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Environmental state of a small intertidal estuary a decade after mangrove clearance, Waikaraka Estuary, Aotearoa New Zealand

D.J. Stokes [a,*], H.E. Glover [b], K.R. Bryan [c], C.A. Pilditch [c]

[a] Southern Cross University, Faculty of Science and Engineering, PO Box 157, Lismore, NSW, 2480, Australia
[b] College of Engineering, School of Civil & Construction Engineering, Oregon State University, 101 Kearney Hall, Corvallis, OR, 97331, USA
[c] University of Waikato, School of Science, Hamilton, 3240, New Zealand

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ABSTRACT

Unlike many parts of the world, mangrove coverage has been steadily increasing in Aotearoa, New Zealand since the 1900s. Intentional mangrove removal in Aotearoa New Zealand is often motivated by a desire to remove muddy sediment from sites that were once sand-dominated. Mangrove removal can result in geomorphic and ecologic evolution over decadal timescales, yet monitoring of removal sites is often limited to a few years. This study presents the result of over a decade of monitoring in Waikaraka Estuary, a quiescent embayment where mangroves were removed between 2003 and 2012. Surface elevation was monitored between 2006 and 2019, and sediment cores and surface samples collected in 2019 were analysed for grain size, total organic content, and root biomass to assess geomorphic change. Remaining mangroves areas and benthic macrofauna were surveyed throughout the estuary to assess ecologic change. Despite initially rapid movement of mud out of cleared areas, the estuary has experienced little bed-elevation change over a decade. The upper estuary remains muddy, and much of the root biomass has not decomposed. However, the region near the mouth of the estuary has become sandier with the return of bivalve species *Austrovenus stutchburyi* (cockles) and *Macomona liliana*. These long-term measurements demonstrate the slow pace of recovery following mangrove removal in quiescent embayments. Without major changes to hydrodynamics, mud is unlikely to be flushed out and bivalve habitat is unlikely to be restored even after a decade. Site-specific characteristics should be assessed when evaluating the efficacy of mangrove removal for estuary restoration, to moderate community expectations.

1. Introduction

Mangroves forests are ecosystems that occupy the intertidal zone in the tropics and sub-tropics. In most systems, a diversity of mangrove species provide numerous ecosystem services. Some mangrove species can also extend to temperate regions, providing similar ecosystem services in terms of habitat for terrestrial and marine benthic organisms, sequestering relatively high carbon loads (Morrisey et al., 2010; Bulmer et al., 2016; Johnson et al., 2020; Wong et al., 2021), and dissipating wave energy (Winter et al., 2019). Globally, mangrove forests have been cleared at rates of up to 3.6% per annum (Valiela et al., 2001), although lower rates have been reported since the turn of the century (Friess et al., 2019). Clearing may be initiated for development of aquaculture/agriculture, land development or for forestry and can result in the loss of ecosystem services (Kauffman et al., 2017; Lee et al., 2014; Rogers et al., 2016), including impacting sediment retention (Bulmer, 2017), nutrient cycling and carbon storage (Lang'at et al., 2014; Bulmer et al., 2015; Adame et al., 2018).

Despite extensive loss of mangrove habitat across the globe, surprisingly limited research has documented resulting site-specific changes to coastal environments (Lang'at et al., 2014). The interconnectedness between trees, pneumatophores and below ground root systems and hydro-, geochemical and sediment dynamics of the wider system mean that cascading effects of removal can be complex. For example, synthesis of elevation change across an evolving landscape of mangrove and marsh habitats in southern Florida, USA, demonstrated significant within- and between-site variability related to storm exposure, hydrologic connectivity, sediment supply, and local plant properties (Feher et al., 2022). Furthermore, coastal change may occur non-linearly in response to natural or human impacts. For example, storm surge and sediment deposition can lead to patchy damage (Lagomasino et al., 2021) and delayed tree mortality (Radabaugh et al.,

* Corresponding author.
E-mail address: debra.stokes@scu.edu.au (D.J. Stokes).

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2020). It is imperative to develop a stronger understanding of the ways muddy coastal fringe systems respond to changes in mangrove extent in order to make effective land management decisions as mangrove clearing continues (Bulmer, 2017).

Mangrove clearing and edge trimming has become common practice in Aotearoa New Zealand, in locations where the native *Avicennia marina* has vigorously colonised intertidal flats. Although mangroves are endemic to Aotearoa New Zealand, and their range likely began slowly expanding as estuaries developed after the last glaciation, the ten-fold increase in sedimentation associated with the arrival of Europeans has caused a well-documented expansion within their climatic range on the North Island (Lundquist et al., 2014a; Horstman et al., 2018; Gao et al., 2019; Gao et al., 2020; Swales et al., 2021). Here, mangroves are often perceived as having low ecosystem service value (Horstman et al., 2018; Dencer-Brown et al., 2018; Glover et al., 2022). This negative perception is largely associated with the accretion of muddy sediments that generally accompany mangrove expansion. Motivations for the removal of mangroves in Aotearoa New Zealand, listed by Lundquist et al. (2017) include:

- Improve access for recreation and amenity
- Shift muddy sediments to return habitats to firm sand flats
- Restoration of seagrass and shellfish beds (for both cultural and ecological values)
- Improve/restore access and navigation to the coast

In turn, both legal and illegal removal has taken place in the hope that the original sandy substrate will return, with the associated change in benthic ecology that characterises muddy environments (e.g., crabs) to the historical bivalve-dominated community, coupled with easier access for recreational activities (see Lundquist et al., 2014a and Stokes et al., 2016 for more detail).

Early studies on the impacts of mangrove removal in affected regions in Aotearoa New Zealand highlighted the importance of considering each estuary individually in terms of hydrodynamics and sediment loads, with slow and steady removal of vegetation key to achieving the ‘recovery’ to sandy conditions (Stokes, 2010). Despite this research, some clearing activity is still conducted rapidly, resulting, in some cases, in the removal of up to 100 ha in less than 12 months (Lundquist et al., 2012).

Most monitoring of mangrove removal in Aotearoa New Zealand has also been limited to a two to three-year post-clearance period (reviewed in Lundquist et al., 2014; Stokes et al., 2016), despite reports that suggest some aspects of ‘recovery’ may require more than a decade to be achieved. The most common parameters measured include temporal changes in surface sediment conditions (size, chlorophyll *a* (chl-a), organic matter content), benthic macro-epifauna and infauna (Stokes, 2010; Lundquist et al., 2012; Bulmer, 2017). Relatively few studies have included an assessment of buried biomass (though see Siple and Donahue 2013; Lundquist et al., 2014; Bulmer et al., 2017). Lundquist et al. (2014a) examined 40 removal sites in Aotearoa New Zealand and suggested that belowground biomass may persist for as long as 16 years after mangrove removal. The benefits of the carbon stores from mangrove roots, and the subsequent loss of carbon and increase in CO_2 on mangrove clearing is increasingly important in terms of climate change mitigation (Lovelock et al., 2011; Duarte et al., 2013; Arnaud et al., 2023). The extent of change to a system within 2–3 years is variable, depending on a range of factors including a) size of patch cleared; b) position of patch relative to channel tidal flow; c) freshwater flushing regime; d) sediment delivery; e) sediment profile and age of mangroves (i.e., depth of mud accumulation) and f) exposure to wind waves (Horstman et al., 2018).

