High Times and Dry Times In a Salt Marsh: Budgeting For Supplemental Freshwater Needs at the Big Boggy National Wildlife Refuge, USA

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Abstract

Salt marshes can be vulnerable to reduced freshwater input. Reduced freshwater inflows, particularly during a hot or dry summer season, can be catastrophic for vegetation productivity, organic accretion and inorganic sedimentation, and the ability of a marsh to maintain a sustainable elevation facing relative sea level rise (RSLR). Unfortunately, it is challenging for scientists to obtain inflow records for ungauged watersheds and link them with historical trends of salt marsh loss. We sought to address this challenge in Big Boggy National Wildlife Refuge (NWR), a small watershed in East Matagorda Bay, Texas. Our objective was to link the quantity of freshwater inflow with salt marsh sustainability and recommend management actions for the NWR. We first explored land cover trends and found that this watershed lost more than one-third of its low marsh since 1953. We then measured the streamflow into and out of the watershed, created a water budget, and modeled historical and future inflows from 1953 to 2100. Freshwater inflows have been increasing on average since 1953, but a combination of RSLR, sediment starvation, and punctuated seasonal droughts are likely responsible for the loss of salt marsh. We also estimated supplemental water needs during potential droughts out to 2100. We conclude that managers cannot fundamentally alter the accretion versus RSLR balance in this basin except by modifying freshwater input. Thus, during droughts, they should focus on providing these inputs and avoiding vegetation loss. Our work points to both water purchases and land management options that can achieve this goal.

Introduction

Salt marshes provide immense value through their numerous ecosystem services (Barbier et al. 2011) and yet they are often vulnerable to a reduction in their inflowing freshwater (Alexander and Dunton 2002). Adequate freshwater inflows are crucial to sustaining healthy marsh vegetation (Stachelek and Dunton 2013), and this vegetation provides support for commercial and recreational fisheries (Taylor et al. 2018; Bell 1997), migratory bird habitat (Darnell and Smith 2004), flood damage reduction (King and Lester 1995), and inland erosion mitigation (Shepard et al. 2011).

A reduction in freshwater inflow can even alter or kill vegetation that is well-adapted to salinity (White and Kaplan, 2017), particularly if the reduction occurs during a relatively hot or dry summer season (Alber et al. 2008) or in the context of additional pressures (Silliman et al. 2005; He et al. 2019). The production of biomass by many saltwater plant species is dependent on the quantity of freshwater input (Morris et al. 2002; Hessini et al. 2008), and any reduction also lowers the accretion and deposition of organic materials onto the marsh surface (Peteet et al. 2018). Moreover, for many marshes, particularly arid ones (Zedler and West 2008), the supply of inorganic sediments from upstream locations is critical to mitigate an accretionary deficit and maintain elevation relative to a rising sea level (Cahoon et al. 1996). For locations with high rates of relative sea level rise, a seasonal loss of vegetation can thus instigate a permanent transition into open water (Kirwan et al. 2008).

Marsh scientists and managers would typically need decades of inflow records to link the absolute quantity of inflowing freshwater with a historical trend of marsh loss or conversion. While historical precipitation or sea level records are available online from many meteorological stations (e.g. Huff et al. 2020), inflow records are only available for major rivers (e.g. USGS 2016). For smaller drainages and watersheds, inflow quantities must be modeled to roughly predict ungauged flow (Schoenbaechler, Guthrie, and Lu 2011) or measured directly by gauges in the field. However, the deployment of a network of gauges can be difficult to sustain over time. Thus, a challenge for researchers is to identify how to connect limited measurements of inflow quantity with longer-term trends.

We sought to address this challenge in a relatively small watershed that contained the Big Boggy National Wildlife Refuge (NWR). The inflow in this basin had been hydrologically-modified by agriculture, hydrocarbon extraction, the petrochemical industry, an extensive irrigation and drainage network, and the construction of roads and other barriers to flow. In addition, relative sea level rise has been rapid over the past several decades due to subsidence. Anecdotal stories by managers at the Big Boggy NWR have described repeated and intense summer periods of marsh hypersalinity, vegetation loss, and fish kills.

Our overall objective was to connect the quantity of freshwater inflow with longer-term marsh sustainability and recommend potential management actions for the NWR. Specifically, we sought to (1) identify salt marsh trends from 1953 to 2020, (2) quantify inflow rates and create a water budget to predict historical and future inflow trends during the critical summer seasons, and (3) and recommend actions that will help to sustain the salt marshes.

Methods

Study area

Big Boggy is a coastal watershed that drains into East Matagorda Bay (EMB) eight miles northeast of Matagorda, Texas, USA (Figure 1). The southern Big Boggy watershed consists of alluvium deposits created during the Holocene, deposited by Big Boggy Creek and Peyton Creek watersheds (Texas Water Science Center 2014). The southern reaches of the watershed are dominated by salt marshes. In the northern reaches of the watershed, Big Boggy Creek flows through cattle pastures with stream banks maintained and mowed periodically by the Matagorda County Drainage District. Several weir structures are present along Big Boggy creek that impede aquatic movement upstream and may limit freshwater flow downstream.

The study area encompasses the entire Big Boggy Creek watershed (Figure 1), but we focus in particular on the 1,500 hectares that is operated under the jurisdiction of the U.S. Fish & Wildlife Service (FWS) as the Big Boggy National Wildlife Refuge (NWR). There is an additional 316 hectares of Big Boggy NWR south of Lake Austin; this portion is not considered in the study because it flows to Lake Austin and is in a different watershed, due to hydrologic interruption by Chinquapin Road. The remaining portion of the study area is privately owned and functions primarily as cattle grazing pasture. Ninety acres of rice

paddies are present at Big Boggy NWR, all of which is seasonally planted with rye grass to provide winter browse for waterfowl (FWS 2013).

