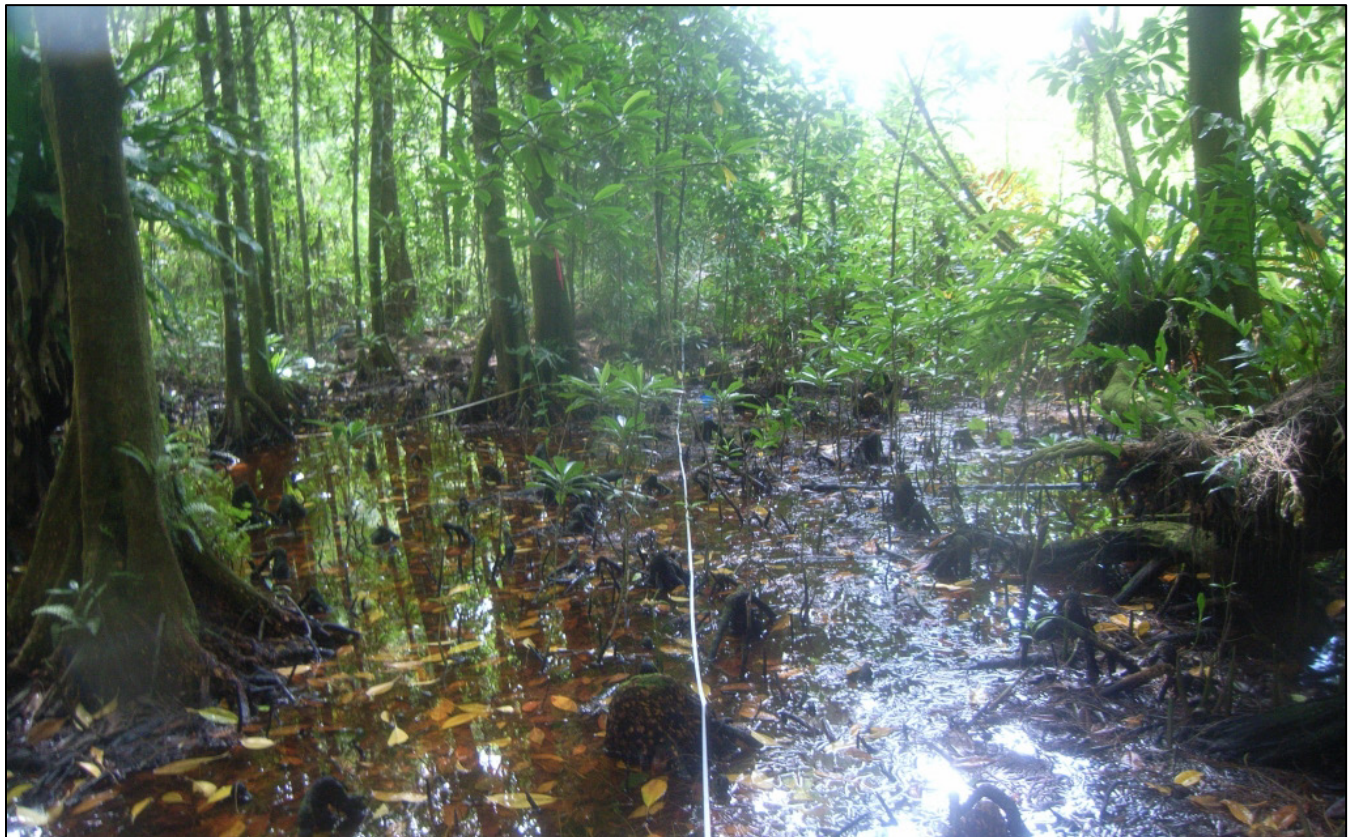




Trends in Water Quality and Assessment of Vegetation Community Structure in Association with Declining Mangroves

A Condition Assessment of American Memorial Park

Natural Resource Report NPS/PACN/NRR—2023/2610



ON THE COVER

The AMME mangrove wetland in 2014.

NPS / JACOB GROSS

Trends in Water Quality and Assessment of Vegetation Community Structure in Association with Declining Mangroves

A Condition Assessment of American Memorial Park

Natural Resource Report NPS/PACN/NRR—2023/2610

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Abstract

American Memorial Park (AMME) in Saipan contains a rare mangrove wetland that is known to support several endangered species. Through monitoring water quality and vegetation characteristics of the wetland for >10 years we documented a declining mangrove population, an increase in invasive plant species, and declining surface water salinity. Comprehensive surveys conducted in 2014 and 2019 quantified declines in the plant community observed by park staff. Surface water salinity declined from 2009 to 2018 and no other trend in surface water quality was observed. Over the time period of the present study, AMME experienced shifts in annual rainfall conditions that could be associated with ENSO cycles. Dry conditions beginning in late 2016 and continuing through mid-2018 resulted in some surface water sampling sites completely drying. Several stressors may have contributed to declines in mangroves adapted to saturated soils directly and allowed competing plants to proliferate, including disruption of hydrologic connectivity with marine waters resulting in reduced surface water salinity, reduced rainfall causing dry soil conditions, and physical storm damage to canopies. Recommendations include study of groundwater salinity, maintaining or modifying a culvert subject to filling with sediment or other excavation work to improve saline water flow to the wetland at high tides, the establishment of a new groundwater monitoring well, adding a surface water monitoring station near the culvert, conducting a spatial assessment of the mangrove habitat suitability within the mangrove wetland, and developing or assisting with mangrove interpretive and outreach programs.

Introduction

American Memorial Park (AMME) is an affiliated area of the U.S. National Park Service (NPS) on the island of Saipan in the Commonwealth of the Northern Mariana Islands (CNMI, Figure 1). The park was established in 1978 to honor the indigenous peoples and American soldiers who died in World War II during the 1944 Mariana Island Campaign. The park consists of memorials, an amphitheater, a marina, recreational facilities, historical ruins, shorelines, a small artificially constructed wetland designed to ameliorate the outflow of a stormwater drain, and more natural areas including a 30-acre mangrove-containing wetland also called Muchot Marsh (Prasad 1997) or Muchot Point (Thomas and Price 1979). This area, hereafter referred to as the “mangrove wetland” was designated by the U.S. Fish and Wildlife Service as a Mitigation Policy Resource Category 2, or “habitat...of high quality for evaluation species and is relatively scarce or becoming scarce on a national basis or in the ecoregion” (Wagner 1990). The site was also designated a “unique or pristine resource” in the Pacific Island Network (PACN) Vital Signs Monitoring Plan (DeVerse and DiDonato 2006).

Mangroves are woody plants adapted to brackish-water conditions, receiving saline water from seawater infiltration of groundwater and/or marine surface flows. Mangrove species are often further categorized into major, minor, and associates based on their reliance on the mangrove environment (specialist vs. generalist), importance to the community structure, and other criteria (Tomlinson 1986). The only “major” mangrove found in the AMME mangrove wetland is *Bruguiera gymnorhiza*, a native species. *B. gymnorhiza* trees are evergreen, reach a height of 30 m, display a straight trunk of 40 to 90 cm in diameter with a buttressed base and knee-like roots that project above the substrate. Extensive mangrove stands across the island were removed for developments during the 20th-century Japanese colonial period and World War II. The AMME mangrove wetland is one of just three remaining relic mangrove stands in CNMI, all of which occur in Saipan (Greene et al. 2019).

The AMME mangrove wetland is an estuarine groundwater-dependent system with ocean-derived saline water inputs and intermittent tidal flooding (Wagner 1990, Perreault 2007). This wetland is a juvenile fish nursery and an edible crab habitat. The wetland helps to limit pollution from developed uplands from entering the sea and controls ongoing and significant coastal erosion (Greene et al. 2019, Mike Gawel personal com.). The wetland provides habitat for three federally listed endangered species which are only known from Guam and the Northern Mariana Islands: the nightingale reed warbler (*Acrocephalus luscinius*), Mariana moorhen (*Gallinula chloropus guami*), and humped tree snail (*Partula gibba*) (Williams et al. 2007). Loss of the AMME mangrove wetland would further exacerbate habitat loss to these endangered species.



Figure 1. American Memorial Park, Saipan, featuring a constructed wetland and a mangrove wetland.

The climate, geology, and hydrology of Saipan are factors relevant to interpreting mangrove condition and water quality data collected in the mangrove wetland. Saipan experiences wet/dry seasonality, with the dry season characterized as November through April and the wet season as May through October. The El Niño Southern Oscillation cycle (ENSO, reviewed by Wang et al. 2016) also has the potential to affect the AMME mangrove wetland by creating multi-year rainfall patterns that could alter the amount of freshwater infiltration into the aquifer and surface water salinity (reviewed by Greene et al. 2019). The park is located along the coast on a mix of carbonate sands and the mangrove wetland is located at least partially on artificial fill overlying carbonate sands (Perreault 2007, Greene et al. 2019). The location of the park and the high porosity of its underlying substrate and low elevation result in the mixing of seawater infiltrating from the shoreline and freshwater derived from direct rainfall and upland groundwater flow. Hence the wetland, park, and island sit above a fresh groundwater lens overlying saltwater (Perreault 2007).

Historically, the mangrove wetland was noted as Muchot Marsh in records as early as 1855 and appeared on a map dated c1914 (Prasad 1997). While the island has been occupied by humans since prehistoric times, the area now designated as the mangrove wetland was developed during World War II (e.g., study sections D and E in Thomas and Price 1979) and later. An archeological survey of the wetland noted a buried sewage line, remains of buildings, a latrine, and assorted artifacts associated with Japanese and American military use during WWII. Prasad (1997) noted other studies which found vegetative evidence of standing water and reviewed historical records noting the presence of the mangrove wetland (Figure 2) and historical landscape changes, concluding that the mangrove wetland may have been connected to the ocean and may have been connected to a patch of mangroves north of Rt. 30 (referred to as Beach Road). A botanical inventory conducted at AMME in 1986 and 1988 (Raulerson and Rinehart 1989) noted that *B. gymnorhiza* was abundant along some transects and recognized as an important component to the ecosystem as the only mangrove species within AMME and Saipan. Raulerson and Rinehart (1989) also noted that “other [“major” or “true” mangrove] species are common to mangrove swamps further south,” possibly referring to *Rhizophora apiculata* and *Rhizophora mucronata* in Guam. A palynological assessment of sediment cores from AMME revealed *B. gymnorhiza* and *Acrostichum aureum* pollen during the Holocene indicating mangrove presence predating that of human arrival (Jarzen and Dilcher 2009). No pollen from the genus *Rhizophora* was recovered from the Jarzen and Dilcher (2009) cores, but the authors state that *Rhizophora* trees did previously occur in the Northern Mariana Islands, including Saipan. McIntosh and Cleghorn (2001) noted the presence of Rhizophoraceae the family containing *Bruguiera*, during archeological surveys. McIntosh and Cleghorn (2010) noted the presence of *B. gymnorhiza* specifically.

The total area and extent of *B. gymnorhiza* within the mangrove wetland, the AMME boundary, and the greater population across Saipan has never been explicitly measured. Cogan et al. (2013) used 2006 satellite imagery and 2010–2011 accuracy assessments to classify vegetation across the entire AMME legal boundary. They estimated a vegetation class with dominant *B. gymnorhiza* cover at 3.5 hectares (8.6 acres; users’ accuracy 80% [90% CI: 40–100%]) and other map classes containing minor *B. gymnorhiza* cover at 17.7 ha (43.7 ac). Using just the dominant *B. gymnorhiza* map class estimate, 2.1 ha (5.3 ac) occurred inside the mangrove wetland, and another 1.3 ha (3.3 ac) occurred

along the coastline on the eastern side of the Smiling Cove causeway. After the Cogan et al. (2013) vegetation map, additional *B. gymnorhiza* has been planted within AMME at other locations in the park, including the constructed wetland, both inland along the constructed wetland's waterway and its costal sandy mouth.



