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Addressing the mismatch between restoration objectives and monitoring needs to support mangrove management

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1. Introduction

Estuaries bridge our terrestrial and marine environments; they are unique and diverse ecosystems that provide invaluable ecosystem services (Barbier et al., 2011). Saltmarsh, mangrove and seagrass vegetation are critical vegetated habitats in estuaries, providing food and protection for numerous estuarine species, including fish, bivalves, crustaceans and birds (Thrush et al., 2013). Unfortunately, estuaries are also sites where impacts from human activity are often amplified, deeming them some of the most heavily used and threatened systems on the planet (Lotze et al., 2006, Halpern et al., 2008) planet. For example, the huge growth in aquaculture has resulted in an enormous loss of coastal/wetland vegetation, with shrimp farming contributing to some ~ 38 % of global mangrove loss and other aquaculture has contributed to a further 14 % loss (Ellison, 2008). Other direct impacts may be driven by land reclamation, dredging, port development and coastal urbanisation, while pollutants such as nutrients, metals and plastics from the surrounding catchment can accumulate within estuaries and harbours (Kennish, 2002). Such impacts can trigger dieback or profound shifts in the composition and health of coastal vegetation and other estuarine habitats (Thrush et al., 2003, Saintilan et al., 2014, Doughty et al., 2015). Degradation of coastal habitats can be addressed by removing threatening processes, however often restoration and rehabilitation is required to support ecosystem recovery (Ferrier and Jenkins, 2010).

Wetland restoration is a common management strategy applied around the globe, although most of this work is taking place in developed countries (see Bayraktarov et al., 2016 for review). A key objective is to restore ecosystem services and halt further loss of vegetation (Zhao et al., 2016). A recent review of 235 coastal restoration projects analysed the successes and costs, all of which involved some form of replanting or reseeding for restoration of coral reefs, seagrass, mangroves, saltmarshes or oyster reefs (Bayraktarov et al., 2016). In temperate regions of north America, replanting of saltmarsh has been central to the success of various restoration projects (Zedler, et al., 2012). Elsewhere, mangrove rehabilitation programs have been implemented to restore forest cover and habitat functionality (Milbrandt et al., 2015, Osland et al., 2012). Rarely has removal of native vegetation been considered a restoration technique, however in recent times resource management agencies in New Zealand have removed mangroves in an attempt to restore tidal flat ecosystems (Harty,2009; Morrisey et al., 2010; Lundquist et al., 2014).

Despite a substantial global decline in mangrove distribution (Giri et al., 2011), many temperate mangrove forests are increasing in distribution (Morrisey et al., 2010). In New Zealand, this typically occurs where sediment loads are high and mangroves colonise seaward across bare mudflats (Lundquist et al., 2014, Swales et al., 2015, Stokes et al., 2010). In southern Australia drought and limited sediment loads (leading to compaction) are suggested causes of landward colonisation by mangroves (Rogers et al., 2005; Saintilan and Rogers, 2013).

When discussing the various approaches to managing mangroves out of coastal and estuarine habitats, Elliot et al. (2007) suggest that for recovery to be truly successful, the community established has to be similar in species composition, population density and size and biomass structure to that which was present prior to mangroves, or similar to that described at a site where mangroves are not present. Understanding how a site responds to the eradication of some or all of its mangroves is vital if we are to develop and implement effective habitat management strategies. There is limited reporting on the impacts of mangrove removal activities in temperate mangrove systems, in both academic and grey literature. This limits the ability to adopt best practice mangrove management techniques that minimise adverse impacts and to identify cost-effective means to achieve restoration success. Furthermore, there is a real risk that an unsubstantiated paradigm shift will take hold in policy development whereby the removal of mangroves becomes identified as a positive ecosystem service. This is a possibility if monitoring of the impacts of their removal continues to be absent or minimal, and any management actions are assumed, but may not actually achieve restoration objectives.

This paper reviews the monitoring programs associated with mangrove eradication in New Zealand, and identifies the key monitoring protocols that should be included in mangrove removal activities. These measures should determine whether mangrove removal leads to the desired environmental outcomes, and also to inform future coastal management. New Zealand community groups and coastal management authorities have been clearing mangroves for over a decade now (de Luca, 2015). This provides us with the opportunity to explore the varied approaches to the removal of mangrove vegetation. Furthermore, we can use existing monitoring data to assess relative success of mangrove management activities to date in achieving restoration objectives.

1.1 Background – Mangroves in New Zealand

Avicennia marina subsp. *australasica* is the only mangrove species occurring in New Zealand (Morrisey et al., 2010). Early European records document the presence of mangroves (Swales et al., 2015) and pollen analysis confirms the presence of mangrove pollen in sedimentary deposits older than 8000 years (Mildenhall, 2001), confirming that this is an indigenous plant of the North Island of New Zealand. Due to a combination of climate gradients (Beard, 2006) and dispersal limitation (de Lange and de Lange 1994), mangroves are presently only found in harbours and bays in the northern half of the North Island, north of around latitude 38°. Phases of mangrove expansion have been mapped from aerial photographs dating back to the 1940s (see Swales et al., 2015). Rates of increase, and the periods during which colonisation was most rapid, have been variable over this time (Swales et al., 2007; Morrisey et al., 2010; Stokes et al., 2010). Drivers of mangrove expansion include estuarine infilling and associated changes in mean bed level (Ellis et al., 2004; Swales et al., 2007; Stokes et al., 2009), and decreased storm and wave activity linked to El Niño (Swales et al., 2015).