Salt marsh occurs primarily in the Big Boggy NWR. The zonation of its plant species is typical and similar to other sites along the central portion of the Texas Coast. *Spartina alterniflora* is the dominant low salt marsh vegetation. Slight elevation gradients within the marsh allows for colonization of halophytic species such as saltwort (*Batis maritima*), dwarf saltwort (*Salicornia bigelovii*), and saltgrass (*Distichlis spicata*). *S. spartinae* is abundant at elevations immediately above the intertidal zone and in areas that are too saline for typical upland grasses. Freshwater marsh vegetation includes cattail (*Typha latifolia*), alligator weed (*Alternanthera philoxeroides*), and common reed (*Phragmites australis*). Upland vegetation includes bushy bluestem (*Andropogon glomeratus*), honey mesquite (*Prosopis glandulosa*), eastern baccharis (*Baccharis halimifolia*), sugar hackberry (*Celtis laevigata*), Chinese tallow (*Triadica sebifera*), and Macartney rose (*Rosa bracteata*).

The Big Boggy NWR is an important destination for migratory waterfowl along the Central Flyway. The Texas Mid-Coast NWR Complex supports more than 100,000 shorebirds annually (FWS, 2013). Among these birds are threatened and endangered species such as the piping plover (*Charadrius melodus*), reddish egrets (*Egretta rufescens*), northern Aplomado falcon (*Falco femoralis septentrionalis*), and the interior least tern (*Sterna antillarum athalassos*). Dressing Point, one of most prominent bird rookeries on the Texas coast, is adjacent to Big Boggy. Dressing Point holds an average of 19 pairs of reddish egrets, in addition to being an important site for roseate spoonbills (*Platalea ajaja*) and royal terns (*Thalasseus maximus*).

Like many coastal areas in the U.S., marshes in the Big Boggy watershed are vulnerable to relative sea level rise (RSLR). Based on the two nearest gauges at Rockport and Galveston, the sea level has been rising at a rate of 0.63 cm/year (Holgate et al. 2013). Relative sea level rise, defined as the rate of rising ocean levels relative to the land at a given location, has the potential to outpace natural accretion rates of salt marsh resulting in the drowning of a marsh platform (Reed 1990).

The natural processes that combat RSLR in the Big Boggy watershed have been weakened by the hydrologic isolation of EMB (Buzan et al. 2011). Historically, the Colorado River of Texas provided direct freshwater flow to EMB, but this was stopped by the creation of the Colorado River Delta and subsequent construction of the Colorado River Navigation Channel (CRNC) (Clay 1949). In present day, the Colorado River flows are split between a flood discharge channel and the CRNC to the Gulf of Mexico. Because of these hydrologic modifications, the only significant freshwater inputs to EMB are attributed to a few small basins, such as that of Big Boggy Creek. Long term resilience of the oyster reefs and recreational fisheries in EMB has been an ongoing concern due to hypersalinity resulting from the hydrologic isolation. Sediment transport in EMB has been equally impacted by the diversion of flows into Matagorda Bay and the Gulf of Mexico (Morton et al. 1976; Morton 1979). The loss of sediment in the tidal water column since 1934 may be negatively impacting salt marshes in the surrounding watersheds by limiting the inorganic vertical accretion (Peteet et al. 2018; Bricker-Urso et al. 1989).

Identify salt marsh and land cover trends from 1953 to 2020

To identify historical changes in salt marsh cover, we analyzed aerial imagery from 1953, 1982, and 2020. These 1-meter horizontal resolution images were chosen based on their image quality and distribution in time, and were obtained through the Texas Natural Resources Information System (TNRIS 1953 and 2020) and the United States Geological Survey's (USGS) EarthExplorer (1982) databases.

Five distinct land cover classes were identified. In the order of increasing relative elevation, the land cover classes were: water, unconsolidated shore, low marsh, high marsh, and upland. The water class was characterized as areas of standing water with no vegetation present. Unconsolidated shore included unvegetated areas directly adjacent to water composed of sand or shell hash. Low marsh intertidal areas were dominated primarily by *Spartina alterniflora* but included halophytes such as *Batis maritima* and *Distichlis spicata*. High marshes were typically dominated by *S. spartinae*, although *D. spicata* was also present. The upland class included all non-marsh classes and human structures or impervious surfaces. In-situ sight identification of vegetation in 2020, and more than 100 geo-tagged photos, were used as a reference for the effort. Each land cover class was digitized using ArcGIS Pro (ESRI, Version 2.8) at a consistent map scale of 1:2,000. The temporal changes among the classified land cover maps were then analyzed in ArcGIS Pro using the Intersect tool to determine the land cover changes from 1953 to 2020.

Quantify average water flow rates into/out of the watershed and create a water budget

The characteristics of the hydrological network were quantified by determining the amount of water flowing through the watershed using flow data from a series of sensors and gauges. A model of the water budget for the watershed was then developed that incorporated the watershed area and precipitation. The resulting model helped us to determine the minimum freshwater inflow required for the watershed to function within normal historical bounds, and to forecast the water budget in the future as precipitation regimes change.

Sensor Stations

To collect the hydrological data, a series of sensors and gauges were deployed from June 23, 2020 through March 5, 2021 (Table 1). The sensors included Conductivity, Temperature, and Depth (CTD) dataloggers (CTD-Diver, Van Essen Instruments), Solinst Leveloggers (Levellogger 5 LTC, Solinst Canada Ltd.), Acoustic Doppler Current Profilers (ADCP; Aquadopp Profiler 1 MHz, Nortek Group), and a precipitation gauge (Onset tipping bucket rain gauge). The CTD dataloggers contained a pressure sensor that measures the hydrostatic pressure of the water to calculate total water depth, as well as a 4-electrode conductivity sensor that measures the specific conductivity of the water—a proxy for salinity. The CTD's were set to record measurements hourly. CTD dataloggers were deployed in a PVC pipe securely inserted into the stream bottom. The ADCP units use acoustic Doppler sensors to measure the flow speed of the

water column. The ADCP units were affixed to a steel fence post using coated steel cables and placed at the center of the stream channel. The precipitation gauge was deployed in close proximity to the other sensors and recorded the amount of rainfall occurring for each rainfall event. Additional hourly precipitation data was obtained from the LCRA rain gauge at Matagorda, Texas (Gauge Matagorda 1 S), 10 miles southwest of the study area.