The body of knowledge is growing as to the ecosystem services provided by mangroves in Aotearoa New Zealand, as well as the physical and ecological changes that occur when they are removed. There are still unknown effects of mangrove removal, however (Bulmer et al., 2017). Analysis of recovery trajectories are imperative to support informed coastal management practices, and to encourage a more balanced and realistic expectation of the perceived outcomes of mangrove clearing.

The aim of this study is to describe changes in the environmental setting of Waikaraka Estuary, Aotearoa New Zealand, a decade after mangrove clearing had ceased. Waikaraka Estuary is one of many sub-estuaries positioned along the western, landward, flanks of Tauranga Harbour where rapid mangrove expansion was observed. Early clearing activity was limited to hand-felling by community groups, followed by more extensive mechanical clearing in 2012 (see Section 2.1.1 for more detail). Subsequent short-term monitoring reported limited bivalve-habitat restoration (Stokes, 2010; Lundquist et al., 2014).

The key questions addressed here are motivated by the original community expectations and whether those expectations have been realised a decade after mangrove removal, being:

a) Have the cleared intertidal flats reverted to a sandy substrate and have bivalve populations increased?
b) How much root mass remains and what trajectory is predicted for complete root decomposition?

2. Methods

2.1. Study site

Waikaraka Estuary is a small embayment on the western flank of Tauranga Harbour, located on the east coast of Aotearoa New Zealand’s North Island (Fig. 1). The 200 km^2 harbour is bounded to the west by the north-south oriented Kaimai Range, resulting in steep and narrow catchments and little capacity for storing catchment-derived sediment load on the land. The estuary has a catchment of around 10 km^2 with two narrow creeks entering its landward (western) tidal flats. Mean

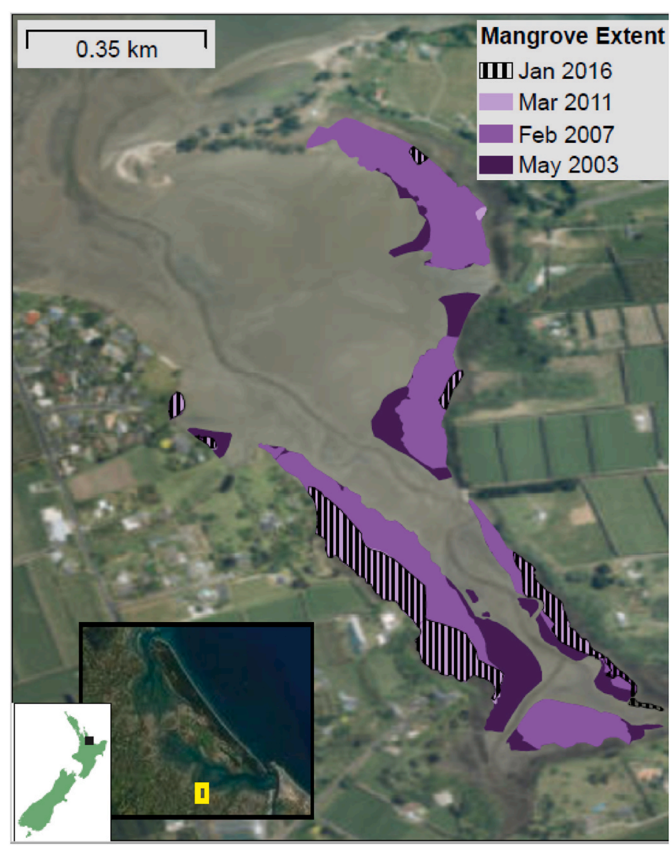

Fig. 1. Waikaraka Estuary and change in mangrove coverage between May 2003 and January 2016 (background image from Google Earth).

annual flow of these creeks has been reported at around 90 l s^{-1} (Hope, 2002). Tauranga annual rainfall averages 1190 mm, with a typical winter maximum. Episodic intense rainfall occurs with the movement of occasional thunderstorms. These, and rare extreme events can cause heavy flooding (Chappell, 2013). The native forest has been cleared from the catchment which is now a mix of residential housing, and kiwifruit and citrus orchards (Hope, 2002). Tides at the entrance of Waikaraka Estuary are meso-tidal with a 2.1 m spring high attenuating to around 0.6–0.7 m in the upper estuary (Hope, 2002).

2.1.1. Clearing activity and previous monitoring

In the early 1940s, mangroves were limited in extent, mapped mostly in a narrow band around the edges of the upper estuary. Rapid colonisation occurred between 1982 and the late 1990s when mangrove habitat had both widened in the upper estuary, and colonised tidal flats closer to the harbour entrance. By 1996, mangroves covered around 10 ha, compared to 1.6 ha in 1943. Further expansion up to 11.5 ha in 2003 was halted by incidental manual clearing of vegetation and removal of seedlings (Stokes, 2010).

A community group was granted a resource consent in 2004 to manually remove mangrove vegetation that had colonised the Waikaraka Estuary since 1994 (Fig. 1). The majority of clearing had taken place by 2012, with narrower strips of fringing mangrove cleared from the western stand after 2012.

Multiple studies have examined grain size and mud content across Waikaraka Estuary, in 2001 (Hope, 2002), 2007 (Parks, unpublished data), and 2006 and 2007 (Stokes, 2010). The community group was required to undertake basic monitoring of epifauna and surface elevation changes. Additional research focused on changing substrate elevation, surface grain size and changes to infaunal and epifaunal communities, although this was limited to a two-year period from 2005 to 2007 (Stokes, 2010). The methods in the present study are consistent with these prior studies to facilitate comparisons.

2.2. Measuring sediment characteristics

In June 2019, surface sediment grabs were collected at randomised intervals along transects across the estuary (see Fig. 2 for sampling locations). Transects were designed to capture the key geomorphic features and habitats of the estuary (noting that the estuary is small). Transect 1 started at the uppermost section of the estuary, with Transect 6 positioned toward the estuary mouth. Transects 6 and 5 crossed the lower estuary (west to east), Transects 2 to 4 crossed the narrower mid-estuary (west to east), and Transect 1 the uppermost zone south of the dissecting tidal channels (south to north). Samples were collected across existing mangroves, cleared tidal flats and undisturbed tidal flats.

Samples were repeatedly treated with 10% hydrogen peroxide over 4–5 days to remove organic material. Calgon was added as a deflocculant before analysis for grain size using the Malvern Mastersizer 3000. A sub-sample of each surface grab was taken to measure sediment total organic content (TOC) by Loss on Ignition after 5.5 h in a muffle furnace at 550 °C (Dean, 1974). Additional sediment cores were collected in the cleared habitat (Fig. 2) using a 5 cm diameter AMS split corer to analyse grain size, mud content and TOC at depths 0–1 cm, 1–2 cm, 4–5 cm, 9–10 cm and 15–16 cm.

Early sampling by other investigators (Hope, 2002; Parks 2007) was not designed to explore impacts of mangrove removal. As such, separate statistical analyses were undertaken to investigate differences in sediment characteristics between Hope and Parks sampling, and the author's subsequent sampling. Results from sampling undertaken by Hope (2002) and Parks (2007) were compared using a two-way PERMANOVA (PRIMER version 7) to assess change over time, and habitat. Similarly, a two-way PERMANOVA was used to test for changes through time between 2006, 2007 and 2019 (data collected by Stokes), specifically comparing cleared and mangrove habitat. Fixed factors were time (year) and habitat (mangroves, cleared, sandflat/mudflat). Post-hoc tests were used to examine differences in mean grain size and % mud if significant (p_{perm}<0.05) differences were identified. A one-way PERMANOVA was used to assess depth-wise changes to grain size, % mud and TOC in the short cores collected in cleared habitat in 2019.

