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MANGROVE MANAGEMENT IN AOTEAROA NEW ZEALAND:

A BIRD'S EYE REVIEW

OCTOBER 2021

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Report prepared in association with the Office of the Prime Minister's Chief Science Advisor (OPMCSA) and the Human-Wildlife Interaction Research Group (HWIRG) at Massey University

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Satellite imagery of the Firth of Thames in Waikato. A dark green line of mangroves stretches along the southern coastline, their growth fuelled by substantial increases in sediment and nutrients in waterways from human-driven land use changes. Miranda – a world-renowned area for wading birds – sits in the upper left-hand corner of the image, diagonally opposite the town of Thames across the bay. This region represents a prime case study of how mangroves are situated at the intersection of ecology, conservation, sociology, and politics in Aotearoa New Zealand.

SUMMARY

Aotearoa New Zealand's mangroves are expanding rapidly, fuelled by human-induced changes to river catchments. Mangrove expansion has prompted communities to remove mangrove vegetation, a process controlled by regional councils using resource consents. This review synthesises data from these consents, highlighting the extent of mangrove removal and its potential effects on avifauna.

In total, this review synthesised data from 148 resource consents granted since 1994 by relevant regional councils. Regional councils granted a total of 330 hectares of mature mangrove removal, a fraction of the area of mangroves gained over the same period. Resource consents predominantly targeted the removal of mature mangrove stands and the average duration of consents – the time during which mangrove removal was permitted and maintained – was 18 years. Most mangrove removal was in Auckland and Bay of Plenty, removing 1.8 and 9.7 percent of their current mangrove forest areas respectively.

The predominant rationale for large-scale mangrove removals (>1000m^2) was the restoration and improvement of recreational and amenity values in coastal environments. By contrast, small-scale removals (<1000m^2) were typically undertaken for the development and/or maintenance of coastal or road infrastructure.

Avifauna were poorly represented among consent conditions and assessment of environmental effects (AEE) reporting was seldom informed by scientifically rigorous monitoring. Review findings highlight the lack of adequate monitoring processes associated with mangrove removals, particularly for large removals. In this respect, this review has quantified the depth of the mangrove-avifauna knowledge gap, rather than filled this gap. Nevertheless, a limited number of case studies indicated that some coastal birds are likely to benefit from mangrove removal, and few adverse effects were documented for mangrove-using birds (excluding banded rails) in removal sites where large areas of mangroves were retained. Contrastingly, available evidence suggests that banded rail populations may decline after mangrove removal. However, mangrove-avifauna findings should be interpreted cautiously; case studies are context-specific, and insights are limited by a small sample size. The implementation of standardised monitoring protocols for resource consents would serve to deepen this evidence and lead to improved management practices.

The paucity of standardised monitoring in mangrove forests has hindered effective adaptive management of these habitats, while a complex statutory framework does not reflect the catchment-scale drivers of mangrove expansion. Reorienting policies to mandate monitoring and reflect large-scale ecological processes is a priority for mangrove management in Aotearoa New Zealand.

POLICY RELEVANCE

Report findings have several implications for the direction and scope of mangrove-relevant policies:

1. *Improving monitoring*
    1.1. *Issue identified*: Mangrove removals lack a standardised monitoring framework, hindering the ability to track the restoration success of removal projects. Monitoring of avifauna is seldom mandated by resource consents (*see* Figure 4), monitoring practices differ among regional councils and removal sites, and monitoring practices for assessment of environmental effects (AEE) reporting are typically informal, lacking standardised methodologies.
    1.2. *Policy shift*: Regional council policies should facilitate standardised monitoring of mangrove removal projects, particularly for large-scale removals. Standardising monitoring allows for informed management decisions, measurable results, and inter-council comparisons of management outcomes. While this report focuses on avifauna, standardised monitoring practices should encompass a range of abiotic and biotic factors.
    1.3. *Recommended action*: Regional councils need to collaborate to define monitoring targets, standardised techniques, and timeframes to inform evidence-based mangrove management. A conceptual framework is provided by Stokes *et al.* (2016), while avifauna-specific recommendations are provided here and in Appendix Table A2.

2. *Adopting adaptive management*
    2.1. *Issue identified*: The management and removal of large areas of mangroves does not follow a consistent management framework. Different regional councils undertake mangrove removal and disposal via different methods, implement different monitoring strategies, and infrequently make use of monitoring-based decision-making.
    2.2. *Policy shift*: Mangrove removal should follow the principles of adaptive management, incorporating standardised monitoring, trial removals, and control sites to inform a stepwise, evidence-based management process. While a broad management framework is needed, decision-making should be tailored to individual sites given site-specific differences to important abiotic and biotic factors.
    2.3. *Recommended action:* Regional councils need to collaborate to define an adaptive management framework for large-scale mangrove removals. A conceptual starting point is provided here (see Figure 7).

3. *Following a holistic approach*

3.1. *Issue identified:* Current policy, both regionally and nationally, presents conflicting messages on mangrove removal and conservation. More importantly, policies largely fail to account for the interconnected nature of estuarine systems. As such, the removal of mangrove stands within estuaries targets the outcome of catchment-scale processes (mangrove growth) rather than the processes themselves (sedimentation and eutrophication).

3.2. *Policy shift*: In Aotearoa New Zealand, mātauranga Māori provides a compelling lens with which to view mangrove ecosystems and shape future policy such that management reflects the interrelated nature of estuaries' component parts. To tackle long-term changes, coastal policy needs to address estuaries holistically and target drivers of change. In the short term, mangrove management decisions should occur on a site-specific basis, but must recognise that this is not a sustainable solution to mangrove expansion.

3.3. *Recommended action*: Within national resource management policy, estuaries need to be recognised as connected to and part of river catchments and placed under the same policy framework. Within regional policies, local-scale mangrove management must be complemented by catchment-scale initiatives to reduce sedimentation and eutrophication in waterways.

4. *Prioritising restoration success*

4.1. *Issue identified*: Currently, mangroves are most frequently removed in large contiguous patches (see Figure 3) despite evidence suggesting this form of removal is unlikely to meet restoration objectives and may have adverse ecological effects.

4.2. *Policy shift*: Mangrove removals which focus on preventing further expansion and retain some mangrove habitat should be preferred to large contiguous removals. Available evidence suggests this form of management is more likely to see recovery of sandy substrates and retain a variety of habitats for avifauna.

4.3. *Recommended action*: Regional council policy should prioritise the retention of longshore mangrove strips in large removal areas (i.e., seaward-strip clearances – see Figure 3), while accounting for site-specific variance in biotic and abiotic factors.

1 INTRODUCTION

Mangrove forests are salt-tolerant plant communities found in coastal and estuarine ecosystems in the intertidal zone between land and sea (Spalding *et al.* 2010). Globally, there are approximately 70 species of true mangroves within 19 families (Morrisey *et al.* 2010) found between ±32°N to ±38°S (Quisthoudt *et al.* 2012). Despite this wide latitudinal range, mangroves are restricted to narrow strips along coastlines covering 13.4 million hectares (Thomas *et al.* 2017), less than one percent of the area of tropical forests (Spalding *et al.* 2010; FAO 2015). Despite providing valuable ecosystem services (Barbier *et al.* 2011), an estimated 20–35% of mangroves were lost globally from 1980 to 2005 (Agardy & Alder 2005; FAO 2007) and declines have largely continued in recent decades (Thomas *et al.* 2017). Mangrove declines are driven by multiple factors, including clearance for aquaculture and urbanisation, overexploitation for timber, erosion, coastal landfill, and deterioration as an indirect effect of pollution and upstream land use (Duke *et al.* 2007; UNEP 2014).

1.1 MĀNAWA: AOTEAROA'S MANGROVES

Counter to global trends, a few temperate regions have seen increases in mangrove forest extent (Morrisey *et al.* 2010). In Aotearoa New Zealand (henceforth Aotearoa), increases in sedimentation and eutrophication in coastal environments have fuelled the mangrove expansion and densification (Horstman *et al.* 2018; Suyadi *et al.* 2019). Mangroves in Aotearoa, comprising a single species *Avicennia marina* var. *australasica*, cover approximately 26,000 hectares (Spalding *et al.* 2010) along the northern coastlines of the North Island. Counter to global trends, these coastal forest have increased at an average rate of 3-4% per year since the mid-1940s (McBride *et al.* 2016; Horstman *et al.* 2018), the equivalent of 1,000 hectares gained in 2010 alone. However, recent estimates of total mangrove cover (ca. 26,500 hectares; LINZ Data Service 2021) do not indicate gains of this magnitude, and mangrove growth rates appear site specific and irregular (Suyadi *et al.* 2019).

Historically, mangrove forests in Aotearoa have undergone both increases and decreases in spatial extent (Figure 1). With the arrival of European settlers, Aotearoa lost mangrove forest to coastal development, grazing, pollution and land reclamation for ports and agriculture (Morrisey *et al.* 2007, 2010; Stokes *et al.* 2016), a scenario which echoes contemporary global trends. Given that this loss largely occurred before aerial photographic surveys began in the 1930s (Morrisey *et al.* 2010), its full spatial extent is unknown. This decline was likely halted by a combination of coastal policy and accelerating rates of sedimentation. The New Zealand's Harbours Amendment Act of 1977 made seabed reclamations for agricultural purposes illegal (Morrisey *et al.* 2010), while mangroves were granted protected status under the 1994 New Zealand Coastal Policy (Harty 2009a). This latter policy meant that mangrove clearance required approval from local regulatory bodies (regional councils) via a formal resource consent process (Stokes *et al.* 2016).

While changes to national policies made mangroves more difficult to remove, changing landscape use facilitated the rapid expansion of mangrove forests (Lundquist *et al.* 2014b). Large-scale deforestation of river catchments and rapid coastal development over the last 50 years have accelerated sedimentation in estuarine environments (Swales *et al.* 2009). The resultant increase in fine terrigenous sediments has caused estuary infilling and built extensive tidal flats, which coupled with increased nutrient inputs, warming climate and structural modifications to estuarine environments (Schwarz 2003; Nicholls *et al.* 2004; Lovelock *et al.* 2007; Morrisey *et al.* 2007, 2010) provides additional suitable habitat for mangroves (Swales *et al.* 2009; Lundquist *et al.* 2014b; Horstman *et al.* 2018). Subsequently, mangroves have grown more densely, and colonised seaward across bare mudflats and landward up tidal channels (Lundquist *et al.* 2014b; Swales *et al.* 2015; Suyadi *et al.* 2019).

1.2 MANGROVE REMOVAL

Increased rates of sedimentation in estuaries and the resultant expansion of mangroves have raised concerns among coastal communities about negative changes to recreational, amenity, and ecological values of estuaries in Aotearoa (Thrush *et al.* 2004; Harty 2009a; Lundquist *et al.* 2014b; Horstman *et al.* 2018). To address these concerns, communities have undertaken mangrove removal – both legally and illegally – at varying scales in mangrove forests on the North Island. Typically, the management objectives of such operations is to use mangrove removal as a restoration tool (Stokes *et al.* 2016) to return coastal areas to unvegetated intertidal sandflats (Harty 2009b; Horstman *et al.* 2018).

While legal removals require resource consent from appropriate regional councils, the full extent of mangrove removal in Aotearoa is unclear; illegal removal is not uncommon (Morrisey *et al.* 2007; Horstman *et al.* 2018) and may make up more than 50% of all mangroves removed (Lundquist 2021, pers. comm.). Moreover, records of legal mangrove removals are not kept in a central database, but rather distributed among regional authorities. While a few studies provide an indication of the scale of individual mangrove removal sites (e.g., Lundquist *et al.* 2014b; Stokes *et al.* 2016), no work has addressed the total extent of mangrove clearance in Aotearoa.

