Wetlands Ecol Manage (2017) 25:563–582
DOI 10.1007/s11273-017-9536-0

ORIGINAL PAPER

Mangroves buffer marine protected area from impacts of Bovoni Landfill, St. Thomas, United States Virgin Islands

Jessica A. Keller · Kristin Wilson Grimes · A. S. Reeve · Renata Platenberg

Received: 20 July 2016 / Accepted: 14 February 2017 / Published online: 22 February 2017

Abstract One of the many ecosystem services that mangrove systems provide is their ability to act as buffers between the land and sea, protecting human development from storm surges while also trapping terrestrial pollutants. In St. Thomas, United States Virgin Islands, an ecologically-important mangrove system sits between Bovoni Landfill and a marine protected area, the St. Thomas East End Reserves. To characterize the physical processes driving this mangrove system, groundwater hydraulic head, sediment cores, sediment surface temperatures, and water and sediment chemistry were analyzed. Hydraulic head data from January to November 2014 were used to determine vertical and horizontal groundwater flow directions. Water and sediment samples were tested for heavy metals potentially originating from Bovoni Landfill. Stratigraphic context was provided by the sediment cores and used to infer past environmental conditions. Subsamples were taken from these cores and analyzed for dry bulk density, organic matter content (through loss on ignition), and heavy metals using electron microscopy. Vertical groundwater velocity and sediment porosity were determined by calibrating a one-dimensional finite difference heat transport model to near surface temperature data from depths of 0, 7, 14, and 21 cm. Groundwater was found to flow from the terrestrial upland, through the mangroves, and toward the ocean for the majority of the study. Flow reversal was seen after long periods of little precipitation. In the surface and shallow groundwater samples, trace metal concentrations were measured from 23 to 105 μg/L for Cr, Ni, Sn, and Zn. Sediment samples collected near the landfill contained

Electronic supplementary material The online version of this article (doi:10.1007/s11273-017-9536-0) contains supplementary material, which is available to authorized users.

J. A. Keller (✉) · K. Wilson Grimes
Center for Marine and Environmental Studies, University of the Virgin Islands, 2 John Brewers Bay, St. Thomas, VI 00802, USA
e-mail: jessica.keller@myfwc.com

K. Wilson Grimes
Wells National Estuarine Research Reserve, 342 Laudholm Farm Rd., Wells, ME 04090, USA

A. S. Reeve
School of Earth and Climate Sciences, University of Maine, 5790 Bryand Global Sciences Center, Orono, ME 04469, USA

R. Platenberg
College of Science and Math, University of the Virgin Islands, St. Thomas2 John Brewers Bay, VI 00802, USA

Present Address:
J. A. Keller
Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission, 2796 Overseas Highway, Suite 119, Marathon, FL 33050, USA

Bi, Cr, Sn, Ti, and Zn. Very slow flushing of sediment pore water was indicated by the vertical groundwater velocities produced from the heat transport model, which ranged from $\pm 10^{-7}$ to $\pm 10^{-9}$ m/s. This study revealed that the mangrove system is an important buffer system protecting the outer lagoon of the marine protected area from terrestrial contaminants via sediment trapping and slowing of water fluxes from the upland area into the lagoon. The results presented here can be used as a baseline for future studies and are relevant to local managers and to landfill closure plans.

Keywords Groundwater · Mangroves · Heavy metals · Landfill · St. Thomas East End Reserves · Caribbean

Introduction

Mangroves are recognized for the important ecosystem services they provide, including habitat for many species of birds, fish, and crustaceans (Mumby et al. 2004; Montalto et al. 2006; Nagelkerken et al. 2008), cultural resources (Barbier et al. 2011), and most recently, for the ability to sequester and store carbon (Jardine and Siikamäki 2014). Mangroves also buffer interactions between terrestrial and oceanic environments, protecting land and human development from storm surges (Shepard et al. 2011; Zhang et al. 2012; Kathiresan and Rajendran 2005; Mazda et al. 1997), and by trapping terrestrial pollutants before they reach the nearshore marine environment (Tam and Wong 1999; Clark et al. 1998; Cavalcante et al. 2009; Sthevan et al. 2011; Ismail et al. 2014).

Despite the many valuable ecosystem services provided by mangroves worldwide, their area has declined by 35% since the 1980s (Valiela 2001) and improved mapping at the global scale using remote sensing shows their distribution to be even more limited than other recent estimates suggest (Giri et al. 2011). Urban development, deforestation, shrimp aquaculture, overexploitation for resources, and changes in sea level, temperature, and precipitation are all drivers of mangrove loss (Ellison and Farnsworth 1996; Spalding et al. 1997; Alongi 2002; Gilman et al. 2008). Urbanization is one of the major threats to mangroves and other coastal wetland systems, through direct habitat loss and indirect impacts like changes to hydrologic and sediment regimes that result in introductions of pollutants, including heavy metals, to these coastal systems (Lee et al. 2006, 2014; Fonseca et al. 2013; Zhang et al. 2014). In the U.S. Virgin Islands, years of direct habitat loss and indirect stressors, have resulted in few remaining intact stands of mangroves; this is particularly true on the island of St. Thomas (Platenberg 2006).

On St. Thomas, the largest remaining mangrove system occurs within the St. Thomas East End Reserves (STEER), a marine protected area (Horsley Witten 2013, Fig. 1). However, this system is under considerable stress from land-based sources of pollution, including Bovoni Landfill, boatyards and marinas, sewage effluent from a wastewater treatment plant, and surface runoff from its relatively large inland watershed (11.8 km^2, the largest watershed on St. Thomas, Horsley Witten 2013) that is home to over one third of the population of St. Thomas (Horsley Witten 2013). Previous work by Pait et al. (2014) revealed greater heavy metal contamination of marine sediments in parts of STEER closer to the landfill and boatyards (Mangrove Lagoon and northern Benner Bay) for chromium, copper, lead, mercury, nickel and zinc, compared to other areas within STEER's boundaries. Because of the study's sampling design, however, the authors could not pinpoint Bovoni Landfill as a definitive source of lagoonal sediment

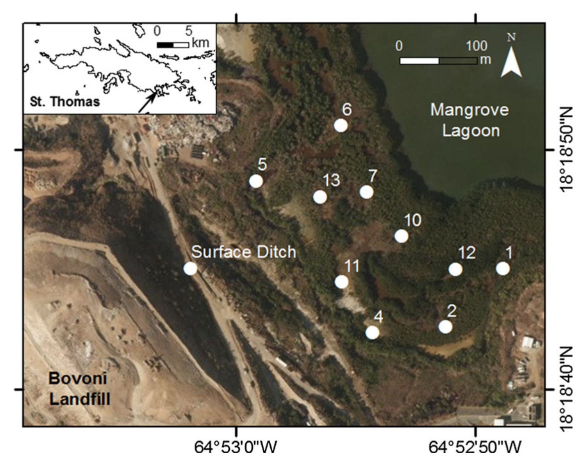


Fig. 1 An aerial view of Mangrove Lagoon, Bovoni Landfill, and the sample locations within the study area. *Arrow* in the *inset* indicates the location of the study area on St. Thomas, USVI

contamination. The authors suggested that both the boatyards and the landfill were likely sources of these heavy metals, but recognized other potential sources including a horse track and ephemeral waterways that could transport upper watershed contaminants into STEER (Pait et al. 2014). In 2006, however, the Environmental Protection Agency (EPA) observed violations of waste management at the unlined landfill, including improper disposal of medical and septic waste, used oil, and lead-acid batteries, as well as the migration of leachate into the adjacent mangrove system (District Court of the Virgin Islands, 2006). Furthermore, in 2012, a representative of The Nature Conservancy in St. Thomas, expressed concern about the mangroves located between the landfill and Mangrove Lagoon due to observed mangrove degradation and apparent loss that had occurred in recent years (Anne-Marie Hoffman, pers. comm.).

Landfills can be significant, well-known sources of heavy metals to the marine environment through surface and groundwater contamination pathways (Jones 2010). Heavy metals like mercury, nickel, copper, chromium, and zinc are problematic because they bioaccumulate within marine organisms, negatively impacting organism function by interfering with metabolic processes (Islam and Tanaka 2004). Coastal habitats, like mangroves, can lessen this effect by intercepting land-derived heavy metals and sequestering them in sediments and vegetation, thereby providing a critical ecosystem service for nearby marine systems (MacFarlane et al. 2007). Knowledge of the hydrological and geomorphological processes within mangrove systems is critical for: (1) understanding their structure (Kjerve et al. 1999; McKee and Faulkner 2000) and ecosystem services (Ewel et al. 1998), (2) identifying the potential spatial and/or temporal variability in the provision of the chemical and sediment transport and buffering capacity (Lee et al. 2014), and (3) developing appropriate conservation and sustainable management practices for these coastal systems (Calderon et al. 2014). Thus, the objectives of this study were three-fold: (1) to document potential heavy metal contamination of sediments and surface and groundwater within the mangrove system abutting Bovoni Landfill, (2) to understand how this mangrove fringe may buffer potential terrestrial impacts to the lagoon through improved documentation of groundwater flow patterns within its boundaries through field observations and modeling, and (3) to explore how the provisioning of this key ecosystem service may change in response to external forcing factors, like wet and dry seasonality, precipitation, and tidal events.

Methods

Study site

St. Thomas is approximately 80 km^2 with a population of 51,000 people and receives approximately 2 million visitors a year (Fig. 1; US Census Bureau 2013; U.S.V.I Bureau of Economic Research 2013). Mangrove Lagoon, located in the western-most portion of STEER, supports local tourism businesses, provides habitat for lobsters, conch and sea turtles, and is a nursery for commercially-important fish (STEER 2011). Mangrove Lagoon was declared an Area of Particular Concern (APC) in 1978 by the Territory's Coastal Zone Management's Planning Office to preserve it as a significant natural area (DPNR 2003). At that time, the Planning Office found the lagoon to have "exceptional natural value" because of: (1) the anchorage provided to boats during storms, (2) its numerous recreational opportunities, and (3) the protection its mangrove-fringed shores provided against shoreline erosion, flooding, and hurricane waves. Mangrove Lagoon was later incorporated into a larger group of marine reserves and wildlife sanctuaries on the east end of the island, collectively called the St. Thomas East End Reserves (Horsley Witten 2013).

Between Mangrove Lagoon and Bovoni Landfill is the largest remaining intact stand of mangroves on St. Thomas (Horsley Witten 2013, Fig. 1). Our study site is located within this mangrove system. Black mangroves, *Avicennia germinans*, dominate the inland area closer to the landfill and red mangroves, *Rhizophora mangle*, dominate the outer fringe that borders Mangrove Lagoon. Over time, the landfill's historic dumping areas have expanded into the adjacent mangrove system and have decreased the distance between the landfill and Mangrove Lagoon (Horsley Witten 2013). During this study, accessing the mangrove system by land required navigating around large piles of scrap metal into the vegetation bordering the eastern most access road (Fig. 1).

