpus officinalis* are now mostly restricted to coastal environments. In this setting, the seasonal interactions of saltwater and freshwater inputs create less than ideal conditions for these forests to survive. Hence, it is crucial to have a better understanding of the hydrologic context of these wetlands. We used water stable isotopes (δ^2H, δ^{18}O) as natural tracers to determine the most important freshwater sources, and the extent of saltwater presence in the *Pterocarpus officinalis* forest of the Humacao Natural Reserve in Puerto Rico, the largest and best preserved of these forests in the US. We sampled freshwater sources (rainfall, runoff) into the forest during a 15-month period, as well as surface water and woody stems from trees adjacent to the surface sample location to determine the source of water uptake by the trees. In addition, we collected groundwater samples in a transect perpendicular to the primary tidal creek that provides a pathway for saltwater into the forest. Isotopic content (δ^2H, δ^{18}O) of the surface water samples revealed that most of the water in the forest was derived from freshwater runoff from the Las Mulas Creek, but there was a seasonal change in its relative contribution to the forest hydrology. During the dry season, high δ values suggested the presence of runoff-derived water that had evaporated, as well as brackish water influences that were found east of the tidal creek where past deforestation created preferential pathways for this brackish water to

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flow. During the wet season, low δ values revealed the presence of water derived from recent rainfall events, although $\delta^{18}O$ values lower that -2.0 ‰ in areas close to the tidal creek suggested that tidal influence was still noticeable in the area. Groundwater data revealed the presence of a saltwater wedge expanding into the forest from the tidal creek. Nonetheless, ^{18}O-enriched stem water suggested that *Pterocarpus officinalis* relies primarily on water accessed from unsaturated soil mounds that surround the tree and are subject to evaporation, as opposed to groundwater at depth. This observation provides further insight to the adaptation mechanisms of *Pterocarpus officinalis* to survive in flooded ecosystems.

4.2   Introduction

Global climate change threatens the survival of coastal freshwater-forested wetlands by altering precipitation patterns and accelerating sea level rise (Day et al. 2008). The combination of these stressors results in increased flooding period and soil salinity in coastal wetlands (Snedaker 1995). In the Caribbean, sea level rise likely will combine with longer dry periods to increase the overall salinity of estuarine wetlands (Larsen 2000, IPCC 2007). In this region, *Pterocarpus officinalis* (hereby *Pterocarpus*)— the dominant species of freshwater forested wetlands (Alvarez-López 1990, Imbert et al. 2013)— is particularly vulnerable to the long-term effects of global climate change (Rivera-Ocasio et al. 2007). Physiological adaptations like a very shallow root system that spreads laterally (Cintrón 1983), the presence of hypertrophic lenticels and aerenchyma tissue (Saur et al. 1998b, Fougnies et al. 2007) and the

concentration of Na away from photosynthetic leaf tissue (Medina et al. 2007) allow

Pterocarpus to withstand prolonged flooding and moderate increases in soil salinity. In

addition, tree buttresses accumulate organic matter that create elevated microsites that

support seedling establishment and soil aeration (Lopez and Kursar 2003). However,

increased soil salinity increases adult mortality rates while reducing growth, recruitment,

and root nodulation (Rivera-ocasio et al. 2007). Salinity also affects flower and litter

production (Eusse and Aide 1999) and reduces colonization of beneficial arbuscular

mycorrhizal fungi (Saint-Etienne et al. 2006, Fougnies et al. 2007).

For hundreds of years many of the coastal valleys in Puerto Rico were cleared

and constantly drained to allow the cultivation of sugarcane crops. This land use

pressure reduced the cover of the once abundant *Pterocarpus* forests to small patches

restricted to the coast (Martinuzzi et al. 2007), near the limits of their physiological

tolerance of salinity (Rivera-Ocasio et al. 2007). As sugar cane agriculture was

abandoned during the industrialization of Puerto Rico in the 1950's, there was no further

need to maintain the pumps and channels that drained the coastal valleys. Over time, the

natural reestablishment of hydrologic conditions allowed for new *Pterocarpus* stands to

colonize the freshwater zones of river estuaries in the island (Lugo 2006, Martínez and

Lugo 2008). Nonetheless, this recovery is constrained by the vulnerability of

Pterocarpus forests to saltwater intrusion in their environment. Furthermore,

deforestation can influence future recovery by altering the geomorphology of the

ecosystem. When mangrove trees are removed from an area— either by landscape-scale

or localized disturbances— there can be a reduction of soil elevation by the combination of increased peat decomposition and lack of organic matter input. The reduction of soil elevation increases flooding period in the disturbed area, which subsequently reduces recruitment success by mangrove seedlings and the area converts from forest to open water (Snedaker 1995). In undisturbed *Pterocarpus* forests, variations in the elevation of the soil and the thickness of the peat layer also influence flooding regimes and plant community structure (Migeot and Imbert 2011).

The vulnerability of coastal *Pterocarpus* forests is predicated on the balance between freshwater and saltwater inputs to the wetland. However, few studies have focused on the hydrology of saltwater-freshwater interactions in these ecosystems. Furthermore, there is no available data on *Pterocarpus* use of different water sources, although this question has been addressed for other types of vegetation from several coastal environments (Greaver and Sternberg 2006, Saha et al. 2010, Wei et al. 2012). As in these studies, we used water stable isotopes ratios (^{2}H/^{1}H, ^{18}O/^{16}O) as natural tracers to study the hydrologic setting of the largest *Pterocarpus* forest in Puerto Rico. In this study we aimed to answer the following questions: 1) What is the temporal variation and spatial extent of saltwater inputs to coastal *Pterocarpus* forests? 2) Is the seasonal influence of freshwater inputs (rainfall, runoff) able to ameliorate the presence of saltwater in the system? 3) What is the primary source of water used by *Pterocarpus* for transpiration, and is there any temporal or spatial variation in the use of different water sources?

4.3 Methods

4.3.1 Study site

Our study site is located in the *Pterocarpus* forest of the Humacao Natural Reserve (HNR), Puerto Rico. This *Pterocarpus* forest is the largest in Puerto Rico (150 ha) and its estuarine location is representative of the hydrologic conditions of the riverine stands on the island (Section 3). The HNR is located in a coastal plain estuary formed by three interconnected valleys and drainages (Ferrer Montaño et al. 2005). One of these three drainages— the Antón Ruiz River— meandered through the *Pterocarpus* forest on its way to the ocean but hydrological modifications have altered its course such that it now leads directly into nearby coastal lagoons (Fig. 4-1). The lower portions of the river that had meandered through the forest are now composed of a primary tidal creek and its distributaries. This tidal creek drains the freshwater outflow of the forest to the mouth of the estuary, saltwater penetrates inland via this tidal creek on periods of low rainfall or high tidal influence (Section 3).

For over a century, the forest cover of the HNR valley was subsequently cleared for several purposes including, sugarcane cultivation, charcoal production, and pasturage (Schwartz 2004). The existing *Pterocarpus* forest is a stand that was left untouched during this

