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Standardised methods to report changes in the ecological integrity of sites managed by regional councils

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Standardised methods to report changes in the ecological integrity of sites managed by regional councils

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Summary

Project and client

- The project is to recommend ground-based methods to monitor biodiversity, and therefore ecological integrity, in response to the management of sites of interest to regional councils throughout New Zealand.
- Hawke's Bay Regional Council led the project on behalf of the Regional Councils' Biodiversity Working Group, and Manaaki Whenua – Landcare Research and Kevin Collins (Collins Consulting) undertook the work.

Objectives

- The objective was to develop a harmonised approach to ground-based monitoring across all regional councils to determine management effectiveness on prioritised ecosystem sites. This is necessary because current efforts are not systematic and often idiosyncratic, which limits the capacity to build a compelling evidence base for reporting, and the capacity to aggregate data across sites. With a standardised approach, greater benefit could be achieved from the current level of investment in monitoring.

Methods

- We aggregated the 152 ecosystems designated in a published schema into seven broad classes for the purposes of monitoring, after discussion with members of the Biodiversity Working Group (BDWG).
- We evaluated point-based methods suitable for measuring change in the ecological integrity of prioritised ecosystems, especially in response to management, and made recommendations for methods to use in the broad ecosystem classes based on discussions and three workshops with members of the BDWG and participants from the Department of Conservation (DOC).
- We sought to standardise approaches across the seven broad ecosystem classes to the greatest extent possible, both to allow for comparability and to recognise that some prioritised sites contain one or more ecotones across broad ecosystem classes. We placed emphasis on well-established methods that have a history of results in peer-reviewed publications.

Results

We developed the following recommendations for ground-based methods to monitor biodiversity in response to the management of sites of interest to regional councils throughout New Zealand

- Use square, fixed-area plots to assess vegetation across all broad ecosystem classes, except those that are dangerous to sample. The fixed area we recommend varies from 1 m^2 plots on transects (dunes and braided rivers) to 400 m^2 plots in forests.
- Conduct a comprehensive species assessment within all plots, with assessments of cover for each plant species.

- Make additional measurements for woody stems in forests, and standard additional measurements of foliar nutrients and water in wetlands.
- Use two 5-minute bird counts at each vegetation plot (other than plots on transects) to quantify the bird communities.
- Employ standard methods to assess ungulate and lagomorph presence (pellet presence by species or species group) and possum abundance (chewcards) at sample points on transects originating from vegetation plots, or along the transects on which small vegetation plots are established (dunes, braided rivers).

Recommendations for further work

We recommend that a consortium of regional councils work with DOC and researchers to:

1 conduct widespread field trials of methods for skeletal ecosystems on mobile substrates (especially on dunes and braided rivers) at a range of scales
2 further develop appropriate means to quantify changes in the ecological integrity of sites that are dangerous to sample
3 evaluate suitable levels of replication and spatial independence for sampling birds and ungulates
4 evaluate the advantages and disadvantages, and the information loss, of changing from existing standard methods to the methods we recommend, especially the consequences of changing from methods used to evaluate changes in structure in herbaceous communities to fixed-area plots
5 evaluate the information loss between 3-night sampling for possums compared with the single-night sampling we recommend
6 evaluate existing forest plot data to assess the information gained from the current seedling plot method or from altering minimum dbh in permanent plots (this would need to involve a collaboration with DOC and the Ministry for the Environment to ensure that any recommendations from the work align with existing monitoring efforts)
7 evaluate areal extents of ecosystems of interest across all regional councils to determine optimal sampling in ecosystems that are very small (e.g. <1 ha)
8 determine optimal sampling intensities and placement of samples for prioritised ecosystems (ideally similar sampling methods should be developed between DOC and regional councils that are easily comprehended and straightforward to use)
9 evaluate opportunities to implement the methods described in this report alongside cultural indicators developed by mana whenua
10 determine opportunities to integrate specific measures of mammalian herbivory (e.g. foliar browse scores) into plots
11 work with DOC to determine options for chewcards made of materials other than corflute to minimise plastic waste onsite
12 work with DOC to investigate the suitability of 5-minute wasp counts to determine pest wasp abundance across ecosystem types
13 develop and maintain databases suitable for use among all agencies to house standardised data for pest mammals and birds.

1 Background

Most regional councils and unitary authorities have programmes to secure, manage, and enhance a set of identified priority ecosystem sites. However, most monitoring of biodiversity in managed ecosystems, especially those that are rare or geographically complex, is *ad hoc*, non-quantitative, and idiosyncratic, and is seldom maintained, discoverable, or capable of providing compelling evidence of change, let alone effectiveness of management. There is currently no framework or process to aggregate or consolidate the data needed to provide compelling evidence of change and effectiveness of management, nor is there capacity to report collectively at a national level.

Some councils do not currently have a framework to monitor the effectiveness of their management of a range of different ecosystem types. Other councils have methods developed using a combination of current best practice and methods developed independently, often in the absence of consensus about national best practice. While there are some specific methods available and in use (e.g. to measure vegetation and some animals in some rare ecosystems), unifying methods, and the principles for applying those methods in a coordinated fashion, are lacking.

Given the expected future requirements to report on the state and trend of biodiversity, this project aims to enable consistency among councils, in turn enabling the consolidation of site-level monitoring data to report at broader scales. The project aligns with most regional councils' strategies, strategic plans and long-term plans, along with the upcoming National Policy Statement for Indigenous Biodiversity. The Department of Conservation (DOC) was also included (e.g. in workshops that were part of the project) to achieve national coordination.