Conditions within the study area vary depending on time of year and the amount of recent rainfall. The Virgin Islands, characterized as a dry tropical climate (Bowden et al. 1970), do not have sharply defined wet and dry seasons (Crossmand and Palada 2003), but on average receive the most rainfall from September through November and the least from January through March (NOAA Average Rainfall, Accessed 11 Apr 2016). The majority of rainfall from May to November results from easterly winds (which may develop into tropical storms and hurricanes), while the majority of rainfall from December to April is the result of cold fronts (Calversbert 1970). Standing water levels in the study area vary depending on location within the mangrove swamp and time of year. Locations closer to the lagoon have standing water year round (or nearly year round), but further inland, the presence of standing water varies. The surface sediment inland is mostly dry or completely dry and cracked during the dry season but contains standing water during the wet season (Fig. 2). The study area is microtidal with a mean range of 0.21 meters (NOAA Tides and Currents 2016).

In this study, ten sites were placed throughout the mangrove system to capture potential hydrologic spatial variation. The ten sites were grouped into three zones based on proximity to Mangrove Lagoon and Bovoni Landfill. The zones serve to highlight the hydrologic and sedimentological differences in different parts of the mangrove swamp. The outer zone encompasses the 5 sites (1, 12, 10, 7, and 6) located close to the mangrove fringe separating the mangrove swamp from the lagoon (Fig. 1). The middle zone contains the two sites (2 and 13) located in the middle of the mangrove swamp. The inner zone is comprised of the three sites (5, 11, and 4) located furthest inland and closest to Bovoni Landfill. Except for the hydraulic head contour maps, all results are presented in terms of these three zones.

Measuring groundwater flow

To determine groundwater flow direction, nineteen groundwater wells (2.54 cm nominal diameter flush threaded polyvinyl chloride (PVC), 30 cm machine slotted screen) were installed across the study area (Fig. 1) by manually driving wells into core holes created with a "Dutch" hand-auger corer. A shallow (1.5 m total length) and a deep (3 m total length) well were placed at each sampling station with two exceptions. At site 7 only a shallow well was installed (due to lack of materials) and at site 5 the deep well had a total length of 2.1 m because it met refusal. Following installation, all wells were developed by surging the water with a sealed plastic pipe and pumping the sediment and water slurry from the well using a hand vacuum pump.

To determine absolute positions and provide reference points for manual surveying, deep wells at four sites (1, 4, 5 and 6) were surveyed using a global positioning system (GPS; Trimble NetR9 GPS receiver and Zephyr II geodetic antenna). GPS data were post-processed using the GPS Precise Point Positioning service run by Natural Resources Canada, Geodetic Survey Division. All remaining wells were manually surveyed to the top of the pipe using a CST/Berger 32x Automatic Level and stadia rod, to calculate the relative elevation differences between wells. Well elevations were verified by comparing optical survey data with GPS data ($\pm$ 3 cm accuracy). GPS coordinates for all wells were recorded with a hand-held GPS unit (Trimble Navigation Limited).

Hydraulic heads were manually measured using an electrical water meter (Solinst Canada Ltd.) and continuously monitored using data-logging pressure transducers (Solinst Canada Ltd. Leveloggers) to determine vertical and horizontal groundwater flow patterns within the mangrove system. Non-vented pressure transducers provided a nearly-continuous history of hydraulic head over the study period (January 6th to November 11th 2014), while the manually-measured values served as a quality control and were used to reference all data to a common vertical datum. Data loggers recorded water pressure and temperatures every ten minutes. Water pressure data were compensated with atmospheric pressure values recorded with a separate barometric pressure transducer at site 11. The compensated values were offset to align with the manual hydraulic head data.

Hydraulic head contour maps were created to determine seasonal changes in horizontal groundwater flow. The depth to water values in each well were manually measured five times in 2014 (2/13, 3/6, 5/6, 7/1 and 11/11) using an electric water level meter. The depth to water data were subtracted from the surveyed well elevations to obtain hydraulic head data relative to mean sea level. Contoured potentiometric surface maps were created by interpolating hydraulic head

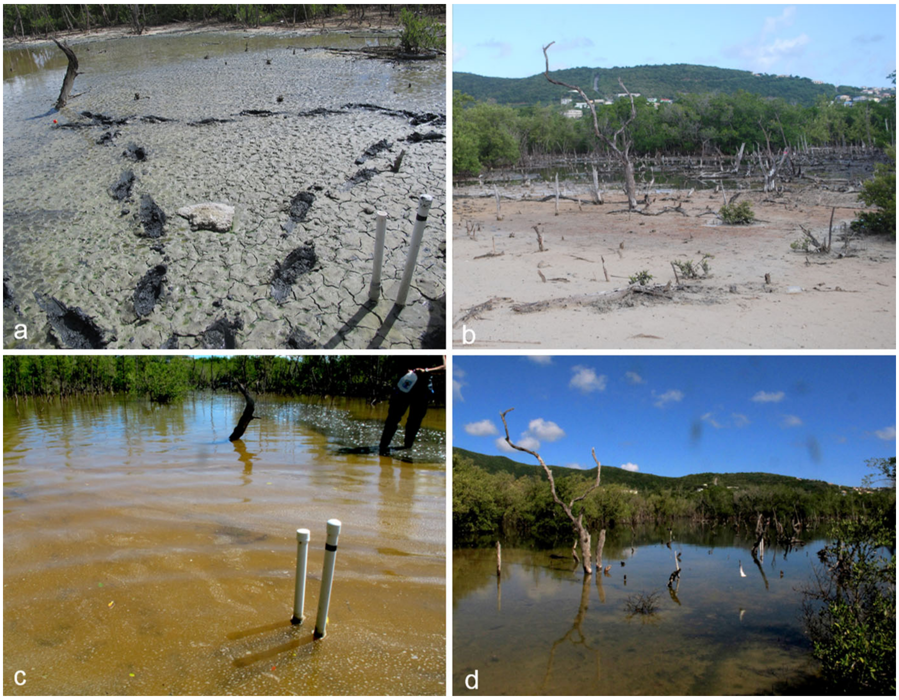


Fig. 2 Comparison between wet and dry seasons at sites 5 (**a**, **c**) and 11 (**b**, **d**). The *top figures* (**a**, **b**) were taken during January and February while the *lower figures* (**c**, **d**) were taken during November. Photo credits: **a**, **c**, and **d** taken by J Keller, **b** taken by K Wilson Grimes

data using kriging routines (Childs 2004; Davis et al. 2009; Rabah et al. 2011) available in ArcGIS 10.2 (ESRI 2013). Empirical Bayesian Kriging was used for shallow hydraulic head interpolations, but could not be used for the deep hydraulic head interpolations because of the small sample size ($n < 10$). In the case of the deep hydraulic head interpolations, simple kriging was used instead.

To test surface water and groundwater for possible contaminates, water samples from both the deep and shallow wells were taken at four sites close to the landfill (2, 4, 5, and 11) and one site close to the lagoon (1) on January 9th, 2014. Surface water samples were also taken at site 5 due to an observed chemical smell during well installation and at a surface ditch containing greenish colored water (Fig. 1). Due to high sediment concentrations, water samples were not filtered in the field, but were filtered in the lab prior to analysis. Samples were collected with no headspace to limit gas exchange, kept in a cooler on ice until refrigerated and processed for pH and specific conductance within 9 h of collection. Samples were shipped to the University of Maine's Sawyer Environmental Chemistry Laboratory, filtered, and tested for copper (Cu), chromium (Cr), nickel (Ni), lead (Pb), tin (Sn), zinc (Zn), and total dissolved nitrogen (TDN). These six heavy metals are found in sheet metal, wire, chrome plating, appliances, batteries and paint (Agency for Toxic Substances and Disease 2014), all of which are likely to be found in Bovoni Landfill. Available resources limited the scope of the chemical analysis and a small suite of heavy metals were chosen based on their likely presence and their potential to affect organism health. Heavy metal analysis was performed using a Perkin-Elmer Optima 3300XL inductively coupled plasma atomic emission spectrometer and TDN analysis was performed with an ALPKEM flow solution IV autoanalyzer. Specific conductance and/or salinity of surface waters were measured using a portable meter (Hach Company) in the field during well installation. Values were not recorded at sites with very little or no standing water.

To determine vertical fluid velocity in shallow groundwater, we measured the spatial and temporal distribution of temperatures in the subsurface. Groundwater movement affects these distributions, and the use of heat as a tracer to assess groundwater interaction with surface water systems has increased over the past two decades as low cost sensors have become available and data analysis techniques have been refined (Rau et al. 2014). One method uses the decrease in amplitude and lag in the diurnal temperature signal with depth to estimate the vertical groundwater velocity (Stallman 1965). This change in the temperature signal was simulated using a heat transport model and the simulated signal was fit to temperature data by adjusting the vertical groundwater velocity. In this study, high-resolution temperature measurements of groundwater were recorded using data logging temperature sensors (Maxim Integrated Thermochron Ibuttons). Four sensors were installed in a vertical array at depths of 0, 7, 14, and 21 cm. These temperature sensor arrays were placed at sites 1, 2, 5 and 7 and left for one week in February 2014. A one-dimensional finite-difference model, similar to that described by Vandenbohede and Lebbe (2010), was calibrated to temperature data collected at these depths while vertical groundwater velocity and sediment porosity were adjusted in simulations to achieve a best fit.

Sediment properties and stratigraphy

A 3-cm diameter Eijkelkamp "Dutch" hand-auger corer was used to collect 1 to 2 m sediment cores at nine of the ten sites on January 6, 2014. No core was recovered at site 12, due to slippage of the sediment from the core tube. Length of the cores varied depending on sediment composition (soil hardness and consistency). Most cores were about 2 m long, but only 70 cm was collected at site 6, and 150 cm at site 5 because we met refusal.

Cores were split, photographed, stratigraphically described, and sub-sampled at 5 cm intervals for dry bulk density and percent organic matter using loss on ignition techniques (Ball 1964; Craft et al. 1991; Heiri et al. 2001). Sediments were interpreted as mangrove peat (high amounts of peat, roots, branches, and/or fine root hairs) or mud flat/pool (clay-rich sediment largely lacking plant macrofossils) environments. Core sub-samples were dried at 80°C for 8 h until a constant mass was achieved. Dry weight and wet volume were used to calculate dry bulk density, where:

$$\text{Dry bulk density } \left(\text{g/cm}^3\right) = \text{dry weight (g)/wet volume } \left(\text{cm}^3\right)$$

Dried samples were ground with mortar and pestle, homogenized over the sample interval, and combusted at 550 °C for 4 h. Percent organic matter by loss on ignition (Ball 1964; Craft et al. 1991) was calculated by mass loss, where:

$$\text{Percent organic matter} = \frac{(\text{dry mass before combustion} - \text{dry mass after combustion})}{\text{dry mass before combustion} \times 100}$$

Sediment shear strength is a measurement commonly used in wetlands to understand differences in sediment properties and susceptibility to erosion (Wigand et al. 2014; Turner 2009; Turner et al. 2011). Sediment shear strength was measured in situ at each well site using a 16 × 32 mm Humbolt inspection vane (Elgin, IL) in July 2014. Sediment profiles of shear strength were created by measuring the stress required to force sediment failure (in kilopascals) every 10 cm from 10 to 100 cm depths. Three replications were performed at each site and the mean value was calculated for each depth.