Table 1
Sensor deployment locations and dates

Equipment Deployment Dates			
Sensor Group	Sensor Type	Start	End
1 - North Boggy	ADCP	6/23/2020	9/19/2020
1 - North Boggy	CTD	6/23/2020	3/5/2021
1 - North Boggy	ADCP	4/9/2021	6/15/2021
1 - North Boggy	CTD	4/9/2021	6/15/2021
1 - North Boggy	HOBO rain gauge	4/9/2021	6/15/2021
2 - South Boggy	ADCP	7/3/2020	9/19/2020
2 - South Boggy	CTD	7/3/2020	3/5/2021
2 - Upland	Barometer	6/23/2020	3/5/2021
3 - Pelton Lake	Solinst	6/23/2020	3/5/2021
3 - Pelton Lake	HOBO rain gauge	6/23/2020	9/19/2020
3 - Chinquapin	CTD	7/3/2020	3/5/2021

We set up the sensors at three stations. The “Upper Boggy” station (or UB) contained an ADCP and CTD sensor and was placed upstream of the salt marsh complex and north of Big Boggy NWR, where the creek banks were more riverine in form and the vegetation indicative of brackish conditions (Figure 1). This station primarily measured the freshwater inflow entering the refuge by way of Big Boggy Creek. The “Lower Boggy” station (or LB) was placed further south within the salt marshes of Big Boggy NWR, where Big Boggy Creek intercepts the Gulf Intracoastal Waterway (GIWW). This second station also had an ADCP and CTD, but primarily measured tidal flow in and out of the watershed. A third group of sensors consisting of one Solinst Levellogger and one precipitation gauge were placed along the eastern edge of Pelton Lake, and an additional CTD gauge was placed in Chinquapin Bayou on the east side of Chinquapin Road, on the other side of a culvert. This group of sensors was put in place to identify the degree of hydrologic isolation of Pelton Lake (the lake immediately below the label numbered 3 in Figure 1), and its connectivity with Chinquapin Bayou. We thus sought to identify the flows between the Big Boggy and Chinquapin watersheds using this third group of paired gauges.

The water level data from the CTD sensors and the flow rate data from the ADCP sensors were then vertically referenced into North American Vertical Datum (NAVD88) units, using a survey-grade Global Navigation Satellite System (GNSS) that included Global Positioning System (GPS) and GLONASS satellites. We then cross-referenced our datasets with the National Oceanic and Atmospheric Administration's (NOAA) Matagorda City tidal gauge (Station ID: 8773146) nine kilometers southwest of the study area.

Hourly stream flow volumes were calculated by multiplying the ADCP-measured, depth-averaged water velocities in a given direction by the cross-sectional area of the channel. The cross-sectional area also varied each hour based on the water level height, and this height was identified by using the accompanying CTD datasets (channel width * hourly water level depth = hourly cross-sectional area). Upstream and downstream flows were determined using the ADCP directional measurements.

Hydrological Budget

We then developed a hydrological budget using the sensor datasets, and used it to ask two questions: (1) what is the quantity of freshwater inflow coming down Big Boggy, how has it changed in the past, and how do we expect it to change over the next 100 years? And (2) are the salt marshes showing signs of hydrologic restriction and hypersalinity? To answer the first question, the water budget was divided into two bins based on the UB and LB stations. At each location, we accounted for the differences between the upstream-versus-downstream flow volumes, and then related them to the precipitation, evaporation, and the expected watershed size. The second question was answered by interpreting data from all sensor stations as well as synthesizing results from the land cover change methods. Groundwater is not explicitly accounted for in this water budget, as it was unlikely to play a large role in this region of coastal Texas due to the low permeability of the clay soils present (Soil Survey Staff 2021). We assume that while there may be losses or gains to groundwater, they balance out over the study period.

The water budget variables at each station included downstream flows, upstream flows, and precipitation. Although we had hourly data available for each of these variables over longer time frames at various stations, we chose to only use the data from 7/4/2020 to 9/19/2020 to build the budget for dates in which ADCP stations were active. During these dates in 2020, the precipitation balance was just below the mean for the period over the past several decades (see Results for more). This period was uniquely important because it was during these summer months when inflows are at their lowest and most critical. We aggregated over this time frame based on an initial investigation of the relationship between measured precipitation and perceived flow at the stations, wherein we concluded that these three months of data were not sufficiently long enough for us to quantitatively account for timing delays caused by complex watershed effects and antecedent conditions. Similarly, there may be limitations in the dataset due to the relatively short period of record. It is possible that flows during our period of record (July, August, September) do not match flow patterns for other periods throughout the year.

For each station, we first found the imbalance between upstream and downstream flow volume. Upstream flows could include both incoming tides and storm surges. Downstream flows could include

outgoing tides and freshwater flows from upstream reaches of the watershed. We then calculated the precipitation volume for the watersheds that fed into each station. To do this, we obtained the precipitation from the Water Data for Texas website operated by the Texas Water Development Board (TWDB) (Texas Water Development Board 2021). This precipitation dataset combines data from different precipitation stations within a quadrant to estimate precipitation throughout the entire quadrant. We next multiplied these datasets by the “total watershed area” and the “effective watershed area” for each station. The total watershed area was identified using the Watershed tool in ArcGIS Pro and a 1-meter Digital Elevation Model (DEM); two total watershed products were produced to delineate the separate sections of the landscape that uniquely contributed to the Upper Boggy and Lower Boggy stations. The effective watershed area was defined as the area across which the precipitation could be assumed to have fallen (minus any evaporation), that would then be equivalent to the observed quantity of inflow reaching each station. In other words, the effective watershed area is the area of capture that is multiplied by a precipitation measurement to obtain the amount of precipitation-driven inflows (precipitation minus evaporation) that are observed flowing past our sensor station.