Figure 2. Aerial photo dated July 1949 showing structures and roads within the area now designated as the mangrove wetland, and an approximate historical extent of the mangrove wetland (red line) as described by Prasad (1997).

Other notable locations on Saipan which contain small but contiguous cover of *B. gymnorhiza* include Sadog Tasi and Tanapag stream. The Commonwealth of the Northern Mariana Islands estimated the length of the Sadog Tasi stand as 755 ft and the length of Tanapag stream stand as 140 ft (CNMI 2019). If a 100 ft buffer is presumed along this length, then the resulting area estimate is: Sadog Tasi stand ~ 0.7 ha and Tanapag stream stand ~ 0.2 ha. There are 2 to 3 smaller, isolated stands between these sites and interestingly one *B. gymnorhiza* individual found further south on the shore of Lake Susupe (Zachary Williams, personal com.). This estimate supports the statement by Williams et al. (2007) that the section of mangroves within AMME are the largest and most intact on Saipan.

Globally, mangrove species richness at the local (i.e., hectare) scale typically includes 6 to 12 species within the Indo-West Pacific and 3 to 4 mangrove species in the Atlantic-Caribbean-East Pacific with diversity decreasing moving away from New Guinea towards the Pacific Islands (Woodroffe 1987, Ricklefs and Latham 1993). All remaining mangrove stands in CNMI are considered monospecific because the only “major” mangrove species present is *B. gymnorhiza*. Conversely, the territory of Guam, located approximately 120 miles (190 km) south of Saipan, contains six “major” mangrove

species: *Avicennia marina*, *B. gymnorhiza*, *Lumnitzera littorea*, *Nypa fruticans*, *Rhizophora apiculata*, and *R. mucronata*. Additionally, three “minor” mangrove species also occur in Guam: *A. aureum*, *Excoecaria agallocha*, and *Xylocarpus moluccensis*. Historically, *Lumnitzera littorea*, *R. mucronata*, and *R. apiculata* were present on Saipan according to palaeoecological studies (Athens & Ward 2004), but now these mangrove species are no longer found within CNMI.

The AMME mangrove wetland has been subject to digging that could have altered subsurface water flow paths throughout its recent history. Alteration of subsurface flow paths could have occurred as a result of laying sewage lines and pipelines during World War II, remediation of pipelines, and backhoe-dug trenches for archeological surveys (e.g., Prasad 1997, McIntosh and Cleghorn 2001). Pacific Island Network staff observed work being conducted near the park that included trench digging in 2015.

Currently, the mangrove wetland is a mix of living, stressed, and dead trees, which have been declining since at least 2007 (Mike Gawel, per. com.). A 2019 Natural Resource Condition Assessment (Greene et al. 2019) rated the condition of the mangroves as “moderate with an unknown trend,” with low confidence in the determination. Mangrove deterioration is the most acute natural resource problem in the park. A road separating the mangrove wetland from the shoreline (Rt. 30 or Beach Road) included a culvert to maintain over-land tidal seawater exchange with the wetland. A qualitative evaluation by the United States Geological Survey (USGS) concluded that conveyance through the culvert is inadequate to promote tidal flooding (Krauss 2016). Blockage of this culvert may entirely restrict or disrupt surface flow of saltwater to the wetland (Figure 3). *Bruguiera gymnorhiza* crowns have also been damaged by typhoon Soudelor in 2015 and Yutu in 2018. Management efforts have included improving stormwater drain infrastructure, removing invasive plants, and closing an adjacent open dump. The small, constructed wetland then drains into a channel which crosses the park and drains at the shoreline. Planting of *B. gymnorhiza* has occurred along the artificially constructed wetland.

The objectives of this work were to examine data collected from monitoring the mangrove and constructed wetlands, particularly given the concerns about changes in observed surface water quality and vegetation conditions (Greene et al. 2019). To examine potential trends, we used data obtained over the last decade by the PACN focal terrestrial plant communities (Ainsworth et al. 2011), water quality protocol (Jones et al. 2011), groundwater dynamics (Izuka et al. 2011), and climate (Schlappa et al. 2011) protocols.

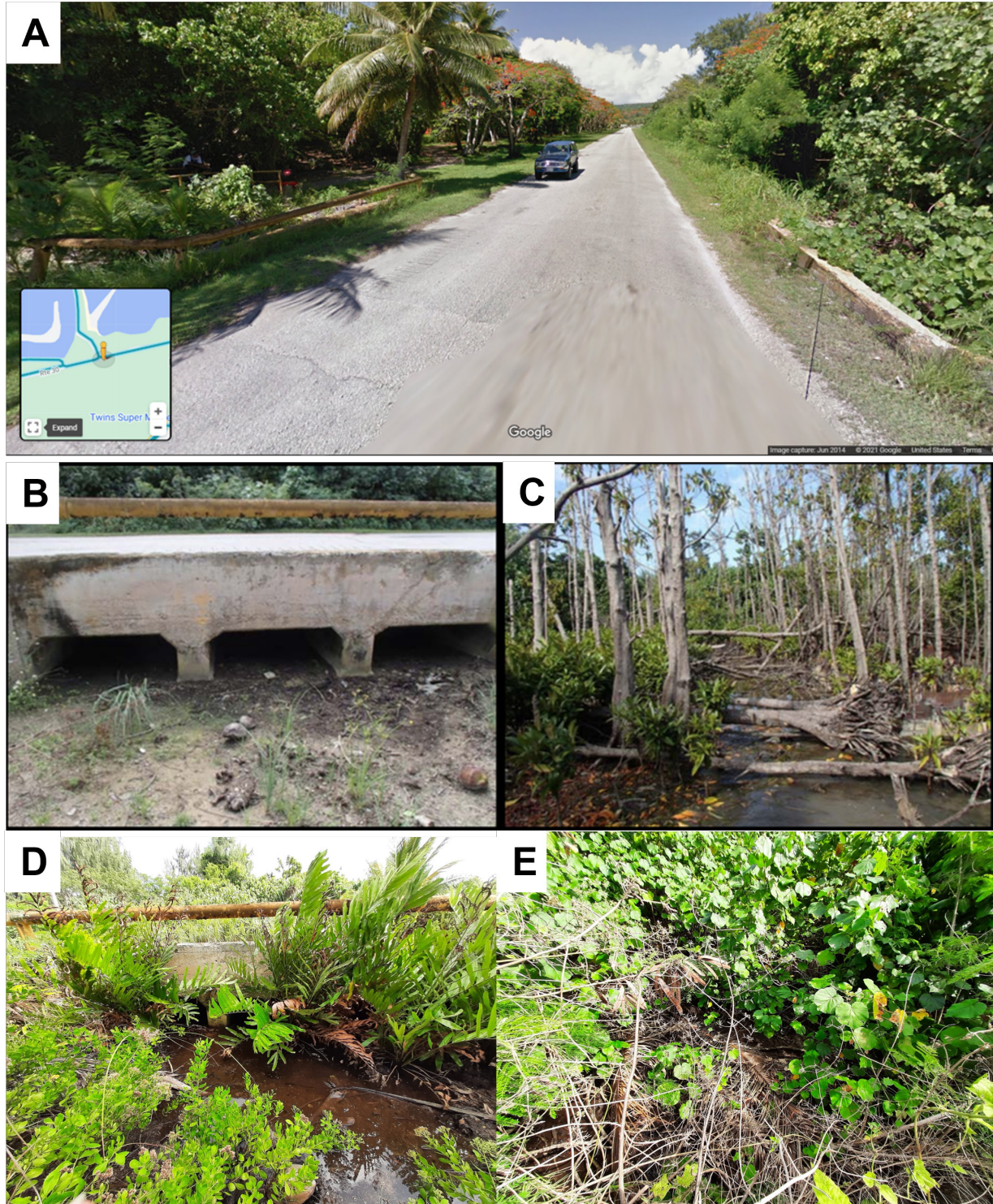


Figure 3. The culvert passing under Rt. 30. A) Culvert in 2014, with the mangrove wetland to the right in this view, and to the south in the inset. B) Seaward facing side of culvert in 2016 (Krauss 2016). C) Area with mangroves on seaward side of the culvert showing inundation in 2016 (Krauss 2016). D) Seaward facing side of culvert in 2023 (photo by A. Farahi). E) Mangrove wetland-facing area adjacent to culvert in 2023 (photo by A. Farahi).

Methods

Focal terrestrial plant communities

Methods are briefly presented here; please consult the protocols for additional information. Sampling within the AMME wetland consisted of ten 20 m x 50 m fixed-location plots (Figure 4). Each plot was monitored once in 2014 and again in 2019. In each plot, all species were recorded and all large trees with a diameter at breast height (dbh) ≥ 10 cm were measured. All small trees ($1 \text{ cm} \leq \text{dbh} < 10$ cm) and large tree ferns (dbh ≥ 10 cm) were recorded in 10 m x 25 m sub-plots, and all seedlings (dbh < 1 cm and height ≥ 50 cm) and small tree ferns (dbh < 10 cm), shrubs and vines (length ≥ 50 cm), were recorded in 2 m x 50 m sub-plots. The coverage of species in the understory was also measured along three 50 m transects located along the two long edges and the middle of the plot. Along these transects, the presence of species was recorded every 0.5 m if one or more species intercepted a 2m height pole (Ainsworth et al. 2011).



Figure 4. Vegetation monitoring plots within American Memorial Park visited in 2014 and 2019. Red circles highlight plots containing *B. gymnorhiza*.

Water Quality

Water samples and sonde measurements were collected from just below the surface (Jones et al. 2011). Water was filtered in the field using an in-line filter (2008–2012) or filter tower (2013–present) equipped with 0.47 μm GF/F filter. Each bottle was rinsed three times with filtered site-water. Three independent grab filtered 125 ml samples were taken at each site, placed in a cooler with ice in the field, transferred to a freezer on the day of collection, and kept frozen until delivery to the EPSCoR Analytical Laboratory at the University of Hawai‘i at Hilo (UHH), Marine Science Department, for analysis. Water samples were analyzed for nutrients. Analytes and their minimum detection limits included total phosphorus and total dissolved phosphorus (TP and TDP, 15.0 micrograms per liter or $\mu\text{g/L}$), total nitrogen and total dissolved nitrogen (TN and TDN, 0.070 milligrams per liter or mg/L), and nitrate + nitrite ($\text{NO}_3^- + \text{NO}_2^-$, 1.0 $\mu\text{g/L}$).

A multi-parameter sonde (YSI 6600) was calibrated a few days before commencing fieldwork. The sonde was used to measure depth (m), temperature ($^{\circ}\text{C}$), specific conductivity (SpC) in milli-Siemens per centimeter (mS/cm), pH, optical dissolved oxygen (ODO) in mg/L , chlorophyll in $\mu\text{g/L}$, and turbidity in nephelometric turbidity units (NTU), which is equivalent to FNU turbidity units (USGS, unpublished). The sonde also reported salinity in parts-per-thousand (ppt) calculated from SpC, and dissolved oxygen in percent saturation (% ODOsat) calculated from ODO and temperature. Salinity and ODOsat are reported here. At each site, the sonde recorded readings every 30 seconds for five to ten minutes. Between sites the sonde was stored with probes immersed in freshwater.