For a more comprehensive understanding of the subsurface structure, stratigraphic interpretations, dry bulk density, water content, organic content, and shear strength profiles for each site were plotted together. Values for dry bulk density, water content, organic content, and shear strength were then categorized as either mangrove peat or mud flat/pool sediments based on our stratigraphic interpretations. Two-sample unequal variance t-tests were performed to compare these values between mangrove peat and mud flat/pool.

To better understand how the subsurface stratigraphy changes throughout the study area, cross-sections of stratigraphic interpretations were created in zones based on proximity to the landfill and Mangrove Lagoon. Three cross-sections (sites 4, 5 and 11; sites 2 and 13; sites 1, 6, 7, and 10) were created to view connectivity across the mangrove swamp while looking for differences among these three zones. A fourth cross-section (sites 5, 13 and 1) was created to more easily examine how the stratigraphy changes from the inner zone toward the lagoon.

Dried sediment samples were analyzed for heavy metal presence using scanning electron microscopy (SEM) and energy dispersive spectroscopy (EDS). Other studies have used SEM/EDS to determine the source of sediments in estuaries and examine sediments for heavy metal presence (Zadora and Brożek-Mucha 2003; Rajkumar et al. 2012). In this study, scanning electron microscopy (Tescan Vega II XMU SEM) and energy dispersive spectroscopy (EDAX Genesis system) were used to detect the presence of heavy metal (besides Fe-containing sediments) in selected, dried sediment samples at the University of Maine. At least two samples (one near the top and one near the bottom) from every core were tested for heavy metals (Online Resource 1), while cores from sites 4, 5 and 11 had additional samples tested due to their proximity to the landfill. Elemental analyses were performed on locations within samples based on backscatter analysis; backscatter analysis allows the rapid identification of locations containing elements with high atomic number (Goldstein et al. 1992). Energy dispersive spectroscopy was only performed on sediment particles with high atomic numbers as identified by the backscatter analysis of the SEM.

Results

Groundwater flow and water chemistry

Horizontal groundwater flow directions, determined from manual measurements (Table 1), varied seasonally with shallow and deep groundwater behaving similarly to each other throughout the study period (Fig. 3). In February and March, groundwater flowed into the center of the mangrove system from the direction of Mangrove Lagoon and from the direction of the landfill. In May, flow was from the lagoon toward the landfill. In July, groundwater flowed from the landfill into the mangroves, although the differences among shallow hydraulic heads were smaller and flow was in a more easterly direction compared to the deep hydraulic head (Fig. 3b). In November, flow was from the landfill toward the lagoon in both deep and shallow groundwater.

Hydraulic head time-series data revealed differences in groundwater responses between seasons and among zones. At all sites, hydraulic head was higher in the wet season, from May until November, than the dry season. Within seasons, hydraulic head patterns varied among zones. Differences between shallow and deep hydraulic heads were larger in the inner zone (5, 11, and 4) than the outer zone (1, 6, 7, 10, and 12), where these differences were negligible Figs. 4 and 5. At sites 4 and 5, recharge occurred for a large portion of the study period when shallow groundwater was higher than deep groundwater, but at site 11, discharge was more common. In the inner zone, patterns in hydraulic head measured in deep wells lagged behind and exhibited a damped response, compared to hydraulic head measured in shallow wells. Hydraulic head data collected at site 2 and site 13, both located in the middle zone, contained patterns similar to the outer and inner zones, respectively (Online Resource 2).

Comparing hydraulic head time series of sites 1 and 5 to the recorded mean sea level and precipitation revealed differences between the inner and outer zone in hydraulic head responses to external factors (Fig. 6). Shallow and deep hydraulic head at site 1 responded quickly to spring high tides in the beginning of the study period and did not have an obvious response to two large rain events on May 7th and August 22nd. Conversely, site 5 was more responsive to precipitation events than tidal changes (Fig. 6). This site had a quick increase in shallow hydraulic head with a slower increase in deep groundwater during the large rain event in May. However, hydraulic head of neither the shallow nor deep wells at site 5 showed much of an increase after the second large rain event August 22nd.

Hydraulic head of both shallow and deep groundwater at site 1 (Fig. 7a) responded mainly to sea-level changes during the first large rain event May 7–10 (Fig. 7). In the shallow well, the hydraulic head increased approximately 2 cm on May 7th, but this increase may be from the incoming high tide and not from precipitation. Site 5 (Fig. 7b) did not have the tidal pattern seen at site 1, but had an abrupt 20 cm increase in shallow groundwater and a slow increase of approximately 5 cm in deep groundwater during the

Table 1 Well locations, elevations, and hydraulic head of each manual measurement

Well	Latitude	Longitude	Well elevation	Hydraulic head				
				2/13/14	3/6/14	5/6/14	7/1/14	11/11/14
1s	18.31250	−64.88023	0.907	41.5	42.9	46.0	47.8	48.6
1d	18.31250	−64.88023	0.879	42.9	43.1	45.6	48.4	49.4
2s	18.31183	−64.88090	1.346	36.6	38.6	45.1	47.8	48.0
2d	18.31183	−64.88090	1.361	39.6	37.4	44.7	48.5	48.3
4s	18.31176	−64.88175	1.113	39.3	36.7	29.4	45.9	50.7
4d	18.31176	−64.88175	1.133	41.2	38.8	36.9	51.2	58.2
5s	18.31352	−64.88310	1.031	49.9	50.8	33.7	53.2	55.8
5d	18.31352	−64.88310	0.960	46.1	49.9	33.1	49.8	56.4
6s	18.31416	−64.88211	0.982	44.4	41.4	45.2	47.3	48.2
6d	18.31416	−64.88211	1.010	42.2	41.5	43.0	46.1	49.1
7s	18.31339	−64.88181	0.932	43.6	41.4	44.6	46.1	47.7
10s	18.31288	−64.88141	0.998	40.3	41.8	45.3	46.6	47.6
10d	18.31288	−64.88141	0.958	40.8	42.1	44.9	37.2	48.2
11s	18.31235	−64.88210	1.041	39.5	40.2	44.6	47.4	48.4
11d	18.31235	−64.88210	0.988	44.7	43.8	44.1	47.8	52.7
12s	18.31250	−64.88078	0.967	42.1	42.2	43.3	45.5	49.1
12d	18.31250	−64.88078	1.005	42.4	42.6	45.1	47.9	48.2
13s	18.31334	−64.88235	0.888	37.3	39.1	44.0	48.5	48.4
13d	18.31334	−64.88235	0.848	41.9	44.2	43.9	49.4	53.4

In the well column, s or d refers to the shallow or deep well. Well elevation is the elevation, in meters, at the top of the well. Hydraulic head measurements are in centimeters

rain event. The slower increase in deep hydraulic head suggests the sediments at depth have a low hydraulic conductivity, insulating the deeper well from surface conditions. At site 1, hydraulic head dropped after the large increase on May 7th, while shallow hydraulic head remained high at site 5.

The hydraulic head at these two sites responded differently to the second large rain event August 21–23 (Fig. 8). Shallow and deep groundwater at sites 1 (a) and shallow groundwater at site 5 (b) have similar patterns to the recorded water level, while the deep groundwater at site 5 had no tidal signal. Hydraulic head in all four wells had a slight alteration of their pattern during the peak rain day, August 22nd, but no large response to the precipitation was recorded. At site 1, recharge occurred immediately following the incoming tide, and discharge occurred when the shallow hydraulic head fell faster than the deep hydraulic head. At site 5, hydraulic head did not show an increase in response to rain and recharge occurred throughout the rain event. Deep hydraulic head increased slowly after the rain event, but the recorded mean sea level also appeared to increase during this time.

Daily fluctuations of hydraulic head occurred from May to November and appear to match the diurnal tidal signal (Figs. 4, 5, 6). In the outer zone, these daily fluctuations in hydraulic head were measured in both shallow and deep wells. In the inner zone, daily fluctuations were observed in shallow, but not deep, hydraulic head measurements. This daily tidal signal only appeared when hydraulic head values were above ~45 cm. Below this threshold, monthly spring high tides affected hydraulic head, but the diurnal signal was not present.

Chromium was found in all water samples, but other heavy metals detected (nickel, tin, and zinc) were only found at site 5 or at the surface ditch (Table 2). The surface ditch sample contained nickel, tin, and the highest concentrations of chromium and TDN. Lead and copper were not found above their detection limits (80 μg/L for Pb and 100 μg/L for Cu) in any sample.

Small vertical velocities produced by the heat transport model (Table 3), indicate very slow vertical movement of shallow groundwater. Site 5 was the only site to have a positive fluid velocity value, representative of groundwater recharge during the sensor