Hindcasted and forecasted inflow volumes

Finally, we hindcasted and forecasted the freshwater inflow volume at the Upper Boggy and Lower Boggy stations over the time period from 1954 to 2100. Because of the relatively short period of data collection (6/23/2020-3/5/2021 for CTDs; 7/4/2020-9/19/2020 for ADCPs), there may have been additional uncertainty in hindcasting and forecasting, due to the potential variability in how the watershed reacts to periods of high or low precipitation. Because of this, the forecasting scenarios were estimated based on the mean trend. For the hindcasting, the estimated historic freshwater inflows for each year from 1954 to 2019 was aggregated during the same summer months as the budget (July, August, and September). The values were derived from the TWDB precipitation dataset, in the same manner as for the budgeting described previously. To identify past years where net flows were above and below what was considered a typical year for rainfall in the region, we calculated the mean trend across the years and then found the root mean square error (RMSE) deviation from that trend line. We then graphed both the mean trend line and the RMSE ranges, to help depict the most aberrant years. For the aberrant years that fell outside of and below the lower RMSE bound (the drought years), we calculated the amount of supplementary inflow that would be needed to bring the total inflow back up to the lower RMSE bound, as well as back up to the mean. We considered these two values as indicative of the estimated range of supplemental volume that would be needed to bring the inflow out of drought conditions.

For the forecasting, we evaluated the effect of three potential climate change scenarios on the mean inflow volumes, again using the budgeted months. Additionally, the mean trend line used in hindcasting was continued through 2100 as a fourth scenario in which the rates of change between 1954 and 2019 were maintained moving forward. The three climate change scenarios were defined by the Intergovernmental Panel on Climate Change (IPCC) and statistically downscaled by Jiang and Yang (2012), and include the A1B, A2, and B1 scenarios. Each scenario predicts future trajectories of climate change depending on global changes in demographics, and economical or technological developments

(Nakicenovic et al. 2000), wherein A2 is the most severe outcome, A1B is intermediate, and B2 is the most favorable.

Results

Identify salt marsh and land cover trends from 1953 to 2020

Between 1953 and 2020, the Big Boggy watershed experienced notable changes in land cover composition. Of the five cover classes, three classes lost more than a quarter of their total area (Figure 2). Low marsh lost 35% of its original area, with 43% of this loss converting to open water. Of the nearly 800 hectares of low marsh present in 1953, 512 hectares remain in 2020. High marsh experienced a net loss of 27%. Some low marsh transgressed onto the former uplands and high marshes north of Pelton Lake and Lake Kilbride (the east-west lying lake to the west of Pelton Lake, but east of Big Boggy Creek). Interestingly, the low marsh is not transgressing further up the main channel of Big Boggy Creek. The total coverage of water in the study area more than doubled from 321 hectares to 681 hectares.

Quantify average water flow rates into/out of the watershed and create a water budget

The total precipitation observed during the study period (June 24th, 2020 – March 4th, 2021) was 60 cm (23 in.; Figure 3a). During this period, two tropical cyclones passed by the area, Hurricane Hanna and Tropical Storm Beta. Precipitation in 2020 (32.36 cm) was 5% lower than the average rainfall from 1954 to 2020 (34.12 cm).

We found that the water level and salinity at all three stations were affected by both tides and precipitation events (Figure 3b). The UB station was most responsive to precipitation as shown with its brief peaks following precipitation events, followed by a generally rapid return to its baseline. The average daily tidal range at UB was approximately 10 cm. Over the course of the study period, the average salinity at UB was 13 PSU, the lowest among the three stations (Figure 3c). Salinity at UB was influenced by the amount of precipitation during an event as well as the storm surge from the GIWW. For example, the precipitation and storm surge from Hurricane Hanna began on July 21, 2020. While large amounts of precipitation were reported (4 cm in an hour at peak), salinity at UB remained relatively steady at 3.3 PSU then sharply rose to 10 PSU on July 26, 2020, as the saltwater wedge from LB made its way upstream into the more riverine channel at UB. Over the coming days and weeks, salinity gradually declined to 4 PSU as saline floodwaters were flushed out of the station area by inflowing freshwater. During periods of sparse precipitation, salinity at UB gradually increased to a peak salinity of 27 PSU.

At the LB station, water level is largely driven by the tide as well as storm surges. Precipitation can also cause water levels at LB to rise, just slightly below those at UB (1.4 m at UB compared to 1.3 m at LB on September 22, 2020). The average daily tidal range at LB was approximately 10 cm. Average salinity at

LB was the highest between the three stations at 21 PSU. Through much of the study period, salinity remained relatively stable dropping down to as low as 10 PSU following a moderate precipitation event in late December.

The Chinquapin station was the most unique of the three. At times, the Chinquapin station appeared to respond differently to precipitation events, and the water levels generally took longer to decrease after these events, suggesting that it was responding to different inflow sources (i.e., Lake Austin). Further, the daily tidal range at Chinquapin was 4 cm, the lowest of the three stations, potentially indicating a hydrologically isolated location. Salinity at the Chinquapin station was the most variable between the three. For example, during Hurricane Hanna, salinity peaked at 30 PSU during the storm surges. Following the storm surge, salinity dropped to 6 PSU. Salinity at Chinquapin rose sharply on August 20, 2020, from 13 PSU to 24 PSU on August 21, 2020. The salinity at Chinquapin maintained levels between 20 PSU and 35 PSU before dropping to 4 PSU due to events not captured in our data. This further suggests response to different inflow sources.

The ADCPs provided stream flow volume and direction (Figure 4b). When coupled with water elevation data from the accompanying CTD (Figure 4a), we obtained a more complete understanding of flows at the UB and LB stations. The volume of flow between UB and LB can differ drastically. At the peak of Hurricane Hanna (July 25, 2020), almost 65,000 m³/hour was flowing at LB. In contrast, the peak upstream flow at UB was only 19,000 m³/hour. The flow volumes also varied drastically during regular tidal periods, with a difference of almost 5,000 m³/hour between the incoming and outgoing tides and at LB, and a difference of only 200 m³/hour at UB. As shown in Figure 4b, the downstream flows at LB were not in balance with upstream flows. This further supports the notion that there were alternate outlets for LB flow exiting into the GIWW (other than out Big Boggy Creek).