Site names for water quality sampling used by PACN were formatted to designate fixed (F) sites, park (e.g., AMME), a number (e.g., 01), and habitat type (“fw” for freshwater). Four fixed sample sites were sampled in the mangrove wetland and four fixed sites were sampled in the constructed wetland. Because the constructed wetland was made to mitigate high nutrient concentrations before stormwater drain effluent entered the ocean, the linear pattern of nutrients along the length of the wetland channel was examined. Note that FAMME01_fw is located at the outlet of the stormwater drain, FAMME09_fw is located within an area typically overgrown with emergent aquatic plants, and that the channel flows north-easterly to the shoreline (Figure 5).

Water quality data from all sites in the mangrove wetland was pooled for analysis due to some sites not containing water during some sampling events. Analysis was conducted for surface water quality to detect trends and examine possible causes of stress to mangroves. Choice of statistical methods for trend analysis was influenced by the nature of the data. The primary factors affecting choice of statistical tests were data censoring, seasonality, and sampling design. While nutrients, chlorophyll, and turbidity had the potential to be censored, only TDP had any censoring (values below the detection limit) at a rate of 4%. Due to low censoring trends data were analyzed by linear mixed model (VanLeeuwen et al. 1996). Data analysis was completed using R v.3.5.0 (R Core Team 2017). Mixed models were evaluated with the *lmer* function in the *lmerTest* package (Kuznetsova et al. 2016). Measurements from a series of water samples or sonde readings taken at an individual site within minutes of each other were assumed to be normally distributed around the true value of the parameter, as the parameter should not change appreciably over the short time required to take multiple samples or readings. Sample variation should be most influenced by external factors.



Figure 5. The constructed wetland at American Memorial Park. Sewage-contaminated effluent enters at FAMME01_fw and flows to the shoreline.

Seasonality was assessed for parameters prior to performing linear mixed model tests to determine if seasonal factors should be included in the model. Seasonality was evaluated using the multiple linear regression model function `lm` in R. Wet and dry seasons were defined November through April (dry season) and May through October (wet season). Two models were considered: change over time and the effect of calendar-year quarters (q1, q2, q3), and change over time and the effect of the wet and dry seasons. It is necessary to exclude a time-interval, e.g., “q4” and “dry”, in tests as these serve as references for the model. Season was included in mixed model analyses for trend when the effect of quarter or season was identified as significant ($\alpha < 0.05$) in the seasonality `lm` models. The linear mixed model allowed for effects to be divided into fixed factors that impact the mean structure (e.g., season or quarter) and random factors that impact the variance structure (year). Variance was estimated by restricted maximum likelihood (REML), which is appropriate for unbalanced data (Piepho and Ogutu 2002). Error for the random factors were partitioned into different sources of variation (e.g., year-to-year, Starcevich 2013). The model included fixed factors for time (WYEAR), and seasonal factors when deemed appropriate by the seasonality analysis (Quarter and/or Season).

Output of the lmer function included effect estimates and t-statistics for each fixed factor in the model and for the interaction between the time component WYEAR and quarter when seasonality was considered. Full output of the linear mixed model displays the slope of the effect of the factor and the standard error for that value. Full output also displays the intercept estimate and its statistical significance, indicating if the slope of the factor is significantly different from zero. Additionally, p-values are estimated using Satterthwaite's approximations (Kuznetsova et al. 2016). Effect estimates are not given for random factors considered unobserved variables (Bates 2010, Bates et al. 2014).

Climate

A Remote Automatic Weather Station (RAWS) was installed at AMME in 2014 (Schlappa et al. 2011). This station measures and records ambient temperature, relative humidity, precipitation, solar radiation, wind speed and direction, barometric pressure, and soil temperature. Instantaneous values are recorded every ten minutes and stored locally, averaged hourly values are transmitted via GOES satellite uplink. Fine-scale (ten minute) data are downloaded physically during biannual sites visits. Data storage, QA/QC, and technical assistance are provided by the Desert Research Institute/Western Regional Climate Center (DRI/WRCC) through a cooperative agreement with NPS. Certified data are publicly available through both the DRI/WRCC website and the NPS Integrated Resource Management Applications (IRMA) Portal. Representation of El Niño conditions was defined by the NOAA Oceanic Niño Index (NOAA 2020).

Groundwater dynamics

Conductivity-temperature-depth (CTD) probes were deployed in two groundwater monitoring wells in the park (Izuka et al. 2011). The groundwater monitoring wells had been installed by a study not affiliated with PACN prior to the creation of the PACN groundwater protocol. These wells may have been the pre-existing wells used by Perreault (2007). Both vented and non-vented CTDs (In-Situ Aqua TROLL 200) with an independent barometer were used. Groundwater monitoring was conducted from 2010 to 2016. Groundwater monitoring data was processed and evaluated by the U.S. Geological Survey, Pacific Islands Water Science Center, Honolulu, HI.

Results

Focal terrestrial plant communities

Bruguiera gymnorhiza were present in 4 out of 10 monitoring plots (02, 04, 06, 09) located around the interior of the AMME mangrove wetland (Figure 4). From 2014 to 2019, three of the four plots containing *B. gymnorhiza* trees experienced substantial declines in large (>10 cm DBH) individuals (Figure 6). The greatest declines were witnessed within Plot 6, where 57 large trees died within the 1000 m² sampling area, translating to a loss of 570 large trees per ha. Medium-sized trees (5<10 cm DBH) and seedling-sized (<1 cm DBH) also declined in several plots, while small-sized *B. gymnorhiza* (1<5 cm DBH) showed signs of recruitment. Three plots showed increased *B. gymnorhiza* cover within the understory (0–2 m from the ground; Figure 7), likely related to the increased density of small-sized mangrove. Within the four mangrove plots, six plant species showed greater increases in cover than the mangrove, these species included: the native langyao fern (*Acrostichum aureum*); the native pago tree (*Hibiscus tiliaceus*), a facultative wetland species with a low-sprawling form; the non-native Indian pluchea shrub (*Pluchea indica*); a non-native vine (*Mikania* sp., likely *Mikania scandens*); an aggressive vine-like native shrub, gasoso (*Colubrina asiatica*), recognized as highly invasive in other locales; and a native banago vine (*Jasminum marianum*). Total understory cover of both native and non-native plants increased within the mangrove plots (Figure 8), presumably from the increased light availability of downed canopy trees during typhoons Soudelor in 2015 and Yutu in 2018. Comparison of photographs taken in 2014 and 2019 illustrate the change in the overall character of vegetation within three of the plots containing mangrove (Figures 9–11).

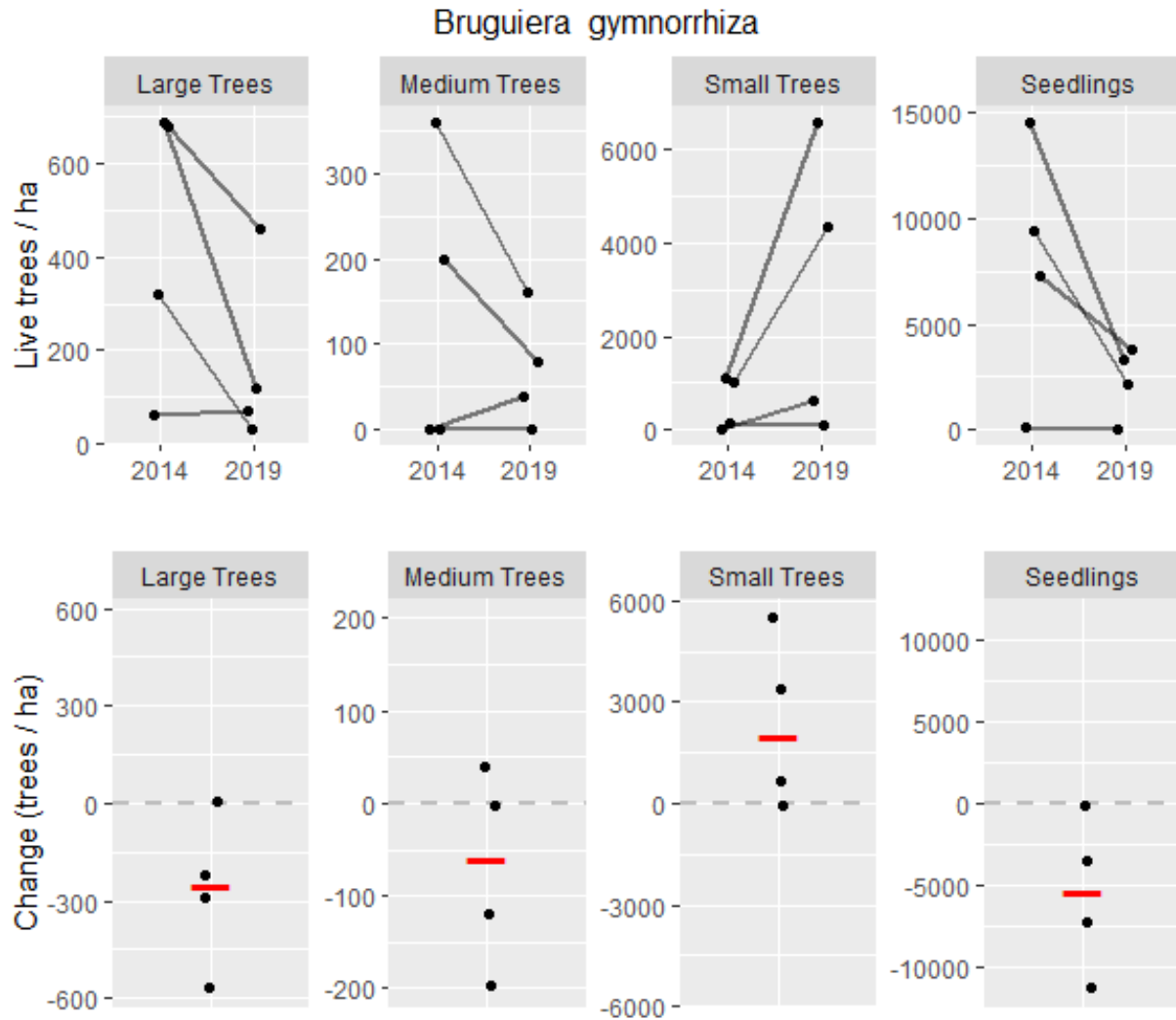


Figure 6. Change in mangrove density in American Memorial Park, Saipan. Top row: Paired-plot measurements of *B. gymnorhiza* density by size class. Bottom row: Points represent the change in tree density for each plot. The red line represents the calculated median for the four plots (Median values: Large Trees = -255, Medium Trees = -60, Small Trees = 2000, Seedlings = -5350). *Bruguiera gymnorhiza* was present in four out of the ten vegetation monitoring plots sampled in 2014 and revisited in 2019. Size classes include: Large trees (> 10 cm DBH), Medium Trees (5<10 cm DBH), Small Trees (1<5 cm DBH), and Seedlings (<1 cm DBH).

