



Tidal Impoundment and Mangrove Dieback at Cabbage Tree Basin, NSW: Drivers of Change and Tailored Management for the Future

E. Asbridge¹ · R. Clark¹ · P. Denham¹ · M. G. Hughes^{1,2} · M. James¹ · D. McLaughlin¹ · C. Turner¹ · T. Whitton¹ · T. Wilde¹ · K. Rogers¹

Received: 6 October 2023 / Revised: 24 July 2024 / Accepted: 20 August 2024 / Published online: 7 September 2024
© The Author(s) 2024

Abstract

Major storms can cause significant changes to coastal and wetland environments. A series of storm events in 2020 resulted in closure of the historically open estuary at Cabbage Tree Basin, Port Hacking, New South Wales (NSW), Australia. Prolonged tidal impoundment (3 months) led to substantial changes in hydrological and sedimentological processes, resulting in widespread mangrove dieback. This study aimed to quantify the degree of impact and recovery for mangroves, identify factors contributing to dieback, and consider the implications for carbon sequestration. This was achieved using remotely piloted aircraft structure-from-motion approaches, aerial photography, and field-based assessments of vegetation health and above-ground biomass (AGB). Mangroves were classified as ‘dead’, ‘partially dead’, and ‘live’. In October 2019, there was 10.8 ha of live mangroves, with this reduced to 6.6 ha by August 2022. Digital surface models (DSMs) were intersected with classified mangroves to assess the vertical distribution of each zone. All mangroves classified as ‘dead’ were distributed at elevations < 0.4 m with respect to the Australian Height Datum (AHD), suggesting these regions were persistently inundated, which was confirmed by water level loggers (inundated during logger deployment). Field data confirmed substrate elevation related to dieback with the proportion of ‘live’ mangroves greatest at elevations > 0.6 m AHD. Substrate elevation and distance to the estuary mouth were significantly correlated with species, with *Avicennia marina* located at lower tidal positions and closer to the entrance compared to *Aegiceras corniculatum*. The dieback event equated to a loss of 81.5 ± 48 Mg of above-ground biomass, 38.1 ± 22.5 Mg C, or 140 ± 82 Mg CO₂ equivalence (CO₂e). This study provides an important baseline for monitoring dieback events. Continued monitoring is crucial to assess recovery and to tailor management strategies.

Keywords Mangroves · Coastal wetland · Tidal impoundment · Storm events · Elevation

Introduction

Globally mangrove forests are threatened by many types of disturbances, including environmental stress as a result of anthropogenic and natural drivers. Despite the historic loss of mangroves, these ecosystems have been described as disturbance adapted communities (Lugo et al. 1981) as they are highly dynamic and resilient, responding rapidly

to extreme infrequent environmental conditions, such as cyclones, flooding, and storm surges (Asbridge et al. 2018; Krauss and Osland 2020). Sometimes, the system may not return to pre-disturbance conditions (i.e. structure – tree height and species); however, resilient disturbance adapted forests often have the capacity to recover to a new stable state. Historically, mangroves have demonstrated adaptive success enduring significant changes in sea level during the Quaternary period, reflecting their capacity for resilience and ecological stability (Woodroffe 1992). However, when environmental change is extreme, rapid, widespread, or persistent, mangroves may undergo structural and functional change, potentially leading to dieback (Blasco et al. 1996; Duke et al. 2010; Sakho et al. 2011). In this scenario, the resilience of mangroves is threatened as adaptive change to disturbances may not be possible before the disturbance occurs (Koh et al. 2018). Since 2001, extreme weather

Communicated by R. Scott Warren

✉ E. Asbridge
emmaa@uow.edu.au

¹ School of Earth, Atmospheric and Life Sciences, University of Wollongong, Wollongong, NSW 2522, Australia

² Science Economics and Insights Division, NSW Department of Planning and Environment, Parramatta, NSW, Australia

events, such as tropical cyclones, droughts, and heat waves, have accounted for 11% of global mangrove loss (Goldberg et al. 2020). Projected accelerating sea-level rise combined with increased frequency and intensity of extreme events provides strong motivation to improve understanding of the spatial patterns of dieback and recovery and provide tailored management strategies.

Mangrove dieback has received recent attention due to the large-scale dieback event in late 2015/early 2016 across northern Australia (Duke et al. 2017; Asbridge et al. 2019; Sippo et al. 2020a) and Western Australia (Lovelock et al. 2017). This event was linked to climatic variability arising from the El Niño Southern Oscillation and Indian Ocean Dipole, coincident with conditions of lower-than-average rainfall and mean sea level. Elsewhere, dieback has been associated with extreme weather conditions, such as hurricanes and cyclones (Ellison 1999; Lagomasino et al. 2020, 2021; Osland et al. 2020).

Persistent flooding of wetlands can modify substrate salinities and concentrations of porewater sulfides and phytotoxins (Lagomasino et al. 2021), and/or lead to extreme sediment deposition that smothers pneumatophores and aerial roots, limiting soil aeration and oxygen availability (Ellison 1999). Estuary entrance closure due to altered sediment dynamics has been reported to lead to significant changes in hydrological connectivity and hydroperiod, leading to major disturbances for mangrove forests (Breen and Hill 1969; Choy and Booth 1994; Mbense et al. 2016). The changes to natural water flow and inundation have the capacity to limit seedling establishment and decrease productivity for many years following the initial disturbance (Milbrandt et al. 2006; Harris et al. 2010). If the condition (health) of a mangrove forest is reduced prior to an extreme event due to changing environmental conditions, particularly those altering hydrological and sediment dynamics, resistance to extreme weather may be compromised. This may lead to more a severe and widespread impact, placing the system vulnerable to collapse (Vogt et al. 2012).

Differentiating the cause of mangrove dieback following an extreme event can be difficult, particularly in the absence of in situ measurements of physiochemical conditions and pre- and post-measurements of biophysical conditions. Under these circumstances, spatial patterns in mangrove dieback and coastal geomorphological change are often used to infer mechanisms. Post-event analyses commonly involve in situ field vegetation surveys to investigate structure and condition, typically measuring tree height, density, and diameter at breast height (DBH) and recording the number of species affected, impact status, canopy cover, and impact to surrounding vegetation communities (Rogers et al. 2005, 2022; Fatoyinbo et al. 2008; Duke et al. 2017; Owers et al. 2018a; Sippo et al. 2020b; Rogers et al. 2022). Remotely piloted aircraft (RPA) surveys have also been used to collect high-resolution aerial

images over a large area, to quantify forest extent, species, zones, and structural features (Lucas et al. 2017; Bartholomeus et al. 2022). This is particularly useful for remote and inaccessible areas. Remote sensing, both airborne and space-borne, provides valuable opportunities to map pre-event conditions and extent, along with time-series mapping to indicate immediate impact and the ongoing degree of recovery (Duke et al. 2017; Asbridge et al. 2019; Abhik et al. 2021; Lagomasino et al. 2021). Light detection and ranging (LiDAR) technologies allow highly accurate three-dimensional models to be created using point cloud data to derive structural canopy metrics (i.e. canopy height, density, and biomass) and to estimate changes in vegetation structure (Lagomasino et al. 2021). Moreover, LiDAR can be used to create digital elevation models (DEMs) and digital surface models (DSMs) to provide insights into potential causes of dieback, recovery, and structural changes (Wannasiri et al. 2013; Asbridge et al. 2016; Feliciano et al. 2017).

Mangroves in populated areas of temperate Australia are highly valued because of the important ecosystem services they provide, and they are regarded to be a natural asset of high value (Rogers et al. 2016). Attention is often placed on tropical mangrove forests in Australia, as these systems regularly experience disturbances from natural hazards such as cyclones (Woodroffe and Grime 1999; Asbridge et al. 2018; Duke et al. 2024). However, temperate mangrove forests in NSW can still be exposed to extreme storm events that cause dieback. Cabbage Tree Basin, located in Port Hacking, NSW, is within the Royal National Park and is positioned on the margins of Australia's most populated city, Sydney. This site supports one of the largest mangrove forests (10.8 ha) in the Sydney region. In early 2020, multiple major storm events along the southeast Australian coast, associated with an intense La Niña phase of the ENSO, resulted in substantial changes to beach and barrier morphology. The historically open estuary partially closed in February 2020 (closed at mid tide and below) and fully closed in May 2020. Prior to this event, there was no evidence of estuary entrance closure, with mangroves historically thriving in the area with frequent tidal flushing (West and West 2000). Following February 2020, mangrove condition diminished (i.e. substantial dieback) due to tidal impoundment and elevated water levels. A new entrance was excavated in the barrier in June 2020, connecting Cabbage Tree Basin to Simpsons Bay; this action is likely to have released the impounded tidal waters. However, the mangrove dieback continued for several months, likely due to the significant physiological stress already experienced.

Given the considerable ecosystem services and values offered by mangroves at this site (site of cultural significance for indigenous peoples, fishing, recreation, and habitat for birds, bats, crustaceans, and fish), this study aimed to characterise the factors contributing to mangrove dieback and

quantify the degree of impact and post-impact recovery so as to provide tailored advice and guidance for natural resource managers. This was achieved by (i) quantifying the extent of mangrove dieback using Earth observation data, RPA, and field surveys, (ii) assessing vegetation health and characterising the varying degree of impact from field surveys, (iii) applying allometric equations to mangroves measured in plots to determine loss of above-ground biomass (AGB) and indicate implications for blue carbon storage services, and (iv) describing the spatial patterns in dieback to ascertain the likely drivers and contributing factors. We hypothesised that mangrove dieback was associated with the storm event and tidal impoundment or root burial following the deposition of sediment at the entrance, and these possible mechanisms formed the basis for the investigation. Moreover, it was theorised that tidal impoundment would ‘drown’ mangroves positioned at lower elevations and this would be evident in their health and AGB, whilst root burial would impact mangroves located nearer to the entrance only where deposition of sediment associated with the storm was greatest. Consideration was also given to the modulating effect of different species distributions on the degree of dieback and loss of AGB.

Methods

Study Site

Port Hacking, NSW, is a tide-dominated drowned river valley estuary (Roy et al. 2001) on the southern margins of Sydney and drains a relatively small catchment of ~ 225 km² (Williams and Meehan 2004). Cabbage Tree Basin (34.085°S, 151. 151°E) drains into Port Hacking and is situated between Bundeena and Maianbar on the southern side of Port Hacking and is within the Royal National Park (Fig. 1a). Tidal exchange between Cabbage Tree Basin and Port Hacking is constricted by a sand barrier at Simpsons Bay, and prior to the 2020 storm events, the basin was connected to Port Hacking via a passage that ran between Deeban Spit and east Maianbar. The basin is filled predominantly with marine sand. Port Hacking has a maximum tidal range of approximately 2 m whereas the maximum tide range inside the basin was attenuated to about 0.5 m by its constricted entrance.

The presence of mangroves in Cabbage Tree Basin suggests the channel entrance has historically remained open allowing for tidal exchange, as mangroves are usually absent from NSW intermittently closed and open lakes and lagoons (Hughes et al. 2019). Two species of mangrove are present, *Avicennia marina* and *Aegiceras corniculatum*. *Avicennia marina* is one of the most common and widespread species in NSW able to withstand a wide range of temperatures

and salinities (Duke 1991, 2006). *Avicennia marina* exhibits heights of 2–12 m in the Port Hacking region and is generally distributed lower in the tidal frame; *Aegiceras corniculatum* is shorter in stature at 1–3 m and distributed at slightly higher elevations, often where substrates are less saline. The upper intertidal saltmarsh is dominated by *Sarcocornia quinqueflora*, *Sporobolus virginicus*, and *Juncus kraussii*, whilst *Casuarina glauca* occupies supratidal flats.

Deeban (Port Hacking) is an important site for Aboriginals, with shell middens, engravings, and charcoal paintings in rock shelters providing an important insight into the lives of the Gweagal people (Bursill 2018). This is also one of only a few sites in the Sydney region with relatively extensive coastal wetlands, providing important educational and recreational services. NSW National Parks and Wildlife Service protect this area for these important cultural, environmental, economic, and social values and as such sampling was restricted to non-destructive approaches.