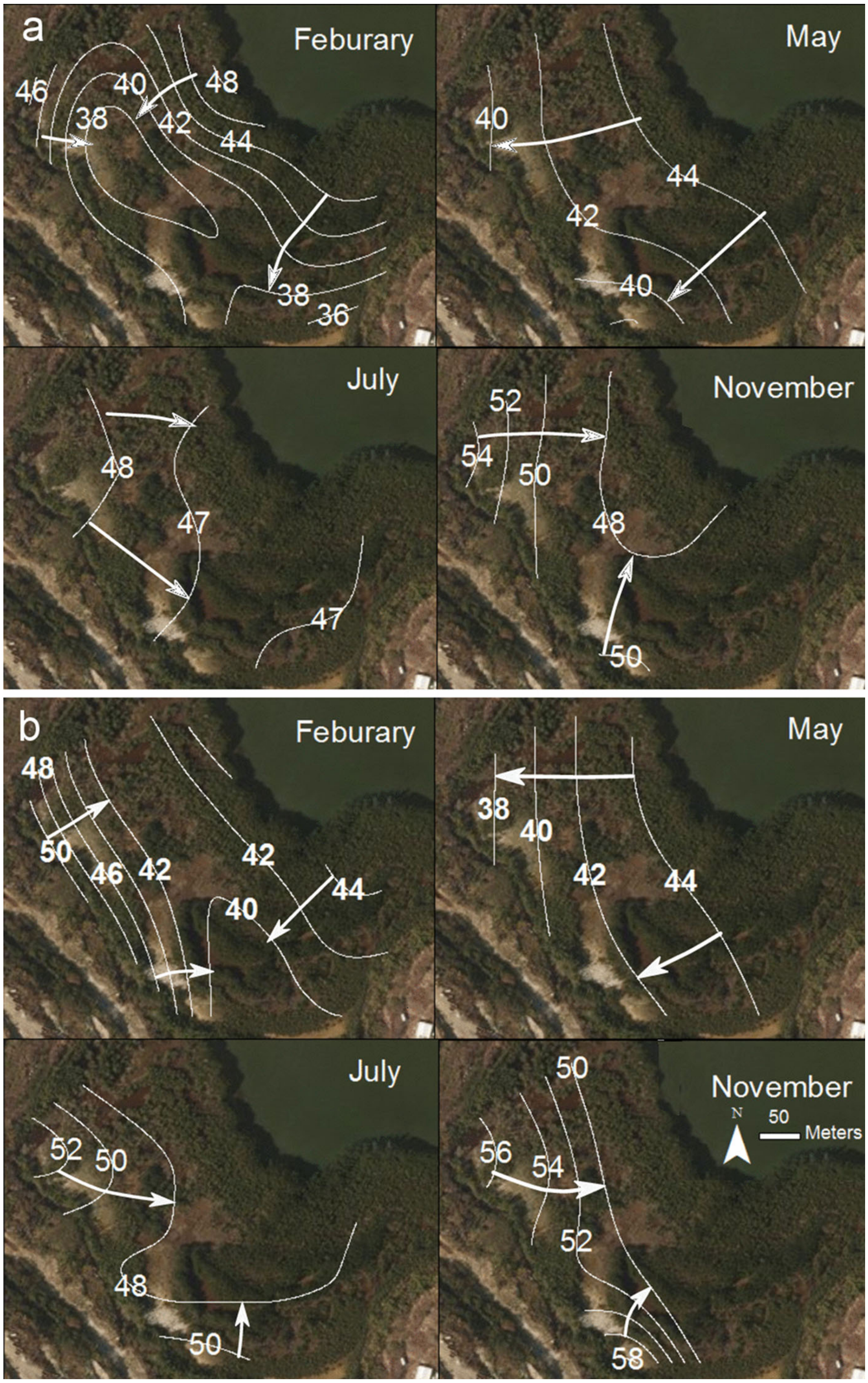
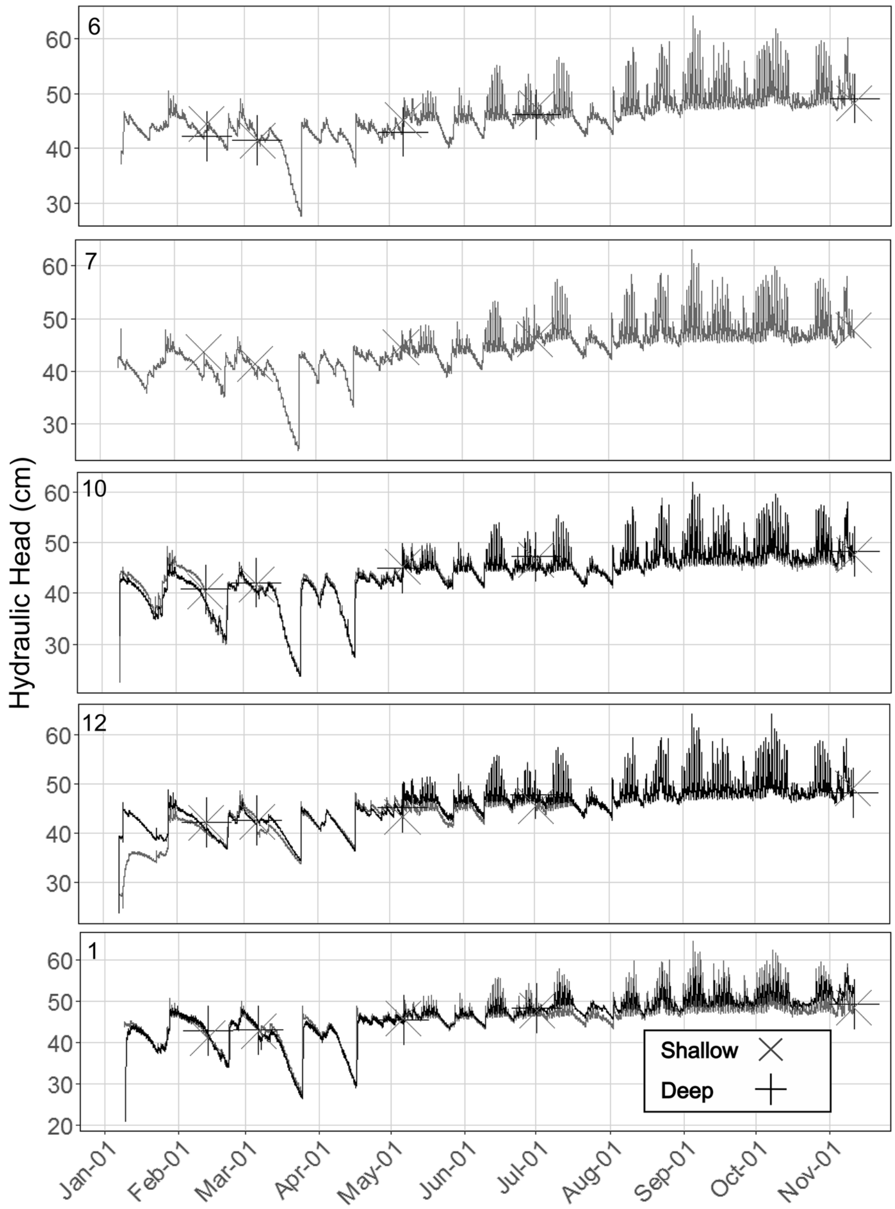


Fig. 3 Hydraulic head (cm) contours for shallow (**a**) and deep (**b**) groundwater with *equipotential lines* and *arrows* to indicate direction of flow. Groundwater flowed into the mangroves from both the upland and from the lagoon in February, into the mangroves from the lagoon in May, and into the mangroves from the upland in July and November. The *north arrow* and *scale bar* apply for all eight images. Bovoni Landfill is to the southwest of the study area

Fig. 4 Hydraulic head time series of both shallow (*grey*) and deep (*black*) wells installed in the outer zone, near the lagoon. Hydraulic head plots are stacked based on spatial location of each site (Fig. 1). Continuous pressure transducer values are shown as *lines* and *symbols* indicate manually-measured values. The pressure transducers malfunctioned in the deep well at site 6 and from the shallow well at site 12 after July



deployment in February. Negative fluid velocities at sites 1, 2 and 7 represent groundwater discharge. Porosity was lowest at site 5, the only inner zone site, although values were similar between all four sites.

Sediment properties and stratigraphy

Cross–sections of the subsurface stratigraphy from cores revealed clear differences between inner and outer zones. Sediment cores from the outer zone were dominated by mangrove peat (Fig. 9a), while cores from the inner zone had substantial muddy deposits characteristic of mud flat or pool sediments interspersed with sections of peat (Fig. 9b). A cross-section of subsurface stratigraphy of sites perpendicular to the mangrove fringe shows the transition of sediments from the inner to outer zones (Fig. 10), with a decrease in mud flat/pool sediments moving toward Mangrove Lagoon.

The mean values for dry bulk density, percent water content, percent organic content, and sediment shear strength were all significantly different between

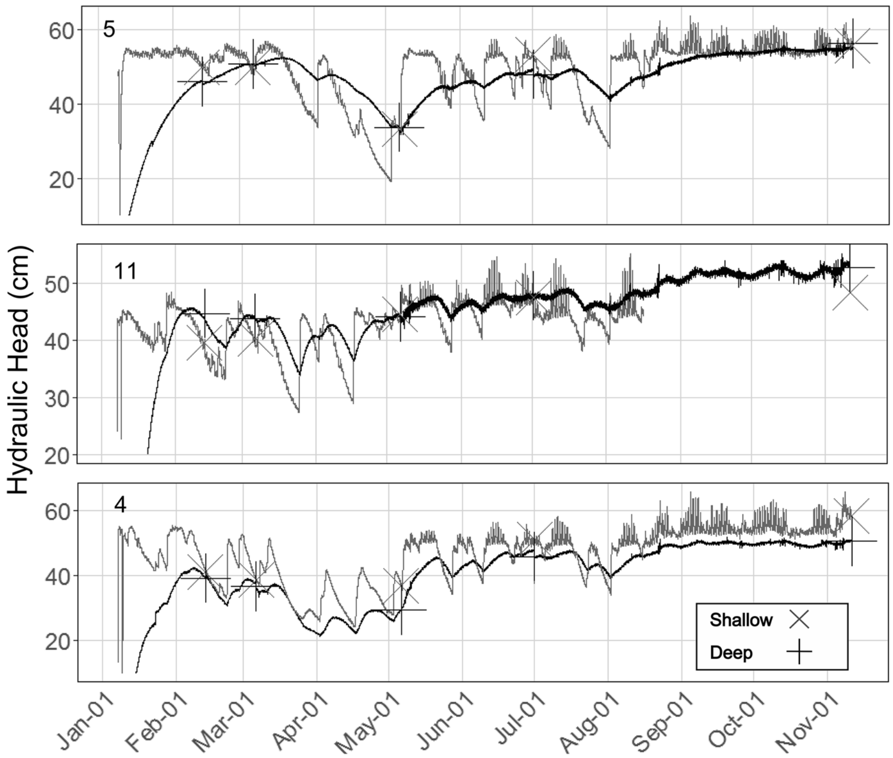


Fig. 5 Hydraulic head time series of both shallow (*grey*) and deep (*black*) wells installed in the inner zone, closest to the landfill. Hydraulic *head plots* are stacked based on spatial location of each site (Fig. 1). Continuous pressure transducer values are shown as *lines* while *symbols* indicate manually-measured values. When shallow hydraulic head is above deep hydraulic head, recharge is occurring and when the opposite is true, discharge is occurring

mangrove peat and mud flat/pool interpreted sediments (Table 4). Sediments interpreted as mud flat/pool had greater dry bulk density values, lower percent water and percent organic contents, and greater sediment shear strengths compared to samples interpreted as mangrove peat. Elemental analyses of dried sediment samples revealed many particles that contained high concentrations of heavy metals at depths between 12 to 112 cm at sites 4 and 5 (Table 5). Elements found include: Bismuth (Bi), Copper (Cu), Cobalt (Co), Lead (Pb), Titanium (Ti), Tin (Sn), and Iron (Fe). The majority of the heavy metals were found between depths of 12 to 22 cm, which contained fine grain sediments classified as mud flat/pool at both sites 4 and 5.

Discussion and conclusions

Mangrove swamp buffers lagoon from upland landfill

Within the mangrove system, horizontal groundwater flow direction changed seasonally (Fig. 3). From February to the beginning of May (the dry season), groundwater either flowed from the lagoon toward the landfill or flowed from both the landfill and the lagoon toward the center of the mangrove swamp. In July and November (the wet season), groundwater flowed from the landfill into the mangroves and out toward the lagoon (Fig. 3).