Hydrological Budget

At the UB station, the total upstream flow during the budgeted period of 7/4/2020 to 9/19/2020 was 1,127 ac-ft (Figure 5) consisting of incoming tides. Total downstream flow measured 1,719 ac-ft, consisting of outgoing tides and freshwater flows. The imbalance between upstream and downstream flows was 592 ac-ft and represented the freshwater inflow quantity. It thus represented the difference between the total precipitation and evaporation in the watershed further upstream, excepting any unbudgeted losses or gains (see Methods).

At the same time, we found that the total watershed area upstream of the UB station was 7,927 hectares (Figure 5). The quantity of precipitation multiplied by this area resulted in a far higher value than the 592 ac-ft observed inflow volume. Thus, the effective watershed area was calculated as 225 hectares, which was only 2.84% of the total potential area. This 2.84% for the effective watershed area matched what one could expect given direct capture of precipitation minus evaporation in the system only, meaning our sensor station may have only observed water delivered directly into open water bodies, low marsh, and high marsh areas, while overland flows may not play as large a role as previously assumed. There is not

a simple explanation regarding the small effective watershed area. One possible explanation is due to the low relief that is present at upland areas. These flat upland areas may not readily flow into the stream and instead might have been subject to high rates of evaporation without ever making it to Big Boggy Creek.

At the LB station, the total upstream flow during the study period was 5,393 ac-ft, consisting of incoming tides and storm surges. Total downstream flow measured 3,422 ac-ft, consisting of outgoing tides and freshwater flows. The difference between upstream and downstream flows was 5,985 ac-ft. Precipitation was estimated as 4,022 ac-ft.

It is important to note that there was a large imbalance in the water budget at LB. There was approximately 5,985 ac-ft unaccounted for, and when taking into consideration the downstream flows from UB into LB as well (1,719 ac-ft), there was a total of 7,704 ac-ft. This large surplus of water suggests that there were losses to other outlets in the marsh complex (Figure 6). These outlets likely only connect and move water into the GIWW when water levels exceeded 0.45 m (NAVD88). The only alternate outlet that may not be water-level dependent was the culvert beneath Chinquapin Road.

We found that the total watershed area upstream of the LB station (minus that upstream of UB) was 2,780 hectares (Figure 7). The effective watershed area was calculated as 1,532 hectares, which was 55% percent of the total potential area. This percent was much higher as compared with UB, because much of the LB watershed was effectively directly capturing the precipitation in open water, low marsh, or high marsh areas.

Hindcasted and forecasted inflow volumes

We found that the average inflows at UB and LB were 207 acre-feet per month and 1,414 acre-feet per month, respectively. The hindcasted inflows at UB had a positive mean trend over time, increasing over the budgeted time period from 1954 to 2019 (Figure 8a, 8c). The inflows at LB followed the same trend although due to the sheer size difference in area between the Upper and Lower Boggy effective watersheds, LB experienced a much greater volume of inflow accompanied by greater variation between wet and dry years. Note that for each year only three months were used to match the study period (July, August, and September).

The forecasted mean inflow scenarios at UB and LB did not strongly deviate from one another through the year 2100, as compared to the annual variability that was possible (Figure 8b, 8d). The historical trend resulted in inflows that will slowly increase at 39% for UB and LB, and this was the best-case scenario. However, this scenario does not account for the complexities and nuance in predictions for a changing climate. The second-best scenario was the B1 scenario. Under this climate scenario, a 4% increase in inflow was predicted at UB and LB. The third scenario, A1B, estimated a 3% increase in inflow at UB and LB. Finally, A2 yielded the worst-case prediction, where inflow at UB and LB were predicted to decrease by 4%. It is crucial to note that these were just predicted changes in the mean trend of inflow. Years with inflows higher or lower than the mean and outside of the RMSE bounds will likely occur, and

these are more likely to alter ecosystem functioning. In summary, inflow quantities will be less sensitive to mean changes in the climate and more sensitive to its variability over time.

Discussion

The Big Boggy watershed lost more than one-third of its low marsh area since 1953. However, during this same period, the freshwater inflow had been increasing on average during the summer months when droughts are most damaging. The combination of these findings suggest that there is likely an alternative cause for the loss, beyond changes in the mean inflow rate during these months. Instead, the datasets point to three interconnected causes: relative sea level rise (RSLR), sediment starvation, and inflow variability.

First, RSLR often results in a similar patterning of marsh loss as that seen in the Lower Boggy (LB) marsh areas, wherein the interior of individual marsh islands is lost while the edges remain. In these cases, the vertical accretion of the marsh interior cannot match RSLR (Morris et al. 2002), while the marsh edges receive sufficient sediment inputs and are able to vertically accrete to keep up with RSLR (Duran Vincent et al. 2021). These conditions result in the fragmentation patterns that were observed. The quantity of precipitation and freshwater inflow is known to be a critical feature that can help counteract low marsh loss by promoting vertical marsh accretion (Craft 2007) through organic growth and deposition (Więski 2014).

Second, sediment starvation is a common factor of marsh loss in other regions (Peteet et al. 2018; Mariotti and Fagherazzi 2013) and the historical loss of sediment-rich flows from the Colorado River means that marshes in Big Boggy—and the surrounding watersheds in EMB—rely almost solely on organic sediment accretion to match RSLR. Before the Colorado River was diverted, tidal waters from a more sediment-rich EMB likely inundated marsh platforms and deposited more inorganic sediments. In the absence of this sediment-rich environment, marsh platforms have been forced to maintain elevation through organic accretion alone. However, based on the fragmentation patterns in the low marsh and its transgression into former areas of high marsh, organic accretion does not appear to be capable of keeping pace with RSLR at contemporary rates for this location.