RPA Data Acquisition and Processing

RPA data were acquired on 23 August 2022 and 29 August 2022 using a Phantom 4 RPA real-time kinematic global positioning system (RTK-GPS) to collect RGB aerial imagery. Flights were conducted at 60 m altitude in relatively clear weather conditions (no rain and few clouds) with low wind speed (< 10 km/h). The RPA was flown using the specifications in Supplementary Table 1. The RTK-GPS capabilities of the RPA allowed image capture to be directly linked to GPS coordinates at the time of data acquisition, which reduced the number of ground control points required. For this study, 30 × 30 cm black and white tiles were used as ground control points and placed at various elevations and aspects, each tile had an RTK-GPS point taken from the centre. The ground control points were required to cross-reference the orientation and position information with the data from the Phantom 4 RTK.

RPA images were processed in Agisoft Photoscan Professional (Metashape 1.2) using structure from motion techniques to generate an orthomosaic (processing parameters in Supplementary Table 1). The orthomosaic was used to delineate mangroves that were dead, live, and partially live. Structure from motion techniques were used to process the RPA images and produce an elevation model. Although Agisoft refers to this model as a DEM, it is more representative of a DSM, as the model includes tree canopies and objects that do not represent the substrate. Henceforth, the RPA-derived elevation model is referred to as a DSM. Confidence in the DSM as an indicator of substrate elevations was assessed by comparison with field-based RTK-GPS measurements, using a matched pairs *t*-test. The analysis confirmed a tendency for the RPA-derived DSM to indicate higher elevations than the RTK-GPS measurements

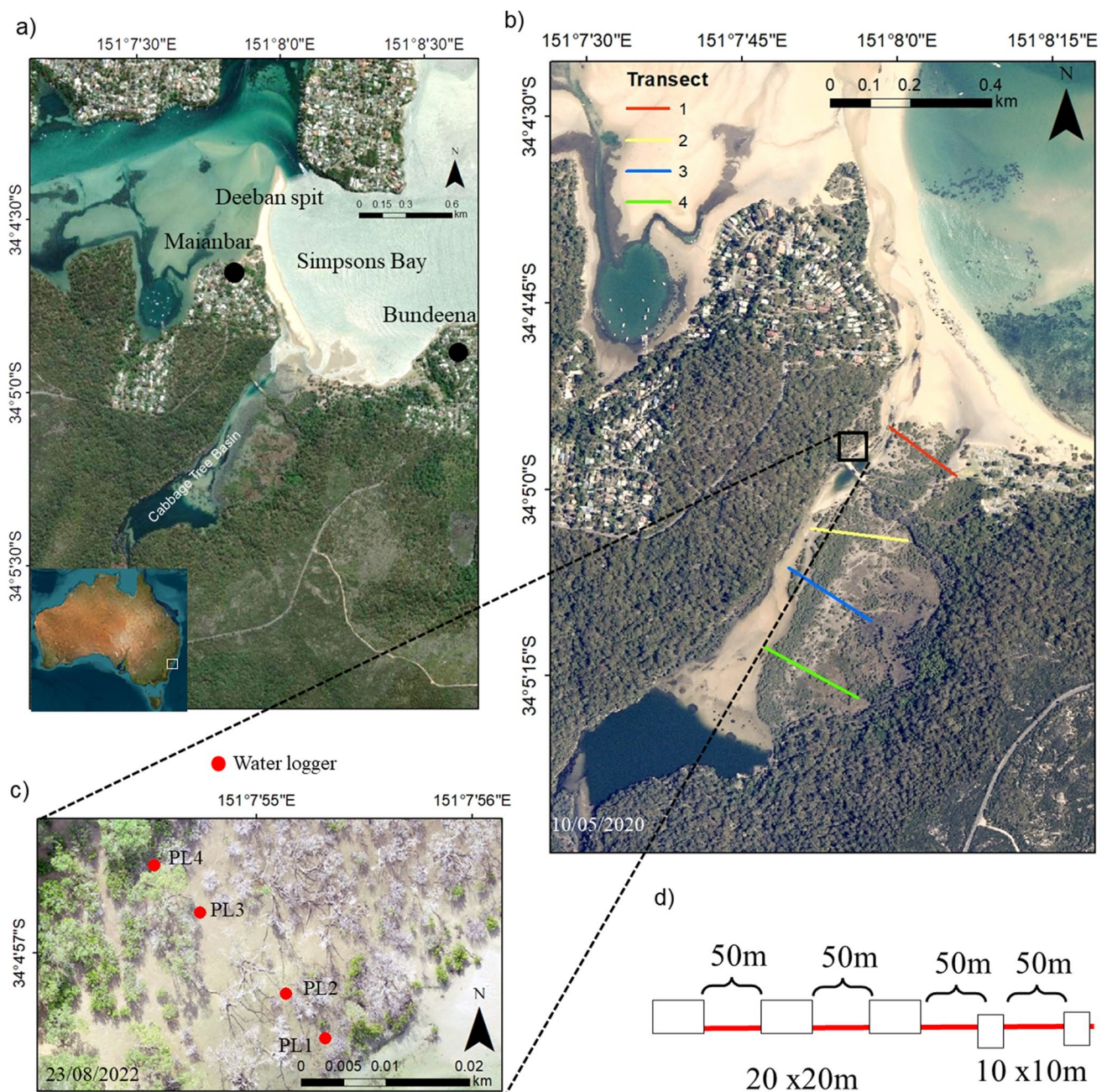


Fig. 1 **a** Cabbage Tree Basin, Port Hacking, NSW, Australia. **b** Numbered transects indicate field surveys, locations derived from a real-time kinematic global positioning system (RTK-GPS); **c** water logger locations derived from RTK-GPS, and **d** the design of field survey

using quadrat/plot transect method to measure observed variables in plots at 50 m intervals. Note the example showing different size plots: *A. marina* dominated 20×20 m and *A. corniculatum* dominated 10×10 m

($P > |t| = < 0.0001$; $P > t = < 0.0001$; $P < t = 1.0000$); however, the correlation between datasets was high ($R^2 = 0.9618$) and the mean difference was low at (0.04 ± 0.006 m) and within the anticipated error range.

LiDAR-derived point clouds are known to provide reliable DEMs indicating substrate elevations. Two LiDAR-derived DEMs were assessed (2013 and 2020) from Geoscience Australia's Elevation Information System (ELVIS)

data repository to assess their suitability for indicating substrate elevations at Cabbage Tree Basin. The 2020 DEM was assessed for accuracy by comparison against 94 RTK-GPS measurements (reported vertical and horizontal accuracy of 15 mm and 8 mm, respectively) acquired during vegetation health field surveys. Matched pairs *t*-tests were undertaken to determine the accuracy, and this preliminary assessment indicated that the correlation between the datasets

was low ($R^2=0.5980$), the difference was highly variable (0.0529 ± 0.0187), and there was a tendency for the LiDAR DEM to indicate higher elevations than the RTK-GPS measurements ($P > |t| = 0.0056$; $P > t = 0.0028$; $P < t = 0.9972$). Later inspection of the LiDAR survey data indicated that the survey had been undertaken at the time of tidal impoundment and did not provide a reliable indication of substrate elevations.

Aerial Photography Data Acquisition and Processing

Historical aerial photography was used to understand previous mangrove extent and condition at Cabbage Tree Basin. The imagery was accessed from the NSW Government Spatial Services portal: Historical Imagery Viewer. Images were chosen based on their clarity and resolution; those considered most useful, 1956 and 1994, were georectified using high-resolution Nearmap imagery as spatial controls. To track recent changes in barrier morphology, dieback, and signs of recovery, Nearmap imagery was accessed for 22/20/2019, 10/05/2020, 01/06/2020, and 02/08/2022.

Water Levels and Depths

To assess the tidal inundation of mangroves within the basin, a transect of four HOBO U20L-04 water pressure loggers was deployed (Fig. 1) from 06/04/2020 through to 07/05/2020. The variation in water depth over the sediment surface and variation in water level relative to the Australian Height Datum (AHD) were determined. A fifth pressure logger was deployed in the air to enable corrections in logger output for variations in atmospheric pressure over the deployment period. The instrument transects extended from the most seaward mangrove (*A. marina*) to the most landward mangrove (*A. corniculatum*) (Fig. 1), covering an elevation range of 0.318 to 0.427 m relative to the Australian Height Datum. Water levels may have a maximum error of ± 28 mm due to logger errors of ± 4 –8 mm and RTK error of ± 20 mm (LEICA RX1200 RTK Survey System).

Time-Series Vegetation Classification

To classify vegetation type and land cover, object-based image analysis (OBIA) approaches were applied. Aerial imagery (historical and Nearmap) and the RPA orthomosaic were used in Trimble eCognition Developer 9 image analysis software. A multi-resolution segmentation approach was used, with processing parameters outlined in Table 1. The nearest neighbour classification was undertaken to delineate the distribution of classes. Classifications were exported as polygon shapefiles and manually cleaned in ESRI ArcMap to correct misclassifications. This was achieved using high-resolution imagery (Google Earth and Nearmap) as a reference.

Spatial Patterns of Mangrove Dieback

Following image classification, the spatial distribution of dieback was assessed to assist with identifying likely drivers and contributing factors to the dieback. We specifically tested whether the spatial distribution of mangrove dieback was (i) delimited to a threshold elevation representing the water levels associated with tidal impoundment, or (ii) delimited to a distance from the entrance, representing regions where sand deposition smothered pneumatophores.

The vertical distribution of living and dead mangroves was discerned by intersecting the RPA-derived image classification with the DSM using zonal statistics in ESRI ArcMAP 10.8.1 to establish the mean elevation and standard deviation for living, partially live and dead mangroves. Following this, the shapefiles were converted to raster and the elevation of dead, partially live, and live mangrove pixels was extracted from the DSM for further statistical analyses. Statistical differences in the elevation of living and dead mangroves were determined using ANOVA.

The distance from mangrove dieback to the estuary entrance was calculated for the study area to identify whether sediment smothering (i.e. large amount of sediment accumulation during the storm) of pneumatophores contributed to dieback. This was generated by digitising a line indicating the entrance, and then generating a distance layer using the Euclidean Distance tool in ArcMap. The mean distance and standard deviation of the dead and live mangroves were determined using zonal statistics applied to the RPA image

Table 1 eCognition segmentation parameters and classes for Cabbage Tree Basin analysis

Image	Scale	Shape	Compactness	Classes
Historical imagery	25	0.05	0.3	Mangrove, sand, water, and mudflats
Nearmap imagery	25	0.05	0.3	Live mangrove, dead mangrove, shadows, forest, sand (wet and dry), water, mudflats
RPA orthomosaic	65	0.05	0.3	Live mangrove, dead mangrove, shadows, forest, sand (wet and dry), water, mudflats

classification and the distance layer. Statistical differences in the distance of living and dead mangrove pixels from the estuary entrance were determined using ANOVA. The digitised boundary between dead and live mangroves was also used to determine the mean distance and standard deviation of this boundary; a small standard deviation was used to confirm whether a threshold distance could be identified.

Spatial Patterns in Vegetation Health

Field surveys conducted during August and September 2022 measured the impact to vegetation health/condition, distribution, and structure, to understand the changes to AGB. A quadrat/plot transect method was used. Four transects were evenly distributed across the basin in both *A. corniculatum* and *A. marina*-dominated stands, with plots at 50 m intervals (Fig. 1c). This followed procedures advocated by English et al. (1997) and applied by Clarke (1995) at nearby Jervis Bay, NSW. Plots dominated by *A. marina* were 20 × 20 m and plots dominated by *A. corniculatum* were reduced to 10 × 10 m, reflecting differences in structure (*A. marina* are taller) and density (*A. corniculatum* patches are denser). Transects and plots were geolocated using an RTK-GPS and measurements were taken to determine the mean elevation of each plot. The mean elevation of each plot was established by averaging the elevation for the vertices of each plot.

Mangrove species, height (cm), diameter at breast height (DBH; cm), and crown foliage area (m²) were recorded for each individual tree within each plot to understand how health (i.e. condition, structure, and live vs dead) varied across the site and to inform AGB assessments. Individuals shorter than 50 cm were not recorded. Tree heights were measured using a graduated staff. DBH was measured at 1.3 m and multi-stemmed individuals were measured at 1.3 m along the length of each branch as per Owers et al. (2018a). Total stem diameter was determined by the sum of the converted circumference measurements to DBH for mangroves with multiple trunks and a visible connection at the base. The crown area was determined from two perpendicular diameter measurements of the visible crown extent, regardless of leaf coverage, using tape measures; these measurements were subsequently used to estimate the area of an ellipsoid.