2 Introduction

The aim of the work is to develop a harmonised approach to ground-based monitoring across all regional councils to determine management effectiveness of prioritised ecosystem sites, with the goal of maintaining or enhancing the ecological integrity of managed sites. This is necessary because current efforts are not systematic and are often idiosyncratic, which limits the capacity to build a compelling evidence base for reporting and the ability to aggregate data across sites. With a standardised approach, greater benefit could be achieved from the current level of investment in monitoring.

The success of the project rests on collective buy-in from multiple regional councils. The Envirolink funds sought for the project were unanimously supported by the regional councils' Biodiversity Working Group (BDWG) and were strongly supported and co-funded by this group. Hawke's Bay Regional Council (HBRC) led the project on behalf of the BDWG, and Manaaki Whenua – Landcare Research and Kevin Collins (Collins Consulting) undertook the work.

Three stages were identified as being necessary to deliver a ground-based biodiversity monitoring framework.

1 Scope the development of a tier 2 terrestrial biodiversity outcome monitoring framework that will assess the effectiveness of management at sites (e.g. ecosystem prioritisation sites) and identify gaps in current methods.

2 For ecosystems where there are current gaps, develop standardised methods for those ecosystems for which current methods are idiosyncratic (including a pilot to test them), and evaluate options for those where none currently exist.

3 Determine the power of detection and ability to aggregate site-level data to regional- and national-level biodiversity reporting.

Future stages are subject to change, depending on the outcome of stage 1, which is the subject of this report.

We convened workshops among all members of the BDWG monitoring sub-group and DOC. The goals of the workshops were to identify, across all regional councils:

- a way to aggregate the 152 ecosystems in *A Classification of New Zealand's Terrestrial Ecosystems* (Singers & Rogers 2014) into broad classes for the purposes of monitoring
- the suitability of current methods to report on the extent of those broad classes of ecosystems, and the condition of their biodiversity (especially in response to management)
- ecosystem classes for which there are either no or only poorly developed methods (e.g. rock bluffs).

The assessment of suitable methods relates in part to those developed for use by regional councils for standardised terrestrial biodiversity indicators for systematic regional and national (Tier 1) monitoring in a previous Envirolink project (Bellingham, Overton et al. 2016). However, only some of those methods will be suitable for measuring outcomes in the types of ecosystems being managed. For example, some of the methods will be unsuitable to address biodiversity outcomes in rare or geographically confined ecosystems. Moreover, the emphasis in this report is to ensure national coordination for measuring the management effectiveness, suitability and feasibility of methods (including where and how) while taking into account issues that are often unique to ecosystems on private land (such as partial protection/management of contiguous ecosystems).

The programme of work was as follows.

1 Before the workshops, HBRC and MWLR aggregated the 152 ecosystems in Singers & Rogers 2014 into broad classes (e.g. those in ephemeral or mobile habitats, those dominated by tall woody vegetation). We sent this material to all workshop participants and sought their feedback to achieve a consensus on the broad classes.

2 Before the workshop, we assembled all extant methods suitable for measuring change in the ecological integrity of the broad ecosystem classes identified in (1) and completed a gap analysis for ecosystems and ecotones for which no suitable methods exist, or for which potential methods are untested.

3 We sent this information to workshop participants to examine and comment on before convening the workshop. We then collated and circulated the comments before the workshop.

4 We convened three online facilitated workshops in May 2020 involving the BDWG Monitoring Sub-group and DOC to meet the goals outlined above.

5 This report summarises the workshops, recommends methods for determining change in ecological integrity in broad ecosystem classes, and makes recommendations for further investigation and investment.

3 Objective

The objective was to develop a harmonised approach to ground-based monitoring across all regional councils to determine management effectiveness on prioritised ecosystem sites, with the goal of maintaining or enhancing the ecological integrity of managed sites. This is necessary because current efforts are not systematic and often idiosyncratic, which limits the capacity to build a compelling evidence base for reporting, and the capacity to aggregate data across sites. With a standardised approach, greater benefit could be achieved from the current level of investment in monitoring.

4 Methods

4.1 Aggregating the 152 ecosystems in Singers & Rogers 2014 into broad classes for the purposes of monitoring

Our approach to aggregating the 152 ecosystems in Singers & Rogers 2014 into broad classes for the purposes of monitoring was to split them into sometimes overlapping groups, based on:

- whether ecosystems were predominantly woody (i.e. those in which woody stems ≥2.5 cm diameter at 1.35 m tall) predominate vs. those without; we then divided predominantly non-woody ecosystems (in which woody plants ≥2.5 cm diameter at 1.35 m tall are currently absent or rare) into ecosystems that are perpetually in that state (e.g. alpine ecosystems) vs. those that are currently mostly non-woody but which could undergo succession to woody communities
- wetlands (in which the water table is at or above the soil surface perpetually or for extended periods) vs. non-wetlands
- whether the ecosystems occur on skeletal surfaces (e.g. gravel, ultramafic rock) – these were further divided into those on stable vs. mobile substrates (e.g. fresh river gravels, dunes)
- whether the ecosystems are on steep surfaces that are dangerous to sample and other potentially dangerous substrates (e.g. geothermal sites).

