Early Responses to Runnels in Southern New England Salt Marshes

A technical report on results from a collaborative experiment testing the efficacy of runnels, an emerging climate adaptation tool, in Buzzards Bay.

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Contributions: Besterman led project design and coordination, sample collection, analyses and report writing. Jakuba and Deegan contributed to project design and conception, field sampling, and data analysis. Ferguson and Brennan contributed to site selection, runnel design and implementation, adaptive management and other guidance. Sullivan led the sample collection, processing, and analysis for data on soil water content, porewater salinity, redox potential, and decomposition. Costa provided training, equipment, and analytical support on elevation surveys. Neil Ganju and Noa Randall (USGS) provided and analyzed turbidity data. Anastasia Pulak (Woodwell Climate Research Center) provided significant field and lab support for multiple analyses.

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Summary

- This project tested runnels, an emerging climate adaptation technique used to restore tidal hydrology and revegetate marshes experiencing interior open-water conversion.

- Runnels were tested at two salt marshes in Buzzards Bay. The study includes 10 sites with experimental runnels and 10 reference sites, split between both marshes. Sites were monitored before (2020) and after (2021) runnels were installed. This report presents data from before and after runnel creation, collected at 12 of the 20 sites where we completed intensive sampling (6 sites each marsh).

- Little Bay in Town of Fairhaven is a fringing marsh exposed to an open embayment. Ocean View Farm in Town of Dartmouth is a sheltered marsh within a back-barrier salt pond, separated from Buzzards Bay by barrier spit and connected by a narrow tidal inlet. Marshes and sites within marshes differed in platform elevations, level of peat degradation, depth shallow water in areas of vegetation dieback, landscape position (proximity to upland and/or creek), and degree of vegetation loss. Tidal range differed between the two marshes as well.

- Early responses showed evidence of a runnel-effect at both marshes. At Little Bay, visual evidence of vegetation change from photographs illustrated revegetation occurring. At Ocean View Farm, water table heights decreased significantly, from chronically above the soil surface to below the soil surface.

- Responses differed between marshes. At Little Bay, water table heights and soil properties related to soil moisture indicated conditions were either the same, or wetter in 2021 (after runnels) than in 2020 (before runnels). This is likely due to less severe initial conditions at Little Bay (shallower water features, higher platform elevation), in combination with precipitation differences. In addition, large differences in precipitation between the years probably masked some runnel effects (2020 dry, 2021 very wet).

- At Ocean View Farm some revegetation was beginning to occur at runnel-sites (based on visual inspection of photographs), but changes appear lower in magnitude than at Little Bay. This is probably because conditions were more degraded at Ocean View Farm than Little Bay initially, as described above (lower platform elevation and deeper water features, as well as greater vegetation dieback and bare ground cover).

- We did not observe evidence that runnels would over-drain marshes, lead to altered decomposition patterns, or platform subsidence.

- We did not observe evidence that runnel installation altered hydroperiods, or decreased sediment deposition on marshes.

- Additional years of sampling will be needed to quantitatively assess vegetation changes, and understand how hydrology, soil processes, sediment dynamics, and geomorphology will change with runnels.

1. Introduction

1.1. Background

Salt marshes are productive coastal wetlands that provide important ecosystem services such as nutrient removal, carbon sequestration, and storm protection for coastal properties. Direct (dredging, draining, filling, tidal flow restriction) (Gedan et al. 2009, Burdick et al. 2020) and indirect (sea level rise) (Kearney and Turner 2016, Mariotti 2016, FitzGerald and Hughes 2019) human activities have contributed to salt marsh loss. A prominent form of salt marsh loss is interior conversion to open water, which occurs when water becomes impounded on the surface of a marsh, stresses vegetation, and leads to plant death. Communities and resource managers are urgently in need of tools to address this problem of expanding shallow water in marshes. Over the past ten years, creating “runnels” has emerged as a tool in New England salt marshes to address marsh loss to interior shallow water (Besterman et al. 2022).

Runnels are shallow channels that were originally developed in Australia to control mosquitoes by draining standing water (Hulsman et al. 1989). Studies have demonstrated runnels are an effective mosquito-management technique with low-environmental impacts in Australia (Knight et al. 2021, and references therein). However, less data is available on runnels as a conservation strategy. In the context of marsh conservation, runnels work by draining shallow water from the marsh surface and restoring tidal hydrology, allowing revegetation to occur (Wigand et al. 2017, Babson et al. 2020, Perry et al. 2021, Besterman et al. 2022). When used in coordination with other management strategies, they may help marshes adapt to rising sea level over longer time horizons (Wigand et al. 2017, Besterman et al. 2022).

Runnels appear to be a promising conservation strategy based on several projects conducted over the past 10 years in the northeast U.S. (Perry et al. 2021, Besterman et al. 2022), especially when used for restoring vegetation and decreasing surface water depths. However, few projects have experimental designs that included monitoring before and after implementation, and of treatment as well as reference sites (a Before-After-Control-Impact, or BACI design). Projects have not typically included experimental replicates, or testing along environmental gradients. Further, ecosystem-scale responses to runnels including soil dynamics, sediment transport, elevation change, and hydrodynamics have only been measured in a few projects (e.g., Perry et al. 2021). Data from most of these projects is not yet available publicly or through publications. As a result, knowledge of runnel efficacy across a range of environmental conditions and marsh types is generally qualitative, and difficult to generalize beyond practitioner experience. With growing interest in using runnels from natural resource managers, quantitative data are needed to support both regulatory approval processes and effective application of the technique.

1.2. Objectives and Approach

Our team initiated an experiment in 2020 to test runnels using best practices identified from team-member experience. Our objectives were 1) to experimentally test the efficacy of runnels using a replicated BACI-design, 2) to test runnel efficacy across a range of characteristics (platform elevation, depth of shallow water area, level of peat degradation, tidal range, wind exposure), and 3) test ecosystem-scale processes in response to runnels that provide insight into how marshes will respond long-term. In this report we present our study design and methods, background site characteristics, and some early responses to runnels (one-year post-implementation). While we are measuring a large suite of variables in this project, we have limited this report to those variables which are likely to have responded within a single year. For a few variables, we tested to see if any change had occurred in one year, and after determining there were no differences, proceeded to present background-only data. These variables are specified below.

2. Methods

2.1. Site Selection

We selected two marsh complexes within Buzzards Bay, Massachusetts. We initially identified marshes where we observed shallow water and bare areas in aerial imagery, and then conducted site visits, field assessments, and meetings with local partners and municipalities. We assessed the characteristics in Table 1 and selected marshes that met the “good candidate” criteria in as many categories as possible. These characteristics were identified as important to runnel project success by experienced project partners, as well as through a workshop on runnels held in 2020 (Besterman et al. 2022). There are other factors groups could consider during site selection (Besterman et al. 2022), but these were the priorities for our project team based on our goals and available resources, and the environment of Buzzards Bay.

Little Bay Marsh in Town of Fairhaven and Ocean View Farm in Town of Dartmouth were selected as study marshes (Figure 1). These two marshes are both protected lands with protected upland space to migrate, have supportive landowners with whom we partnered, and are located in towns where we had municipal, public, partner, and county mosquito control support to initiate and maintain the project. These marshes have both been historically ditched, and have many shallow water areas present that appear to have formed recently.

Little Bay (LB) is a fringing marsh exposed to an open embayment, with high exposure to wind-waves and a tidal range of 1.15 m. Adjacent upland at LB is covered by low-lying forest, including red maple swamp habitat, and we have observed fresh surface water inputs where the high marsh borders the upland. Ocean View Farm (OVF) is a sheltered marsh with adjacent uplands covered by hay fields. OVF sits within a back-barrier salt pond and is connected to Buzzards Bay through a narrow tidal inlet. The tidal range estimated from a tide station outside of the salt pond is smaller than at LB (0.96 m), and the tidal inlet further restricts the tides within the pond. Approximately every five years the inlet migrates, narrows, and closes. Prior to a full closure, the tidal range becomes further restricted, with tidal connectivity eventually cut off completely. A local non-profit and homeowner's group manage the re-opening by dredging a new inlet within weeks of a full closure. A full closure occurred between December 2020 and January 2021, and the inlet was reopened in February 2021. Based on partial tidal data obtained from a water level data logger in the pond (Onset HOBO U20L-04), it appears the tidal range in 2020 was around 0.35 m in the pond. After dredging, the tidal range increased appreciably, but tidal elevations were still slightly dampened relative to tide station data, so the tidal range at OVF was still less than then 0.96 m estimated from outside the pond.

In addition to the tidal range and landscape differences between LB and OVF, these marshes also differ in platform elevation and degree of degradation within shallow water areas. Both marshes exhibited a within-marsh gradient of elevation and condition. Within each marsh we selected sites (shallow water areas) for our study that spanned a range of horizontal size, platform elevation, depth, vegetation cover, and peat degradation. These characteristics ranged from meeting "good candidate" characteristics, to nearly "poor candidate" characteristics (Table 1). We did not select sites that would qualify as poor candidates across all environmental categories, as we wanted to test sites that resource managers could realistically expect a response from runnel adaptation.

Table 1. Marsh characteristics used to select study sites. Based on authors' experience using runnels over the past ten years, and knowledge synthesized from a 2020 workshop on runnels. Marsh characteristics are divided into Environmental Characteristics and Logistics and Community Considerations (practical factors affecting the implementation and sustainability of a runnel project). Characteristics were sorted into "good" and "poor" categories. Many sites exhibited features in-between these end-points.

Marsh Characteristic	Good Candidate	Poor Candidate
Environmental Characteristics		
Shallow water areas	• Features present • Bed of shallow water area is firm, with intact peat • Evidence of recent formation • Evidence of horizontal spread/expansion	• Features not present • Bed of shallow water area is soft and covered with layer (>10 cm) of unconsolidated material • Evidence of older formation (40+ years) • Stable border, no signs of horizontal spread/expansion
Microtopography and water flow	• Embankments, levees, ditch spoils, and/or clogged ditches that create barriers to flow	• No evidence of topographic barriers to flow • Barriers that cannot be fixed with a runnel (e.g., undersized culvert)
Elevation	• Platform around shallow water feature is at or above mean high water • Bed of shallow water area sits 20 cm or less below the platform	• Platform around shallow water feature is close to mean sea level • Bed of shallow water area sits greater than 20 cm below the platform
Adaptation potential	• Adjacent upland has a low topographic slope, no hardened barriers to migration • Marsh complex is large with significant amount of marsh area found at or above mean high water	• Topographic slope or hardened barrier prevent migration • Marsh complex is small, fringing, narrow, and/or mostly sits below mean high water
Logistics and Community Considerations		
Public and municipal interest	• Public health issues due to standing water (mosquito breeding) • Expansion of invasive *Phragmites australis*	• Unsupportive municipality and community

	associated with shallow water areas • Municipality concerned about marsh loss; supportive of restoration/adaptation activities • Marsh provides coastal defense to local community/property	
Landowner interest	• Landowner concern about marsh loss; supportive of restoration/adaptation	• Unsupportive landowner
Adaptation potential	• Marsh and adjacent upland protected from development	• Existing and extensive infrastructure directly adjacent to marsh (e.g., dense housing)
Stewardship potential	• Established partnerships with volunteer-community, municipality, mosquito control agency, landowners • Interest among partners to maintain runnels	• No local partners
Access	• Marsh is easy to access by road and foot • Accessible to partners • Machinery (if needed) for runnel creation or monitoring can access marsh	• Only accessible by boat • Highly restrictive access (difficulty gaining permission for future monitoring/maintenance) • No access for machinery (if machinery needed)

2.2. Experimental Design

We used a replicated-BACI (Before-After-Control-Impact) design to test the effect of runnels on salt marsh hydrology, vegetation, sediment and soil dynamics, and other ecosystem processes. We selected 10 distinct areas of shallow, standing water at both LB and OVF as study sites (20 sites total). As described above, the degree of vegetation loss, elevation loss, and peat degradation within each of these shallow water areas varied. Of the 20 sites, twelve were intensively monitored with greater replication and more variables monitored. This report presents methods and data from the intensive sites only.

Sites within each marsh complex were hydrologically independent, separated by a microtopographic barrier such as a levee, ditch, or creek. At each site we identified an approximate mid-point within the shallow water area (centroid), and then established a monitoring transect that bisected the shallow water area and extended from the high marsh toward the low marsh. Three zones were established for sampling: Zone 1 (0–5m upland and seaward of the centroid), Zone 2 (5–15m upland and seaward of the centroid), and Zone 3 (15–30m upland and seaward of the centroid). We established 1-m^2 monitoring plots along the transect, assigning five plots to Zone 1, four plots to Zone 2, and four plots to Zone 3. In some cases, shallow water areas were located too close to the upland or seaward edge of the platform to fit all of the Zone 2 or Zone 3 plots; fewer monitoring plots were used on those transects. One side of each transect was designated for walking and disturbance (e.g., collection of soil cores), and the other was dedicated to vegetation monitoring and left undisturbed. Monitoring took place in the summer and fall of 2020 before runnels were created at all sites, and during the winter, spring and summer of 2021 after runnels were created. Monitoring frequency differed across variables; details on each variable monitored are presented below.

2.3. Background Variables

2.3.3. Turbidity

Sediment suspended in the water column floods marshes with each high tide, deposits onto the marsh platform and contributes marsh vertical accretion. Degraded marshes undergoing erosion may lose sediment, contributing to suspended sediment concentrations in adjacent waters. Water turbidity provides a measurement of suspended sediment concentration, and can be measured using sensors to understand the marsh sediment balance, i.e., whether a marsh is gaining or losing sediment (Nowacki and Ganju 2019). We deployed water quality sondes (YSI

EXO2) to measure turbidity at LB and OVF. One sonde was deployed on the bed of the open embayment adjacent to each marsh. Sondes were mounted on platforms 15 cm off the bed, and protected with anti-fouling copper tape and wiper blades. We deployed these instruments for four weeks in August 2021 to measure available sediment to Buzzards Bay marshes, and determine the average sediment balance at each marsh.

Sondes measured turbidity in Nephelometric Turbidity Units, or NTUs. Sondes were calibrated using NTU standards prior to deployment in accordance with USGS protocols. We calculated standard summary statistics and percentiles for turbidity throughout the deployment. We also calculated the difference between turbidity during flood and ebb tide, which has been shown to correlate well with whether a marsh is losing or gaining sediment on average. Finally, we examined how patterns of turbidity corresponded with wind events at LB and OVF to assess how sensitive the sediment dynamics at these marshes are to wind. Wind speeds were gathered from a NOAA Buoy at the mouth of Buzzards Bay, buoy number BUZM3.

2.3.1. Elevation

At each site, elevations were measured along transects against benchmarks using a digital laser level (Leica Sprinter 250m) and barcode staff. Along each transect the barcode staff was placed on the vegetation-side of the transect. Measurements were collected at every monitoring plot, as well as every 2 m along the transect and at any visible microtopographic transitions (e.g., depressions or hummocks on marsh platform). The benchmarks were NGS rod style benchmarks installed for this study in 2019 and 2020. The elevations of benchmarks were documented using a GPS system (Juniper Systems Geode) and software (EZSurv) using post-processed kinematic (PPK) survey technology. Vertical elevations were corrected to the North American Vertical

Datum of 1988 (NAVD88). At least two elevation observations were made for each benchmark, and observations were generally repeatable to 3 cm.

Elevation surveys were undertaken during the growing season in 2020 and 2021. Reference transects at LB were established and surveyed prior to initiating this experiment (2019), providing an additional time point of elevation measurements. A few transects were surveyed twice in a season to confirm measurements. After inspecting the data, we determined variation between surveys was within the expected error range of the method, so all data were included in analyses. We did not expect to see a difference in marsh platform elevations from runnels after one year, and visually inspected the data to confirm this expectation. Measurements were within the expected error range, thus we averaged across all surveys to estimate elevations.

Elevation measurements were used to calculate two metrics: platform elevation and depth of shallow water areas. Measurements were collected at slightly different horizontal positions between surveys, although some positions were consistent (e.g., established monitoring plots). To process the data, we first averaged measurements collected at the same horizontal locations across surveys. To create a smoothed profile across measurements collected at different horizontal positions in different surveys, we calculated a rolling average across three or four values for each transect (determined for each transect separately to avoid over-smoothing or over-interpreting unique elevations). We restricted the rolling-average analysis to elevations measured within Zone 1 and Zone 2, as those were most relevant for the geomorphology of the shallow water area.

The platform elevation was interpreted as the median value (Watson et al. 2017) of the smoothed profile. To determine the depth of the shallow water area we needed to compare the elevations within the shallow water area to the surrounding platform, while accounting for a negative slope between the upland and seaward end of the transect. The negative slope was removed from the data by detrending the smoothed profile. The minimum detrended elevation within the shallow water area was compared with the detrended platform elevation (median) to estimate the depth of the shallow water area. This approach provided a reproducible method that avoided over-interpreting any individual elevation measurement (e.g., small holes or hummocks can bias estimates).

2.3.4. Soil Shear Strength

Soil shear strength, or the amount of shear stress a soil can withstand without moving, provides information on the stability of a soil. In these marshes, greater shear strength corresponds with higher root density, more intact and drier peat, while lower shear strength measurements would be found in more saturated, less consolidated soils with lower root density. Shear strength may also increase if platform subsidence occurs within areas of dieback, vegetative and elevation context is important to interpreting shear strength measurements. We used a field inspection vane (Humboldt Field Vane Shear Set) to measure soil shear strength in shallow water areas. Measurements were collected in a well-vegetated high marsh area along each monitoring transect for comparison, as well as in the shallow water area. In the shallow water area shear strength was measured on the walking-side of the transect (but away from walking paths) in a location corresponding with the centroid, and the nearest upland and seaward monitoring plots. While measurements were not taken in monitoring plots, we selected ground patches with similar cover to each plot (similar water depth, cover of vegetation and bare peat). In each of the four monitoring locations, three replicate vertical profiles were tested within a 1-m^2 area. In the high marsh, measurements were taken at 10 cm depth. In the shallow water areas measurements were taken at 5-cm, 15-cm, and 30-cm depths. In total, nine measurements were made at each depth at each site within the shallow water area, and three measurements in the high marsh (fewer in a couple cases due to rocks). These measurements were averaged for analysis. Shear strength was measured in 2020 before runnels were created, and again in 2021 after runnels were created. As no differences between the before and after periods were detected, we present only pre-runnel data from 2020 as background information about the sites.

2.3.2. Sediment Grain Size

Sediment grain size distributions were used in this report to interpret exposure of shallow water areas to flooding tides, and vulnerability to erosion. Across the marsh platform coarser sediments are deposited closer to the platform-water interface, while little coarse material is transported to the interior of the marsh. Thus, we would expect to see coarser sediments near to creeks and the marsh bank. Surface sediments (depth of 2-cm) were collected within the shallow water area to quantify the distribution of sediment grain sizes in 2020 before runnels were created. We calculated the percent of fine sediments (grains < 0.05 mm) in the shallow water areas as an indicator for water and sediment dynamics, and vulnerability to erosion. Sediment samples were processed to remove organic material. Samples were then processed using laser diffraction to estimate the distribution of sizes.

2.4. Response Variables

2.4.1. Water Level

Water levels were monitored using Onset HOBO Water Level Loggers (U20L-04) deployed in PVC wells at each site. Wells were installed on the "walking side" of each transect, at least a meter away from the transect line but within the deepest part of the shallow water area. Perforated PVC pipes were inserted into the marsh platform and loggers were suspended by nylon-coated steel wire from a locking well cap. In 2020, 0.40-m pipes were inserted to a depth of approximately 0.30 m in the marsh. We modified the design in summer 2021 to use 1.20-m pipes installed to approximately 1.0 m-depth. The deeper wells improved vertical stability. In both designs loggers were deployed so that the tip of the logger rested at the base of the well (~0.30 m and ~1.0 m below the soil surface in 2020 and 2021, respectively). All water level measurements were adjusted relative to the soil height at each well to account for small differences in deployment depth, so that a depth of 0.0 m is equal to the soil height.