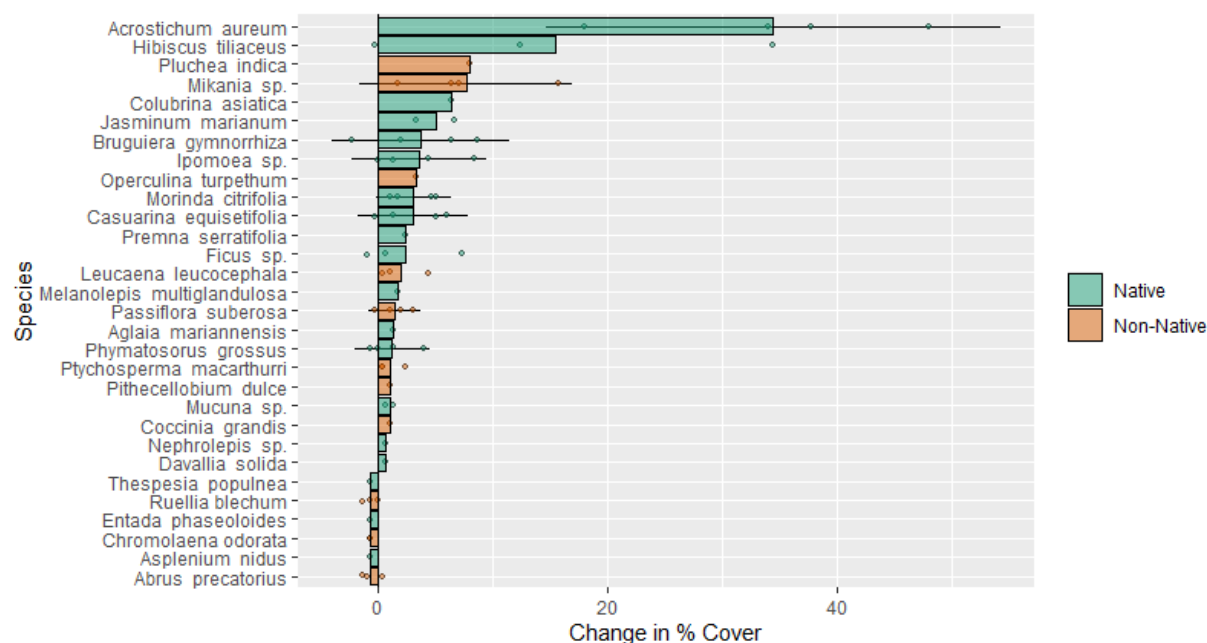


Figure 7. Mean change in understory cover (0–2 m from ground) by species across the four plots in which *B. gymnorhiza* was present. Error bars represent the 95% confidence interval. Error bars were not calculated when the species was detected in three plots or less. Points represent change within one plot.

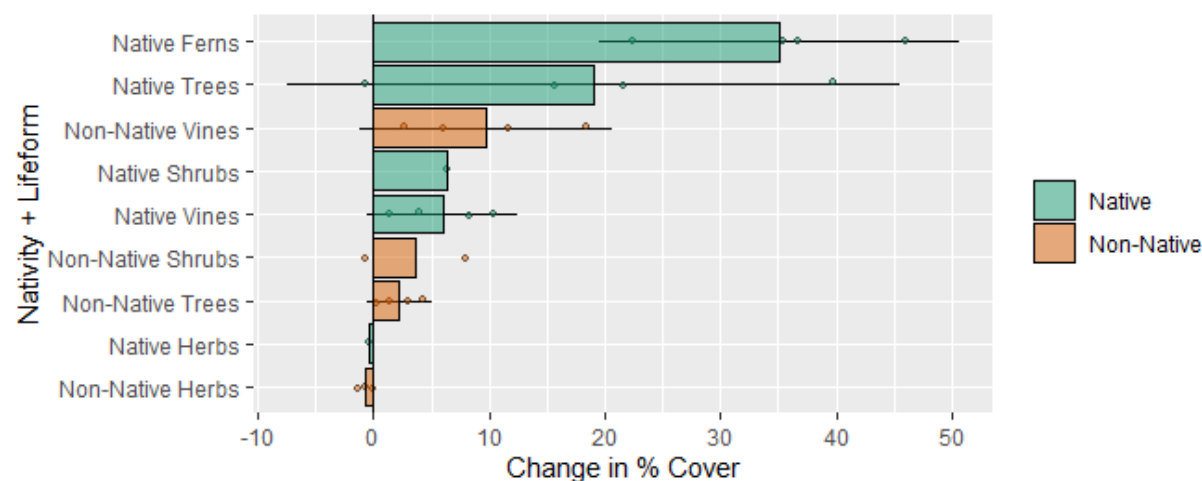


Figure 8. Mean change in understory cover (0–2 m from ground) by plant lifeform across the four plots in which *B. gymnorhiza* was present. Error bars represent the 95% confidence interval. Error bars were not calculated when the species was detected in three plots or less. Points represent change within one plot.



Figure 9. AMME vegetation plot 09 in 2014 (top) and 2019 (bottom) showing decrease in canopy cover and increase in understory density.

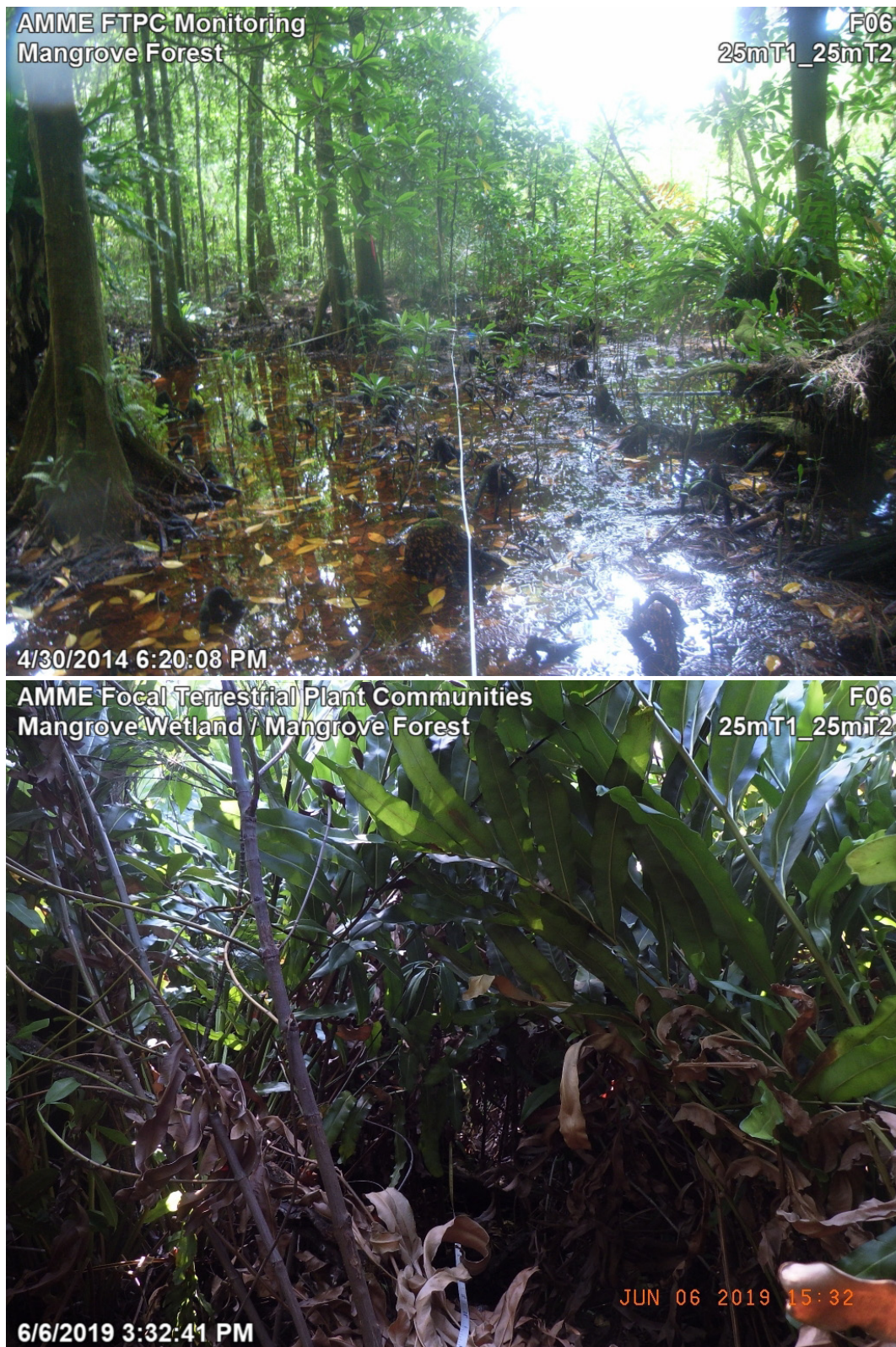


Figure 10. AMME vegetation plot 06 in 2014 (top) and 2019 (bottom) showing decrease in open water and increase in understory density.



Figure 11. AMME vegetation plot 04 in 2014 (top) and 2019 (bottom) showing decrease in open water and increase in understory density, in this case *A. aureum*).

Mangrove Wetland Surface Water Quality

Seasonality was observed in oxygen, salinity, and temperature and thus included in the linear mixed model for those parameters (Table 1). Seasonality was not detected in other parameters. Time series for surface water quality parameters are presented in Figure 12. A trend of decreasing salinity was detected from 2009 to 2018 (Table 2, Figure 13). Although trend analysis showed a statistically significant change over time in TN and TP, these changes were driven by one datum near the end of the study period for both analytes, thus these patterns are not trends (Figure 13). Ranges of water quality parameters in the mangrove wetland are presented in Table 3.

Table 1. Results of seasonality tests for water quality parameters in the mangrove wetland of American Memorial Park. Significant results shaded in gray.

Parameter	Wet / Dry				Quarterly					
	R ²	Main (<i>p</i>)	Time (<i>p</i>)	Wet(<i>p</i>)	R ²	Main (<i>p</i>)	Time (<i>p</i>)	Q1 (<i>p</i>)	Q2 (<i>p</i>)	Q3 (<i>p</i>)
TDN	0.15	0.22	0.11	0.41	0.49	(<i>p</i> < 0.05)	0.11	0.62	0.07	0.61
TDP	0.11	0.35	0.28	0.29	0.32	0.13	0.27	0.08	0.89	0.41
NO ₃ +NO ₂	0.14	0.26	0.22	0.31	0.32	0.16	0.20	0.12	0.92	0.95
TP	0.30	0.24	0.21	0.27	0.56	0.23	0.22	0.11	0.87	0.90
TN	0.31	0.22	0.23	0.22	0.52	0.23	0.32	0.17	0.81	0.93
Chlorophyll	0.09	0.45	0.79	0.22	0.07	0.88	0.81	0.72	0.99	0.71
Oxygen	0.30	(<i>p</i> < 0.05)	(<i>p</i> < 0.05)	0.12	0.44	0.03	(<i>p</i> < 0.05)	0.15	0.16	(<i>p</i> < 0.05)
pH	0.06	0.49	0.25	0.74	0.17	0.40	0.26	0.17	0.48	0.19
Salinity	0.24	0.05	(<i>p</i> < 0.05)	0.12	0.47	(<i>p</i> < 0.01)	0.07	(<i>p</i> < 0.05)	(<i>p</i> < 0.01)	(<i>p</i> < 0.05)
Temperature	0.31	(<i>p</i> < 0.05)	0.21	(<i>p</i> < 0.05)	0.50	(<i>p</i> < 0.01)	0.09	0.08	0.31	0.51
Turbidity	(<i>p</i> < 0.05)	0.65	0.39	0.84	0.21	0.38	0.33	0.15	0.53	0.17