Sometimes groups were not mutually exclusive (e.g. forested wetlands). We sent our provisional aggregation of Singers & Rogers 2014 ecosystems to members of the BDWG for comment and made further adjustments after their input (for the final groupings see section 5.1).

4.2 Evaluating the suitability of current methods to report on the extent of those broad classes of ecosystems and the condition of their biodiversity (especially in response to management)

We conducted interviews with the staff of nine regional councils to determine point-based methods they currently use to evaluate changes in biodiversity at local sites across a range of ecosystems. We conducted three workshops in May 2020 (convened remotely because of the Covid-19 pandemic). We presented and discussed with regional council staff and participants from DOC the proposed methods and their suitability to determine changes in ecological integrity in response to management. The workshops focused on forests, wetlands, and other ecosystems. We had subsequent discussions with some workshop participants.

We evaluated the suitability of methods used in systematic regional evaluations (Tier 1; Bellingham, Overton et al. 2016) and presently in use by Greater Wellington Regional Council and Auckland Council for regional reporting (Bellingham et al. 2020). These align closely with methods used nationally by DOC in Tier 1 reporting. In determining a minimum set of methods, we focused on vegetation, bird communities, and some pest mammals, in the same way that these were agreed by regional councils as a basis for region-wide assessments (Bellingham, Overton et al. 2016). The methods suitable for regional evaluations are used in widespread ecosystems (forests, some wetlands, perennial grasslands).

We evaluated FORMAK (the Forest Monitoring and Assessment Kit).[1] FORMAK methods are used by some regional councils, landowners, land-care groups, and community groups to monitor ecological state and trends in forests, including the response of forests to management. Vegetation plots advocated for use in forests in FORMAK are 4 m × 20 m (based on a central 20 m transect). Only the central 20 m transect is permanently marked, potentially leading to errors in reconstituting the plot boundaries. These plots are one-fifth the size of the 20 m × 20 m plots widely used in New Zealand (e.g. Hurst & Allen 2007a), which we believe is insufficient to sample tree densities and tree diversity in many forests (Bellingham et al. 1999; Holdaway et al. 2017). Similarly, replication of seedling plots in FORMAK forest plots is low (five per plot), two of which are located at the edge of the 4 m × 20 m forest plot, each extending 0.5 m^2 beyond the forest plot boundary. FORMAK bird methods for forests are very similar to those we recommend, and FORMAK methods to assess ungulates and possums are also similar to those we recommend.

Current methods suitable for region-wide assessment are often not used or potentially unsuitable for rare or restricted ecosystems, including ecosystems on mobile substrates (e.g. on sand dunes and braided rivers) and wetlands. We evaluated methods used in determining change in plant communities over time on sand dunes, especially from New Zealand (e.g. Hilton et al. 2005), as well as from point-in-time studies on sand dunes (e.g. Drobner et al. 1995; Stubbs & Wilson 2004; Pegman & Rapson 2005; Verhoeven et al. 2014). We also evaluated methods to quantify vegetation on other mobile substrates,

[1] https://www.formak.co.nz/webfolder.html; accessed 15 September 2020.

including gravel beaches and braided rivers (Williams & Wiser 2004; Wiser et al. 2010; Brummer et al. 2016). Methods to assess changes in biodiversity (especially plants) in wetlands are well developed in New Zealand and conform with international methods (Clarkson et al. 2004, 2014; Clarkson & Bartlam 2017).

We also evaluated FORMAK's equivalent for wetland monitoring, WETMAK (Wetlands Monitoring and Assessment Kit).[2] There is a high degree of overlap between the methods we recommend for monitoring ecological integrity in wetlands and those recommended in WETMAK, except for a few details (e.g. we do not recommend the use of 2 m × 2 m plots to monitor wetland vegetation).

Methods for measuring vegetation in permanent grasslands are well developed in New Zealand and have focused on structural measurements (Wraight 1962; Dickinson et al. 1992; Wiser & Rose 1997). DOC has used fixed area (20 × 20 m) plots in permanent grasslands in its Tier 1 programme, and this has been adopted in grasslands by Greater Wellington Regional Council and Auckland Council. In this report we advocate for fixed-area plots rather than methods suitable to evaluate structure.

We have not recommended photopoints to monitor vegetation. Photopoints are widely advocated in ecological monitoring (e.g. FORMAK, WETMAK) and can be effective ancillary tools in showing changes in vegetation with time (e.g. Wilson 1994). Our recommendations focus on methods that provide quantitative data, and we do not recommend photopoints alone as a means of assessing vegetation change or the effectiveness of management because it is difficult to generate quantitative data from them. However, they can be used effectively as ancillary data to support quantitative data from permanent vegetation plots; for example, to illustrate the responses of vegetation to herbivores (Cruz et al. 2017) or to changes in fire and grazing management (Mark & Dickinson 2003). We therefore suggest using photopoints in conjunction with quantitative plot-based methods.

Remote sensing techniques are an important tool to delimit ecosystem boundaries, with greatest confidence around woody vegetation (forests and shrublands), and less with herbaceous communities, grasslands, etc. They can be used to determine change in areal extent of some ecosystems (e.g., Weeks et al. 2013, Cieraad et al. 2015, Monks et al. 2019, Robertson et al. 2019), but afford much less insight into change in condition and composition within areas. Measures of change in components of ecological integrity such as maintaining ecosystem composition cannot be achieved currently by remote sensing, and others such as reducing spread and dominance of exotic species can only be detected when major transformations have taken place (e.g., major invasions of non-native conifers into grasslands or willows into wetlands) that alter structure and dominance. In the latter case, this would trigger management intervention only once invasions had already transformed ecosystems, and remote sensing would not detect subtler invasions of plants beneath extant canopies (e.g., seedlings of the deciduous tree sycamore, *Acer pseudoplatanus*, invading forest understoreys; Williams 2011).

