



Mangrove Surface Elevation Loss after Tree fall during Extreme Weather

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Abstract

Fringing mangrove forests are likely to experience increasingly severe weather events under climate change, including an increased frequency of tropical storms that may lead to mangrove mortality. Mangrove mortality can result in soil surface elevation loss through the death and compaction of roots, which can in turn alter the hydrology of the ecosystem and reduce its resilience to sea-level rise. In subtropical Queensland, Australia, an extreme weather event in early 2021 resulted in the death and toppling of large *Avicennia marina* trees on a creek edge in mainland Quandamooka (Moreton Bay). A set of three rod surface elevation tables were located meters away from the uprooted mangroves and were opportunistically measured for twenty months after the tree deaths. Surface elevation in the mangrove forest decreased by 26.3 ± 3.59 mm year⁻¹ following the severe weather event, when estimated using a traditional annualization approach for surface elevation loss data. Loss estimates using generalised additive models were able to account for non-linear change in surface elevation and were of the same direction as annualised estimates but an order of magnitude lower (-4.82 ± 0.03 mm year⁻¹). Surface elevation loss was the result of local root compaction, and subsequent recovery of elevation was the result of mangrove forest recovery through neighbouring tree root growth and canopy infilling. This study was the first to follow the change in soil surface elevation after the uprooting of *A. marina* at a single tree scale, providing insights for mangrove forest resilience under future sea-level rise.

Keywords Canopy gap · Subsidence · Flood · Disturbance · RSET · GAM

Introduction

Ongoing global climate change will impact weather patterns and place increasing stress on marine and intertidal ecosystems (Friess et al. 2022). In subtropical Queensland, Australia, seaward fringing mangrove forests are likely to experience an increased frequency of tropical storms, delivering stronger winds, higher rainfall, storm surges and prolonged flooding, creating a higher risk of tree mortality (Ward et al. 2016). Mangrove tree mortality can result in local soil surface elevation loss as extensive mangrove root systems die, compact and decompose, causing soil subsidence (Cahoon et al. 2006; Morris et al. 2023). Localised subsidence can alter hydrology in mangroves (Chambers et

al. 2019), leading to increases in tidal inundation that may increase vulnerability to sea-level rise and decrease suitability of subsided patches for propagule establishment and seedling growth (Lugo 1997; Temmerman et al. 2023). While subsidence of mangrove soils due to mortality after intense storms has been assessed in the peat soils of the Caribbean (Cahoon et al. 2003; Osland et al. 2020), there have been few assessments in soils with higher proportions of mineral content (Gijssman et al. 2021) or at sites in the Indo-Pacific region. Additionally, most studies of elevation loss associated with mangrove tree death have monitored large scale damage or dieback events (hectares), so there is limited knowledge of the change in surface elevation following small scale tree death (10s of m²) or individual tree falls (Gijssman et al. 2021; Krauss and Osland 2019; Lang'At et al. 2014).

Mangrove tree mortality can result in surface elevation losses in the soil profile from rapidly decomposing root biomass (Morris et al. 2023). Decomposing root systems and compression of the soil volume following tree mortality

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have caused significant soil elevation loss. For example, subsidence of 11 mm year⁻¹ was observed in a Honduras basin forest composed of peat soils and dominated by *Rhizophora mangle*, following mass mangrove mortality from Hurricane Mitch (Cahoon et al. 2003). After Hurricane Wilma in Tampa Bay, the mass mortality of *R. mangle* in peat soils caused subsidence of 8.8 mm year⁻¹ (Osland et al. 2020). In minerogenic sediments, a surface elevation loss of 10.6 mm year⁻¹ was measured in Micronesia following the harvesting of *Xylocarpus granatum* (Krauss et al. 2010), and in Kenya, experimental girdling of *R. mucronata* mangroves growing in mineral soils resulted in subsidence of 32 ± 8.4 mm year⁻¹ (Lang'At et al. 2014). Locally, decomposition experiments with *A. marina* roots in Quandamooka reported up to 50% decomposition within six months, demonstrating that root decomposition can advance rapidly even in anoxic mangrove sediments (Hayes et al. 2017). These studies suggest that in both peat and minerogenic settings root mortality may have a large effect on soil surface elevation.

The mortality of individual trees results in canopy gaps, and subsequent regeneration and maintenance of a multi-generational forest is an important process in maintaining the health of the mangrove ecosystem (Duke 2001). Increases in light availability caused by mangrove canopy gaps increase seedling survival and growth, and gap regeneration can increase surface elevation and enhance accretion (Amir and Duke 2019; Krauss et al. 2008; Kumara et al. 2010), but recovery of soil surface elevation in canopy gaps has not yet been studied in minerogenic soils.

More frequent intense storms are expected for south-east Queensland under future climate change (Herold et al. 2021), causing increased rainfall and winds, and subsequently increased river flows into estuaries and increased local sea levels, which are broadly expected to negatively impact coastal fringing mangrove forests. Saintilan et al. (2020) recently predicted that mangroves will be able to withstand sea-level rise of 6.1 mm year⁻¹ by maintaining their relative elevation in the tidal range through increasing their surface elevation. Increases in surface elevation are due to a range of biological and physical adaptive processes that vary on local and regional scales (Cahoon 2024; Krauss et al. 2014). Mangroves have differing tolerances of extreme events and differing capacities to adapt following disturbance. For example, *Avicennia* can resprout by epicormic growth after disturbance (Granek and Ruttenberg 2008). Mangrove recovery from disturbance following extreme events may also influence rates of surface elevation change which ultimately may influence vulnerability to sea-level rise.