Leaf coverage was visually estimated as the observable coverage of a healthy crown area; this was indicated in increments of 10%. These scores were then scaled to a health rating ranging between 1 and 5 based on physical characteristics and the appearance of live biomass (Supplementary Fig. 1). The scale rating categories were '1' ≥ 90% dead; '2' ~ 75% dead; '3' ~ 50% dead; '4' ~ 75% live; and '5' ≥ 90% live.

The vertical distribution of species with respect to elevation and the lateral distribution of species from the entrance were considered to characterise whether one species may have been more impacted due to its landscape position. A logistic model was applied to determine whether there was a statistical difference in the vertical and lateral distribution of species and to assess the interaction effect. Similar analyses were undertaken to assess the relationships between the health of individual trees, elevation, and distance.

The proportion of live trees per plot was estimated as the ratio between live trees with health ratings of 2–5 and dead trees with a health rating of 1 to ascertain whether there was a threshold vertical elevation and lateral distance from the entrance at which a plot was likely to exhibit a high proportion of dieback. Regression analyses (linear and non-linear) were undertaken between the proportion of live trees and elevation, and the proportion of live trees and distance from the entrance; partitioning was undertaken to ascertain whether a threshold elevation or distance that could be detected.

Implications of Mangrove Dieback for Above-Ground Biomass

Mangrove AGB allometric equations were used to estimate AGB from field-based mangrove structural measurements; these were modified to account for the impact of the dieback event. The allometric equations developed by Owers et al. (2018b) were regarded to be the most appropriate as they were derived from destructive measurements of the same mangrove species in the same biogeographic region. These equations (Eq. 1) were modified based on the structure of individual trees (i.e. tall, shrub, dwarf) and biomass components being estimated (i.e. trunk, branch, leaves plus inflorescence, total biomass). Tree height was used to determine whether the tall, shrub, or dwarf parameters, established by Owers et al. (2018b), were applied. Parameters and the correction factor in Eq. 1 were parameterised as per the values and units provided by Owers et al. (2018b) and indicated in Supplementary Table 2.

$$AGB = (\exp(a + b) \times \ln(x_1) + c \times \ln(x_2) + d \times \ln(x_3)) \times CF \quad (1)$$

where

x_1	height (cm)
x_2	total stem diameter (cm)
x_3	crown area (m ²)
CF	correction factor

a, b, c, d parameters

The estimated AGB in the trunk, branch, and leaves plus inflorescence were used to quantify AGB for each tree prior to the dieback event. This estimate of total AGB at full health (i.e. prior to the dieback) was separated into the components that were lost following the dieback (i.e. leaves and inflorescences) and the woody components (i.e. trunk, branches) that remained following the dieback event. As trees were regarded to be live, partially live, and dead, based on leaf coverage, the leaves and inflorescence component of AGB at full health were adjusted based on the estimated percentage of remaining leaf cover. To account for the loss of woody components, the trunk and branch components of AGB at full health were adjusted based on the nominal health weighting. The estimated AGB post-dieback was determined by totalling the adjusted AGB for the trunk, branches, leaves, and inflorescences. The total AGB lost was quantified as the difference between the estimated AGB pre-dieback and the adjusted AGB post-dieback.

Estimated AGB were quantified for each individual and reported based on mean biomass per species. AGB per individual was converted to Mg per hectare by standardising the pooled AGB within plots by the plot size. The influence of elevation and distance on pre-dieback and post-dieback AGB (Mg/ha) was determined using two-way analysis of variance. Partition decision trees were used to establish threshold elevations or distances where dieback diminished.

The implications of mangrove dieback for carbon emissions were estimated based on the concentration of dead AGB below threshold elevations and the area classified as dead mangrove. Lost AGB was converted to lost carbon using a density value of $46.8 \pm 0.5\%$ C, as per Owers et al. (2018a) and represents the density of tall *A. marina*. This

value was subsequently converted to CO₂e using a factor of 3.67.

Results

Timeline of Events

Interpretation of automatically reported wave and weather records, together with several site visits, enables the establishment of the timeline shown in Table 2. The storm event began in early February 2020, with wave heights and rainfall peaking over the next 6 days leading to reported tidal impoundment on 16/02/2020. Approximately 1 month later (March 2020), *A. corniculatum* dieback was observed, followed by *A. marina* (early May 2020). A subsequent prolonged storm event towards the end of May completely closed the estuary to tidal flow before the estuary was opened again in June 2020.

Water Depths and Levels

Pressure loggers (PL) were positioned perpendicular to the wetland (i.e. PL4 (highest elevation) to PL1 (lowest elevation)) (Fig. 1). The available water level data (derived from pressure loggers) shows that the basin tide was only about 17% of the open coast tide range during the observation period, and the mean water level shows evidence of spring tidal pumping, where there is a fortnightly oscillation in the mean water level. Both observations are characteristic of heavily restricted tidal entrances (Hinwood et al. 2005; Nielsen 2009). During the first spring tides, the areas adjacent to all four loggers remained inundated (> 0 m water depth). For the first 6 days of the deployment period, the water depth remained > 0.18 m at PL1 and > 0.08 m at PL4. During the first neap tides (14–19 April), the water drained

Table 2 Timeline of events and significant observations entrance closure

Date	Event	Significance
03–04/02/2020	Storm wave conditions	Short-lived (4 h) storm event with peak Hs = 5.1 m and Ts = 10.5 s
09–10/02/2020	Storm wave conditions	Storm event (21 h) with peak Hs = 7.1 m, Ts = 10.6 s and heavy rainfall
16/02/2020	Local residents contact NPWS	Report that the estuary does not appear to be draining at low tide
25/03/2020	Local resident contacts NPWS	Report that some mangroves are dying
30/03/2020	Site visit	Inspection of mangrove dieback (only <i>A. corniculatum</i> dieback apparent)
06/04/2020	Site visit	Inspection of mangrove dieback (only <i>A. corniculatum</i> dieback apparent); installation of depth loggers
07/05/2020	Site visit	Inspection of mangrove dieback (both <i>A. corniculatum</i> and <i>A. marina</i> dieback apparent); recovery of depth loggers
22–26/05/2020	Storm wave conditions	Long-lived (91 h) storm event with peak Hs = 6.4 m and Ts = 12.7 s
24/05/2020	Site visit	Estuary observed to be closed to tidal exchange
01/06/2020	Site visit	Estuary (possibly artificially) opened

from all four pressure loggers at low tide and only returned to PL1 at high tides. During the second spring tide (22–27 April), the areas adjacent to PL1 and PL2 remained inundated whilst PL3 and PL4 drained to zero water depth daily. During the third neap tide (28 April to 2 May), the areas adjacent to PL2, PL3, and PL4 drained to zero water depth daily whilst the area adjacent to PL1 remained inundated. The pattern for the third spring tide of the deployment period was like the first (Fig. 2).

The longest period that each logger was inundated continuously during the deployment period was 18, 10, 8, and 8 days for P1, P2, P3, and P4, respectively (Fig. 2). The percentage of the deployment period that the water

depth exceeded 0.01 m is listed for each of the loggers in Table 3. The percentage of time that a range of other indicative depths are listed to facilitate comparison with the range of heights that pneumatophores and lenticels typically exist. *A. marina* pneumatophores were measured to be 0.05 to 0.15 m in height, whereas *A. corniculatum* lenticels reached < 0.05 to 0.1 m up the stems. Thus, at PL1, for example, 0.10 m high pneumatophores associated with *A. marina* would have been inundated for 75% of the deployment period. Similarly, lenticels at the base of the trunks of *A. corniculatum* would have been inundated for 89% of the time.

Fig. 2 **a** Water depth and **b** water level relative to the Australian Height Datum measured by pressure loggers (PL) 1, 2, 3 and 4. PL1 was lowermost on the transect and P4 uppermost (see Fig. 1). The water level record from the Sydney tide gauge is shown as representative of the open coast tide

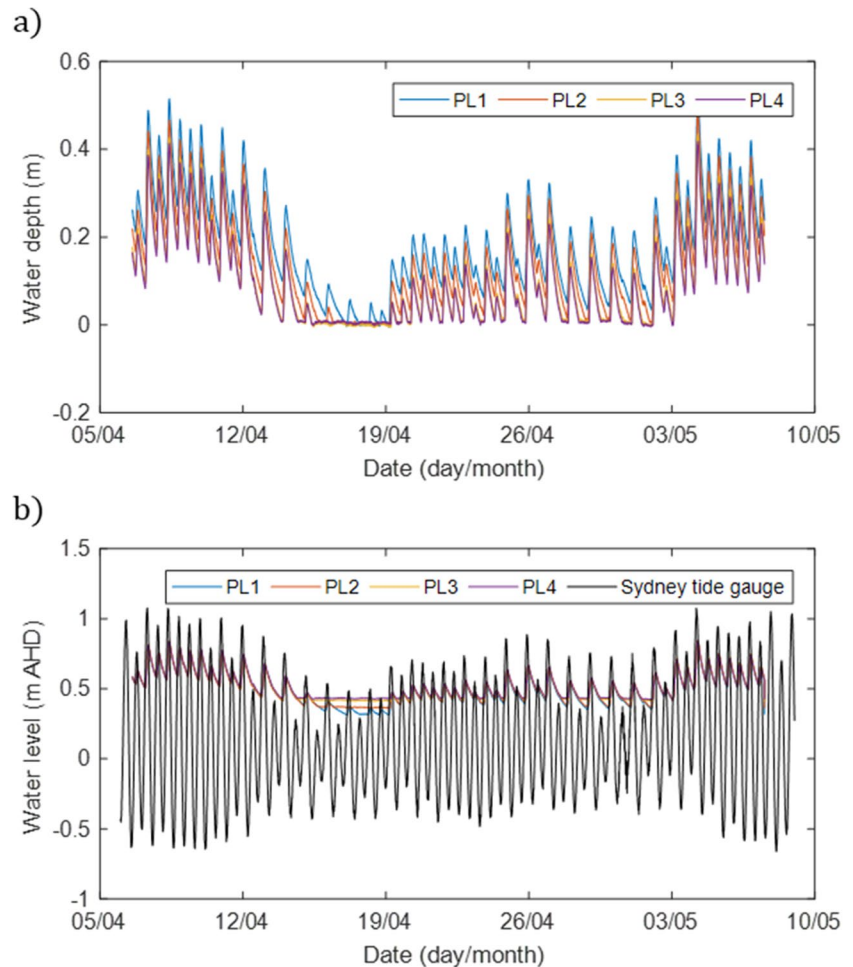


Table 3 The percentage of the deployment period (31 days) that the water depth exceeded the listed level at each of the pressure loggers. The substrate elevation at the pressure logger is listed in Table 1

		Water depth (m)				
		0.01 m	0.05 m	0.10 m	0.15 m	0.20 m
Pressure Logger (PL) Number	PL1	96	89	74	55	40
	PL2	89	78	58	43	29
	PL3	78	59	44	31	19
	PL4	73	56	41	28	17

Time-Series Vegetation Classification

Historic aerial photography indicated that mature mangroves occupied the site in 1956 (10.88 ha) and persisted until 1994 (11.50 ha). There was little evidence of previous dieback events, but the distribution of mangroves did shift with increased coverage near the entrance and thinner coverage in the western reaches (Fig. 3).

Dynamics of the barrier positioned at the entrance and mangrove dieback could be distinguished from the Nearmap imagery (Fig. 4). In 2019 (Fig. 4a), the entrance was open indicating that the large area of ‘live’ mangroves (10.8 ha, Table 4) would have experienced tidal flushing. The barrier at the entrance was partially closed following storms in February 2020 and fully closed in May 2020 (Fig. 4b). Whilst tidal impoundment and dieback were reported by local residents in early March 2020 and observed during a site visit in late March 2020 (Table 2), the dieback was not pronounced in aerial imagery, with only 0.1 ha classified as dead mangrove (Fig. 4b, Table 4). The barrier was penetrated, and a new entrance was created in June 2020, re-establishing tidal exchange (Fig. 4c). However, mangrove mortality became

increasingly evident with 0.9 ha of mangroves identified as dead (Fig. 4c, Table 4). It is likely that the mangroves succumbed to extended periods of inundation in the previous months when the entrance was partially closed (Fig. 2). This pattern of increasing mortality continued until August 2022, with 4.9 ha of dead mangrove (Fig. 4d, Table 4).