Loggers recorded pressure every 15-minutes, which was converted into a water depth using a standard conversion procedure. Loggers were deployed from July – October 2020, January or February 2021 – June 4, 2021, and June 8, 2021 – August 2021. At a few sites a deployment was missed due to equipment malfunctions. Across the twelve sites and three deployments we collected 289,000 water level measurements. To understand fluctuations of the water table over time we calculated the daily minimum water level, and used that as a proxy for the water table height.

To analyze changes in water level over time we statistically tested the effect of the runnels using a three-way interaction linear model including: treatment (runnel or reference), time (before or after runnels), and marsh (LB or OVF). This model tested if a change in water level between before and after runnels were created varied between reference and runnel-treatment sites, while also allowing LB and OVF to follow different patterns.

2.4.2 Hydroperiod

Hydroperiod is a measure of the length of time a wetland is inundated with water. For this study, we were interested in the tidal hydroperiod specifically (i.e., discounting the impounded, permanent inundation at some sites). Tidal hydroperiod can influence the amount of marine-sourced sediment deposited on the marsh, with longer hydroperiods providing greater potential for sediment deposition. Most of the shallow water sites we studied were inundated with water either permanently or intermittently and did not drain as the tide receded. We needed a method to differentiate the tidal flooding from the longer-term inundation to understand tidal hydroperiod. To accomplish this, we used a software package developed in the R programming language for detecting high and low tides in water level time series, and human interpretation to adjust parameters and ensure we were not capturing spurious water level fluctuations. After identifying high and low tide moments in the time series, we calculated the length of time between each high and subsequent low, then multiplied that value by two to estimate the hydroperiod for each flooding tide. Because differentiating tidal flooding from background water levels required a detailed and iterative process, we focused on a subset of sites to determine whether hydroperiod changed with runnelling. We selected one reference and one runnel site at both LB and OVF. We tested a model comparing hydroperiod length from before and after runnels were installed, from both reference and runnel sites.

2.4.3. Water Table Dynamics

The water table is impacted by factors other than runnelling, and these variables may modify the efficacy of the runnel. These factors include precipitation, the tidal phase (spring vs. neap tide), location on the platform relative to the upland and creek, and depth of the depression within the shallow water area. For this analysis we focused on daily precipitation measured at local weather stations, and the tidal phase. Tidal phase was interpreted by calculating the maximum daily water level at each site, presumed to be the height of the highest tide occurring each day. We compared precipitation and tidal phase in 2021 with water table heights across sites and treatment groups at both OVF and LB. We compared the relative effects of tidal phase and precipitation on water table heights between sites, and treatment groups.

2.4.4. Visual Ecosystem Changes

At each site we installed photo posts to collect standardized, long-term photographs of sites over time. Photographs were taken before runnels were created, and after runnels were created at multiple time points. In this report we present photos taken during the autumn between late-September and early-November in both years. Photos were always taken at the same angle using the camera in a mobile "smart" phone. As one year is insufficient to quantitatively analyze changes in vegetative cover and community composition, these photographs provide an early indicator of how the sites overall responded to runnels.

2.4.5. Sediment Deposition

Sediment deposition on the marsh platform occurs with incoming tides, and helps marshes to vertically accrete. We measured sediment deposition within shallow water areas to understand the importance of suspended sediment in Buzzards Bay marshes in general, and whether runnels affected the magnitude of sediment deposition in shallow water areas. Deposition was measured using sediment traps constructed with acrylic plates (~100-cm^2) and glass microfiber filters (9-cm diameter). We fixed filters to plates using UV-resistant rubber bands. Plates were then secured to the soil surface using aluminum gutter spikes (20.34 cm length). These sediment traps were deployed on the vegetation-side of the transect, at least 1-m away from our vegetation plots. This location was chosen to avoid any disturbance from walking or sampling. A trap was placed at distances corresponding with each vegetation plot in Zone 1 (n = 5), in an area with similar ground cover to the vegetation plot.

Traps were deployed for ~2 weeks in 2020. However, some filters were lost or damaged when deployed for this long. Filters with significant damage were not included in analyses. Traps were deployed for 2-3 days in 2021 during spring tide cycles to ensure tides would reach the traps, but limit exposure to water and waves to reduce damage. This revised method was a significant improvement, and only a couple of traps were excluded due to damage or erroneous measurements. Traps at OVF suffered more damage, and we also faced some logistical issues with deployments. As a result, our sample size was insufficient for statistical comparisons at OVF, and we proceeded with analyses at LB only. One reference site at LB was also problematic because sediments within the dieback were mineral and loosely packed due to intensive crab burrowing (LBSB, Table 2). A massive quantity of sediment washed onto traps at this site from the surrounding soil, and we had no good way to differentiate this sediment from the surrounding sediments. Thus, we excluded this site from analysis. In total we analyzed data from two deployments in 2020 for 5 sites (n_{site} = 3 – 10, fewer than 10 due to losses and trap damage), and two deployments in 2021 for 5 sites (n_{site} = 10), yielding a total of 88 observations at LB.

After traps were collected in the field, they were stored frozen until analysis. Traps were dried at 60° C until constant weight, then dry weights were compared with initial filter weights to calculate total accumulated sediment. Filters were then ashed in a muffle furnace for 4 hours at 450° C to calculate the quantity of inorganic sediment accumulated on the trap. Inorganic sediment is a more useful measure of sediment deposition because organic deposits typically decompose rapidly, and do not contribute as significantly to vertical accretion. Since the deployment length changed between years, we normalized accumulated sediment mass to the number of tides that flooded shallow water areas by at least 10 cm. Tides were identified and counted using the same approach as the hydroperiod analysis. We tested for an effect of runnels using a linear model with an interaction effect between time period (before and after runnels) and treatment (runnel and reference sites), while accounting for between-site differences.

2.4.6. Soil Moisture Content and Porewater Salinity

Soil moisture refers to the quantity of water contained within a soil matrix. Too much soil water can stress and kill plants, while too little can lead to decomposition of the organic matter in soils, resulting in elevation loss. We measured soil water content to understand the effect of runnels on soil moisture. Porewater salinity can vary in salt marshes depending on the salinity of flooding waters, distance from creeks, inundation frequency, and effects of upland freshwater from both surface runoff and groundwater sources. We measured soil moisture content and porewater salinity using a modified 60-mL syringe to collect soil cores to a depth of 5 cm. Duplicate cores were collected at 6 – 7 locations distributed along the walking-side of the sampling transect in May and October 2020, and in May, August and October in 2021. Duplicate cores were combined in a single centrifuge tube, and frozen until analysis.

To measure porewater salinity, tubes were thawed, centrifuged, and supernatant was extracted from the soil core. Salinity was measured on the supernatant of each core using a laboratory probe. To measure soil moisture content, a 5-g subsample was taken from each soil core, weighed, dried for 24 hours at 60° C, then reweighed. The difference between the wet weight and dry weight was interpreted as the mass of water in the soil. For analyses, core-locations were categorized into three groups: Up (Zone 2 and 3, landward of the shallow water area), Center (Zone 1), and Down (Zone 2 and 3, seaward of the shallow water area). These three groups each contained 2–3 sampling locations.

2.4.7. Redox Potential

The redox potential of a soil is an electrochemical indicator for how easily organic matter can be decomposed. Values are measured in millivolts (mV); positive and higher values indicate a greater potential for organic matter decomposition, while negative and lower values indicate lower potential for decomposition. Soils of wetlands become saturated, which lowers oxygen and other electron acceptor availability in the soil (gases diffuse more slowly into liquid than air). Reduced electron acceptor availability lowers the redox potential, and the rate of decomposition. For this study, we were interested in redox potential as indicator for how microbial decomposition might change in response to runnels. We measured redox potential using a probe (Extech RE300 ExStik ORP Meter) inserted 5 cm into the soil. Measurements were collected adjacent to the 6 – 7 locations where cores were collected (above), once per month between May and October in both 2020 and 2021. Measurements were grouped into the Up, Center, and Down categories as described above.

2.4.8. Decomposition

Decomposition of organic material in soils is an ecosystem process affected by the quantity of organic matter and the redox potential of a soil. We were interested in comparing decomposition rates from before and after runnels were created to determine whether draining surface water would alter decomposition rates (specifically, if rates would increase). We measured decomposition with an established protocol using green and herbal (red) tea bags. This "tea-bag experiment" involves drying and weighing bags, burying them for 80 days, then collecting, drying, and reweighing. The difference in mass lost between the green and red tea bags is used to calculate a decomposition constant (K). Higher K-values indicate faster decomposition, while lower K-values indicate slower decomposition rates. We buried bags at three locations corresponding with the Up, Center, and Down zones along the walking-side of the transect at a depth of 5 cm. In 2020, 5 green and 5 red tea bags were buried in each location.

In 2021, 10 green and 10 red tea bags were buried in each location. Bags were buried during the growing season in both years.

3. Results and Discussion

3.1. Soil Structure and Geomorphology

Platform elevations varied within and between marshes. OVF sites were lower in elevation (0.272 – 0.472 m NAVD88) than LB (0.543 – 0.833 m NAVD88) (Table 2). Similarly, depression-depths within shallow water areas varied within marshes (Table 2, Fig. 2), with OVF sites ranging between 0.056 m and 0.181 m, and LB sites ranging between 0.011 m and 0.148 m. LBSB differed from other sites in that the unvegetated area did not coincide with a depression in elevation, so we excluded this site from depth calculations. Soil shear strength also varied between and within marshes, indicating a range of peat degradation. Lower soil shear strength within the dieback areas indicates peat has undergone greater decomposition and soil is less consolidated. OVF had generally lower soil shear strength in the surface 5 cm of soil than LB, indicating greater peat degradation (Fig. 3). Across both marshes, sites with deeper depressions generally also exhibited lower soil shear strength throughout the profile (Figs. 2 & 3), indicating these sites were likely more degraded to begin with than sites with shallower depths and greater soil shear strength.

Table 2. Background variables on site soil structure and geomorphology. Sites are organized by treatment and marsh, and averages with standard deviation (SD) presented. Platform elevation as meters above NAVD88, the depth of the depression within the shallow water area as meters below the platform, and percent of fine sediments within surface 2-cm of shallow water areas are displayed.

Site	Platform Elevation (m NAVD88)	Depth of Dieback (m)	Percent fines
Ocean View Farm Reference Sites			
OVFD/Reference 1	0.399 m	0.058 m	71.4%
OVFF/Reference 2	0.382 m	0.067 m	72.2%
OVFG/Reference 3	0.272 m	0.181 m	79.6%
Average (SD)	**0.351 (0.07) m**	**0.102 (0.07) m**	**74.4 (4.5) %**
Ocean View Farm Runnel Sites			
OVFA/Runnel 1	0.472 m	0.056 m	37.9%
OVFE/Runnel 2	0.393 m	0.098 m	95.5%
OVFH/Runnel 3	0.290 m	0.142 m	75.1%
Average (SD)	**0.385 (0.09) m**	**0.099 (0.04) m**	**69.5 (29.2) %**
Little Bay Reference Sites			
LBNA/Reference 4	0.734 m	0.148 m	60.7%
LBSC/Reference 5	0.543 m	0.021 m	60.7%
LBSB/Reference 6	0.833 m	NA	39.7%
Average (SD)	**0.703 (0.15) m**	**0.085 m**	**53.7 (12.1) %**
Little Bay Runnel Sites			
LBND/Runnel 4	0.706 m	0.025 m	39.8%
LBNF/Runnel 5	0.682 m	0.011 m	56.2%
LBSM/Runnel 6	0.644 m	0.071 m	74.2%
Average (SD)	**0.677 (0.03) m**	**0.036 (0.03) m**	**56.7 (17.2)%**

3.2. Visual Changes in Vegetation

Initially OVF had greater bare ground cover than LB within shallow water areas (Figs. 4–26, even numbers). Areas were larger, with less total vegetation and more contiguous loss as opposed to the patchier vegetation-loss at LB. At both LB and OVF, 1-year after photographs show some positive revegetation at runnel sites (Figs. 4 – 8, 16 – 20 even numbers). The amount of change varied across sites, but consistent positive responses were observed. LB generally appears to have greater revegetation at runnels sites than OVF. Reference sites showed little or no change in vegetative cover (Figs. 10 – 14, 22 – 26 even numbers).

3.3. Water Levels

Water table heights and tidal amplitudes experienced at each site differed within and between marshes (Figs. 5–27, odd numbers). As discussed above, tidal ranges are known to differ between LB and OVF due to different geomorphic settings, and between years at OVF due to the tidal inlet dredging. Tidal amplitudes increased in 2021 relative to 2020 across OVF sites as a result of re-opening the inlet; however, differences were still apparent between runnel and reference sites. With deeper shallow water areas and lower platform elevation, we observed higher water table heights at OVF than LB in both years, and longer periods of continuous inundation above the soil surface (Figs. 5–27, odd numbers).

3.3.1. Effects of Tides and Precipitation on Water Levels

To understand how differences in tidal phase and precipitation affected water table heights we tested various linear models. There were no interactions between treatment (runnel vs. reference sites) and either tidal phase or precipitation at marshes. At OVF, both precipitation and tidal phase significantly affected water levels, and effects differed across sites ($p_{model} < 0.0001$, $p_{precipXsite} = 0.02$, $p_{tidalXsite} <0.0001$). Precipitation had the largest impact at OVFA/Runnel 1, where 10mm of rainfall led to an increase of 0.9mm in the shallow water area (9% increase). This effect was similar in magnitude at OVFD/Reference 1, and OVFE/Runnel 2, and larger than effects at OVFF/Reference 2, OVFG/Reference 3, and OVFH/Reference 3. Tidal phase predictably had a much larger effect on water levels at OVF than precipitation. The effect was largest at OVFA, with a 10mm increase in tidal height leading to a 1.8mm increase in water table heights (18% increase). Patterns were similar as with precipitation, with OVFD and OVFE appearing similar, and OVFF, OVFG, and OVFH showing dampened effects of tidal phase on water levels. Water levels at OVFF, OVFG and OVFH were more consistent through time than

OVFA, OVFD, and OVFE, regardless of tides or precipitation. This suggests water table heights at these three locations are controlled by landscape position, platform elevation, depression depth, and microtopographic features that can block hydrologic flow. Note that we used 2021 data for this analysis, after runnels were created. Since OVFE and OVFF are similar in platform elevation, landscape position, and depression depth, the difference between these two sites likely indicates the runnel successfully breached a microtopographic barrier, allowing flooding to resemble tidal patterns more closely. Meanwhile, water levels at OVFF remain constant, indicating continued impoundment.

At LB we also observed effects of precipitation and tidal phase, but precipitation effects did not differ across sites ($p_{model} < 0.0001$, $p_{precip} = 0.03$, $p_{tidalXsite} < 0.0001$). Across the marsh precipitation had a lesser effect than at OVF, with a 10mm rainfall event corresponding with a 0.6mm rise in water table heights (6% increase). Tidal phase also affected water table heights, with a larger effect of tidal flooding at LB than at OVF on water levels, and differing effects across sites. The largest effect of tidal height was seen at LBSB/Reference 5, where an increase in tidal height of 10mm led to a 3.4mm increase in the water table (34% increase). Tidal effects were mostly similar across sites, except for LBSM/Runnel 6 where tidal effects were dampened to a 1.1mm increase in water table height per 10mm higher tide (11% increase in water level).

3.3.2. Water Table Heights and Soil Water

We use a linear mixed-effects model to test for an effect of runnels on water table heights. We included a three-way interaction between time (before or after runnels installed), treatment (runnel or reference site), and marsh (LB vs. OVF), while accounting for differences between individual sites. We found that runnels significantly reduced water levels at OVF ($p_{treatXtimeXOVF} < 0.0001$) relative to reference sites, but not at LB ($p_{treatXtimeXLB} = 0.68$) (Fig. 28).

There were, however, visible differences in the length and frequency of inundation events at LB runnel sites (Figs. 4–27, LB sites). A few factors may have contributed to the absence of a statistical effect. First, water levels were lower across LB, and especially at reference sites, prior to runnel installation. Thus, any runnel effect on water table heights would have been smaller in magnitude and more difficult to detect. Second, precipitation was much lower in 2020 than in 2021, so a small effect of lowered water table heights from runnels may have been countered by overall wetter conditions. Precipitation may not have mediated an overall runnel effect at OVF because precipitation had little significance on water levels for 3 of the 6 sites.

The hypothesis that precipitation reduced runnel effects on water table heights at LB is supported by soil moisture content and porewater salinity data (Figs. 29 & 30). Though we did not perform statistical tests on these data, soil moisture content appeared higher in 2021 than 2020 across all experimental zones and at all sites (Fig. 29). Porewater salinity also appeared lower across all sites and experimental zones (Fig. 30). Differences in soil moisture and porewater salinity between the two years were present at both OVF and LB, though appeared larger at LB. Thus, it seems likely that higher precipitation at LB increased soil moisture, decreased soil salinity, and masked any effects of the runnel on the water table.

Porewater salinity, independent from precipitation effects, could either increase or decrease with runnelling. Porewater may freshen with runnels if runnels prevent tidal water from becoming impounded on the marsh where it becomes hypersaline over time with evaporation. High marsh areas are particularly vulnerable to forming hypersaline conditions since they are flushed less frequently than more seaward locations. Porewater may become more saline after runnels if high freshwater inputs from upland sources were contributing initially to water impoundment, and runnels restore tidal connections to shallow water areas. At both LB and OVF in 2020, average porewater salinities were slightly higher (around 35–45 ppt) than those of flooding waters (~33 ppt). Porewater salinity decreases occurred everywhere in 2021, though effects were especially pronounced in the 'Up' zone adjacent to upland areas. That effects appeared more pronounced in the Up zone further supports an effect of precipitation and freshwater input on differences in marsh hydrology between 2020 and 2021, and the hypothesis precipitation differences masked a runnel effect, if any occurred. Determining how runnels will ultimately affect porewater salinity and overall marsh hydrology at these marshes will require further years of sampling since the 2020 – 2021 period was strongly affected by precipitation patterns.

3.3.3. Hydroperiod

We analyzed hydroperiods at two reference and two runnel sites at OVF and LB (OVFE/Runnel 2, OVFF/Reference 2, LBNF/Runnel 4 LBNA/Reference 4) to determine whether runnels impacted hydroperiod. The runnel at LBNF did not impact hydroperiod relative to LBNA ($p_{treatXtimeXLB} = 0.9$). OVF did show a decrease in hydroperiod at both OVFE and OVFF, with a greater decrease at the reference (OVFF). The decrease in hydroperiod is most likely driven by the dredging of a new tidal inlet at OVF. The inlet had already narrowed substantially by summer 2020, before closing fully over the winter. Thus, the dredging should have increased the tidal flushing in the summer of 2021 relative to 2020. Given the lack of an effect at LB, and the decrease in hydroperiod at both runnel and reference sites at OVF, we conclude that runnels at the scale constructed for this project do not alter hydroperiods.

3.4. Soil Chemistry and Processes

Redox potential and decomposition (Figs. 32 & 33) followed patterns that would be expected based on the soil moisture data and pattern of increased precipitation. We did not perform statistical tests on these data, but observed an apparent decrease in redox levels between 2020 and 2021 in all three experimental zones at runnel and references sites at both marshes (Fig. 32). Wetter conditions correspond with lower redox potential. Rainfall likely explains the difference between these two years. Similarly, wetter conditions and lower redox potential can result in decreased rates of decomposition, which we observed in 2021 relative to 2020 at all sites (Fig. 33). Long-term measurements over a few years will be needed to determine whether runnels have a net impact on soil chemistry and processes.

3.5. Sediment Dynamics

3.5.1. Turbidity

Turbidity, an indicator for suspended sediment, was generally low at both LB and OVF (Fig. 34). We used a formula developed elsewhere to convert NTUs to suspended sediment concentration for comparisons with other systems (Nowacki and Ganju 2019). While a system-specific conversion is needed, these values provide a first-order approximation of suspended sediment in the system. At LB, sondes measured an average of 6.9 mg L^{-1} of sediment, (SD = 7.9 mg L^{-1}). Values were similar, but slightly lower at OVF, with an average of 6.3 mg L^{-1} (SD = 3.4 mg L^{-1}) measured. These values are near the minimum suspended sediment concentrations (4 mg L^{-1}) measured across 13 different marsh complexes in North America. This follows expectation, as Buzzards Bay is a low sediment system. The implication is very little inorganic sediment is available to subsidize elevation in marshes. During significant wind events, sediments were resuspended at LB and temporarily led to higher turbidity in the water column (Fig. 34). Resuspension events have the potential to transport sediment onto marsh platforms. With OVF protected behind a barrier island and narrow inlet, winds were unable to generate enough wave energy to resuspend sediments, and no spike in suspended sediment was observed (Fig. 34).