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Figure 4-1. Study area. (A) Location of the Humacao Natural Reserve, east coast of Puerto Rico. (B) Tidal creek, Antón Ruiz River and Las Mulas Creek are highlighted by a blue line. The Antón Ruiz River drains directly into the coastal lagoons. Las Mulas Creek drains directly into the *Pterocarpus offinicalis* forest that is located northeast of the coastal lagoons. Freshwater outflow is directed to the ocean through a tidal creek that also allows seasonal incursion of saltwater into the *Pterocarpus* forest. Groundwater transect area is enclosed in white rectangle. (C) Close-up of groundwater transect perpendicular to the tidal creek.
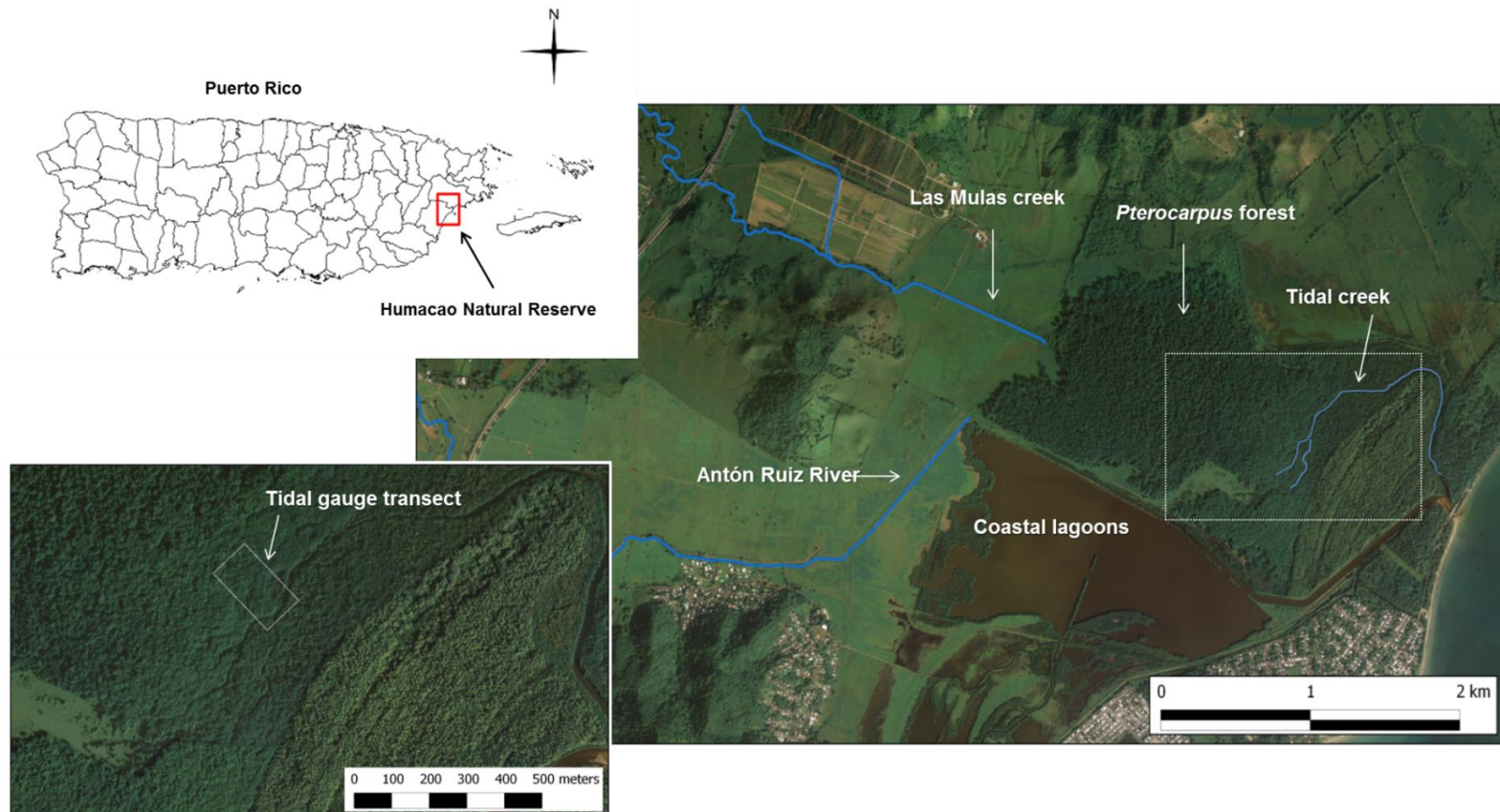
development except for the area surrounding the tidal creek, which remained deforested until the 1950's (Fig. 4-2). This area has recovered since, forming a secondary forest where *Pterocarpus* remains the most abundant species but unlike the old-growth portions of the stand, understory vegetation includes salt-tolerant species like *Acrostichum aureum* and *Rhizophora mangle* (Alvarez-López 1990, Toledo-Rodríguez 2013).

Vegetation in the portions of the forest further downstream is dominated by *Rhizophora mangle* and *Laguncularia racemosa* on the northeast across the tidal creek, the palm tree *Cocos nucifera* (grown commercially in the past) directly to the east towards the coast, and herbaceous marsh to the north. Runoff to the forest is provided by sheet flow from the marsh and by a perennial stream (Las Mulas Creek) that drains directly to the northern part of the forest (Fig. 4-1).

As in much of the Caribbean, contrasting rainfall amounts during the dry (DS) and wet season (WS) mediate this interaction between upstream freshwater sources and the ocean. Recorded rainfalls during the study period (National Weather Service, station JNCP4) show the DS occurring between January and April, and the WS from May to December (Fig. 4-3).

The HNR sits in an alluvial valley with unconfined aquifer conditions (Graves 1989). Deep groundwater is recharged locally by rainfall at different locations in the valley and travels east towards the coast and is ultimately discharged into the ocean as subsurface flow (Veve and Taggar 1996). At the location of the forest, the freshwater-

saltwater interface in deep groundwater is estimated to be 120 feet below the soil.

During an extremely dry period (March 1984), groundwater salinity at deep wells near the coast was measured at 1-1.5 ppt (Graves 1989). Although this is considered saltwater intrusion into the freshwater supply when accessing this supply through wells, its contribution to the shallow groundwater considered in this study (defined as less than one meter of depth) is minimal. In addition, the Ghyben-Herzberg principle— which states that for every foot the water table is located above sea level there are forty feet of fresh groundwater — is often used to estimate the location of the freshwater-saltwater interface in alluvial estuaries. Following this principle we do not expect a measurable contribution of the freshwater-saltwater interface to influence the salinity of the forest (Savenije 2012).

In the specific locality of the *Pterocarpus* forest, the soils are characterized as swamp-marsh deposits with a shallow water table (1-2 feet). At the scale of our study forest (< 2 km^2) and at shallow groundwater depths (< 1 m of depth), salinity in the *Pterocarpus* forest soil is ultimately supplied by the nearby tidal creek, as this water penetrates horizontally into the soil or overland through flooding of the tidal creek.

4.3.2 Sampling protocol

Possible sources of water (end-members) were collected monthly during a 15-month period to develop a local database of isotopic data for comparison purposes. End-members sampled included rainfall, runoff (Las Mulas Creek), and ocean. All water samples were collected in 250 ml Nalgene laboratory plastic bottles, capped tightly, wrapped with parafilm to prevent cap loosening, and refrigerated until analysis.

Groundwater samples were collected along three locations on a 100 m transect perpendicular to the tidal creek at 10m, 30m and 100m distance from the creek (Figure 4-1, tidal gauge transect). At each location three groundwater sampling wells were established at three different depths: 30cm, 60cm and 90cm. Sampling wells consisted of a 2" diameter PVC pipe with horizontal slits to allow adequate flow of water through the pipe at the respective sample depth. During sampling, the wells were drained two times and water was allowed to flow back in. The sample was taken on the third time and salinity was measured in the field using a handheld refractometer (LW Scientific, Inc.). Groundwater samples were taken on April 20, June 16 and June 20 of 2011, and February 18 and May 21 of 2012.