Third, inflow variability, as opposed to a change in the mean trend of inflow, plays a major role and can modulate the RSLR-accretion balance. Variable or extreme inflows that fall well above and below the mean trend can impose a greater influence on marsh productivity, than small deviations over time in the trend itself. The key feature for the Big Boggy NWR is whether the marshes receive the inflowing water during the time periods, and in the spatial locations, that they require it for specific ecological processes. For instance, the large area of low marsh on the eastern side of the NWR, further away from the main channel of Big Boggy Creek, shows signs of hydrologic restriction. In this portion of the NWR, there is a relatively low tidal range and a highly variable response to precipitation as shown by the Chinquapin gauge. There are also several elevation-dependent flow outlets as shown by the LIDAR datasets (e.g., Fig. 6). Excavating these elevation-dependent outlets would increase the hydrological connectivity of the

marsh system, potentially alleviating concerns of hypersalinity in Pelton Lake. Further, freshwater inputs are likely to be especially crucial for this area during the summer months when the tidal range is quite small, and heat and evaporation are greatest.

Since managers cannot fundamentally alter the RSLR-accretion balance other than by modifying freshwater input – and they cannot turn the clock back to the pre-Colorado River diversion conditions of 1934 - their efforts can instead focus on providing consistent freshwater inflows, during particularly dry summers.

Potential management and restoration actions to increase inflow

To determine the supplemental flow quantities that managers could provide to offset a drought or drought-like conditions, we developed an inflow decision tool for the Big Boggy Creek watershed (downloadable from cml.tamu.edu). This tool uses inputs of precipitation to calculate the amount of inflow, and then determines the supplemental flow needed to fall within the range “normal”, defined here as a value falling between the mean trend and the lower RMSE bound. It can also estimate the total freshwater contribution to East Matagorda Bay. For example, using 2011 as an example of a severe drought year, we can estimate the precipitation needed to bring inflows arriving at the upstream portion of the NWR (at the UB station) up to the lower RMSE bound of what we consider acceptable (see Figure 8a). Approximately 4.7 inches of additional rainfall would have raised the net inflows arriving to the lower RMSE bound, and 11.8 inches to reach the mean trend (during July to September of that year). The supplemental inflow volumes that one would have needed to acquire between July and September 2011 were 218 ac-ft per month and 547 ac-ft per month, respectively.

As part of its duties, the Lower Colorado River Authority (LCRA) sells water to users in the Colorado River basin for municipal, agricultural, and wildlife management uses (Austin et al., 2015). Existing agriculture in the area utilizes irrigation canals to transport water, and at least one of the canals is adjacent to the eastern portion of the marsh that is the most hypersaline (near the Chinquapin gauge location; some relatively minor elevation modifications would be required; FWS, 2013). In 2021, water purchases from LCRA were available at a rate of \$66.14 per acre-foot delivered (E. Ray, personal communication, October 26, 2021). If the 2011 drought happened in the summer of 2021, it would cost ~\$14,000 per month to return the conditions to within the normal RMSE bounds. This cost is relatively low compared to typical salt marsh restoration actions.

An alternative perspective is to think about how much land is needed to capture precipitation and convert it into supplemental inflow - in other words, to increase the effective watershed area and rely on precipitation rather than purchasing supplemental flows. Resource managers can use the tool to estimate the additional effective watershed area needed to offset the decline in inflows. Using the 2011 drought example, an additional 362 hectares would be needed to meet the lower bound, and 885 hectares would be needed to reach the mean trend. This would more than double the effective watershed at a minimum.

A series of impounded ponds located immediately north of Big Boggy NWR present themselves as potential restoration targets, but the effective watershed would increase by only 17 hectares. 386 hectares of upland areas could also be graded to drain into an existing channel more readily and facilitate upslope marsh migration in response to sea level rise. However, most of these lands are privately owned and incentives would likely be necessary for landowner cooperation (e.g., fee simple land acquisition or conservation easements). In summary, the most realistic and immediately impactful action would be to purchase water through the LCRA during times of drought.

Conclusion

Freshwater inflows are critical to the long-term resilience of salt marshes in the Big Boggy Creek watershed and to the estuarine waters of East Matagorda Bay. Relative sea level rise, sediment starvation, and inflow variability are likely responsible for the loss of salt marsh in this watershed. The natural processes responsible for maintaining healthy salt marsh ecosystems have been disrupted in the past and the negative impacts of sea level rise will likely be intensified by future climate change. Two restoration actions have been identified as having the greatest likelihood for improving salt marsh health: supplemental water purchases and increasing the effective watershed area through marsh creation. Our approach provides a framework that can be replicated in ungauged watersheds around the world to answer questions about the contribution of inflows and their relation to long-term marsh sustainability.

Declarations

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Competing Interests

The authors have no relevant financial or non-financial interests to disclose.

Data Availability

The datasets generated during the current study are available at cml.tamu.edu.

Author contributions

MJM, RAF, TPH designed the research; MJM, TPH, BB performed the data collection duties, assisted in field efforts, MJM, TPH analyzed data. MJM, RAF primarily wrote the manuscript, and all have edited it.