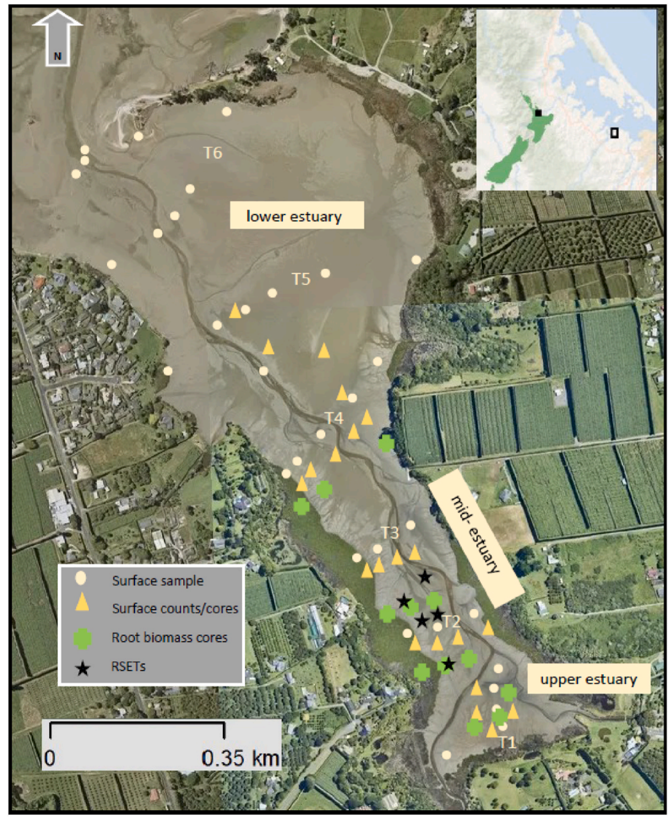

Fig. 2. Location of sample collection for surface sediment samples, epifauna (surface) and infaunal macroinvertebrate counts, root biomass cores and Rod Surface Elevation Tables (RSETs).

2.3. Root biomass

A series of triplicate sediment cores were collected from areas where mangroves had been cleared in the 2003–2005 period, and also from areas that were cleared after 2005. A split corer with a 5 cm diameter was used to collect samples to between 25 and 30 cm depth (and total core length noted). Sediment was put through a 1 mm sieve on site to collect mangrove root material. Structural roots (>2 mm diameter) and fine roots (<2 mm diameter) were thoroughly washed in fresh water, bagged and dried at 60 °C for 24 h. Structural roots and fine roots were measured separately and a dry weight volume calculated and reported in kg/m^2.

2.4. Mangrove characteristics

The mangroves of the study site are mostly shrub and dwarf trees of the single species *Avicennia marina* var. *Australasica*. Within the remaining continuous stand of mangroves, two quadrats were marked. Every tree within each 2 × 4 m quadrat was measured, recording tree stem diameter at 5 cm above substrate as the typical measurement at breast height (DBH) is not suitable for trees that are both <3 m tall and branch close to the substrate (Woodroffe, 1985; Stokes, 2010; Bulmer et al., 2016b; Owers et al., 2018), tree height, and canopy area (Kauffman and Donato, 2012). Stem density and average tree height were calculated.

2.5. Surface elevation

A series of Rod Surface Elevation Tables, or RSETs (Cahoon et al., 2002) and erosion pins were installed in 2006, following partial mangrove clearing in 2005. RSETs have two components; the first is a permanent 'base' that sits above the substrate with 10–15 m of stainless-steel rod buried vertically into the substrate to maintain stability. A coupling device allows for the installation of the RSET 'arm' on each field visit. The arm is levelled on the vertical and horizontal and 9 numbered pins are inserted through holes in the arm, with the base of each pin connecting with the surface of the substrate. The height of each pin that sits above the arm is recorded for each of four positions around the base (i.e. north, south, east and west). Over time, these averaged pin values provide a measurement of temporal surface elevation change (see Cahoon et al., 2002 for detail of the device design). Three transects of RSETs were positioned across the mid-estuary, with two RSETs of each transect stationed inside existing mangroves and two positioned on the tidal flats, with the most north-easterly of these positioned in (what was then) recently cleared substrate (see Stokes et al., 2009 for more detail).

Of the twelve original RSETs, five were still in place and functional in 2019. The others had either been damaged during mechanical removal of mangroves (from tractors) or could not be found in the existing mangrove fringe. Only one RSET was functional in Transect 1. T1-3 station reflects a site that was vegetated in 2007 and subsequently