Figure 4-2. **(A) Historical image showing deforested area in the HNR** *Pterocarpus* **forest in 1951. (B) Close-up of deforested area around the tidal creek. (C) Small channels where the tidal creek drains east (black arrows), not visible in recent imagery due to increased canopy cover.**
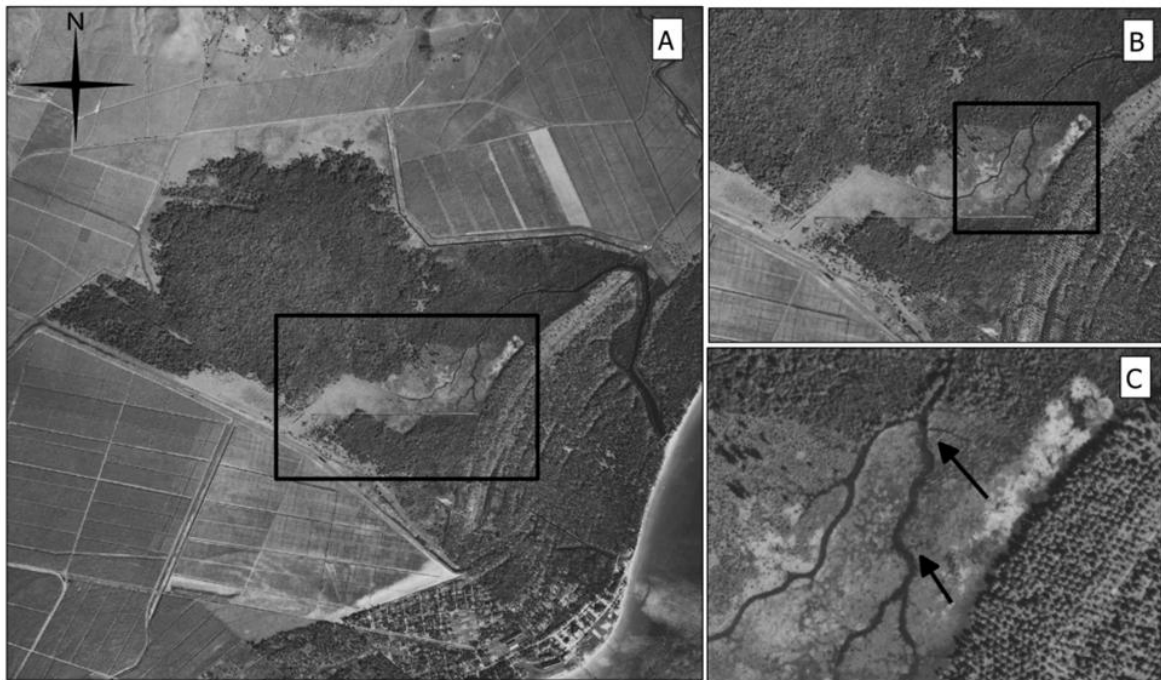
Figure 4-3. 2011-12 Rainfall (National Weather Service, station JNCP4). Surface and tissue water sampling dates are highlighted by red arrows.

Rainfall 2011-2012
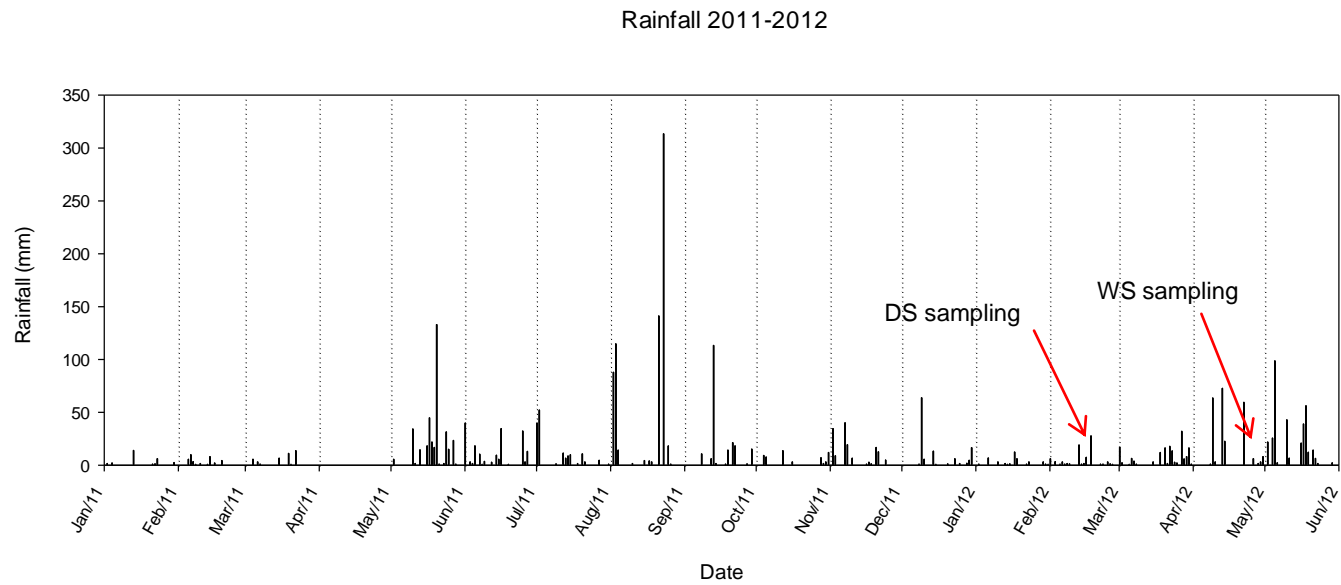
To determine if there was seasonal variation in water utilization by *Pterocarpus* trees, two sampling trips were conducted during February (DS) and May (WS) of 2012. During these sampling trips, surface water and tree stem samples were collected at random locations along the *Pterocarpus* forest, in the general vicinity of the groundwater transect (Figure 4-1) and in the previously deforested portions of the forest (Figure 4- 2) and is subject to both saline to freshwater conditions. Caution was taken to sample tree stems from suberized, non-green tissue far from the compound leaf tip to avoid isotopic enrichment due to transpiration (Sternberg and Swart 1987). Stem samples were collected in airtight Pyrex tubes and frozen until time of analysis. Subsequently, water was extracted from the stems using cryogenic vacuum distillation (Sternberg et al. 1991). All water samples were analyzed using a continuous-flow pyrolysis isotope ratio mass spectrometry technique (TC/EA-IRMS; Delta V Isotope Ratio Mass Spectrometer; Thermo Scientific, Waltham, MA). Each sample was analyzed for both hydrogen and oxygen isotope ratios. Stable isotope ratios are expressed as follows:

$$\delta(\%_{00}) = \left[\frac{R_{sample}}{R_{standard}} - 1\right] \times 1000$$

where R_{sample} represents the ratio of heavy to light isotope (here either $^2H/^1H$ or $^{18}O/^{16}O$) and $R_{standard}$ is the isotope ratio of (in this case) Standard Mean Ocean Water (SMOW). Laboratory reference waters that have been calibrated to SMOW are used in each run for normalization and quality control.

The ratio of the heavy (less naturally abundant) to light (abundant) stable isotopes in the molecules of water is affected by the different phase changes of the hydrologic cycle (Gat 2010). Water that undergoes evaporation becomes disproportionally enriched in the heavy isotopes ^{18}O and ^{2}H (hereby deuterium, D). This process— called *fractionation*— creates isotopic "signatures" that can be used to trace the source of water samples and detect processes like evaporation by observing the deviation of the samples from the linear relation between δD and $δ^{18}O$ of meteoric waters described by the Global Meteoric Water Line (GMWL, Craig 1961). Water uptake by the tree roots does not fractionate water, thus there is a strong correlation between plant stem water and water sources, creating a useful method to study changes in plant water use (Saha et al. 2010).

4.4 Results

Isotopic content of end-members resembled δD and $δ^{18}O$ values reported for the east coast of Puerto Rico (Scholl et al. 2009). Rainfall samples varied from -6.4 to +0.6 ‰ for $δ^{18}O$, and -41.9 to +13.3 ‰ for δD. Meanwhile, runoff water collected from Las Mulas creek remained stable, varying from -1.9 ‰ to -2.9 ‰ for $δ^{18}O$, and -15.5 to -5.1 ‰ for δD. Average δD and $δ^{18}O$ ocean water was +4.79 and +0.84 ‰ respectively (Fig. 4-4). Overall, the average isotopic content of the end-members in the *Pterocarpus* forest hydrologic cycle are presented in Fig. 4-5.