Reports produced in the 1970s (Chapman et al., 1976a and b) alerted authorities to dwindling mangrove habitat following land reclamation and grazing impacts. As a result, New Zealand mangroves were granted protected status under the New Zealand Coastal Policy (Harty, 2009). Local regulatory bodies such as Regional Councils operate according to the NZ Coastal Policy and Resource Management Act, and as such, any mangrove clearing must be approved via the resource consent process (Harty, 2009). However, minimal monitoring information is available from prior mangrove removal activities to inform decision making with respect mangrove removals. In the New Zealand context, natural resource management falls under the jurisdiction of district and regional councils (Harty, 2009).

Interestingly, a recent re-evaluation of the NZ Coastal Policy (2010) has altered the status of mangroves, in which the policy statement “no longer identifies mangroves as a species worthy of conservation”. In line with this change, some regional councils have updated their regional policies. For example, Northland Regional Council have described conditions under which mangrove removal can be undertaken as a ‘discretionary activity’ (i.e. do not require the full consent process for approval, and does not require monitoring of impacts). Waikato Regional Council continues to require consent application for mangrove removal, and they are involved in large consented clearings in both Whangamata and Tairua Harbours. Detailed monitoring has been undertaken in Whangamata Harbour (see Bulmer and Lundquist, 2015 and 2016 for monitoring results). Auckland Council and Bay of Plenty Regional Council have draft plans that are relatively permissive for removal of mangroves and seedlings (DeLuca, 2015), although the extent of the removal is bounded by an arbitrary historical mangrove boundary (i.e. back to where mangroves fringed the site in 1996). Recently (August 2016), the proposed 1996 mangrove boundary for the Auckland coastal plan was removed after consultation hearings, and the final plan is expected to return to less permissive mangrove removal conditions.

Mangrove clearing across New Zealand has been undertaken both with consent (and so legally), and without consent (illegally). A number of methods have been applied to clear the above-ground mangrove vegetation, with earlier work mostly done using chainsaws and/or slash cutters, supported by the hand-pulling of seedings across the seaward mud/sandflats. The size of any area cleared at one time was mostly limited by the manpower, and of those activities that were documented, the extent of removals ranged from < 0.5 ha to ~ 2 ha. More recently, the introduction of wide-track machinery has resulted in larger areas being cleared at any one time. The largest was cleared in Tauranga Harbour in 2010, which totalled approximately 101 ha, however cleared plots within the sub-estuaries ranged in size from 1.5 ha to 20 ha (see Lundquist et al., 2014 for a more comprehensive list of clearing activities).

The treatment of the mangrove debris has also varied across management jurisdictions. Hand-cleared vegetation has often been stockpiled on site, and incinerated once dried. Some hand-cleared material was removed from the intertidal zone which required access for a truck or tractor to transport the material off-site. In 2010, vegetation which had been cleared with wide-track machinery, was subsequently mulched and deposited on the intertidal mudflat, resulting in a persistent blanket of debris (Lundquist et al., 2014).

We summarise national monitoring of mangrove removals, focussing on three detailed case studies of legally consented mangrove removal in New Zealand (see Figure 1). Firstly we identify what specific information is needed to inform sustainable management of mangrove habitats, and secondly to determine whether the desired management objectives for mangrove removal were detected by monitoring. We then present monitoring variables that have been successfully used to elucidate success at achieving restoration objectives in mangrove management.

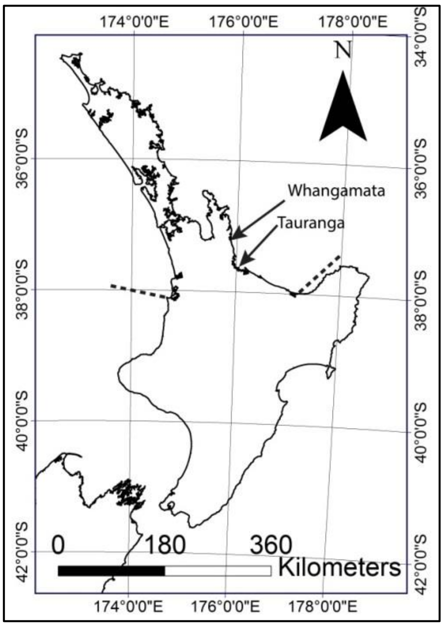

Figure 1. Location of Whangamata Harbour and Tauranga Harbour, on the north island of New Zealand. The dashed line marks the approximate southern extent of mangrove habitation.

2. Case Studies

Management objectives for mangrove removal in New Zealand generally cite two key desired outcomes: a) to return a site to sandy benthic habitat; and b) to see a return of benthic communities that were present prior to mangrove presence, with healthy bivalve abundance (Stokes 2010). As such, monitoring designs have tended to focus on these attributes. Limited long term studies are available to support any attempt at predicting the scale of the potential impacts of mangrove removal and the rate of recovery. To date, detailed studies of physical and ecological changes are mostly limited to 2 or 3 years post-clearance. What can be concluded from these studies, however, is that each site responds differently according to exposure to tidal currents, agitation of sediments from wind and waves, the depth of muddy material, the size of the cleared area and the removal methods used (Park, 2004; Alfaro, 2010; Stokes et al., 2010; Lundquist et al., 2014)

2.1 Case study 1(a) – Tauranga Harbour: mangrove clearance by community groups

Tauranga Harbour is a large barrier estuary covering 200 km^2. Numerous small sub-estuaries nestle into steep sub-catchments that introduce considerable sediment loads to the harbour. The city of Tauranga wraps around the margins of the southern section with a population of around 100,000. By 2003, mangrove coverage in some of these sub-estuaries increased from narrow marginal fringes in the 1950s (Park 2004) to occupy up to 38 % of each embayment, totalling more than 500 hectares across the Harbour.