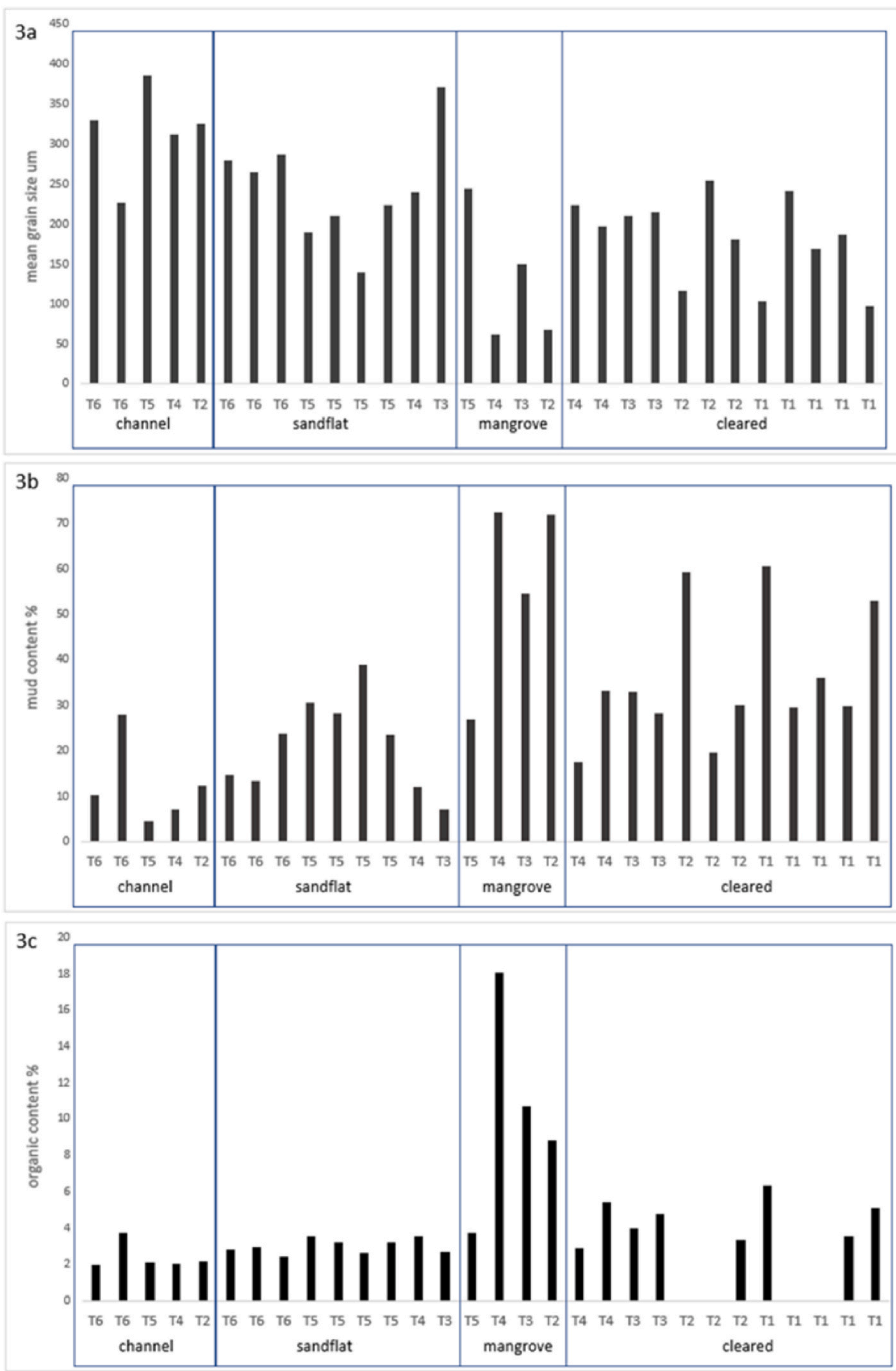

Fig. 3. % Total organic content (3a) mean grain size (3b) and % mud (3c) from single surface samples collected across six transects in Waikaraka Estuary (2019), lower estuary (T5, T6) mid-estuary (T2, T3, T4) and the upper estuary (T1).

cleared, T3-2 within mangrove vegetation and T2-3, T2-4 and T3-4 are all positioned on what has been bare sandflat since installation. RSET No. 1 (Fig. 2) could not be measured in 2019 due to a build-up of sediment in the receiver.

2.6. Macrofauna

Surface counts of epifauna and crab burrows were undertaken at random locations across the estuary (Fig. 2). All macroinvertebrates and crab burrows were counted within triplicate 0.25 m^2 quadrats. In addition, sediment within each quadrat was excavated using a benthic corer (15 cm diameter to 20 cm depth) to assess the presence/absence of bivalves. Any bivalves that were found after sieving the cores through a 1 mm mesh, were measured (length) and species identification confirmed using WoRMS (World Register of Marine Species).

3. Results

Sediment characteristics.

Surface sediment characteristics show spatial variability (Fig. 3a), with general coarsening of sediments toward the lower estuary (Transect 5) where mean grain size at the mouth of the estuary measured between 350 and 538 μm, and channel sediments 300–400 μm. In contrast, the unconsolidated fine sediments in the cleared flats of the upper estuary (Transect 1) averaged 160 μm (range of 97–240 μm).

Mangrove sediments had >50% mud, while mud content of cleared substrate in the mid-estuary ranged from 20 to 40%. Mud content in the cleared substrate of the upper estuary was high, between 30 and 61%. The cleared substrate east of the channel mid-estuary was slightly less muddy, at 20–33%, while the broad intertidal flats of the lower estuary ranged in mud content from 5% in the channel to 30% toward the outer reaches (Fig. 3b).

Total organic content of surface sediments was <4% across the lower estuary. Sediment collected within the fringing mangroves was higher at between 9 and 18% whereas sediment TOC of intertidal samples of the mid and upper estuary ranged between 3.5 and 11% (Fig. 3c).

Sediment characteristics of mangrove sediments sampled in 2001, 2006, 2007 and 2019, and cleared sites sampled in 2005, 2007 and 2019 were compared (Fig. 4). Mangrove sediments sampled in 2001 were coarser than the surface sediment grain size reported for mangroves from 2006 to 2019, suggesting some fining over time. Mangrove and cleared surface sediments were similar in grain size and mud content in 2005 and 2007, with a notable increased mean grain size in 2019 for cleared sites. Two-way PERMANOVA analysis of monitoring data collected in 2006, 2007 and 2019 identified some statistical support for significant differences in sedimentology by both habitat (cleared, mangrove, sandflat), ($p_{perm} = 0.08$) and time ($p_{perm} = 0.03$). The habitat × time interaction was not significant ($p_{perm} > 0.02$) Post-hoc tests found differences in sediment characteristics between cleared and mangrove habitats in the mid-estuary, while the upper estuary was significantly different to all other habitats due to its higher mud content and fine sediment size.

Results from a two-way PERMANOVA showed a significant effect of habitat (mangrove, sandflat) but not time when comparing mud content of samples collected by Hope (2002) and Parks (2007). There was no 'cleared' habitat as part of their sampling regime, with each habitat of sandflat and mangrove significantly different ($p < 0.01$).

3.1. Short core analysis – cleared intertidal flats

All cores were found to have a larger grain size at the surface compared to sediments from 2 cm down to 10 cm, below which a subtle coarsening occurred between 10 and 15 cm. Similarly, mud content was lower at the surface, increasing to between 20 and 40% at depths below 2 cm. Variability in TOC with depth was minimal, with average TOC between 4 and 5% at all depth intervals (Fig. 5). A one-way PERMANOVA test, however, found no significant differences ($p_{perm} = 0.36$) in measured parameters with depth.

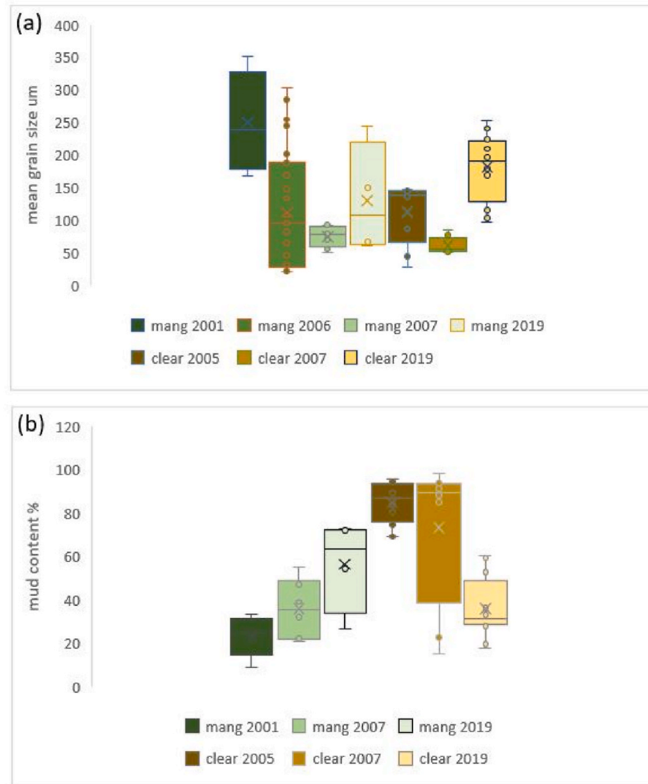

Fig. 4. Grain size (a) and mud content (b) of mangrove (2001 [Hope 2002], 2007 and 2019) and cleared zones, (2005, 2007 and 2019). Note that mangrove sediment mud content wasn't measured in 2005.

3.2. Mangrove characteristics and root biomass

The remaining uncleared mangroves measured between 1.1 m (± SE 0.02) and 1.3 m (±0.07) in height, with stem diameter averaging 32 cm within both quadrats (±3.9 and 4.5). Tree density varied between 1 and 1.75 trees/m^2.