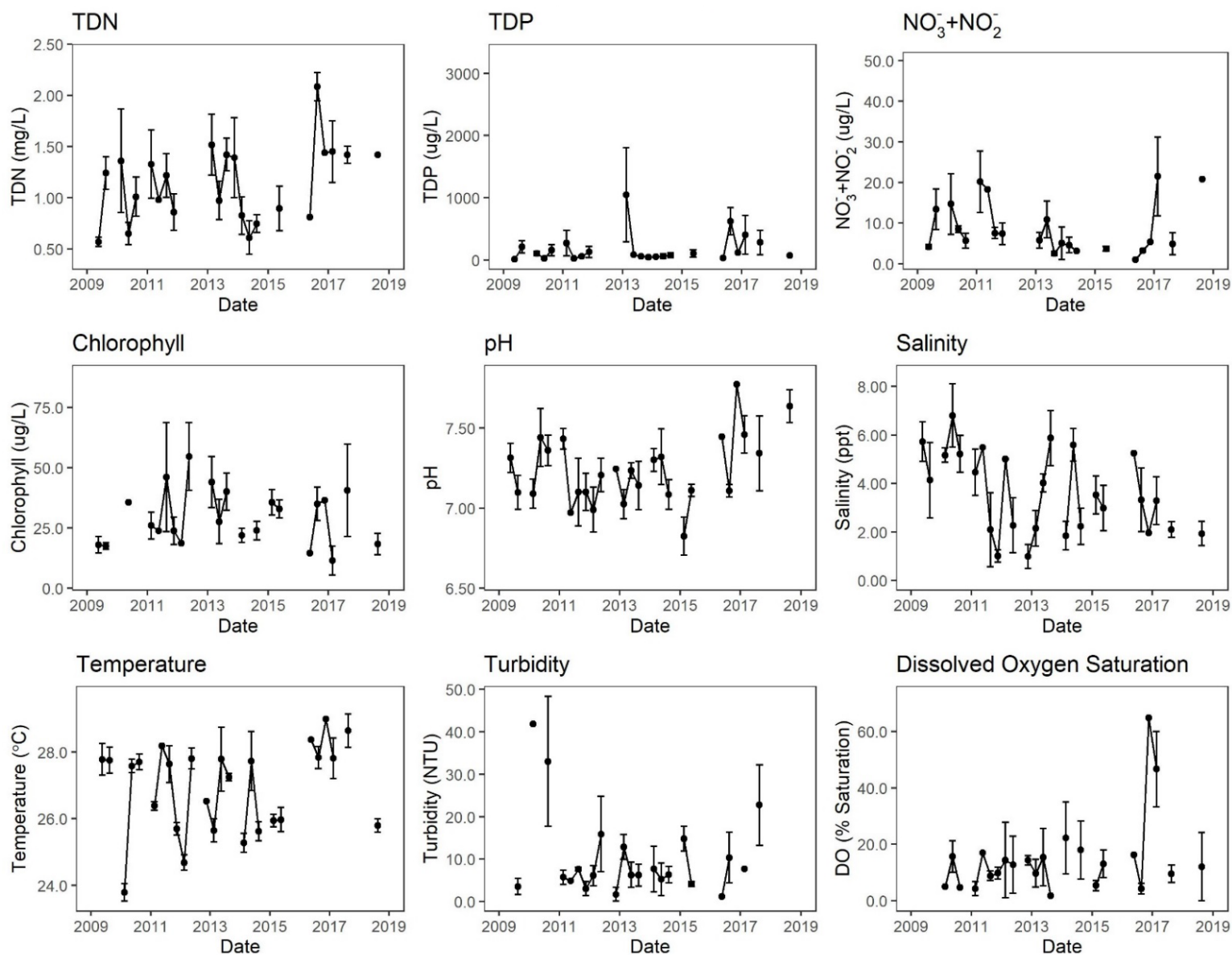


Figure 12. Surface water quality in the mangrove wetland of American Memorial Park.

Table 2. Summary of trend tests in the mangrove wetland of American Memorial Park. Significant results shaded in gray.

Parameter	Seasonality	Censoring (%)	Time (<i>p</i>)	Quarter (<i>p</i>)	Time:Quarter (<i>p</i>)	Wet (<i>p</i>)	Trend
NO ₃ ⁻ + NO ₂ ⁻	N	0	0.63	0.62	0.37	–	None
TDP	N	4	0.66	0.54	0.97	–	None
TDN	N	0	0.98	0.30	0.39	–	None
TP	N	0	(<i>p</i> < 0.01)	(<i>p</i> < 0.05)	(<i>p</i> < 0.05)	–	None
TN	N	0	(<i>p</i> < 0.01)	(<i>p</i> < 0.05)	(<i>p</i> < 0.05)	–	None
Chlorophyll	N	0	0.26	0.25	0.22	–	None
Turbidity	N	0	0.20	0.09	0.17	–	None
pH	N	0	0.41	0.34	0.29	–	None
Oxygen	Y	0	0.08	0.45	0.43	0.14	None
Temperature	Y	0	0.40	0.50	0.49	(<i>p</i> < 0.0001)	None
Salinity	Y	0	(<i>p</i> < 0.05)	0.21	0.33	(<i>p</i> < 0.01)	Decreasing

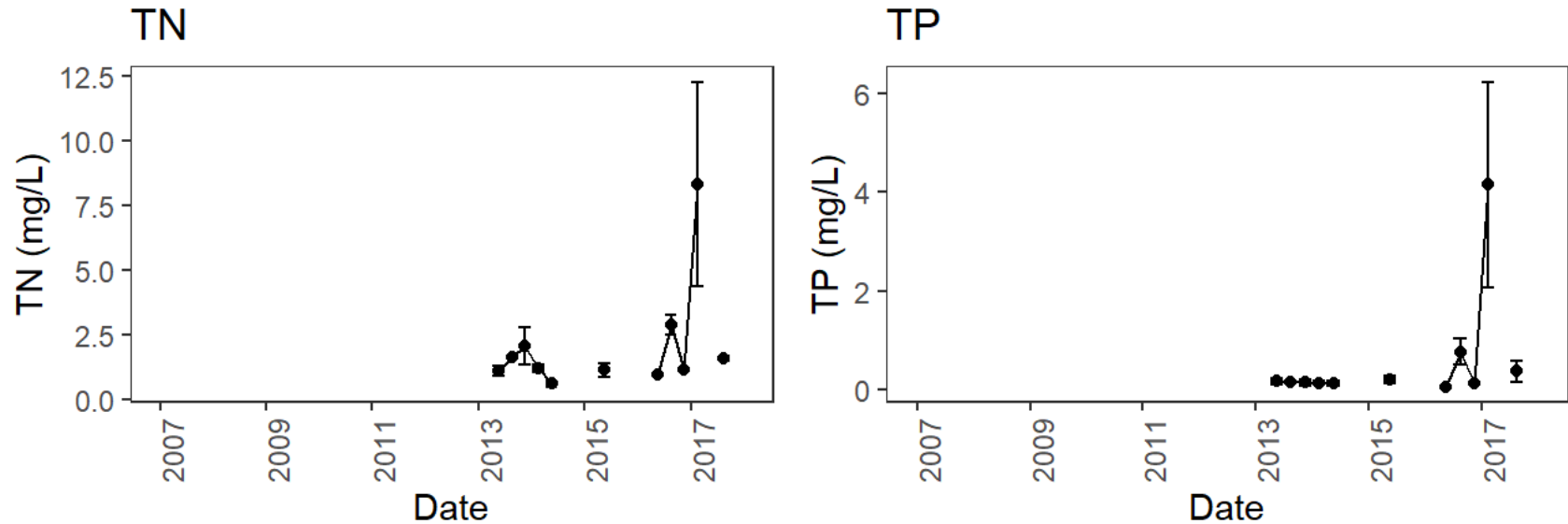


Figure 13. Total nitrogen (TN) and total phosphorus (TP) in the mangrove wetland of American Memorial Park.

Table 3. Ranges of water quality parameters in the mangrove wetland of American Memorial Park 2009–2018.

Statistic	TDN (mg/L)	TDP (mg/L)	NO ₃ ⁻ + NO ₂ ⁻ (mg/L)	TN (mg/L)	TP (mg/L)	Chlorophyll (µg/L)	Oxygen Sat. (%)	pH	Salinity (ppt)	Temperature (°C)	Turbidity (NTU)
n	67	67	63	29	29	70	70	88	88	88	70
Maximum	2.4	3.3	0.05	18.9	8.6	88.3	174.8	8.9	25.9	37.7	48.4
3rd Quartile	1.4	0.16	0.01	1.8	0.4	34.7	19.9	7.5	5.5	28.2	12.8
Mean	1.1	0.21	0.09	2.4	0.8	27.4	16.8	7.3	4.2	27.4	10.6
Median	1.1	0.07	0.005	1.4	0.2	24.1	9.8	7.2	3.7	27.2	7.8
1st Quartile	0.7	0.03	0.003	0.9	0.1	15.5	4.2	7.1	2.0	26.1	3.8
Minimum	0.4	0.02	0.001	0.5	0.1	0.1	1.0	6.6	0.0	23.5	0.1

Constructed Wetland Surface Water Quality

Seasonality was detected for temperature in surface waters in the constructed wetland (Table 4). No trends in surface water quality were detected (Table 5, Figures 14–15). Ranges for water quality parameters in the constructed wetland observed from all sites are presented in Table 6. Mean TDN in the constructed wetland was more than twice that of the mangrove wetland, with maximum observed TDN more than 15x higher. TDP values were comparable with the mangrove wetland. Mean and maximum nitrates in the constructed wetland were twice that of the mangrove wetland. Temperature, pH, and dissolved oxygen were comparable between wetlands. The constructed wetland had higher turbidity and lower salinity than the mangrove wetland. Nutrient concentrations were generally much higher at the stormwater drain outflow releasing effluent into the constructed wetland compared to downstream sites; with the exception of nitrates (Figures 5, 16). Other parameters were generally similar between sites along the channel, with the exception of the downstream site FAMME_03 which displayed greater and highly variable salinity (Figure 14).

Table 4. Results of seasonality tests for water quality parameters in the constructed wetland of American Memorial Park. Significant results shaded in gray.

Parameter	Wet / Dry				Quarterly					
	R ²	Main(p)	Time (p)	Wet (p)	R ²	Main (p)	Time (p)	Q1 (p)	Q2 (p)	Q3 (p)
TDN	0.14	0.16	0.17	0.20	0.27	0.12	0.18	0.09	0.07	0.04
TDP	0.18	0.09	0.09	0.18	0.24	0.18	0.11	0.30	0.13	0.10
NO ₃ +NO ₂	0.09	0.35	0.16	0.77	0.29	0.11	0.17	0.03	0.16	0.46
TP	0.15	0.33	0.21	0.37	0.22	0.53	0.22	0.39	0.44	0.20
TN	0.09	0.33	0.21	0.34	0.23	0.41	0.36	0.13	0.29	0.15
Chlorophyll	0.13	0.18	0.07	0.73	0.14	0.45	0.07	0.68	0.53	0.87
Oxygen	0.14	0.12	0.56	0.06	0.14	0.41	0.42	0.39	0.97	0.38
pH	(p < 0.01)	0.94	0.79	0.85	(p < 0.05)	0.85	0.84	0.33	0.39	0.37
Salinity	0.10	0.21	0.39	0.10	0.08	0.69	0.49	0.48	0.32	0.20
Temperature	0.35	(p < 0.01)	0.67	(p < 0.0001)	0.43	(p < 0.01)	0.77	0.24	0.08	(p < 0.05)
Turbidity	0.09	0.29	0.21	0.32	0.22	0.17	0.11	0.78	0.07	0.95

Table 5. Summary of trend tests in the constructed wetland of American Memorial Park. Significant results shaded in gray.