After some unconsented clearing of mangroves during the 1980s and 1990s, the regional council (Bay of Plenty Regional Council– BoPRC) and the local District/City council supported the creation of 'Estuary Care Groups' to facilitate a more regulated approach to mangrove management. Using volunteer labour these groups removed < 20 hectares of mangroves from the harbour in the space of 5 - 10 years. The Regional Council also provided groups with community monitoring guidelines to evaluate surface elevation changes, surface macrofauna communities and intertidal vegetation boundaries (Schwarzet al., 2004). The resulting data were delivered to the regulatory bodies (council), not only as evidence of on-going mangrove expansion but also to support consent applications for further clearance of mangroves. Unfortunately the results and analysis undertaken by these care groups are not readily available or useful for two reasons, firstly that substantial inconsistencies in methodologies occurred in this volunteer monitoring, both in methods and in regularity of monitoring, and secondly that poor documentation and record keeping at council level occurred such that results are not available to inform further management activities.

The extent of mangrove removal undertaken by community groups at this time was limited by the amount of volunteer labour available to each group, and the limitations of the methods employed (mainly chain saws and brushcutters which were used to partially clear mangrove stands). Consent conditions stipulated that only mangroves which had established since 1984 could be cleared, while older stands were to remain (Stokes, 2010).

While detailed results of this activity have not yet been compiled by council or any data made available, independent postgraduate research monitored changed over an 18 month period within one of the sub-estuaries. The study reported a coarsening of surface sediment, in the first 12 months after mangrove removal coupled with a marked fall in surface topography of up to 38 mm. In an analysis of benthic macroinvertebrates no links between community structure and 'restoration' could be confirmed, at least in the 1 mm sieve size fraction (Stokes, 2010).

2.2 Case Study 1(b): machinery assisted clearing in Tauranga Harbour

In 2010, Bay of Plenty Regional Council (BoPRC) authorised the use of wide-track machinery to clear approximately 110 hectares of mangroves from 10 sub-estuaries of Tauranga Harbour. Vegetation was mulched as it was cleared, and the mulchate was left in-situ (Lundquist et al., 2014). Under the issued consent, no collection of baseline data was required prior to this large-scale clearance (Consent 66505 and 65693). The consent was based on a small trial of mechanical clearing, whereby epifauna were observed to rapidly recolonise tracks within a tidal cycle (Park 2008, in Harrison Grierson Consultants Ltd, Resource Consent Application 2012); however, the tracking and mulching methods that did occur during consent works had little correspondence with methods tested in the small trial. Post-clearing monitoring required by the resource consent was limited to qualitative observations, with site visits required 'every two days for no less than 1 week and up to two months'. No formal technical reports or data was collected, as these visits were primarily interested in the lack of dispersal of mulchate, and thus, lack of likely impact on neighbouring habitats. Photographs were taken to record evidence of wading bird activity, distribution of mulch material, and changes to sediment characteristics. Later, after concerns were raised by community groups, further monitoring was initiated annually in two of the ten estuaries exposed to mulching.

Due to concerns by independent scientists at the National Institute of Water and Atmospheric Research about the scale of the mangrove removals in the consent, and the lack of data to inform the consent activity, independent research was initiated. This included the collection of baseline samples prior to mangrove removals, to gather detailed data on the impacts of persistent mulchate. The study assessed the general condition of three of the ten sub-estuaries, though delays in consent activities meant that sampling only occurred within two sub-estuaries. Transect surveys of macrofauna and sediment characteristics were undertaken at 3, 6 and 12 months post-clearing in both estuaries. Additional sampling by NIWA, as part of a separate research project, occurred in five sub-estuaries approximately three years after removals. In both studies, visual surveys of sites at positions within mulch zones, and in adjacent mangrove and adjacent tidal flats were combined with sediment cores (grainsize, organic content, chlorophyll a) and macrofaunal cores to interpret both recovery trajectories and document adverse impacts. Early results identified limited dispersal or decomposition of the mulch material which resulted in anoxic sediments, elevated levels of phosphorous and ammonium, algal blooms and some occurrences of lowered dissolved oxygen below ANZECC guidelines (Lundquist et al., 2012). No evidence of site recovery was found within 12 months at either site, and sites all exhibited degraded conditions at 3 years post removal (Lundquist, unpublished data).

2.3 Case Study 2: Whangamata Harbour

In 2012, Waikato Regional Council (WRC), taking note of the Tauranga Harbour experience, approved clearance of up to 23 hectares of mangroves in Whangamata Harbour. Consent conditions required detailed baseline surveys and regular monitoring for up to three years post-clearance (Bulmer and Lundquist 2014, 2015, 2016). WRC chose to trial and monitor both mechanical and hand removal methods, at locations with varying degrees of tidal flushing and exposure to wind waves and storm conditions. Here, however, all mulchate was stockpiled and incinerated rather than spread across the cleared substrate (Bulmer and Lundquist, 2014).