$δ^{18}O$ values of groundwater samples varied with distance from the tidal creek and sampling depth, although all samples remained between -2.0 and -0.8 ‰ during both the

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DS and the WS at all sampled depths (Figs. 4-6 and 4-7). At shallow depths, the influence of brackish waters on the isotopic composition of the samples decreased with distance from the tidal creek. For example, at 30 cm depth the dry season samples show a trend towards more depleted sources of water as distance to the tidal creek increases (Figure 4-6). Increased rainfall during the 2011 wet season (Fig. 4-3) made this trend more noticeable at depths of 30 and 60 cm (Fig. 4-7). At 90 cm depth, δ^{18}O values showed a different variation with distance, with slightly more enriched values at TG7 than at TG8, as expected.

Standing surface waters during the DS ranged between -2.02 and +0.42 ‰ for δ^{18}O, and -7.75 to +10.25 for δD; during the WS the surface water ranged from -2.87 to -1.52 ‰ for δ^{18}O, and -15.36 to -3.38 ‰ for δD (Fig. 4-8). The seasonal difference in these signatures corresponds to the amount of rainfall and its control on saltwater influence in the forest. During both seasons, the relatively less negative δ values of surface water were spatially constrained to the east of the tidal creek (Fig. 4-8), an area that receives most of the overflow of the tidal creek due to water pathways created by the deforestation of the area prior to the 1950's (Fig. 4-2). Likewise, surface samples with relatively depleted δ values are usually found outside of this area.

Figure 4-4. δD vs. δ¹⁸O scatter plot of meteoric (rainfall) and runoff (Las Mulas Creek). Local Meteoric Water Line (LMWL, dashed line) is calculated from rainfall samples. Global Meteoric Water Line (GMWL, solid line) is shown for reference (Craig 1961).
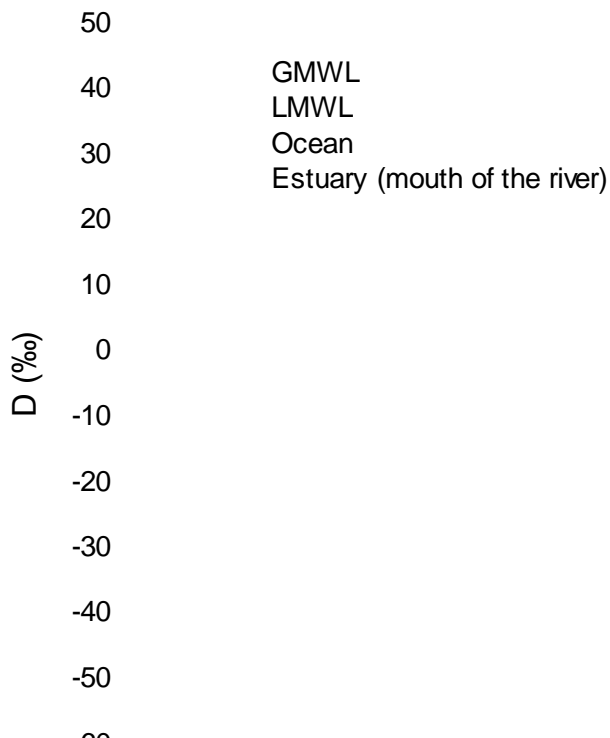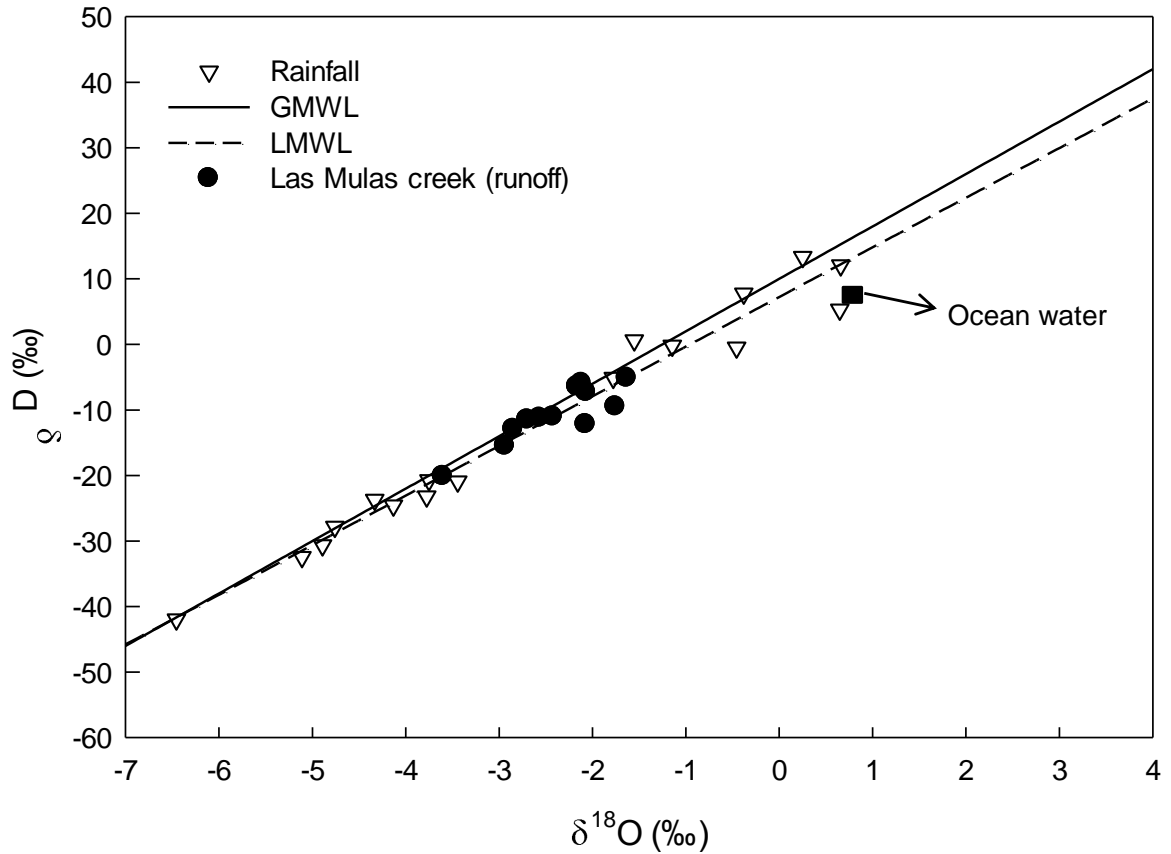
Figure 4-5. Hydrologic cycle of the *Pterocarpus* forest with isotopic content of different phases. δD and δ^{18}O values (‰) taken from Scholl et al. 2009 (vapor) and data from this study (rainfall and runoff).