References

1. Alber M, Swenson EM, Adamowicz SC, Mendelssohn IA (2008) Salt Marsh Dieback: An overview of recent events in the US. *Estuar Coast Shelf Sci* 80:1–11. doi: 10.1016/j.ecss.2008.08.009
2. Alexander HD, Dunton KH (2002) Freshwater Inundation Effects on Emergent Vegetation of a Hypersaline Salt Marsh. *Estuaries* 25:1426–1435. doi: 10.1007/BF02692236
3. Austin B, Kennedy A, Osting T, Walker C (2015) Evaluation of Freshwater Delivery Alternatives to East Matagorda Bay. *AquaStrategies*
4. Barbier EB, Hacker SD, Kennedy C, Koch EW, Stier AC, Silliman BR (2011) The value of estuarine and coastal ecosystem services. *Ecol Monogr* 81:169–193. doi: 10.1890/10-1510.1
5. Bell FW (1997) The economic valuation of saltwater marsh supporting marine recreational fishing in the southeastern United States. *Ecol Econ* 21:243–254. doi: 10.1016/S0921-8009(96)00105-X
6. Bricker-Urso S, Nixon SW, Cochran JK, Hirschberg DJ, Hunt C (1989) Accretion Rates and Sediment Accumulation in Rhode Island Salt Marshes. *Estuaries* 12:300–317. doi: 10.2307/1351908
7. Buzan D, Cook B, Fontenot M, Hardy T, Hoffpauir R, Kennedy K, Thornton O, Trungale J, Wakefield C, Watters S (2011) BBEST Environmental Flow Regime Recommendations Report. Colorado and Lavaca Rivers and Matagorda and Lavaca Bays Basin and Bay Expert Science Team
8. Cahoon DR, Lynch JC, Powell AN, Estuarine (1996) coastal and shelf science 43:19-32. doi: 10.1006/ecss.1996.0055
9. Clay C (1949) The Colorado River Raft. *Southwestern historical quarterly* 52:410–426
10. Craft C (2007) Freshwater Input Structures Soil Properties, Vertical Accretion, and Nutrient Accumulation of Georgia and U.S. Tidal Marshes. *Limnol Oceanogr* 52:1220–1230. doi: 10.4319/lo.2007.52.3.1220
11. Darnell TM, Smith EH (2004) Avian Use of Natural and Created Salt Marsh in Texas. *USA Waterbirds* 27:355–361.. doi: 10.1675/1524-4695(2004)027[0355:AUONAC]2.0.CO;2.
12. Duran Vinent O, Herbert ER, Coleman DJ, Himmelstein JD, Kirwan ML (2021) Onset of runaway fragmentation of salt marshes. *One Earth* 4:506–516. doi: 10.1016/j.oneear.2021.02.013
13. Feagin RA, Ravens TM, Möller I, Yeagei KM, Baird AH, Thomas DH, Lozada-Bernard SM (2009) Does Vegetation Prevent Wave Erosion of Salt Marsh Edges?. *Proceedings of the National Academy of Sciences - PNAS* 106:10109-10113. doi: 10.1073/pnas.0901297106
14. Hanson A, Johnson R, Wigand C, Oczkowski A, Davey E, Markham E (2016) Responses of *Spartina alterniflora* to Multiple Stressors: Changing Precipitation Patterns, Accelerated Sea Level Rise, and Nutrient Enrichment. *Estuaries Coasts* 39:1376–1385. doi: 10.1007/s12237-016-0090-4
15. He Q, Silliman BR, van de Koppel J, Cui B (2019) Weather fluctuations affect the impact of consumers on vegetation recovery following a catastrophic die-off. *Ecology (Durham)* 100:e02559-n/a. doi: 10.1002/ecy.2559
16. Hessini K, Ghandour M, Albouchi A, Soltani A, Werner KH, Abdelly C (2008) Biomass production, photosynthesis, and leaf water relations of *Spartina alterniflora* under moderate water stress. *J Plant Res* 121:311–318. doi: 10.1007/s10265-008-0151-2

17. Holgate SJ, Matthews A, Woodworth PL, Rickards LJ, Tamisiea ME, Bradshaw E, Foden PR, Gordon KM, Jevrejeva S, Pugh J (2013) New Data Systems and Products at the Permanent Service for Mean Sea Level. *J Coastal Res* 29:493–504. doi: 10.2112/JCOASTRES-D-12-00175.1
18. Huff T, Feagin R, Figlus J (2020) Enhanced tide model: Improving tidal predictions with integration of wind data. *Shore and beach* 40–45. doi: 10.34237/1008824
19. Jiang X, Yang Z (2012) Projected changes of temperature and precipitation in Texas from downscaled global climate models. *Climate Res* 53:229–244. doi: 10.3354/cr01093
20. King SE, Lester JN (1995) The value of salt marsh as a sea defence. *Mar Pollut Bull* 30:180–189. doi: 10.1016/0025-326X(94)00173-7
21. Kirwan ML, Murray AB, Boyd WS (2008) Temporary vegetation disturbance as an explanation for permanent loss of tidal wetlands. *Geophys Res Lett* 35:L05403. .. doi: 10.1029/2007GL032681
22. Mariotti G, Fagherazzi S (2013) Critical width of tidal flats triggers marsh collapse in the absence of sea-level rise. *Proceedings of the National Academy of Sciences - PNAS* 110:5353-5356. doi: 10.1073/pnas.1219600110
23. Morris JT, Sundareshwar PV, Nietch CT, Kjerfve B, Cahoon DR (2002) Responses of Coastal Wetlands to Rising Sea Level. *Ecology (Durham)* 83:2869–2877. doi: 10.1890/0012-9658(2002)083[2869:ROCWTR]2.0.CO;2
24. Morton RA, Pieper MJ, McGowen JH (1976) Shoreline changes on Matagorda Peninsula (Brown Cedar Cut to Pass Cavallo. an analysis of historical changes of the Texas Gulf shoreline
25. Nakicenovic N, Alcamo J, Davis G, de Vries B, Fenhann J, Gaffin S, Gregory K, Grubler A, Jung TY, Kram T, La Rovere EL, Michaelis L, Mori S, Morita T, Pepper W, Pitcher HM, Price L, Riahi K, Roehrl A, Rogner H, Sankovski A, Schlesinger M, Shukla P, Smith SJ, Swart R, van Rooijen S, Victor N, Dadi Z (2000) Special Report on Emissions Scenarios: a special report of Working Group III of the Intergovernmental Panel on Climate Change. Cambridge University Press, New York, NY (US). United States.
26. Peteet DM, Nichols J, Kenna T, Chang C, Browne J, Reza M, Kovari S, Liberman L, Stern-Protz S (2018) Sediment starvation destroys New York City marshes' resistance to sea level rise. *Proceedings of the National Academy of Sciences - PNAS* 115:10281-10286. doi: 10.1073/pnas.1715392115
27. Reed DJ (1990) The impact of sea-level rise on coastal salt marshes. *Prog Phys Geogr* 14:465–481. doi: 10.1177/030913339001400403
28. Schoenbaechler C, Guthrie C, Lu Q (2011) Coastal Hydrology for East Matagorda Bay. Texas Water Development Board
29. Shankman D (1996) Stream Channelization and Changing Vegetation Patterns in the U. S. Coastal Plain. *Geogr Rev* 86:216–232. doi: 10.2307/215957
30. Shepard CC, Crain CM, Beck MW (2011) The protective role of coastal marshes: a systematic review and meta-analysis. *PLoS ONE* 6:e27374. doi: 10.1371/journal.pone.0027374
31. Silliman BR, van de Koppel J, Bertness MD, Stanton LE, Mendelssohn IA (2005) Drought, Snails, and Large-Scale Die-Off of Southern U.S. Salt Marshes. *Science (American Association for the*