Hydraulic head responses to external factors also varied. Sites further inland responded more to precipitation events than tides while tides were more influential for sites located closer to the lagoon. A large rain event in May increased hydraulic head in the inner zone (with little change in the outer zone), but a second large rain event in August did not cause the same increase (Figs. 7, 8). The response of the inner zone to precipitation events may be partly determined by the hydraulic head at the time of the event. During the rain event in August, the hydraulic head of the shallow groundwater at site 5 was approximately 53 cm, compared to 33 cm during the rain event in May. The higher hydraulic head in August may indicate that sea-level rose above a threshold point after May, inundating the swamp, raising hydraulic head, and lessening the impacts of precipitation. This

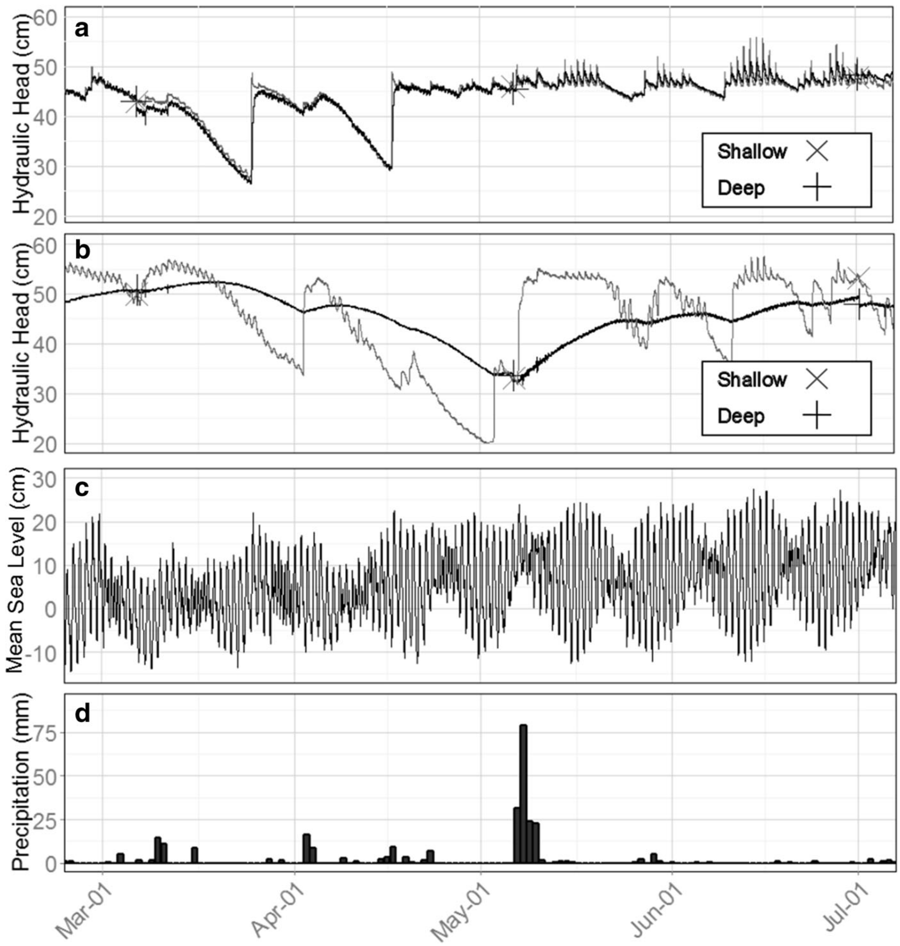


Fig. 6 Time series of hydraulic head (cm) for both shallow and deep groundwater at site 1 (**a**) and site 5 (**b**). *Lines* (*grey* for shallow, *black* for deep) represent values recorded with pressure transducers while symbols represent manually measured values. Below these are recorded water levels based on mean sea level from Charlotte Amalie Harbor (**c**) (NOAA/NOS/Center for Operational Oceanographic Products and Services) and precipitation from the Bovoni weather station (**d**) (The Weather Channel 2015)

threshold point may be about 45 cm above mean sea level, below which monthly spring high tide signals are observed in hydraulic head, but diurnal signals are not present (Figs. 4, 5, 6). An alternative explanation for the lack of hydraulic response may be that the rain was absorbed by soil that was dry due to high air temperatures and limited rain after the May event.

This threshold in hydraulic head responses to tide suggests the presence of a hydraulic barrier that restricts the interconnection between the mangrove swamp and the ocean from January to May. In the mangrove swamp, the red mangrove fringe (Fig. 1) separating the swamp from the lagoon may trap sediment and limit the hydraulic connection between the lagoon and swamp. Aerial photographs of the study area suggest the presence of a mineral sediment ridge at the interface between Mangrove Lagoon and the mangrove swamp and field investigations from Mangrove Lagoon by kayak confirm its presence. However, in this study, sediments sampled in the outer portion of the mangrove swamp were dominated by peat which is more permeable than inland mineral-rich sediments. Further work investigating the presence and hydrologic impact of the sediment ridge at this location is needed.

Stratigraphic connectivity can be seen throughout the inner zone as all three cores contained mud flat/pool sediments at the surface, followed by a mangrove

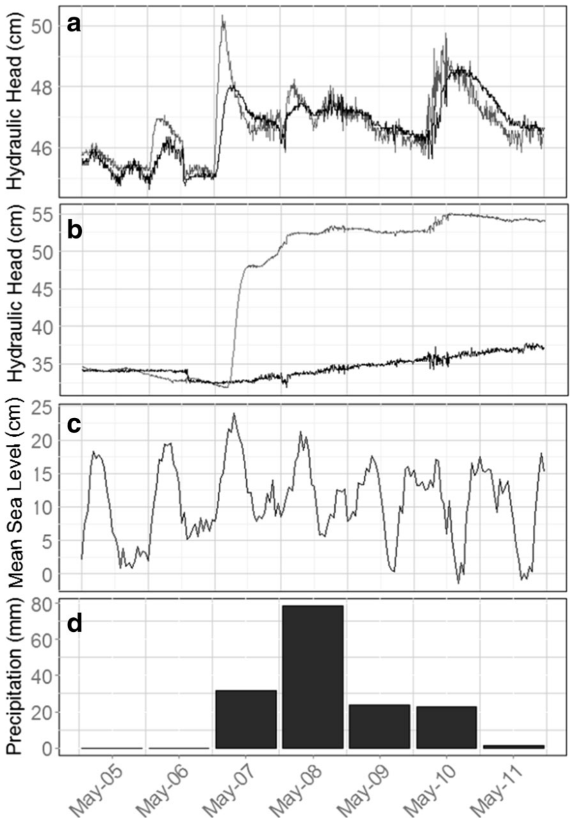


Fig. 7 Hydraulic head at site 1(**a**) and at site 5 (**b**) with sea-level and precipitation during a rain event May 7th–10th. *Grey lines* represent shallow hydraulic head and the *black lines* represent deep hydraulic head. Recorded water levels (**c**) are based on mean sea level from Charlotte Amalie harbor (NOAA/NOS/Center for Operational Oceanographic Products and Services) and precipitation (**d**) is from the Bovoni weather station (The Weather Channel 2015)

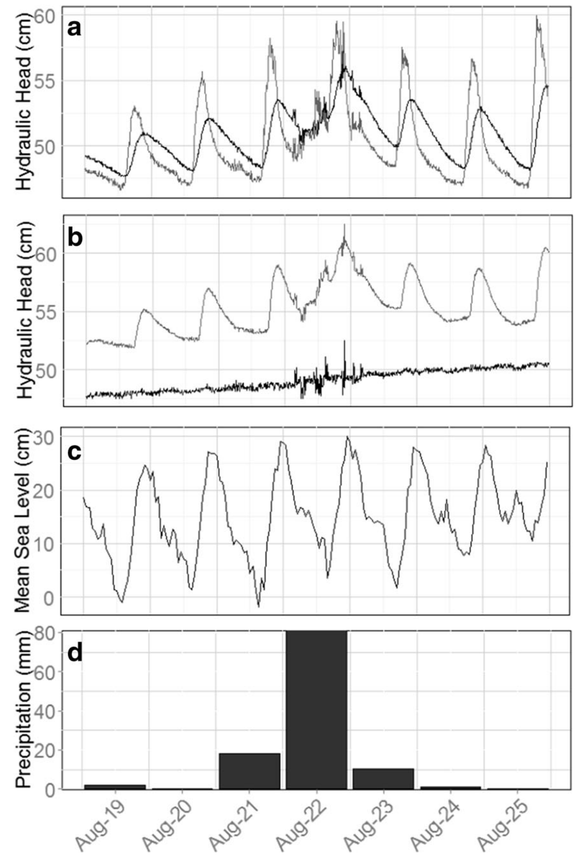


Fig. 8 Hydraulic head at site 1(**a**) and at site 5 (**b**) with sea-level and precipitation during a rain event August 21st–23rd. *Grey lines* represent shallow hydraulic head and the *black lines* represent deep hydraulic head. Recorded water levels (**c**) are based on mean sea level from Charlotte Amalie harbor (NOAA/NOS/Center for Operational Oceanographic Products and Services) and precipitation (**d**) is from the Bovoni weather station (The Weather Channel 2015)

peat section before transitioning back into a mud flat/pool environment (Fig. 9b). The mud flat/pool interpreted sediments common in the inner zone had significantly higher densities and shear strengths, and lower percent water and organic contents than mangrove peat sediments (Table 4). Significant differences between sediment properties are important as groundwater velocity is affected by the properties of the medium it flows through (Anderson 2007), and sections of denser fine sediments are less permeable than mangrove peat and impede groundwater flow (Harvey and Odum 1990). The denser, finer sediments in the mud flat/pool sections would slow vertical and horizontal groundwater flow, which was observed in the lagged and dampened responses of the deep hydraulic head compared to the shallow hydraulic head in the inner zone (Fig. 5). The alternating mangrove peat and mud flat/pool sediments in the inner zone (Fig. 9b) suggest that the swamp is dynamic and that favorable conditions for mangrove growth have occurred only at certain times in the past, likely due to changes in the water flow regime (sea-level change, alteration of freshwater availability from the upland environment, or storm events) or an increase in sediment supply from the upland.