Seaward ←———————————————————————————→ Landward

Rainfall average
δ^{18}O = -3.0‰
δD = -14.0‰

Vapor
δ^{18}O = -11.0‰
δD = -70.0‰

Evaporation from the ocean

Ocean enters the system via the tidal creek

Runoff average
δ^{18}O = -2.3‰
δD = -10.0‰

Ocean
δ^{18}O = 0.8 ‰
δD = 4.7 ‰

R. mangle

Recovered forest

Old growth forest

Freshwater marsh

Upland watershed

Pterocarpus forest
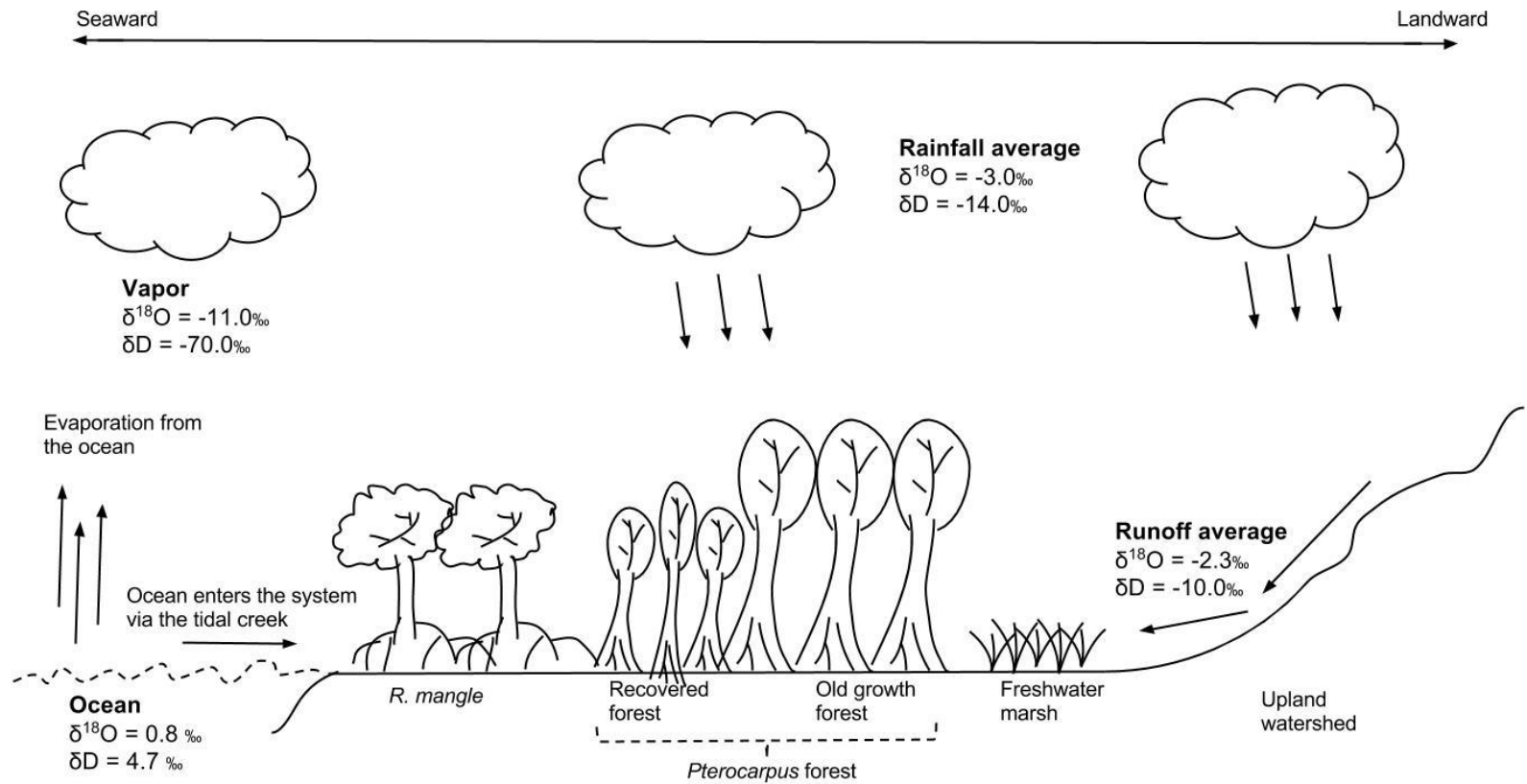
Figure 4-6. $\delta^{18}O$ (‰) of dry season groundwater sampled on April 20, 2011 (a) and February 18, 2012 (b).
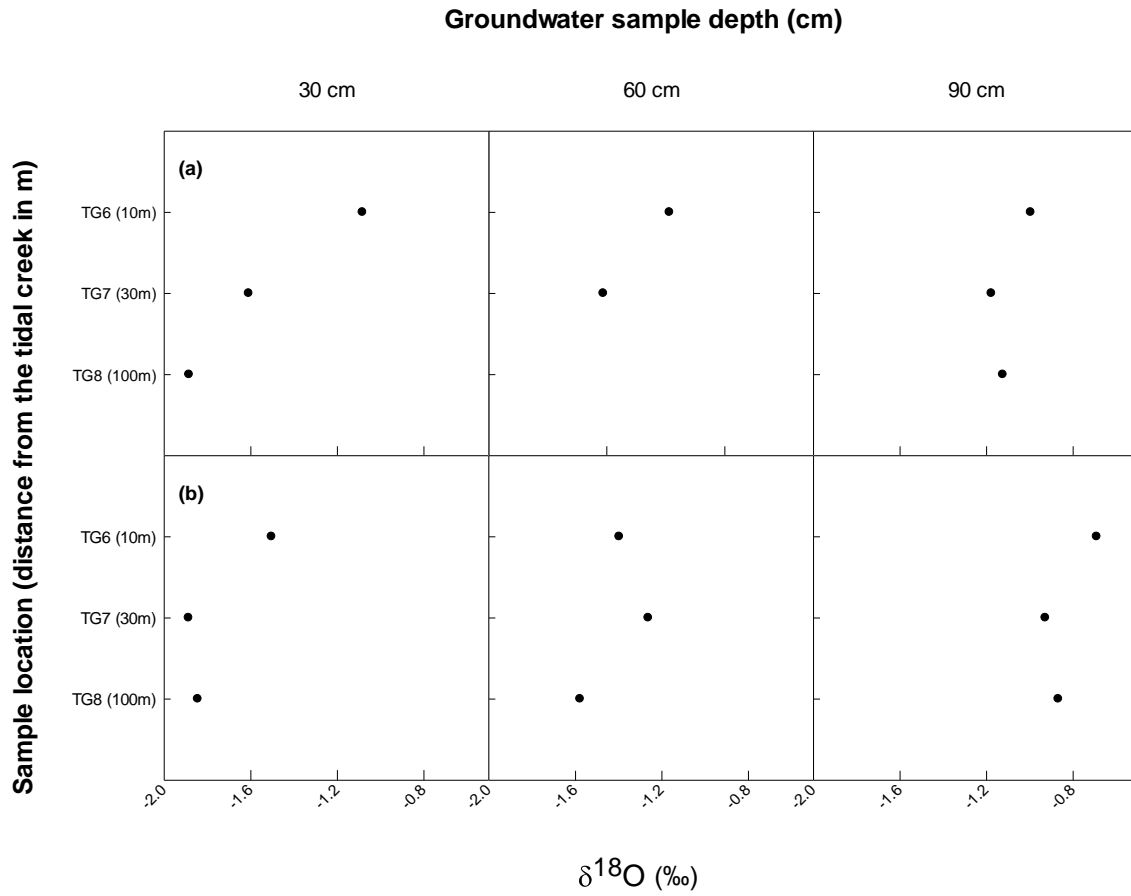
Figure 4-7. δ^{18}O (‰) of wet season groundwater sampled on June 16 (circles) & June 20, 2011(squares) (a) and May 21, 2012 (b).
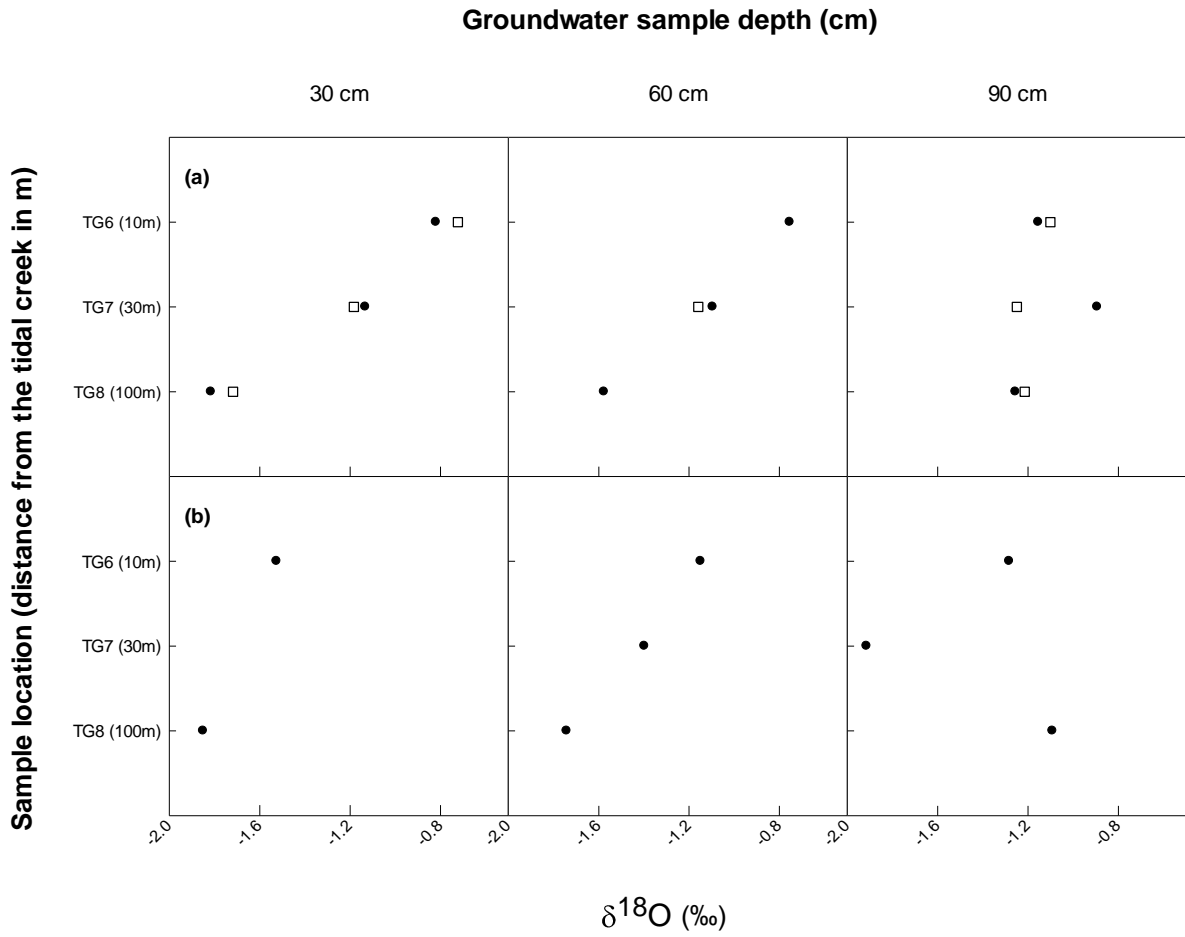
Isotopic content of stem water showed seasonal variation similar to the surface water. During the dry season, stem water ranged from -1.3 to +0.6 ‰ for δ^{18}O, and -6.9 to +2.5 ‰ for δD; during the wet season, stem water ranged from -3.1 to -0.8 ‰ for δ^{18}O, and -29.7 to -6.9 ‰ for δD (Fig. 4-9). These ranges of values suggest that surface water (derived from rainfall and runoff) is an important source in tree water uptake. In addition, stem water δ values point to a water source that has undergone evaporation, as evidenced by the slope of the δD vs. δ^{18}O scatterplots (Fig. 4-9).