Parameter	Seasonality	Censoring (%)	Time (<i>p</i>)	Quarter (<i>p</i>)	Time:Quarter (<i>p</i>)	Wet (<i>p</i>)	Trend
NO ₃ ⁻ + NO ₂ ⁻	N	0	0.60	0.38	0.95	–	None
TDP	N	0	0.96	0.73	0.40	–	None
TDN	N	0	0.51	0.65	0.20	–	None
TP	N	0	0.64	0.60	0.83	–	None
TN	N	0	0.64	0.60	0.83	–	None
Chlorophyll	N	0	0.87	0.37	0.93	–	None
Turbidity	N	0	0.74	0.86	0.21	–	None
pH	N	0	0.86	0.91	0.94	–	None
Oxygen	N	0	0.11	0.07	0.19	–	None
Temperature	Y	0	0.73	0.84	0.47	(<i>p</i> < 0.0001)	None
Salinity	N	0	0.14	0.46	0.13	–	None

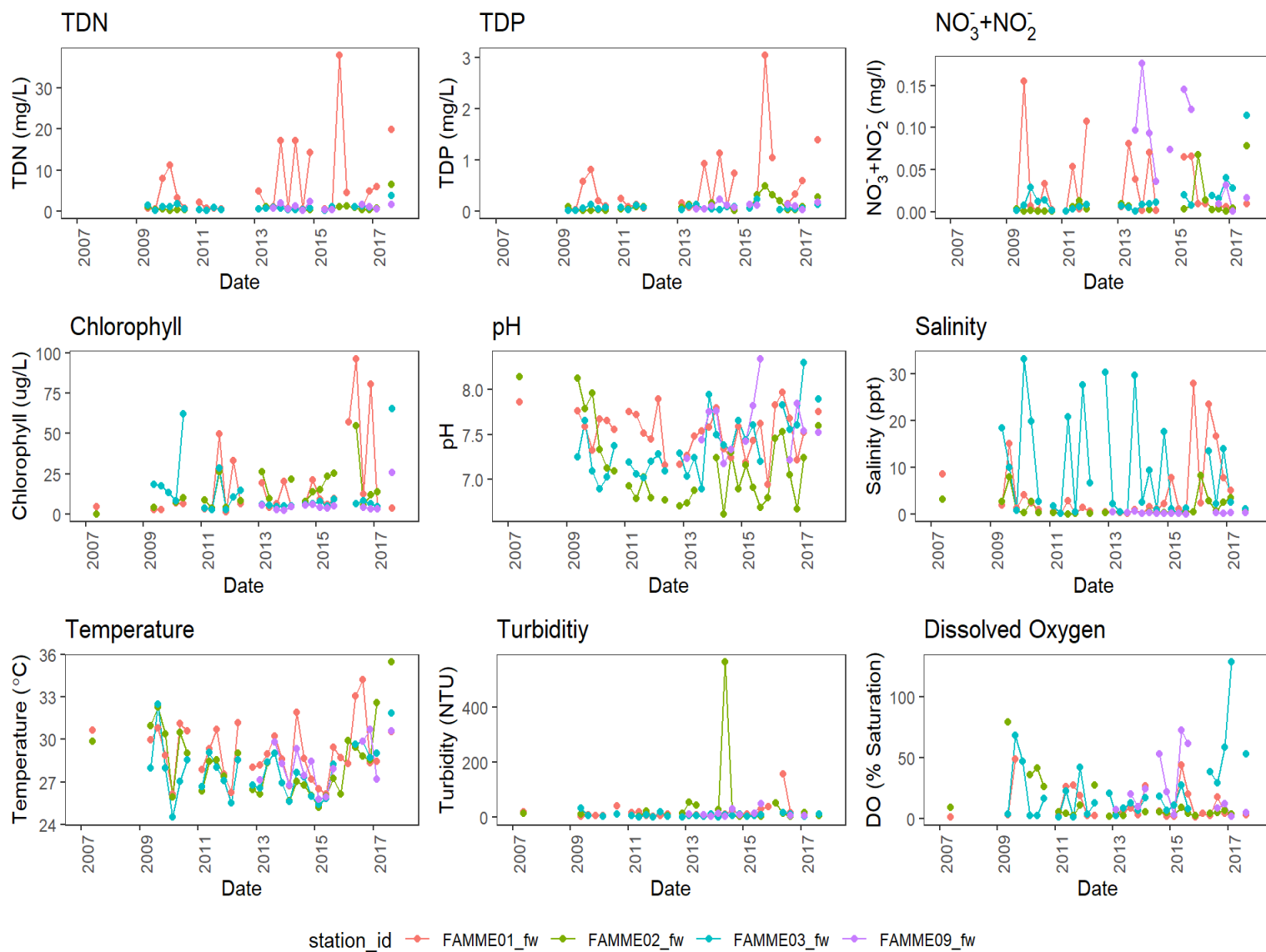


Figure 14. Surface water quality in the constructed wetland of American Memorial Park.

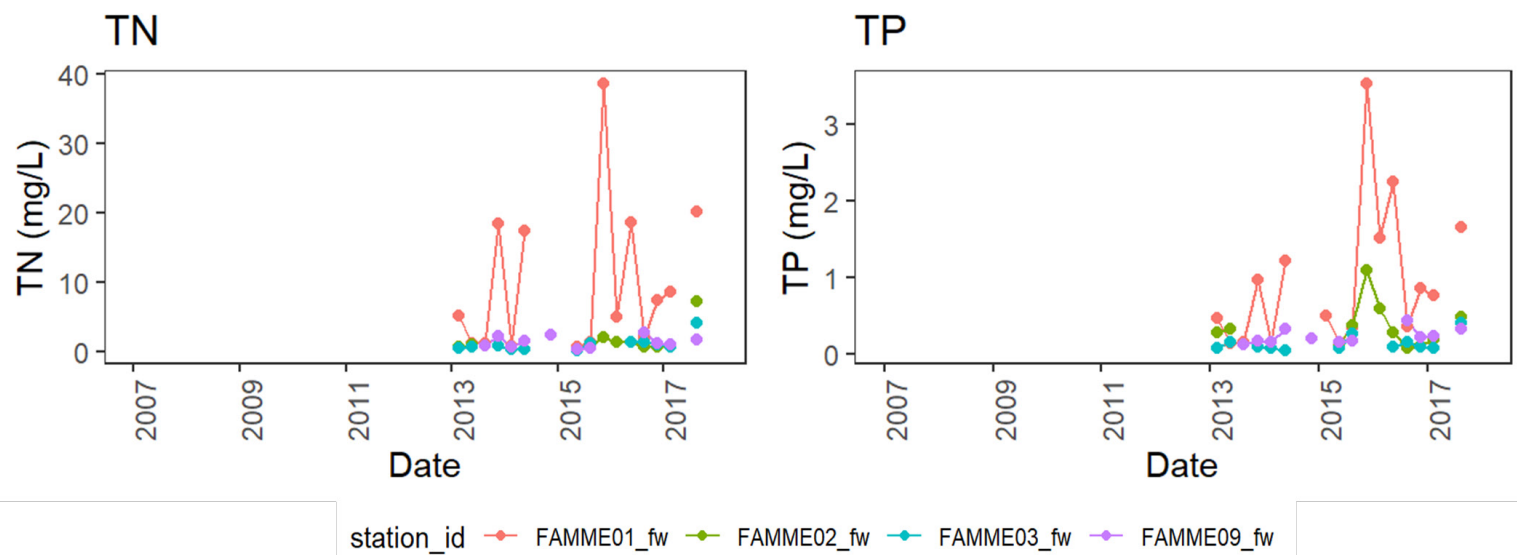


Figure 15. Total nitrogen (TN) and total phosphorus (TP) in the constructed wetland of American Memorial Park.

Table 6. Ranges of water quality parameters in the constructed wetland of American Memorial Park 2009–2018.

Statistics	TDN (mg/L)	TDP (mg/L)	NO ₃ ⁻ + NO ₂ ⁻ (mg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	Oxygen Sat. (%)	pH	Salinity (ppt)	Temperature (°C)	Turbidity (NTU)
n	87	87	80	49	50	87	93	103	104	104	88
Maximum	37.8	3.1	0.17	38.5	3.5	96.5	129.1	8.3	33.3	35.5	564.4
3rd Quartile	1.4	0.17	0.03	2.4	0.5	14.4	22.7	7.7	3.0	29.8	15.1
Mean	2.5	0.21	0.027	3.9	0.5	13.7	16.7	7.4	4.4	28.4	20.1
Median	0.8	0.09	0.01	1.3	0.2	6.6	6.6	7.4	0.9	28.4	9.3
1st Quartile	0.5	0.05	0.002	0.7	0.1	4.4	2.6	7.1	0.3	26.8	5.1
Minimum	0.2	0.02	0.001	0.2	0.1	0.1	0.9	6.6	0.0	24.5	0.1

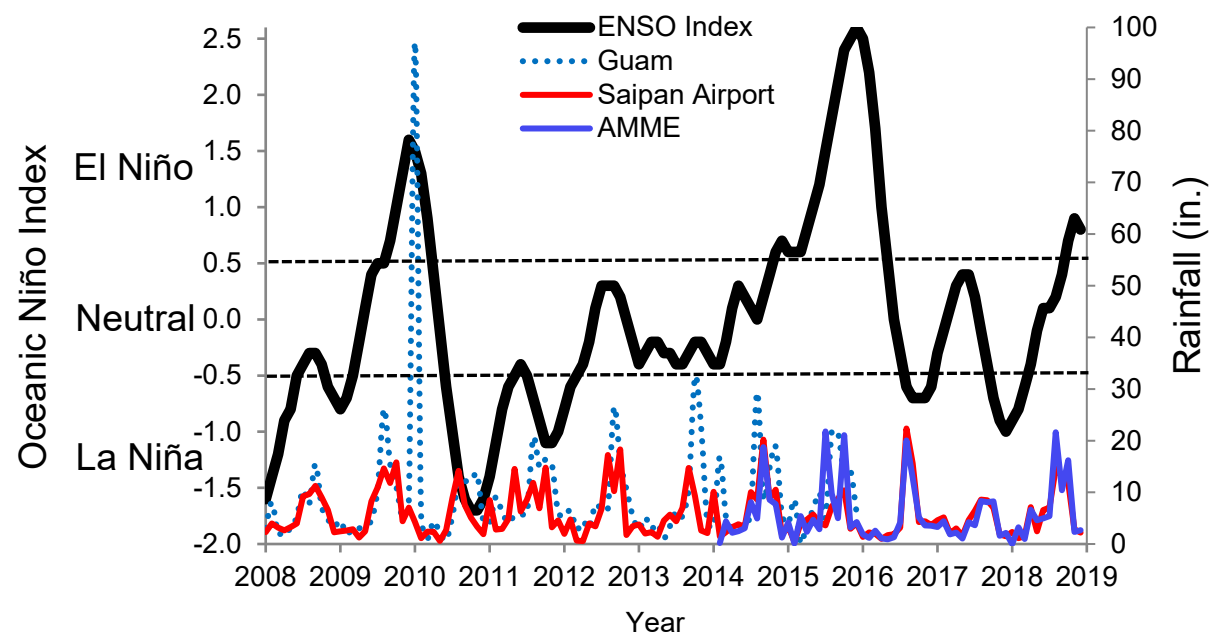


Figure 16. El Niño Southern Oscillation (ENSO) cycles and precipitation in Guam and the Commonwealth of the Northern Marianas Islands.

Climate

Over the course of the study period two ENSO cycles occurred (Figure 16). According to the NOAA Oceanic Niño Index, the ENSO cycle moved from La Niña conditions in 2008 to El Niño in 2009 to 2010, back to La Niña, stayed in the neutral condition from 2012 to 2014, and moved into El Niño in 2015 and back to La Nina in 2016 (NOAA 2020). Rainfall in Guam and Saipan were very similar across the study period, with two exceptions: a storm event caused high rainfall in Guam in 2010 that was not experienced in Saipan, and AMME experienced two rainfall events in 2015 that were not recorded at the Saipan Airport to the south of the park. Rainfall was much lower in 2017 at AMME compared to other years. Many surface water sites were dry in 2017 and 2018, which is reflected in fewer surface water data points visible in time series graphs during those times (Figure 12).