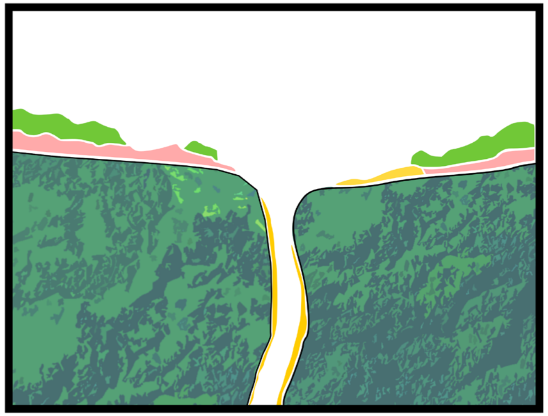
Ultimately, the expansion of Aotearoa's mangrove forests in recent decades and a concomitant increase in applications for their removal (Lundquist *et al.* 2014b) has resulted in a polarity of public attitudes towards mangrove habitats (Morrisey *et al.* 2007; Dencer-Brown *et al.* 2018). Key to this polarity of attitudes is the trade-off between the recreational and ecological values of open habitats, and the ecological values of mangrove forests. This trade-off is exacerbated by an elaborate and

Pre-human arrival

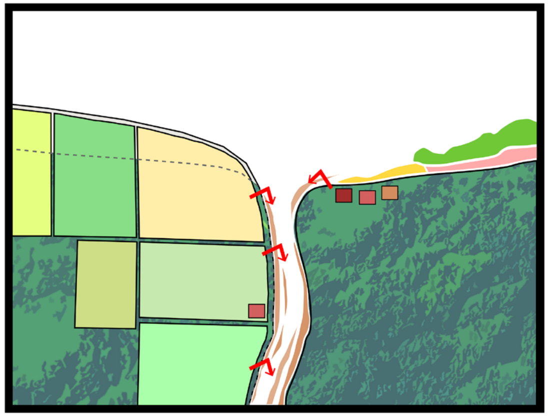
Late 1800's

Early 1900's

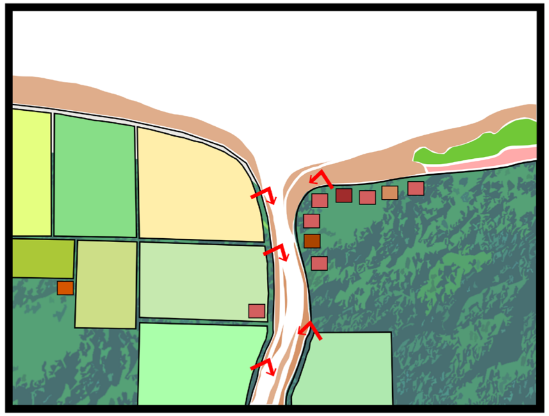
Timeline continues below



Mangrove forests (light green), sandy shores (yellow), and saltmarshes (pink) have co-existed on many of Aotearoa's coastlines for millions of years, long before humans arrived.



Early settlers often reclaimed coastal land for agriculture, removing other coastal habitats like mangrove and saltmarsh in the process. Inland, deforestation for agriculture and infrastructure development (red blocks) enabled erosion, and loss of sediment into rivers (red arrows).

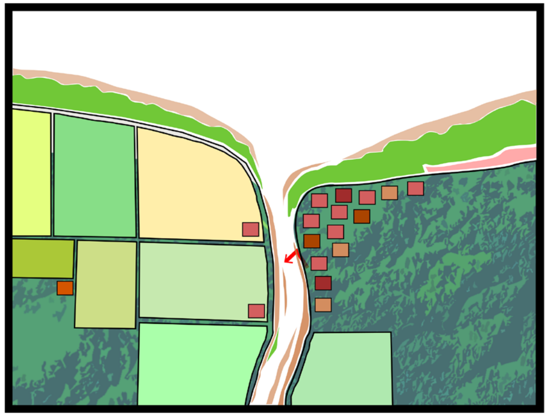


Over time, continuous sedimentation has resulted in growing mudflats (brown) along coasts and within estuaries. This mud has displaced valuable habitats like seagrass and shellfish beds. Additionally, run-off from farms has led to high nutrient loads in waterways.

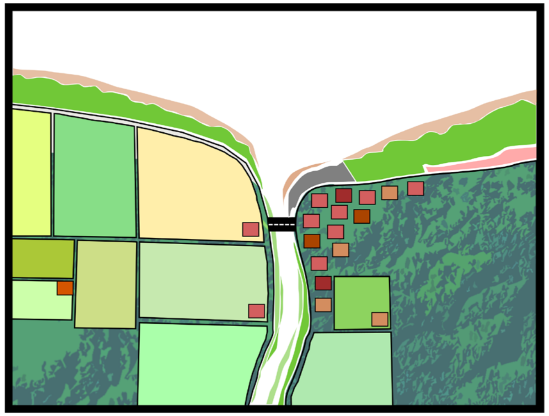
Continued timeline

Late 1900's

Present day



Growing coastal towns and the continued loss of inland forest further contributed sediment run-off into waterways, building expansive, nutrient-rich mudflats and facilitating the rapid expansion of mangrove forests in many harbours.



Infrastructure such as causeways (black) has interrupted the flow of water, providing stable conditions for mangrove growth upstream. Today, some communities remove mangroves (light grey) to restore more open habitats, but without stopping underlying sedimentation these projects seldom meet restoration goals.

Figure 1: A brief, illustrated history of mangrove forests, including their expansion and removal, in Aotearoa New Zealand (as per Morrisey et al. 2007 and Horstman et al. 2018)

sometimes conflicting statutory context which explicitly protects rights to coastal access and amenity values, while simultaneously protecting coastal habitats and their fauna.

1.2.1 STATUTORY CONTEXT

At a national level, policies within the 1991 Resource Management Act (RMA) and the 2010 New Zealand Coastal Policy Statement (NZCPS) provide rationales for both the removal and conservation of mangrove forests. The RMA seeks to protect "areas of significant indigenous vegetation and significant habitats of indigenous fauna", while simultaneously providing for the "the maintenance and enhancement of public access to and along the coastal marine area" (New Zealand Government 1991; RMA Sec. 6, Sec. 7). Additionally, the RMA prohibits the destruction or disturbance of the foreshore in "a manner likely to have an adverse effect on the foreshore or seabed, or on plants or animals or their habitat", but concedes that regional councils can expressly allow such activities if in line with a regional coastal plan or specific resource consent (New Zealand Government 1991; RMA Sec. 12e).

The NZCPS seeks to avoid significant adverse effects on indigenous ecosystems and habitats found only in the coastal environment but does not include mangroves as a habitat deemed vulnerable to modification (New Zealand Department of Conservation 2010; NZCPS Policy 11). Additionally, while the NZCPS policies seeks to preserve and restore the natural character of the coastal environment (New Zealand Department of Conservation 2010; NZCPS Policy 13, Policy 14), a 2010 revision removed specific references to mangroves under these policies, allowing regional councils to either protect or clear mangroves based on local considerations (Parliamentary Commissioner for the Environment 2020).

Conflicting national policy has left regional authorities to determine the limitations on of mangrove management in Aotearoa. Four regional councils – Northland (NRC), Auckland (AC), Bay of Plenty (BOPRC), and Waikato (WRC) – have coastlines which are home to mangroves, and the concurrent responsibility of their management (Figure 2). In the 1990s and early 2000s, regional council policies adopted a conservative approach to mangrove management, in line with national policy of the time which highlighted mangroves as a valuable natural feature, vulnerable to modification (New Zealand Department of Conservation 1994; NZCPS Policy 1.1.2, Policy 3.4.3). However, subsequent mangrove expansion and pressure from communities to limit mangrove spread have resulted in more permissive approaches to mangrove removal in updated council policies (ca. 2010) and revised regional coastal plans (Table 1).

Current council policies reflect the juxtaposition of removing mangrove forests for public coastal access, while simultaneously protecting valuable coastal habitats (including mangroves) and fauna (Table 1).

To balance these competing imperatives, councils make use of 'activity rules' and relevant 'overlays' to evaluate applications for mangrove removal on a case-by-case basis. Activity rules define the degree of council oversight required for a given activity, ranging from permitted activities (least council oversight) to discretionary activities (most council oversight). Overlays delineate the coast marine area spatially, and include areas defined by their ecological, biological, and natural character values. Such overlays and rules help determine whether mangrove removal applications are approved, identify the likelihood of adverse environmental effects, and clarify the degree of council oversight required.

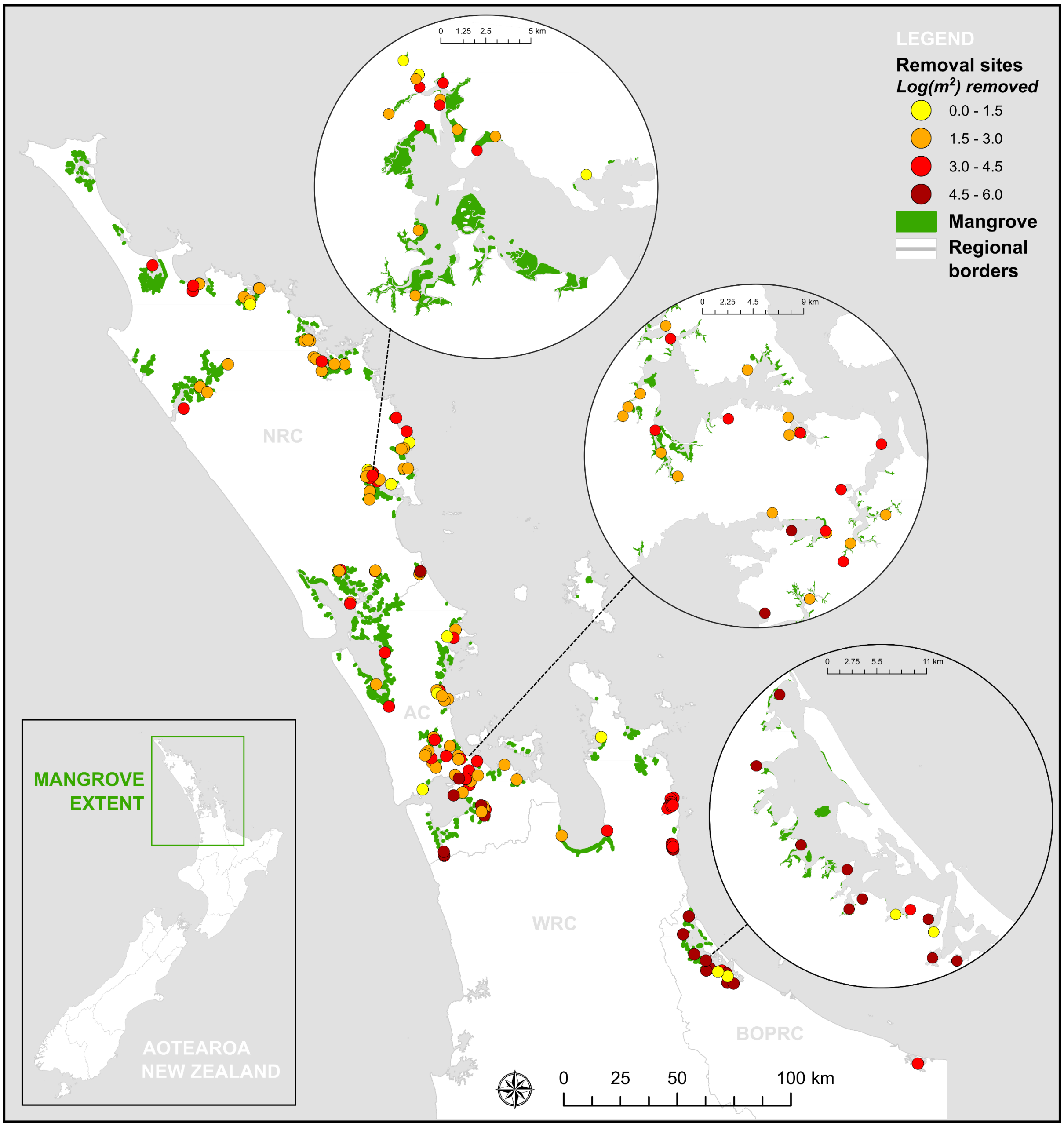


Figure 2: The distribution of mangrove forests (green areas) and mangrove removal sites (coloured circles, not to scale) in northern New Zealand. Large circles show enlarged views of removal sites in major cities, including Whangarei (top), Auckland (middle), and Tauranga (bottom). Colours of removal sites indicate the area of mangrove removed, presented on a log scale.