4.5   Discussion

4.5.1   *Saltwater inputs in the Humacao Natural Reserve, Puerto Rico*

During our field sampling efforts there was no indication of dead or dying trees in the areas adjacent to the tidal creek near our sample locations, except for downed trees after strong rainfall or wind events. Adult trees can grow and even reproduce on areas of high salinity— 15 to 20 ppt— (Eusse and Aide 1999). However, trees in these areas are unlikely to be replaced by their offspring because salinity reduces recruitment success (Rivera-Ocasio et al. 2007). As the sea level rises, the saltwater wedge will migrate further inland in the channel and sideways through the forest soil, increasing the spatial extent of salinity-related stress (Section 3). In addition to increased salinity, sea level rise can alter the flooding regime of coastal wetlands, creating an additional stress on estuarine vegetation. An increase in flooding periods reduced root depth and stomatal conductance, and increased the root:shoot ratio on *Pterocarpus* seedlings in a greenhouse experiment (Lopez and Kursar 2003). These physiological responses are

common in plants that are not tolerant to flooding; an indication that prolonged flooding can also stress trees that are in areas of low salinities.

4.5.2   **Pterocarpus officinalis** *water use*

As an estuarine wetland, salinity in the HNR *Pterocarpus* forest varies by season and is mediated by freshwater inputs. The important contribution of rainfall for *Pterocarpus* water use resembles ecohydrological processes in other coastal ecosystems. In Australia, riparian species occurring along an estuarine ecotone were shown to rely mostly on shallow water derived from rainfall, even during the dry season when plant water use was expected to shift to groundwater (Wei et al. 2012). The contribution of rainwater for coastal vegetation has also been reported for

Figure 4-8. Map of δ¹⁸O (‰) of surface waters sampled in the dry season (left panel) and wet season (right panel).
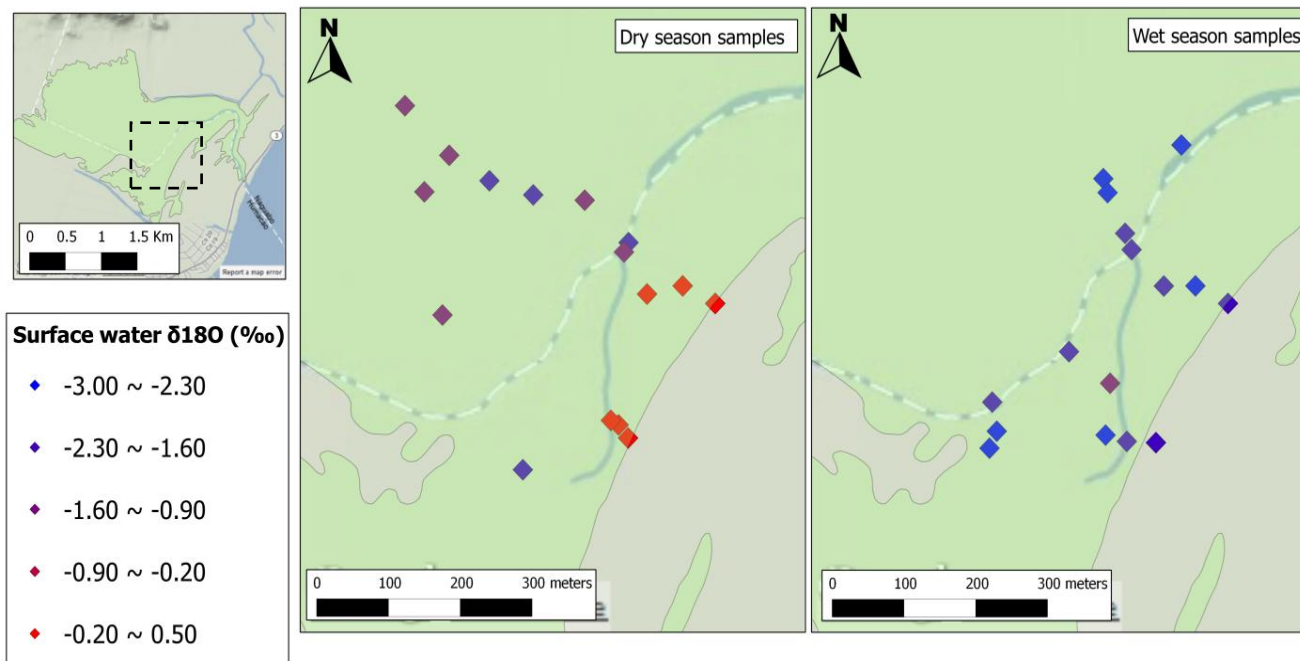
Figure 4-9. δD vs. δ¹⁸O scatter plots of pool surface and tissue water for February 2012 (dry season) and May 2012 (wet season).
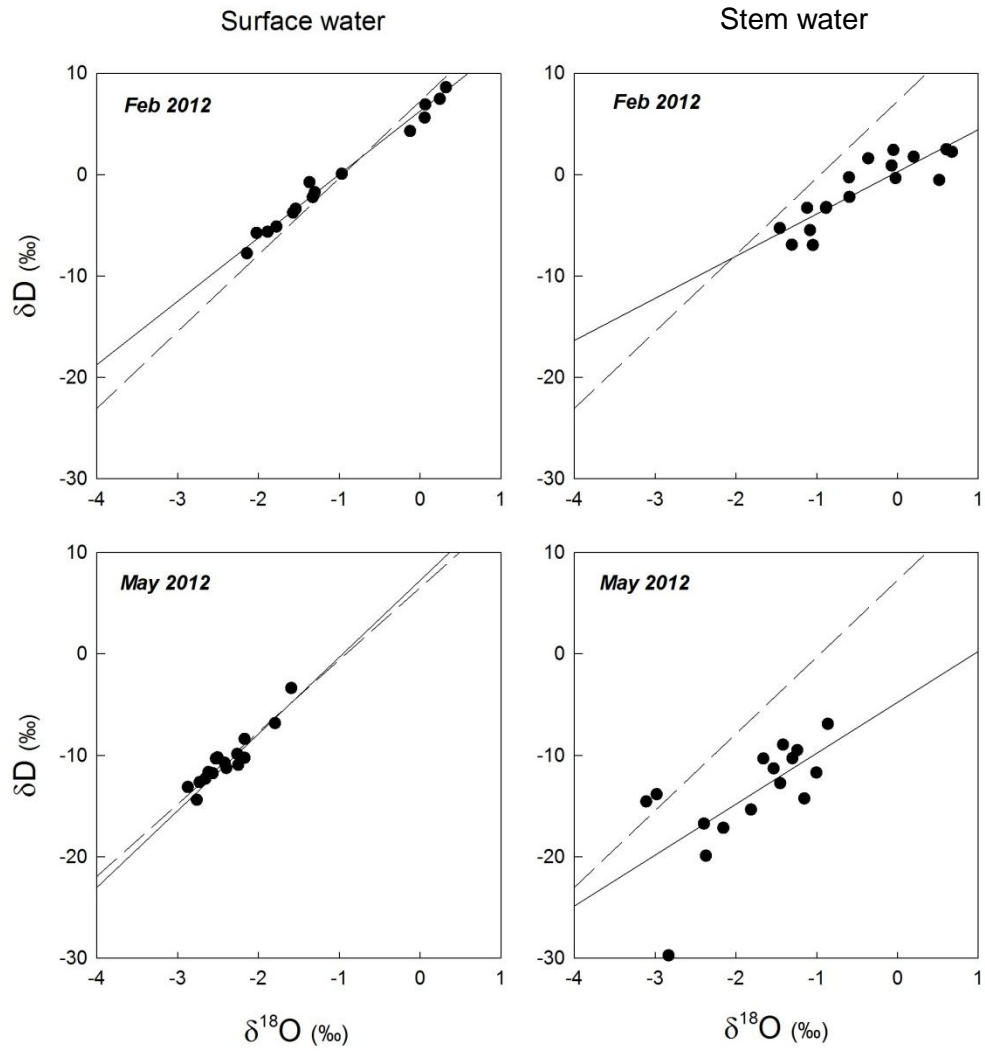
coastal dune plants and sea-side mangroves (Sternberg and Swart 1987, Greaver and Sternberg 2006). Moreover, patterns of coastal plant water use can vary seasonally, as observed in the Everglades (Ewe et al. 2007). Our results provide evidence that the high dependence on rainwater observed in estuarine vegetation (Wei et al. 2012) also extends to non-halophyte species in the Caribbean such as *Pterocarpus*.