We assessed the impact of extreme storm events on a creek fringing mangrove in Quandamooka (Moreton Bay),

Queensland, Australia, through the monitoring of surface elevation, canopy growth and root growth at a single-tree-neighbourhood scale. Following two tree fall events, we tested the following hypotheses:

1. Mangrove mortality causes an overall loss of soil surface elevation due to soil subsidence surrounding dead trees. Based on the reports of the extensive horizontal distribution of *A. marina* roots, subsidence may occur up to 20 m from the dead tree — “cable roots originate at the base of the trunk and extend horizontally for long distances through the substrate” (Tomlinson 1986); and “the pneumatophores of *Sonneratia* and *Avicennia* are slender and cone shaped, standing erect and aligned on the cable root of 1–20 m or more in length, which spread horizontally in the soil” (Srikanth et al. 2016).
2. Recovery of soil surface elevation will be positively correlated with regrowth of vegetation and roots.

Methods

Site Description

Warra Warra (Point Halloran Conservation Area) is adjacent to Eprapah Creek in the shire of Talwalpin (Redland Bay), Queensland, Australia. The 52 km creek runs from Jungalpin (Mount Cotton) in the west and opens into the southern part of Quandamooka with a catchment area of around 39 km² and is impacted by flash flooding and tidal inundation (Redlands City Council 2023). First Nations people lived on the banks of Eprapah Creek before European invasion and utilised the food and materials of both the upstream freshwater sections, and the downstream tidal sections and mangroves (Redlands (Qld.), Council Library Services & McNeill, 2021). The conservation area was officially opened in 1994 and contains a well-developed fringing mangrove forest, comprising seaward areas dominated by large *A. marina*, in front of landward *R. stylosa*, and backed by landward saltmarsh and cyanobacterial mats. In the supratidal zone, there is a well-developed *Melaleuca* and *Casuarina* forest (Fig. 1a).

Characterisation of the Extreme Events

There were two extreme weather events associated with tree falls at our site: a severe storm in March 2021, and severe flooding in March 2022.

In March 2021, there was unusually high monthly rainfall of 344 mm (cf. average monthly rainfall for March 2014–2020 = 171 mm), with 230 mm falling during the last week

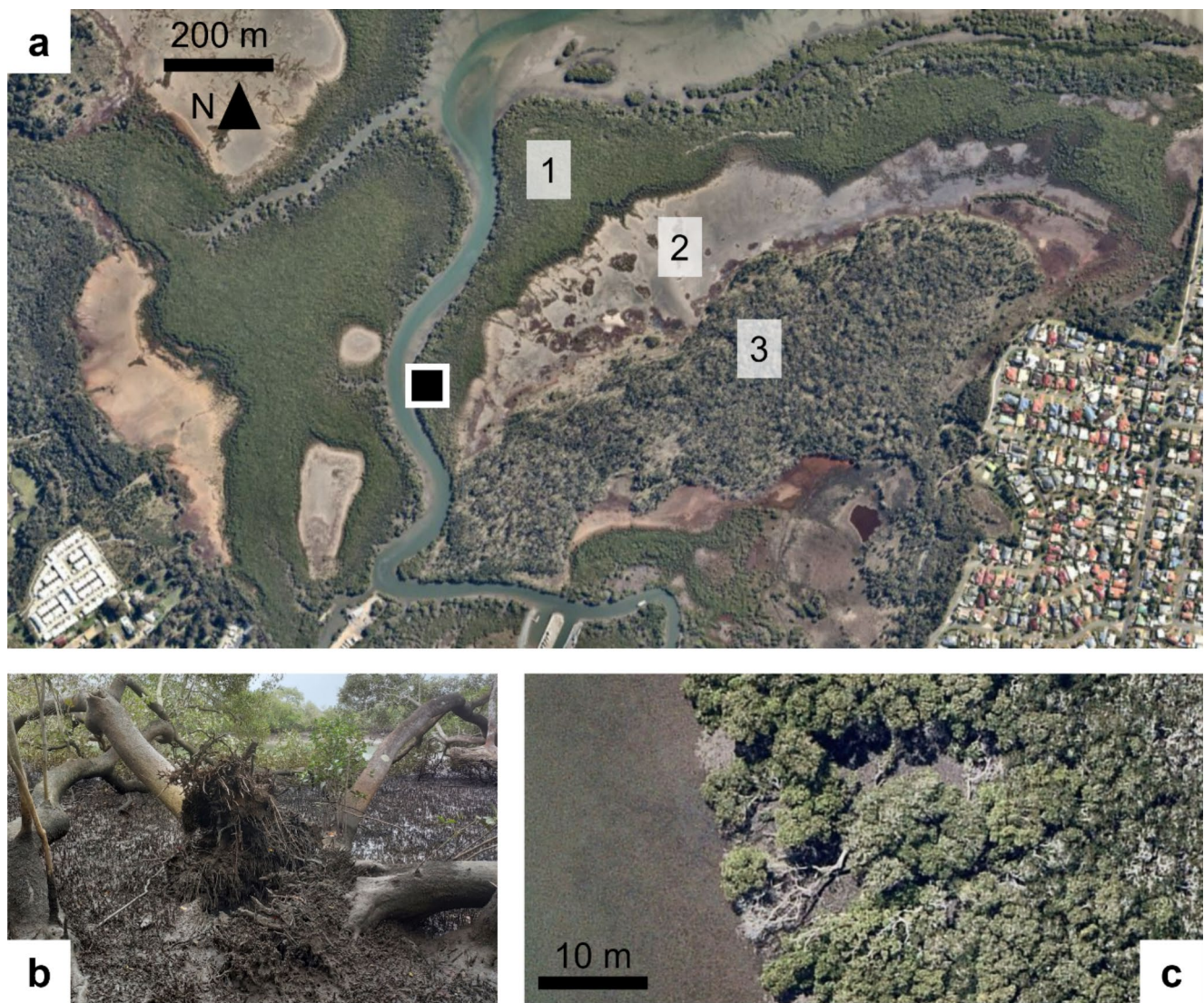


Fig. 1 Panel (a) shows location of Warra Warra (Point Halloran Conservation Park) on the south-western side of Quandamooka (Moreton Bay), Queensland, Australia, noting (1) the fringing mangrove forest along Eprapah Creek; (2) the landward saltmarsh and cyanobacterial mats; and (3) the supratidal forest of *Casuarina* and *Melaleuca*, backed