To date, these monitoring conditions are the most detailed required by any mangrove removal consent in New Zealand. As such, it is likely that a better estimate of the timeframes required to achieve a state of restoration will become possible. In addition, rather than simply reporting on point-in-time physical and biological conditions, the monitoring program includes an assessment of trends and triggers. This approach provides an opportunity to adopt an adaptive management approach whereby further clearances can be discontinued if any harmful trends are identified, or site-specific remediation can be undertaken should it be deemed appropriate. Importantly, this approach attempts to quantify recovery or restoration using a range of physical and biological parameters. Criteria included in the monitoring at Whangamata Harbour are summarised in Table 1.

Table 1. Key criteria used to identify 3 year trends, characterised as positive or negative trends.

CRITERIA	POSITIVE TREND & TRIGGERS	NEGATIVE TREND & TRIGGERS
Sediment characteristics (mud content)	decrease in mud content; increase in Redox Potential Depth (RDP)	increase in mud content of > 5 % and/or RPD decreases by > 10 %
Sediment characteristics (compaction)	compaction class (firm, slightly soft, soft, very soft) trending towards more compact sediments	A decrease in compaction relative to initial conditions
Saltmarsh and bank erosion	No significant change to bank condition	Significant erosion (characterised on a case-by-case basis)
Mangrove encroachment	No significant change in saltmarsh-mangrove boundary	Landward transgression in the saltmarsh-mangrove boundary of more than 30 cm in a 12 month period
Seagrass extent and density	Increase or no significant reduction in seagrass cover (defined as > 10 % areal coverage in any 3 month period and > 20 % in any 12 month period)	Decreases in areal coverage > 20 % over twelve months (noting seasonal variability)
Benthic fauna	Community composition and abundance trends towards open intertidal tidal flat. > In abundance to at least 20 % of the abundance of the top 5 ranked taxa found on adjacent tidal flats	Minimal colonisation by macrofauna; presence of macroalgae blooms.

Following 24 months of monitoring, Bulmer and Lundquist (2015) reported that larger sites (greater than 1 hectare) showed some increase in mud content, suggesting an undesirable trend (Bulmer and Lundquist, 2015). It was unclear, however, whether this sediment deposition was related to mangrove removals or land-based activities (such as forestry) in the neighbouring catchment. Macrofaunal communities were more similar to intact mangrove community structure than adjacent tidal flats and this was similar across all sites regardless of size, exposure or removal methods. However, some trends toward sandflat communities were observed three years after removals (Bulmer and Lundquist, 2016). The study reported a more rapid coarsening of surface sediments where mangroves were cleared by hand, while the width of the area cleared correlated with recovery trends. Sites where width of removal was < 30 m showed a more rapid change in sediment characteristics, as did any cleared habitat positioned near tidal channels (and increased hydrodynamic forces). Adverse impacts, as witnessed in Tauranga Harbour (i.e. long-lasting anoxic sediments, macroalgal blooms and delayed colonisation by macrofauna) appeared to be minimal across Whangamata Harbour, although trends toward sandier substrates were slower than anticipated.

The case studies outlined above identify the varied approaches to monitoring associated with mangrove removal in New Zealand. What is evident is that the numerous management authorities have applied inconsistent approaches to mangrove removal, and most alarmingly the larger scale removals can result in adverse impacts. The introduction of mechanical clearing techniques, and poorly documented monitoring (post-clearance) may delay the adoption of best practice in terms of appropriate scale of removal plots, and the timeframes identified to undertake the clearing activities. It is important that coastal managers are aware of the potential impacts associated with their preferred approach to mangrove management, to ensure the likelihood of negative outcomes are mitigated (Lundquist et al., 2012, 2014). A new monitoring framework, incorporating monitoring data to date, can improve the cost-effectiveness of the techniques employed. It can also provide an improved opportunity to identify and prioritise areas where restoration activities are likely to result in successful restoration after mangrove clearing.

3. Knowledge Gaps

'Ecological restoration' implies an endeavour to return an ecosystem back to, as much as possible, it's 'original' condition (Dale et al., 2014). Under this principle, the ultimate goal of wetland restoration is" to create a self-organising, self-maintaining and functioning natural ecosystem that is resilient to perturbation without further assistance" (Zhao et al., 2016). Using this definition, how do we evaluate the success of using mangrove clearing as a restoration tool? A cleared site should reflect similar physical conditions as those present in the adjoining tidal flats, while the biology should also mirror that of a typical tidal flat habitat. This implies minimal mud content in the sediments, no mangrove root biomass, dissipation of inherited nutrients and no evidence of overabundant surface macroalgae. Importantly, macrofauna community structure should also be analogous to the adjacent tidal flat communities.

There are two main issues relating to monitoring of restoration/rehabilitation following mangrove clearing: firstly, the range of parameters that are monitored; and secondly the monitoring time frames. Monitoring of mangrove removal in New Zealand to date, has been variable in terms of temporal sampling, and as yet, there are no cases where repeated monitoring has continued beyond three years (e.g., Alfaro, 2010; Bulmer and Lundquist, 2014, 2015, 2016). Typically wetland restoration projects report some satisfactory level of recovery long after a 5 year timeframe, although this is usually in the context of planting for rehabilitation rather than clearance of vegetation. Siple and Donahue (2013) suggest that cleared sites in Hawaii may take decades to return to a pre(mangrove)-clearing/pre-invasion condition. Given the lack of available data in New Zealand, clearly any attempt at a synthesis of findings is going to be greatly limited. Stokes (2010) and Bulmer and Lundquist (2016) have reported minimal coarsening of sediment after 18 months to 3 years, while other sites with more regular and stronger flow velocities achieved a more substantial coarsening (Stokes, 2009; Alfaro, 2010). Conversely, where cleared areas are large in relation to flushing potential, mud content has been observed to increase, at least within that 2 year timeframe (Bulmer and Lundquist, 2015).