The resolution of the 2022 RPA orthomosaic and DSM is markedly greater than the historical aerial photography and Nearmap imagery (Fig. 5). This increase in resolution and some observed foliage recovery in the warmer temperatures of spring likely contributed to a marginal increase in live mangrove extent between August and September 2020. The trajectory of live and dead mangrove extent is indicated in Table 4.

Spatial Patterns of Mangrove Dieback

In 2020, live mangroves occurred at higher elevations (0.5 ± 0.1 m AHD) than dead mangroves (0.4 ± 0.2 m AHD) ($p < 0.0001$) (Fig. 6a). This elevation threshold is consistent with the elevation at which extended inundation occurred in the water level records (Fig. 2b), particularly if

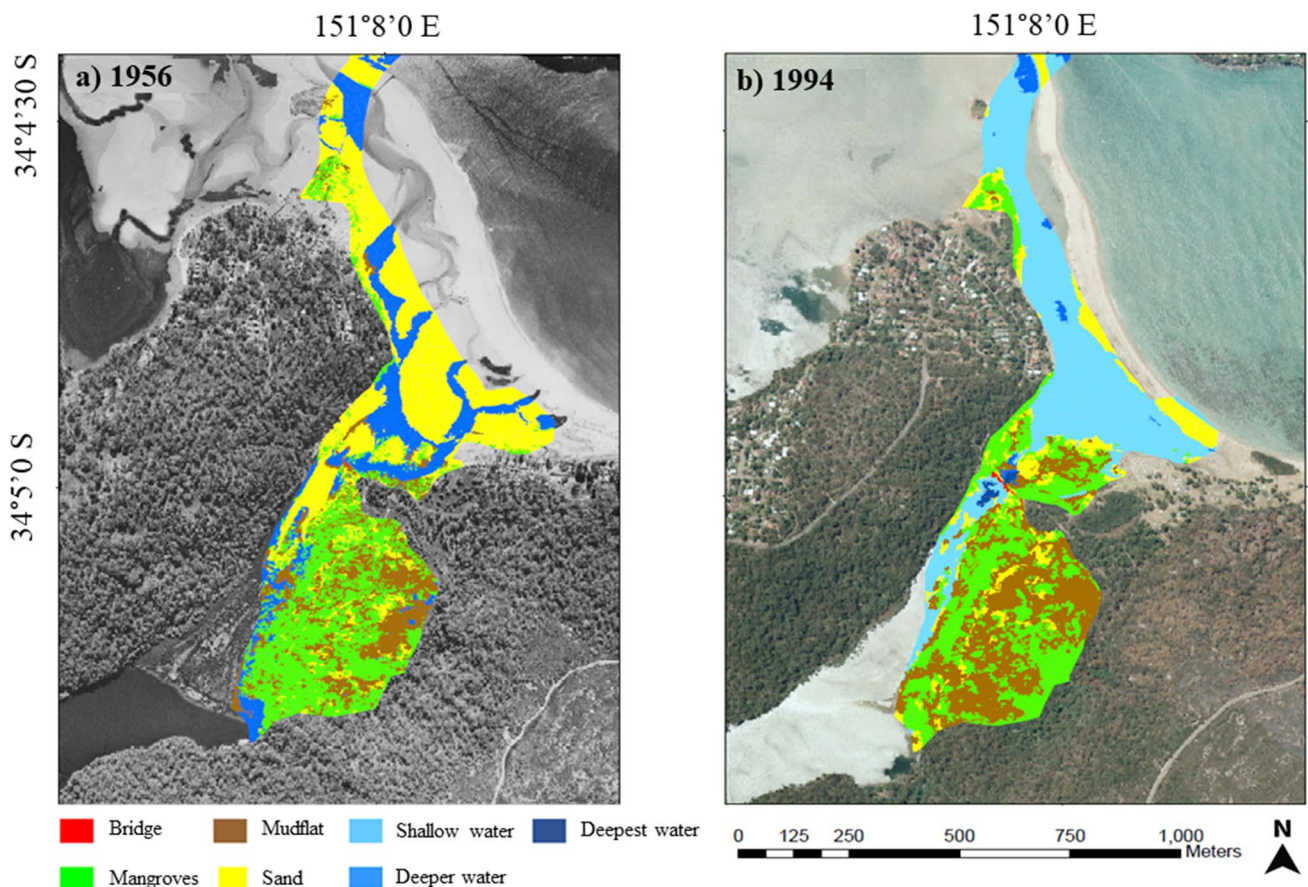


Fig. 3 Change in vegetation extent and sand barrier morphology from **a** 1956 to **b** 1994. See Supplementary Fig. 2) for historical aerial images without classification overlay

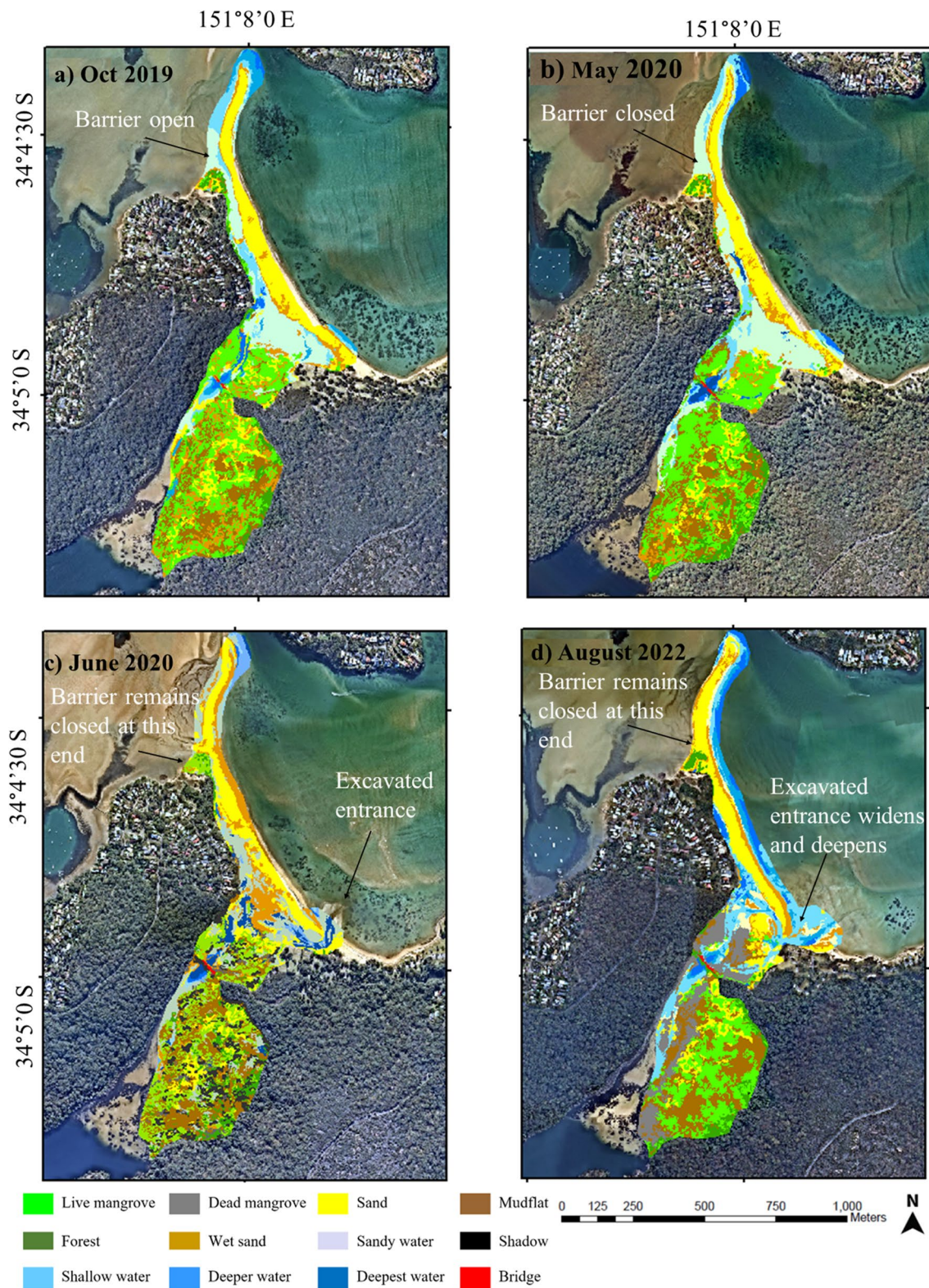
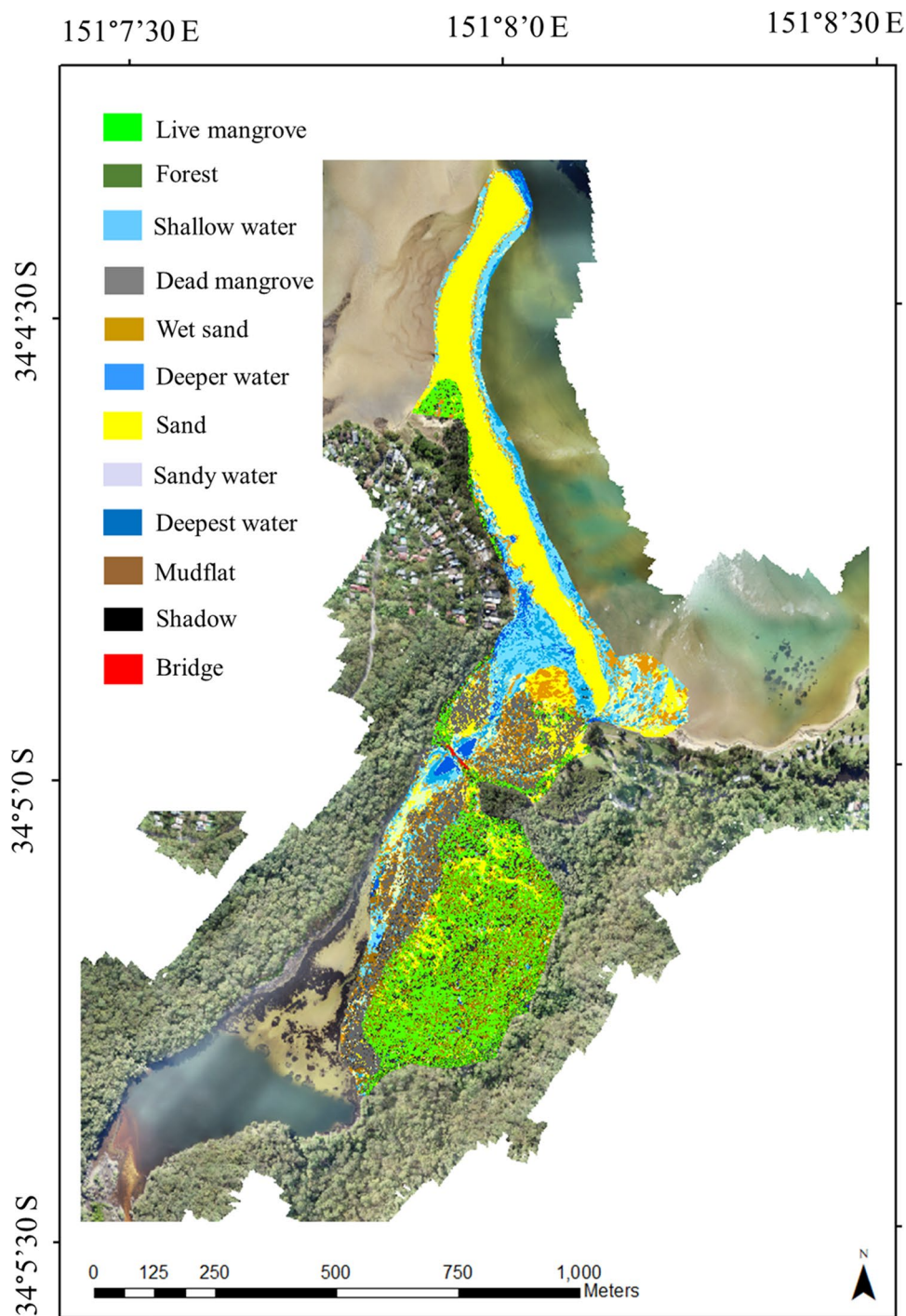


Fig. 4 Nearmap OBIA classifications showing change in mangrove extent and condition, sand, mudflat, and water (**a** October 2019, **b** May 2020, **c** June 2020, and **d** August 2022). See Supplementary Fig. 3 for Nearmap images without classification overlay