by a terrestrial forest at the urban edge. Black square in the mangrove represents the location of the fallen trees. Panel (b) shows the first fallen *Avicennia marina* in April 2021. Width of the uprooted root ball was 73 cm. Panel (c) shows the canopy gap from this and adjacent fallen trees, visible on satellite imagery in the months following

of March and 125 mm in one day alone (Bureau of Meteorology 2021). Local sea levels for April 2021 following the prolonged severe storms were the third highest recorded monthly sea levels since 2000, with only the early 2011 and early 2022 south-east Queensland floods having higher sea level records (Inner West Bar, Station ID 822; Holgate et al. 2013; Tide Gauge Data 2023). High tides during the week of April 2021 were also predicted to be at 2.5 m (maximum king tides for the area = 2.78 m) and were much higher than predicted (Bureau of Meteorology 2023). Additionally, prevailing moderate northerly winds throughout March and April aligned with the northerly entrance of Eprapah Creek from Quandamooka. These winds had long fetch distances

across central Quandamooka, further increasing water levels and wave attack at the mouth of the creek.

Following these months of extreme weather, in late March 2021, a large *A. marina* individual (diameter at breast height (DBH) = 93 cm) fell (Fig. 1a). The uprooted mangrove left a 34 cm deep hole in the sediment (Fig. 1b). Eight meters to the north, another smaller *A. marina* tree (DBH = 24 cm) also fell.

In March 2022, similar extreme weather caused extensive flooding and the second highest recorded monthly sea levels for the locality since 2000 (Holgate et al. 2013; Tide Gauge Data 2023). Shortly afterwards, another medium-sized *A. marina* (DBH = 25 cm) toppled about 50 m to

the north-east of the previously fallen trees. Both sets of fallen trees were near to an array of rod surface elevation benchmarks (RSETs, see section below). There were other uprooted mangroves within the forest after both events, but not in the proximity of the RSET benchmarks.

Measurements of Soil Surface Elevation

Twenty-five meters to the north of the original fallen *A. marina* were a series of three RSET benchmarks, approximately 25 m apart, that were installed in 2007 and have been re-measured annually (Bennion et al. 2024). Changes in soil surface elevation can be measured to 1 mm accuracy (reported error ± 1.3 mm) using RSETs which are attached to a permanent benchmark (a series of connected steel rods) anchored deep in the soil profile (Cahoon et al. 2002). Benchmarks are installed into bedrock (3.6 m depth at our site). To measure elevation, a detachable arm is attached to the benchmark and nine fibreglass pins are carefully lowered to touch the soil surface. The change in surface elevation is recorded as the length of each pin protruding above the measurement arm, with measurements taken from four cardinal directions and averaged to show change for a given RSET.

Surface elevation measured in December 2020 was used as the reference point for elevation change after the March 2021 weather event. The RSETs were measured in early April 2021 with subsequent monitoring at two-week intervals for eight weeks, followed by monthly and then two-monthly measurements to December 2021. Surface elevation was measured again following the March 2022 flooding, with follow-up monitoring in July and December 2022 to record any surface elevation change. Three saltmarsh RSET benchmarks to the landward of the mangroves were measured at the same time and used to evaluate loss or gain of elevation that may be due to deeper geological processes occurring over the entire reserve.

Surface accretion (sediment accumulating on the surface of the soil) was measured using marker horizons. A white powder (feldspar) is poured over the surface of the soil with the accumulation of sediments on top of the white layer, after some time, measured by pushing a modified syringe (diameter 3 mm) through the soil surface to take a small core and measure the depth of the buried white layer. Measurements of surface accretion were taken annually.

Soil thickness in the root zone (top 25–40 cm of sediment, Adame et al. 2017) at each RSET benchmark was measured using five 1 mm diameter stainless steel pins inserted 25 cm into the sediment, where the change in the length of pin protruding from the soil surface was indicative of changes in the thickness of the root zone soil. Changes in the soil thickness in the root zone surrounding the uprooted mangrove

were measured from April 2021 using 20 pins inserted into the soil to two depths (8 pins at 25 cm depth and 12 pins at 50 cm depth). We used two different pin depths to evaluate whether changes in root zone soil thickness mainly occurred in the surface or deeper layers. Changes in the length of pin protruding from the surface were recorded as changes in root zone soil thickness, which could be positive or negative.

The relative elevation of the three mangrove sites was similar (elevation relative to AHD; mangrove 1–0.407 m, mangrove 2–0.412 m, and mangrove 3–0.436 m) (Lovelock et al. 2011). Tidal inundation of each mangrove RSET benchmark occurs within 10 min (pers. obs.). Previous studies found that sediment delivered to the fringing mangroves was homogenous over the site at 1.83 ± 0.13 g cm⁻² per spring tide, decreasing towards the landward saltmarsh (0.03 ± 0.03 g cm⁻²) (Adame et al., 2010). Dissolved inorganic nitrogen and phosphorus in tidal water ranged between 0.32 and 1.2 mg L⁻¹ and 0.041–0.28 mg L⁻¹ respectively throughout 2021/22, with highest values recorded following extreme hydrological events (Department of Environment and Science 2024). The suspended sediment concentrations in Eprapah Creek were 7 nephelometric turbidity unit (NTU) before flooding (15/02/2022) and 72 NTU following flooding (7/3/2022) (Department of Environment and Science 2024). Previous fallen mangroves within our study forest were identified using ultra-high-resolution aerial imagery (Nearmap 2024). Two canopy gaps related to fallen trees were observed in imagery from 2008 (following defoliation in 2006, The State of Queensland 2024), with gaps equalling 132.8 m² and 179.3 m² respectively. Gap infilling was observed in 2018 (10 years later) and by 2023, 80% of gap one and 66% of gap two were infilled.