Sediment condition drives benthic community structure (Cummings et al., 2003, Thrush et al., 2003). There is little evidence available to suggest that benthic communities within cleared substrates become consistent with tidal flat community structure within the first few years (Stokes, 2010; Lundquist et al., 2012; Bulmer and Lundquist, 2015). Outside of New Zealand, assessments of benthic food webs following clearance of red mangrove (*Rhizophora)* canopy, suggests that a recovery from removal occurs gradually (and is not governed by top-down effects (Siple and Donahue, 2013). An initial increase in species abundance, sometimes coupled with increased species diversity, is often noted, but assumed to be a response of opportunistic species to the liberation of benthic food sources (Felsing 2006, Alfaro 2010, Sweetman et al., 2010, Lundquist et al., 2012, Siple and Donahue 2013). Long term monitoring is required to separate seasonal fluctuations, the influx of short-term colonising species, and the development of stable benthic diversity. Again, this reinforces the necessity for a minimum of 5 years monitoring to confirm whether a return to tidal flat biogeochemistry and biological community structure occurs.

The monitoring parameters outlined in these case studies seem to have overlooked some potential unfavourable impacts of mangrove clearance. Typically these relate to the measure of biogeochemical change that develops on and below the surface. Firstly, surface topography is typically not measured. Mangrove-dominated substrates consists of sediment fines mixed with large volumes of mangrove root material (Kauffman and Donato 2012). How quickly the surface topography changes in response to mangrove clearing will be partly linked to the existing belowground biomass (mangrove root material). The depth and density of mangrove root material, and its decomposition over time, is already typically ignored in monitoring schemes. Root material varies temporally and spatially (Saintilan, 1997; Komiyama et al., 2008; Lovelock, 2008), making it challenging to measure decomposition rates. Stokes and Harris (2015) described a considerable coarsening of sediment between 3 and 4 years post-clearance which they hypothesised may be linked to the slow rate of root decomposition. The timeframe sits within model predictions developed by (Gladstone-Gallagher et al., 2014) where buried wood and pneumatophores decompose to half their original weight between 317 and 613 days. Mangrove roots resist decay because they grow in saturated, low oxygen soils (Middleton and McKee, 2001) and their rate of decay is influenced by species (Middleton and McKee 2001), location and tidal elevation (Huxham et al., 2010). Generally, however, complete decay of woody material is expected to take several years (Mackey and Smail, 1996; Middleton and McKee, 2001; Siple and Donahue, 2013), potentially decades (Gladstone-Gallagher et al., 2014; Lundquist et al., 2014).

Another aspect of change in surface topography that is rarely included in monitoring of mangrove removals is that of site elevation. If sediments and biological material are flushed from a site, then a considerable deflation of the intertidal surface can be expected. This phenomenon has been reported in New Zealand, where surface elevation fell by 9 – 38 mm within 2 years of mangrove removal (Stokes et al., 2009). This is consistent with findings reported in other countries, where above-ground mangrove vegetation was cleared for development (Hayden and Granek, 2015), and where mangroves were destroyed during a hurricane (Cahoon et al., 2003). A lowering of surface levels has potential implications for nutrient loading and sedimentation on adjacent nearshore habitats. Such geomorphological change also results in an increase in relative sea level which may impact on adjacent saltmarsh habitat (Rogers et al., 2013; Saintilan and Rogers, 2013). Additionally, exposed shorelines may be more vulnerable to erosion, particularly if localised relative sea level rise is coupled with predicted global sea level rise (Hayden and Granek, 2015).

Finally, while a few consent monitoring schemes have looked at adverse impacts such as macro-algal blooms and bacterial blooms, in most cases in New Zealand, these have not been incorporated into monitoring designs. Granek and Ruttenberg (2008) suggest even small-scale clearing of mangroves can lead to changes in biotic conditions. Their study in the Caribbean identified a link between cleared sites and increased macro-algal biomass and cyanobacterial growth, both earlier than 12 months and after 8 years. Strong links between nutrients and algal blooms exist in many estuaries (Anderson et al., 2002), particularly urbanised locations (Wallace and Gobler, 2014). Macroalgae blooms, specifically opportunistic Ulvalean species are a common symptom of eutrophication (Valiela et al., 1997). Macroalgal blooms may be driven by changes to the N:P or N:Si ratio, while shifts in macroalgae species are linked to an increase in the ratio of DOC:DON (Anderson et al., 2002). Measures of bacterial or algal biomass have not been part of monitoring protocols in New Zealand post mangrove removal, however significant macro-algal blooms associated with persistent mangrove debris were reported in Tauranga Harbour and elsewhere (Lundquist et al., 2014). It is important to have some knowledge of the background nutrient loading of an estuary in order to determine the likelihood of blooms following mangrove eradication.