The flood-ebb turbidity differential appeared to be near neutral at both OVF and LB. OVF slightly favored sediment import, with a positive differential of 0.4. LB very slightly favored sediment export, which can indicate erosion occurring (-0.1). However, this value is near enough to zero to consider the sediment balance neutral between import and export. For comparison, the highest import observed across marsh sites measured in Nowacki and Ganju was +5, while the largest sediment export differential measured was -17 (Nowacki and Ganju 2019). While very little sediment is available to LB and OVF in general, neither marsh is losing sediment, and wind-driven resuspension can lead to sediment import events at LB.

3.5.2. Sediment Deposition

Across all sites and both years the average sediment deposition was 31.8 mg per tide of 10 cm or more (mg T^{-1} hereafter), with SD of 26.3 mg T^{-1} (excluding LBSB/Reference 5). Due to changes in methodology between 2020 and 2021, a reduced sample size we suggest sediment deposition results be interpreted as tentative until more sampling can be conducted. Statistical tests indicated that inorganic sediment deposition increased within shallow water areas at LB runnel sites relative to reference sites after runnels were installed ($p_{treatXtime} = 0.01$, $F = 6.47$, df = 83) (Fig. 35). Increases in sediment deposition were apparent from 2020 to 2021 at both reference and runnel sites (due to either methodological or environmental differences). However, runnel site deposition increased by 31.2 mg T^{-1} from 2020 to 2021, while reference sites increased by 3.5 mg T^{-1} (difference of 27.7 mg T^{-1}).

To ensure these increases were not simply caused by disturbance to soils from runnel creation, and subsequent sediment mobility, we compared the percent organic content of deposited sediments. If the higher deposition was only caused by mobilized surface sediments within the dieback areas, we would expect the percent organic to increase from 2020 to 2021, and the percent organic sediment to resemble the percent organic content of the surrounding soils. First, average organic content of sediment deposited on filters (18% in 2021) was lower than organic content of soil cores collected from sites (> 40% for most samples, data not presented here). Second, the percent organic decreased at runnel sites relative to reference sites between 2020 and 2021 ($p = 0.07$), although high variability suggests this result requires greater scrutiny. Whether or not the relative increase in inorganic material deposited is supported with subsequent sampling, it does not appear that the increased deposition could be driven by eroding or mobilized marsh surface sediments.

Tentative evidence suggests inorganic sediment deposition may increase within shallow water areas as a result of runnels. If further data collection supports this finding, it may be caused by runnels increasing connectivity between the interior marsh platform and open embayment. With microtides and low sediment in Buzzards Bay, very little sediment would make it into the interior platform areas where the shallow water dieback areas occur. The runnels may provide a more direct conduit for sediment to be transported into the marsh interior. However, more robust sampling and testing at both OVF and LB is needed to confirm these results.

4. Conclusions

Early responses of two salt marsh complexes in the Buzzard Bay Estuary show promising results for runnels as a climate adaptation technique. Visual evidence of vegetation changes, and water table dynamics show that tidal hydrology and marsh vegetation are beginning to show signs of restoration, with variation across sites and marshes. Variation is likely caused by large differences in background conditions at the two marshes, and among sites, including platform elevation, depth of dieback area, landscape position, peat degradation and tidal range. In addition, temporal factors such as precipitation significantly affected water dynamics at the two marshes, and resulting soil chemistry and processes. The higher rainfall occurring in 2021 over 2020 confounded our ability to detect an effect of runnels on multiple marsh properties and to determine the “footprint” of a runnel across the marsh platform (between experimental zones). We found very tentative evidence of increased inorganic sediment deposition within dieback areas after runnels were created; however, this result needs further study to confirm. Additional years of data collection are needed to fully understand the responses of these marshes to runnels, and how a number of spatial and temporal factors interact with the runnelling approach to affect marsh hydrology, geomorphology, and vegetation.

Figure 1. Site maps of LB and OVF. Green lines represent sampling transects at reference sites, and purple lines indicate sampling transects at runnel sites. Light blue lines illustrate the runnels created in October and November 2020. The stars indicate the intensive sites. All reporting in this document focuses on the intensive sites only.

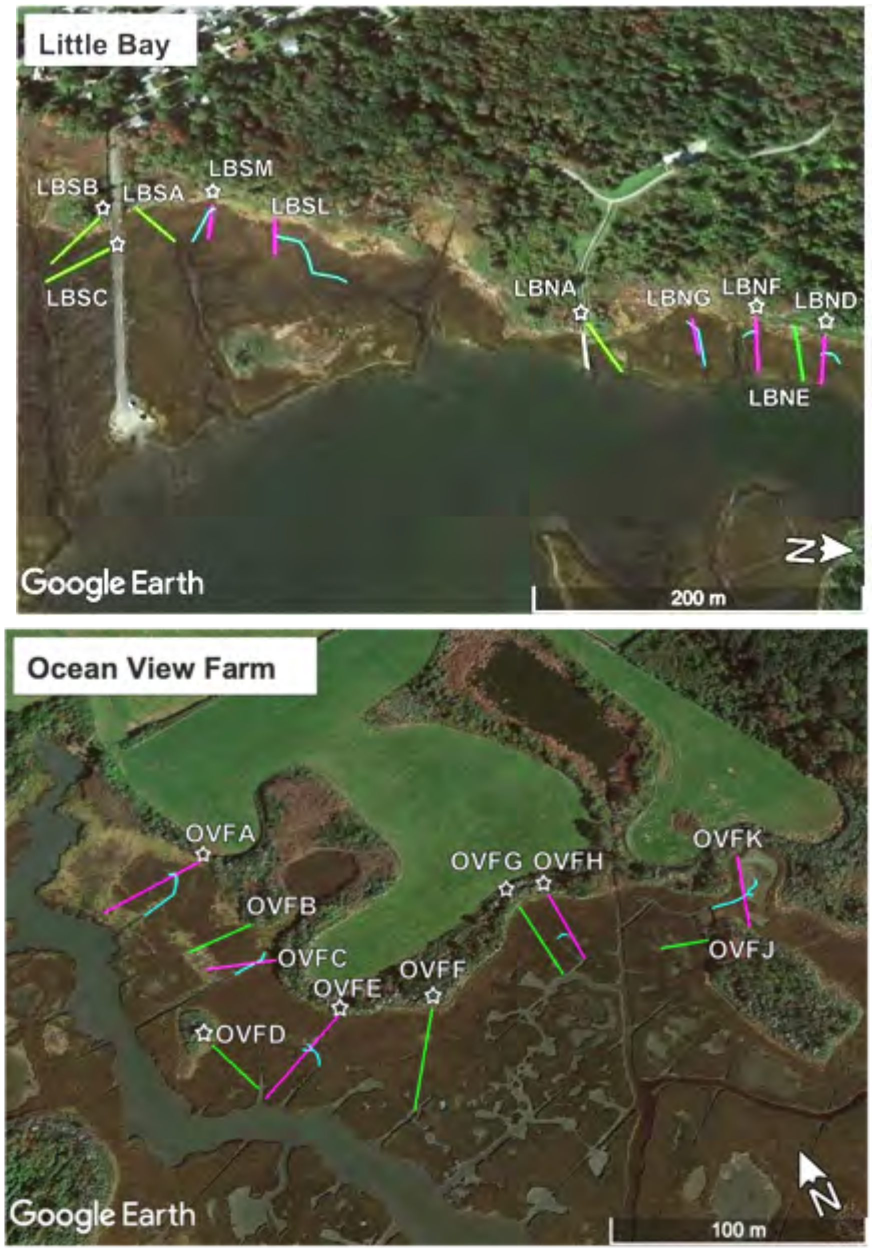
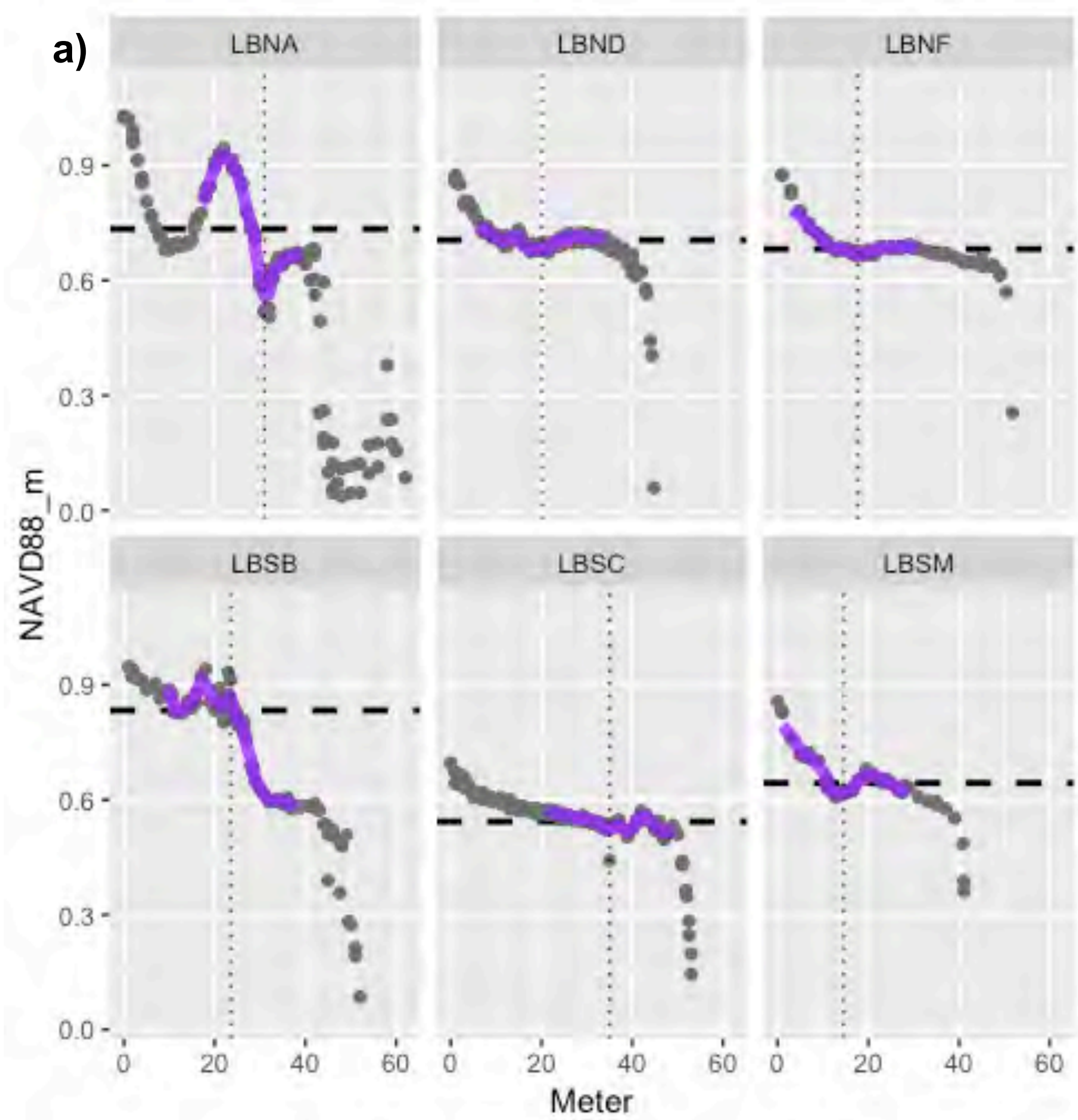
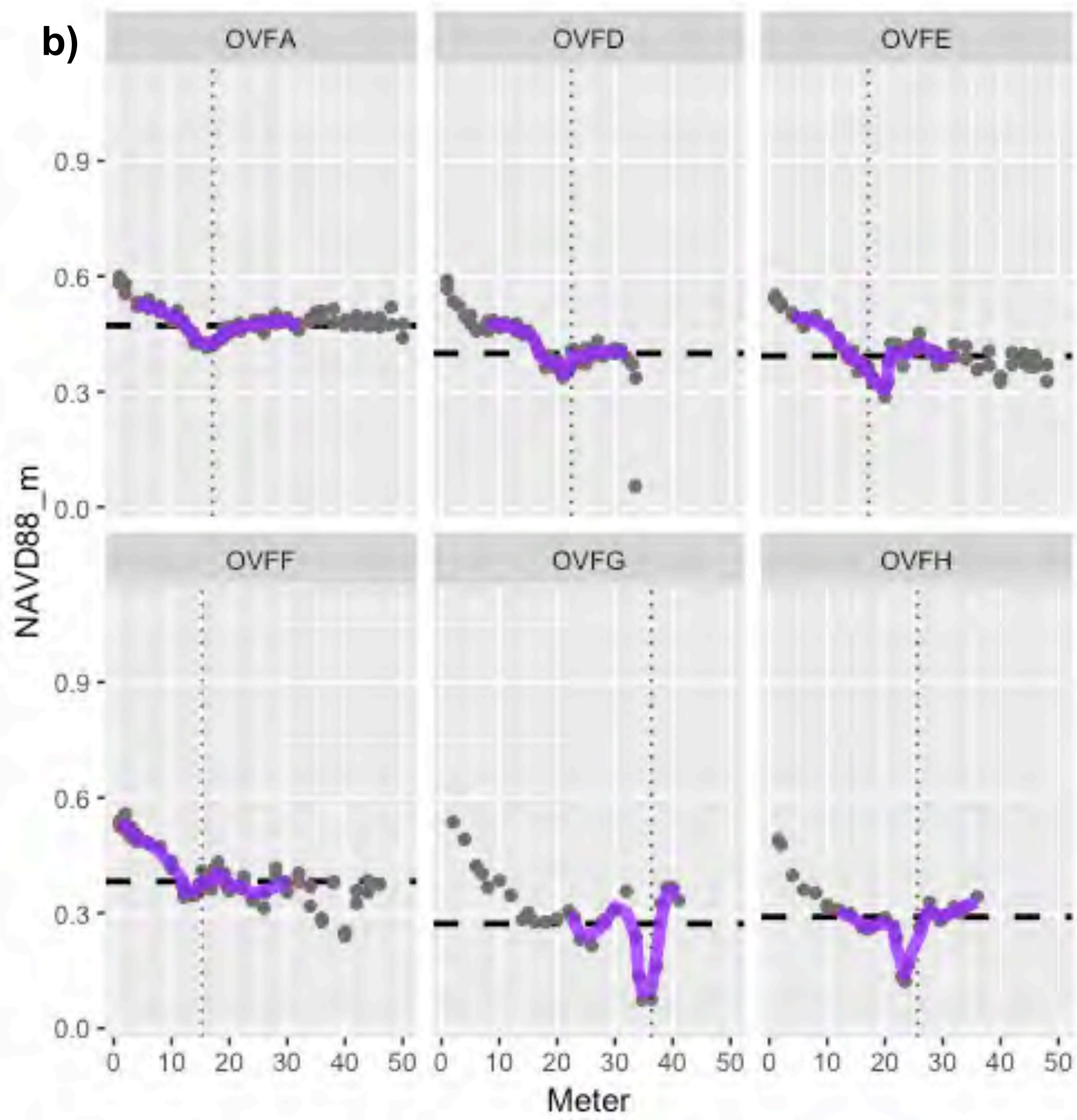


Figure 2. Elevation profiles at **a)** LB, and **b)** OVF. See Table 2 for experimental treatment units of each site (4-letter codes). Profiles extend from the high marsh-upland border (0 on x-axis) seaward toward open water. Gray dots are averaged elevation measurements collected along sampling transects, and the smoothed depth profile of the shallow water area is illustrated with the purple line. Long-dashed horizontal line shows the median elevation for each transect, interpreted as the platform elevation. The dotted vertical line illustrates the centroid of the shallow water area along transect.



b)



OVFA
OVFD
OVFE
OVFF
OVFG
OVFH
NAVD88_m
0.0
0.3
0.6
0.9
0
10
20
30
40
50
Meter

Figure 3. Soil shear strength vertical profiles (mean +/- SE) collected at **a)** LB and **b)** OVF. See Table 2 for experimental treatment units of each site (4-letter codes). Black line illustrates the shear strength profile at 5, 15, and 30 cm depths within the shallow water area (or "dieback"), and the green point shows the shear strength in a well vegetated patch of marsh from the upland edge of the transect (usually high marsh vegetation) collected at 10 cm depth.

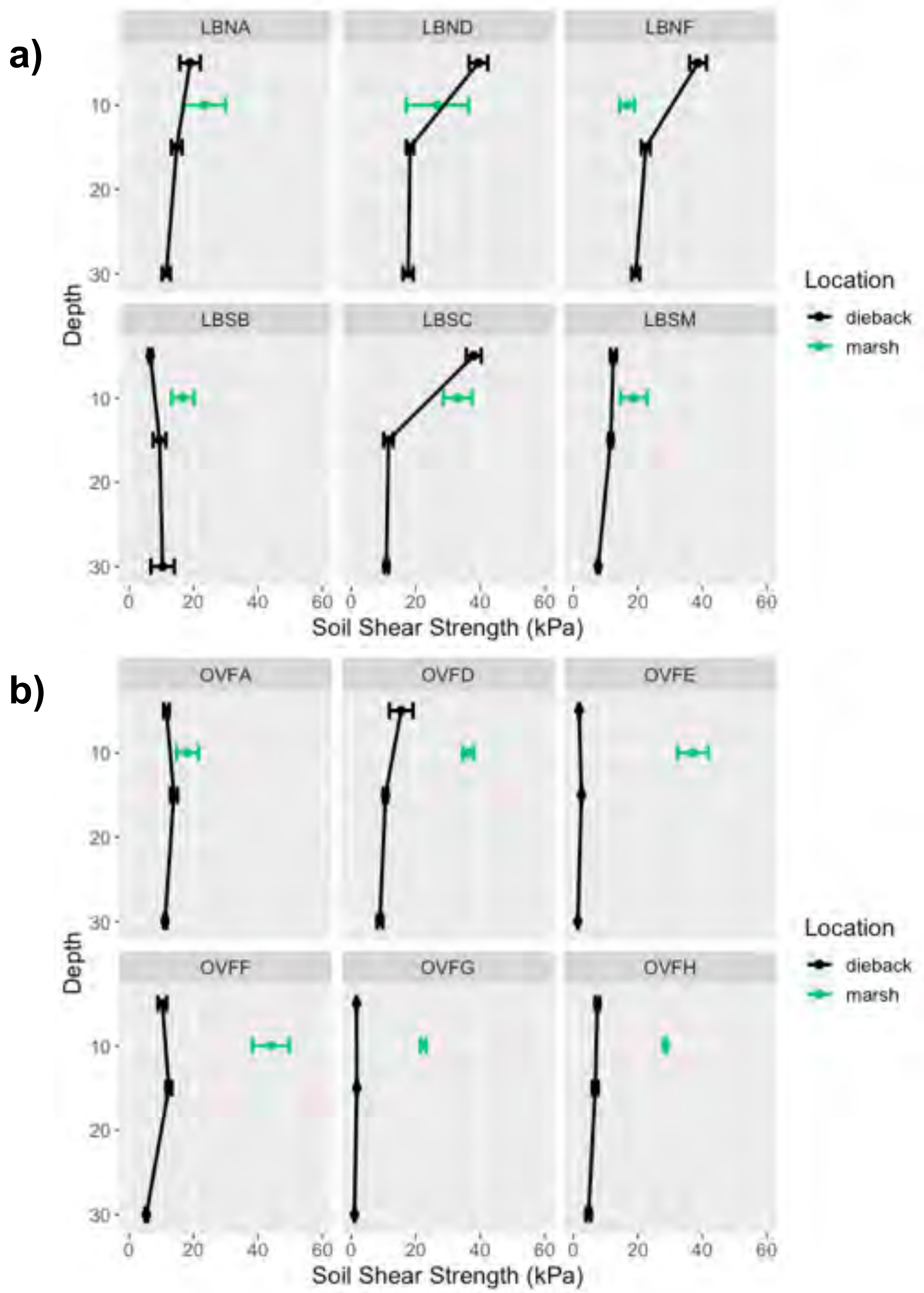


Figure 4. Before and after photograph at OVFA, Runnel Site 1.