Table 1: The mangrove management objectives and avian-relevant mangrove rules specified by the coastal policies of the four relevant regional councils in Aotearoa New Zealand. 'Removal' refers to the removal of mature mangrove forest, unless otherwise stated

	Auckland	Bay of Plenty	Northland	Waikato
Policy document	Auckland Unitary Plan Section F2 Coastal (operative 2018)	Bay of Plenty Regional Coastal Plan (operative 2019)	Northland Regional Coastal Plan (appeals version 2020)	Waikato Regional Coastal Plan (operative 2004, updated 2012)
Avian-specific overlays	D9 Significant Ecological Areas (SEAs) Appendix 5 Wading Bird Areas (WBAs)	Schedule 2: Indigenous Biological Diversity Areas A and B (IBDA-A, IBDA-B)	Significant Bird Areas (SBAs) Significant Ecological Areas (SEAs) Significant Marine Mammal and Seabird Areas	Areas of Significant Conservation Value (ASCVs)
Mangrove management objectives	Retain mangroves with significant ecological[1] and hazard mitigation value[2] Restore and maintain natural character and ecological values, including wading bird habitat, public access, navigation, and/or amenity[3] Reduce sedimentation and eutrophication at catchment level[4, 5] Reflect Mana Whenua values of mātauranga and tikanga in mangrove management[6] For significant areas of removal, follow adaptive management practices[7]	Provide for small-scale, low-impact removals in appropriate areas[11] Prevent net loss of established mangroves in IBDAs[12] Where appropriate, remove mangroves to restore amenity, cultural, and/or recreation values as part of a catchment plan that recognises habitat and natural character values, coastal erosion risk, and mitigates increasing sedimentation[13, 14] Avoid removal of mangroves which are ecologically significant, at the edge of their range, or buffer coastal erosion[15]	Where appropriate, remove mangroves to maintain, restore, or improve biodiversity and ecosystem health, habitats displaced by mangroves, areas where mangroves have been previously lawfully removed, public recreation and walking access, amenity values, and/or legal coastal infrastructure[20] Removals must consider adverse effects, in particular effects on ecological values, increased risk of coastal erosion, effects on cultural values, amenity impacts, changes to sediment and hydrodynamic characteristics, and changes to natural character[21]	No mangrove-specific objectives Mangroves managed under rules for the removal or eradication of indigenous plant species, which aims to protect areas of significant indigenous vegetation, habitat, and fauna[24]. Concomitant rules imply that: Small-scale removal for boating and drainage is a permitted activity, while removal within ASCVs is not[25] Larger-scale removal is a discretionary activity, assessed primarily on ecological values rather than societal ones such as recreation and amenity[26]
Avian-relevant removal rules	Seedling removal is a permitted activity in WBAs[8] Removal is a discretionary activity in WBAs[9] Removal in SEAs is a discretionary activity[10]	Seedling removal in IBDA-A must occur outside of breeding and nesting seasons[16] Seedling removal and small-scale removal to maintain roosting/nesting sites is a permitted activity in IBDA-A, with conditions[17] Small-scale removal (<$200m^2$) is a controlled activity outside of IBDA-A requiring a report by a qualified ecologist on adverse effects on threatened/at risk species[18] Larger-scale removal is a restricted discretionary activity, including council discretion on threatened/at risk indigenous species and ecosystems[19]	Seedling removal and minor removal are permitted activities, but *not* in SBAs or during breeding/nesting seasons[22] Small-scale (<$200m^2$) removal is a controlled activity, where effects in SBAs and SEAs are under council control[23]	Removal of indigenous vegetation is contingent on the degree to which the activity affects the ability of the remaining habitat to survive or to support dependent fauna, and to protect faunal migration[27]

Coastal plan sections – AC Unitary Plan, Section F2 Coastal: [1]F2.7.2-1 [2]F2.7.2-2 [3]F2.7.2-3 [4]F2.7.2-4 [5]F2.7.3-2 [6]F2.7.2-5 [7]F2.7.3-4 [8]Table F2.19.4-A45 [9]Table F2.19.4-A46 [10]Table F2.19.4-A48; BOP Regional Coastal Plan: [11]2.8, Objective 39 [12]2.1, Policy NH 11 [13]2.8, Objective 40 [14]2.3.1, Policy DD 18 [15]2.3.1, Policy DD 17 [16]2.3.2, Rule DD 19g [17]2.3.2, Rule DD 20 [18]2.3.2, Rule DD 21 [19]2.3.2, Rule DD 23; Northland Regional Coastal Plan: [20]D.5.28-1 [21]D.5.29 [22]C.1.3.1-4 [23]C1.4.3; Waikato Regional Coastal Plant [24]3.2 [25]16.2.1 [26]16.2.3 [27]16.2.3-iv/v

1.2.2 ECOLOGICAL EFFECTS

Globally, mangrove removal has rarely been considered as a restoration technique (Stokes *et al.* 2016) and there is limited understanding of its short- and long-term ecological effects (Lundquist *et al.* 2014a). A common assumption among communities of mangrove removal is that it restores sandier substrates, creating open habitats with higher recreational value and simultaneously shifting the balance of ecological communities and ecosystem services towards sandflat habitats (Harty 2009a; Horstman *et al.* 2018). This assumption is based on the premise that changes to estuaries are driven by mangrove spread, when in fact changes are likely caused by the factors which *lead* to mangrove spread – such as increased sediment deposition or nutrient inputs – rather than by mangroves plants themselves (Ellis *et al.* 2004; Horstman *et al.* 2018). Indeed, there is little scientific evidence to suggest that muddy substrate will consistently return to sandflat habitat following mangrove clearance (Stokes 2009; Stokes *et al.* 2010; Lundquist *et al.* 2012, 2014b).

A change from muddy to sandy substrate relies on natural site-specific factors which remove fine silt (e.g. high wind/wave exposure) but do not deposit further sediment (Lundquist *et al.* 2014b). Successful restoration of sandflats (by mangrove removal) is typically seen in sites already dominated by sandy substrate, although sandier removal sites represent only a limited proportion of all removal sites (Lundquist *et al.* 2014b; Horstman *et al.* 2018). In the majority of cases, removal sites do not meet restoration objectives and return to a desired sandy state (Lundquist *et al.* 2014a; Stokes *et al.* 2016). Additionally, removal sites may develop unwanted phenomena such as macroalgal blooms, anoxic sediments, lower levels of oxygen in the water column, and macrofauna communities dominated by opportunist species (Lundquist *et al.* 2014b; Bulmer & Lundquist 2016).

While there is documented evidence of changes to abiotic conditions (Lundquist *et al.* 2012; Bulmer *et al.* 2015, 2017b), vegetation (Bulmer & Lundquist 2016), and macrofauna communities (Alfaro 2010; Bulmer *et al.* 2017a) following mangrove removal, there is little evidence of the response of avifauna. Given birds are frequently used as indicators of environmental change (Caro & O'Doherty 1999; Piatt *et al.* 2007; Ogden *et al.* 2014), it is surprising that avifauna are so often omitted when evaluating restoration success of mangrove removals (Stokes *et al.* 2016).

The omission of avifauna among mangrove removal studies reflects the data-poor nature of mangrove-fauna studies globally. Reviews of mangrove faunal diversity (e.g. Macintosh & Ashton 2002; Nagelkerken *et al.* 2008; Luther & Greenberg 2009) are reliant on patchy data (Macintosh & Ashton 2002) and difficult working conditions often hinder mangrove studies (Kutt 2007). Omitting avifauna from assessments of mangrove removal ignores the potential importance of these forests as habitats; global loss of mangrove forests has seen negative repercussions for a variety of mangrove endemic avifauna (Spalding *et al.* 2010). For example, Huang *et al.* (2019) identified 99 separate avian metapopulations reliant on mangroves, of which 94 experienced declines in mangrove habitat from 2000–2015. Of these, 85 metapopulations saw a decrease in metapopulation capacity, a trend driven primarily by the loss and fragmentation of large patches of mangrove habitat.

In Aotearoa, remarkably little is known about the importance of mangroves to native birds (Lundquist *et al.* 2014b; Bell & Blayney 2017; Makowski & Finkl 2018). Although Aotearoa does not have any obligate mangrove species (Crisp *et al.* 1990), multiple bird species are facultative users of mangroves, using these coastal forests to roost, breed, and forage (Cox 1977). The most apparent of these is moho pererū, the banded rail *Gallirallus philippensis assimilis*, a native rail classified as 'at risk - vulnerable' to extinction (Robertson *et al.* 2017) whose distribution in Aotearoa is largely restricted to saltmarsh-mangrove complexes (Beauchamp 2015). Banded rails are a cryptic species and therefore difficult to study; literature on their use of mangroves in Aotearoa is highly limited (Botha 2011; Beauchamp 2015) and their ecology remains poorly understood (Dunlop 1970, 1975; Elliot 1983, 1987, 1989).

1.3 AIMS

While academic literature on mangrove-avifauna relations in Aotearoa is scarce, data from resource consents may provide an opportunity to address this knowledge gap. Obtaining consent for mangrove removal from regional councils requires a pre-removal environmental assessment, and in some instances, a mandated monitoring programme to determine biotic and abiotic responses to removal activities. While such assessments are not published in peer-reviewed journals, they are publicly available and represent the prevailing source of information on mangrove removals and their potential effects on avifauna.

Thus, studying resource consents with an avifauna lens provides a unique opportunity to collate and synthesise information pertaining to:

1. The contemporary state of legal mangrove removal, including data on spatial extent, patterns of clearance, methods of removal, and societal rationales.
2. The effects of mangrove removal on avifauna, including data on avian-related consent conditions, monitoring methods, and perceived or actual changes to avian communities.

2 METHODS

Relevant data associated with mangrove removals were collected from Auckland Council (AC), Bay of Plenty Regional Council (BOPRC), Northland Regional Council (NRC), and Waikato Regional Council (WRC) mid-2020 via official information request. Specifically, I requested the resource consent document, staff report, application document, assessment of environmental effects (AEE) report, and monitoring reports for all consents on record pertaining to mangrove removal. To control for search effort and method, councils were requested to provide any consent document containing the keyword 'mangrove'. Documents received were individually assessed to ensure they pertained to mangrove removal.

2.1 MANGROVE REMOVAL DATA

Using resource consent documents, I extracted data pertaining to the location, extent, and shape of mangrove removal, mangrove removal method, mangrove removal type (the removal of mature plants, saplings, or seedlings), consent durations, and avifauna-relevant conditions. Where the explicit area of mangrove removal was not quantified, I used historical satellite imagery in Google Earth Pro (version 7.3.4.8248) to quantify the area of removal by digitally measuring sites prior to and after removal. All consents which granted removal of >1000m^2 of mature mangrove forest were classified as 'large removals', while <1000m^2 were classified as 'small removals'. To describe the shape of removal, removal shape types were adapted from Lundquist *et al.* (2017) and expanded upon to form a standardised reference set (Figure 3). Application documents were used to determine the applicant's rationale for removal. Staff reports were used to verify data collected from resource consent and application documents.