Groundwater

Data from CTD probes was determined to be not reliable due to failures of wells and monitoring equipment. The AMME wells were installed prior to the implementation of monitoring protocols by other researchers, most likely Perreault (2007), and used opportunistically by NPS. The wells had not been constructed with sealed vaults, allowing surface runoff to flow into the vaults and wells. The infiltration of surface water caused repeated equipment failures and loss of data. The United States Geological Survey (USGS) evaluated CTD data and determined that due to gaps in the time series the dataset was not reliable (unpublished). Hence no groundwater specific-conductance data, and therefore no groundwater salinity data, was available.

Discussion

Results suggested at least two different disturbance types affecting the mangrove wetland: 1) an ecosystem-level shift away from a functioning mangrove state due to altered water quality and hydrology, and 2) the disturbance to the mangrove canopy caused by the 2015 typhoon Soudelor and the 2018 typhoon Yutu. The high understory cover of non-mangrove species within the mangrove wetland points to an ecosystem shift suggesting salinity and/or water availability conditions have been altered because, in general, it is difficult for non-mangrove plants to invade a functioning mangrove system (Lugo 1999).

Focal terrestrial plant communities

From 2014 to 2019, large declines in *B. gymnorhiza* were documented in each tree size class, except small trees with a dbh between 1 and 5 cm (Figure 6). Stands of mature (medium and large) *B. gymnorhiza* previously supported a tall, closed forest canopy with a relatively open understory in 2014 (Figures 9–11). Certainly, typhoons Soudelor and Yutu caused numerous deaths to the mature mangrove trees across the area, which thinned the forest canopy resulting in increased light availability which stimulated growth in the understory (0–2 m from the ground). The natural recovery and regeneration potential for the mangrove forest within the mangrove wetland depends upon both the abiotic factors discussed above and the following biotic factors: 1) the current *B. gymnorhiza* population structure, 2) interspecific competition, 3) propagule and seedling predation, 4) mangrove species diversity, 5) population distribution, each discussed in more detail below.

Bruguiera gymnorhiza population structure

Using density measurements by size class, several inferences can be made about the current *B. gymnorhiza* population (Figure 6). First, the median increase of 2000 small trees per hectare provides some evidence that natural regeneration was indeed occurring between 2014 and 2017. However, this statistic only includes plots with *B. gymnorhiza* already present ($n = 4$). Indeed, 6 of the 10 total monitoring plots in the mangrove wetland did not contain any *B. gymnorhiza* individuals. Therefore, propagule availability is already largely restricted to the center of the mangrove wetland where *B. gymnorhiza* is currently present (Figure 4). Second, given the large reduction in mature *B. gymnorhiza* trees from the recent typhoons, future propagule availability is also expected to decline, thus reducing future seedling densities, at least until the adult canopy can recover to prior numbers. Indeed, vegetation monitoring results noted a dramatic decline in *B. gymnorhiza* seedling density (median loss of 5350 seedlings per hectare), however, we cannot be certain if this reduction is a direct result of mature tree propagule loss or other factors (e.g., environment, interspecific competition). Bosire et al. (2008) referenced a minimum of 2500 well-distributed seedlings per hectare was needed for adequate natural mangrove regeneration in Malaysia. In 2019, the 4 plots with *B. gymnorhiza* present showed densities of 3800, 3300, 2200, and 0 seedlings per hectare (Plots 09, 04, 06, 02, respectively). Therefore, any further reduction in seedling densities could make natural regeneration unobtainable.

Interspecific competition

Percent cover measurements for the mangrove wetland understory (0–2 m from the ground) provide insights to interspecific competition (Figure 7). Again, focusing on the 4 plots where present, *B. gymnorhiza* cover increased in the understory, although at much lower rates than some other species. This increase in *B. gymnorhiza* is likely due to the increased density of small trees (DBH 1–5 cm) as all other size classes showed declines in density measurements from 2014 to 2019. However, because cover measurements, seedling density, small/medium tree density, and large tree density are sampled differently (see Methods), individuals or cohorts cannot be tracked year to year (e.g., a measured seedling in 2014 which grows and becomes a small tree might not be measured again in subsequent monitoring because of different sampling localities within a plot). Percent cover measurements show that *B. gymnorhiza* is competing with several aggressive plants in the understory, six of which showed mean increases greater than its own. These other species include native and non-native vines (*Mikania*, *Colubrina asiatica*, *Jasminum marianum*, *Ipomoea*, *Operculina turpethum*, *Passiflora suberosa*, *Coccinia grandis*), the sprawling beach hibiscus (pāgo, *Hibiscus tiliaceus*), the non-native sourbush (*Pluchea indica*), and the native langayao fern (*Acrostichum aureum*). These species (both native and non-native) are aggressive and can become weedy in disturbed sites. For example, Hamilton and Snedaker (1984) list several studies confirming poor mangrove regeneration in *A. aureum* dominated areas, including one study specifying inadequate mangrove regeneration whenever *A. aureum* occupied 60 percent or more of the area. Indeed, in the Matang mangroves of Malaysia, *A. aureum* thickets are routinely removed by the Forestry Department prior to planting *Rhizophora* mangrove trees (Chan 1989). While *A. aureum* is often found in mangrove ecosystems, researchers have debated whether it should even be considered a “minor” mangrove taxon (thus still considered a mangrove species) or removed from the mangrove list and instead considered a mangrove “associate.” Jayatissa et al. (2002) argue that it is a mangrove “associate” because (1) it is not restricted to mangrove ecosystems in some regions, (2) it is not woody like other minor mangrove components, and (3) it is often considered a pest by local mangrove managers and is rigorously removed when rehabilitating sites. Dahdouh-Guebas et al. (2005) used *A. aureum* as an example of “cryptic ecological degradation” in which a minor species dominates the mangrove at the expense of the major species without a noticeable change in spatial extent of the mangrove system but ultimately resulting in altered wildlife habitat and ecosystem services (e.g., protection from storms, preventing soil erosion, mitigation from sea level rise). Nevertheless, *A. aureum* is recognized as native to the region and was found in very high abundance within Holocene sediment cores taken within the mangrove wetland, indicating an *Acrostichum*-dense environment was present along with *B. gymnorhiza* prior to World War II (Jarzen and Dilcher 2009). *Acrostichum aureum* is highlighted here because it showed the greatest increase in cover within the four *B. gymnorhiza* plots (Figure 7) and thus expected to have high interspecific competition with *B. gymnorhiza*.

Plots on the eastern and western edges of the mangrove wetland, where *B. gymnorhiza* was not found (Figure 4), were dominated by non-native grasses, vines, trees, shrubs, and herbs in the understory, all of which are non-mangrove species, thus indicating non-mangrove habitat. Uniquely, the understory of mangrove forests is almost exclusively seedlings, and saplings of the overstory (typically mangrove species) and often lack non-mangrove species (Lugo 1999); an exception being landward transition edges of the mangrove forest or when soil salinity is low (e.g., mangroves in high

rainfall areas; Krauss et al. 2008). The reason for such low species diversity in mangrove forests is that it is extremely difficult for non-halophytes (plants that do not tolerate salinity) to grow in mangrove environmental conditions (Lugo 1999). Because only a small percentage of the world's flora are halophytes only a small percentage of plants should be able to invade the mangrove wetland, if mangrove conditions are present. Of the 55 “true mangroves” listed by Tomlinson (2016), only two species, *B. gymnorhiza* (major component) and *A. aureum* (minor component, or sometimes considered an associate), are currently present in the mangrove wetland. Therefore, we can conclude that mangrove environmental conditions (e.g., salinity, flooding, prolonged hydroperiod, anoxic conditions, and accumulation of toxins such as H₂S; Lugo 1999) are not presently available in the outer perimeter of the mangrove wetland and may also be somewhat limited in the center. Thus, interspecific competition with non-mangrove plants can be largely addressed by restoring the hydrology and promoting mangrove conditions to the greatest spatial extent possible (Lugo 1999, Lewis 2001, Lewis et al. 2016). Conversely, if salinity and water availability decline further in the study area, suitable habitat for *B. gymnorhiza* would also decline further reducing the competitive advantage for *B. gymnorhiza*, leading to more interspecific competition in the understory, and encroachment of non-native species from the perimeter to the center of the mangrove wetland.

Propagule and seedling predation

Propagule and seedling predation by herbivorous or frugivorous crabs is well documented in the literature and can affect mangrove regeneration and reforestation (Van Nederveelde et al. 2015, Krauss et al. 2008). Insects have also been noted in mangrove herbivory including the stem-boring scolytid beetle, *Coccotrypes rhizophorae*, which caused seedling mortality to *Rhizophora mangle* in Panama and Florida. Furthermore, mongoose (*Herpestes auropunctatus*) and rats (*Rattus exulans*, *Rattus rattus*) have been a suspected cause of *R. mangle* propagule attack in Hawaii but showed significantly lower mortality rate than that of the herbivorous crab *Sesarma erythrodactyla* on *R. mangle* in American Samoa (Steele et al. 1999). Mangrove reforestation projects are lacking in CNMI despite historical loss of stands in Saipan (Zachary Williams, personal comm.), consequently, the rate of propagule predation for *B. gymnorhiza* in Saipan is unknown. A recent mangrove rehabilitation effort (Johnson and Gourley 2022) conducted at AMME as part of a compensatory mitigation plan, established 146 *B. gymnorhiza* trees along the coast of AMME between 2016 and 2021. During the planting of propagules, the authors noted that “a fair number of propagules died from being chewed” and presumed either crabs or rats were responsible.

Mangrove species diversity

Of the classified “true” mangrove species (Tomlinson 2016), currently the only “major” mangrove species present on Saipan is *B. gymnorhiza* and the only “minor” mangrove species present is *A. aureum*. As mentioned previously, the territory of Guam contains six “major” mangrove species (*A. marina*, *B. gymnorhiza*, *L. littorea*, *N. fruticans*, *R. apiculata*, and *R. mucronata*) and three “minor” mangrove species (*A. aureum*, *E. agallocha*, and *X. moluccensis*). Historically, the major species *L. littorea*, *R. mucronata*, and *R. apiculata* were also present on Saipan according to palaeoecological studies (Athens & Ward 2004). When multiple mangrove tree species are present within a mangrove forest, zonation of the “major” species into dominate bands parallel to the shoreline is often witnessed (Smith 1992).

In Guam, zonation follows the patterns described by Smith (1992) with the *Rhizophora* species occurring in the maximum salinity seashore and reef flat areas while *B. gymnorhiza* is often restricted to the more inland areas with less salinity. Therefore, it is important to note that the AMME seaward populations of *B. gymnorhiza* (outside the “mangrove wetland”) are growing in areas that would historically be dominated by *Rhizophora* mangroves. While the “mangrove wetland” discussed here is indeed where *B. gymnorhiza* dominance would be expected historically. Ecosystem resilience management suggests that systems with greater levels of functional diversity and redundancy will have greater resilience (Chambers et al. 2019). Therefore, considering species diversity alone, the remanent mangrove forest on Saipan (including the mangrove wetland) and the services they provide would have greater resiliency if locally extinct mangrove species were reintroduced. Accordingly, a CNMI project is underway to source propagules from Guam and reintroduce *L. littorea*, *R. mucronata*, and *R. apiculata* concurrent with *B. gymnorhiza* reforestation at specific sites along the western coastline of Saipan (Zachary Williams, personal com.).