Table 4 Area of live and dead mangroves (ha) derived from different image sources, including historical aerial imagery (H), Nearmap (NM), and RPA-derived imagery, acquired at different times

Image acquisition	1956	1994	10/2019	05/2020	06/2020	08/2022	09/2022
Image source/class	Historical aerial imagery	Historical aerial imagery	Nearmap	Nearmap	Nearmap	Nearmap	Remotely piloted aircraft
Live mangrove (ha)	10.9	11.5	10.8	10.4	7.7	6.6	8.0
Dead mangrove (ha)	0.0	0.0	0.0	0.1	0.9	4.9	3.9

Fig. 5 OBIA classified RPA orthomosaic (September 2022). The excavated entrance has widened and eroded along the shoreline. Large areas of dead mangrove are dead towards the centre of the channel i.e. the lower elevation regions. See Supplementary Fig. 4 for orthomosaic image without classification overlay



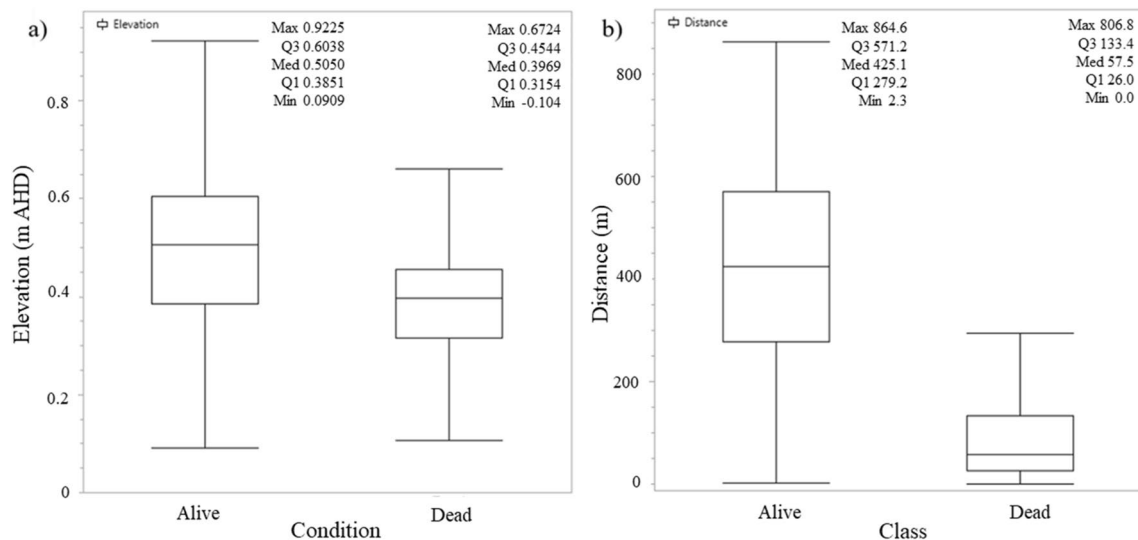


Fig. 6 The distribution of live and dead mangrove is significantly ($p < 0.0001$) related to **a** elevation (m AHD) and **b** distance (m) to the estuary entrance. The median elevation for live and dead mangrove

was 0.5 m and 0.39 m, and the median distance for live and dead mangrove was 425.1 m and 57.5 m respectively

pneumatophore and lenticel heights are around 10–15 cm. Live mangrove was also located further from the entrance (424 ± 195 m) than dead mangroves (107 ± 127 m) (Fig. 6b) ($p < 0.0001$).

Spatial Patterns in Vegetation Health

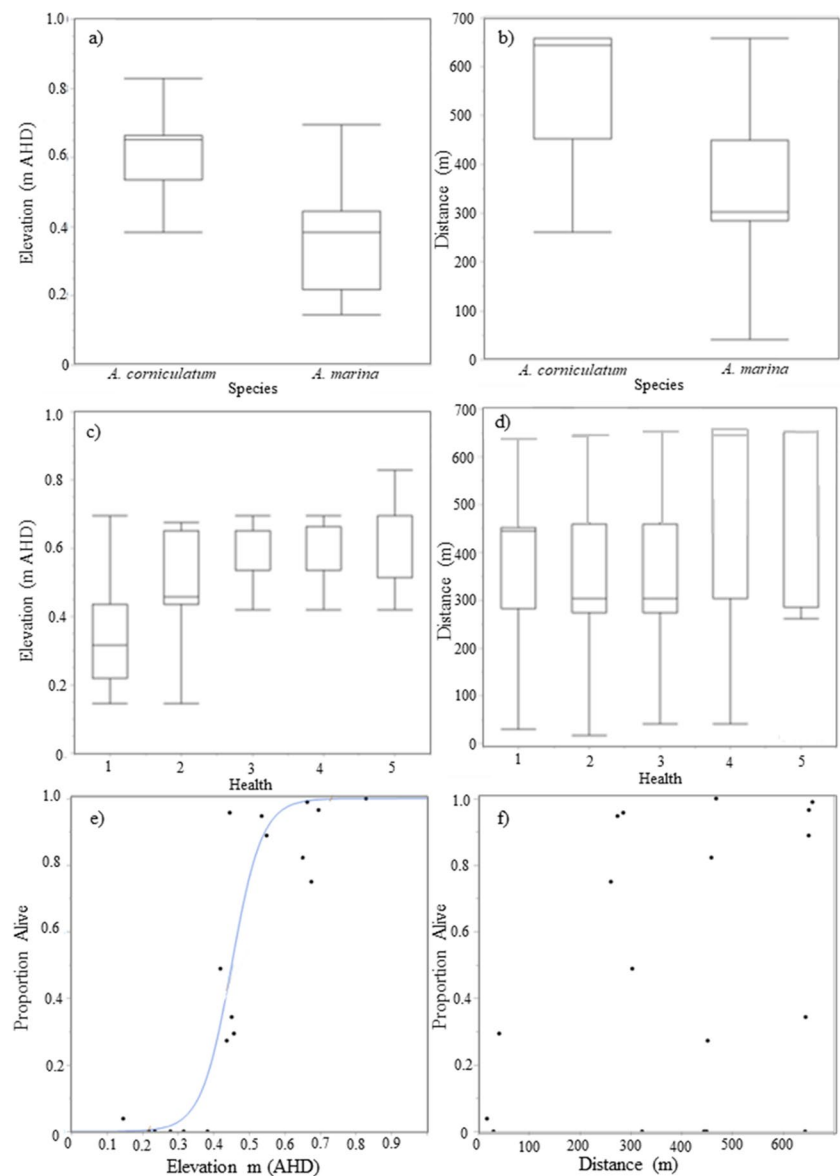
A. marina tended to be distributed at lower tidal positions (0.3 ± 0.1 m AHD) than *A. corniculatum* (0.6 ± 0.1 m AHD) (Fig. 7a) and these differences were statistically significant ($p < 0.0001$). *A. marina* also tended to occur at positions closer to the estuary entrance (312 ± 173 m) than *A. corniculatum* (520 ± 163 m), and this relationship was also significant ($p < 0.0001$) (Fig. 7b), but the interaction effect between elevation and distance was not significant ($p = 0.4323$). Lower health ratings (based on physical characteristics and the appearance of live biomass (Supplementary Fig. 1)) were also evident at lower elevations (Fig. 7c) and the differences were statistically significant ($p < 0.0001$). Health ratings did not show an obvious relationship with distance from the entrance ($p = 0.0993$); however, the interaction effect was significant ($p < 0.0001$) (Fig. 7d).

The proportion of live individuals per plot was related to elevation using a 2-parameter logistic model ($R^2 = 0.8345$, $RMSE = 0.1765$) (Fig. 7e). Partitioning indicated that the proportion of live individuals was very low at elevations < 0.45 m AHD; moderate at elevations of 0.45 – 0.65 m AHD, and a high at elevations exceeding 0.65 m AHD. A robust relationship was not established between the proportion of live individuals and distance (Fig. 7f).

Implications for Mangrove Above-Ground Biomass

Analyses of pre- and post-dieback AGB indicated substantially more biomass was lost from *A. marina* (dead biomass = 33.1 ± 3.9 kg), than *A. corniculatum* (dead biomass = 0.55 ± 0.05 kg) (Supplementary Fig. 5). The distribution of biomass was significantly modified by elevation and distance ($p = 0.0129$); however, only the elevation parameter was significant ($p = 0.0036$), with both distance ($p = 0.8490$) and the interaction effect ($p = 0.4001$) not being significant. The bulk of dead biomass was distributed at lower elevations, with dead biomass still evident at greater distances from the entrance (Fig. 8a). A distinct decision tree partition was established at an elevation of 0.54 m; below this elevation, dead AGB was in the order of 20.9 ± 12.3 Mg/ha; above this elevation, dead AGB was 2.01 ± 1.36 Mg/ha. A similar partition was established based on distance from the entrance of 452 m; below this distance, dead AGB was 19.8 ± 14.8 Mg/ha, whilst dead AGB was 5.65 ± 5.31 Mg/ha at greater distances (Fig. 8b). This variation was largely an outcome of substrates exhibiting higher elevations at greater distances (i.e. elevation is the predominant influence on dieback). The total area of dead mangroves in September 2020 (3.9 ha) equated to a loss of 81.5 ± 48 Mg of AGB, 38.1 ± 22.5 Mg C, or 140 ± 82 Mg CO_2e .

Fig. 7 Box and whiskers plots of field-based measurements of mangrove species distribution with respect to **a** elevation (m AHD) and **b** distance (m) to estuary entrance; and box and whiskers plots of field-based measurements of mangrove health with respect to **c** elevation (m AHD) and **d** distance (m) to estuary entrance. The relationships between the proportion of live individuals within plots, **e** elevation (fitted with logarithm -parameter model ($1/(1 + \exp(-a*(\text{Elevation} - b)))$)) shown in the blue line and, **f** distance to the entrance



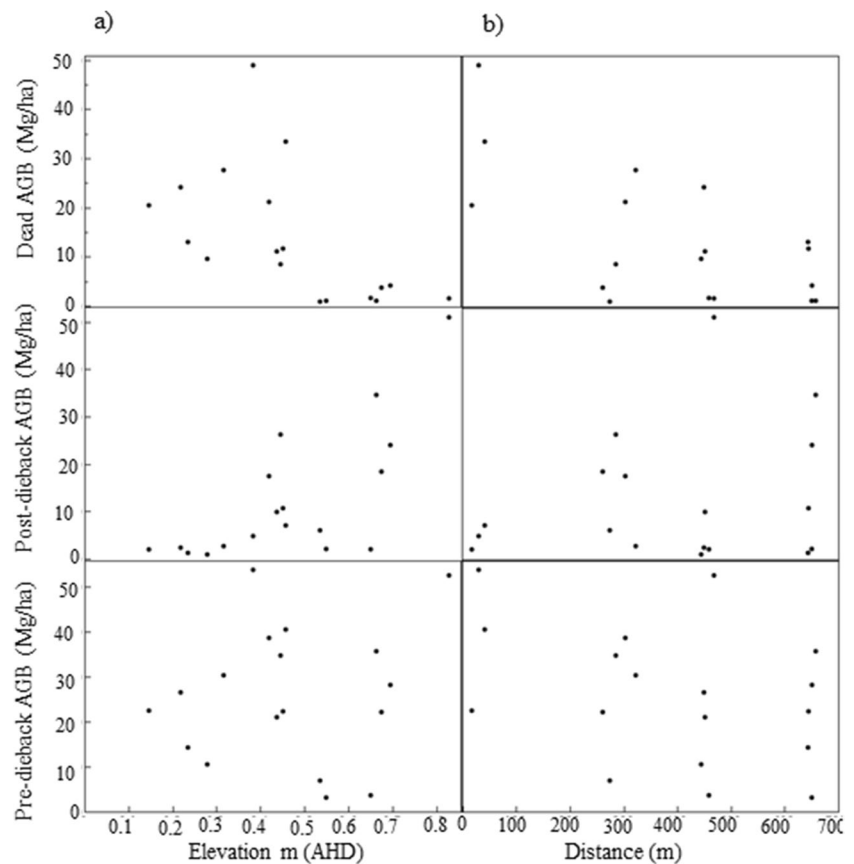
Discussion

Historical assessments revealed this was the first recorded mangrove dieback event at Cabbage Tree Basin, with no evidence of previous entrance closure. The dieback coincided with several storm events in early 2020 with deepwater significant wave heights > 7 m (recorded in February 2020 from the Sydney waverider buoy). The large swell from several storms led to significant sediment transportation and deposition along Deeban Spit, resulting in almost complete closure of the entrance, tidal impoundment, persistent inundation, and dieback of mangroves. The intent of the study was to characterise the drivers of dieback, with reference to whether elevation of the substrate or distance from the entrance was related to the extent of dieback and the condition of mangroves more broadly.