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[2] https://www.landcare.org.nz/resource-item/wetmak; accessed 15 September 2020).

5 Results

5.1 Aggregating the 152 ecosystems in Singers & Rogers 2014 into broad classes for the purposes of monitoring

The 152 ecosystems of Singers & Rogers 2014 were aggregated into seven classes, described below and listed fully in Appendix 1. For each of the seven classes, we describe the key drivers of change in those ecosystems that are related to human activities or result from human introductions of weeds and pests. We also describe management actions intended to maintain or improve ecological integrity in each class, and attributes of ecological integrity that could be measured to evaluate the effectiveness of management.

1. Forests

Generic ecosystem types: forests, tall shrublands.

Indicative Singers & Rogers ecosystems: all forests (subtropical SF1; warm temperate WF 1–14; North and South Island mild MF 1–25; cool forest and scrub CLF 1–12; cool forest and scrub CDF 1–7) and some azonal vegetation (e.g. cold temperate inversion TI 1, 2, 4, 5); also forests dominated by non-native tree species not described by Singers & Rogers.

Key drivers: stock grazing; pest animals (e.g. deer, goats, pigs, brushtail possums, rodents and mustelids); weed invasion; fire; adjacent land use and cover.

Recurrent management activities: fencing to exclude livestock; wide-scale pest animal control (especially brushtail possums, goats); local pest animal control (rodents and mustelids, deer); weed control; restoration planting or managing natural regeneration to augment forest area or enhance connectivity among remnants.

Indicators of ecological integrity (examples): indigenous dominance measured as the relative cover of native and non-native plants, relative abundance of native and non-native birds, and the abundance of pest animals; species occupancy (size–class structures and distributions of widespread species or species' groups typical of forests and shrublands).

2. Wetlands

Generic ecosystem types: non-forested freshwater wetlands.

Indicative Singers & Rogers ecosystems: wetlands (WL1–21); possibly some other ecosystems with a tall (>2 m height) woody component (e.g. WF8), and those invaded by non-native trees (especially *Salix* spp.), not described by Singers & Rogers.

Key drivers: drainage; excessive nutrient input causing eutrophication (sometimes caused by adjacent land management); stock grazing; weed invasion; pest mammals; fire; changes in land-cover matrix.

Recurrent management activities: reinstatement or maintenance of hydrology regime; buffer planting next to adjacent land; fencing to exclude livestock; wide-scale pest animal control (especially brushtail possums, goats); local pest animal control (rodents and mustelids, deer); weed control.

Indicators of ecological integrity (examples): indigenous dominance measured as the relative cover of native and non-native plants, relative abundance of native and non-native birds, and the abundance of pest animals; plant and animal species occupancy (distributions of widespread species or species' groups typical of wetland types).

3. Perpetual herbaceous communities (including some woody components)

Generic ecosystem types: herbaceous communities (some with low, woody components).

Indicative Singers & Rogers ecosystems: tussock and shrublands (AL1–9), alpine communities (AH2), some saline environment communities (e.g. SA5, 6, 10), and temperature inversion lichen field and shrubland (TI3).

Key drivers: stock grazing; pest mammals; weed invasion; fire; land-cover matrix.

Recurrent management activities: fencing to exclude livestock; wide-scale pest animal control (especially brushtail possums, goats); local pest animal control (rodents and mustelids, deer); weed control.

Indicators of ecological integrity (examples): indigenous dominance measured as the relative cover of native and non-native plants, relative abundance of native and non-native birds, and the abundance of pest animals; species occupancy (distributions of widespread species or species' groups).

4. Herbaceous communities on stable ground undergoing succession

Generic ecosystem types: herbaceous communities.

Indicative Singers & Rogers ecosystems: tussock ecosystems affected by combustion or volcanic activity (VS11–14). Also abandoned or retired agricultural ecosystems.

Key drivers: stock grazing; pest mammals; weed invasion; fire; land-cover matrix.

Recurrent management activities: fencing to exclude livestock; wide-scale pest animal control (especially brushtail possums, goats); local pest animal control (rodents and mustelids, deer); weed control; planting of native woody species.

Indicators of ecological integrity (examples): indigenous dominance measured as the relative cover of native and non-native plants, relative abundance of native and non-native birds, and the abundance of pest animals; species occupancy (distributions of widespread species or species' groups).

5. Skeletal ecosystems on stable substrates

Generic ecosystem types: sparse, mostly herbaceous, plant communities set among large areas of bare ground.

Indicative Singers & Rogers ecosystems: alpine herbfields (AH1–4), tussocks on ultramafic substrates (UM3), erosion pavements (EP1).

Key drivers: pest mammals; weed invasion.

Recurrent management activities: wide-scale pest animal control (especially deer, chamois, tahr); weed control.

Indicators of ecological integrity (examples): indigenous dominance measured as the relative cover of native and non-native plants, relative abundance of native and non-native birds, and the abundance of pest animals; species occupancy (distributions of widespread species or species groups).

6. Skeletal ecosystems on mobile substrates

Generic ecosystem types: sand dunes, gravel beaches, braided riverbeds, possibly lake shores.