Mangrove root biomass measured between ~9 and 13 kg m^2, with structural roots and fine roots contributing similar amounts to the total weights of root material (Fig. 6). A clear spatial trend in sub-surface biomass was evident along each transect. There was higher root biomass in existing mangrove habitat, with reduced biomass in cleared areas. Some temporal trend in decomposition was apparent, with samples collected closest to mangroves (and therefore cleared later) measuring higher biomass relative to material found in cores collected further from the mangrove fringe. Root biomass remaining in cleared areas was much lower, with total biomass of <1.5 kg m^2 in the upper reaches, or 'head' of the estuary (south of the tidal channel), increasing to up to 4 kg m^2 where mangroves were cleared mid-estuary (Transects 2 to 4).

3.3. Surface elevation changes

Five of the twelve RSETs in the mid-estuary were still in place and functioning in 2019. T1-3 station reflects a site that was cleared in August 2005, T3-2 was originally within mangrove vegetation, while T2-3, T2-4 and T3-4 are all positioned on what has been bare sandflat since installation (Fig. 7).

At RSET T1-3 a fall in elevation of 31 mm occurred between May 2006 to March 2007, after which sedimentation reduced the change in

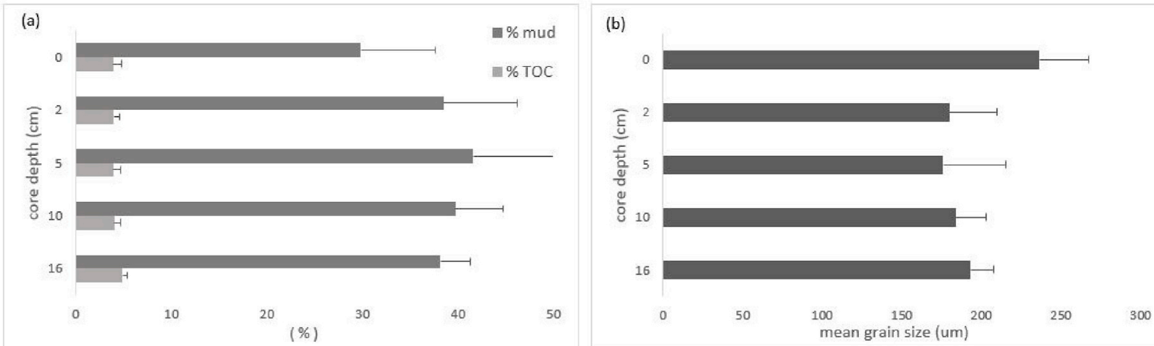

Fig. 5. Sediment characteristics: (a)mud content and organic content and (b) mean grian size at depth intervals to 16 cm from short cores collected from intertidal substrate cleared of mangroves (n = 6).

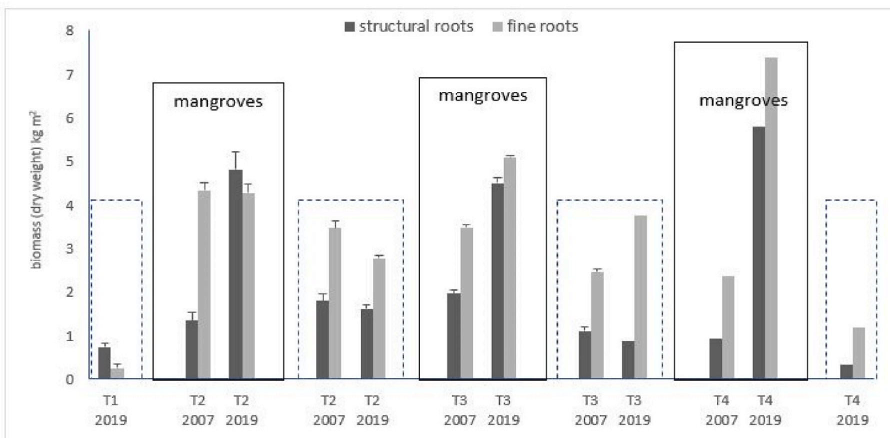

Fig. 6. Mangrove root biomass (dry weight) separated in to structural roots and fine roots (+se), from four transects covering existing mangroves (in solid boxes) and cleared substrate (dashed boxes) (n = 3).

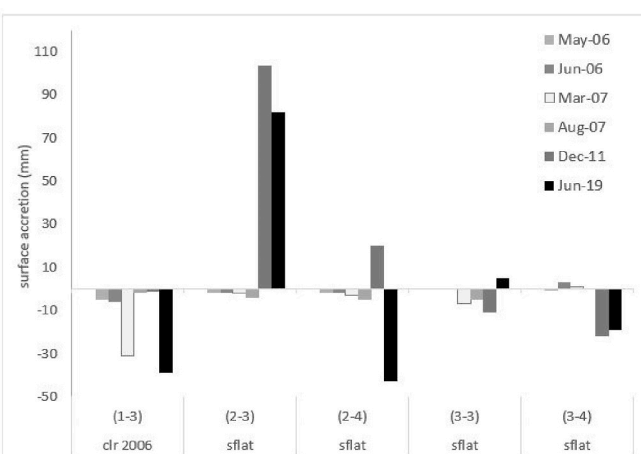

Fig. 7. Surface elevation change between March 2006 (when all stations recorded 0 change) and June 2019 from RSETs. Labels are consistent with original RSET transect numbering in 2006 (Stokes, 2010).

elevation to only – 2 mm in December 2011. Between 2011 and June 2019, surface elevation continued to fall, resulting in an overall change of – 39 mm.

T2 sandflats experienced very little change in elevation between 2006 and 2007, < −5 mm. T2-3 measured an elevation change of 103 cm in December 2011 then a lower level of sedimentation of 82 mm in 2019. It is clear from photos of the RSET bases in 2006 and 2019 that sediment accretion has occurred because most of the base has been covered. However, site T2-4 shows the reverse trend, with an overall drop in surface elevation of 43 mm between March 2006 and June 2019.

Elevation changes were less marked for the sandflat locations at Transect 3. T3-3, originally 10 m seaward of the mangrove fringe in 2005, measured no change until March 2007, when −7 mm was measured. In December 2011, elevation change was −11 mm. A small increase in elevation of 5 mm was measured in June 2019. T3-4 varied less than 2 mm between May 2006 and August 2007, after which erosion of −22 mm was recorded in 2011, and -19 mm in 2019.

3.4. Macroinvertebrates

The spatial distribution of key epi and in-fauna (to 20 cm depth) was variable, with the cleared area at the head of the estuary (Site 1, C1 in Table 1) devoid of gastropods, crabs and bivalves. Abundance of epifauna sampled across the mid-estuary was limited to <2 individuals per 0.25 m^2 quadrat for key gastropod species of *Zeacumantus*, and <0.5 individuals for *Amphibola* and *Cominella* (Fig. 8). Occasional bivalves were encountered, with cockles (*Austrovenus)* and *Macomona* of, on average, <0.5 individuals per quadrat. Bivalves were mostly limited to the lower estuary with *Macomona* averaging 2.4 per quadrat and average length of 2.4 cm (±0.4). The cockle *Austrovenus* averaged 0.4 individuals per quadrat, with lengths of 1 cm–2 cm (average = 1.3 cm ± SE 0.3) (Fig. 9).