Relationships Between Mangrove Condition, Species, Elevation, and Distance

The spatial patterns of dieback and recovery at Cabbage Tree Basin highlight how tidal impoundment and position in the tidal frame (i.e. elevation) influenced mortality, resulting in loss of AGB and carbon. Field and aerial surveys revealed that dead individuals and those with lower health (condition) ratings were primarily confined to lower elevations compared to live individuals. Considering these analyses, our hypothesis was correct: mangroves experiencing dieback were primarily located at lower elevation positions within the tidal frame and generally located closer to the estuary entrance. These findings may be somewhat related to the fact that pre-dieback there were more mangroves located further from the estuary entrance. Considering this, it is unlikely

Fig. 8 Pre-dieback, post-dieback, and dead AGB (Mg/ha) with respect to **a** elevation (m AHD) and **b** distance (m) to estuary entrance



that dieback was an effect of sediment smothering, as an overwash event extending to distances of > 250 m is highly unlikely. Irrespective, it is likely, and observations did confirm, smothering of mangrove pneumatophores in very close proximity to the entrance, aligning with observations that have been made elsewhere (Ellison 1999).

Both *A. marina* and *A. corniculatum* experienced dieback at similar levels; however, field-based assessments identified *A. marina* individuals were primarily located at lower tidal positions, closer to the estuary entrance compared to *A. corniculatum* (Fig. 6a and b). Species position in the tidal frame (vertical distribution) often reflects species-specific physiological tolerances in relation to salinity, sediment moisture, and nutrient availability across the intertidal gradient; this is largely moderated by surface elevation and local geomorphology. *A. marina*, often described as a pioneer species as it is often the first to colonise, recover, and adapt following environmental change due to its capacity to survive across a broad physiochemical gradient, including lower elevation positions (Ball 1988; van Hespén et al. 2022). In that regard, it is unlikely the dieback is linked to increases in salinity concentration as mangroves, particularly *A. marina*, can adapt and withstand relatively rapid and large fluctuations in salinity (Khan and Aziz 2001; Ye et al. 2005; Nguyen et al. 2015; Li et al. 2020). Differences in AGB loss

between species is due to the difference in structure at Cabbage Tree Basin, with *A. marina* generally much taller than *A. corniculatum*.

Elevation and Position in the Tidal Frame Can Drive Dieback Due to Prolonged Inundation

This investigation is consistent with published literature addressing similar inundation events in coastal wetlands, highlighting that although mangroves thrive in dynamic intertidal zones, persistent inundation for weeks and months can result in physiological stress and dieback (Erftemeijer and Hamerlynck 2005; Adams and Human 2016; Yu and Gao 2022) as oxygen exchange in the root zone is limited and the environment becomes anoxic (Li et al. 2020). This idea is supported by the gradient of impact at Cabbage Tree Basin, with greater dieback at lower elevations. During the period of prolonged inundation and dieback, it is likely root turnover was reduced due to the mortality of individual trees; this exacerbates the impact of the disturbance as surface elevation loss can occur rapidly (Krauss et al. 2010) increasing the depth and duration of flooding, further exacerbating the physiological stress experienced by the individual tree.

Similar dieback events have been documented with maximum elevations of impact indicating persistent inundation. For example, Lagomasino et al. (2021) found that storm surges in wetlands with poorly drained soils led to persistent flood conditions, resulting in mangrove dieback, likely due to a reduction in gas exchange. Moreover, studies at Kosi Bay and St Lucia estuary, South Africa, identified mangrove dieback was associated with low-elevation waterlogged sites with a high redox potential and sediment anoxia (Breen and Hill 1969; Adams and Human 2016). There are also many studies identifying estuary mouth closure events have led to mangrove dieback due to persistent waterlogging and reduced capacity for gas exchange i.e. negative (reduced) redox potential (Branch and Branch 1981; Choy and Booth 1994; Mbense et al. 2016).

Implications of Mangrove Dieback

The implications of mangrove dieback at Cabbage Tree Basin (and dieback events across Australia and globally) range from local impacts to recreation and tourist value and loss of biodiversity to wider scale impacts such as the wetland's capacity to immobilise pollutants and sequester blue carbon (Queiroz et al. 2022). In recent years, mangroves have received considerable attention for their value as a blue carbon ecosystem, with approximately 50–99% of coastal blue carbon stored below ground in mangroves, tidal marshes, and seagrass meadows (Donato et al. 2012; Sasmito et al. 2019). The dieback at Cabbage Tree Basin resulted in a considerable loss of 81.5 ± 48 Mg of AGB and 38.1 ± 22.5 Mg C, which is equivalent to 140 ± 82 Mg CO₂e. Carbon-related impacts are likely to persist as the wetland recovers with soil carbon stocks continuing to fall rapidly post-dieback potentially increasing the carbon impacts of the dieback event (Sippo et al. 2019).

Recovery of mangroves will require a regular and plentiful supply of propagules (Duke and Burns 1999). This may be possible at Cabbage Tree Basin as there are large areas of live mangrove at higher elevations; however, establishment and growth may be limited as sea levels rise (Church et al. 2006; Dowdy et al. 2014) potentially pushing saplings beyond their tolerance thresholds for inundation and salinity (Lovelock et al. 2017; Riascos et al. 2018; Sippo et al. 2020a). For example, Balke et al. (2011) identified that the successful establishment of *Avicennia alba* propagules requires an initial period without inundation, to allow the roots to grow beyond a minimum length where after the seedling can withstand waves and currents. Seedling establishment and growth may also be limited by future physical disturbances such as flooding, storm surge, and wave action (Proisy et al. 2009; Mbense et al. 2016). Recovery may therefore be limited temporally and spatially as seedling establishment requires windows of opportunity where

environmental conditions are ideal for propagule growth, dispersal, and successful establishment. Monitoring the success of propagule establishment and understanding these thresholds, particularly for each species, is fundamental for identifying conditions that offer windows of opportunity, and for management of coastal wetlands, particularly when considering active (i.e. planting) or passive restoration strategies (Balke et al. 2011; van Bijsterveldt et al. 2022). Many studies and management agencies advocate for the restoration of the mangrove habitat (i.e. reinstatement of tidal flows, erosion control, and removal of barriers) to promote natural recruitment rather than actively planting seedlings. This is often due to high failure rates for planting and widespread misunderstanding of the conditions required for successful establishment (Lewis III 2005; Winterwerp et al. 2013; Balke and Friess 2016). This study highlights the need to understand the conditions required for successful propagule establishment, particularly across a range of species and geomorphic settings.

Mangrove recovery at Cabbage Tree Basin may also be further compounded as there are only two mangrove species that can grow in south-eastern NSW, and the stress threshold has already been exceeded for a large portion of these trees resulting in reduced resilience and slower recovery (Jimenez et al. 1985). Studies suggest that successful recruitment and regeneration following dieback events could take several decades depending on the hydrological conditions and suitability of the conditions (Lewis III 2005; Duarte et al. 2015).

Continuing variation in the environment is likely with climate change, and in the short term there are likely to be ongoing changes in sediment dynamics as the newly excavated channel becomes the dominant entrance. This may have longer-term implications for near-shore inundation patterns that could be further exacerbated by sea-level rise and increased frequency of storm surges during La Niña periods (Zhao et al. 2016). Although the new channel was created with the best intentions to release tidally impounded waters and reduce inundation stress on mangroves, these on-ground actions resulted in significant erosion and should have been carefully considered and legally approved to ensure hydrological restoration improved the overall ecological condition. Considerations could have included an assessment of the DEM to identify preferential paths of shallow water flow and inform site selection for the excavation of primary and secondary channels (Pérez-Ceballos et al. 2020). The potential for sediment subsidence due to the decomposition of dead roots should also be considered as this adds complexity to flow pathways (Krauss et al. 2018). By assessing these factors and perhaps using hydraulic modelling to simulate various restoration options (Marois and Mitsch 2017), the severity of erosion at the entrance may have been reduced, whilst reactivating the hydroperiod and reconnecting the wetland with Simpsons Bay to allow for regular tidal

flushing (Pérez-Ceballos et al. 2020). Ongoing monitoring of erosion, sediment transport, water levels, and biogeochemical parameters is recommended to understand the short- and long-term impacts and to allow for early identification of further hydrological stress that may impair wetland recovery.

The extent to which the system will recover to its pre-existing status of a permanently open system is unclear. Experience elsewhere on the NSW coast, such as Lake Illawarra, indicates that if the desire is for an intermittently closed system to be open then ongoing entrance management is required. Transition to a different stable state (sometimes called state shift or regime shift) is likely as the hydrological and sediment conditions have changed considerably. Regime shifts have been noted for many wetlands following mangrove dieback as ecological (and species) tolerance thresholds are surpassed. For example, Mbense et al. (2016) noted a shift from a healthy *A. marina*-dominated forest to mudflats and eventually saltmarsh following estuary entrance closure as a result of large sea storm deposits. Rapid conversion from mangrove to mudflat has also been observed following cyclone-induced dieback and resultant peat collapse (Osland et al. 2020). The changes in sediment elevation significantly alter inundation and salinity regimes limiting mangrove recovery. Rapid conversion from mangroves to saltmarsh or mudflats decreases blue carbon storage and may lead to a loss of habitat for fish, birds, and other invertebrates (Cavanaugh et al. 2019).

Sea-level rise may facilitate opportunities for higher elevation areas to become suitable environments for wetland vegetation potentially extending the mangrove area inland and offsetting some of the carbon losses (Sippo et al. 2018). The RPA DSM indicates that most of the live mangroves are already higher in the tidal frame (0.5 ± 0.1 m AHD, with a maximum range to 0.9m), with further large areas that could be suitable for mangrove colonisation. However, inland retreat may be limited by propagule supply if established mangroves are not able to recover to sufficient levels.

Recommendation for Future Management

Mangrove dieback has been associated with extreme climatic events such as tropical cyclones (Asbridge et al. 2018; Krauss and Osland 2020; Lagomasino et al. 2021). As the intensity of such events increases with climate change, many wetlands may experience the next extreme event before getting the opportunity to recover from the previous disturbance. This reduces the resilience of the system (McLeod and Salm 2006; Huxham et al. 2010; Walcker et al. 2019). The frequency of dieback events and their capacity for recovery is further compounded by increasing anthropogenic threats and risks such as modified freshwater flows and clearance for agricultural and urban development (Day et al. 2018; Asbridge and Dwyer 2023). It is for this reason

that management of Cabbage Tree Basin must be tailored and include ongoing monitoring to understand the dynamic responses to climate, environment, and anthropogenic drivers. This site also provides a unique opportunity to educate and promote the importance of coastal wetlands within the Greater Sydney region as for many residents this may be their only access to a wetland system.

Ongoing monitoring at Cabbage Tree Basin is required to assess impact and recovery and guide tailored management approaches. This is particularly important as the site is within the Royal National Park and is recognised as a site of Aboriginal cultural significance with many remaining artefacts, rock carvings, hand paintings, and middens in the area (West and West 2000). Management plans need to consider monitoring the newly excavated channel and the wider implications for erosion and accretion along the coastline to understand the implications for hydrological regimes and mangrove recovery. Changing sediment dynamics influencing mangrove extent and condition were observed in the Cambridge Gulf-Ord River, Western Australia, where tall *Avicennia* trees were undercut exposing root systems leading to riverbank collapse and mangrove mortality. The unstable sediment limited seedling growth with uprooting and undercutting of sediment (Thom et al. 1975). Similarly, along the coast of French Guiana, the fluctuating phases of erosion and accretion demonstrate the significant influence of sediment dynamics reshaping the mangrove coastline. During periods of high accretion, hundreds of hectares are colonised by mangrove seedlings, yet during successive erosion phases large areas of mangroves are lost as sediment and trees are undercut (Fromard et al. 1998; Proisy et al. 2009). These studies highlight the importance and driving force of sediment and hydrological dynamics in shaping the wetland environment and the conditions required for propagule establishment and growth. This study documents somewhat unique and localised phenomena in terms of tidal impoundment from a series of storm events, and subsequent barrier excavation. However, understanding the drivers, processes, and impacts at Cabbage Tree Basin is important for documenting and highlighting the short- and long-term implications and to direct tailored management centred on abrupt ecosystem shifts in coastal wetlands and inform tipping point thresholds for mangroves. This information is globally applicable and timely as natural resource managers attempt to develop protocols for early detection of physiological stress and mitigate the effects of widespread dieback in a changing climate (Eslami-Andergoli et al. 2015).