2.2 AVIFAUNA DATA

Collation of avifauna data was restricted to large removals and collected from associated AEE reports and avifauna monitoring reports. Data manually captured included report authors, bird groups assessed, monitoring methods, the perceived or actual effect of removal (categorised as positive, minor adverse, adverse, no effect, and not considered), and reasoning provided for effect judgements. To verify manual data capture, I cross-referenced our synthesis with keyword searches for all sentences or words containing phrases 'bird', 'banded rail', 'wader', 'monitor', 'fauna', or 'habitat' within reports using package *pdfsearch* (LeBeau 2019, v.0.3.0) in RStudio (RStudio Team 2020).

Where avifauna monitoring was undertaken both before and after mangrove removal (henceforth 'case studies'), I distinguished monitoring efforts between coastal birds (wading and shore birds which use open habitats), mangrove-using birds (excluding the banded rail), and banded rails. For banded rails, I captured temporal and spatial data footprint surveys from available maps and data tables in monitoring reports. I used paired sample *t*-tests to determine differences on a per site basis between (1) the number of footprints recorded prior to and after clearance, and (2) the number of footprints found within clearance areas or adjacent mangroves, both corrected for sampling effort.

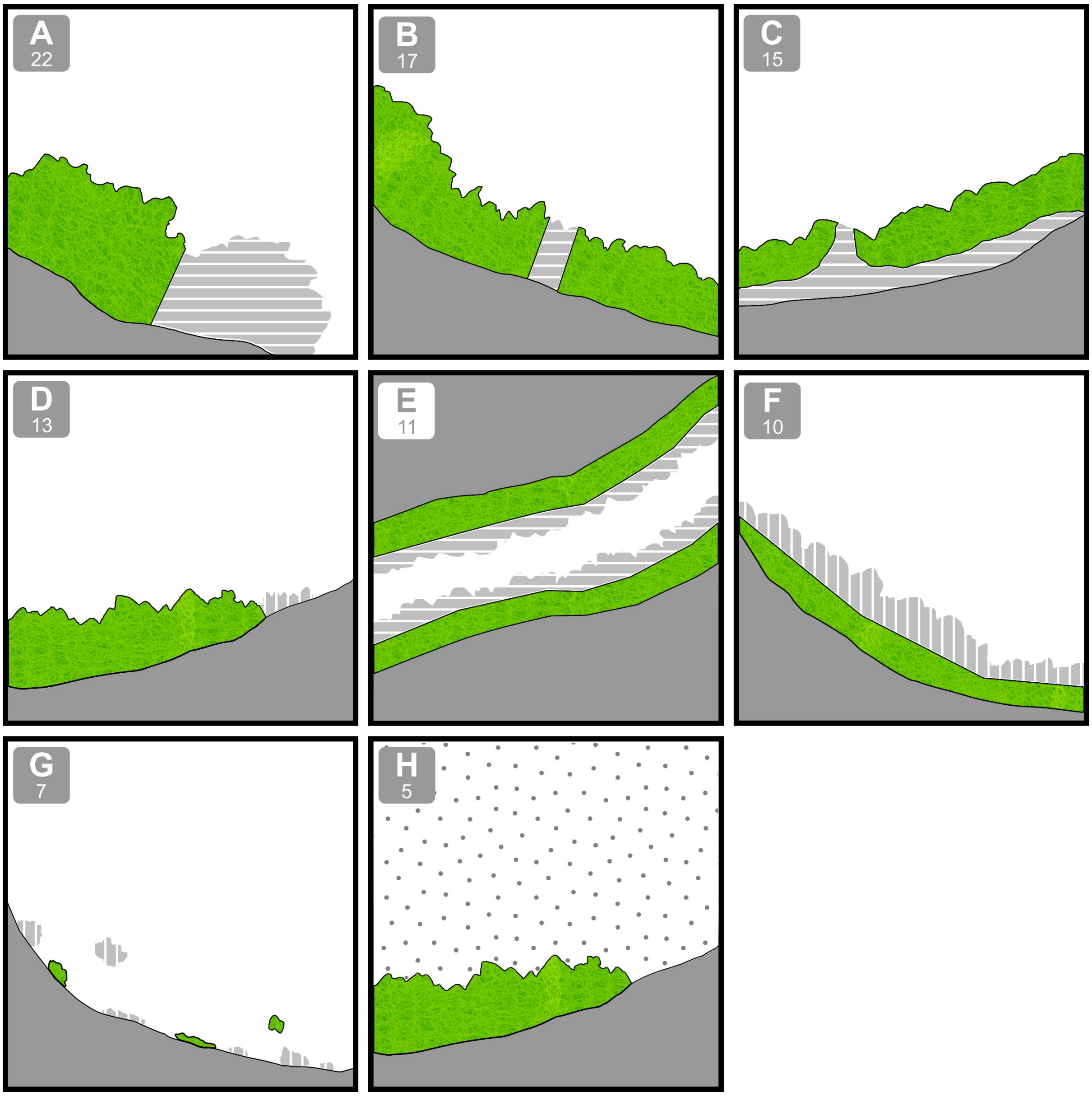


Figure 3: Mangrove removal clearance shapes (letters) ordered by their respective frequencies in the collated data set (numbers, indicating percentage): A = large-contiguous; B = pathway; C = inshore; D = small-contiguous; E = channel; F = seaward-strip; G = patchwork; H = seedling-only. Green areas indicate mangroves, grey-striped areas indicate mature mangrove removal, and grey dots indicate seedling removal. Clearance shapes adapted from Lundquist et al. (2017) and expanded upon.

3 RESULTS

3.1 MANGROVE REMOVAL

In total, 148 resource consents pertaining to mangrove removal were received from relevant regional councils: Auckland, Bay of Plenty, Northland, and Waikato (Table 2). We identified 161 discrete removal areas of mature mangrove removal totalling 333.0 hectares of mangrove forest cleared between 1994 and 2020 (Figure 2, Figure 4). On average, consents were granted for a duration of 17.8 years (SE ± 1.1, range 0.2–35). Eighty-eight percent of consents allowed for removal of mature mangrove plants, while just seven percent excluded mature mangroves and targeted only saplings or seedlings. The largest total area of mangrove removal granted by a single consent was 75 hectares, granted by Auckland Council in 2015. Conversely, the smallest area granted by a single consent was two square-metres, one of ten consents to remove ten square-meters of mangrove forest or less.

The most frequent type of adult mangrove clearance was large-contiguous clearing (22%) while the least frequent was seedling-only clearing (5%) (Figure 3). More than a third of all sites (37%) were cleared using hand-held machinery, 17% of sites were cleared using heavy machinery, while a combination of both methods was used for 12% of sites. Nearly a quarter of all resource consents did not specify a method for mangrove removal. Almost two-thirds of the consents required felled mangroves to be removed from the coastal marine area, while other methods included burning (10%), a combination of mulching and/or burning (6%), mulching only (1%) or tidal dispersion of chopped material (1%). Almost a quarter of consents did not stipulate a required disposal method for felled mangrove vegetation.

Our synthesis of community rationales found more than half of all large removals were motivated, at least in part, by community desire for improved coastal amenity and recreation values (Figure 5). Ecological restoration and bird conservation were cited as a rationale in 18% and 7% applications for large removals respectively, but neither were cited for smaller removals. The most frequent rationale provided for small removals was the maintenance or installation of infrastructure in the coastal marine area (26%), closely followed by road infrastructure (22%) and boating access (22%).

Table 2: An overview mangrove removal resource consents granted by four regional councils in Aotearoa New Zealand

	Auckland	Bay of Plenty	Northland	Waikato	Total
Total mangrove area, ha (% of total)	8,256 (31)	1,126 (4)	14,213 (54)	2,879 (11)	**26,474**
Consents granted (sites)	56 (56)	16 (16)	65 (65)	11 (24)	**148 (161)**
Consents by removal size					
0-500m2	24	2	32	2	**60**
500-1000m2	5	0	6	0	**11**
1000-10,000m2	13	0	14	4	**31**
>10,000m2	8	11	4	2	**25**
Total area removed, ha	151.3	109.1	27.0	45.6	**333.0**
Proportion removed, %	1.8	9.7	0.2	1.6	**1.3**
Average removal size, ha ± SE	3.2 ± 1.6	8.4 ± 1.9	0.5 ± 0.3	5.7 ± 3.6	-
Most frequent removal shape (n)	Large contiguous (14)	Seaward strip (10)	Pathway & Inshore (13)	Large contiguous & Juvenile (2)	-
Avg. consent duration, years ± SE	17 ± 2	11 ± 1.7	20 ± 1.6	17 ± 3.8	-

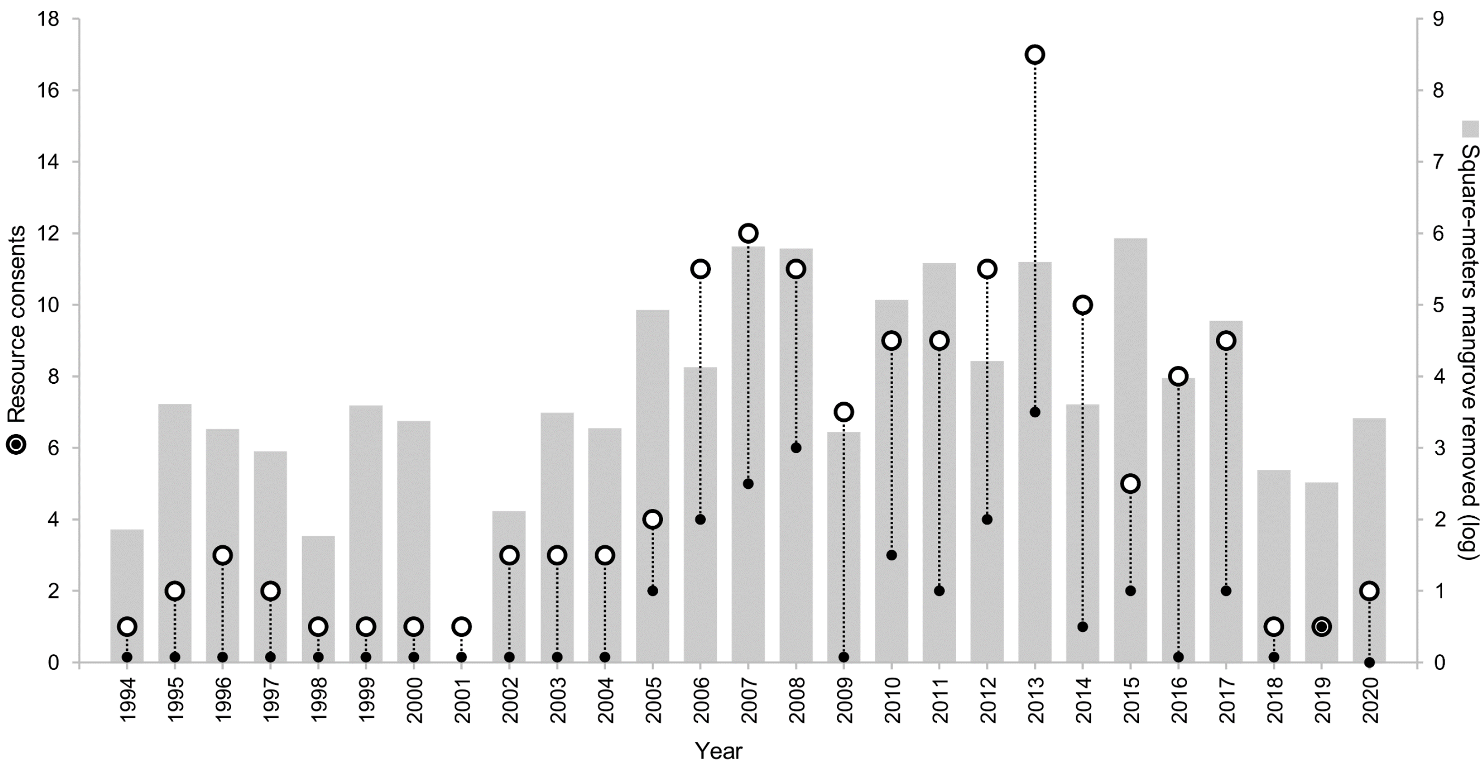


Figure 4: A comparison of resource consents granted annually (white circles) since 1994 with the amount of mangrove removal granted by consents in the same year (grey bars). Dotted lines indicate the disparity between all consents (white circles) and those containing avifauna-relevant stipulations (black circles)