Indicative Singers & Rogers ecosystems: dune ecosystems (DN1–4); all three braided river ecosystems (BR1–3); some of the saline ecosystems (SA4, 7, 8) where they occur on or intergrade with mobile substrates; many other ecosystems will occur infrequently where sampling includes recently stabilised sediments (e.g. back dunes; margins of river beds).

Key drivers: rates of substrate deposition; rates of substrate erosion (by water and wind); sea-level rise and available land; afforestation through planting or natural succession; invasion by substrate-stabilising species (e.g. gorse, marram, pine); stock grazing; population dynamics of pest animals (e.g. rabbits, pigs); direct human disturbance (e.g. 4WD, subdivisions); fire; land-cover matrix.

Recurrent management activities: weed control (e.g. marram, broom, wilding pines, bone seed, boxthorn); wild animal control (e.g. rabbits, pigs, possums, cats); access management (e.g. 4WD, walking tracks); restoration planting.

Indicators of ecological integrity (examples): indigenous dominance measured as the relative cover of native and non-native plants, relative abundance of native and non-native birds, and the abundance of pest animals; species occupancy (size–class structures and distributions of widespread species or species groups typical of those ecosystems).

7. Ecosystems that are dangerous to sample

Generic ecosystem types: cliffs; steep, rocky or clay habitats; some geothermal habitats.

Indicative Singers & Rogers ecosystems: coastal and inland cliff and talus ecosystems (CL1–11), geothermal systems (GT1 in part and GT2 in total), and cave entrances (CV1).

Key drivers: rates of substrate erosion (by water and wind); rates of substrate deposition; sea-level rise and available land; weed invasion (e.g. gorse, old man's beard, pine); invasive browsing mammals; fire; land-cover matrix.

Recurrent management activities: weed control (e.g. gorse, old man's beard, pines, bone seed, boxthorn); wild animal control (e.g. goats, deer, rabbits, hares).

Indicators of ecological integrity (examples): indigenous dominance measured as the relative cover of native and non-native plants, relative abundance of native and non-native birds, and the abundance of pest animals; species occupancy (size–class structures and distributions of widespread species or species groups typical of those ecosystems).

5.2 Ecological integrity and management

The objective is to determine whether the ecological integrity of prioritised ecosystem sites being secured, enhanced or maintained by management requires tools to monitor the various objectives that comprise ecological integrity. Reporting environmental outcomes in the context of ecological integrity is enshrined in law in New Zealand (Environmental Reporting Act 2015), and the Biodiversity Assessment Framework used by DOC was developed explicitly in the context of eight outcome objectives that comprise ecological integrity (Table 1), each with explicit measures.

Table 1. Outcome objectives of ecological integrity in the Department of Conservation's Biodiversity Assessment Framework (McGlone et al. 2020)

Outcome objectives	Description
Maintaining ecosystem processes	The extent to which the environment is capable of supporting indigenous ecosystems and the degree to which they are free of disturbance factors that lead to poor ecological outcomes.
Reducing spread and dominance of exotic species	Documentation of the presence, dominance and rate of increase of exotic species in the natural environment.
Preventing declines and extinctions	Conservation status of all species in the New Zealand biota (per the New Zealand Threat Classification System); security of threatened and at-risk taxa; loss of genetic diversity in critically reduced taxa.
Maintaining ecosystem composition	Demography of functional groups, their representation, abundance of common and widespread taxa and changes in species diversity.
Ensuring ecosystem representation	The extent, protection status and ecological condition of indigenous ecosystems.
Adapting to climate change	Documentation of changing climates, and the biological responses.
Human use and interaction with natural heritage	Documentation of how humans interact with natural ecosystems in their harvesting of both indigenous and exotic taxa, through recreating in them, and how they use them to gain spiritual and physical well-being.

In workshops convened during this project and in subsequent discussions there was no consensus on what constitutes management. Some regional councils had a strict view of commonly applied management actions applied at local scales to prioritised ecosystems (e.g. fencing, pest and weed control). Other regional councils took a broader view to include not only management actions applied at local scales but also the effectiveness of policy, or of directives such as National Policy Statements. We have not attempted to reconcile these viewpoints, but have worked to provide methods that are suited to narrow and broad views of management and reporting the outcomes of either.

Commonly applied management activities relate to at least one outcome objective of ecological integrity, and methods are needed that are suitable to measure progress towards maintaining or enhancing those objectives. Following are some examples.

- Weed control pertains, at the very least, to reducing the spread and dominance of exotic plant species and maintaining ecosystem composition (of native plant species). When the weed species are functionally distinct (e.g. combustible species, such as gorse, pines, or *Hakea* spp.), their control may also be aimed at maintaining ecosystem processes (e.g. reducing the flammability of the ecosystems). When rare native plant taxa are present, weed control may also be focused on preventing declines and extinctions if the weeds are presumed to be competing for resources (e.g. light) with the rare native plants.
- Possum or ungulate control typically focuses on maintaining ecosystem composition (of native species) by ensuring the plant species they consume preferentially are represented. If there are particular rare species consumed by these animals (e.g. mistletoe species consumed preferentially by possums), then their control might be focused on preventing declines and extinctions. Since the plant species they consume are typically functionally distinctive (usually thin-leaved, high in foliar nutrient concentrations, and often decomposing rapidly; Wardle et al. 2001), possum and ungulate control might also focus on maintaining ecosystem processes.
- Control of predatory mammals may focus on maintaining ecosystem composition (of native bird species) and on preventing declines and extinctions of rare native bird species. It might also focus on maintaining ecosystem processes (e.g. predator control in successional landscapes focused on boosting populations of kererū/kukupa; *Hemiphaga novaeseelandiae*) to ensure dispersal of seeds of native tree species that they alone can disperse from residual populations (Clout & Hay 1989).