It is recommended that the combination of approaches presented in this study should be considered as a guide, when designing dieback monitoring programs, within Australia and internationally. Integrating field and remote sensing data at a variety of spatial and temporal scales allows for a comprehensive assessment of changes in extent and

condition. Lewis III et al. (2016) suggested that regular field and remote sensing monitoring should be incorporated into coastal management plans for early detection of stress in mangrove forests and to guide and direct rehabilitation strategies. In some cases, transformative solutions, reactive strategies, and pre-emptive management with a focus on physiological thresholds and adaption pathways may prevent severe mortality or loss in mangrove conditions (Swart et al. 2013).

Supplementary Information The online version contains supplementary material available at <https://doi.org/10.1007/s12237-024-01426-8>.

Acknowledgements The authors would like to acknowledge the land on which this research was undertaken and the Gweagal people who managed the Cabbage Tree Basin for millennia. NSW National Parks and Wildlife provided a scientific license for fieldwork activities and Mishy McKensy provided considerable logistical support. We would also like to thank the reviewers of the manuscript who dedicated considerable time and effort, with their feedback greatly improving the paper.

Funding Open Access funding enabled and organized by CAUL and its Member Institutions.

Open Access This article is licensed under a Creative Commons Attribution 4.0 International License, which permits use, sharing, adaptation, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if changes were made. The images or other third party material in this article are included in the article's Creative Commons licence, unless indicated otherwise in a credit line to the material. If material is not included in the article's Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this licence, visit <http://creativecommons.org/licenses/by/4.0/>.

References

- Abhik, S., P. Hope, H.H. Hendon, L.B. Hutley, S. Johnson, W. Drosowsky, J.R. Brown, and N.C. Duke. 2021. Influence of the 2015–2016 El Niño on the record-breaking mangrove dieback along northern Australia coast. *Scientific Reports* 11: 20411.
- Adams, J., and L. Human. 2016. Investigation into the mortality of mangroves at St. Lucia Estuary. *South African Journal of Botany* 107: 121–128.
- Asbridge, E., R. Lucas, C. Ticehurst, and P. Bunting. 2016. Mangrove response to environmental change in Australia's Gulf of Carpentaria. *Ecology and Evolution* 6: 3523–3539.
- Asbridge, E., R. Lucas, K. Rogers, and A. Accad. 2018. The extent of mangrove change and potential for recovery following severe Tropical Cyclone Yasi, Hinchinbrook Island, Queensland, Australia. *Ecology and Evolution* 8: 10416–10434.
- Asbridge, E., R. Bartolo, C.M. Finlayson, R.M. Lucas, K. Rogers, and C.D. Woodroffe. 2019. Assessing the distribution and drivers of mangrove dieback in Kakadu National Park, northern Australia. *Estuarine, Coastal and Shelf Science* 228: 106353.
- Asbridge, E., and P. Dwyer. 2023. Marine vegetation management strategies: a framework for estuary wide prioritisation of protection and rehabilitation. *Restoration Ecology* 31: e13819. <https://doi.org/10.1111/rec.13819>
- Balke, T., and D.A. Friess. 2016. Geomorphic knowledge for mangrove restoration: A pan-tropical categorization. *Earth Surface Processes and Landforms* 41: 231–239.
- Balke, T., T.J. Bouma, E.M. Horstman, E.L. Webb, P.L. Erftemeijer, and P.M. Herman. 2011. Windows of opportunity: Thresholds to mangrove seedling establishment on tidal flats. *Marine Ecology Progress Series* 440: 1–9.
- Ball, M.C. 1988. Salinity tolerance in the mangroves *Aegiceras corniculatum* and *Avicennia marina*. I. Water use in relation to growth, carbon partitioning, and salt balance. *Functional Plant Biology* 15: 447–464.
- Bartholomeus, H., K. Calders, T. Whiteside, L. Terryn, S.M. Krishna Moorthy, S.R. Levick, R. Bartolo, and H. Verbeeck. 2022. Evaluating data inter-operability of multiple UAV–LiDAR systems for measuring the 3D structure of Savanna Woodland. *Remote Sensing* 14: 5992.
- Blasco, F., P. Saenger, and E. Janodet. 1996. Mangroves as indicators of coastal change. *CATENA* 27: 167–178.
- Branch, G., and M. Branch. 1981. *The living shores of southern Africa*. Struik publishers.
- Breen, C., and B. Hill. 1969. A mass mortality of mangroves in the Kosi Estuary. *Transactions of the Royal Society of South Africa* 38: 285–303.
- Bursill, L. 2018. *The story of Deeban*. Worrige, NSW: Dharawal Publications.
- Cavanaugh, K.C., E.M. Dangremond, C.L. Doughty, A.P. Williams, J.D. Parker, M.A. Hayes, W. Rodriguez, and I.C. Feller. 2019. Climate-driven regime shifts in a mangrove–salt marsh ecotone over the past 250 years. *Proceedings of the National Academy of Sciences* 116: 21602–21608.
- Choy, S.C., and W.E. Booth. 1994. Prolonged inundation and ecological changes in an *Avicennia* mangrove: Implications for conservation and management. *Hydrobiologia* 285: 237–247.
- Church, J.A., J.R. Hunter, K.L. McInnes, and N.J. White. 2006. Sea-level rise around the Australian coastline and the changing frequency of extreme sea-level events. *Australian Meteorological Magazine* 55: 253–260.
- Clarke, P.J. (1995) The population dynamics of the mangrove shrub *Aegiceras corniculatum* (Myrsinaceae): fecundity, dispersal, establishment and population structure. *Proceedings of The Linnean Society of New South Wales* 115: 35–44.
- Day, R.H., S.T. Allen, J. Brenner, K. Goodin, D. Faber-Langendoen, and K.W. Ames. 2018. Ecological resilience indicators for mangrove ecosystems. Ecological Resilience Indicators for Five Northern Gulf of Mexico Ecosystems. NatureServe, NatureServe, Arlington, Virginia, USA.
- Donato, D.C., J.B. Kauffman, R.A. Mackenzie, A. Ainsworth, and A. Pfleger. 2012. Whole-island carbon stocks in the tropical Pacific: Implications for mangrove conservation and upland restoration. *Journal of Environmental Management* 97: 89–96.
- Dowdy, A.J., G.A. Mills, B. Timbal, and Y. Wang. 2014. Fewer large waves projected for eastern Australia due to decreasing storminess. *Nature Climate Change* 4: 283–286.
- Duarte, C.M., A. Borja, J. Carstensen, M. Elliott, D. Krause-Jensen, and N. Marbà. 2015. Paradigms in the recovery of estuarine and coastal ecosystems. *Estuaries and Coasts* 38: 1202–1212.
- Duke, N. 1991. A systematic revision of the mangrove genus *Avicennia* (Avicenniaceae) in Australasia. *Australian Systematic Botany* 4: 299–324.
- Duke, N.C. 2006. *Australia's mangroves: the authoritative guide to Australia's mangrove plants*. MER.
- Duke, N.C., J.M. Kovacs, A.D. Griffiths, L. Preece, D.J. Hill, P. Van Oosterzee, J. Mackenzie, H.S. Morning, and D. Burrows. 2017. Large-scale dieback of mangroves in Australia's Gulf of

- Carpentaria: A severe ecosystem response, coincidental with an unusually extreme weather event. *Marine and Freshwater Research* 68: 1816–1829.
- Duke, N.C., A.D. Canning, and J.R. Mackenzie. 2024. More intense severe tropical cyclones in recent decades cause greater impacts on mangroves bordering Australia's Great Barrier Reef. In *Oceanographic Processes of Coral Reefs*, 226–253. CRC Press.
- Duke, N.C., and K.A. Burns. 1999. *Fate and effects of oil and dispersed oil on mangrove ecosystems in Australia*. Australia: Australian Institute of Marine Science Townsville.
- Duke, N., A. Haller, S. Brisbane, A. Wood, and B. Rogers. 2010. Sinking centres in Moreton Bay Mangroves. Maps showing areas of unusual anoxic ponds and mangrove dieback in tidal wetlands of the Bay area in 2003–08. School of Biological Sciences.
- Ellison, J.C. 1999. Impacts of sediment burial on mangroves. *Marine Pollution Bulletin* 37: 420–426.
- English, S., C. Wilkinson, and V. Baker. 1997. Survey manual for tropical marine resources.
- Erftemeijer, P.L., and O. Hamerlynck. 2005. Die-back of the mangrove *Heritiera littoralis* dryland, in the Rufiji Delta (Tanzania) following El Nino floods. *Journal of Coastal Research* 228–235.
- Eslami-Andergoli, L., P. Dale, J. Knight, and H. McCallum. 2015. Approaching tipping points: A focussed review of indicators and relevance to managing intertidal ecosystems. *Wetlands Ecology and Management* 23: 791–802.
- Fatoyinbo, T.E., M. Simard, R.A. Washington-Allen, and H.H. Shugart. 2008. Landscape-scale extent, height, biomass, and carbon estimation of Mozambique's mangrove forests with Landsat ETM+ and Shuttle Radar Topography Mission elevation data. *Journal of Geophysical Research: Biogeosciences* 113: G02S06.
- Feliciano, E.A., S. Wdowski, M.D. Potts, S.-K. Lee, and T.E. Fatoyinbo. 2017. Estimating mangrove canopy height and above-ground biomass in the Everglades National Park with airborne LiDAR and TanDEM-X data. *Remote Sensing* 9: 702.
- Fromard, F., H. Puig, E. Mougin, G. Marty, J. Betoulle, and L. Cadamuro. 1998. Structure, above-ground biomass and dynamics of mangrove ecosystems: New data from French Guiana. *Oecologia* 115: 39–53.
- Goldberg, L., D. Lagomasino, N. Thomas, and T. Fatoyinbo. 2020. Global declines in human-driven mangrove loss. *Global Change Biology* 26: 5844–5855.
- Harris, R.J., E.C. Milbrandt, E.M. Everham, and B.D. Bovard. 2010. The effects of reduced tidal flushing on mangrove structure and function across a disturbance gradient. *Estuaries and Coasts* 33: 1176–1185.
- Hinwood, J., E. Mclean, and M. Trevethan. 2005. Spring tidal pumping. In *Coasts and Ports 2005: Coastal Living-Living Coast*; Australasian Conference; Proceedings, pp. 601–606. Barton, ACT: Institution of Engineers, Australia.
- Hughes, M.G., K. Rogers, and L. Wen. 2019. Saline wetland extents and tidal inundation regimes on a micro-tidal coast, New South Wales, Australia. *Estuarine, Coastal and Shelf Science* 227: 106297.
- Huxham, M., M.P. Kumara, L.P. Jayatissa, K.W. Krauss, J. Kairo, J. Langat, M. Mencuccini, M.W. Skov, and B. Kirui. 2010. Intra- and interspecific facilitation in mangroves may increase resilience to climate change threats. *Philosophical Transactions of the Royal Society B: Biological Sciences* 365: 2127–2135.
- Jimenez, J.A., A.E. Lugo, and G. Cintron. 1985. Tree mortality in mangrove forests. *Biotropica* 17: 177–185. <https://doi.org/10.2307/2388214>.
- Khan, M.A., and I. Aziz. 2001. Salinity tolerance in some mangrove species from Pakistan. *Wetlands Ecology and Management* 9: 229–233.
- Koh, H., S. Teh, and Kh'ng X and Raja Barizan R., 2018. Mangrove forests: Protection against and resilience to coastal disturbances. *Journal of Tropical Forest Science* 30: 446–460.
- Krauss, K.W., and M.J. Osland. 2020. Tropical cyclones and the organization of mangrove forests: A review. *Annals of Botany* 125: 213–234.
- Krauss, K.W., D.R. Cahoon, J.A. Allen, K.C. Ewel, J.C. Lynch, and N. Cormier. 2010. Surface elevation change and susceptibility of different mangrove zones to sea-level rise on Pacific high islands of Micronesia. *Ecosystems* 13: 129–143.
- Krauss, K.W., A.W. Demopoulos, N. Cormier, A. From, J.P. McClain-Counts, and R.R. Lewis Iii. 2018. Ghost forests of Marco Island: Mangrove mortality driven by belowground soil structural shifts during tidal hydrologic alteration. *Estuarine, Coastal and Shelf Science* 212: 51–62.
- Lagomasino, D., T. Fatoyinbo, E. Castañeda-Moya, B.D. Cook, P.M. Montesano, C.S. Neigh, L.A. Corp, L.E. Ott, S. Chavez, and D.C. Morton. 2021. Storm surge and ponding explain mangrove dieback in Southwest Florida following Hurricane Irma. *Nature Communications* 12: 1–8.
- Lagomasino, D., L. Fatoyinbo, E. Castaneda, B. Cook, P. Montesano, C. Neigh, C. Lawrence, L. Ott, S. Chavez, and D. Morton. 2020. Storm surge, not wind, caused mangrove dieback in southwest Florida following Hurricane Irma. *Nature Communications* 12: 4003.
- Lewis Iii, R.R. 2005. Ecological engineering for successful management and restoration of mangrove forests. *Ecological Engineering* 24: 403–418.
- Lewis Iii, R.R., E.C. Milbrandt, B. Brown, K.W. Krauss, A.S. Rovai, J.W. Beever Iii, and L.L. Flynn. 2016. Stress in mangrove forests: Early detection and preemptive rehabilitation are essential for future successful worldwide mangrove forest management. *Marine Pollution Bulletin* 109: 764–771.
- Li, H., Z. Li, Z.-J. Shen, M.-R. Luo, Y.-L. Liu, M.-Y. Wei, W.-H. Wang, Y.-Y. Qin, C.-H. Gao, and K.-K. Li. 2020. Physiological and proteomic responses of mangrove plant *Avicennia marina* seedlings to simulated periodical inundation. *Plant and Soil* 450: 231–254.
- Lovelock, C.E., I.C. Feller, R. Reef, S. Hickey, and M.C. Ball. 2017. Mangrove dieback during fluctuating sea levels. *Scientific Reports* 7: 1680.
- Lucas, R., C.M. Finlayson, R. Bartolo, K. Rogers, A. Mitchell, C.D. Woodroffe, E. Asbridge, and E. Ens. 2017. Historical perspectives on the mangroves of Kakadu National Park. *Marine and Freshwater Research* 69: 1047–1063.
- Lugo, A.E., G. Cintrón, and C. Goenaga. 1981. *Mangrove ecosystems under stress*. Chichester, UK: John Wiley and Sons Limited.
- Marois, D.E., and W.J. Mitsch. 2017. A mangrove creek restoration plan utilizing hydraulic modeling. *Ecological Engineering* 108: 537–546.
- Mbense, S., A. Rajkaran, U. Bolosha, and J. Adams. 2016. Rapid colonization of degraded mangrove habitat by succulent salt marsh. *South African Journal of Botany* 107: 129–136.
- Mcleod, E., and R.V. Salm. 2006. *Managing mangroves for resilience to climate change*. Switzerland: World Conservation Union (IUCN) Gland.
- Milbrandt, E., J. Greenawalt-Boswell, P. Sokoloff, and S. Bortone. 2006. Impact and response of southwest Florida mangroves to the 2004 hurricane season. *Estuaries and Coasts* 29: 979–984.
- Nguyen, H.T., D.E. Stanton, N. Schmitz, G.D. Farquhar, and M.C. Ball. 2015. Growth responses of the mangrove *Avicennia marina* to salinity: Development and function of shoot hydraulic systems require saline conditions. *Annals of Botany* 115: 397–407.
- Nielsen, P. 2009. *Coastal and estuarine processes*. World Scientific Publishing Company.