Vegetation Monitoring

Morphometrics (DBH, cm and height, m) of the fallen mangroves were recorded in April 2021. Additionally, we calculated the openness of the canopy gap (%) using a densiometer (Lemmon 1956), with subsequent measurements in December 2023 to record any regrowth or gap infilling. Root production was measured using root in-growth bags (Steen and Håkansson 1987). Five in-growth bags were deployed from December 2021 to June 2022 surrounding the fallen tree and at each mangrove RSET (total $n = 20$). A second set of root in-growth bags were deployed from June 2022 to December 2022. Bags were made of 70% nylon shade cloth (2 mm gaps) and filled with a mix of peat moss, perlite, and sand (ratio 1:1:2) with a bulk density of 0.9 g cm⁻³. Although the root bag fill was denser than the surrounding sediment at our site (0.44 ± 0.04 g cm⁻³, Bennion et al. 2024), we did not expect this to impede root in-growth, as a previous study in Quandamooka found no significant

differences in root growth into root bags of different densities ($0.2\text{--}1.2\text{ g cm}^{-3}$) (Ola et al. 2019). After collection, root bags were sieved to 0.5 mm and the clean roots were dried to constant mass to determine root biomass (g of dry root per g of soil).

Data Analysis

We analysed our surface elevation change data with two methods to maximise the utility of our report in the broader RSET literature. To reflect emerging analysis methods which can account for non-linear rates of change, we modelled surface elevation change over time (mm year^{-1}) using generalized additive models (GAMs) (Feher et al. 2023). We considered the rate of change to be meaningfully non-linear over time when the effective degrees of freedom (edf) of a GAM was greater than 2 (edf = 1 indicates a linear trend) (Wood 2017; Zuur et al. 2009). Allowing for non-linear change is particularly important when assessing the impacts of extreme event disturbances, which were hypothesized to produce sub-annual variation in surface elevation. To derive estimates of surface elevation rates of change from GAMs, we calculated 500 equally spaced first derivatives of the fitted function for each RSET (Feher et al. 2023). Rates were estimated as the mean (and standard error) of these derivatives, either across the complete monitoring period, or before and after the first tree fall event.

Additionally, to allow for more comparisons with the existing RSET literature, we also estimated surface elevation change using an annualisation method, which reflects the linear analysis techniques of established standard operating procedures (Cahoon et al. 1995; Lynch et al. 2015). We multiplied surface elevation change from the month following each of the extreme events by 12 to give an estimate of annual change. This method of calculation allowed comparison with other studies that used the same method but is not ideal because we expected to see sub-annual variation in the rate of surface elevation change during post-event recovery.

We used a beta regression to test for differences in root growth using our root in-growth bag data (because these data were proportions of root weight per soil weight), between sites and between deployment periods. We used a t-test to test for differences in change in root zone soil thickness around the fallen tree between our two root bag deployment periods. To determine if changes in root growth were correlated with changes in root zone soil thickness, we considered whether there were significant changes in each of those variables over time. We could not directly test for an effect of root growth on root zone soil thickness because individual sediment pins were not paired with individual root bags. Significance for all models was set at $p \leq 0.05$ and

p-values and test statistics were reported to three significant figures. Bennion et al. (2024) described the pre-processing required to standardize and clean the RSET data from these sites.

All analyses were performed using R in the RStudio environment (v4.1.2, R Core Team 2021). Data were cleaned and manipulated using *tidyverse* (Wickham et al. 2019), extended with *readxl* (Wickham and Bryan 2023). We used *betareg* (Cribari-Neto and Zeileis 2010) for beta regression models, and *mgcv* (Wood 2011) for generalised additive modelling with *gratia* (Simpson 2023) for model diagnostics and derivative samples. Data were plotted with *ggplot2* (Wickham 2016), extended with *ggforce* (Pedersen 2022) and *ragg* (Pedersen and Shemanarev 2023).

Our raw data and code to reproduce all results and figures are available on Zenodo: <https://doi.org/10.5281/zenodo.13920091>.

Results

Canopy cover in the gap around the large, uprooted *A. marina* was 28% immediately after the tree fall. Thirty-two months later, canopy cover was 38%, indicating some infilling of the light gap by surrounding tree canopies. There were no seedlings present in the canopy gap during the thirty-two months of observation. The stem, branches and canopy of the uprooted tree started to decompose six months after the uprooting event (Figure S1).

For mangrove RSETs closer to the fallen trees, elevation change was non-linear over time (GAM: $\text{edf}_{\text{mangrove2}} = 5.73$, $\text{edf}_{\text{mangrove3}} = 3.91$) (Fig. 2). At those nearby RSETs, surface elevation rates of change from GAMs reverted from positive to negative after the tree fall and were more variable than for RSETs further from the fallen tree (Table 1). Using the annualisation approach, surface elevation in the mangrove forest declined by $26.3 \pm 3.59\text{ mm year}^{-1}$ following the severe storms of early 2021, and by $9.11 \pm 9.46\text{ mm year}^{-1}$ following the flooding of early 2022. In contrast, surface elevation change was positive and constant in the saltmarsh and for the mangrove RSET most distant from the fallen tree (GAM: $\text{edf}_{\text{saltmarsh}} = 1.00$, $\text{edf}_{\text{mangrove1}} = 1.03$) (Fig. 2; Table 1).

Surface accretion (the accumulation of sediments on top of the marker horizon) in the mangrove forest was positive between December 2020 and December 2021 ($10.0 \pm 2.57\text{ mm}$). Unfortunately, surface accretion was not measured in April 2021.