Even in cases of an assumed response to these commonly applied management activities, covariates and interpretive data are needed, since the responses of ecosystems to management are highly context-dependent. Simple cause–effect relationships resulting from management actions do exist (e.g. only complete removal of all predatory mammals will allow populations of tīeke [*Philesturnus* spp.] to be sustained or increase in forests), but they are rare.

Some assumed cause–effect relationships have been shown to be incorrect, but more commonly the signal of a management effect is drowned by other influences (e.g. drought, soil fertility) and endogenous attributes (e.g. stand structure). The difference made by management (Overton et al. 2015) is contingent on the initial state of the ecosystem (e.g. its state of degradation, including legacies of past land use, loss of soil organic matter, depletion of residual tree species that might colonise a site, trampling by livestock, below-ground effects of exotic plants).

The rate at which change in response to management will occur will be similarly contingent on environmental influences; for example, more rapid change could be expected in milder sites than in colder sites, in wetter sites than in drier sites, and in sites with fertile vs. infertile soils. The more contextual data are collected as part of a biodiversity monitoring programme, the more defensible the interpretation of trends (Peltzer et al. 2014).

It is also important to collect contextual information during the monitoring programme. For example, major disturbances (such as Cyclone Ita on West Coast forests in 2014, or the Kaikōura earthquake of 2016) can have a far greater effect on the ecosystem than management activities, through disruption of plant communities and soils. These disturbance events can often provide a basis for refocusing management activities. For example, fencing forests to exclude ungulates immediately after canopy disturbances (e.g. after storms or cyclones) will have a much greater effect on recruitment of plants they consume than under intact canopies (Mason et al. 2010).

5.3 Issues of design

This report focuses on the minimum requirements for monitoring change in the ecological integrity of prioritised ecosystem sites. It is challenging to ascribe changes to the effectiveness of management, since management occurs against a background of pervasive change. For example, pervasive climate change has resulted in extended periods of summer drought (e.g. during the summer of 2019/20; Ministry for the Environment & Stats NZ 2020). In forests this could result in a significant reduction of seedling regeneration and high mortality of existing seedlings, and this may be especially high in seedlings of tree species with high leaf area per unit mass (specific leaf area; e.g. Greenwood et al. 2017). Species favoured by mammalian herbivores often have high specific leaf area (e.g. Lloyd et al. 2010; Mason et al. 2010), and a lack of regeneration of these species and high mortality could be wrongly ascribed to a failure of management if the effects of drought were not considered.

Conducting management in an explicitly experimental framework can be a way of discerning management effects, but this often poses difficulties. For example, paired experimental control and treatment areas are often hard to achieve. Environmental differences between treatment and non-treatment areas are often the major drivers of the responses observed, and this is particularly the case for small lowland ecosystems that are naturally uncommon, or are small fragments of ecosystems that were once more extensive. Replicates are either hard to find or it is undesirable to leave some untreated by management (e.g. deliberately leaving some rare forest fragments unfenced).

Interpreting data from local prioritised ecosystem sites will also be strengthened if site-level trends can be set in the context of regional or national trends determined from systematically located sample points measuring the same biodiversity components (i.e. on an 8 km grid; Bellingham et al. 2020). Widespread plots (Tier 1) will allow comparability in some widespread ecosystems between managed and unmanaged sites. These plots could provide context to pervasive influences (e.g. climate change, drought) that local management cannot influence. However, many prioritised sites in naturally rare ecosystems or severely depleted ecosystems (such as lowland forest fragments on fertile soils) cannot be readily matched with the spatially extensive data set.

Recent statistical advances in assessing the effects of management at a small scale allow their interpretation using as many wide-scale data as are available (e.g. from national or regional systematic data sets; such as Tier 1 data from the 8 km grid). Propensity scoring can be used to reduce systematic differences in confounding variables and evaluate the signal that results from management treatment. For example, after taking account of 13 potentially confounding variables (related to stand attributes, climatic conditions, soil nutrients, physiography, etc.), in the context of widespread data the effects of local management of possums could be linked with confidence to improved canopy conditions of one tree species (southern rātā, *Metrosideros umbellata*) but not of another three common tree species (Ramsey et al. 2019).

To achieve the best integration of local-scale or site-based data with regional or national data, it is critical that in-common methods and objective sampling techniques be used at local and regional scales. In this way it is possible to leverage wider investment (by DOC, MfE, and other regional councils). Solutions like this will not be possible for many ecosystems of interest to regional councils, especially severely depleted lowland ecosystems (e.g. some lowland forests), and rare ecosystems in which all extant sites are of interest and all are subject to management (e.g. all geothermal sites in the Bay of Plenty region).

Designs that are comparable across the broad ecosystem classes will allow not only comparisons among similar ecosystems but also across them. This will be especially useful where a managed site includes several ecosystems with ecotones among them (e.g. from foredunes to freshwater wetlands in the back dunes, to old-growth forests further inland).