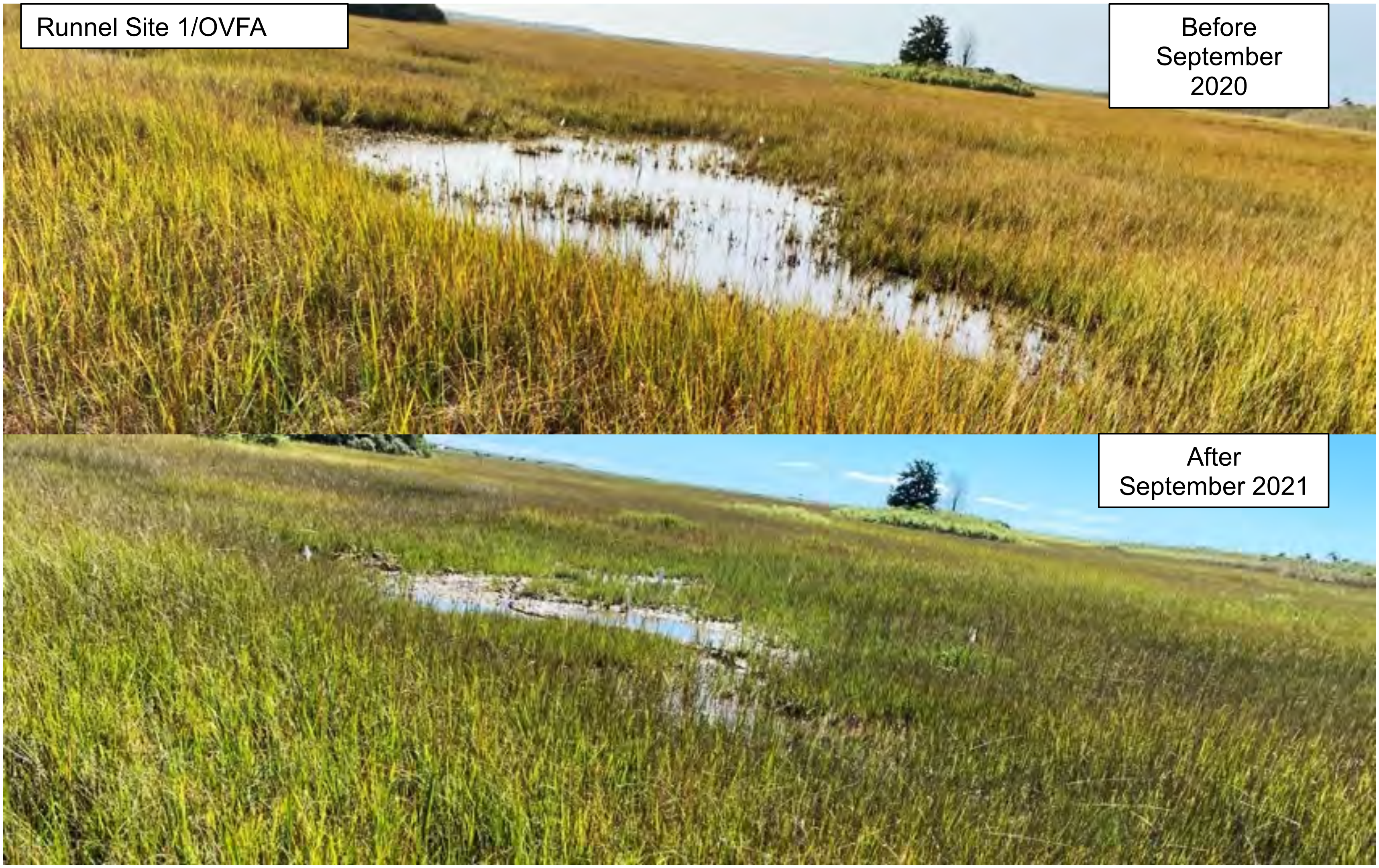


Figure 5. Local precipitation, and water levels from before runnels were installed (2020) and after (2021) at OVFA, Runnel Site. 1. Light blue line shows 15-minute data, and the dark blue line shows the water table height.

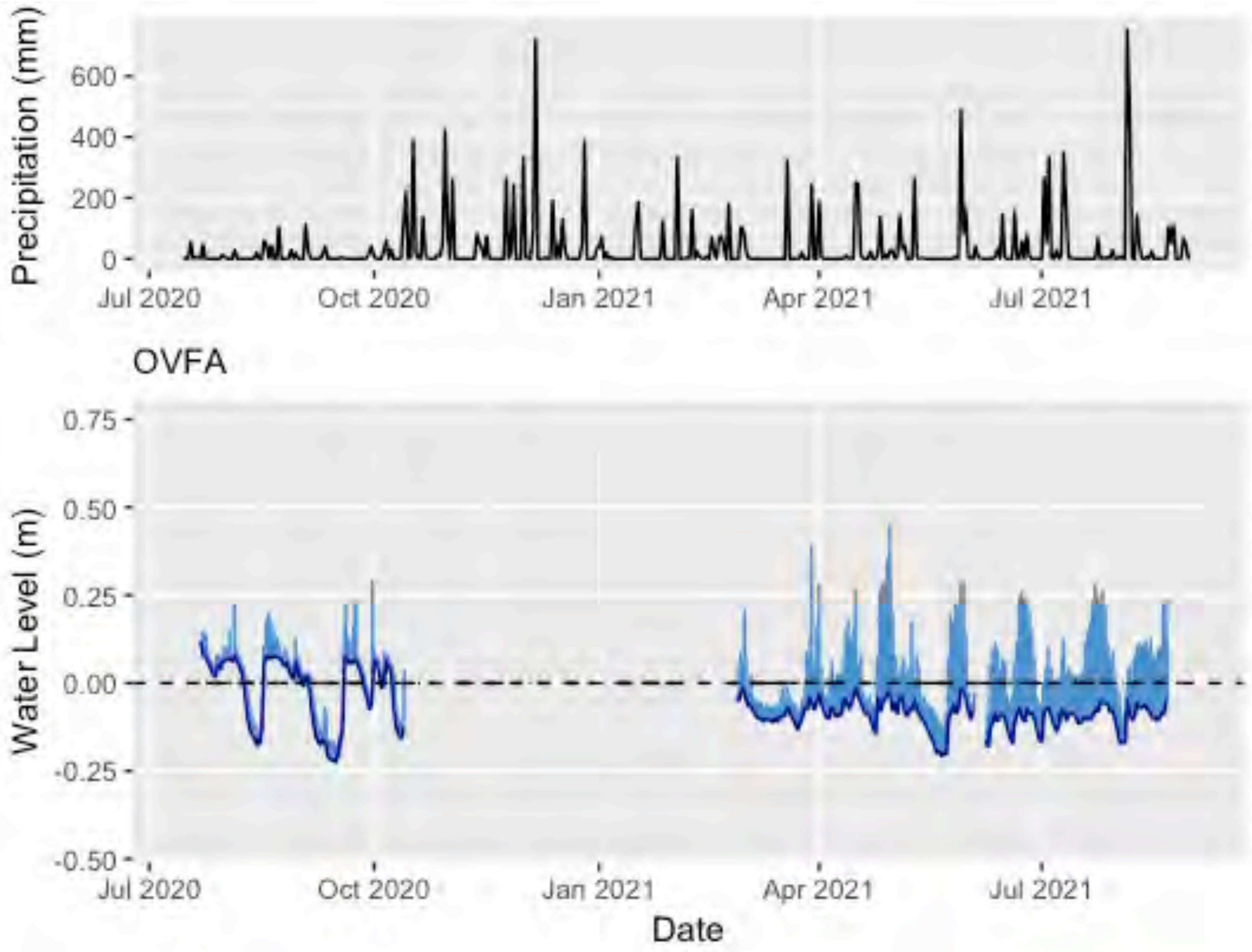


Figure 6. Before and after photograph at OVFE, Runnel Site 2.

Figure 7. Local precipitation, and water levels from before runnels were installed (2020) and after (2021) at OVFE, Runnel Site. 1. Light blue line shows 15-minute data, and the dark blue line shows the water table height.

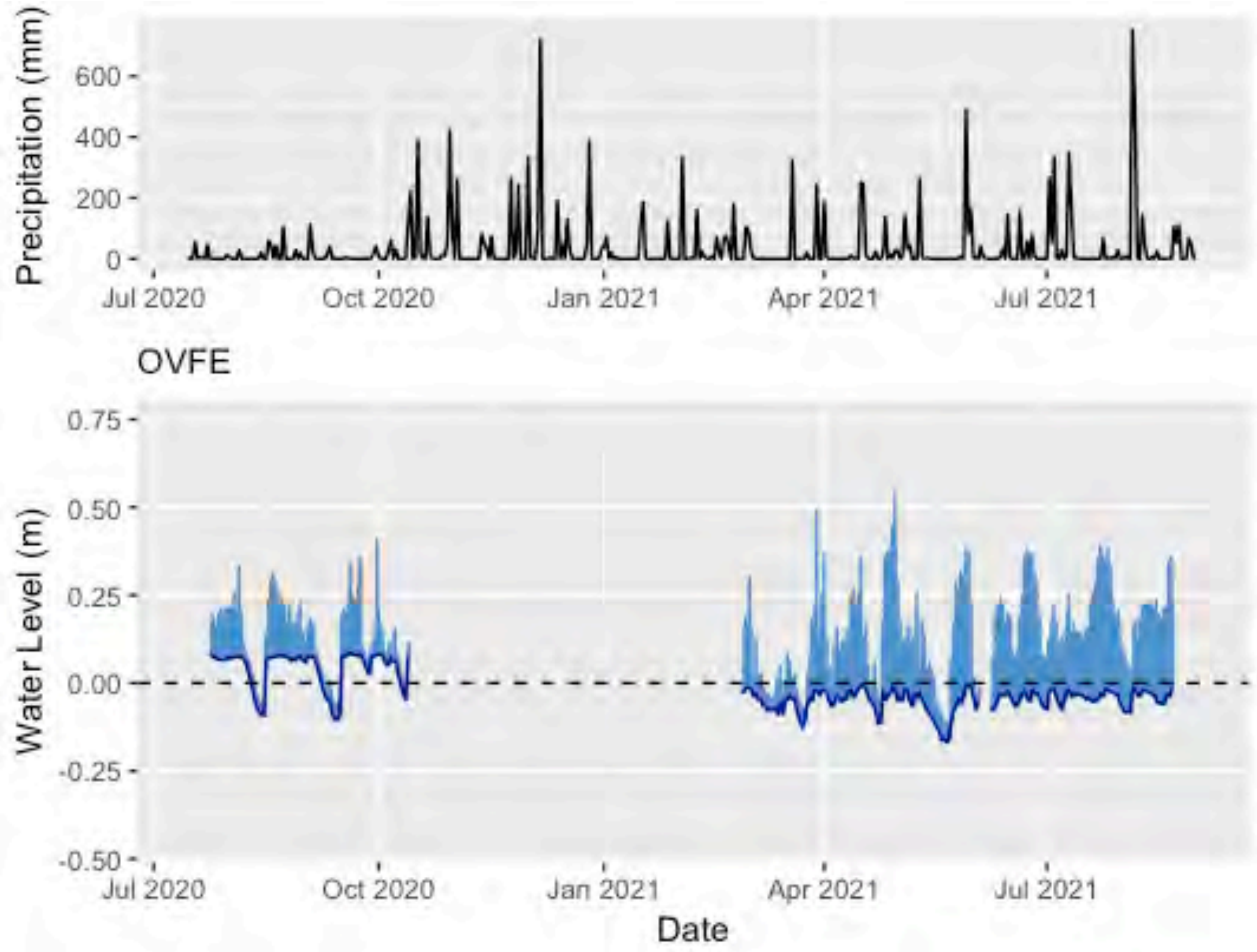


Figure 8. Before and after photograph at OVFH, Runnel Site 3.

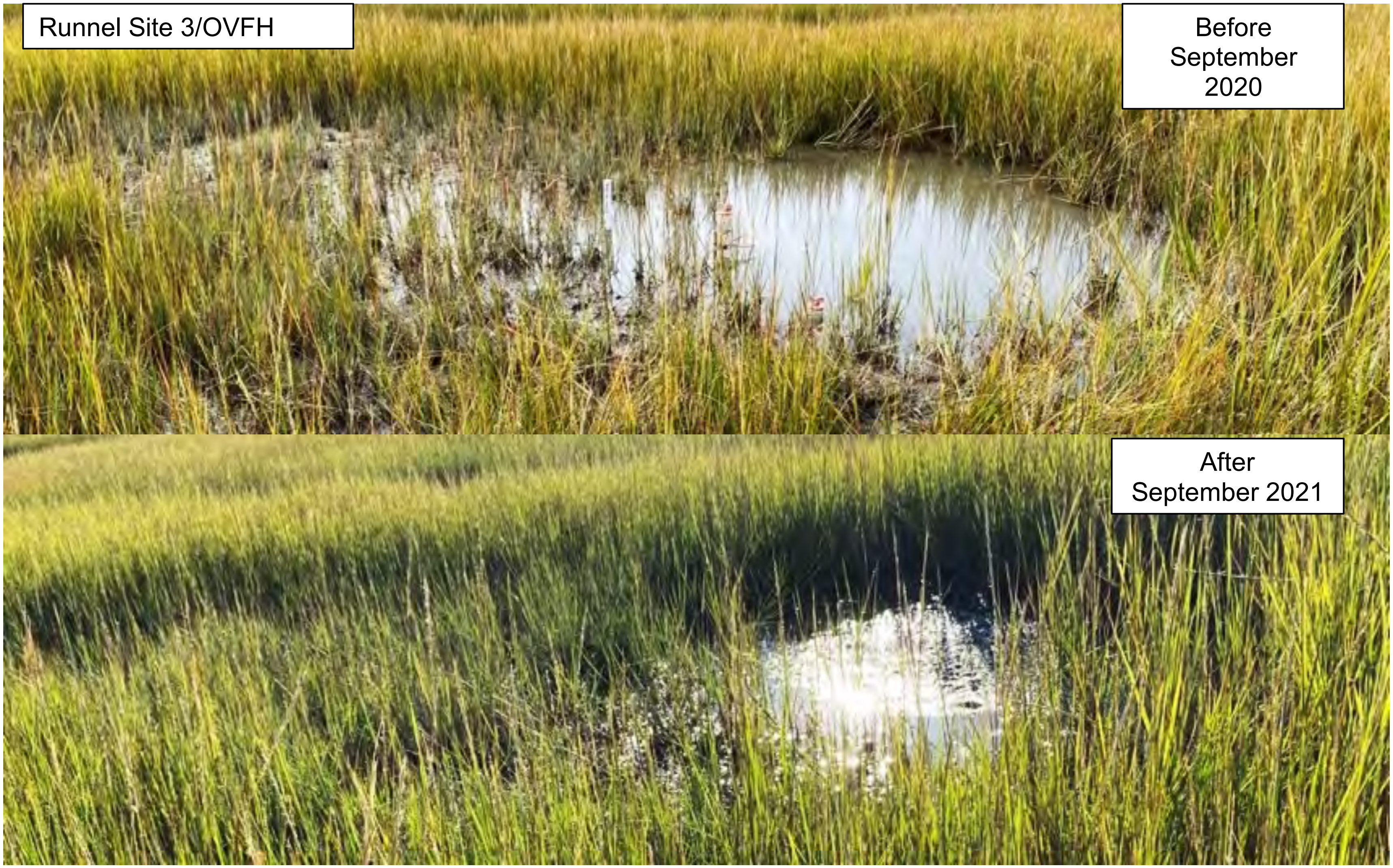


Figure 9. Local precipitation, and water levels from before runnels were installed (2020) and after (2021) at OVFH, Runnel Site. 3. Light blue line shows 15-minute data, and the dark blue line shows the water table height.

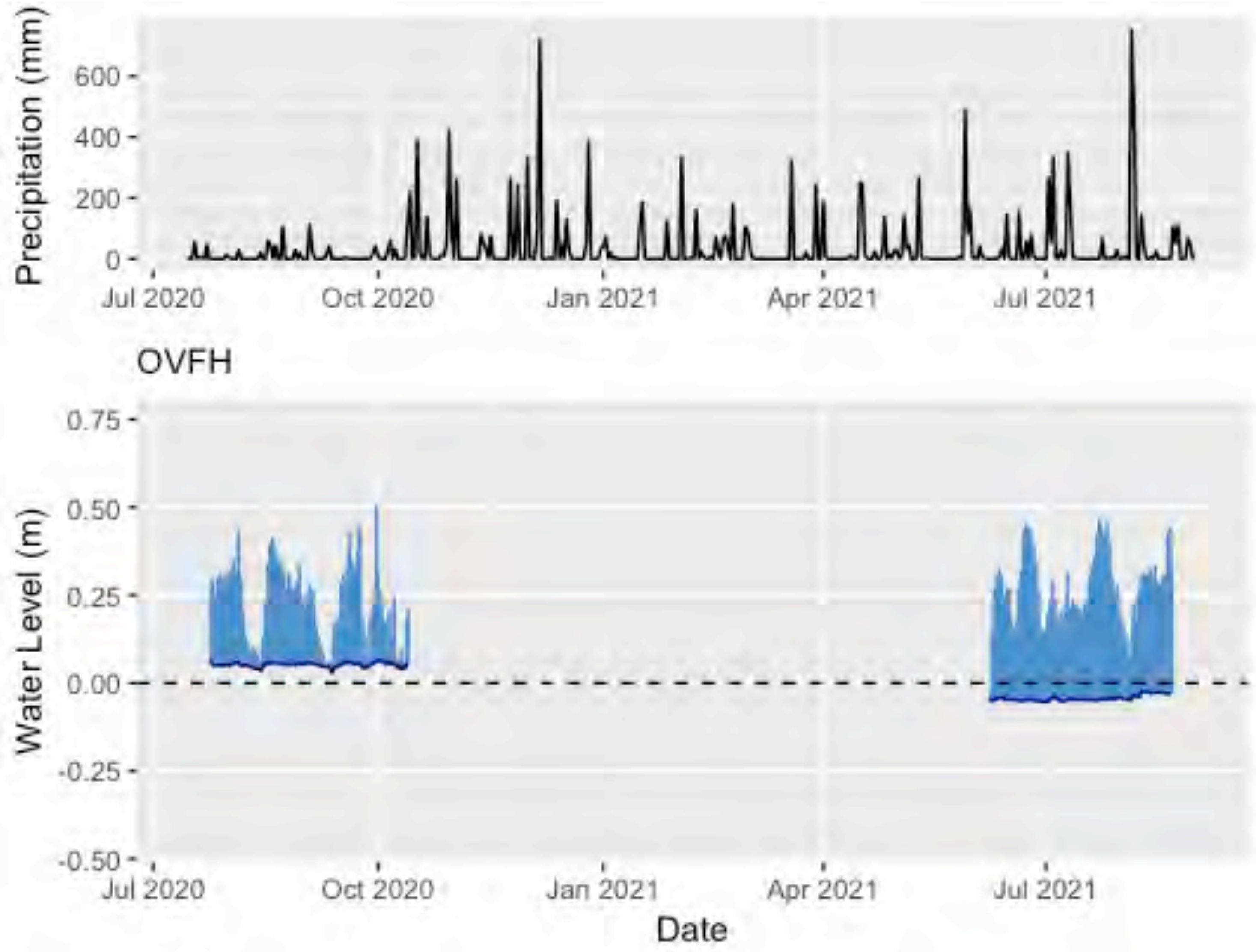


Figure 10. Before and after photograph at OVFD, Reference Site 1.