From our data, we produced a conceptual model (Fig. 11) of the geologic structure of the mangrove swamp depicting idealized groundwater flow patterns during the wet and dry seasons. This model is based on the sediment cores of sites 5, 7, and 13 (Fig. 10) and inferred stratigraphic changes from the inner zone near

Table 2 Physical and chemical properties of groundwater and surface water samples

Sample ID	TDN Mg/L	Cr µg/L	Ni µg/L	Sn µg/L	Zn µg/L	PH Std. units	SC MS/cm
Reporting limit	0.1	20	40	100	40		
Site 1 Shallow	6.21	35.5	–	–	–	6.65	71.7
Site 1 Deep	4.45	39.2	–	–	–	6.7	70.3
Site 2 Surface							63.3
Site 2 Shallow	14.1	47.1	–	–	–	6.7	90
Site 2 Deep	4.71	41.7	–	–	–	6.93	86.9
Site 4 Surface							90
Site 4 Shallow	4.94	23.4	–	–	–	7.46	58.6
Site 4 Deep	5.14	30.9	–	–	–	6.6	72.3
Site 5 Surface	20.9	37.3	130	–	67.7	9.09	12.9
Site 5 Shallow	–	33.5	82.2	–	–		12.9
Site 5 Deep	15.0	51.7	–	–	–	6.75	101.2
Site 6 Surface							24.3
Site 7 Surface							23.2
Site 10 Surface							49.8
Site 11 Surface							43.3
Site 11 Shallow	15.2	47.1	–	–	–	6.92	84.6
Site 11 Deep	9.75	35.4	–	–	–	7.41	74
Surface Ditch	120	74.5	99	105	–	8.02	7.3
Site 12 Surface							53.2
Site 13 Surface							64.4

Presence of Pb and Cu were tested, but were not detected in any sample. SC is specific conductance. All samples were tested or collected for testing on January 9th, 2014 and filtered prior to analysis for heavy metals. *Dashes* represent samples that were not above the detection limit and *blank cells* indicates no analyses were performed

Table 3 Fluid velocity, sediment porosity and error (sum of squares) produced by fitting subsurface temperatures to the heat transport model

Site	Fluid velocity	Porosity	Error
1	-2.58×10^{-9}	0.9	31.8
2	-3.85×10^{-7}	0.9	23.5
5	2.78×10^{-7}	0.8	27.5
7	-9.18×10^{-8}	0.9	24.2

the landfill toward the outer zone near the lagoon for areas with no data. During the wet season, groundwater recharges into the mangrove swamp and flows from the direction of the landfill toward Mangrove Lagoon (Fig. 11a). During the dry season, groundwater discharges into the swamp (potentially driven by evapotranspiration) with flow from Mangrove Lagoon toward the landfill (Fig. 11b)

Mangrove swamp provides key ecosystem service

Spatial trends in the water chemistry indicate chemicals are being transported through surface water and shallow groundwater. Due to the seasonal variation in groundwater flow, chemical constituents from terrestrial sources are likely preferentially transported into the Mangrove Lagoon during the wet season (Fig. 3). However, geochemical trends suggest that the mangrove swamp is acting as a buffer, slowing or preventing heavy metals from entering the lagoon via groundwater or sediments (Tables 3). Groundwater recharge during periods of inundation, coupled with the seasonal shifts in lateral groundwater flow patterns and slow vertical groundwater velocities (Table 4) indicate that any dissolved chemical constituents will be trapped within the mangrove swamp. Our manual measurements indicate that hydraulic heads measured in shallow (1.5 m) sediments are similar to the hydraulic head of the surface water, suggesting that surface waters will have flow patterns similar to those measured in the shallow (1.5 m) sediments. These observations suggest that surface water exchange between Mangrove Lagoon and the mangrove swamp will also be limited and sediments carried by these flows will also be trapped in the mangrove swamp.

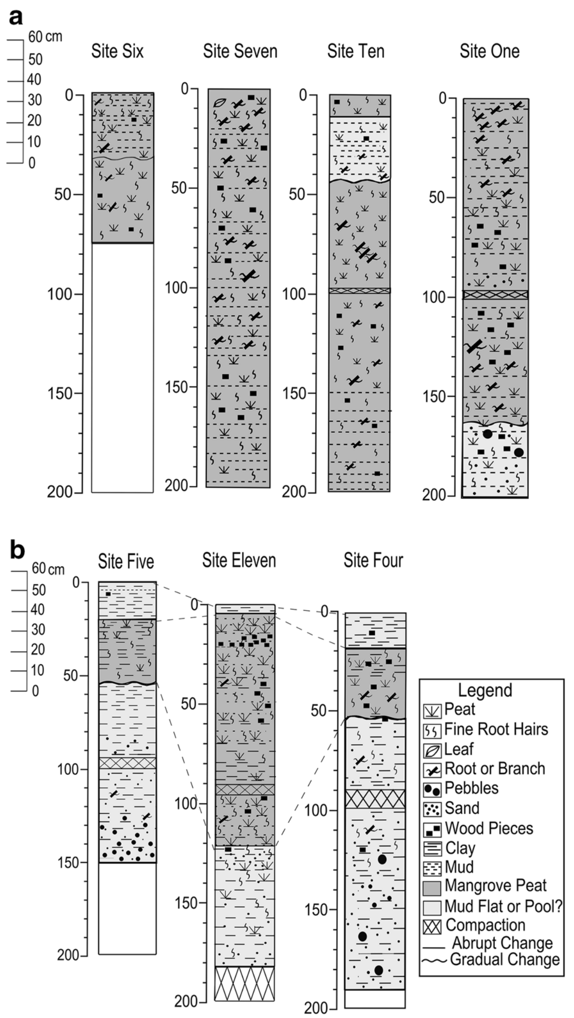


Fig. 9 Cross-section of core stratigraphy for the inner zone (**a**) and outer zone (**b**). Cores are corrected for topographic elevation. *Dotted lines* in between cores connect interpreted environments (mangrove peat or mud flat/pool

Chromium, nickel, and zinc were detected in groundwater samples, with nickel concentrations high enough to potentially impact aquatic organisms. The Criterion Continuous Concentration (CCC) and Criterion Maximum Concentration (CMC) are designated by the Environmental Protection Agency (EPA) as concentrations that can affect an aquatic community through continuous or brief exposure (EPA 2014). Nickel concentrations were above both CCC and CMC values in the surface water sample at site 5, the shallow groundwater sample at site 5, and the surface ditch (Table 6). However, nickel concentrations were not above the detection limit in any other groundwater

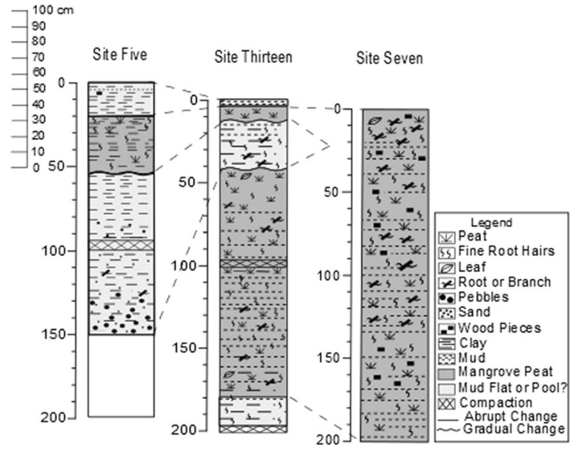


Fig. 10 Cross-section of core stratigraphy for sites 5, 13, and 7. Cores are corrected for topographic elevation. *Dotted lines* in between cores connect interpreted environments (mangrove peat or mud flat/pool)

sample, indicating that the metal is either not traveling widely from its source, or that it is becoming diluted as it is moved through the mangrove system. Zinc was not above its CCC or CMC in any sample. Safe drinking water standards were also surpassed at the surface ditch for chromium and nickel and site 5 for chromium, nickel, and pH (Table 6). The surface water samples represent overland flow that runs into the mangroves. The elevated concentrations of contaminants in these samples could affect not only the mangroves themselves, but also other organisms in the system (e.g., birds that drink from surface water puddles).

Any heavy metals that are potentially transported to Mangrove Lagoon would increase the already present risk to local fish and invertebrate communities as heavy metal accumulation has been found to limit fish size, impair DNA in gastropods, and interfere with reproduction in crustaceans (Canli and Atli 2003; Sarkar et al. 2014; Rodriguez et al. 2007). Sediments within Mangrove Lagoon have already been shown to contain high mean concentrations of chromium (26.7 µg/g), copper (48.4 µg/g), lead (18.7 µg/g), mercury (0.061 µg/g), nickel (10.5 µg/g), and zinc (109 µg/g; Pait et al. 2014). Four of these heavy metals were found in either the groundwater or sediments of the mangrove swamp, suggesting that some heavy metals may be transported through the swamp despite our findings, or that they are originating from another source. In this study, all water and

Table 4 Mean values of dry bulk density, percent water content, and percent organic content for both mangrove and mud flat interpreted environments and p-values of two-sample, unequal variance t-tests

Sediment property	Mangrove peat mean	Mud flat/pool mean	df	*P* value (two-tailed)
Dry bulk density (g/cm^3)	0.31	1.15	105	3.65E−19*
% Water content	71.73	36.10	126	5.56E−33*
% Organic content	38.19	14.09	169	4.03E−21*
Shear strength (kPa)	47.94	61.84	117	0.000356*

df degrees of freedom

* Indicates significant differences between sediment types

Table 5 Metallic elements found in sediment samples using a SEM and EDS

Site	Depth (cm)	Element	Formula	Weight percent
4	12	Cobalt	CoO	3.17
		Titanium	TiO2	0.84
4	22	Titanium	TiO2	57.79
4	22	Bismuth	Bi2O3	48.06
5	12	Tin	SnO2	37.18
5	12	Copper	CuO	16.28
		Nickel	NiO	3.11
5	19	Bismuth	Bi2O3	15.35
5	112	Titanium	TiO2	35.14
		Lead	PbO2	0.75

The weight percent (considered semi-quantitative) is the percent each element represents in the sediment particles analyzed under EDS. Metallic elements (besides Fe) were only found at sites 4 and 5, both located in the inner zone closest to the landfill

sediment samples with high heavy metal concentrations were collected from sites located closest to the landfill and no samples located close to the lagoon (sites 6, 7, 10, 12 and 13) tested above detection limits for heavy metals. The green-colored, leachate-like fluid in the surface ditch, however, had elevated levels of Cr, Ni, and Sn, compared to the mangrove samples that tested positively for these heavy metals, suggesting strongly that the landfill is one source for these contaminants. Additional sampling at higher spatial and temporal resolutions would help determine contamination hotspots within the mangrove swamp and additional migration pathways for these heavy metals.

Continued protection of the mangrove swamp is critical to marine protected area integrity

Continued rapid urbanization of the coastal zone in St. Thomas, U.S. Virgin Islands compromises the capacity of mangrove systems to provide key ecosystem services. Urbanization may have already altered the hydrology of the study area as the red mangroves near the lagoon were found to be inundated nearly year round when mangroves typically have a more restricted inundation by high tides (Lewis 2005). A technical document (Towle 1985) from the 1970s reports similar inundation periods to those found in this study, however there is very limited data about the study area before the development of the landfill. Furthermore, what historic information is available is focused on the Mangrove Lagoon, not the mangrove swamp. Runoff and sedimentation from the landfill seem like likely factors to have altered hydrology in the mangrove swamp, but a lack of data before this study limit inferences.