Population distribution

The lack of precise information on the current extent of *B. gymnorhiza* highlights the need for accurate monitoring of this ecosystem during and proceeding restoration efforts. The use of electromagnetic radiation and spectral reflectance signatures to delineate mangrove species, including *B. gymnorhiza*, has been utilized in other studies (Zulfa et al. 2020 & 2021) and may be an appropriate monitoring method for *B. gymnorhiza* in Saipan. Following ecosystem resilience management (Chambers et al. 2019), mangrove reforestation in Saipan should focus on expanding current mangrove stands and wherever possible, creating suitable mangrove habitat within propagule dispersal distance from core sites, thus encouraging greater connectivity between stands. Increased connectivity (i.e., open waterways) between the mangrove stand within the mangrove wetland and the mangrove stand along the coastline on the eastern side of the Smiling Cove causeway will not only improve environmental conditions (e.g., salinity levels) but also improve spatial resilience by way of propagule exchange. Additionally, spatial resilience will also vary at smaller scales with particular importance on small-scale elevation changes and topographic position in relation to tidal inundation and freshwater sources. Mangrove forest composition, structure, and productivity can be analyzed alongside microsite characteristics to improve our understanding of *B. gymnorhiza* habitat suitability and prediction of appropriate mangrove restoration sites.

Water quality

The declining trend in salinity observed here began prior to 2009, based on previous studies. Surface water salinity in the park in December 1986 (wet season) was measured as high as 6.5 ppt. (USACE 1987). Field measurements of surface water salinity ranged from 5.5 to 8 ppt in the mangrove wetland in April 1990 (dry season, Wagner 1990). Since the onset of monitoring by PACN from 2009 to 2018, surface water salinity began near 6 ppt and dropped to ~2 ppt. Such reductions in salinity can be expected to negatively impact mangroves. Smith (1992) found *B. gymnorhiza* maximum groundwater salinity measured in the field at sites where the species was growing was 50 ppt and that the salinity for optimum growth based on culture studies was 8 to 34 ppt.

Variation in AMME wetland salinity was observed in 1986 by USACE (1987) from 0.5 to 6.5 ppt and was attributed to spatial variation and rainfall conditions, with the lowest saline measures occurring after a typhoon. The study concluded that measured salinities indicated brackish conditions, rather than a freshwater wetland. Salinities in AMME mangrove wetland surface waters in 2005 were also observed by Perreault (2007) to decrease following typhoon rainfall. In general, rainfall has been observed to affect the vertical profile of water including the freshwater lens in Pacific islands (Lau and Mink 2006). These observations illustrate the interplay between meteorological and subterranean sources of water for terrestrial plants. Rainfall simply dilutes saline water near the surface. Such influxes of freshwater must then infiltrate the saturated zone of the soil and mix with underlying saline water, which then must percolate toward the surface.

ENSO effects in this part of the Pacific generally result in three conditions relevant to mangroves: higher rainfall during El Niño conditions followed by lower-than-average rainfall, an increased risk of typhoons, and sea level decrease during strong El Niños (NOAA 2015). All of these effects would act to reduce salinity near the surface. El Niño conditions occurred in 2010 and 2015. Surface salinity dropped in 2011, rebounded in 2012, and fell again in 2016, whereafter it did not recover. Hence meteorological conditions clearly affect surface salinity measurements at AMME, which complicates interpretation of historical records. Previous studies were not long-term monitoring efforts, however, and the long-term salinity trend observed here includes variation over El Niño cycles suggesting reductions in salinity are not short-term meteorological effects.

Staff at AMME had reported that the mangroves were not thriving prior to 2015. This decline could have several possible causes or be the result of multiple concurrent stressors. Surface water monitoring in the mangrove wetland showed not only decreasing salinity over time, but more recently a greater number sites without surface water, which could contribute to mangrove tree decline. A decline in surface water salinity and presence concurrent with stable groundwater salinity might suggest that surface water flows derived from ocean tides have been reduced. Such flows might only exist at high tides. While a comparison of surface and groundwater conditions was not possible due to failure of the groundwater monitoring equipment, other data provided insights.

First, at Laura, Majuro Atoll, Republic of the Marshall Islands, the freshwater lens was reduced in thickness under drought conditions and groundwater pumping (Preseley 2005). This had the effect of raising the brackish transition zone closer to the surface. Lower rainfall at AMME was observed in 2017 without a concurrent increase in surface salinity, suggesting that even if the brackish transition zone moved closer to the surface, without a saturated layer at the surface there was no mechanism for dissolved salt to be transported upward to plants adapted to brackish conditions.

Second, alteration of “horizontal” surface and subsurface flow paths in the park could change the balance between the freshwater lens and underlying saltwater under the AMME mangrove wetland. Perreault (2007) observed surface freshwater flow derived from typhoon rainfall diverted from the mangrove wetland by roads and channels. Injection wells, road construction, pipelines installation, and other digging projects could alter subsurface flow paths in the area. Potentially crucial, however, would be alteration of surficial ocean tide connectivity that reduces periodic influxes of seawater.

Krauss (2016) qualitatively surveyed the mangrove wetland and culvert, stating “...the conveyance is currently inadequate to promote any tidal flooding. A lack of tidal conveyance through this culvert is likely the result of more than just in-situ ditch dynamics over time, as adjacent development has no doubt altered the way tides enter the conveyance as originally designed. While tidal fluctuations may still be occurring in the groundwater underlying the forest complex, this hydrographic character is certainly not within the hydrologic requirements for mangrove forests as surface tides are now either rare or non-existent. A restoration plan would need to consider re-engineering of this culvert and tidal conveyance.” Hence while the culvert should be maintained to allow as much conveyance of water as possible, it is unclear whether the culvert is adequate to convey tidal water even when completely open. Inspection of the culvert in 2023 showed substantial plant growth immediately upstream (Figure 3), which could further reduce water conveyance.

Third, patterns over short periods of time must be considered. Perreault (2007) found that groundwater levels in AMME were directly affected by ocean tides. Groundwater level fluctuation lagged behind tide levels differentially across an east to west gradient, suggesting that the eastern portion of the park was more hydraulically conductive than areas to the west. We can hypothesize that differential hydraulic conductivity could explain why higher salinities in surface water were observed further from shore, whereby saline water is not flushed as quickly by meteorologically derived freshwater, as observed by previous studies (e.g., USACE 1987). Such complexity reinforces the importance of understanding groundwater influence on surface water salinities.

Over the long term the ecosystem may be perturbed by potentially antagonistic effects of climate change. Predicted effects of climate change in the region include increasing ocean water temperatures, sea level rise, ocean acidification, wetter wet seasons, drier dry seasons, and altered typhoon activity (EPA 2016). Thus, while rising sea level would result in increased inland infiltration of saline groundwater, improving conditions for mangroves, rising sea level could also result in shoreline erosion and loss of park land including the wetland itself.

Conclusions and Recommendations

We recommend several actions. The highest priority is to improve tidal connection of the mangrove wetland with the sea, which could include re-engineering the culvert under Rt. 30 or other excavation work to promote tidal connection. This recommendation is in specific agreement with those of Krauss (2016) and general agreement with the need to restore physical ecosystem processes that are essential for the long-term stability of mangroves (UNEP 1984). Secondly, a properly designed and constructed permanent groundwater monitoring well should be installed to monitor a hydrologically complex system that cannot be fully characterized by surface water monitoring alone. The USGS should be consulted as to the best well location. The USGS recommends that a new well should be constructed of 4-in PVC casings to provide sufficient room for multiple instruments, include a concrete pad, and extend two to three feet above ground level to prevent flooding by surface flows. PACN should also add a surface water monitoring station in the mangrove wetland if a permanent surface water body exists near the culvert. Such a monitoring station would better address temporal patterns in surface water salinity affected by tides.

The ecological importance of mangroves is recognized by the Pacific Regional Environmental Programme, of which CNMI is a member (SPREP 2011). Thus, restoration of the mangrove wetland as a mangrove wetland supports the Pacific Regional Environment Programme regional wetland action plan. Further, the AMME mangrove wetland is designated as Class 1 by the CNMI Department of Environmental Quality. As such, these waters are intended to remain in their natural state as nearly as possible with a minimum of anthropogenic pollution with no wastewater discharges. Wagner (1990) conducted reconnaissance to make recommendations for a monitoring program at the site. PACN later identified “All groundwater within the WQ area of interest including various western Saipan coastal aquifer components” to be monitored (DeVerse and DiDonato 2006). Perreault (2007) recommended “additional monitoring wells would need to be installed... to improve understanding of the effects of nearby faults and injection wells on local hydrology... A long-term plan that includes monitoring of surface and subsurface water levels and salinities could be useful to identify trends that could affect the park’s critical habitat, including its endemic and endangered species.” While PACN initiated a monitoring program that included groundwater, equipment failure resulted from use of existing wells not constructed for long-term groundwater monitoring.

Lewis et al. (2016) emphasized the importance for early warning of degradation in mangrove forests. Given the declines in *B. gymnorhiza* density within the mangrove wetland and the uncertainty of mangrove recovery due to decreased habitat suitability and invasive plant encroachment, restoration actions should be taken as soon as possible. Restoring the tidal connection to the mangrove wetland should be prioritized as restoring mangrove conditions would exclude weedy non-native species from the core areas and allow *B. gymnorhiza* to naturally recolonize. Indeed, Lewis (2001) points out that the greatest cost-savings in mangrove restoration can be achieved when hydrology is restored in areas where natural recolonization is still occurring. *B. gymnorhiza* seedling densities in the four plots where the mangrove was present indicated natural regeneration was occurring but very near what Bosire et al. (2008) referenced as the minimum needed for natural regeneration in forestry practices (2500 per hectare). While significant effort in mangrove forest restoration could be saved by restoring the hydrology before the ability of the system to naturally regenerate is lost, restoration of hydrologic processes in this system may be difficult.

It is unclear whether clearing and maintenance of the existing Rt. 30 culvert would be adequate to restore and maintain salinity conditions in the mangrove wetland sufficient to support a population of mangroves. Krauss (2016) suggested that excavating parts of the forest may be necessary to improve tidal connection, which suggests the culvert alone is not adequate to maintain tidal connection between the sea and the mangrove wetland. We recommend a synoptic survey of salinity using specific conductance during both wet and dry seasons, and concurrent short-term deployment of specific-conductance and water level instruments to address this question. Such a survey should be designed to evaluate spatial patterns and variability related to ocean tides, and could leverage existing monitoring wells to some extent if any remain intact (e.g., Perreault 2007). Evaluation of the amplitude of tidal variation, if any, would contribute to the evaluation of the status of hydrologic connection with the ocean.

We further recommend the restoration of the tidal connection as soon as possible, annual assessment of the natural regeneration of *B. gymnorhiza*, non-mangrove plant removal that enhances natural regeneration, and out planting in suitable locations experiencing low natural regeneration. In addition, several projects are also recommended which would support mangrove restoration within AMME and across Saipan: 1) develop an appropriate remote sensing approach (e.g., spectral imagery) to inventory and monitor *B. gymnorhiza* populations at higher temporal frequency within AMME and across Saipan, 2) obtain high-accuracy elevation data within the mangrove wetland to better understand mangrove habitat availability and suitability, 3) develop or support public outreach projects that convey the value and importance of the mangroves to Saipan residents and visitors at AMME. Despite the diminished extent on Saipan, mangrove forests still provide incredible value as habitat for threatened/endangered wildlife, nursery habitat for fisheries species, shoreline protection, sediment trapping/filtration, and aesthetic and cultural enjoyment (UNEP 1984). This understanding has led to commitments in several organizations to restoring mangroves across Saipan. The largest percentage of core mangrove forest on Saipan occurs within American Memorial Park with a majority of that occurring within the mangrove wetland. However, mangrove habitat suitability within the mangrove wetland is in a state of decline and without prompt action a devastating loss to this core population appears likely. In 2019, natural regeneration still appeared feasible in certain areas, but only if quick action is taken to restore the appropriate hydrology.

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