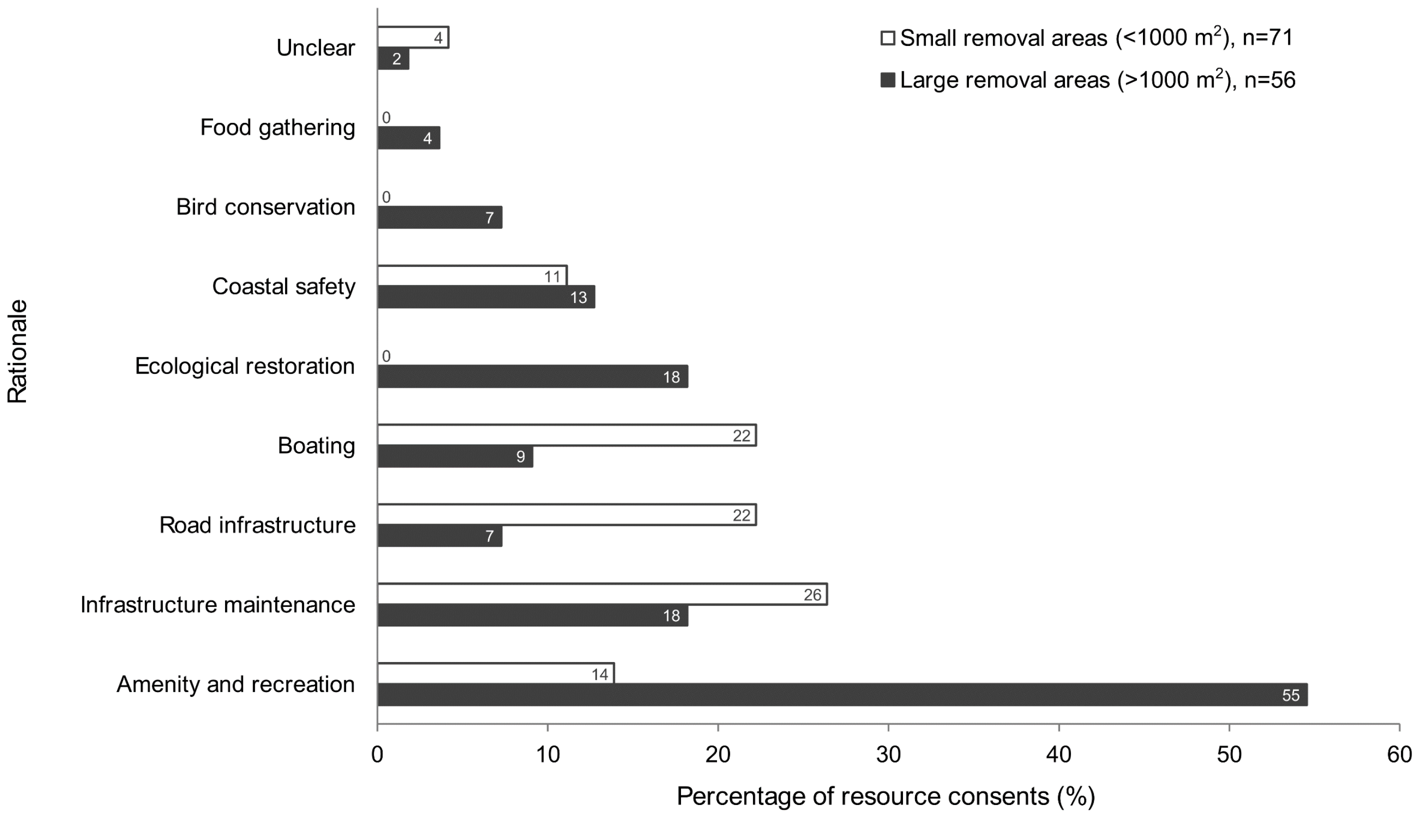


Figure 5: A summary of applicant rationales for mature mangrove removal among large and small removal sites, as determined by application documents for mangrove removal

3.2 AVIFAUNA

Although there has been a marked increase in the number of consents since 1994, the number of consents containing avifauna-specific conditions has been consistently lower than the total number of consents granted each year (Figure 4). Fewer than a third (27%) of all consents contained avifauna-specific conditions, of which fewer than half related to monitoring of avifauna populations either before or after mangrove removal. The most frequent avian-specific condition required that removal take place outside of the avian breeding season (21% of all consents), followed by the requirement for a post-removal bird survey (12%) and/or a pre-removal bird survey (7%).

For large consents (n=56), a total of 50 unique AEE reports were sourced. With respect to avifauna, 32% of reports did not consider birds at all, while the remaining 68% considered different bird groups with varying frequency. Of all reports, 48% considered banded rail specifically, 40% considered mangrove-using birds (often defined in reports as ‘marsh birds’), and 34% considered coastal birds. AEE assessments of mangrove-using birds and banded rails (n=44) suggested that mangrove removal would have no adverse effects on groups of avifauna in 29 of 44 cases. Reporters cited the small proportional losses of mangroves, a lack of mangrove dependency in birds, and the site-specific absence of banded rails as chief justifications for this assessment. Contrarily, 15 of 44 reports suggested adverse effects on mangrove-using avifauna or banded rails, citing a loss of foraging habitat in most cases. Coastal birds were considered to benefit from mangrove removal in 14 of 17 AEE reports, primarily due to the expected restoration of open habitats.

AEE reports were seldom informed by monitoring data; just 9 of 44 assessments of mangrove-using birds and banded rails used standardised, scientifically-recognised monitoring methods (henceforth 'formal surveys') such as footprint surveys, passive-acoustic surveys, playback surveys, census counts, or five-minute counts (Dowding 2012, Appendix Figure A1) to determine species presence or habitat use within mangroves. A further 12 assessments consisted of site visits and non-standardised visual surveys (henceforth 'informal surveys'), while the remaining 23 assessments provided no indication of avifauna surveys of any kind. For coastal birds, just 3 of 29 AEE reports used formal surveys to inform assessments.

Aside from monitoring undertaken for AEE reports, 13 large removals (2 WRC, 1 AC, 10 BOP) undertook formal avifauna monitoring before *and* after mangrove removal as part of consent conditions (case studies). For all removals, environmental consultants monitored banded rail (Appendix Table A1) across 16 discreet sites (Figure 6), while mangrove-using avifauna were monitored in Whangamata, and coastal avifauna in Pahurehure Inlet, Auckland. Although I identified further surveys of coastal and mangrove-using birds undertaken in Tauranga, these were community-led and resultant data were patchy, not standardised among groups, nor formally collated. Thus, insights into the effects of mangrove removal on coastal and mangrove-using birds (excluding banded rail) are restricted to Whangamata and Pahurehure case studies.

Case studies indicated minor adverse effects of removal on banded rail in nine of the sixteen sites (Tauranga - Rowson 2012), and no adverse effects in four sites (Whangamata - Richardson *et al.* 2019; Tairua - Wium *et al.* 2019), while reporting from the remaining three sites in Pahurehure Inlet did not qualify the degree of adverse effects on banded rail (Don 2015) (Appendix Table A1). Banded rails remained present in all sites which employed seaward strip clearing (13 of 16 sites) although banded rail presence was principally restricted to remaining mangrove stands. Correcting for survey effort across these sites, analysis indicated significantly fewer footprints in post-removal surveys compared with pre-removal surveys, ($P < 0.05$, $t = 2.39$, $df = 12$) and significantly fewer footprints in removal areas than in mangroves directly adjacent to removal areas ($P < 0.01$, $t = 3.77$, $df = 12$).

Remaining case study sites (3 of 16) were large-contiguous clearances in Pahurehure Inlet, where pre-removal surveys estimated that there had been 10 banded rail pairs across throughout sites in the Inlet (Southey 2009). Post-removal surveys observed five banded rail individuals using cleared areas adjacent to rush-saltmarsh in two of the three sites ('north' and 'south') but no banded rails in the eastern site (ca. 11 ha), despite footprint evidence of their presence prior to removal.

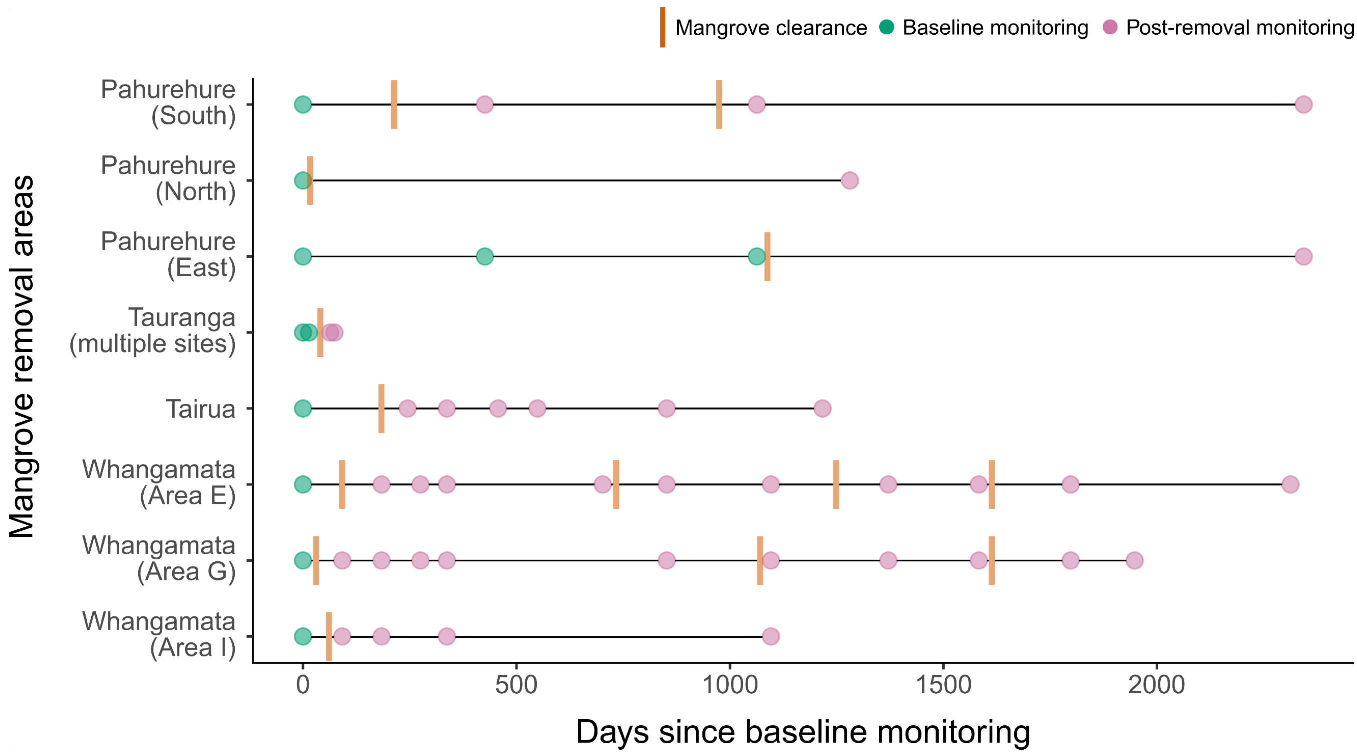


Figure 6: The timing and frequency of banded rail monitoring surveys (circles relative to mangrove clearance events (vertical orange bars) at case study sites. Green circles surveys prior to any mangrove clearance (baseline monitoring), while pink circles indicate surveys after clearance events (post-removal monitoring).