The mangrove swamp is currently facing impacts from multiple human activities, not just the landfill. For example, fine sediments and heavy metal particles in the inner zone of the swamp are likely due in part to erosion from upland development. Increased sediment input could be one reason for recent mangrove health decline, as excessive sedimentation can result in filling of wetlands (Horner 2000) and alteration of the hydrological regime, causing changes in plant community structure and composition (Lee et al. 2006). Further, changing climate will impact this mangrove swamp, particularly if sea level rises above the hydraulic barrier, increasing the inundation of the system and potentially increasing nutrient discharge

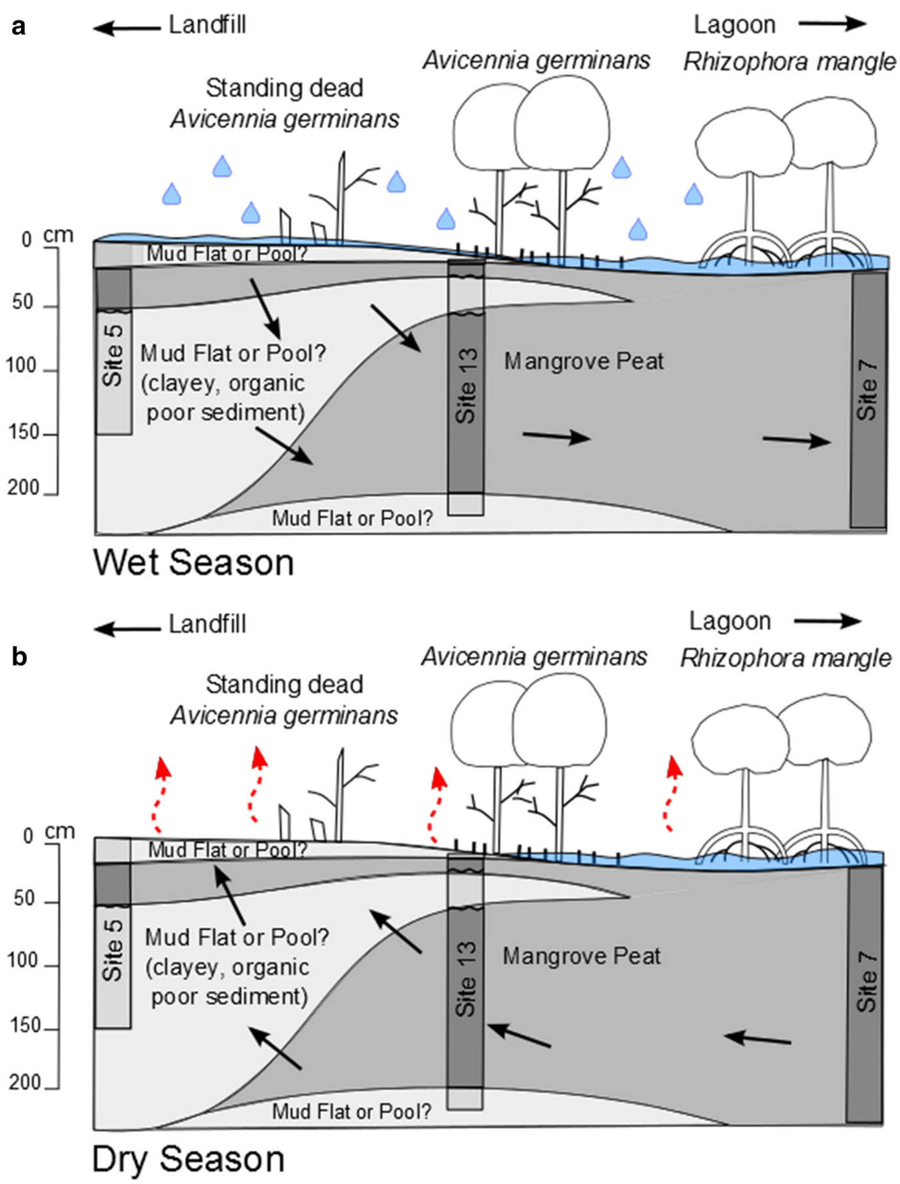


Fig. 11 Groundwater flow shown in a cross-sectional cartoon view of the mangrove swamp where Bovoni Landfill is to the *left* and Mangrove Lagoon is to the *right*. **a** During the wet season (or after any large rain event) and spring high tides, there is recharge and flow out to Mangrove Lagoon. **b** During low tides and after long periods of little precipitation (*red arrows* indicate evapotranspiration), there is discharge and flow into the mangrove swamp from Mangrove Lagoon

from the wetland to the lagoon (Wilson and Morris 2012).

Most pressing, however, is that this mangrove swamp is also facing the threat of direct habitat loss due to upcoming closure plans for Bovoni Landfill. Proposed closure plans (USA v. The Government of the Virgin Islands 2006) will result in cover materials and access roads being placed much closer to, or in, the mangrove swamp. Activities related to the landfill closure should be carefully planned to limit their impact on the mangrove swamp, as this study demonstrates one important ecosystem service that it provides. Removal or burial of part of this system could reduce its ability to act as a buffer, and disturbance of sediments may release trapped heavy metals and allow them to be carried toward the lagoon, a marine protected area containing significant natural and cultural resources (Horsley Witten 2013). Mangrove rehabilitation is an option for this study area, but careful consideration is needed before the start of any effort. Removal of stressors and rehabilitation of mangrove forests have shown to be more successful when started early and with an understanding of the mangrove forest's characterizations and source of

Table 6 Criterion Continuous Concentration (CCC), Criterion Maximum Concentration (CMC), and drinking water standards compared to surface ditch samples for Chromium, Nickel, Zinc and pH

Contaminant	CCC[a]	CMC[a]	Drinking water standards[b]	Site 5 (surface)	Site 5 (Shallow)	Site 5 (Deep)	Surface Ditch
Cr	50	1100	50	37.3	33.5	51.7	74.5
Ni	8.2	74	70	130	82.2	–	99
Zn	81	90	–	–	–	–	67.7
pH	6.5–8.5	–	6.5–8.5	9.09		6.75	8.05

All values except pH are in µg/L

[a] EPA (2014)

[b] WHO (2011)

– Standard not established/sample value below detection limit

stress (Lewis et al. 2016). Further study of the alterations of the area and decline in mangrove health should be completed before any rehabilitation efforts or further alterations are made to insure the survival and recovery of this important mangrove swamp.

Acknowledgements We would like to thank the USVI Department of Planning and Natural Resources, the Lana Vento Charitable Foundation, the Virgin Islands Water Resources Research Institute through 104b funding from the United States Geological Survey, and the Virgin Islands EPSCoR through the National Science Foundation for funding this work. We thank Paul Jobsis and the Center for Marine & Environmental Sciences at the University of the Virgin Islands for grant administration support, Anne-Marie Hoffman of The Nature Conservancy for project support and well installation, and Jen Kisabeth, Jon Jossart, and Joe Sellers for field assistance. Marty Yates provided assistance with the SEM/EDX at the University of Maine and Amelie Jensen, Emily Harris, Emma Swartz, Dana Cohen-Kaplan, and Florence MacGregor assisted with lab analyses at the Wells National Estuarine Research Reserve. This paper is contribution #168 from the Center for Marine & Environmental Studies at the University of the Virgin Islands.

Funding Mangroves buffer marine protected area from impacts of Bovoni Landfill, St. Thomas, United States Virgin Islands. Wetlands and Ecology Management. Jessica Keller*, Kristin Wilson Grimes, A.S Reeve, Renata Platenberg. * Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission, 2796 Overseas Highway, Suite 119, Marathon, FL 33050. Email: jessica.keller@myfwc.com. Virgin Islands Water Resources Research Institute, through 104b funding from the United States Geological Survey: Grant recipients: Kristin Wilson Grimes and Andrew Reeve. Virgin Islands Department of Planning and Natural Resources, Coastal Zone Management, no award number, Grant recipient: Kristin Wilson Grimes. Virgin Islands EPSCoR through funding from the National Science Foundation, award # 1355437. Lana Vento Charitable Foundation.

References

Agency for Toxic Substances and Disease Registry (2014) Toxic Substances Portal. http://www.atsdr.cdc.gov/. Accessed 17 Apr 2015

Alongi D (2002) Present state and future of the world's mangrove forests. Environ Conserv. doi:10.1017/s0376892902000231

Anderson M (2007) Introducing groundwater physics. Phys Today 60:42–47

Ball DF (1964) Loss-on-ignition as an estimate of organic matter and organic carbon in non-calcareous soils. J Soil Sci 15:84–92

Barbier E, Hacker S, Kennedy C, Koch EW, Stier AC, Sillman BR (2011) The value of estuarine and coastal ecosystem services. Ecol Monogr 81:169–193. doi:10.1890/10-1510.1

Bowden MJ, Fischman N, Cook P, Wood J, Omasta E (1970) Climate, water balance, and climatic change in the north-west Virgin Islands. Caribbean Research Institute, College of the Virgin Islands, St. Thomas

Calderon H, Weeda R, Uhlenbrook S (2014) Hydrological and geomorphological controls on the water balance components of a Mangrove Forest during the dry season in the Pacific Coast of Nicaragua. Wetlands 34:685–697. doi:10.1007/s13157-014-0534-1

Calvesbert RJ (1970) Climate of Puerto Rico and the US Virgin Islands. Climatography of the United States No. 60-52. US Department of Commerce

Canli M, Atli G (2003) The relationships between heavy metal (Cd, Cr, Cu, Fe, Pb, Zn) levels and the size of six Mediterranean fish species. Environ Pollut 121:129–136. doi:10.1016/s0269-7491(02)00194-x

Cavalcante R, Sousa F, Nascimento R et al (2009) The impact of urbanization on tropical mangroves (Fortaleza, Brazil): evidence from PAH distribution in sediments. J Environ Manag 91:328–335. doi:10.1016/j.jenvman.2009.08.020

Childs C (2004) Interpolating surfaces in ArcGIS spatial analyst. ArcUser (July-September): 32–35

Clark M (1998) Management implications of metal transfer pathways from a refuse tip to mangrove sediments. Sci

Total Environ 222:17–34. doi:10.1016/s0048-9697(98)00283-6

Craft C, Seneca E, Broome S (1991) Loss on ignition and kjeldahl digestion for estimating organic carbon and total nitrogen in estuarine marsh soils: calibration with dry combustion. Estuaries 14:175. doi:10.2307/1351691

Crossman S, Palada MC (2003) Production constraints to agricultural development. In: Proceedings of the First US Virgin Islands agricultural forum: prospects for sustainable agriculture in the VI, 22–23 April

Davis H, Marjorie Aelion C, McDermott S, Lawson A (2009) Identifying natural and anthropogenic sources of metals in urban and rural soils using GIS-based data, PCA, and spatial interpolation. Environ Pollut 157:2378–2385. doi:10.1016/j.envpol.2009.03.021

Department of Planning and Natural Resources (DPNR) (2003) Mangrove Lagoon and Benner Bay Area of Particular Concern management plan

Ellison A, Farnsworth E (1996) Anthropogenic disturbance of caribbean mangrove Ecosystems: past impacts, present trends, and future predictions. Biotropica 28:549. doi:10.2307/2389096

Environmental Protection Agency (EPA) (2014) National Recommended Water Quality Criteria. In: Epa.gov. https://www.epa.gov/wqc/national-recommended-water-quality-criteria-aquatic-life-criteria-table. Accessed 16 Apr 2015

ESRI (2013) ArcGIS 10.2 for Desktop. Version 10.2.0.3348

Ewel K, Twilley R, Ong JI (1998) Different kinds of mangrove forests provide different goods and services. Global Ecol Biogeogr 7(1):83–94

Fonseca E, Baptista Neto J, Silva C (2013) Heavy metal accumulation in mangrove sediments surrounding a large waste reservoir of a local metallurgical plant, Sepetiba Bay, SE, Brazil. Environ Earth Sci 70(2):643–650. doi:10.1007/s12665-012-2148-3

Gilman E, Ellison J, Duke N, Field C (2008) Threats to mangroves from climate change and adaptation options: a review. Aquat Bot 89:237–250. doi:10.1016/j.aquabot.2007.12.009