Root growth around the fallen tree was significantly lower than at the RSETs, immediately following the tree fall (beta regression: $p = 0.017$) (Fig. 3). Root growth near the fallen tree increased significantly from twelve to eighteen months

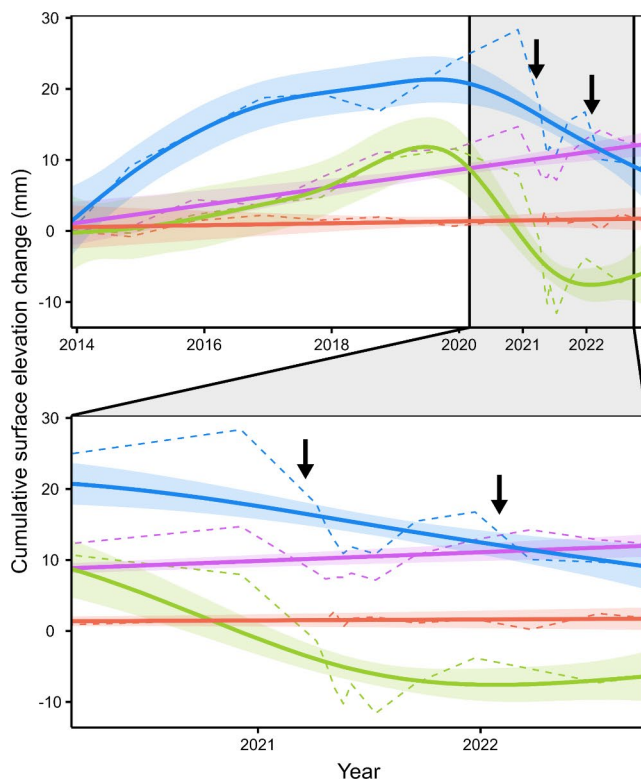


Fig. 2 Surface elevation change (cumulative mm) in mangroves (blue, green, purple lines) and saltmarsh (orange lines) at Warra Warra (Point Halloran Conservation Park), Queensland, Australia, measured using RSETs. Top left panel shows surface elevation since 2014; bottom left panel focuses on a period including two extreme weather events (black arrows; first arrow — severe storms; second arrow — severe flood-



ing). Dashed lines show empirical observations and solid lines (with shading) show GAM fits (with 95% credible intervals) for each RSET. Saltmarsh data are the mean of measurements from three RSETs. Right panel shows where a mangrove tree fell during the first extreme event (black square), impacting nearby mangrove RSETs (coloured circles which correspond with coloured lines on graph)

Table 1 Rates of change in surface elevation (mm year^{-1}) in mangrove and saltmarsh at Warra Warra (Point Halloran Conservation Park), Queensland, Australia, measured using RSETs — either across the full range of data, and before or after two large trees fell 20 m to the south of the “Mangrove 3” RSET. The ‘Mangrove 1’ RSET was furthest from the fallen tree; ‘Saltmarsh’ RSETs were not impacted by the fallen tree. Rates are the mean (standard error) of derivatives sampled from a GAM fit for each RSET ($n_{\text{full time range}} = 500$, $n_{\text{before tree fall}} = 388$, $n_{\text{after tree fall}} = 112$), after Feher et al. (2023)

RSET	Surface elevation change		
	Full time range	Before tree fall	After tree fall
Mangrove 1	1.24 (6.20×10^{-4})	1.25 (6.43×10^{-4})	1.23 (7.96×10^{-5})
Mangrove 2	-0.62 (0.22)	0.02 (0.24)	-2.81 (0.49)
Mangrove 3	0.76 (0.19)	2.37 (0.17)	-4.82 (0.03)
Saltmarsh ^a	0.14 (1.55×10^{-6})	0.14 (1.53×10^{-6})	0.14 (1.53×10^{-6})

^a mean of observations from three saltmarsh RSETs.

after the storm event (beta regression: $p = 0.027$) (Fig. 3), though fine root (< 2 mm) growth remained visually absent (Figure S2). This significant increase in root growth was correlated with a significant increase in the thickness of the root zone soil around the fallen tree, measured with sediment pins (t-test: $p = 1.09 \times 10^{-3}$) (Fig. 4a).

After the initial tree fall, there was a reduction in root zone soil thickness on the seaward side of the fallen tree trunk, and to the landward, there were variable changes in soil thickness in the root zone, resulting in significantly increased root zone soil thickness in landward compared to

seaward positions (GAM: $p = 1.67 \times 10^{-4}$) (Fig. 4a). During the further flooding in early 2022, there was erosion on both sides of the fallen trunk, followed by recovery of sediments in November 2022 (Fig. 4a). At the mangrove RSET sites, variable root zone soil thickness trajectories were observed over the same period (Figure S3).