- Advancement of Science) 310:1803–1806. doi: 10.1126/science.1118229
32. Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture (2021) Web Soil Survey. Available online at the following link: <http://websoilsurvey.sc.egov.usda.gov/>. Accessed 5 May 2021
 33. Stachelek J, Dunton K (2013) Freshwater inflow requirements for the Nueces Delta, Texas: *Spartina alterniflora* as an indicator of ecosystem condition. *Texas Water Journal* 4:62–73
 34. Taylor MD, Gaston TF, Raoult V (2018) The economic value of fisheries harvest supported by saltmarsh and mangrove productivity in two Australian estuaries. *Ecol Ind* 84:701–709. doi: 10.1016/j.ecolind.2017.08.044
 35. Texas Water Development Board (2021) Lake Evaporation and Precipitation. Water Data for Texas. <https://www.waterdatafortexas.org/lake-evaporation-rainfall>. Accessed 10 June 2021
 36. Turner RE, Swenson EM, Milan CS Organic and Inorganic Contributions to Vertical Accretion in Salt Marsh Sediments Concepts and Controversies in Tidal Marsh Ecology. Springer Netherlands, Dordrecht, pp583–595
 37. United States Department of Agriculture (1953) Matagorda USDA historic imagery, 1953-02-08
 38. United States Department of Agriculture (2020) Texas NAIP imagery, 2020-04-01.
 39. United States Fish and Wildlife Service (2013) Texas Mid-Coast National Wildlife Refuge Complex, Brazoria, Fort Bend, Matagorda, and Wharton Counties, TX; Comprehensive Conservation Plan and Environmental Assessment. Interior Department Documents and Publications
 40. United States Geological Survey (2016) USGS Water Data for the Nation
 41. Vinent O, Herbert E, Kirwan M (2019) Lower threshold for marsh drowning suggests loss of microtidal marshes regardless of sediment supply
 42. Voss C, Christian R, Morris J (2013) Marsh macrophyte responses to inundation anticipate impacts of sea-level rise and indicate ongoing drowning of North Carolina marshes. *Mar Biol* 160:181–194. doi: 10.1007/s00227-012-2076-5
 43. White E, Kaplan D (2017) Restore or retreat? saltwater intrusion and water management in coastal wetlands. *Ecosystem health and sustainability* 3:e01258-n/a. doi: 10.1002/ehs2.1258
 44. Więski K, Pennings S (2014) Climate Drivers of *Spartina alterniflora* Saltmarsh Production in Georgia, USA. *Ecosystems* (New York) 17:473–484. doi: 10.1007/s10021-013-9732-6
 45. Zedler JB, West JM (2008) Declining Diversity in Natural and Restored Salt Marshes: A 30-Year Study of Tijuana Estuary. *Restor Ecol* 16:249–262. doi: 10.1111/j.1526-100X.2007.00268.x

Figures

Figure 1

Project study area located 70 miles southwest of Houston, Texas. The red outline displays the total watershed. The interior red line separates the watersheds of Upper and Lower Boggy. The smaller blue outline displays the boundary of Big Boggy NWR. The dashed black line displays the extent to which land cover was classified. The sensor stations are also displayed at Upper Boggy (1), Lower Boggy (2), and Chinquapin/Pelton Lake (3).

Figure 2

Land cover change from 1953 to 2020 (top panel). Note the discontinuous y-axis to accommodate the large quantity of upland area. Land cover classification from 1953 (middle panel) to 2020 (bottom panel) is displayed, focusing on the Big Boggy NWR extent. Note the loss of high and low marsh.

Figure 3

Precipitation (a), water elevation (b), and salinity (c) as obtained from the LCRA rain gauge (a) and the deployed CTDs (b, c). Water elevation is in NAVD88 meters. Salinity is in Practical Salinity Units (PSU), which is similar to a parts per thousand (ppt) measurement.

Figure 4

Water elevation (a) and flow volume/direction (b) for ADCP sites at Upper and Lower Boggy. Positive values indicate upstream flows, negative values indicate downstream flows.

Figure 5

A visualization of the water budget for Big Boggy from July 2020 to September 2020.

Figure 6

Alternate outlets for flowing water (white arrows) and water elevations needed to flood portions of the watershed (NAVD88 meters). Upper Boggy (a), Lower Boggy (b), and Chinquapin (c) sites are also depicted.

Figure 7

The Big Boggy watershed as derived from a 1-meter DEM and processed using ArcGIS Watershed tools.

Figure 8

Hindcasted (a, c; 1954 – 2019) and forecasted (b, d; 2021 – 2100) inflows at Upper and Lower Boggy. The inflow rates we observed in the field in 2020 are depicted (dashed line) for comparison. The inflow patterns are identical, but please note the much greater volumes contributed by Lower Boggy due to its larger effective watershed area. For each year, only three months were used to match the 2020 study period (July, August, and September).