5.4 Principles of monitoring, including state vs. trend

Key principles of monitoring for all ecosystem types/groups are reviewed in Lindenmayer & Likens (2010, 2018; see also Allen et al. 2003) and summarised below:

Any method must be repeatable by different people, within an acceptable level of error.
Data must be fit for purpose for reporting on three aspects of biodiversity: • state • change (trend) • responses to management.
Simple or rapid methods must yield data that are comparable with components of more complex methods. For example, a rapid measurement of forest stand structure using a basal area prism sweep is comparable with a measurement based on individual tree diameters.
Methods must be fit for reporting on the condition of individual sites and for pooling across sites to report on broad ecosystem types (and fit for scaling to national levels, if necessary).
Objectivity should be incorporated into the sampling design.

Major abiotic gradients influence the composition and structure of characteristic biota of most ecosystems and should be recognised and incorporated into the sampling design. Examples include: distance from forest margin for small forest fragments); water depth for some wetlands; distance to high tide line for coastal systems (equates to disturbance, salinity); heat gradients in geothermal areas.
Statistical power comes from replication, so it is better to apply a simple method at more points than to conduct a detailed survey at a few points. This will be particularly important in skeletal habitats, where many plots will have few plants.
All monitoring methods are a compromise because you cannot measure everything (e.g. most methods do not measure very small seedlings, bryophytes, worms, etc.).

For some of the types of sites councils are monitoring, stratification could greatly influence sampling efficiency. In complex sites where multiple broad ecosystem groupings may be present, a site might be stratified according to vegetation height/structure, with different methods applied within them (e.g. from foredunes to freshwater wetlands in the back dunes, to old-growth forests further inland).

Ad hoc observations of species of interest are widely collected but almost never stored in such a way as to be useful beyond the realm of the individual who collected them. To enhance their importance we advocate the following as best practice.

- For all taxa, take a photograph and use iNaturalistNZ to archive the observation. iNaturalistNZ gives an option to obscure locations, and this can be applied, if necessary, to safeguard landowner interests.
- For plants, collect a voucher specimen and send this to a herbarium (if the plant is too rare at the site for this, use iNaturalistNZ to archive the observation).
- Some taxa have specific sites to lodge observations (e.g. for kea; https://www.keaconservation.co.nz/support-kea/kea-sightings/).

5.5 Current methods suitable to report on the extent of the broad classes of ecosystems and the condition of their biodiversity (especially in response to management)

Methods recommended for determining changes in the ecological integrity of broad ecosystem classes, and especially the effectiveness of management, are described below. Most methods recommended are already widely applied and well established. Most of the methods have been used in peer-reviewed published studies in journals, which should give regional councils confidence in their use.

Note that the brief for this report focuses on point-based measures of ecological integrity. The brief specifically excludes measures of the extent of ecosystems, but we wish to emphasise that extent is a critical component of ecological integrity. Since loss of extent of threatened ecosystems is occurring in New Zealand, contributing to overall biodiversity loss (Cieraad et al. 2015; Robertson et al. 2019), reliable methods to quantify extent and change in extent are essential for assessing the effectiveness of the management of prioritised ecosystems alongside those we present.

Methods to detect changes in extent rely predominantly on remote imagery, from satellites (e.g. Landcover Database; LCDB) to aerial imagery, to drones. The focus in the methods recommended below for point-based measures makes no mention of these important remote technologies. However, these technologies are important in conjunction with point-based measures. Measurement of extent can be enhanced through ground-based assessments made when point-based measures are made, to ground-truth remote imagery and to resolve ecosystem boundaries and ecotones.

In discussions with regional councils before compiling this report there was agreement that we would not recommend dropping components of established methods in setting a minimum requirement. Our recommended approach is to start measuring ecosystem integrity in prioritised ecosystems with a comprehensive use of methods and then consider what to drop in subsequent remeasurements, rather than starting less comprehensively and deciding what to add. We describe suitable methods to measure attributes of ecological integrity that pertain to vegetation and birds, and to measure pressures on ecological integrity (weeds and some pest mammals), for each of seven classes of ecosystems.

5.5.1 Forests

Vegetation

- Establish multiple permanent 20 m × 20 m plots per site, with locations assigned in advance.
- Use a GPS to record the location of the centre of each plot.
- Record slope, aspect, topographic index, mean canopy top height, elevation off Topomap, and disturbances (per Hurst & Allen 2007b), and make detailed notes and a diagram to relocate the plot.
- If budget permits, collect a soil sample from the top 100 mm of mineral soil (i.e. after scraping off the litter and organic horizon) from each of four quarters of the 20 m × 20 m plot, pool the samples, and analyse for pH, and total carbon (C), nitrogen (N), and phosphorus (P).
- Record all vascular plant species in each plot and allocate percentage live cover to each species as a relevé (or 'recce' plot) bounded by the plot edge. Use an ordinal cover score for each species (<1%, 1–5%, 5–25%, 25–50%, 50–75%, 75–100%) within fixed height tiers (<0.3 m; 0.3–2 m; 2–5 m; 5–12 m; 12–25 m; 25 m+) and record presence, not cover, of epiphytes (Hurst & Allen 2007b).
- Divide each 20 m × 20 m plot into 16 contiguous 5 m × 5 m subplots.
- Permanently tag (metal tags nailed to stems) all stems ≥2.5 cm diameter at 1.35 m height above ground (dbh), record the subplots in which they occur, record species identity, and record stem diameter to the nearest 0.1 cm (using diameter tapes) (Hurst & Allen 2007a). Tag all stems of multi-stemmed trees and note that they are attached to one another. Liana stems are not tagged, nor are their diameters recorded.
- For tree ferns and palms (which do not exhibit radial growth), tag and record species identity (long nails are required) and the subplots in which they occur, and measure height from ground to the point of the stem from which fronds or leaves emerge, to the nearest 0.1 m.