The MDS ordination shows a separation between sandflat and mangrove macrofaunal communities, whereas cleared sites appear to be

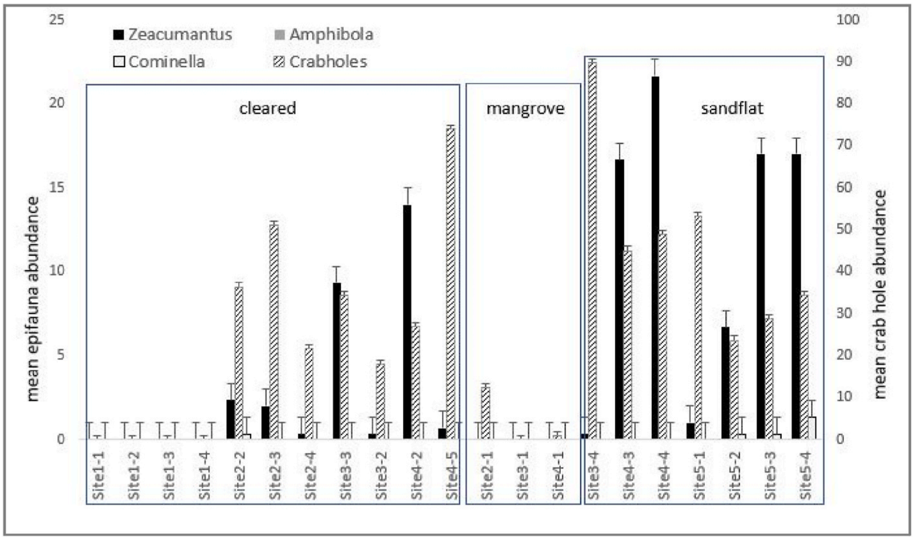

Fig. 8. Mean abundance per 0.25 m^2 quadrat (±SE) of surface macrofauna species at each site sampled in Waikaraka Estuary (2019 (n = 5). Site/Transect 1 is the most landward site, and Site/Transect 5 closest to the mouth of the estuary.

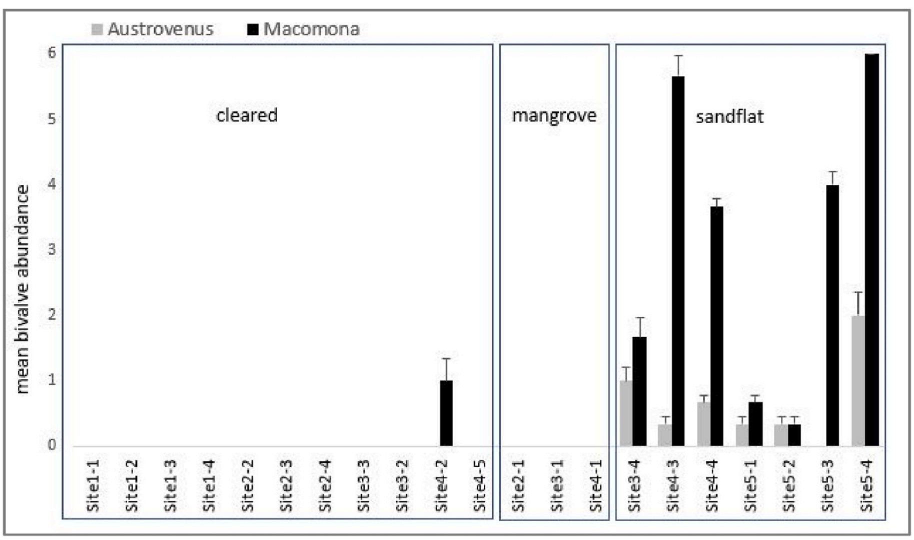

Fig. 9. Mean abundance per 0.25 m^2 quadrat (±SE) of bivalve species at each site sampled in Waikaraka Estuary (2019 (n = 5). Site/Transect 1 is the most landward site, and Site/Transect 5 closest to the mouth of the estuary.

more variable in community structure (Fig. 10). The site was found to be a significant factor in determining macrobenthic community structure (Table 1) from PERMANOVA analysis ($p_{perm} < 0.001$). Pairwise comparisons found the greater differences to be associated with both location and habitat, with cleared sites in the upper estuary significantly different to cleared habitat in the lower estuary while differences in community structure of mangrove and cleared habitat was not significant. The greatest site differences were between the upper cleared site (C1) and the sandflat locations (12–16; p_{perm} 0.001 and 0.004).

4. Discussion

4.1. Geomorphology and bed characteristics

The objective of this study was to assess the extent of change to the geomorphology and ecology of Waikaraka Estuary throughout the decade after mangrove clearing. Since 2003, 8 of the total 11.5 ha of mangroves have been cleared from Waikaraka, with roughly 90% of that cleared by 2012. The removal of above-ground mangrove vegetation was undertaken with the hope that a) the muddy sediment associated with the mangroves would be flushed out of the estuary; and b) a more diverse, bivalve dominated macroinvertebrate community would establish. Though the estuary has had close to a decade to adjust or respond to that clearing, much of the estuary has not yet changed to a state that would be considered 'restored'.

Muddy sediments continue to dominate the upper estuary. The head of the estuary, south/landward of the drainage channels, maintains surface sediments with 30–50% mud which persisted to sampled depths of more than 10 cm. Relatively little mangrove root biomass was recovered at this site, measuring less than 1.5 kg m^2, where the sediment was watery and unconsolidated, with a notable absence of macrofauna and bivalves. Glover et al. (2022) reported water velocities consistently <25 cm/s in this region and minimal net sediment transport. The low tidal velocities and minimal freshwater flows suggest that mud will not be flushed out of this zone in the future. Further, extended periods of exposure during mid to low tide will likely increase erosion thresholds (Nguyen et al., 2019).

The mid-estuary has undergone some bed texture and elevation

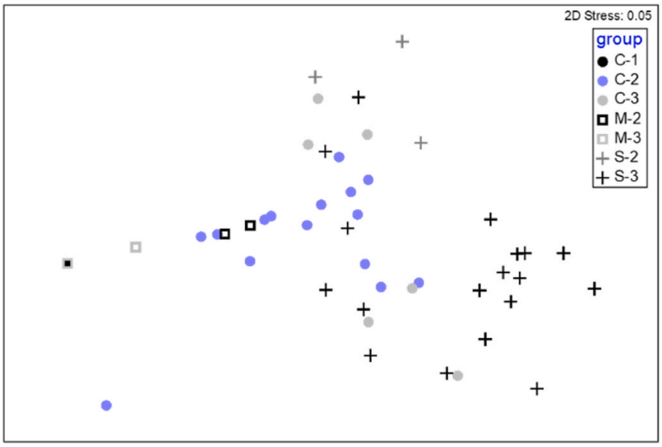

Fig. 10. Multidimensional scaling (MDS) ordination of macrofauna data from cleared sites (circles; C1 at Site 1; C2 at Site 2; C3 at Site 4); mangrove sites (squares; M2 at Site 2; M3 at Site 3); and sandflat sites (crosses; S2 at Site 3 and S3 at Site 4).