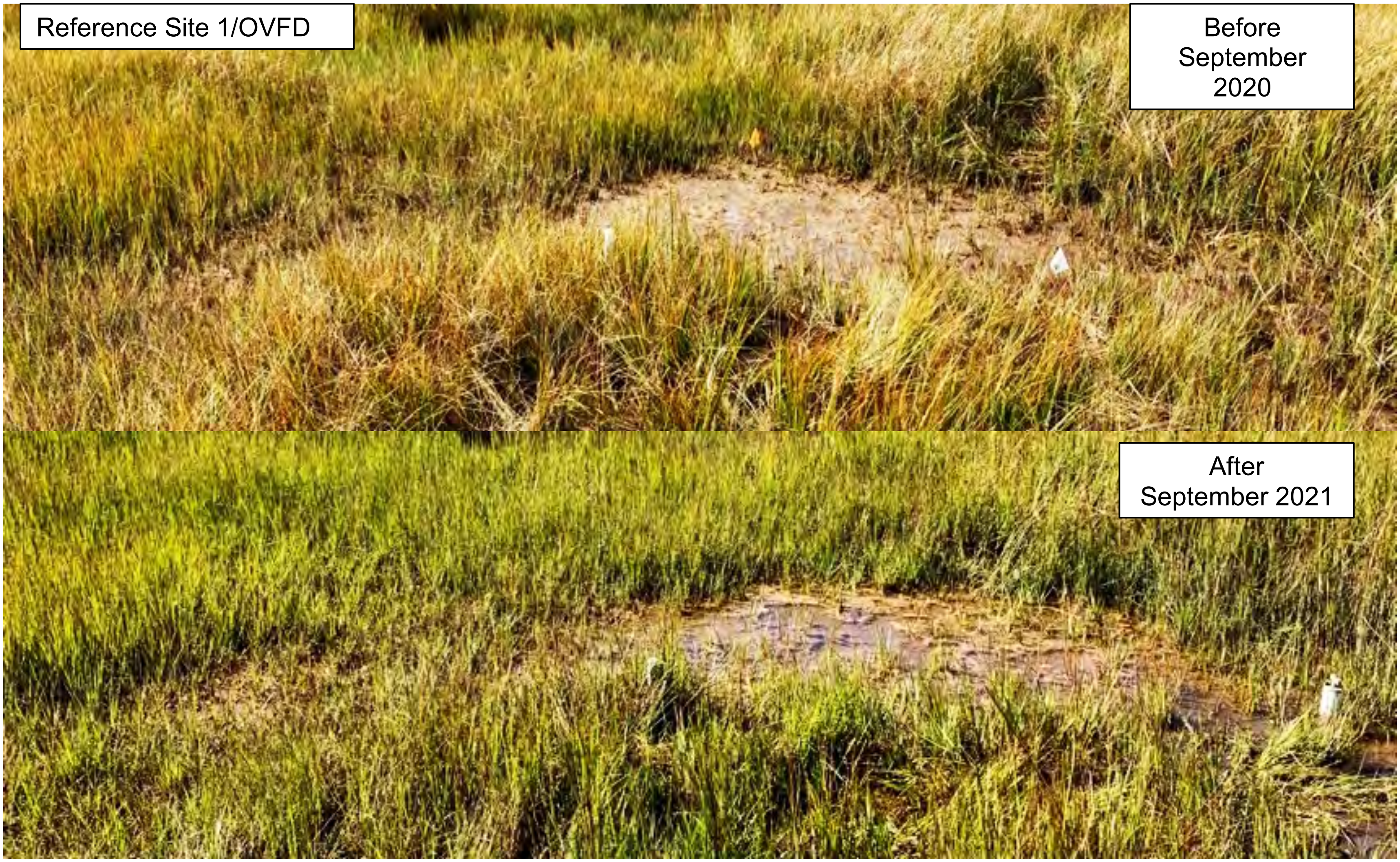


Figure 11. Local precipitation, and water levels from before runnels were installed (2020) and after (2021) at OVFD, Reference Site 1. Light blue line shows 15-minute data, and the dark blue line shows the water table height.

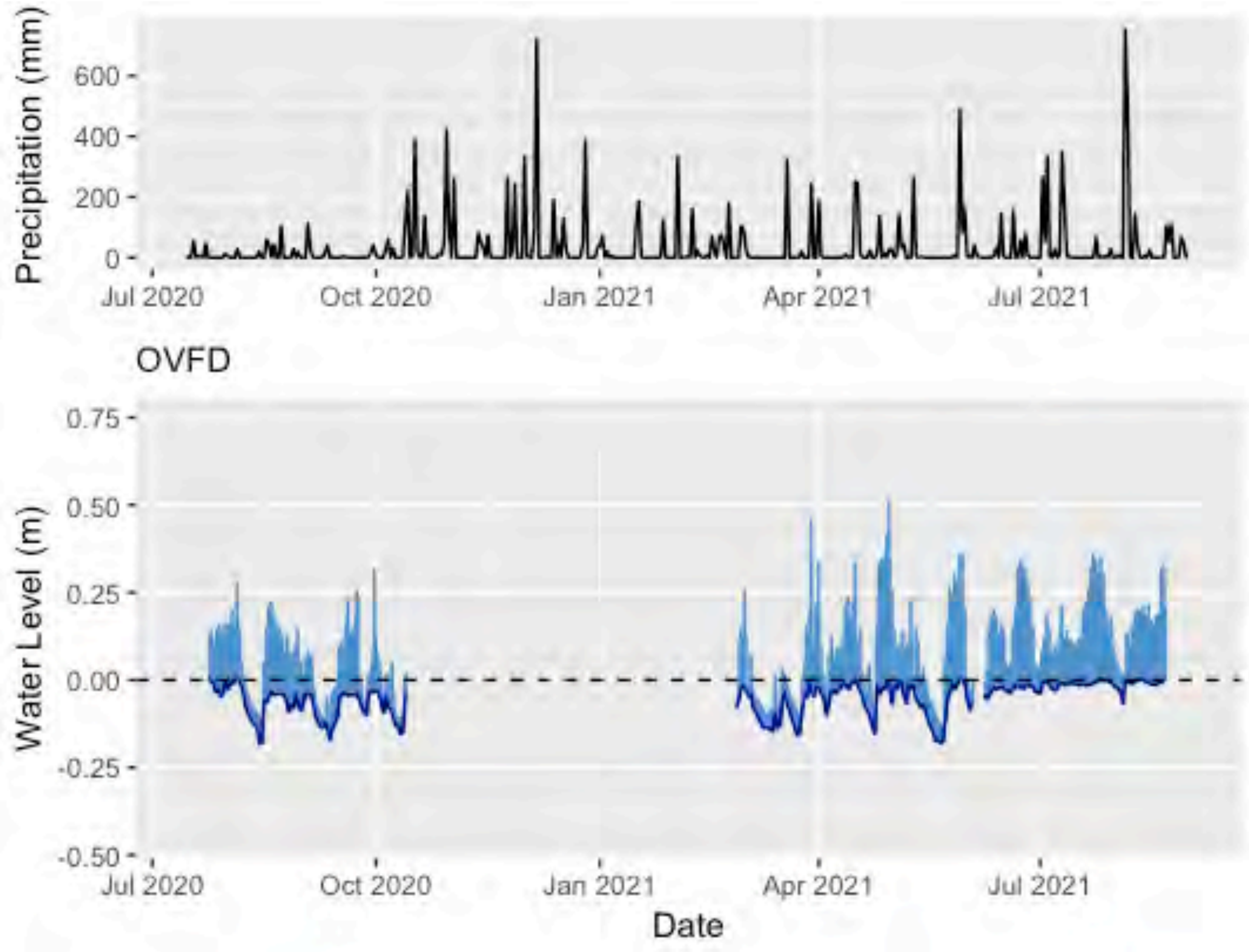


Figure 12. Before and after photograph at OVFF, Reference Site 2.

Figure 13. Local precipitation, and water levels from before runnels were installed (2020) and after (2021) at OVFF, Reference Site 2. Light blue line shows 15-minute data, and the dark blue line shows the water table height.

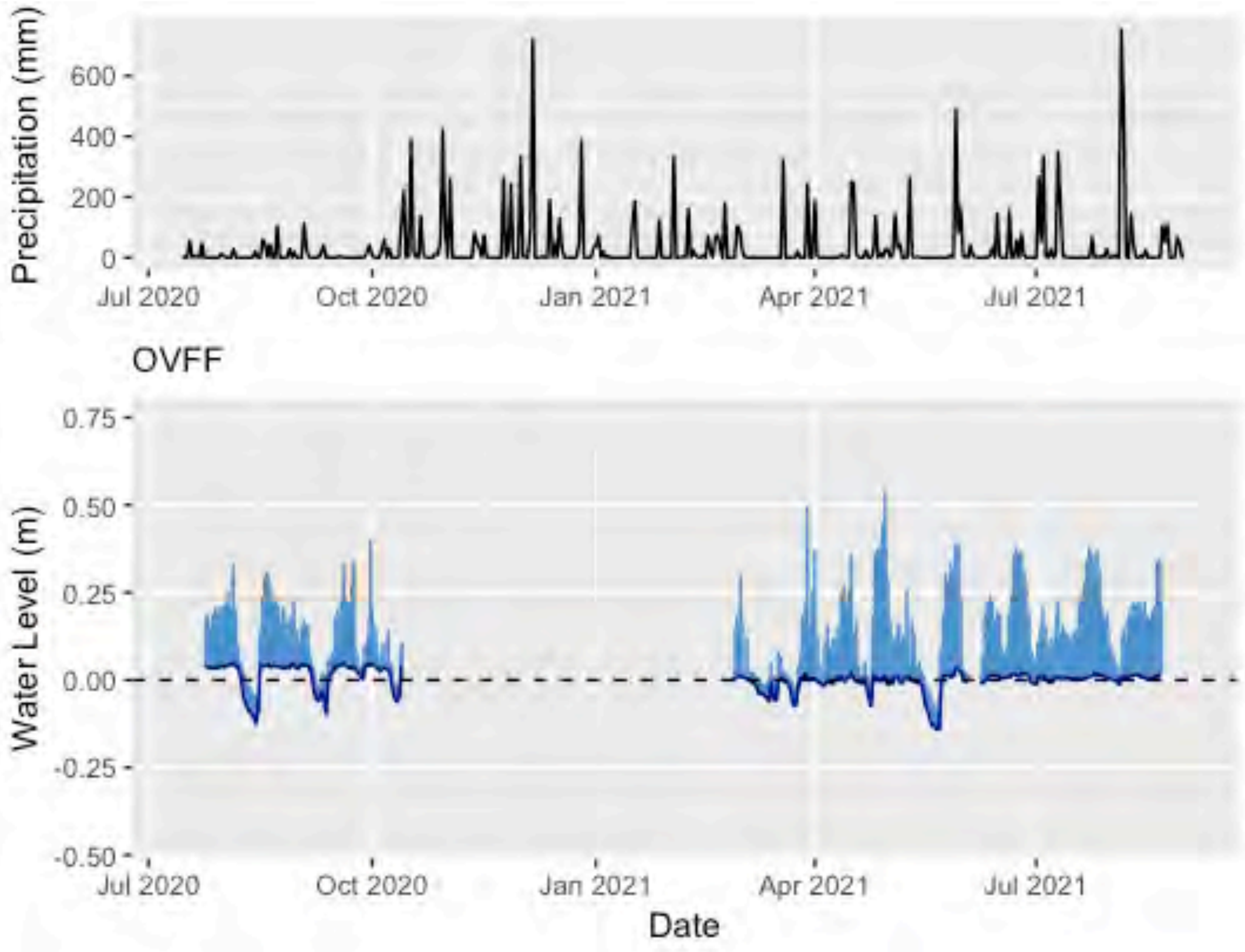


Figure 14. Before and after photograph at OVFG, Reference Site 3.

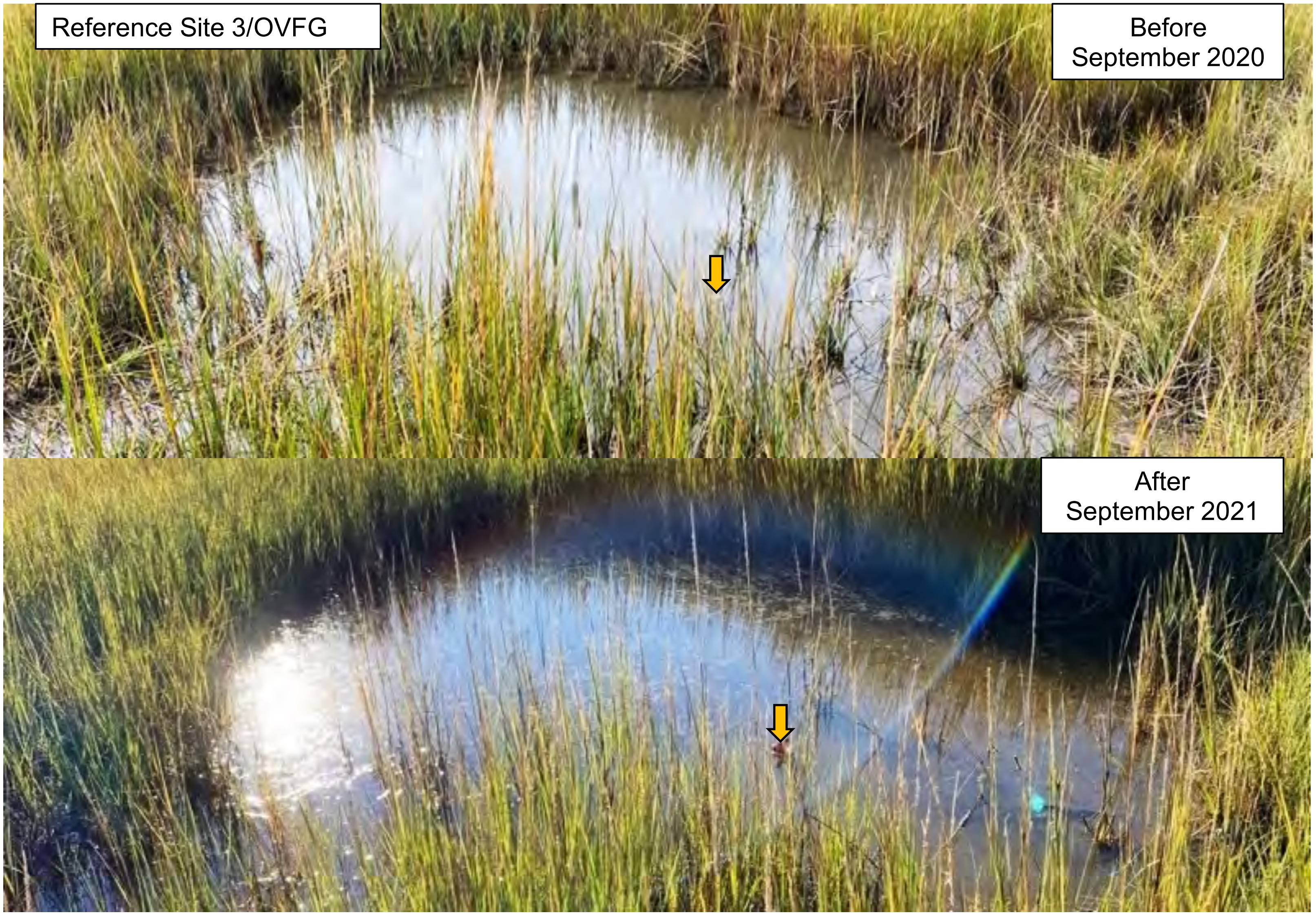


Figure 15. Local precipitation, and water levels from before runnels were installed (2020) and after (2021) at OVFG, Reference Site 3. Light blue line shows 15-minute data, and the dark blue line shows the water table height.

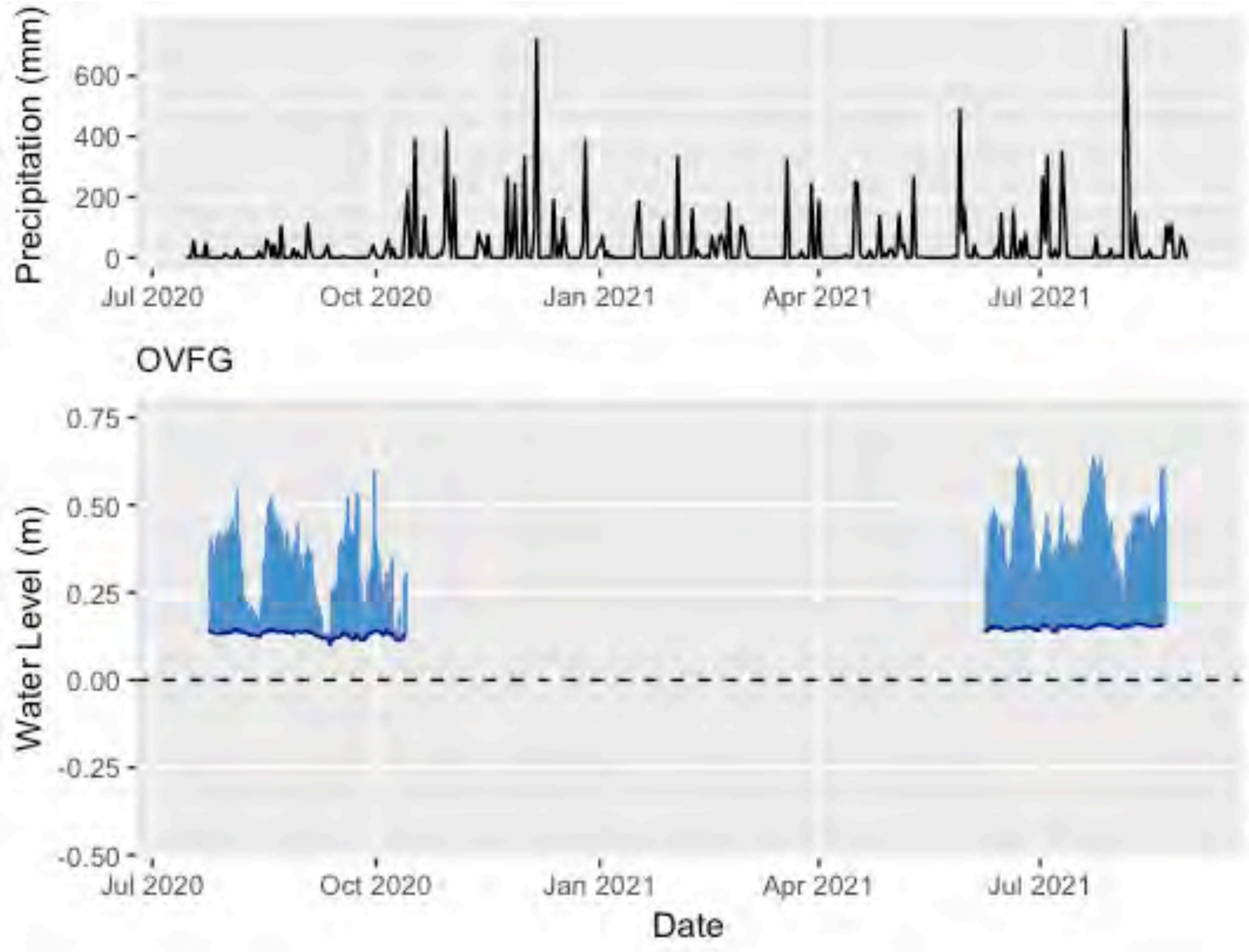


Figure 16. Before and after photograph at LBND, Runnel Site 4.

Figure 17. Local precipitation, and water levels from before runnels were installed (2020) and after (2021) at LBND, Runnel Site 4. Light blue line shows 15-minute data, and the dark blue line shows the water table height.