It is important to highlight here two additional environmental benefits provided by mangroves that have not been considered in the development of mangrove management strategies in New Zealand. Firstly, mangroves are known to contribute to the health of coral reef and fisheries (Mumby and

Steneck, 2008), although these services may be less important in temperate than in tropical mangrove forests (Morrisey et al., 2010). Reef filter feeders, for example, take up mangrove derived nutrients; while mangrove organic matter can be transported further afield by fish movement (Lee, 1995). Mangroves may therefore be an important source of nutrients to adjacent reef systems and therefore any reduction in mangroves may lead to reduced nutrient availability for sessile reef organisms (Granek et al., 2009). A more recently recognised ecosystem benefit is that of climate change mitigation. Numerous studies have identified the enormous carbon burial capacity of mangroves (Kristensen et al., 2008). Any activity that impacts on carbon stores, such as the removal of mangrove vegetation, and in turn adds to atmospheric carbon dioxide, can be viewed as a negative outcome for climate change mitigation (Bulmer et al., 2015 and 2016). This aspect of ecosystem management could also be considered in the context of mangrove removal in New Zealand.

4. Monitoring objectives to identify restoration goals following mangrove eradication

Even though regulatory authorities today are often constrained by limited budgets and staff, an opportunity exists to link industry needs with research objectives. Below, we provide a discussion on what we consider to be key monitoring parameters. Collectively, these parameters can be used to asses if and when management objectives are achieved. We have detailed the recommended monitoring approaches using three levels (A, B and C in Table 2), grouped according to necessity, complexity and expense (Table 2). Level A monitoring parameters reflect methods that can be utilised with limited budgets, and can also be performed by community or volunteer groups with some supervision of techniques and data reporting. Level B monitoring parameters are those that would require some laboratory analysis and/or specialist knowledge (i.e. benthic macrofauna analysis) while Level C monitoring parameters could be included in well-funded, comprehensive monitoring programs. Information needs are compiled under three key themes; physical, chemical and biological, detailed in Sections 4.1 – 4.3. Additionally, as levels of replication must be considered when planning any monitoring program, we have included recommended levels of replication relative to the size of proposed clearance areas (Figure 2).

Table 2. Hierarchical measurements suggested for monitoring of mangrove removals, as per details in text. Details of the methodologies and limitations are described in Sections 4.1 – 4.3.

MEASURING PHYSICAL & CHEMICAL CHANGES (sampling intact mangroves; cleared; tidal flat)		
LEVEL A monitoring	LEVEL B monitoring	LEVEL C monitoring
Description of surface sediment using field-based methods http://soilquality.org.au Sediment compaction (Schwarz et al., 2004 or Bulmer & Lundquist, 2014) Redox Potential Depth (Bulmer & Lundquist, 2014) Surface elevation – basic method (Schwarz et al., 2004) Bank erosion – photopoints (Schwarz et al., 2004)	Grain size: clay; silt, sand, gravel fractions Root biomass within macrofauna cores (Bulmer & Lundquist, 2014) Sediment organic content (Stokes, 2009) Grain size changes with depth using field-based methods (Stokes, 2009) Sediment chlorophyll-a content Sediment organic content (LOI method)	Surface elevation changes – long term RSET method (Stokes et al., 2009) Nutrient concentrations (N, P) Sediment accumulation in adjacent vegetated habitats (sediment traps; Stokes et al., 2010) Grain size changes with depth using lab-based analysis of size fractions Sediment dating – to develop mangrove sediment accretion history and rates of carbon accumulation Stable isotope analysis – to identify carbon sources

MEASURING BIOLOGICAL CHANGES		
LEVEL A monitoring	LEVEL B monitoring	LEVEL C monitoring
Habitat change: seagrass (Bulmer & Lundquist, 2014) Habitat change: saltmarsh (Bulmer & Lundquist, 2014) Habitat change: shellfish (Bulmer & Lundquist, 2014)	Habitat change: sampling established/pre-existing shellfish abundance	Habitat change: Shellfish condition – adjacent sandflat and established/pre-existing shellfish beds (Bulmer & Lundquist, 2014)
Surface observations - Epifaunal species/abundance - No. of dead pneumatophores - Any new seeds/seedlings - % cover of macroalgae/algal blooms - Crab burrow abundance - Shallow infaunal bivalves (handraking upper 5 cm) - Depth to root mat (& note presence/absence of root mass)	Benthic macrofauna counts (0.5 mm sieve; Bulmer & Lundquist, 2014) Bird monitoring (Schwarz et al., 2004)	

4.1 Monitoring physical parameters

The following parameters must be considered (before and after mangrove removal) in order to develop a detailed understanding of the physical changes that occur within an estuary following mangrove removal;

a) Grain size change on the surface
b) Grain size change with depth (compaction would be associated with this)
c) Vertical elevation changes and change to relative sea level (tidal inundation)
d) Potential for bank erosion

Sediment grain size is a key measure of 'recovery', as benthic community structure is most likely to respond to changes in mud content (Thrush, 2003). Surface sediments should be sampled in order to identify which sediment size fraction is flushed following mangrove removal. The sampled depth can vary, although ideally this should be limited to the upper 2 – 5 mm. Results can be reported using % mud and % sand, via sieving methods, although a more useful characterisation of surface sediment might include % silt, % clay, and at least 3 sand fractions.