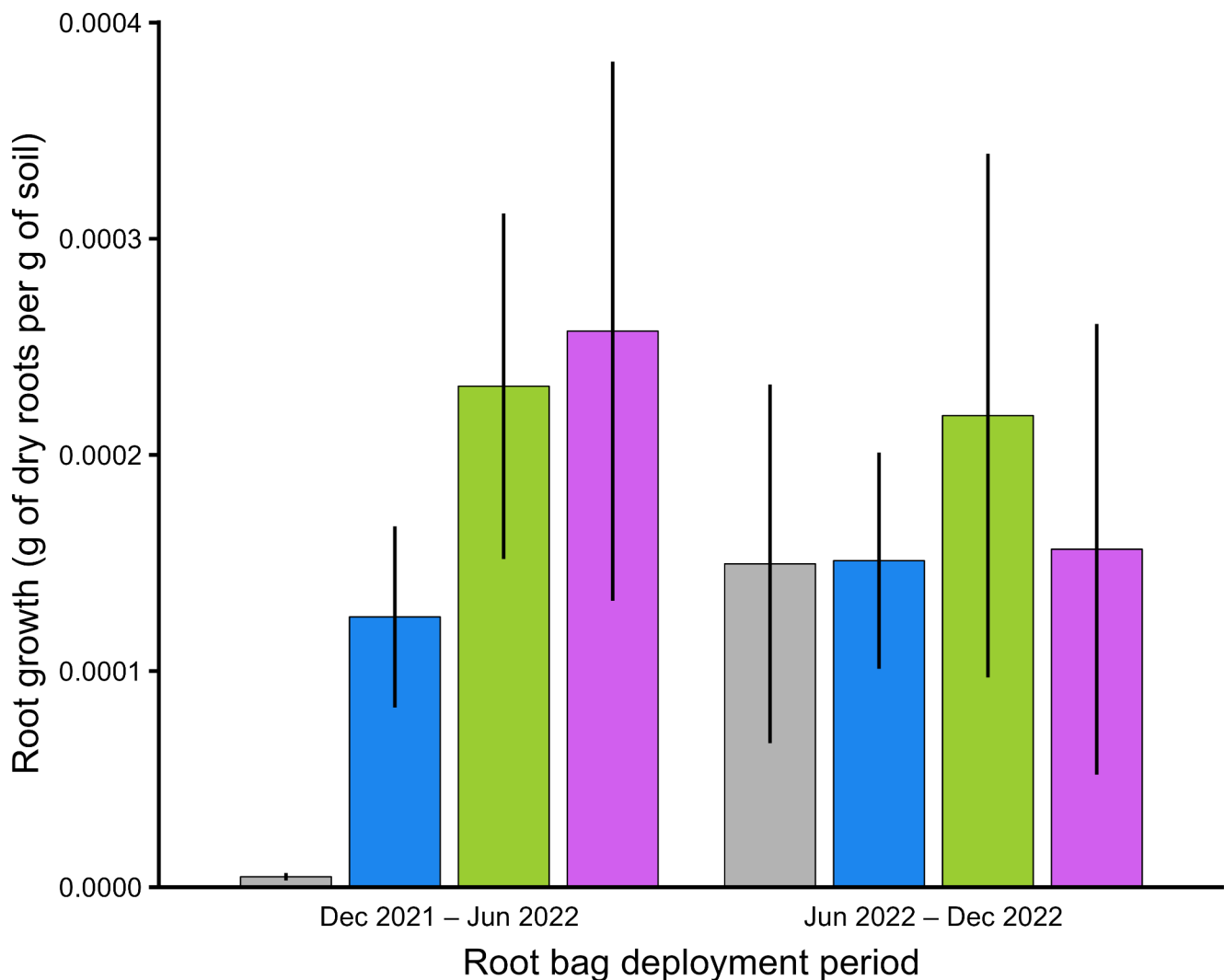


Fig. 3 Root growth (mean dry root mass (g) per gram of soil) in mangroves at Warra Warra (Point Halloran Conservation Park), Queensland, Australia, for two six-month periods between December 2021 and December 2022, measured using root in-growth bags. Bags were inserted near an *Avicennia marina* that fell in March 2021 (grey

bars), and at nearby RSETs (coloured bars; ordered by distance of each RSET from the fallen tree — blue bar was closest to the fallen tree and purple bar was furthest; colours correspond with Fig. 1). Error bars show standard error

Discussion

We found that individual tree mortality in fringing, low elevation mangroves after extreme weather events caused loss of surface elevation in the surrounding forest. Additionally, surface elevation recovery over time was complex and varied with distance from the fallen tree and exposure to open water of the creek. Possible recovery of elevation was delayed by the slow regrowth of belowground roots. Our results contribute to understanding the effects of small-scale tree mortality in mangrove forests, which is important for understanding broader mangrove forest dynamics under a future of increasingly intense coastal weather and sea-level rise.

Our observed rates of mangrove surface elevation loss after tree death were similar to previous reports (Table 2). Using our annualised rates of loss (to allow for direct comparisons with the literature), our estimate was very similar to the few other published reports of surface elevation loss around individual dead trees, and about one to two times larger than reports of surface elevation loss from sites which experienced tree death on a broader scale (Table 2). Using GAM-derived loss rates, our estimates were of a similar magnitude as most published reports (Table 2). GAMs have only recently been applied to coastal wetland surface elevation data (Feher et al. 2023), but the structure of the models and their potential to include non-linear trends in surface elevation over time is more appropriate for modelling