With respect to coastal birds, formal surveys in Pahurehure Inlet, Papakura (Don 2015) found increases in both the abundance and diversity of coastal bird species after clearance of some 27 hectares of mangrove forest, with species richness increasing from 16 species before clearance to 21 species after clearance. Notable species recorded only after mangrove clearance were kuaka (bar-tailed godwits; *Limosa lapponica*), little egret (*Egretta garzetta*), and kōtuku ngutupapa (royal spoonbill; *Platalea regia*).

As for coastal birds, just one set of surveys formally assessed the response of mangrove-using birds to mangrove removal. Repeated five-minute point counts, undertaken in Whangamata (Richardson *et al.* 2019), documented 33 bird species in mangrove habitats, 19 indigenous and 13 exotic. Frequently-observed indigenous species included Silvereye (tauhou; *Zosterops lateralis lateralis*), welcome swallow (*Hirundo neoxena neoxena*), kotare (New Zealand kingfisher; *Todiramphus sanctus vagans*), pīwakawaka (North Island fantail; *Rhipidura fuliginosa placabilis*), pūkeko (*Porphyrio melanotus melanotus),* riroriro (grey warbler; *Gerygone igata*) and tūī (*Prosthemadera novaeseelandiae novaeseelandiae*), while 'threatened' or 'at risk' species detected during counts included matuku

(Australasian bittern; *Botaurus poiciloptilus*), mātātā (North Island fernbird; *Bowdleria punctata vealeae*), moho pererū (banded rail; *Gallirallus philippensis assimilis*), and tarāpunga (red-billed gull; *Larus novaehollandiae*). For most species, relative abundance and/or frequency of occurrence before and after mangrove clearance remained stable or increased; a pattern attributed to fewer pre-clearance surveys and increasing sampling effort over time. For seven species - all classified as either Not Threatened or Introduced (Robertson *et al.* 2017) – declines were observed but attributed to differences in sampling effort and seasonal timing rather than removal effects. Importantly, all monitored sites retained substantial areas of mangrove post-removal, given clearance in Whangamata was a seaward-strip shape (Figure 3).

4 DISCUSSION

The seaward expansion of mangrove forests in Aotearoa has fuelled community demand for mangrove removal, a process controlled by regional councils using resource consents. In collating data from these consents, this study sheds light on mangrove removal itself as well as its potential effects on avifauna. This study is the first to quantify the total spatial extent of legal mangrove removal in Aotearoa: 330 hectares granted by 148 resource consents since 1994. While the number of consents for mangrove removal has seen notable increases in the last two decades, the number of consents containing avifauna-relevant conditions has been consistently lower.

Fewer than a third of all resource consents accounted for avifauna among consent conditions, and fewer still assessed avifauna before or after mangrove removal. Most of these assessments failed to use scientifically recognised monitoring methods, although thirteen case studies were identified where formal monitoring of avifauna was undertaken before and after removal. Monitoring of removal sites where large areas of mangroves were retained (seaward-strip sites) indicated few minor adverse effects on mangrove-using avifauna, although banded rail abundance appeared to decline after removal. Conversely, removals of large contiguous areas of mangrove (large-contiguous sites) appeared to benefit coastal birds, but likely lead to banded rail displacement and/or decline. Ultimately however, a lack of monitoring and targeted scientific research continues to undermine our understanding of the effects of mangrove removal on native avifauna in Aotearoa.

4.1 MANGROVE REMOVAL

The total amount of mangrove removal documented, 330 hectares, represents 2.6% of the total mangrove area in Aotearoa (Table 2). Almost a third of removal was within BOPRC jurisdiction, despite mangroves in the region accounting for only 4% all of Aotearoa's total mangrove habitat (Table 2). Conversely, less than 8% of mangrove removal was within Northland, despite the region containing 54% of all of Aotearoa's mangrove forest (Table 2). While the high density of people, mangroves, and removals in Tauranga (BOPRC) likely explains this difference, this reasoning does not hold true for the highly populated Auckland region, where the area of mangroves removed was proportionate to the area of mangroves that the region contains (Table 2).

Our study suggests that mangrove removal projects are unlikely to meet restoration objectives in many instances. While mangrove clearance will create open habitats by definition, the likelihood of regaining sandflats is context-specific and dependent on removal conditions; removal sites are more likely to recover where non-mechanical removal techniques are used, smaller areas are cleared, and mangrove cuttings are removed from the coastline (Lundquist *et al.* 2014a). However, these conditions were not consistently met; mechanical machinery was commonly used to remove mangroves, particularly among large removal sites. While just 7% of clearances of sites allowed for mangrove disposal via mulching, this covered a disproportionately large area of ca. 130 hectares in Auckland and Bay of Plenty. In Pahurehure Inlet and Tauranga, mulched sites triggered macroalgal blooms and decreased oxygen concentration in the water column (Lundquist *et al.* 2014a; Stokes *et al.* 2016). Even though smaller removal areas can exhibit faster trends towards recovery – i.e., loss of muddy sediments – this process can take decades, and is altogether unlikely in sheltered locations not exposed to strong coastal erosion (Lundquist *et al.* 2014a). Thus, while community desires for more open habitats may be met by mangrove removal, these are mostly restricted to improved visual amenity, but are unlikely to extend to the ecological or recreational benefits associated with sandy coastal habitats.

While this review has documented removal projects throughout mangroves' distribution, it is highly unlikely that it captures the full extent of mangrove removal in Aotearoa. Firstly, several sources note the occurrence of illegal mangrove in Aotearoa (Morrisey *et al.* 2007; Lundquist *et al.* 2014b) which may account for more than 50% of all removal in the country (pers. comm. C. J. Lundquist). Secondly, prior to 1994, mangroves were not protected under national policy and were removed without resource consent. Thirdly, although our study controlled for search effort when sourcing data from regional councils, it is possible that our final dataset does not include all mangrove removal on record. Thus, it is not unrealistic to suggest that at >650 hectares of mangrove forests have been removed since 1994, at a rate of 25 hectares per year. This rate is likely substantially lower than mangrove expansion over the same period (Suyadi *et al.* 2019), and barring substantial reductions in sediment supply, mangroves are expected to continue their rapid expansion (McBride *et al.* 2016). In turn, mangrove removals are likely to continue, making an understanding of their ecological ramifications key to effective management practices (Stokes *et al.* 2016).

4.2 AVIFAUNA MONITORING

Despite more than 25 years of regulated mangrove removal in Aotearoa, this review found sparse evidence to advance our understanding of mangrove removal effects on avifauna. Fewer than half of all AEE reports considered mangrove-using birds or banded rails in their evaluation of removal effects, while fewer still assessed coastal birds. AEE reports adjudged impacts of removal on mangrove-using birds (including banded rail) to have no effect in 63% of assessments, but seldom based assessments on data collected from site-specific formal surveys, instead relying on references to limited literature, informal surveys, or educated guesswork.

Arguably, AEE reporting for mangrove removals is predisposed to provide limited insight. Firstly, AEEs take place in advance of removals and thus effects judgements are always pre-conceived, rather than retrospective. Secondly, within the current policy framework, AEE reporting is applicant-led rather than authority-led. Thus, the quality of an AEE is determined by the applicant, although regional councils can vet the standard of AEE reporting. Thirdly, the cost to benefit ratio for informative AEE reporting may be prohibitive; the costs of a monitoring removal areas (including abiotic changes, macrofauna communities, and avifauna) can reach $125,000 NZD for harbour-wide removal operations (Lundquist *et al.* 2017). Nevertheless, our review of large consents indicated that AEE reporting was typically outsourced to environmental consultancies. While these yielded some formal monitoring reports, the absence of baseline data in many instances reveals a notable shortcoming in pre-removal assessments of avifauna.

4.2.1 CASE STUDIES

Using birds as indicators of the success of restoration success of mangrove removal requires substantial efforts to elucidate spatial and temporal (intra- and inter-annual) patterns of habitat use (Stokes *et al.* 2016). While unlikely to meet these standards, the case studies identified by this study represent the sole efforts to quantify mangrove removal effects on avifauna to date in Aotearoa. Data collated from case studies were almost exclusively limited to banded rail (Appendix Table A1), although coastal birds and other mangrove-using birds were assessed on occasion.

While coastal and mangrove-using birds were poorly represented in case studies, two reports (Don 2015; Richardson *et al.* 2019) have undeniably advanced our understanding of avifauna responses to mangrove removal. In Whangamata, the removal of seaward strips of mangrove forest had few adverse effects on mangrove-using birds (Richardson *et al.* 2019). Importantly, this monitoring recorded 33 avifauna in mangrove habitats of Whangamata, a much-needed update to similar surveys by Cox in 1977 (23 species in Kaipara Harbour). In Pahurehure Inlet , large-scale contiguous clearing of mangroves coincided with an increase in abundance and species richness of wading birds (Don 2015). Given the near-complete clearance of mangroves in the inlet, this likely impacted negatively upon mangrove-using birds (although this was not measured).

Four mangrove removal consents - located in 16 discrete sites across Tauranga (Rowson 2012), Tairua (Wium *et al.* 2019), Whangamata (Richardson *et al.* 2019), and Auckland (Don 2015) – assessed banded rail footprints (Appendix Table A1), concluding that removal had minor or no adverse effects on banded rail populations, given the continued presence of footprints before and after removal in all but one site. While banded rails remained present in almost all sites, our analysis of this footprint data suggested that banded rail made little use of cleared areas and potentially occur in lower numbers after removal has taken place. However, these findings are inconclusive; reports from Tauranga indicated that cleared areas scarcely retained footprints given excess mulched material, while survey effort of cleared areas was inconsistent among sites. Nevertheless, it is possible that seaward-strip removal of mangroves could reduce the density of local banded rail populations given the reduced availability of foraging area, however further formal study is required to test this hypothesis.

The only site in which banded rail was not observed in post-removal surveys was following the large-contiguous clearance of mangroves in the eastern section of Pahurehure Inlet (Don 2015). Additionally, further evidence from Pahurehure Inlet suggests a decline in the local population of banded rail following large-scale contiguous mangrove removal; pre-removal estimates suggested ten banded rail pairs were found throughout the area (Southey 2009), while post-removal estimates documented five individuals in the same area, an apparent population decline of seventy-five percent. However, this estimate should be viewed with caution; it is difficult to accurately estimate abundance for cryptic species (Suwanrat *et al.* 2015) and estimates based on footprint surveys are not recognised as a strong proxy for abundance (Dowding 2012).

Based on a limited number of examples, it appears that large-contiguous removals have larger adverse effects on banded rails and other mangrove-using avifauna than seaward-strip removals. This finding aligns with mangrove removal recommendations by Lundquist *et al.* (2017), who indicated that seaward-strip removals (which made up 10% of all documented sites in this study) should be prioritised over large-contiguous removals (22% of all sites) as they are more likely to result in positive restoration trends. Conclusions from the avifauna case studies highlighted by this study provide further evidence for this recommendation, but further research is required to establish firm conclusions.