As sediments become more compacted, amenity value (i.e. walking access) increases. Sediment compaction is most easily monitored by assessing the depth to which footprints penetrate the surface (Bulmer and Lundquist, 2014). Commercial penetrometers can also be used, however for the sake of cost and simplicity, community groups generally utilise a steel rod with a pointed end. This is dropped from a known and consistent height and depth of penetration into the substrate is measured (Alfaro, 2010; Schwarz et al., 2004). Logic demands that as any remaining mangrove roots die and decompose, the surrounding sediments become softer and more penetrable, and conversely, as muds are flushed out of the system, sediments will compact and become firmer underfoot. However, we recommend that sampling is augmented by a basic stratigraphical assessment, as sand armouring can camouflage persistent muds.

A recent study has identified the potential for sand armouring where mangroves are cleared in locations that receive low flow velocities (Stokes and Harris, 2015). This suggests that a reported positive change in surface grain size from mud dominated to sand dominated, may mask the trapping and retention of buried mud-dominated sediments. Simple stratigraphical descriptions could address this. A 30 cm core could be described using field-based textural assessments at 2 – 5 cm intervals (see http://soilquality.org.au/factsheets/soil-texture as an example)***.***

An important, but often overlooked impact of change following mangrove removal is the potential for significant lowering of the surface topography as a result of sediment flushing, sediment compaction and root collapse (Cahoon et al., 2003; Stokes et al., 2009). A lower seabed increases relative sea level, which, as a result of increased inundation may impose physiological stresses on adjoining saltmarsh communities or increase the risk of bank erosion. Schwarz et al., (2004) describe a low cost approach for monitoring bed-level whereby 1 m long wooden stakes are hammered into the sediment, along a transect. This technique was utilised by community groups to capture 'surface elevation change'. One limitation of this approach is the unknown vertical movement of the stake which may artificially amplify results. Use of longer, thinner stainless steel pegs is desirable as these are more stable than wooden pegs, and being of smaller diameter, it is less likely that erosion will occur around the peg itself. Replication at each sampling station would be required, preferably with 6 – 12 pegs positioned within a ½ m plot (Stokes, 2009). A more reliable (but equally more expensive) method employs Rod Sediment Elevation Tables (RSET) see (Stokes et al., 2009). These RSETs are driven many metres into the sediment and provide the opportunity for long term monitoring of inflation or deflation of the intertidal substrate (Cahoon et al., 2000).

4.2 Monitoring chemical parameters

To develop an understanding of chemical changes in the substrate following mangrove removal we must consider:

a) Nutrient loading and algal bloom occurrence
b) Background nutrient loads
c) Oxic layer/redox

Estuaries have enormous capacity to accumulate sediments, nutrients and contaminants (Anderson et al., 2002). If a site already harbours high nitrogen and phosphorous levels, introducing decomposing mangrove litter, and liberating carbon rich mangrove sediments is likely to shift the system into a eutrophic state. Algal blooms are associated with nutrient rich systems, and can pose a real threat to estuarine organisms (Howarth et al., 2011). As such, baseline surveys need to include a spatial assessment of sediment nutrient loads, along with a record of the presence or absence of existing algal cover. Sampling of porewater nutrients can quantify whether removal areas have elevated nutrients, and whether these nutrients are likely transported into neighbouring habitats. Monitoring of temporal/seasonal variability in macro-algal blooms can determine how long-lasting nutrient impacts are following mangrove removal. In addition, nutrient sources can be determined, using elemental ratios (i.e. Yamamuro, 2000), which would identify sites loaded by mangrove detrital decomposition and/or sites smothered by macroalgae.

Measurement of sediment redox characteristics should also be included in any prescribed monitoring, as the development of oxic sediments is a requirement for recolonization of sand-dwelling benthos. Measurements can be made via direct redox probe readings, or simply by measuring the depth to which black/dark grey sediments occur (reflecting the end of the oxic layer), as described in Lundquist et al. (2012).

4.3 Monitoring biological parameters

To develop an understanding of biological changes following mangrove removal we must consider:

a) Benthic fauna
b) Vegetative condition of clearance sites
c) Impact on neighbouring habitats (e.g. bivalve beds, seagrass meadows, salt marsh)

Macroinvertebrates are often used as an indicator of environmental change (Hilty and Merenlender, 2000). As surface fauna (i.e. epifauna), are easily identified and counted, this approach has been incorporated into both community and consented monitoring. Limited species knowledge of epifaunal taxa is required as only 4 to 6 grazing species are typically encountered in visual surveys. In addition, hand raking for bivalves is a rapid sampling method, and generates important information on the recovery status of the site. Conversely, enumerating other buried macroinvertebrates, or infauna, is a time-intensive process that often requires specific taxonomic expertise, particularly to identify the numerous polychaete species typically associated with mangrove and tidal flat habitats. Assessments of species abundance and richness of benthic macroinvertebrates is commonly used to assess change; however caution must be observed when interpreting results as these univariate metrics document only number of species or individuals, which may not reflect successful restoration of ecological function. Rather high values of these univariate metrics may instead reflect macrofaunal communities dominated by opportunistic or disturbance tolerant species, which has often been documented in degraded systems typical of unsuccessful mangrove removals (Lundquist et al., 2012, 2014). Multivariable statistical approaches are a preferred method to analyse restoration success of benthic macrofaunal communities, and allow differentiation between communities dominated by ephemeral, opportunistic colonisers from species that are indicative of a rehabilitated site.

As fish and shorebirds (at least in temperate mangroves in New Zealand) tend to only spend small proportions of the tidal cycle within mangrove ecosystems, typically they are poor indicators of restoration success unless substantial efforts are put into elucidating spatial and temporal (intra- and inter-annual) patterns in fish and bird usage of mangrove habitats. To date, information to confirm the relative role of mangroves for key species is not yet available, though monitoring in Whangamata Harbour of bird responses to mangrove removals suggests minimal impacts on birds after removals, where mangrove habitats remain (Wildlands, 2014).