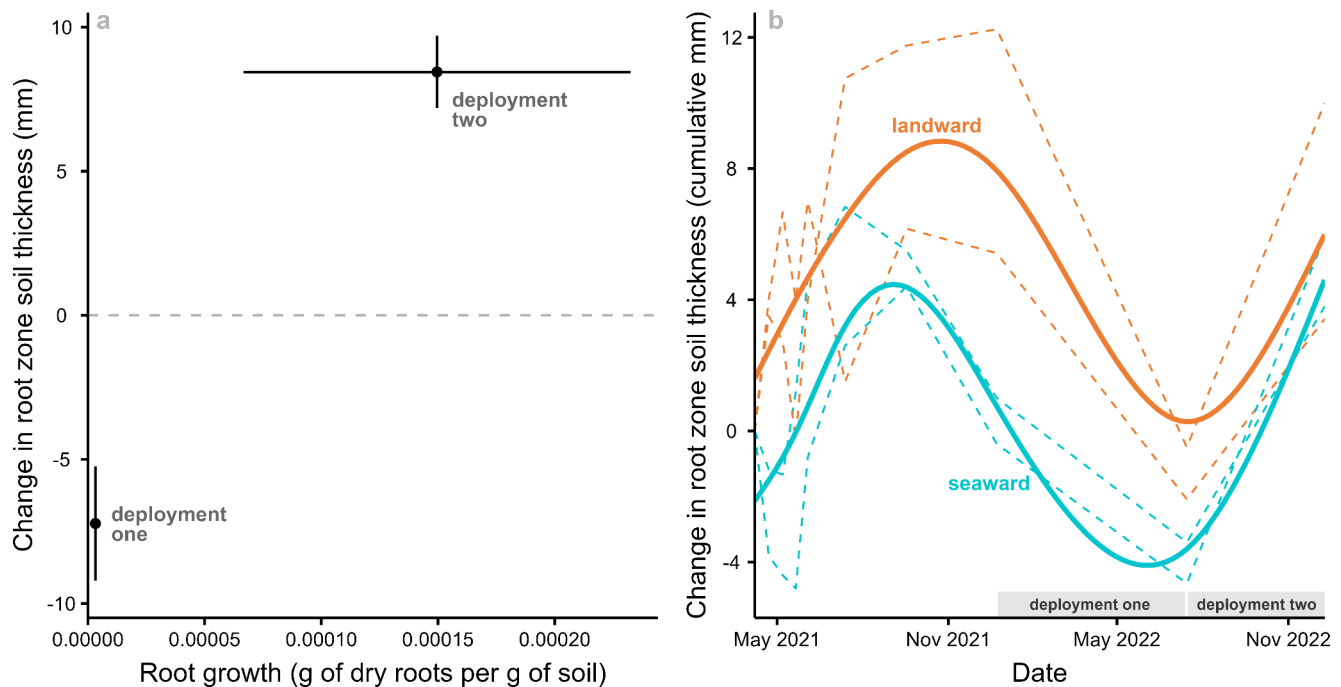


Fig. 4 Panel (a) shows that increased root growth (g of dry roots per g of soil in root in-growth bags, over two deployment periods) was correlated with root zone soil thickness changes (mm, measured using sediment pins) around a fallen tree in mangroves at Warra Warra (Point Halloran Conservation Park), Queensland, Australia. Grey dashed line shows zero change in root zone soil thickness, and error bars show standard error for both variables. Panel (b) shows empirical cumulative soil thickness changes in the root zone (dashed lines) from 8 shal-

low pins (approx. 25 cm depth) and 12 deeper pins (approx. 50 cm depth) that had statistically similar changes in root zone soil thickness, on either the landward (orange) or seaward (light blue) side of the fallen trunk. Solid lines are GAM fits for each setting. Grey bands on the x-axis show the deployment periods of the root bags; these same periods were used to calculate the changes in root zone soil thickness for panel (a)

Table 2 Rates of change in surface elevation (mm year^{-1} , mean and standard error) in mangroves after tree death events of varying causes and scales

Location	Surface elevation change	Tree death process	Scale of disturbance	Interpretation of surface elevation change	Reference
Australia, Qld	-4.82 (0.03) from GAMs	Uprooting	Individual tree <i>A. marina</i>	Root death (compaction of root tissue and soil macropores, and some decomposition)	This study
Australia, Qld	-26.3 (3.59) from annualization	Uprooting	Individual tree <i>A. marina</i>	Root death (compaction of root tissue and soil macropores, and some decomposition)	This study
Honduras	-11	Hurricane Mitch	hectares	Peat collapse	(Cahoon et al. 2003)
USA, Florida	-3.1	Lightning gaps	m^2	Subsidence below root zone	(Whelan 2005)
New Zealand (Aotearoa)	-9 to -38	Cleared	hectares	Sediment erosion	(Stokes et al. 2009)
USA, Florida	-9.7 (2.1)	Hurricane Wilma	hectares	Sediment scouring	(Feher et al. 2020; Whelan et al. 2009)
Micronesia	-10.6	Individual tree harvest	m^2	Soil subsidence	(Krauss et al. 2010)
Australia, NSW	30.3	Storms (<i>El Niño</i>)	m^2	Sediment deposition during storm	(Rogers et al. 2013)
Kenya	-32.1 (8.4)	Experimental girdling	Individual trees (<i>R. muncronata</i>)	Decomposition of soil organic matter	(Lang'At et al. 2014)
USA, Louisiana	4.4 (0.7)	Freezing	m^2	Sediment deposition	(McKee and Vervaeke 2018)
USA, Louisiana	32	Hurricane Gustav	hectares	Sediment deposition during storm	(McKee et al. 2020)
USA, Florida	-8.8	Hurricane Wilma	hectares	Peat collapse	(Osland et al. 2020)
USA, Florida	-0.6	Unknown	hectares	Peat collapse	(Cormier et al. 2021)
Kenya	-10.9 (2.4)	Hydrological disruption	hectares	Reduction in minerogenic sediments and enhanced decomposition	(Kimeli et al. 2022)

responses to disturbance events than other types of analysis (Swales et al. 2019).

Surface accretion (sediments accumulating over marker horizons) at the mangrove RSET sites throughout the year (10.0 ± 2.57 mm) indicated that there was no erosion of surface sediments in the mangroves between December 2020 and December 2021. Thus, it is most likely that subsidence was caused by root death, and possibly by the early phases of compression and decomposition of highly aerenchymatous mangrove roots (Middleton and McKee 2001). Unfortunately, surface accretion was not measured in April 2021 following the first weather event, but this information could have improved our understanding of the soil surface dynamics.

Mangrove tree mortality has been observed to cause subsidence (termed ‘sinking centers’ by some authors; Duke et al. 2010), with root mortality, decreased soil shear strength and soil compaction leading to soil subsidence (McKee and McGinnis 2002) and pooling of water occurring around areas of tree death (Radabaugh et al. 2021). Regeneration in these areas occurred by colonization, sometimes from different species (Sherman et al. 2000; Smith et al. 1994), and surface elevation was recovered through a range of processes including root growth and accretion on the soil surface. With small scale individual tree death, neighboring trees may contribute to the recovery of surface elevation and root regrowth, for example, by an exchange of resources through grafted root networks as has been reported for *Avicennia* (Vovides et al. 2021).