Table 1
Summary of Results of pairwise tests from PERMANOVA

Groups	t	P(perm)	unique perms
C-1, C-2	**7.7469**	**0.001**	**978**
C-1, C-3	**8.5197**	**0.001**	**58**
C-1, M-2	2.9183	0.032	8
C-1, M-3	2.2804	0.186	2
C-1, S-2	**16.333**	**0.004**	**8**
C-1, S-3	**12.061**	**0.001**	**994**
C-2, C-3	1.1973	0.245	989
C-2, M-2	3.5824	0.001	969
C-2, M-3	3.4108	0.004	510
C-2, S-2	2.6734	0.003	570
C-2, S-3	**4.3201**	**0.001**	**998**
C-3, M-2	3.8358	0.004	202
C-3, M-3	3.6187	0.013	63
C-3, S-2	1.8407	0.09	84
C-3, S-3	2.0654	0.024	994
M-2, M-3	0.86173	0.587	11
M-2, S-2	**4.94**	**0.015**	**42**
M-2, S-3	**6.5345**	**0.001**	**994**
M-3, S-2	6.1055	0.11	7
M-3, S-3	**5.5585**	**0.002**	**649**
S-2, S-3	2.9153	0.006	709

changes. The contiguous stand of mangroves on the western flank of the estuary was gradually and systematically cleared, commencing in the early 2000s. Surface sediments here measured more than 80% mud within months of clearing in 2005, reducing to around 60% mud in 2007. Similar cleared habitat fronting the remaining mangrove stand continued to show mud content of 30–50% in 2019, with a coarsening of surface sediments toward the tidal channel. The intertidal flats closer to the channel showed the lowest mud content, suggesting some winnowing associated with tidal flow. Channels likely provide a major conduit for sediment transport out of the estuary, and restoration of sandy conditions could be promoted by modulating tidal dynamics in these conduits (Glover et al., 2022).

Interestingly, short core stratigraphy from the mid-estuary revealed a coarser surface sediment (although not significantly different), with a higher mud content found at depths to 16 cm. Stokes and Harris (2015), described the development of 'sand caps' in another estuary cleared of mangroves. This sand cap can inhibit further erosion of subsurface mud by providing a barrier to resuspension. The hydrodynamic and ecological conditions in Waikaraka Estuary do not provide a mechanism for the breakdown of this sand cap, to liberate the muddier sediments underneath. The bed shear stresses observed and modelled in the mid-estuary were consistently 0.7 Pa which is unlikely to rework the sand cap (Glover et al., 2022). Waves can rework the bed to greater depths than tidal action. Unfortunately, the potential for wind waves in Waikaraka Estuary is limited due to the geometry and limited fetch of the system. In more energetic environments, major storm events have been shown to drive morphologic regime shifts (Osland et al., 2020). The role of bioturbating invertebrates in redistributing and reworking sediment is also limited at this site, with few to no bivalves inhabiting this mid-estuary intertidal area. The formation of these resistant surface layers should be considered when estimating recovery time frames for estuaries (Friedrichs, 2011).

The broad geomorphology of the system also appears to have changed little in response to mangrove clearing. Field sampling and hydrodynamic modelling results suggest Waikaraka is an estuary close to a dynamic geomorphic equilibrium, with minimal net tidal sediment transport (Glover et al., 2022). The lower flats appear to have a higher trapping capacity toward the landward edges, which is characteristic of estuarine sediment dynamics (Friedrichs, 2011). The upper estuary experiences tidal patterns that are unlikely to flush the muddy sediment seaward but instead will typically simply redistribute the mud within the upper estuary. Rare storm events and floods are therefore vital to the longer-term flushing of muddy sediment in the upper estuary.

Given that a decade has passed after clearing, it is possible that a return to a consolidated substrate with reduced mud content may not happen, at least within the next 10–20 years. Monitoring of other sub-estuaries in the greater Tauranga Harbour has reported similar sediment retention following mangrove removal at locations where the geometry limits turbulence associated with wind waves and/or greater distance from tidal channels (Stokes et al., 2016). The rate of sediment coarsening has also been linked to the size of the area cleared, with smaller areas showing a more rapid change in characteristics over time (Bulmer et al., 2017). Mangrove removal is more likely to result in significant and rapid change when sites are consistently exposed to wind waves and/or high velocity tidal flows (Winterwerp et al., 2013; Swales et al., 2015; Besset et al., 2019; Brunier et al., 2019). However, changes to grain size following mangrove removal or mortality are fundamentally related to the characteristics of supplied sediment. Sediment mineralogy was not monitored here, but could play an additional role in sediment retention, for example, in limestone rich sites. In regions dominated by mud import, mangrove removal can simply result in a permanent shift to bare mudflats (Osland et al., 2020).

Overall, Waikaraka Estuary demonstrates the slow timeline of recovery following mangrove removal in quiescent environments. The importance of storm/flood reworking also suggests that recovery can be strongly non-linear over time. In systems exposed to storm disturbance, sediment reworking and removal will likely occur in a stepwise manner associated with energetic events (Lagomasino et al., 2021). The possibility of rapid evolution following an event also supports the need for better long-term monitoring of mangrove removal sites.

4.2. Ecology

Mangroves are known to invest much of their biomass to their root system, in response to anoxic and saline growing conditions (Komiyama et al., 2008). In Waikaraka Estuary, mangrove belowground (root) biomass has continued to increase in remaining mangrove stands. Mangrove root biomass measurements reported in 2005 (Stokes, 2010) showed a site average mass of between 2.5 and 3.5 kg m^2 compared to the 2019 average of 10.6 kg m^2 (±1 SE 3.5). It appears that the temperate, growth limited mangroves here have concentrated growth to belowground fine and structural root development. In 2005, the mangroves were mostly <30 years old, suggesting that another 14 years of growth has resulted in this increased load, with a higher percentage (30–60%, average 45%) of the total weight associated with structural roots in 2019. The tree height has changed little, averaging 1.18 m (±0.36) in 2019 which is similar to average tree heights in the same stretch of mangroves in 2005 (Stokes, 2010). Although detailed studies of biomass allocation are few, general estimates suggest belowground root material can contribute up to 60% of the total biomass (Adame et al., 2017), with recent studies suggesting younger trees produce relatively higher belowground biomass (Arnaud et al., 2021). Here, in a temperate setting, the root:shoot ratio may be even higher (Bulmer et al., 2016). This growth distribution should be investigated further in terms of carbon sequestration services.

Mangrove root material has also persisted in cleared areas despite more than 10 years of decomposition time. Root material collected from cleared areas of the estuary show some spatial trend, with persisting root material of <2 kg in the uppermost part of the system, most of which was structural roots, suggesting the fine roots have decomposed or reworked to greater depths. There was also a decrease in biomass across transects from the existing mangrove fringe toward the tidal channel in the mid-estuary. There was 2–4 kg m^2 of biomass toward the fringe mangroves, representing the more recently cleared habitat (~7–8 years), while the intertidal flats closest to the channel measured generally <1.5 kg m^2. The lower values closest to the tidal channel could be the result of increased oxygenation and interstitial water/nutrient flow associated with both longer inundation periods and higher flow velocities, while the cleared areas more landward are only inundated by slow and shallow tidal flows for the highest part of the tide. Other studies of mangrove roots have concluded that site characteristics, climate and tree species will all influence both the density of root material and the rate at which it will degrade (Saintilan, 1997; Middleton and McKee 2001; Komiyama et al., 2008; Lovelock, 2008; Ola and Lovelock, 2021).

It is difficult to establish a confident estimation of decomposition rate because of the spatial and temporal variability of mangrove biomass and the rate of mangrove clearing activity. However, this decadal monitoring demonstrates that at least two decades is required for a near complete decomposition of belowground biomass. Sampling at other Aotearoa New Zealand sites found no significant difference in root biomass between established mangroves and cleared habitat up to 2 years after mechanical clearing activity (Lundquist et al., 2014), while Gladstone-Gallagher et al. (2014) estimated mangrove root material in a NZ temperate setting would decay to 50% its mass between 317 and 613 days. Approximate calculations from this field data suggest decomposition to 50% was not reached within the decadal time-scale, further supporting the notion that complete decay of woody material will take decades (Siple and Donahue, 2013; Lundquist et al., 2014).