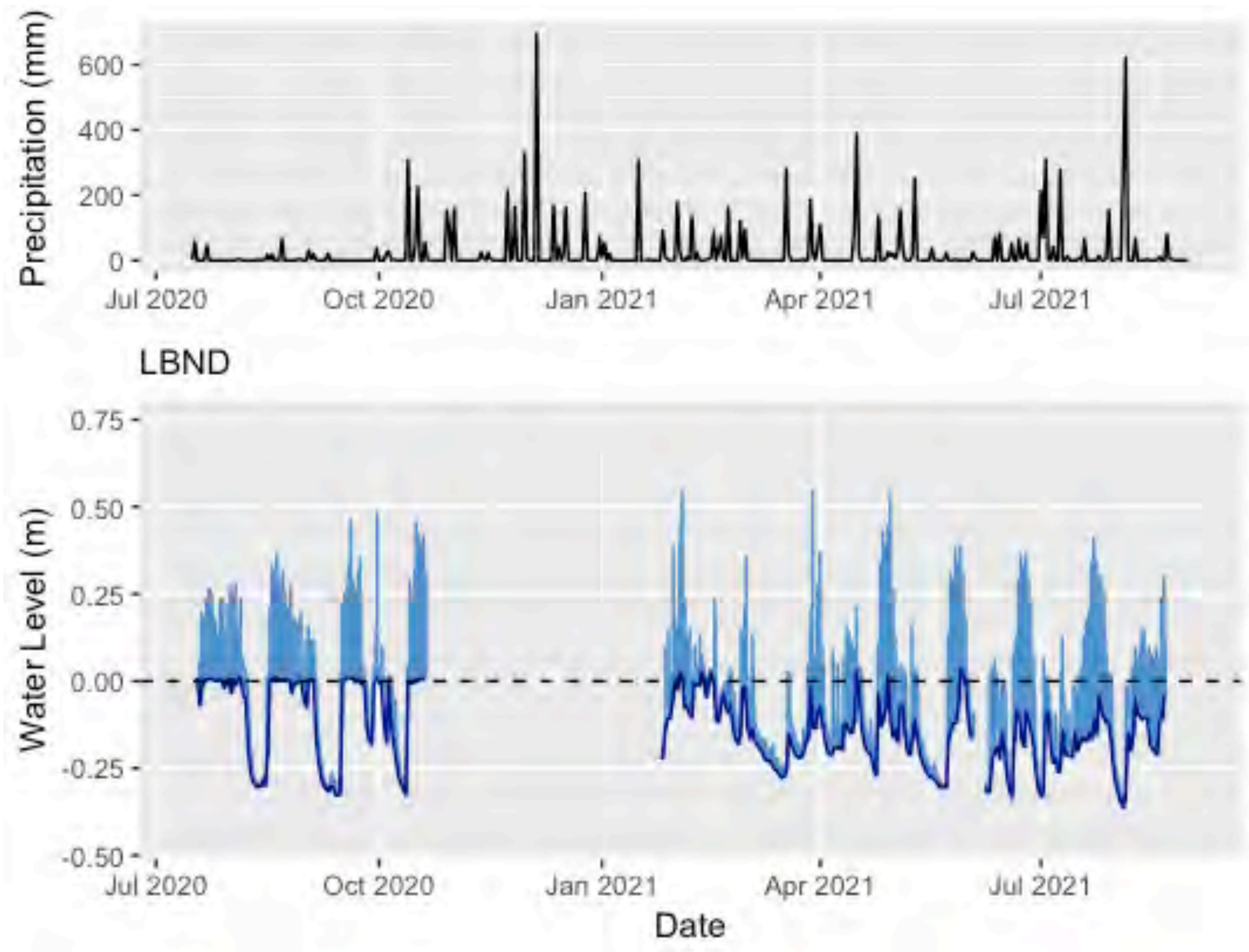


Figure 18. Before and after photograph at LBNF, Runnel Site 5.

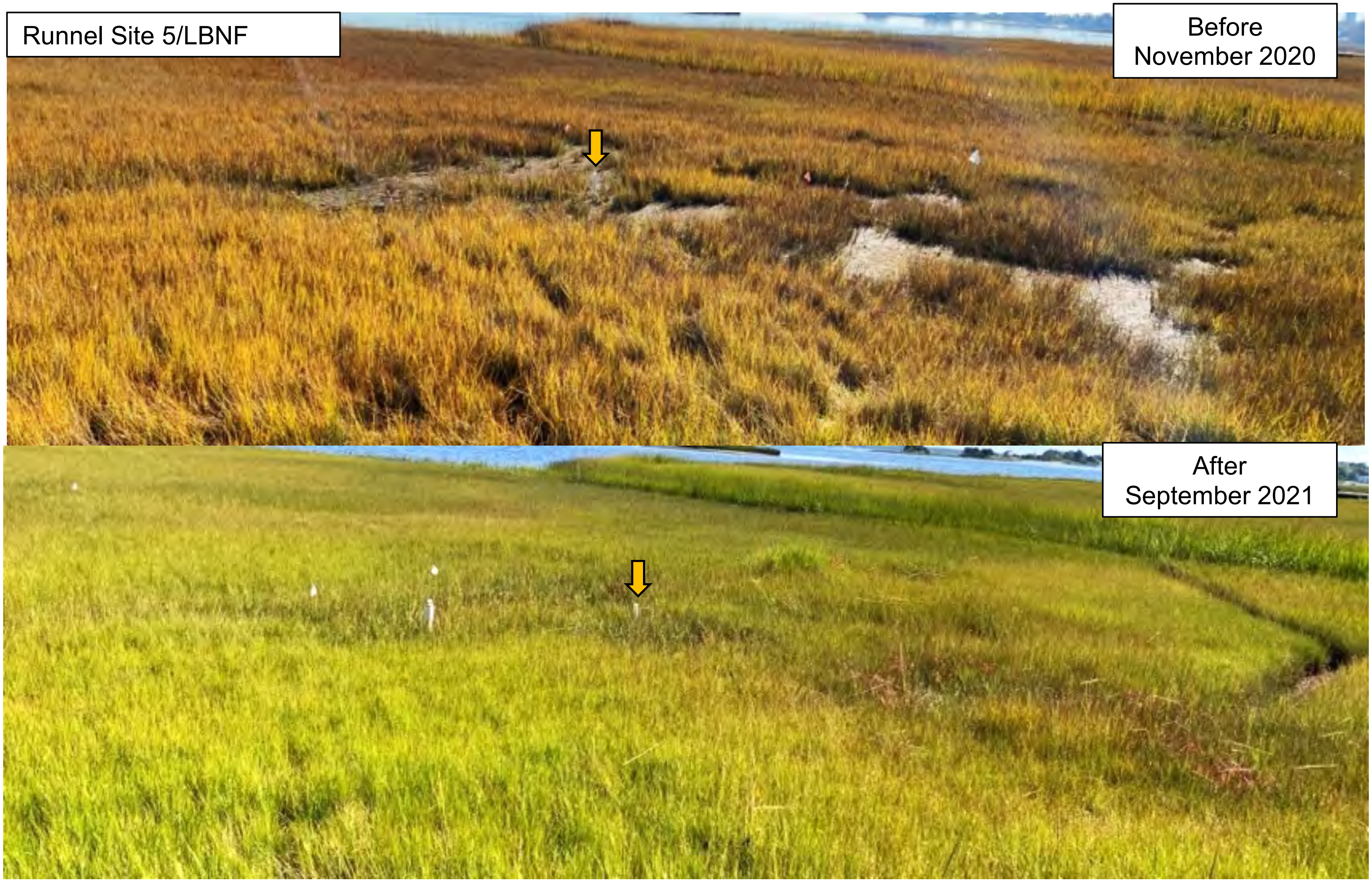


Figure 19. Local precipitation, and water levels from before runnels were installed (2020) and after (2021) at LBNF, Runnel Site 5. Light blue line shows 15-minute data, and the dark blue line shows the water table height.

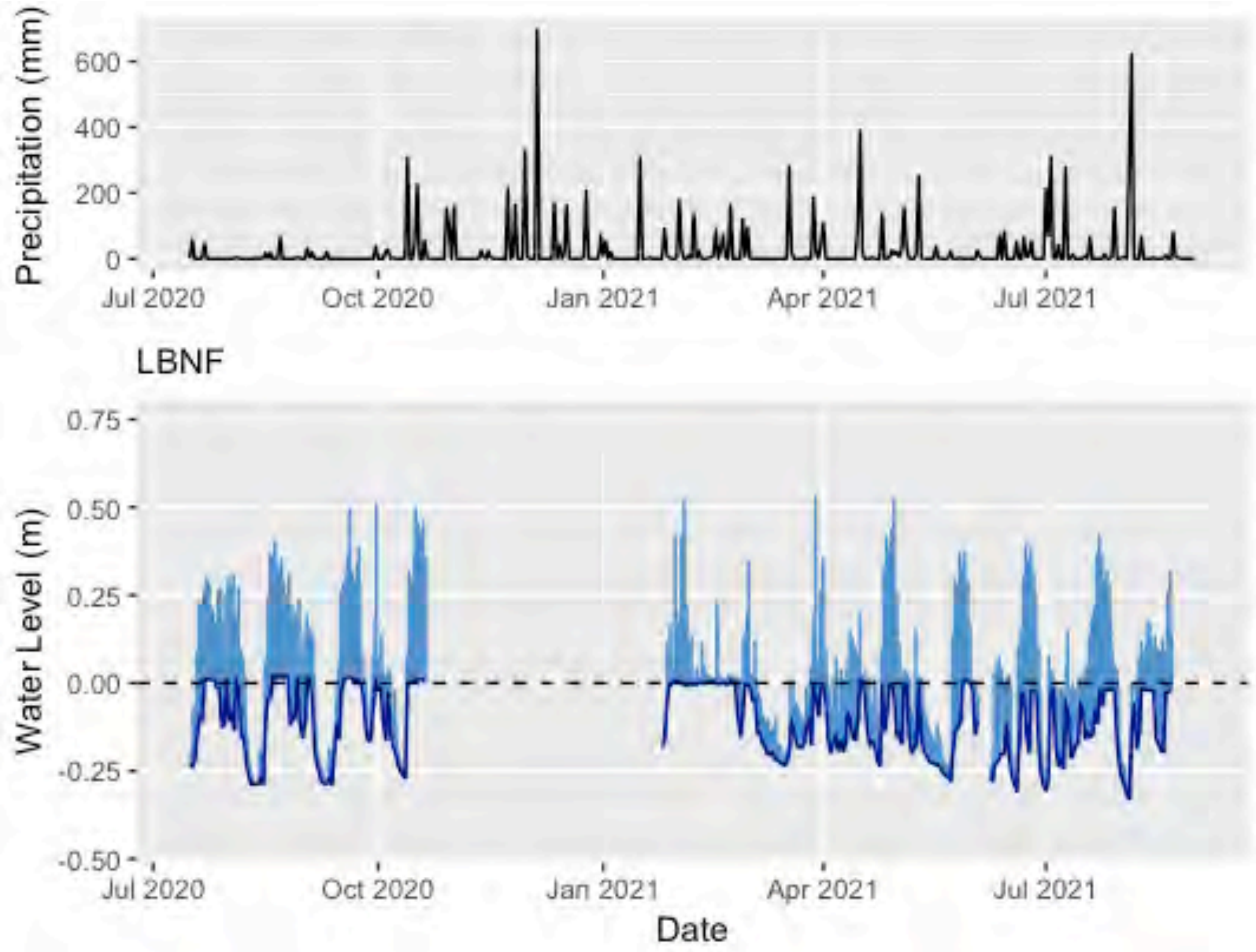


Figure 20. Before and after photograph at LBSM, Runnel Site 6.

Figure 21. Local precipitation, and water levels from before runnels were installed (2020) and after (2021) at LBSM, Runnel Site 6. Light blue line shows 15-minute data, and the dark blue line shows the water table height.

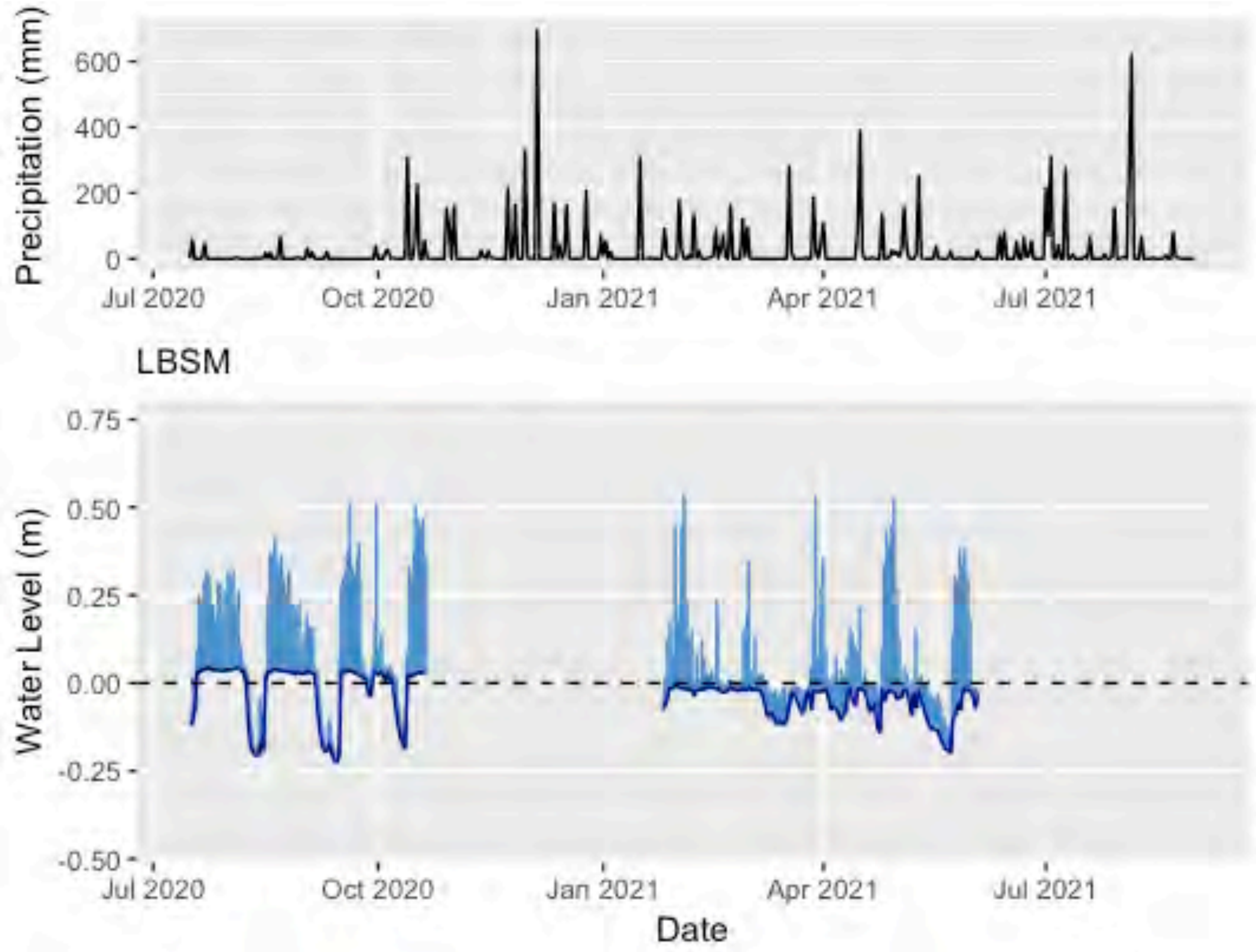


Figure 22. Before and after photograph at LBNA, Reference Site 4.

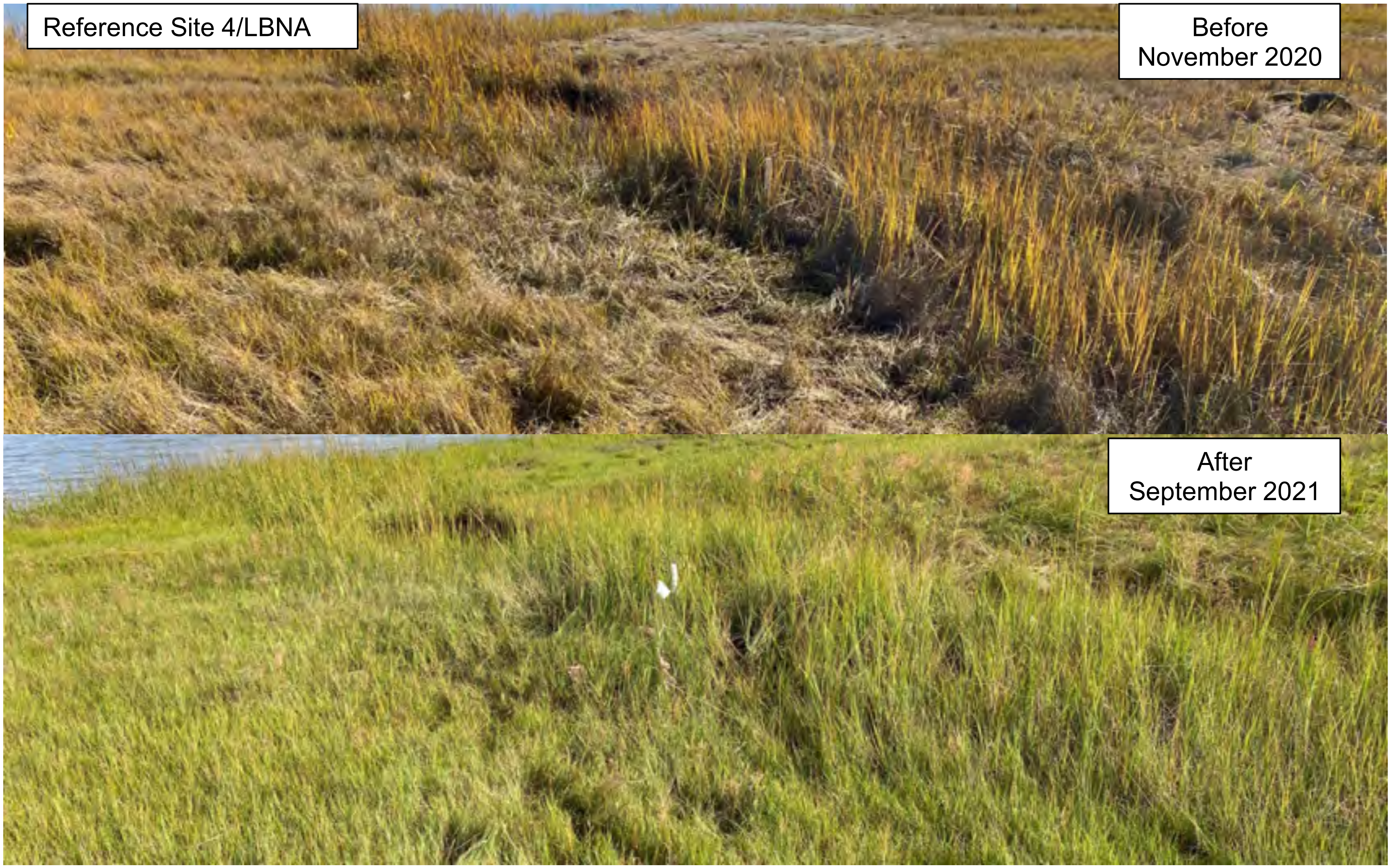


Figure 23. Local precipitation, and water levels from before runnels were installed (2020) and after (2021) at LBNA, Reference Site 4. Light blue line shows 15-minute data, and the dark blue line shows the water table height.