Giri C, Ochieng E, Tieszen LL, Zhu Z, Singh A, Loveland T, Masek J, Duke N (2011) Status and distribution of mangrove forests of the world using earth observation satellite data. Glob Ecol Biogeogr 20:154–159. doi:10.1111/j.1466-8238.2010.00584.x

Goldstein J, Newbury DE, Echlin P, Joy DC, Romig AD Jr, Lyman CE, Fiori C, Lifshin E (1992) Scanning electron microscopy and X-Ray microanalysis: a text for biologists, materials scientists and geologists, 2nd edn. Plenum Press, New York

Harvey JW, Odum WE (1990) The influence of tidal marshes on upland groundwater discharge to estuaries. Biogeochemistry 10:217–236. doi:10.1007/bf00003145

Heiri O, Lotter AF, Lemcke G (2001) Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. J Paleolimnol 25:101–110

Horner RR (2000) Introduction. In: Azous AL (ed) Wetlands and Urbanization Implications for the Future. Lewis Publishers, Boca Raton, pp 1–21

Horsley Witten (2013) St. Thomas East End Reserves watershed management plan. Prepared for NOAA Coral Reef Conservation Program, VI Department of Planning and Natural Resources, and The Nature Conservancy p. 73

Islam MS, Tanaka M (2004) Impacts of pollution on coastal and marine ecosystems including coastal and marine fisheries and approach for management: a review and synthesis. Mar Pollut Bull 48:624–649. doi:10.1016/j.marpolbul.2003.12.004

Ismail S, Saifullah SM, Khan SH (2014) Bio-Geochemical studies of Indus Delta mangrove ecosystem through heavy metal assessment. Pak J Bot 46(4):1277–1285

Jardine S, Siikamäki J (2014) A global predictive model of carbon in mangrove soils. Environ Res Lett 9:104013. doi:10.1088/1748-9326/9/10/104013

Jones R (2010) Environmental contamination associated with a marine landfill ('seafill') beside a coral reef. Mar Pollut Bull 60:1993–2006. doi:10.1016/j.marpolbul.2010.07.028

Kathiresan K, Rajendran N (2005) Coastal mangrove forests mitigated tsunami. Estuar Coast Shelf Sci 65:601–606. doi:10.1016/j.ecss.2005.06.022

Kjerfve B, Drude de Lacerda L, Rezende CE, Coelho Ovalle AR (1999) Hydrological and hydrogeochemical variations in mangrove ecosystems. p. 71-82. In: A. Yáñez–Arancibia y A. L. Lara–Domínquez (eds.). Ecosistemas de Manglar en América Tropical. Instituto de Ecología A.C. México, UICN/ORMA, Costa Rica, NOAA/NMFS Silver Spring MD USA. 380 p

Lee SY, Dunn RJK, Young RA, Connolly RM, Dale PER, Dehayr R, Lemckert CJ, Mckinnon S, Powell B, Teasdale PR, Welsh DT (2006) Impact of urbanization on coastal wetland structure and function. Aust Ecol 31:149–163. doi:10.1111/j.1442-9993.2006.01581.x

Lee SY, Primavera JH, Dahdouh-Guebas F, Mckee K, Bosire JO, Cannicci S, Diele K, Fromard F, Koedam N, Marchand C, Mendelssohn I, Mukherjee N, Record S (2014) Ecological role and services of tropical mangrove ecosystems: a reassessment. Glob Ecol Biogeogr 23:726–743. doi:10.1111/geb.12155

Lewis RR (2005) Ecological engineering for successful management and restoration of mangrove forests. Ecol Eng 24:403–418

Lewis RR, Milbrandt EC, Brown B, Krauss KW, Rovai AS, Beever JW, Flynn LL (2016) Stress in mangrove forests: early detection and preemptive rehabilitation are essential for future successful worldwide mangrove forest management. Mar Pollut Bull 109(2):764–771

MacFarlane G, Koller C, Blomberg S (2007) Accumulation and partitioning of heavy metals in mangroves: a synthesis of field-based studies. Chemosphere 69:1454–1464. doi:10.1016/j.chemosphere.2007.04.059

Mazda Y, Magi M, Kogo M, Hong PN (1997) Mangroves as a coastal protection from waves in the Tong King delta. Vietnam Mangroves Salt Marshes 1:127–135

McKee KL, Faulkner PL (2000) Restoration of biogeochemical function in mangrove forests. Restor Ecol 8(3):247–259. doi:10.1046/j.1526-100x.2000.80036.x

Montalto F, Steenhuis T, Parlange J (2006) The hydrology of piermont marsh, a reference for tidal marsh restoration in the Hudson river estuary, New York. J Hydrol 316:108–128. doi:10.1016/j.jhydrol.2005.03.043

Mumby PJ, Edwards AJ, Arias-Gonzalez JE, Lindeman KC, Blackwell PG, Gall A, Gorczynska MI, Harborne AR, Pescod CL, Renken H, Wabnitz CCC, Llewellyn G (2004) Mangroves enhance the biomass of coral reef fish communities in the Caribbean. Nature 457:533–536. doi:10.1038/nature02286

Nagelkerken I, Blaber SJM, Bouillon S, Green P, Haywood M, Kirton LG, Meynecke JO, Pawlik J, Penrose HM, Sasekumar A, Somerfield PJ (2008) The habitat function of mangroves for terrestrial and marine fauna: a review. Aquat Bot 89:155–185. doi:10.1016/j.aquabot.2007.12.007

NOAA Average Rainfall. http://www.srh.noaa.gov/sju/?n=averagerainfall. Accessed 11 Apr 2016

NOAA Tides and Currents. http://tidesandcurrents.noaa.gov/. Accessed 08 Mar 2016

Pait AS, Hartwell SI, Manson AL, Warner RA, Jeffery CFG, Hoffman AM, Apeti DA, Pittman SJ (2014) An assessment of chemical contaminants in sediments from the St. Thomas East End Reserves, St. Thomas, USVI. Environ Monit Assess 186(8):4793–4806. doi:10.1007/s10661-014-3738-1

Platenberg R (2006) Wetlands conservation plan for St. Thomas and St. John, U.S. Virgin Islands. Available: https://data.nodc.noaa.gov/coris/library/NOAA/CRCP/other/other_crcp_publications/Watershed_USVI/steer_exisiting_studies/wetlandconservationplansttandstj.pdf

Rabah FKJ, Said MG, Ali AS (2011) Effect of GIS interpolation techniques on the accuracy of the spatial representation of groundwater monitoring data in Gaza Strip. J Environ Sci Technol 4(6):579–589. doi:10.3923/jest.2011.579.589

Rajkumar K, Ramanathan A, Behera P (2012) Characterization of clay minerals in the Sundarban mangroves river sediments by SEM/EDS. J Geol Soc India 80:429–434. doi:10.1007/s12594-012-0161-5

Rau GS, Andersen MS, McCallum AM, Roshan H, Scworth RI (2014) Heat as a tracer to quantify water flow in near-surface sediments. Earth Sci Rev 129:40–58. doi:10.1016/j.earscirev.2013.10.015

Rodríguez E, Medesani D, Fingerman M (2007) Endocrine disruption in crustaceans due to pollutants: a review. Comp Biochem Physiol A 146:661–671. doi:10.1016/j.cbpa.2006.04.030

Sarkar A, Bhagat J, Sarker S (2014) Evaluation of impairment of DNA in marine gastropod, Morula granulata as a biomarker of marine pollution. Ecotoxicol Environ Saf 106:253–261. doi:10.1016/j.ecoenv.2014.04.023

Shepard C, Crain C, Beck M (2011) The protective role of coastal marshes: a systematic review and meta-analysis. PLoS ONE 6:e27374. doi:10.1371/journal.pone.0027374

Spalding M, Blasco F, Field CD (1997) World Mangrove Atlas. The International Society for Mangrove Ecosystems, Okinawa, p 178

Stallman R (1965) Steady one-dimensional fluid flow in a semi-infinite porous medium with sinusoidal surface temperature. J Geophys Res 70:2821–2827. doi:10.1029/jz070i012p02821

STEER (2011) St. Thomas East End Reserves Management Plan. St. Thomas, USVI

Sthevan R, Krishnan SR, Patterson J (2011) Preliminary survey on the heavy metal pollution in Punnakayal Estuary of Tuticorin Coast, Tamil Nadu. Global J Environ Res 5(2):89–96

Tam NFY, Wong Y (1999) Mangrove soils in removing pollutants from municipal wastewater of different salinities. J Environ Quality 28:556. doi:10.2134/jeq1999.00472425002800020021x

Towle EL (1985) The island microcosm. Island Resources Foundation, Caribbean Office

Turner RE (2011) Beneath the salt march canopy: loss of soil strength with increasing nutrient loads. Estuaries Coasts 34:1084–1093

Turner R, Howes B, Teal J et al (2009) Salt marshes and eutrophication: an unsustainable outcome. Limnol Oceanogr 54:1634–1642. doi:10.4319/lo.2009.54.5.1634

US Census Bureau (2013) 2010 Census of Population and Housing, U.S. Virgin Islands Summary File: Technical Documentation

USA v. The Government of the Virgin Islands, Virgin Islands Port Authority, Virgin Islands Waste Management Authority, Hodge J, and Hodge Z. 2006. No. 3:10-cv-48. District Court of the Virgin Island

U.S.V.I Bureau of Economic Research (2013) U.S. Virgin Islands annual tourism indicators. Retrieved from http://www.usviber.org/publications.htm

Valiela I, Bowen JL, York JK (2001) Mangrove forests: one of the world's threatened major tropical environments. Bioscience 51(10):807–815. doi:10.1641/0006-3568(2001)051[0807:mfootw]2.0.co;2

Vandenbohede A, Lebbe L (2010) Parameter estimation based on vertical heat transport in the surficial zone. Hydrogeol J 18:931–943. doi:10.1007/s10040-009-0557-5

Wigand C, Roman C, Davey E et al (2014) Below the disappearing marshes of an urban estuary: historic nitrogen trends and soil structure. Ecol Appl 24:633–649. doi:10.1890/13-0594.1

Wilson A, Morris J (2012) The influence of tidal forcing on groundwater flow and nutrient exchange in a salt marsh-dominated estuary. Biogeochemistry 108:27–38. doi:10.1007/s10533-010-9570-y

World Health Organization (WHO) (2011) Guideline for drinking water quality, 4th edn. World Health organization, Geneva, p 541

Zadora G, Brożek-Mucha Z (2003) SEM–EDX—a useful tool for forensic examinations. Mater Chem Phys 81:345–348. doi:10.1016/s0254-0584(03)00018-x

Zhang K, Liu H, Li Y, Xu H, Shen J, Rhome J, Smith TJ (2012) The role of mangroves in attenuating storm surges. Estuar Coast Shelf Sci 102–103:11–23. doi:10.1016/j.ecss.2012.02.021

Zhang Z, Xu X, Sun Y, Yu S, Chen Y, Peng J (2014) Heavy metal and organic contaminants in mangrove ecosystems of China: a review. Environ Sci Pollut Res 21:11938–11950. doi:10.1007/s11356-014-3100-8