4.3 TOWARDS HOLISTIC MANGROVE MANAGEMENT

4.3.1 IMPROVING AVIFAUNA MONITORING

Consent-related monitoring of avifauna had two notable shortcomings: the use of informal survey methods, and a lack standardised monitoring among removal. To address the former shortcoming, monitoring should use habitat- or species-specific methodologies (Appendix Figure A1) to target relevant avifauna groups (Dowding 2012). Based on successful monitoring in Tauranga and Whangamata, the following survey methods are proposed: census counts for coastal avifauna in open habitats (as per Don 2015), five-minute counts for mangrove-using avifauna (as per Richardson *et al.* 2019), footprint surveys and call counts for banded rail (as per Richardson *et al.* 2019, but see below for methodological improvements), and species-specific playback call counts for cryptic marsh birds (e.g. Williams 2016). In order to compare the results of such monitoring among sites, the methods, frequency, and timing of the avifauna surveys should be standardised (Stokes *et al.* 2016).

In addition to general monitoring protocol, there is scope for several improvements to banded rail monitoring specifically (see Appendix Table A2). Firstly, the timing of footprint surveys should be standardised relative to low tide to allow for inter-site comparisons. For example, starting footprint surveys at low tide represents a standard time which likely improves the detection of footprints given banded rails forage extensively in mangroves during the hours around low tide (de Satgé, unpubl. data). Secondly, when looking for banded rail footprints, observers should note substrate conditions along transects (e.g., wet, soft, hard, dry) as this affects footprint detectability (Elliot 1987). Thirdly, transect locations should stay consistent throughout the monitoring process; transects should target mangrove stands, their seaward edge, and any cleared areas to allow for explicit comparisons between habitat zones. Finally, acoustic surveys of banded rail calls can complement footprint surveys and are most effective during breeding season (September-November) during crepuscular hours (Beauchamp 2015, de Satgé unpubl. data).

4.3.2 MONITORING AS A MANAGEMENT TOOL

Monitoring is key to effective adaptive management (Schreiber *et al.* 2004), a practice which incorporates research into action (Salafsky & Margoluis 2003) by treating management actions as experiments to monitor and learn from (West *et al.* 2019). Adaptive management connects monitoring design and decision structure (Lyons *et al.* 2008), and is a practice which should underpin mangrove management strategies in Aotearoa. Auckland Council cites adaptive management as an explicit objective for mangrove management (Auckland Council 2019), and case studies in Papakura (AC) Tauranga (BOPRC), and Whangamata (WRC) all exhibited elements of an adaptive management framework, included the use of staged decision-making, the use of control areas, and monitoring-informed removal timelines. Drawing on these examples, a conceptual framework is proposed for incorporating avifauna monitoring into mangrove management projects (Figure 7).

While this review has focused on avifauna monitoring, a variety of abiotic and biotic factors should be considered to inform mangrove management decision-making. Stokes *et al.* (2016) argue that monitoring multiple parameters – physical, chemical, and biological – is key to meeting mangrove management objectives in a cost-effect and site-specific manner. Moreover, explicitly linking different monitored parameters can provide insight into how different ecological factors interact and respond to different management practices. For example, monitoring of oxygen concentrations in sediment and macrofauna communities has indicated that *in situ* mulching to dispose of mangroves is likely to have substantial adverse ecological effects within mangrove removal sites (Lundquist *et al.* 2012).

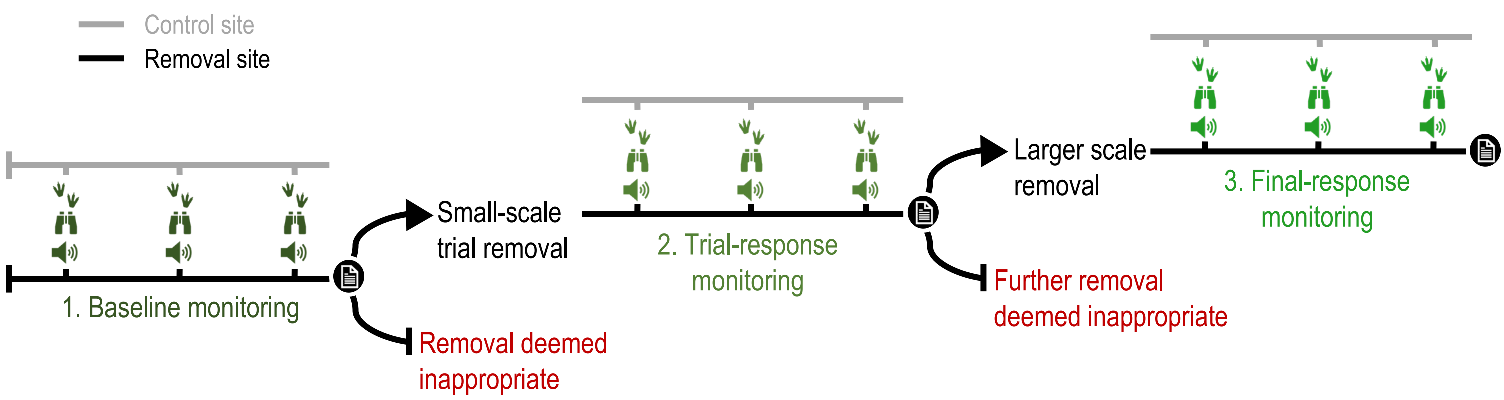


Figure 7: Incorporating repeated, stage-based avian monitoring into existing adaptive management frameworks pertaining to mangrove removal sites and control sites. Green symbols indicate monitoring types: footprint symbol = footprint surveys for ground-feeding birds; binoculars symbol = visual surveys such as five-minute point counts or census surveys for coastal birds; speaker symbol = playback/call surveys for cryptic marsh birds. Black circles containing document symbols indicate review points, where monitoring data is collated, analysed, and communicated to stakeholders to inform the dialogue for ongoing management decisions. Monitoring repeats are indicative, see Stokes et al. (2016) for detailed schedules of various monitoring objectives.

4.3.3 HOLISTIC THINKING

Mangrove management has much to gain by reflecting the tenets of mātauranga Māori (henceforth mātauranga), the continuum of distinct knowledge with Polynesian origins which includes Māori worldview, values, culture and cultural practice (Hikuroa 2017; Clapcott *et al.* 2018). An understanding that the natural world is comprised of multiple, co-dependent parts is a fundamental principle within mātauranga (Harmsworth & Awatere 2013; Salmond *et al.* 2019), while the concept of *taiao* (environment) recognises the natural world as one interconnected system, whose parts respond to and rely on one another and are best interpreted as a whole (Hikuroa 2021), a concept with strong parallels to earth system science (Wilkinson *et al.* 2020). Giving voice to mātauranga and *tikanga* (Māori customs and lore) in policy by acknowledging that people, plants, animals and waterways are inextricably linked, is key transcending fragmented thinking in current freshwater policy (Salmond *et al.* 2019; Taylor *et al.* 2020).

Contemporary mangrove management practices are largely focused on mangrove plants themselves without accounting for their broader context, including underlying drivers of mangrove expansion – human-induced increases in sediment and nutrients in waterways – which operate at catchment-level spatial scales. Scaling mangrove policy and management to reflect catchment processes is key to restoring ecosystem health at multiple spatial scales (Peacock *et al.* 2012; Lundquist *et al.* 2014b; Swales *et al.* 2015). This viewpoint is gaining traction in policy; both AC and BOPRC regional coastal plans explicitly reference catchment-wide approaches to mangrove management (Auckland Council 2019;

Bay of Plenty Regional Council 2019), while the recent *Managing our estuaries report* (Parliamentary Commissioner for the Environment 2020) calls for the integration of catchments at a policy level.

Unfortunately, a complex statutory framework has hindered the development of mangrove policy which reflects catchment-wide processes. Currently, estuaries are managed under a separate management framework to freshwater systems, despite their physical interconnection (Parliamentary Commissioner for the Environment 2020). Moreover, estuaries are subject to at least eight separate statutory frameworks[1], ranging from the 1991 Resource Management Act at a national level to district plans at a local level (Parliamentary Commissioner for the Environment 2020).

A multitude of coastal and freshwater policies can represent conflicting interests; along coastlines the rights of communities to coastal access and recreation are pitted against the protection of indigenous vegetation and valuable habitats, a tension which mangrove expansion has brought into sharp relief. National policy has the potential to address such conflicts and support the adaptive management of socio-ecological systems (Frohlich *et al.* 2018; Waylen *et al.* 2019). By framing of mangrove policies to focus on catchment processes, policymakers have an opportunity to treat estuaries and their catchments as a single identity and ensure management decisions and both wide and enduring (Parliamentary Commissioner for the Environment 2020). While new draft legislation for resource management, the Natural and Built Environments Act, proposes "the adoption of mātauranga Māori, including integrated management of natural and cultural resources such as biosystems, water, urban areas and climate" (Resource Management Review Panel 2020, p.103), it remains to be seen how such policies will play out in practice.

4.4 AVIFAUNA KNOWLEDGE GAPS

It is evident that our understanding of mangrove-avifauna is in its infancy. Report findings highlight a lack of adequate avifauna monitoring associated with mangrove removals, particularly for large removals. In this respect, our study has quantified the depth of the mangrove-avifauna knowledge gap, rather than filled this gap. Limited case studies suggest that mangrove removal is likely to benefit coastal birds but may adversely affect banded rail populations. However, these insights are context specific, and the implementation of standardised monitoring protocols is needed to deepen this evidence and lead to improve management practices.

[1] Statutory frameworks, as per PCE 2020 report: Resource Management Act 1991, Draft National Policy Statement for Indigenous Biodiversity, National Policy Statement for Freshwater Management 2020, New Zealand Coastal Policy Statement 2010, regional policy statements, regional plans, regional coastal plans, and district plans

Before we can understand the effects of mangrove removal on birds, we need to determine the value of mangroves to avifauna. This requires detailed study of the spatial and temporal (inter- and intra-annual) patterns of avifauna habitat use (Stokes *et al.* 2016) over long time frames. Studies of avifauna in Aotearoa are limited to banded rails (Botha 2011; Beauchamp 2015), a specific harbour (Cox 1977), or a part of broader biodiversity assessments (Dencer-Brown *et al.* 2020). To our knowledge, mangrove habitat use by other threatened or at risk avifauna other than the banded rail has not been studied (but see a summary of anecdotal evidence by Bell & Blayney 2017b).

Bell & Blayney (2017b, p.2) identify several research priorities to adequately determine the distribution and abundance of threatened and at-risk birds within mangroves, including:

1. A presence/non-detection inventory of bird distribution within mangrove habitat and adjoining areas.
2. The modification and development monitoring techniques for the monitoring of birds to create a standard protocol that includes the ability to provide estimates of density.
3. Surveys across mangrove sites in Aotearoa to determine how density and carrying capacity are impacted by different habitats and by their connectivity.
4. Research and monitoring on the effects of mangrove removal on bird populations in the short and long term, including measures of density, survival, fecundity, site occupancy, use of mangrove removal areas and migration.
5. Research into life history factors that may contribute to the impact of mangrove removal on birds such as breeding patterns, parental care, and moult regimes.
6. A comparison of the abundance and habitat use distribution patterns of mangrove-using birds across Aotearoa, including perspective on different habitats birds are using, how habitat patch size influences habitat use, predation pressures, and the effects of habitat fragmentation or connectivity

4.5 CONCLUSIONS AND RECOMMENDATIONS

This report has documented substantial legal mangrove removal in Aotearoa New Zealand since 1994, totalling 330 hectares. Although this is likely an underestimation of mangrove removal in Aotearoa, mangrove expansion over the same period has likely far outstripped mangroves lost to removals.

Monitoring data from resource consents has indicated that mangroves provide suitable habitat to a range of avian species, including several at-risk or threatened birds. Mangrove management practices which remove seaward-strips of mangrove appear to have relatively few minor adverse effects on mangrove-using avifauna, although local populations of banded rail may decline after such removals as they lose foraging grounds. Similarly, removals of large contiguous areas of mangrove may cause adverse effects on mangrove-using avifauna and banded rails but have been shown to benefit coastal avifauna such as wading birds.