The presence of mangrove root biomass is another key measure for determining whether a site is transitioning towards tidal flat conditions, as thick mangrove biomass delays both sediment erosion and colonisation by macrofauna. This can be rapidly assessed by counting the number of pneumatophores (if present), and digging into the sediment to detect the presence of a root mass and quantify volumes.

Monitoring should include assessment of potential impacts on neighbouring habitats, including habitats of particular importance such as shellfish beds and seagrass meadows. If the size of a mangrove clearing is large compared to the expected flushing of sediment and nutrients, it is likely that adverse downstream effects on sand-dwelling macroinvertebrates will occur. However, note that to date, no adverse impacts have been detected on neighbouring tidal flat and seagrass meadows, even within locations within a few metres of mangrove removals. This is likely due to the slow rates of sediment erosion observed to date from mangrove removals, such that rates of sediment erosion have been unlikely to impact on neighbouring habitats. Bivalves in particular are an important component of the estuarine trophic system, providing food for foraging birds, fish and people. Some bivalves, such as the New Zealand species *Paphies australis* and *Macomona liliana* prefer a sand-dominated substrate and show adverse sensitivities to an increase in mud content. Similarly, redistribution of fine sediments may lead to increased turbidity and sediment deposition, and decline of seagrass, while landward transport of sediment may smother remaining mangroves or saltmarsh beyond their ability to thrive. Baseline surveys are a necessity, followed by seasonal observations of plant species richness, density and general condition.

In New Zealand, early attempts at mangrove clearing were restricted to clearing above-ground biomass by hand from narrow marginal strips. More recently however, large machinery in the form of wide-track diggers, have been used to cut and mulch larger areas of mangrove vegetation (Lundquist et al., 2014). Observed dieback of mangroves adjacent to some of the cleared plots suggests that large-scale mechanical clearing may be detrimental to neighbouring vegetation. Visual descriptions, coupled with repeated photographic documentation from nominated 'photo-points', are a useful method of picking up any changes to the coverage and condition of remaining vegetation. Along with mangroves, this method can also include an assessment of seagrass and saltmarsh; these techniques have also been used to measure rates of encroachment of mangrove forests into saltmarsh habitats, a process that is anticipated to increase in frequency with sea level rise.

4.4 Replication and frequency of sampling

It is imperative that any restoration program include regular monitoring at an appropriate scale of replication to document rates of recovery and any adverse or unexpected impacts. Lundquist and Bulmer (2015) documented little change in some monitoring parameters within 12 months of mangrove removal in Whangamata. As such, a more staggered repetition of monitoring may be required to ensure longer-term sampling can be incorporated. Figure 2 provides a suggested sampling protocol relative to the size of the area to be cleared of mangroves.

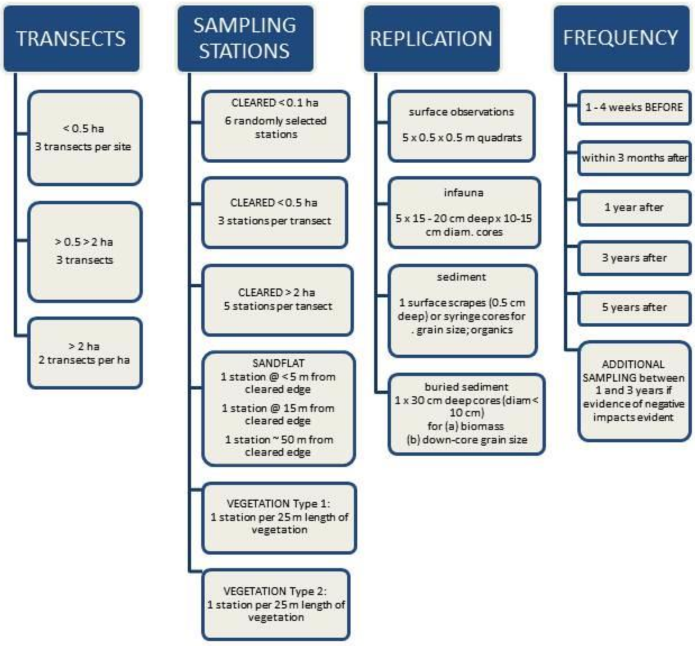

Figure 2. Suggested levels of sampling replication, relative to size of area subjected to mangrove clearing.

5. Final remarks

The suggested monitoring parameters discussed above not only hint at the complexity of estuarine systems but also highlight the need for detailed before, after and repeated monitoring programs where mangrove removal occurs. Unfortunately the mangrove removal programs in New Zealand have not yet led to the development of a consistent management or monitoring approach. Indeed, there is little conclusive evidence that any key management objective has been achieved. Resource managers must first have a clear management objective in mind (e.g. whether a site is to be "restored" to a pre-determined pristine condition or to one that is "fit for purpose" (not necessarily pristine), and in so doing seek a better understanding of a) whether stressors associated with mangrove removal can be stopped or mitigated and b) whether the system will recover on its own or whether continued intervention is required (Elliot et al., 2007). The background, pre-removal conditions of an estuary will influence the rate of restoration. Detailed, repeated monitoring is central to the development of an informed management program designed to support a healthy estuarine ecosystem.

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