The rate of root decomposition has significant implications for morphology and estuary recovery. In existing mangrove stands, dense root mats help to 'hold' sediments in place. Roskoden et al. (2020) suggest mangrove roots initiate sediment dewatering and, combined with the physical root structures themselves, increases consolidation and erosion thresholds. As root material decays after either the removal of above-ground structures or the death of trees from lightning or hurricanes (for example), a loss of root turgidity can result in soupier, less consolidated muddy sediment. In carbonate environments, such as the Everglades (USA), mud tends to persist while the peat decays and the substrate sinks (Parkison and Wdowinksi, 2022; Feher et al., 2022). This sinking response is also occurring at Waikaraka (see Fig. 7). Where flow is sufficient to flush surface fines, this sinking of the substrate may be increasing relative sea level. It appears that both scenarios are currently present at Waikaraka Estuary. The uppermost cleared zone showed deep watery muds. This zone is inundated daily but with low-velocity flows, tidal heights of around 1.2 m, and an average inundation period of ~160 min. Groundwater could be another contributor, keeping the water content high.

RSET measurements suggest that the early trend of rapid falling surface elevation was not maintained over the 13 years following mangrove clearing, with some sediment accumulation and increase in surface elevation at sandflat sites adjacent to cleared habitat. Repeated RSET measurements in the first year following mangrove removal revealed falling surface elevation of between 9 mm and 38 mm per year, compared to a rising surface elevation in existing mangroves up to 14 mm year. Although some studies have addressed the changes to elevation associated with mangrove clearing, most report change within a period of only two to three years (Lang'at et al., 2014; Hayden and Granek, 2015). Similar initial rapid subsidence seen elsewhere has been ascribed to the collapse and decomposition of dying roots coupled with sediment compaction (Cahoon et al., 2003; Lang'at et al., 2014; Osland et al., 2020). It is difficult to draw confident conclusions on the overall changes in surface elevation across the entire estuary from this small dataset. However, this potential change to relative sea level following mangrove clearing is mostly overlooked in the development of monitoring protocols. It can be both challenging and time intensive to evaluate the elevation at mm-scale accuracy across an entire estuary. Yet, this is an important parameter to consider, particularly if a site has fringing saltmarsh that could be negatively impacted from the amplified effect from falling surface elevation coupled with regional sea level rise. This data from Waikaraka Estuary emphasises that changes to surface elevation can be non-linear over decadal scales, consistent with similar results from other regions of the world (Osland et al., 2020; Feher et al., 2022). A range of external factors can influence short-term changes to surface elevation, such as groundwater availability (Rogers et al., 2005; Whelan et al., 2005), drought (Rogers et al., 2006), cyclones and hurricanes (Osland et al., 2020) nutrient availability (Reef et al., 2017) and associated vegetation health (i.e. root growth variability with season/condition). Inter-annual monitoring of surface elevation can detect these perturbations, with annual monitoring a more ideal approach for long-term trajectories of change (Rogers and Saintilan, 2021). Our results are a snapshot of change that can only compare across a decade, without the nuanced seasonal or even yearly incremental changes.

Mangrove removal in Aotearoa New Zealand is often undertaken with some expectation that bivalve communities will recolonise the cleared intertidal areas. Results show spatially variable benthic macroinvertebrate community structure, with two key zones. Thrush et al. (2003) and Anderson (2008) have modelled the relationship between sediment conditions and likelihood of presence of common estuarine macroinvertebrate species in Aotearoa New Zealand. Optimal mud content for common bivalve species such as *Paphies* (pipi) is low, at 3.4%, compared to *Austrovenus* (11% and *Macomona* 16.6%). This mud limitation is consistent with the findings in Waikaraka Estuary. The lower tidal flats at Waikaraka were home to both *Macomona* and *Austrovenus*, where surface sediments contained between 5% mud close to the channel, fining to up to 39% at the outer edges.

The loose, muddy sediments remaining in the cleared intertidal flats at the head of the estuary (Site 1/C1) were devoid of gastropods with no evidence of crab burrows or bivalves. Upper estuarine environments are typically dominated by a mud-tolerant suite of benthic fauna associated with both a higher mud load, limited tidal flushing and higher exposure time during low tides. Other organisms could be present, such as smaller oligochaetes and polychaetes that are suited to a muddy environment (Thrush et al., 2003). Bivalve species were absent in benthic samples from the mid and upper estuary.

Although bivalves were not found across the cleared areas, there was a higher diversity and abundance of epifaunal species compared to neighbouring mangrove habitat. Similar to findings in 2007 (Stokes, 2010), mangrove sites had sparse gastropod abundance in 2019, while cleared sites had a higher abundance of common gastropods such as *Amphibola* and *Zeacumantus*. Seasonal influences of water temperature, light intensity and substrate temperature (Granek and Ruttenberg, 2008) would likely result in some differences to abundance of species, however the broad comparison between mangrove and cleared habitat in this study indicates some ecological changes associated with the clearing and coarsening of surface sediments. Bulmer et al. (2017) reported an initial increase in benthic abundance and diversity in response to the disturbance of removal, followed by smaller changes through the first 36 months following removal. The magnitude of this trend is dependent upon site conditions as well as the method of clearing and whether the detritus is removed from site (Horstman et al., 2018).

5. Conclusion

There are few studies of decadal-scale change following mangrove removal and little guidance for managers on the timescale of monitoring required to assess the future trajectory of an ecosystem following removal. The aim of this study was to describe the geomorphic and ecological changes in Waikaraka Estuary, NZ over the decade following mangrove removal. Waikaraka Estuary is representative of many small embayments in Aotearoa New Zealand, where mangrove removal is undertaken with the hope of restoring sandy habitat for bivalves, and is representative of many vulnerable coastlines, where mangrove removal is primarily motivated by coastal infrastructure expansion.

The limited river inflow combined with the estuary geometry and slow tidal flows have limited changes to the bed. A historically cleared catchment and associated increased sediment loads enhanced sediment infilling, which led to a broader intertidal area where mangroves could easily establish. These same characteristics have meant that this site has had limited 'success' in terms of a return to a sandy intertidal habitat after mangrove removal. There has been limited flushing of mud and bivalve communities have not re-established in much of the estuary. Additionally, mangrove-root decomposition has been slow, further slowing changes to the estuary morphology despite approximately ten years of time for adjustment. Future monitoring of Waikaraka Estuary would provide further refinement on these rates change, however, the >10 year study period has provided a clear picture of the ecosystem trajectory.

Management interventions to halt or remove mangrove encroachment should consider site-specific characteristics. The most important parameters for managers to evaluate are:

1. Is there sediment delivery to the system and what is the composition of imported sediment? A site is unlikely to become sandy if high rates of fine sediment import will continue following mangrove removal.
2. Is the site exposed to energetic wave or storm events? Quiescent environments are less likely to change rapidly following mangrove removal.
3. Is root material going to be removed and if not, do mangrove roots decay rapidly in the region of interest? Mangrove root decay is a critical factor in morphologic and ecosystem evolution following above-ground tree removal.

Moreover, forecasting to the future is important to overlay potential changes associated with increased sea level or further sediment delivery due to land use. Without site-specific evaluation, mangrove removal is unlikely to result in expected outcomes, even over decadal timescales.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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