- Osland, M.J., L.C. Feher, G.H. Anderson, W.C. Vervaeke, K.W. Krauss, K.R. Whelan, K.M. Balentine, G. Tiling-Range, T.J. Smith, and D.R. Cahoon. 2020. A tropical cyclone-induced ecological regime shift: Mangrove forest conversion to mudflat in Everglades National Park (Florida, USA). *Wetlands* 40: 1445–1458.
- Owers, C.J., K. Rogers, and C.D. Woodroffe. 2018a. Terrestrial laser scanning to quantify above-ground biomass of structurally complex coastal wetland vegetation. *Estuarine, Coastal and Shelf Science* 204: 164–176.
- Owers, Christopher J., Kerry Lee Rogers, and Colin D. Woodroffe. 2018b. Spatial variation of above-ground carbon storage in temperate coastal wetlands. *Estuarine, Coastal and Shelf Science* 210: 55–67.
- Pérez-Ceballos, R., A. Zaldívar-Jiménez, J. Canales-Delgadillo, H. López-Adame, J. López-Portillo, and M. Merino-Ibarra. 2020. Determining hydrological flow paths to enhance restoration in impaired mangrove wetlands. *PLoS ONE* 15: e0227665.
- Proisy, C., N. Gratiot, E.J. Anthony, A. Gardel, F. Fromard, and P. Heuret. 2009. Mud bank colonization by opportunistic mangroves: A case study from French Guiana using lidar data. *Continental Shelf Research* 29: 632–641.
- Queiroz, H.M., T.O. Ferreira, V.A. Fandiño, I.O.B.F. Bragantini, D. Barcellos, G.N. Nóbrega, A.D. Ferreira, L.E. De Oliveira Gomes, and A.F. Bernardino. 2022. Changes in soil iron biogeochemistry in response to mangrove dieback. *Biogeochemistry* 158: 357–372.
- Riascos, J.M., J.R. Cantera, and J.F. Blanco-Libreros. 2018. Growth and mortality of mangrove seedlings in the wettest Neotropical mangrove forests during ENSO: Implications for vulnerability to climate change. *Aquatic Botany* 147: 34–42.
- Rogers, K., N. Saintilan, and D. Cahoon. 2005. Surface elevation dynamics in a regenerating mangrove forest at Homebush Bay, Australia. *Wetlands Ecology and Management* 13: 587–598.
- Rogers, K., P.I. Boon, S. Branigan, N.C. Duke, C.D. Field, J.A. Fitzsimons, H. Kirkman, J.R. Mackenzie, and N. Saintilan. 2016. The state of legislation and policy protecting Australia's mangrove and salt marsh and their ecosystem services. *Marine Policy* 72: 139–155.
- Rogers, K., A. Zawadzki, L.A. Mogensen, and N. Saintilan. 2022. Coastal wetland surface elevation change is dynamically related to accommodation space and influenced by sedimentation and sea-level rise over decadal timescales. *Frontiers in Marine Science* 9: 807588.
- Roy, P.S., R.J. Williams, A.R. Jones, I. Yassini, P.J. Gibbs, B. Coates, R.J. West, P.R. Scanes, J.P. Hudson, and S. Nichol. 2001. Structure and function of Southeast Australian estuaries. *Estuarine, Coastal and Shelf Science* 53: 351–384.
- Sakho, I., V. Mesnage, J. Deloffre, R. Lafite, I. Niang, and G. Faye. 2011. The influence of natural and anthropogenic factors on mangrove dynamics over 60 years: The Somone Estuary, Senegal. *Estuarine, Coastal and Shelf Science* 94: 93–101.
- Sasmito, S.D., P. Taillardat, J.N. Clendenning, C. Cameron, D.A. Friess, D. Murdiyarso, and L.B. Hutley. 2019. Effect of land-use and land-cover change on mangrove blue carbon: A systematic review. *Global Change Biology* 25: 4291–4302.
- Sippo, J.Z., C.E. Lovelock, I.R. Santos, C.J. Sanders, and D.T. Maher. 2018. Mangrove mortality in a changing climate: An overview. *Estuarine, Coastal and Shelf Science* 215: 241–249.
- Sippo, J.Z., D.T. Maher, K.G. Schulz, C.J. Sanders, A. McMahon, J. Tucker, and I.R. Santos. 2019. Carbon outwelling across the shelf following a massive mangrove dieback in Australia: Insights from radium isotopes. *Geochimica Et Cosmochimica Acta* 253: 142–158.
- Sippo, J.Z., C.J. Sanders, I.R. Santos, L.C. Jeffrey, M. Call, Y. Harada, K. Maguire, D. Brown, S.R. Conrad, and D.T. Maher. 2020a. Coastal carbon cycle changes following mangrove loss. *Limnology and Oceanography* 65: 2642–2656.
- Sippo, J.Z., I.R. Santos, C.J. Sanders, P. Gadd, Q. Hua, C.E. Lovelock, N.S. Santini, S.G. Johnston, Y. Harada, and G. Reithmeir. 2020b. Reconstructing extreme climatic and geochemical conditions during the largest natural mangrove dieback on record. *Biogeosciences* 17: 4707–4726.
- Swart, R., S. Fuss, M. Obersteiner, P. Ruti, C. Teichmann, and R. Vautard. 2013. Beyond vulnerability assessment. *Nature. Climate Change* 3: 942–943.
- Thom, B.G., L. Wright, and J.M. Coleman. 1975. Mangrove ecology and deltaic-estuarine geomorphology: Cambridge Guildford river, Western Australia. *The Journal of Ecology* 63 (1): 203–232.
- Van Bijsterveldt, C.E., A.O. Debrot, T.J. Bouma, M.B. Maulana, R. Pribadi, J. Schop, F.H. Tonneijck, and B.K. Van Wesenbeeck. 2022. To plant or not to plant: When can planting facilitate mangrove restoration? *Frontiers in Environmental Science* 9: 690011.
- Van Hespén, R., Z. Hu, Y. Peng, Z. Zhu, T. Ysebaert, and T.J. Bouma. 2022. Identifying trait-based tolerance to sediment dynamics during seedling establishment across eight mangrove species. *Limnology and Oceanography* 67: 2281–2295.
- Vogt, J., A. Skóra, I.C. Feller, C. Piou, G. Coldren, and U. Berger. 2012. Investigating the role of impoundment and forest structure on the resistance and resilience of mangrove forests to hurricanes. *Aquatic Botany* 97: 24–29.
- Walcker, R., C. Laplanche, M. Herteman, L. Lambs, and F. Fromard. 2019. Damages caused by Hurricane Irma in the human-degraded mangroves of Saint Martin (Caribbean). *Scientific Reports* 9: 1–11.
- Wannasiri, W., M. Nagai, K. Honda, P. Santitamnont, and P. Miphokasap. 2013. Extraction of mangrove biophysical parameters using airborne LiDAR. *Remote Sensing* 5: 1787–1808.
- West, R., and J. West. 2000. Cabbage tree basin: Natural values and options for management. *Unpublished report for the Hacking Catchment Management Committee* 79. <https://dws.ssec.org.au/environment/bioregion/hacking/resources/reports/basin.pdf>.
- Williams, R. J., and A. J. Meehan. 2004. Focusing management needs at the sub-catchment level via assessments of change in the cover of estuarine vegetation, Port Hacking, NSW, Australia. *Wetlands Ecology and Management* 12: 499–518.
- Winterwerp, J., P. Erftemeijer, N. Suryadiputra, P. Van Eijk, and L. Zhang. 2013. Defining eco-morphodynamic requirements for rehabilitating eroding mangrove-mud coasts. *Wetlands* 33: 515–526.
- Woodroffe, C. 1992. Mangrove sediments and geomorphology. *Tropical Mangrove Ecosystems* 41: 7–41.
- Woodroffe, C., and D. Grime. 1999. Storm impact and evolution of a mangrove-fringed chenier plain, Shoal Bay, Darwin, Australia. *Marine Geology* 159: 303–321.
- Ye, Y., N.F.-Y. Tam, C.-Y. Lu, and Y.-S. Wong. 2005. Effects of salinity on germination, seedling growth and physiology of three salt-secreting mangrove species. *Aquatic Botany* 83: 193–205.
- Yu, M., and Q. Gao. 2022. Prolonged coastal inundation detected with synthetic aperture radar significantly retarded functional recovery of mangroves after major hurricanes. *Landscape Ecology* 1–15.
- Zhao, Q., J. Bai, L. Huang, B. Gu, Q. Lu, and Z. Gao. 2016. A review of methodologies and success indicators for coastal wetland restoration. *Ecological Indicators* 60: 442–452.