Root growth is a critical process for recovery of soil surface elevation. Delayed return of root growth around our downed trees, reaching baseline levels only about 20 months after the tree death events (Fig. 3), may be related to slow surface elevation recovery at our sites. Root growth was linked to 42% of surface elevation gains in a Caribbean mangrove forest (McKee et al. 2007). Fine root production is also important for surface elevation gain and was responsible for 62% of root zone expansion in Chinese mangroves (Lin et al. 2023; Liu et al. 2017; Xiong et al. 2017). In keeping with our quantitative results for root biomass, qualitative fine root growth was not detected in root bags near our uprooted trees for up to 14 months following the extreme events (Figure S2) but was present 20 months afterwards. We propose that this increase in root growth is from (belowground) gap infilling from neighboring trees. To increase our understanding of the relevance of fine roots for surface elevation change, future studies that use root in-growth bags and root cores from mangrove forests could process samples to capture fine (<2 mm) roots to estimate their potential contribution to elevation gain, or use minirhizotrons (Arnaud et al. 2021).

Surface elevation loss was largest at the RSET nearest to the fallen trees, however the impact of the storm on surface elevation was evident as an initial loss of elevation over all three RSET sites. The small loss in elevation across all three sites may indicate that the storm surge compressed the soil across the entire mangrove site, or that prolonged flooding led to widespread root mortality and compression of soil macropores (Xiong et al. 2019). Future research is needed to further understand the mechanisms that result in loss of elevation during extreme weather events, and recovery in the weeks and months following.

Different causes of tree mortality result in different patterns of root death and recovery. For example, lightning strikes resulted in standing dead trees (disturbing an average 30 m diameter area), giving rise to slow decomposition of roots and localised soil subsidence (Sherman et al. 2000). Harvested mangroves induced localised soil surface elevation collapse due to root mortality and decomposition along with sediment compaction (Lang’At et al. 2014). We suggest that mortality due to uprooting would also cause extensive root death and compression of aerenchymatous roots and root channels (macropores). Additionally, the physical rending of roots from aboveground tissues (Fig. 1a, Figure S1) would reduce the possibility of root survival, increasing our confidence in attributing much of the surface elevation losses in our data to early phases of compression and decomposition of aerenchyma-filled mangrove roots (Middleton and McKee 2001). We conclude that rapidly decomposing fine roots (Hayes et al. 2017; Morris et al. 2023) and compaction of the soil (Xiong et al. 2019) as processes influencing surface elevation loss in our study.

Partial canopy closure in the 32 months following tree death occurred by sprouting foliage of the neighbouring mangrove canopies. Complete canopy closure takes decades and is important for mangrove forest maintenance (Duke 2001). Aerial images of older gaps in our study forest indicated that canopy closure visible from above can take up to a decade to occur, highlighting the importance of long-term monitoring for understanding canopy infilling responses. These canopy infilling rates are similar to those of lightning strike gaps, with reported 8–16-year closure times in the Dominican Republic (Sherman et al. 2000) and at most 15 years in Papua New Guinea (Paijmans and Rollet 1976). Additionally, in contrast to reports of mangrove forest recovery following large scale disturbance being mostly due to propagule establishment and development (Smith et al. 1994), we found that canopy infilling was entirely through in-growth of neighbouring canopy branches, as there was no juvenile growth around the fallen trees during our 2.5 years of observation. Our fallen *A. marina* had some new growth from epicormic budding, three years post-uprooting, a trait of only some mangrove genera (Baldwin et al. 1995; Carlos

et al. 2015; Clarke and Allaway 1993; Krauss and Osland 2019). There are very limited data on recovery from epicormic shoots off the boles or fallen stems of damaged trees due to limited long-term assessments (Krauss and Osland 2019). Future monitoring of the site in the present study could add data about this phenomenon.

Under global climate change, an increase in severe storms is expected for south-east Queensland (Herold et al. 2021), which could have devastating effects on fringing mangroves, given their slow pace of recovery following disturbance. Repeated losses in relative soil surface elevation could result in mangrove fringes that are unable to maintain their elevation relative to sea-level rise, leading to an eventual transitioning to open water and retreat of the mangrove. However, another possible mechanism for recovery may be through recruitment and growth of *Rhizophora stylosa*, a species that is beginning to dominate some communities in Quandamooka, has faster growth rates than *A. marina*, and may be more resilient to storms due to its prop root structures (Bennion et al. 2024; Hill et al. 2021, 2024). An increased abundance of *R. mangle* in canopy gaps was observed in the Dominican Republic, and increased occurrence of gaps within the forest over time resulted in a *Rhizophora*-dominated mangrove forest (Sherman et al. 2000; Whelan 2005).

Management could focus on monitoring mangrove condition on shorelines in regions with increasing frequency and intensity of extreme events, investigating strategies for recovering fringing mangroves (e.g. through enrichment planting, Primavera et al. 2016), understanding implications of reduced coastal protection, and evaluating the potential for landward retreat if this is likely. This research could be further improved with measurements of groundwater levels, which can influence surface elevation (Rogers and Saintilan 2008). Future studies could also sample root cores along transects moving away from fallen trees, to confirm that dead roots are more abundant near fallen trees, and to determine at what distance from fallen trees can increased root death be detected.

Our results show that mangrove soil surface elevation is affected at small scales by extreme weather events that cause tree death and localised subsidence. Such loss of surface elevation may meaningfully influence mangrove resilience to future sea-level rise, including pushing vulnerable sites towards sea level tipping points. Saintilan et al. (2020) suggested that sea-level rise of 6.1 mm year^{-1} may be a tipping point for mangrove survival, highlighting the importance of understanding mangrove resilience to surface elevation perturbations and recovery over time. Recovery from extreme events occurred by canopy infilling from neighbouring trees and neighbouring root production that enhanced surface elevation and began recovery to a pre-mortality state.

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Declarations

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

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