Despite evident stress related to increase flooding and salinity, *Pterocarpus* remains the most successful colonizer of flooded landscapes in the Caribbean (López and Kursar 2007, Migeot and Imbert 2011). Analysis of stem water δ values can provide additional insights to this success. The shallow root system of *Pterocarpus* trees concentrates most of its root biomass within 1 m of soil (Alvarez-López 1990). Interestingly, the range of stem δD and $δ^{18}$O values is different from the range of groundwater measured at this depth (Fig. 4-6 and 4-7). Instead, it appears that surface soil moisture is the most important water source for *Pterocarpus* trees, as observed by the similar ranges of stem and surface standing water δ values during both seasons (Figure 4-8). The stem water scatter plots reveal a surface water source that is evaporatively enriched in the heavy isotopes, as evidenced by the regression slopes (Figure 4-8). Evaporative enrichment of soil water is more likely to occur in the vadose (unsaturated) zone of the soil. In the case of *Pterocarpus* forests with a long hydroperiod, these unsaturated zones are restricted to elevated microsites created by hummocks of organic matter with improved aeration (Lugo 1997, Koponen et al. 2004). Toledo-Rodríguez et al. (2013) found that *Pterocarpus* trees exposed to short periods of

flooding grew long roots that reached far from the tree. Likewise, trees in the HNR

Pterocarpus forest had shorter roots (Toledo-Rodríguez et al. 2013). Downed trees in

the HNR forest showed a high concentration of fine roots near the buttress roots (Pers.

Obs.). Additionally, in elevated peats soils where waterlogging is prolonged, the finer

roots occupy the uppermost centimeters of the soil (Migeot and Imbert 2011).

Therefore, we interpret these results as additional evidence that the shallow root system

and the formation of mounds is an adaptation to flooded environments, and furthermore,

that water uptake by *Pterocarpus* trees occurs at depths even shallower than expected

(Fig. 4-10). Furthermore, Eusse and Aide (1999) found *Pterocarpus* with high litter

production on areas with salinity between 9 and 24 ppt, although they point out that

salinity was measured at 60 cm. In this case, the hypothesis that *Pterocarpus* water

uptake is concentrated to very shallow depths or unsaturated portions of soil mounds can

help explain apparent resilience of *Pterocarpus* to high groundwater salinity.

4.5.3   *Disturbance and climate change*

Historical landscape features of the study location seem to constrain tidal

influence to the east of the tidal creek, where the water flow follows a preferential

pathway only visible on historical images that show that section of the forest cleared by

deforestation (Fig. 4-2). The influence of these pathways is particularly noticeable in the

DS, when freshwater inflow is close to zero and the influence of brackish water in the

area increases (Fig. 4-7). West of the tidal creek, freshwater dominates regardless of the

season. In addition to the small channels that drain the tidal creek to the deforested area

(Fig. 4-2), the flow of brackish water to the east of the tidal creek may be enabled by lower soil elevation at the site. When estuarine wetlands suffer mass mortality of vegetation, the upper layer of peat substrate starts to decompose and collapses, reducing the elevation of the soil. In basin mangrove forests impacted by hurricanes, mass mortality of trees caused peat collapse of about 11mm year^{-1} due to decomposition of dead root material and sediment compaction (Cahoon et al. 2003). Peat collapse was also observed in lighting-created gaps in a mangrove forest (Sherman et al. 2000) and following mass mortality of freshwater marshes in Lousiana (DeLaune et al. 1994). When the elevation of the soil is reduced, the increase in water depth prevents the reestablishment of typical wetland vegetation (Lugo 1997). The disturbed areas then converts into open water ponds (DeLaune et al. 1994) or is colonized by faster growing or species more tolerant to flooding (Sherman et al. 2000).