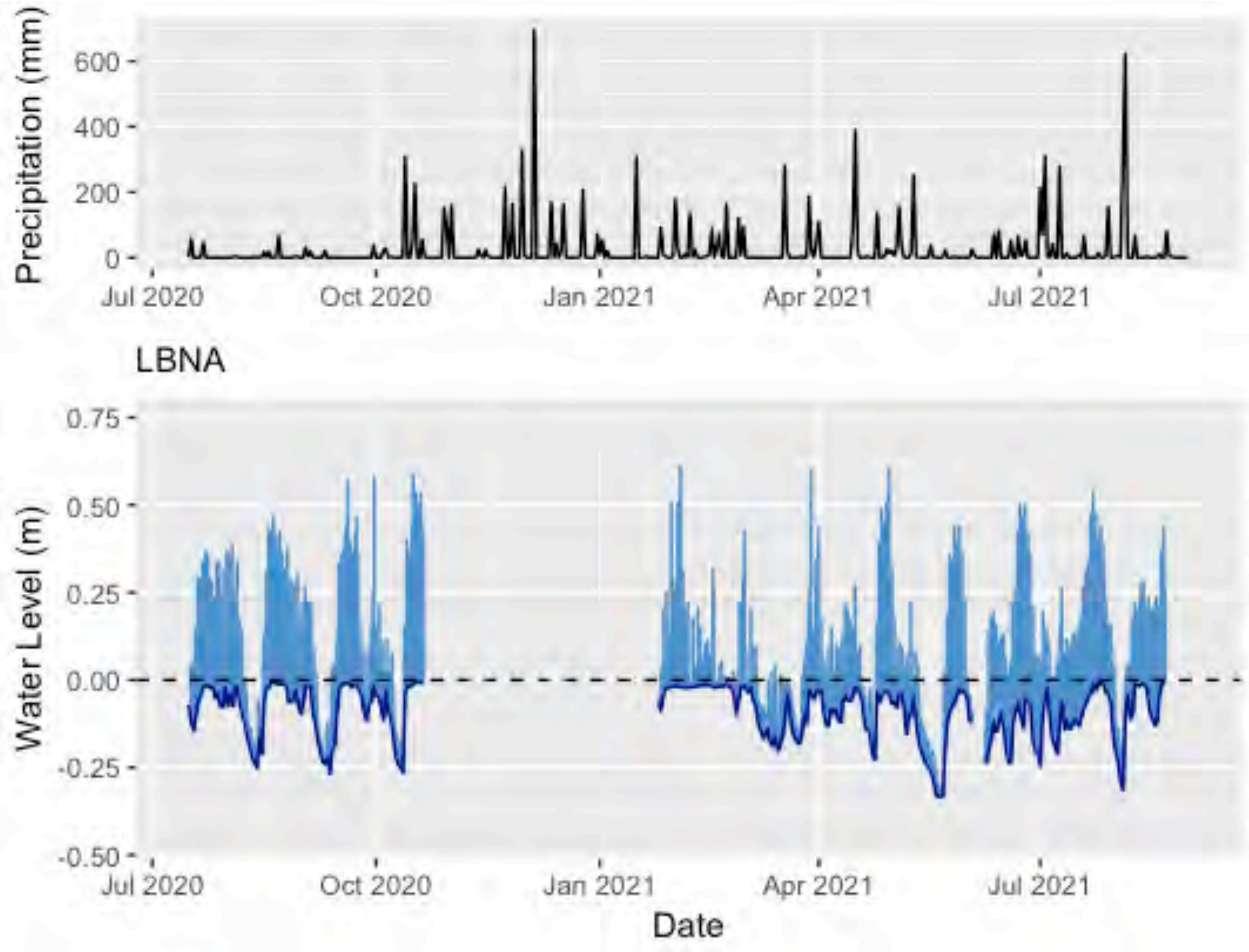


Figure 24. Before and after photograph at LBSB, Reference Site 5.

Figure 25. Local precipitation, and water levels from before runnels were installed (2020) and after (2021) at LBSB, Reference Site 5. Light blue line shows 15-minute data, and the dark blue line shows the water table height.

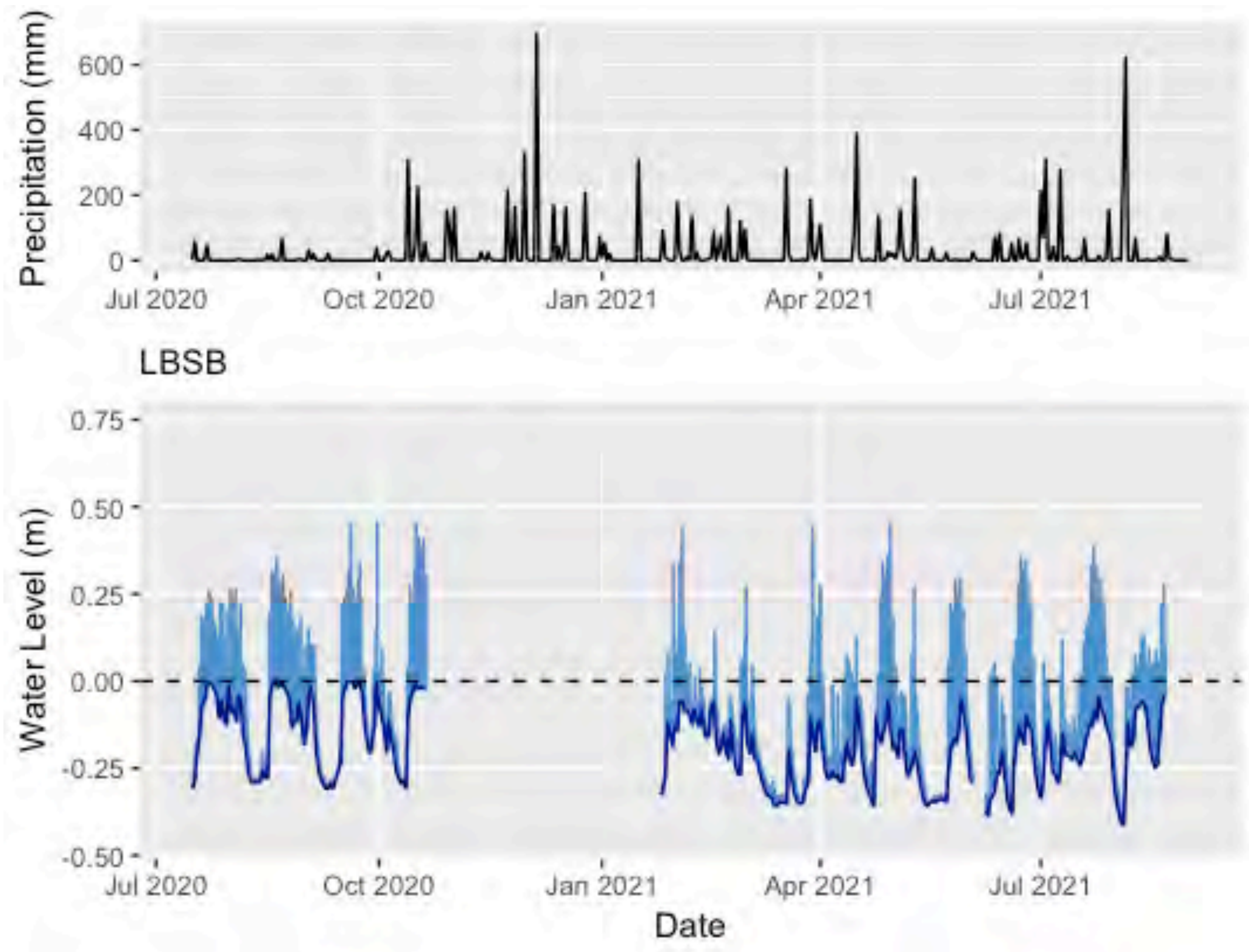


Figure 26. Before and after photograph at LBSC, Reference Site 6.

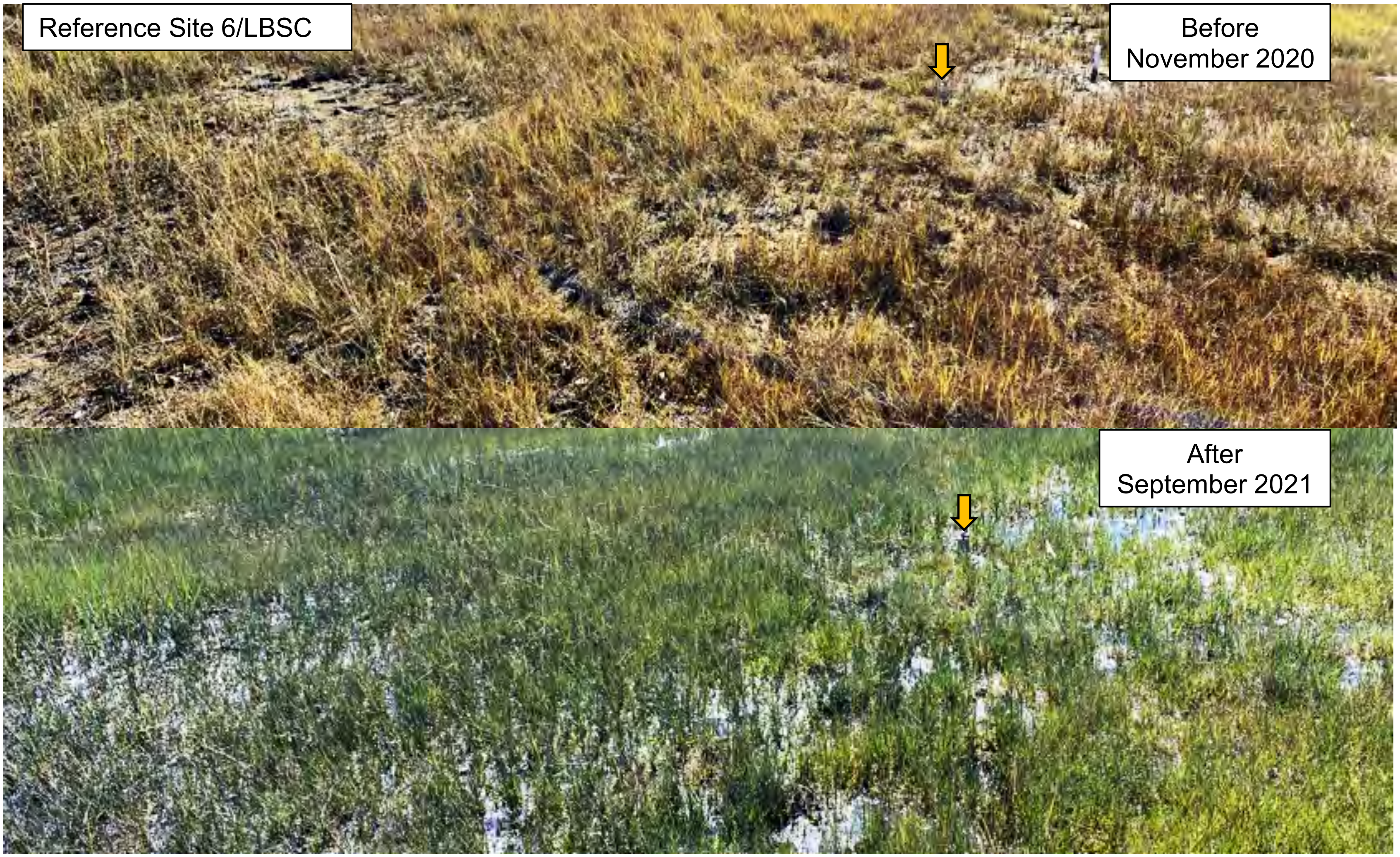


Figure 27. Local precipitation, and water levels from before runnels were installed (2020) and after (2021) at LBSC, Reference Site 6. Light blue line shows 15-minute data, and the dark blue line shows the water table height.

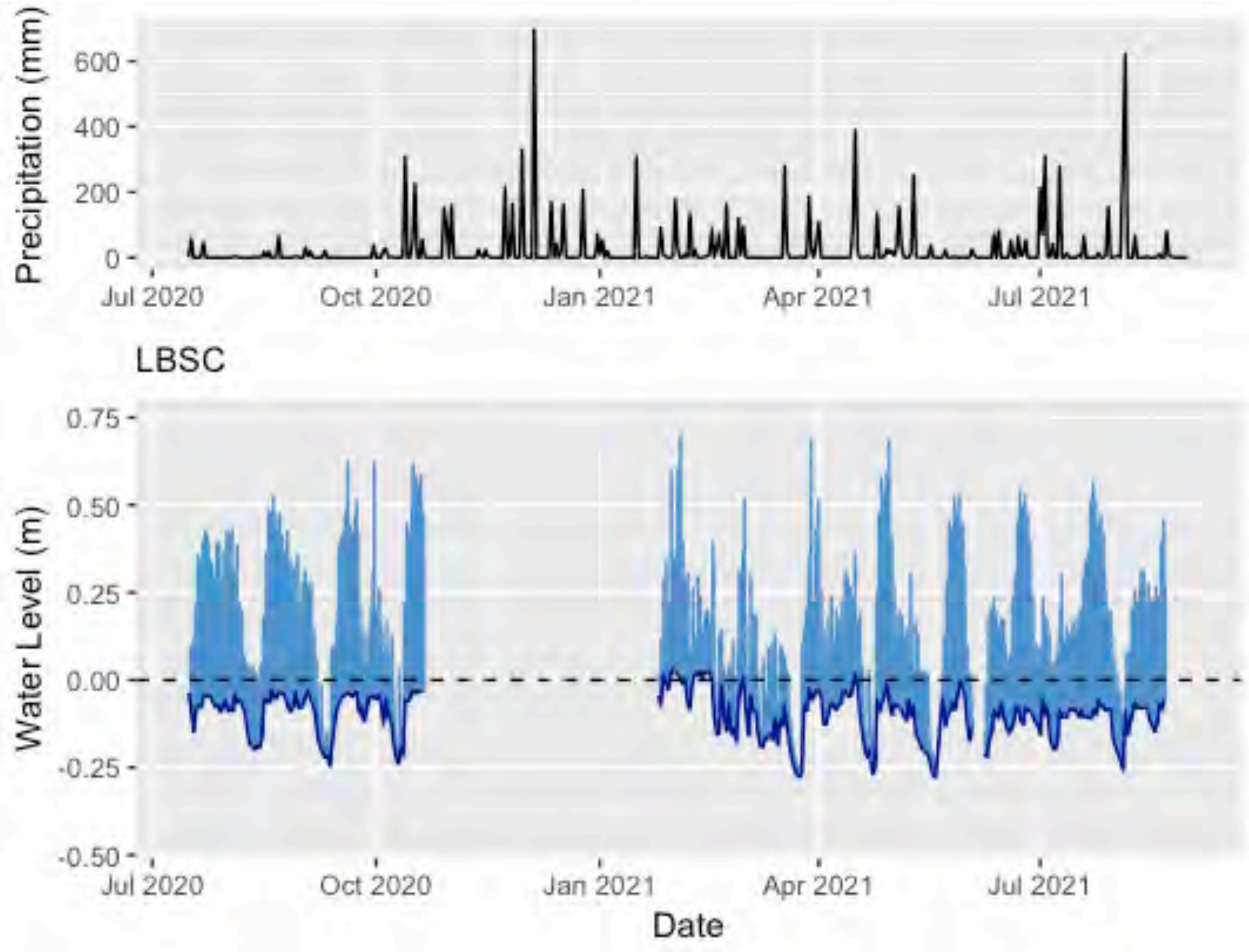


Figure 28. Results of statistical test for an effect of runnels on water table heights at LB and OVF. Displaying mean and water table heights and standard error for runnel and reference sites, before and after runnel-installation, and accounting for between-site differences. Model results indicated runnels decreased water table heights at OVF relative to reference sites, but not at LB ($p < 0.001$).

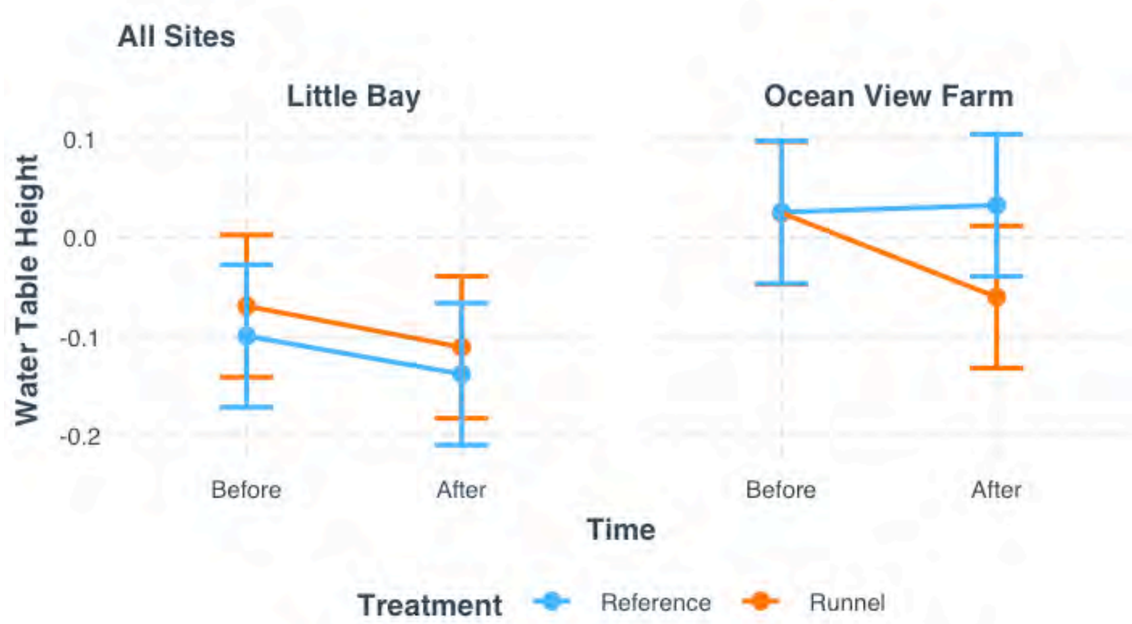


Figure 29. Soil moisture content as percent of wet weight of soil cores measured at a) OVF and b) LB sites. Boxplots show percentages before (purple, dark green) and after (gray, light green) runnel installation, in 'Down' (creekward), 'Center' (within dieback), and 'Up' (upland) experimental zones, at both runnel and reference sites.

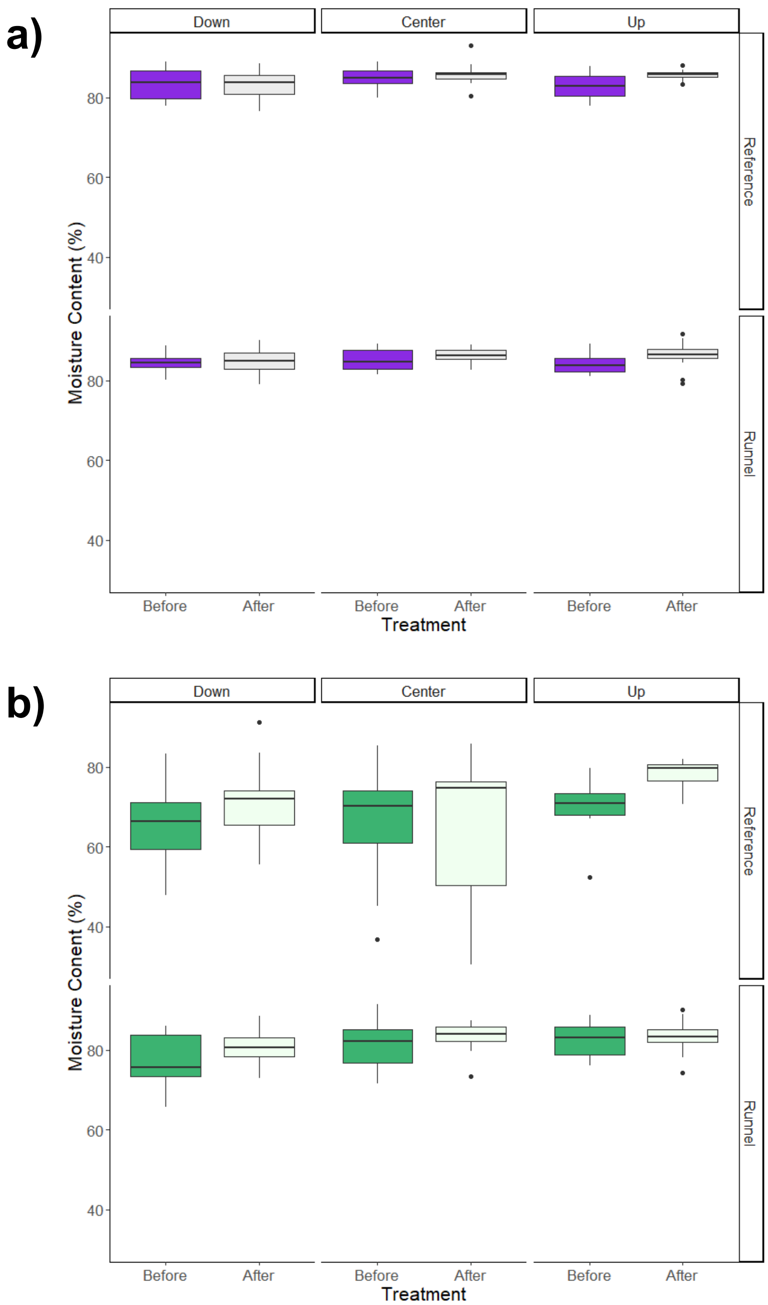


Figure 30. Porewater salinity as parts per thousand (ppt) measured at a) OVF and b) LB sites. Boxplots show percentages before (purple, dark green) and after (gray, light green) runnel installation, in 'Down' (creekward), 'Center' (within dieback), and 'Up' (upland) experimental zones, at both runnel and reference sites.