While the number of consents for mangrove removal issued has increased in the last two decades, the number of consents which require accounting for, and monitoring of, avifauna is consistently lower. This fact, combined with a lack of targeted scientific research, continues to hinder our understanding of the effects of mangrove removal on native avifauna.

In assessing the effects of mangrove removal on avifauna, this review has highlighted several shortcomings in mangrove management practices relating to monitoring, adaptive management, holistic policy, and prioritising restoration goals (see section Policy relevance). As such, the following changes are proposed to mangrove management practice and policy:

1. Implement standardised, species- and habitat-appropriate monitoring of fauna for all large-scale mangrove removal projects
2. Develop an adaptive management framework for large-scale mangrove removals which incorporates monitoring, removal trials, and control sites to inform a stepwise, evidence-based management process
3. Refocus policy to emphasise the drivers of mangrove expansion (sedimentation, eutrophication, and hydrodynamic conditions) rather than mangroves themselves. Similarly, reflecting this shift in thinking in coastal information and education programmes could drive changes in community perspectives.
4. Prioritise measures and methods which are most likely to lead to improved restoration success of mangrove removals, as informed by available evidence

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6 APPENDIX

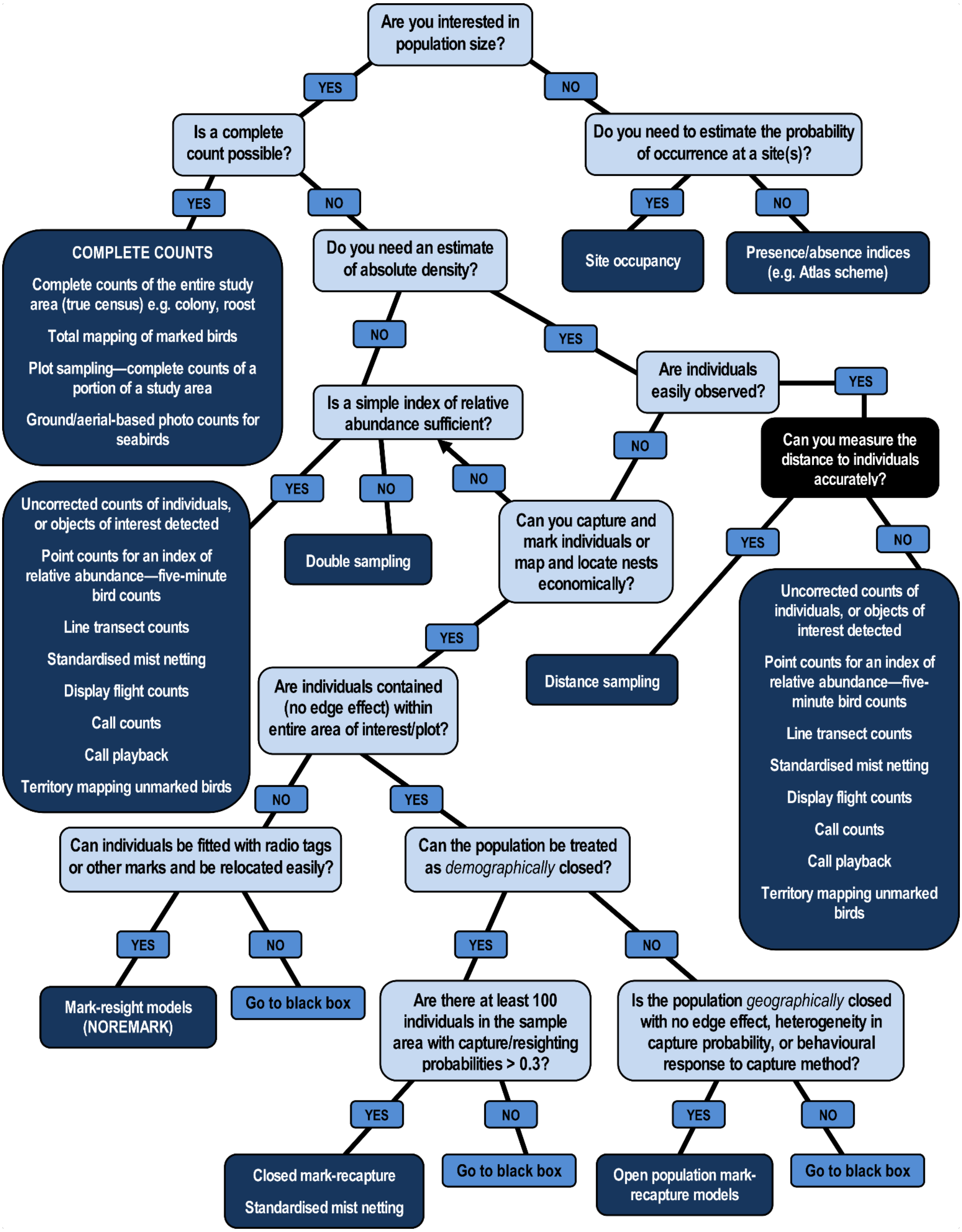

Figure A1: The bird methods decision tree, as per Dowding (2012) in the Department of Conservation's Introduction to Bird Monitoring report, DOCDM-578317.

Table A1: Summary of case study findings on the effects of mangrove removal on banded rail

Council	Consent	Location	Removal area (ha)	Clearance shape	Survey type methods	Pre surveys	Post surveys	Reported effect of removal	BR presence post-removal	BR use of removal area	Reporter observations[1]
Auckland	35053	Pahurehure (North)	9.9	Large contiguous	Footprint	1	1	Effect not stated	Yes, adjacent to rush-marshes	Frequent	BR not observed in eastern half of the site post clearance
Auckland	35053	Pahurehure (South)	6.0	Large contiguous	Footprint	1	3	Effect not stated	Yes, adjacent to rush-marshes	Frequent	BR not observed in eastern half of the site post clearance
Auckland	35053	Pahurehure (East)	10.9	Large contiguous	Footprint	3	1	Effect not stated	Yes, adjacent to rush-marshes	Frequent	BR not observed in site post clearance
Bay of Plenty	65219	Matua	2.0	Seaward strip	Footprint	2	2	Minor negative effect	Yes, in adjacent mangrove	Highly limited	Evidence of reduced BR foraging range
Bay of Plenty	65220	Waikareao	6.5	Seaward strip	Footprint	2	2	Minor negative effect	Yes, in adjacent mangrove	Highly limited	Evidence of reduced BR foraging range
Bay of Plenty	64912	Wainui	12.4	Seaward strip	Footprint	1	1	Minor negative effect	Yes, in adjacent mangrove	Highly limited	Evidence of reduced BR foraging range
Bay of Plenty	64546	Te Puna	16.9	Seaward strip	Footprint	2	2	Minor negative effect	Yes, in adjacent mangrove	Highly limited	Evidence of reduced BR foraging range
Bay of Plenty	65389	Welcome Bay	7.4	Seaward strip	Footprint	2	2	Minor negative effect	Yes, in adjacent mangrove	Highly limited	Evidence of reduced BR foraging range
Bay of Plenty	62776	Waikaraka	8.0	Seaward strip	Footprint	2	2	Minor negative effect	Yes, in adjacent mangrove	Highly limited	Evidence of reduced BR foraging range
Bay of Plenty	63941	Athenree-Tanners Point	24.2	Seaward strip	Footprint	2	2	Minor negative effect	Yes, in adjacent mangrove	Highly limited	Evidence of reduced BR foraging range
Bay of Plenty	64154	Uretara	11.9	Seaward strip	Footprint	2	2	Minor negative effect	Yes, in adjacent mangrove	Highly limited	Evidence of reduced BR foraging range
Bay of Plenty	65026	Omokoroa	12.3	Seaward strip	Footprint	2	2	Minor negative effect	Yes, in adjacent mangrove	Highly limited	Evidence of reduced BR foraging range
Waikato	122986	Whangamata (Area I)	0.1	Seaward strip	Footprint, acoustic	1	4	No adverse effect	Yes, in adjacent mangrove	None	Evidence of reduced BR presence post clearance
Waikato	122986	Whangamata (Area G)	3.2	Seaward strip	Footprint, acoustic	1	10	No adverse effect	Yes, in adjacent mangrove	None	-
Waikato	122986	Whangamata (Area E)	2.9	Seaward strip	Footprint, acoustic	1	12	No adverse effect	Yes, in adjacent mangrove	Highly limited	-
Waikato	125983	Tairua (Oturu)	21.8	Seaward strip	Footprint, acoustic	1	6	No adverse effect	Yes, in adjacent mangrove	None	Limited BR use of site generally

[1]Reporter for Auckland = Bioresearches Consultancy and Ian Southey; Bay of Plenty = Bob Mankelow; Waikato = Wildland Consultancy

Table A2: Recommendations for establishing standardised banded rail (BR) footprint surveys which enable consistent temporal and spatial comparisons of banded rail presence or absence. Recommendations are based on available research, including preliminary research findings (de Satgé, unpubl. data) at time of writing

Recommendation	Description
Sampling method	Choose a scientifically recognised sampling method for finding and counting footprints and maintain this method over the course of the study. Listed below are two options:
A) Transects	Establish transects of equal length within mangroves, along their outer edge, and on adjacent mudflats or within a mangrove removal area. Record all footprints within 1 meter on each side of the transect line. It is important to keep the location of these transects consistent over time to allow for temporal comparisons. Transects along mangroves' outer edges should be adjusted accordingly as/if mangroves are removed. The inclusion of a transect within mangroves is important; recent research has indicated that BR footprints are more likely to be found within mangrove stands than along their outer edge.
B) Stratified-random	Allocate sampling points randomly within mangroves and adjacent mudflats or removal areas. Ensure that the number of sampling points allocated within each habitat is proportional to the habitat size or area of habitat sampled. At or near each randomly allocated sampling point, conduct a search for footprints within a 2x2 meter square quadrat.
Substrate condition	Note the condition or quality of substrate when sampling as the ability of substrate to retain footprints affects their detectability. Create a scoring system for substrate quality, for example, 2 = excellent (substrate retains full footprint), 1 = good (retains partial or faint footprints), 0 = poor (substrate retains little or no footprints as it is too hard/soft/wet). For transect sampling, provide a score of substrate quality at set intervals (e.g., every 5-10 meters). For stratified-random sampling, provide a score for each sampling point. Where possible, select for good and excellent substrates when sampling.
Tide timing	Start searches for footprints at low tide. This allows time between high tides for banded rails to leave footprints behind and time for observers to find prints before the tide comes in again.
Time of day	Note the time of day the survey is undertaken. While it is preferable to keep this standardised over the course of the study, tide timing should be prioritised.
Weather	Avoid performing footprint surveys during or after periods of heavy rainfall. This is likely to degrade substrate quality and in turn decrease the detectability of banded rail footprints.
Observer	Keep a record of the observer undertaking the survey. Different observers may identify footprints inconsistently and it is important to account for this bias.
Correct footprint identification	Accurately determine which footprints belong to BR. As per Elliot (1983), BR footprints are between 36 and 47 mm long and can be confused with footprints from oystercatchers, spur-winged plovers, pied stilts, crakes and young wekas. However, several key differences make BR prints distinguishable from these species. The footprints of waders (oystercatchers, spur-winged plovers, and pied stilts) are more asymmetrical than those of BR, whose prints are mostly symmetrical. Crake prints are usually smaller than BR, although the largest crakes' prints may overlap with the smallest BR prints. Young wekas have similar prints to BR but are likely to be accompanied by adult wekas and their much larger prints. For further guidance on footprint identification on a variety of coastal bird species, see https://www.nztracker.org/.