Figure 4-10. Schematic representation of *Pterocarpus* forest microtopography and hypothesized water uptake of *Pterocarpus* trees during the dry season (top panel) and the wet season (bottom panel). Tree buttress roots form mounds of organic matter that create the hummocks and hollows microtopography. Seasonal changes in freshwater inflows to the forest determine the location of the water table and the area of unsaturated soil where water is accessed by the trees.
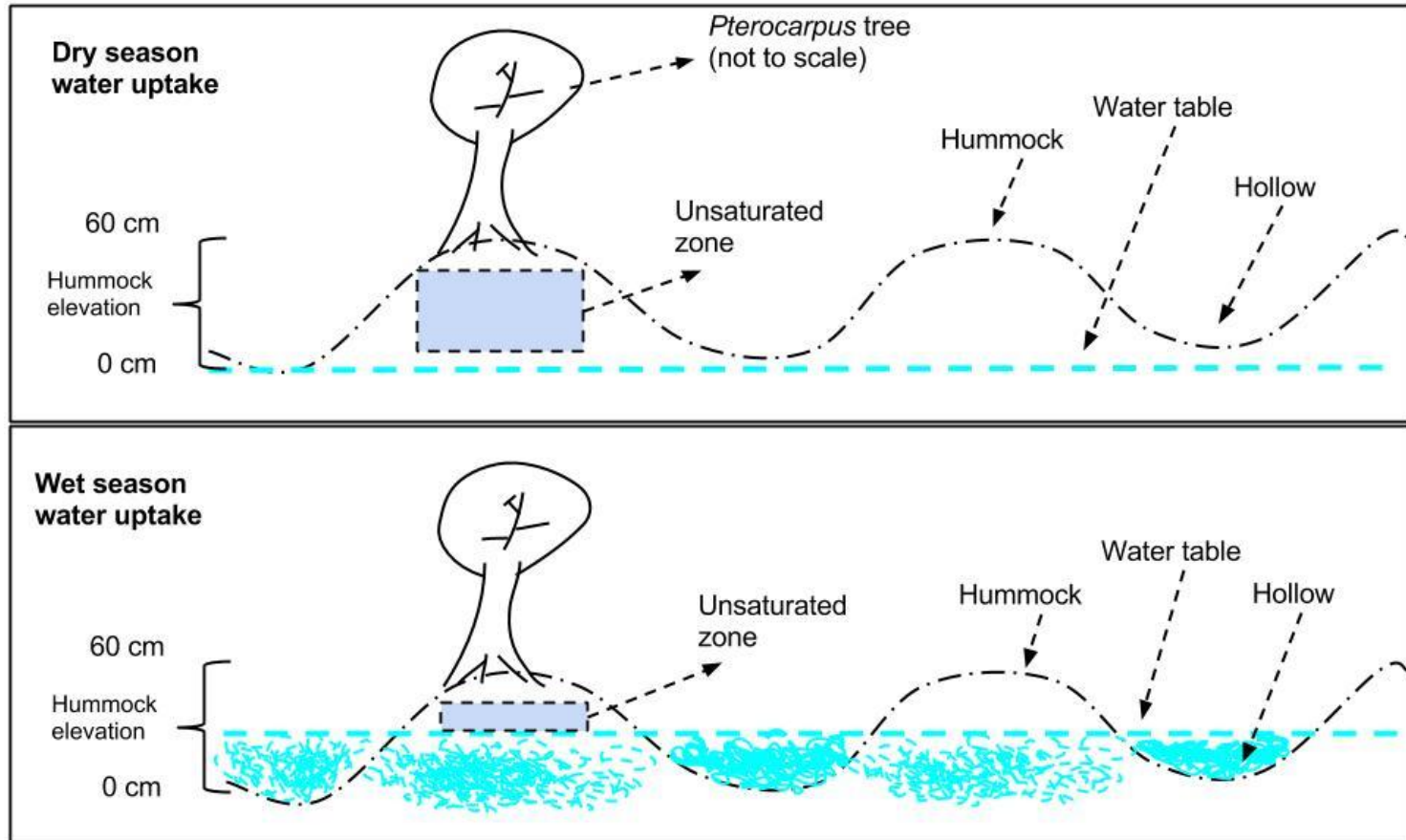
The recovery of the deforested area of the HNR *Pterocarpus* forest (Fig. 4-2) was also influenced by salinity. In this area, tree diameter at breast height (DBH) and canopy cover was lower than in other areas of the forest, while understory vegetation cover of salt tolerant species like the mangrove fern (*Acrosticum aureum*) was higher (Toledo-Rodríguez et al. 2013). In this case, the soil disturbance caused by deforestation allowed brackish water to flood the area, altering the species composition as expected by the general zonation pattern for *Pterocarpus* forests proposed by Migeot and Imbert (2011).

As global climate change advances through the remainder of the century, sea level will continue to rise (Nicholls and Cazenave 2010) and the weather in the Caribbean is expected to continue on a drying trend (Larsen 2000, IPCC 2007). As the long term weather becomes drier, strong rainfall events will become more frequent (IPCC 2007) while hurricanes are projected to become more intense due to increased moisture in the atmosphere (Mousavi et al. 2010). For riverine coastal wetlands, this means that annual freshwater inputs will be concentrated on single, intense-events while the saltwater inputs from the ocean become progressively higher. As we have observed in the HNR *Pterocarpus* forest, human-induced disturbances aggravate the natural threats of global climate change. Inland migration of coastal wetlands is limited by coastal development (Hopkinson et al. 2008). In the Caribbean, rapid population growth and associated developments is increasing human pressure on the coast (Simpson et al. 2012). Anthropogenic disturbances, by themselves or in combination with natural

disturbances, can alter hydrological and geomorphological features of the coastal

landscape to such extent that a shift in the state of the system is created (Lugo 2006).

Accounting for variability of natural occurrences in coastal management can be

challenging, but the resilience of freshwater wetlands in the Caribbean can be enhanced

by allowing natural hydrological conditions to persist and reducing human disturbances

that accelerate undesirable shifts in vulnerable ecosystems.

5. CONCLUSION

In the Matagorda study (Section 2), we observed that pond connectivity— a feature that enhances energy flow in salt marshes— is determined by an elevation threshold in the water level of separate water bodies within the marsh. However, we were unable to clearly identify the factors that drive water levels towards the threshold elevation. Instead, we conclude that a combination of factors drive the surface hydrology of the marsh, as suggested for other microtidal wetlands (Crase et al. 2013). By analyzing the results from Sections 3 and 4, along with the results of the Section 2, we conclude that the vulnerability of microtidal wetlands to climate change is not only based on the limited amount of sediments supplied by the tides, but also on the variable combinations between freshwater inputs and tidal exchange which in turn are influenced by anthropogenic disturbances.

The results from the HNR studies (Sections 3 and 4) show that the recent hydrologic alterations have altered the conditions of the estuary, which now behaves as a saltwater wedge or "blind" estuary (Savenije 2012). This type of estuary is commonly found in river outlets with microtidal conditions, like the ones found in the Caribbean (Miller and Lugo 2009) and the Mediterranean regions (Haralambidou et al. 2010). Furthermore, the results indicate that the dynamics between freshwater and saltwater inputs are still influenced by past disturbances to the site's hydrology. For example, the lack of channelized flow in TG4 is the result of the disconnection between the tidal creek and the Antón Ruiz River several decades ago. Similarly, the concentration of tidal

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influence in standing water east of the tidal creek is the result of past deforestation and eventual recovery of the area.

A report published in 2009 by US Corp of Engineers scientists stated that *"there is a level of uncertainty associated with the data gathered to date because it does not describe the conditions of the Rio Anton Ruiz outlet at the Caribbean Sea"* (Weston and Velez 2009). Indeed, by gathering a detailed dataset of the water level and salinity at the tidal creek of the *Pterocarpus* forest we have found that the conditions at the river outlet have changed dramatically since the construction of the drainage channel. An important contribution of this dataset includes the observation of the seasonal movement of the saltwater-wedge through the *Pterocarpus* forest as the mouth of the river opens and closes. Furthermore, we observed that excess outflows during the wet season actually increased the overall salinity of the system by the establishment of a well-developed saltwater wedge that moves inland during the dry season. In addition, results from Section 4 reveal that *Pterocarpus* trees rely mostly on very shallow soil water for their metabolic processes, providing further evidence to the hypothesis that *Pterocarpus* shallow root system functions as a mechanism to cope with increased salinity in the system.

Although our results are of particular interest to federal and state agencies in charge of the management and conservation of the HNR *Pterocarpus* forest, they contribute to our general understanding of microtidal wetland respond to climate change. At this site, we observed that— at any given magnitude of tidal influence— saltwater

intrusion into the inner parts of the forest is facilitated by lack of freshwater inputs into

the system. This observation is in close agreement with the hypothesis of mangrove

forest response to climate change proposed by Snedaker (1995), which states that

regardless of the relative elevation of the sea, reduced rainfall and runoff inputs act as

the most important stressor. This is particularly important for coastal wetlands in the

Caribbean where long-term weather predictions project drier conditions. In terms of

current management approaches this can be interpreted in a positive context, as

freshwater input is a variable that can be more effectively managed than sea level rise.

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