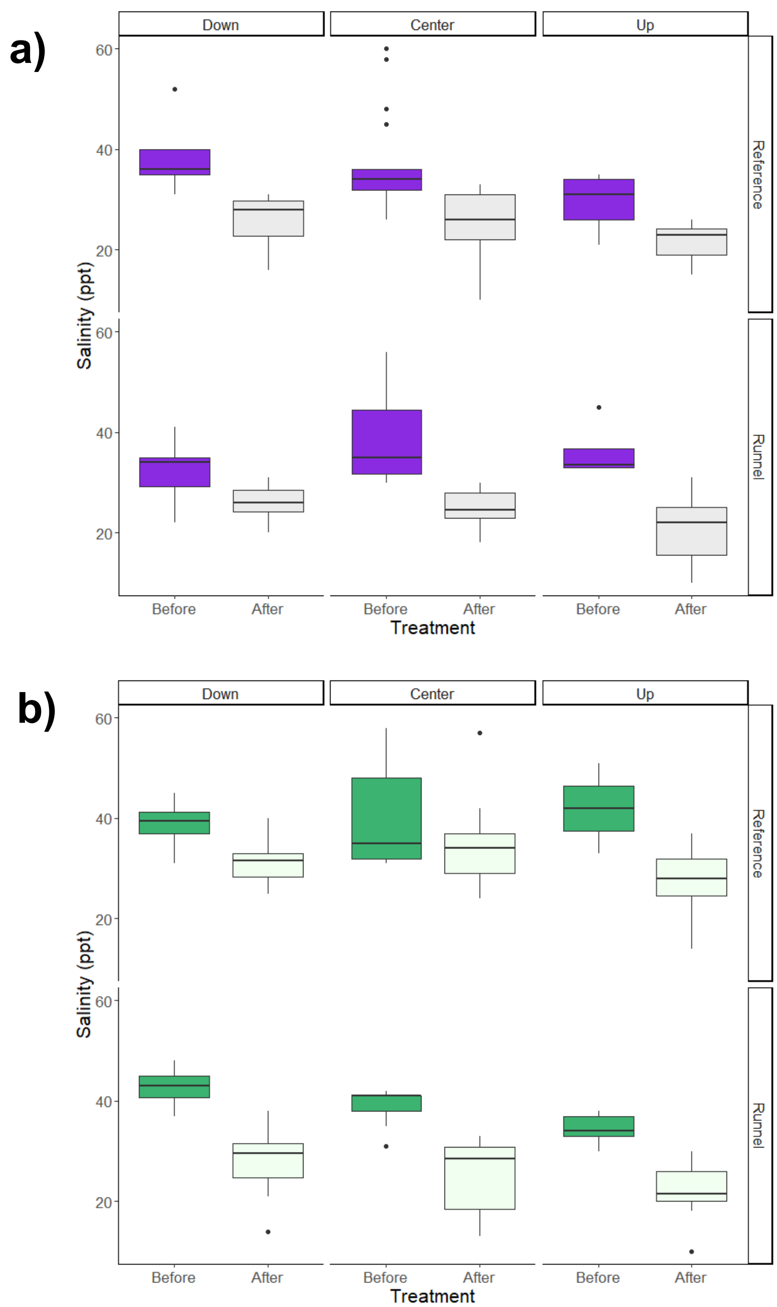


Figure 31. Illustration of method to estimate hydroperiod. 15-min water level data in m shown as a time series, identified high and low tide points identified using software indicated with open circles, and reference minimum water level used to identify low tide points shown as blue dashed horizontal line.

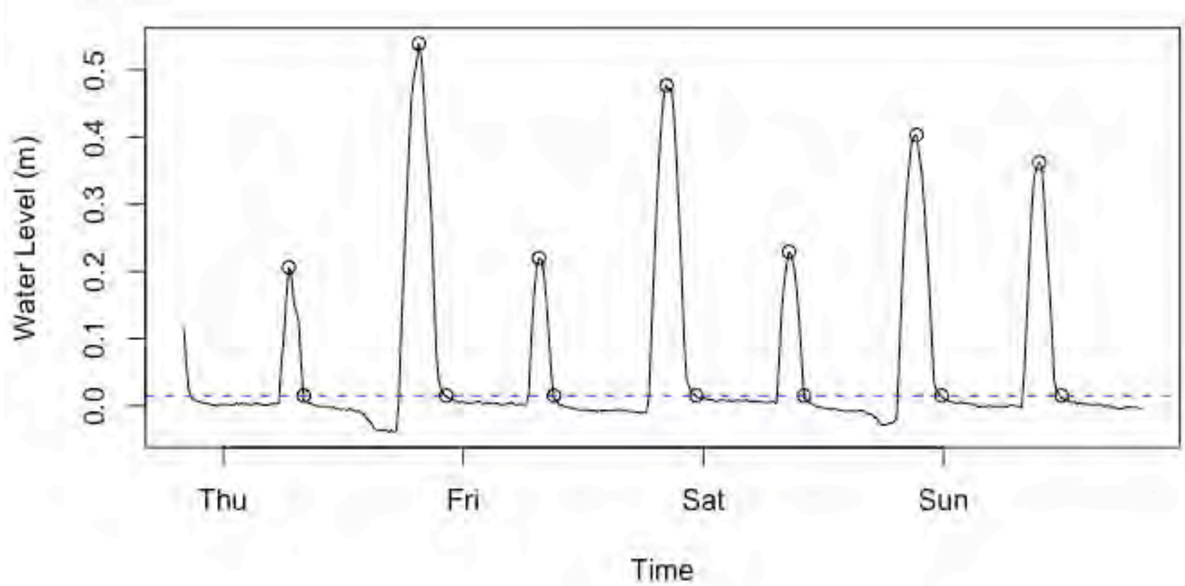


Figure 32. Redox potential measured as millivolts (mV) at a) OVF and b) LB sites. Boxplots show percentages before (purple, dark green) and after (gray, light green) runnel installation, in 'Down' (creekward), 'Center' (within dieback), and 'Up' (upland) experimental zones, at both runnel and reference sites.

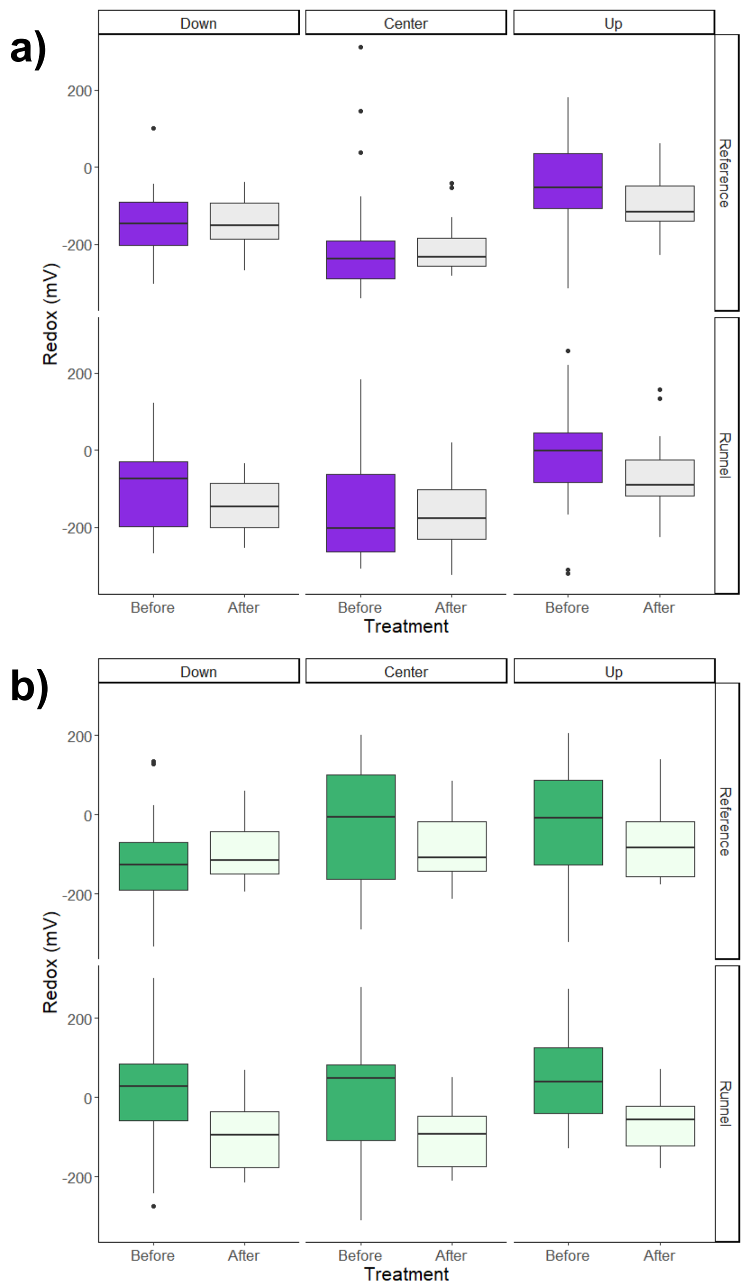


Figure 33. Decomposition of tea bags expressed with the K constant at a) OVF and b) LB sites. Boxplots show percentages before (purple, dark green) and after (gray, light green) runnel installation, in 'Down' (creekward), 'Center' (within dieback), and 'Up' (upland) experimental zones, at both runnel and reference sites.

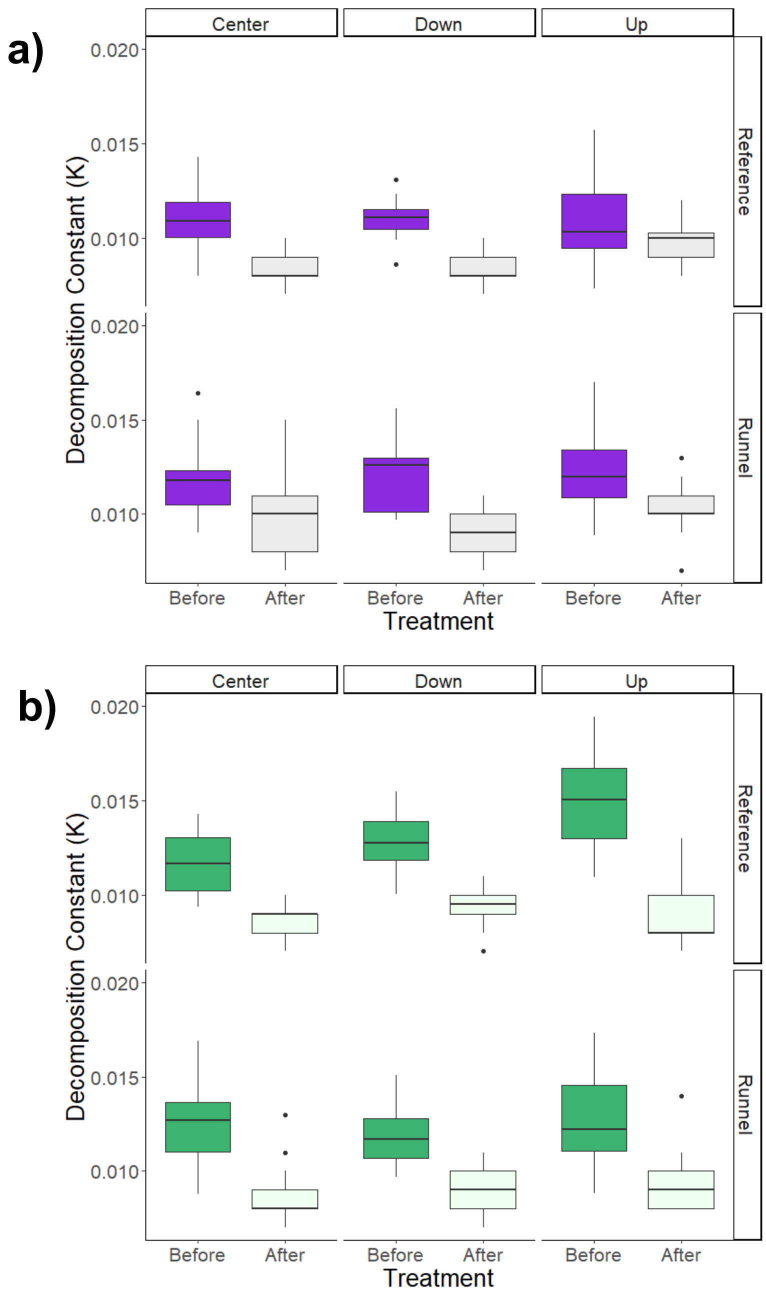
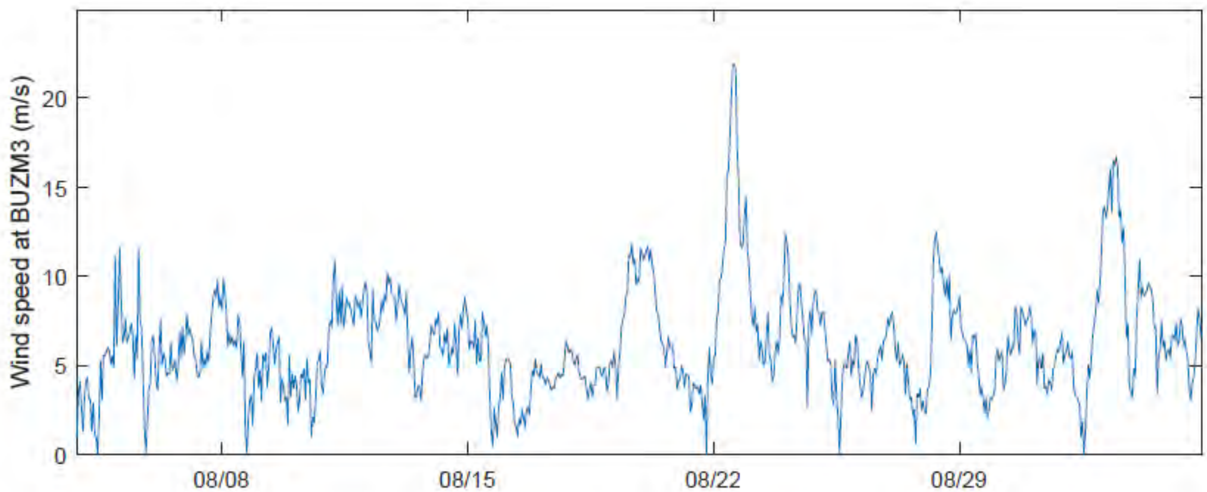


Figure 34. Upper panel: Wind speed at BUZM3 (entrance to Buzzards Bay), lower panel: turbidity at Little Bay channel site (red) and Ocean View Farm channel site (blue). Large peaks at LB represent resuspension of bed material during storms, with perhaps slight export of sediment from the marsh during events (but overall neutral transport over the entire time period). OVF shows no response to wind-wave resuspension.



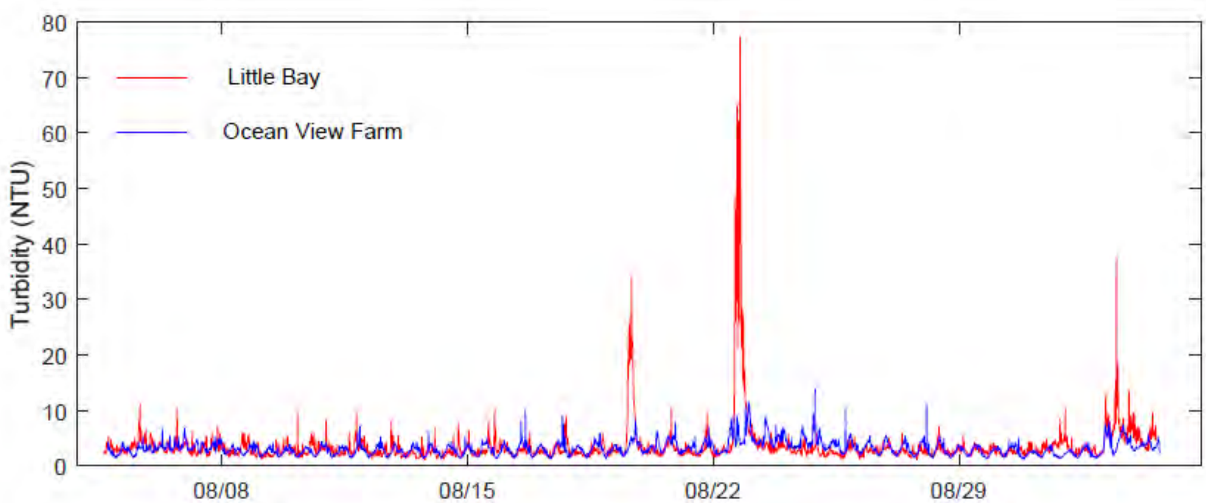
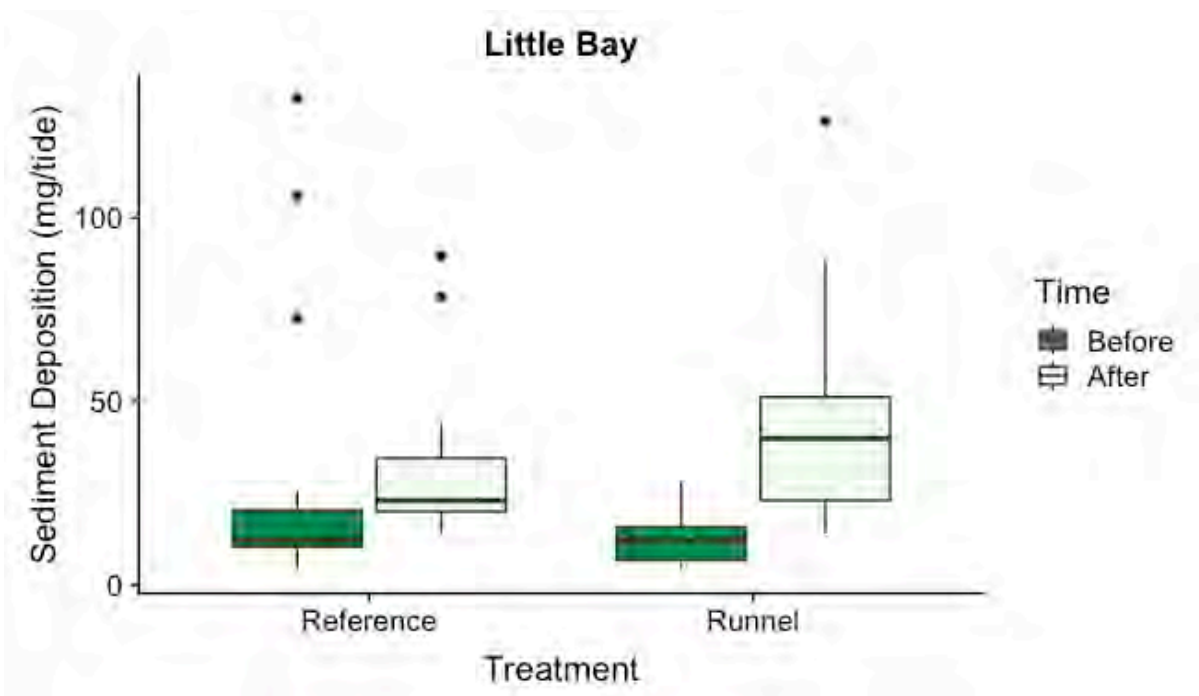


Figure 35. Sediment deposition as milligrams (mg) of inorganic sediment accumulated per flooding tide of 10-cm depth or more at LB. Boxplots show sediment deposition in before (dark green) and after (light green) periods, at both reference and runnel sites. Initial data exploration indicates that sediment deposition might increase within shallow water areas after creating runnels (p = 0.01).



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