- Record a tally of saplings by species in each subplot (saplings are defined as woody stems >1.35 m tall and <2.5 cm dbh).
- Establish 24 seedling subplots at regular points (equidistant 2.5 m along all plot interior edges of 5 m × 5 m subplots; Hurst & Allen 2007a). Seedling subplots are circular about the central datum, 0.49 m radius (i.e. 0.75 m^2 subplots). Record all species in each subplot by maximum height of each individual, in height tiers (<0.15 m, 0.15–0.45 m, 0.46–0.75 m, 0.76–1.05 m, 1.06–1.35 m tall). Record presence only for species <0.15 m, and presence in height tiers for herbaceous species, and tallies in height tiers ≥0.15 m tall for woody species. Do not establish seedling plots in forests that are periodically or permanently inundated (i.e. in any of SA1, WF8; MF4, 13, 14; CLF7; SA1; TI1, 5; VS4, 5; WL1, 2, 4, 12, 20, 21 that are inundated).
- Photograph or preferably collect material of plants that cannot be readily determined in the field for later determination by experts. Collect flowering or fruiting material (or otherwise fertile material, e.g. fern fronds bearing sori) whenever possible to facilitate determination.

Birds

- Use five-minute bird counts (5MBC) to sample the bird community.
- Constrain counts to ≤100 m of the recording point.
- Conduct two counts at the centre of each permanent 20 m × 20 m vegetation plot. DOC's Tier 1 protocol adds counts at the end of transects used to assess presence of mammalian herbivores; we do not recommend these for forests of interest to regional councils because we expect that many are small, and achieving independence of sample points will be challenging, as well as the additional costs of obtaining these data. Conduct one count on arrival at the plot and the second before departure from it; or, if returning the following day for chewcards (see below), conduct the second count on that day. Data are analysed as counts (mean across the two 5MBC at each sample point), but by making two counts at each location, these data are fit for formally estimating occupancy (MacLeod et al. 2012).
- Because bird abundance and conspicuousness vary with time of year, 5MBC should be undertaken at the same time of year to ensure comparability in repeated measures.

Ungulates, lagomorphs

- Collect data on presence of faecal pellets of ungulates and lagomorphs along four 150 m transects extending diagonally from each corner of each permanent 20 m × 20 m vegetation plot.
- Set 1 m radius subplots at 5 m intervals along each transect (excluding any subplots that are permanently inundated in any of SA1, WF8; MF4, 13, 14; CLF7; SA1; TI1, 5; VS4, 5; WL1, 2, 4, 12, 20, 21); 30 subplots per transect. Record the presence of faecal pellets of ungulates in each 1 m radius subplot. Record the presence of faecal pellets of lagomorphs in each of 0.18 m radius subplots, nested at the centre of the 1 m radius subplots (DOC 2020) for ungulate faecal pellets along each transect; distinguish pellets of brown hares vs. European rabbits in the field.

- If the transects are unsafe to complete (e.g. they are too steep), or obstacles are encountered, or they run into non-forested habitat (e.g. adjacent pasture), then treat it as an impassable barrier and turn 90°.

Possums

- Monitor possums using chewcards (9 cm × 18 cm cards made of 3 mm plastic corflute; Sweetapple & Nugent 2011), baited with an aniseed-flavoured paste.
- Nail chewcards to trees or affix through metal stakes c. 30 cm above the ground at 20 m intervals along four 200 m transects extending diagonally from each corner of each permanent 20 m × 20 m vegetation plot, parallel to the 150 m transects used to monitor ungulates and lagomorphs, separated by 3.5 m from them.
- Set chewcards for a single fine night along the transects (per DOC Tier 1 monitoring methods, DOC 2020; see also Forsyth et al. 2018). Collect the following day, and label them according to each sample point, transect and position along them. If a chewcard has been chewed, then the identity of the mammal that chewed it is assigned (brushtail possum, other mammal, or both: other mammals that chew the chewcards include rodents, stoats, and European hedgehogs; all of these can be distinguished and their presence on each card noted; Sweetapple & Nugent 2011).
- If the transects are unsafe to complete (e.g. they are too steep), or obstacles are encountered, or they run into non-forested habitat (e.g. adjacent pasture)), then treat it as an impassable barrier and turn 90°.
- The data are presence of each animal species on each card, summarised as a proportion across all four transects for each plot, and then summarised at a site as the mean proportion across all plots.

5.5.2 Wetlands

Apply the wetland condition index (as per Clarkson et al 2004; Clarkson et al 2014; Clarkson & Bartlam 2017) to the site. The current version is set out below, but elements of this may change and the most up-to-date version should be followed:

Map vegetation types

- Delineate major vegetation structural classes of each wetland (per Atkinson 1985; the extent of the wetland having already been defined), based on grey literature, local knowledge, aerial photos, and other relevant information, and store this information in a GIS. This may need to be periodically updated. Ensure each vegetation type is homogeneous. Where the wetland is a complex mosaic not suitable for splitting into vegetation types, treat the wetland as a single sampling unit.

Sample vegetation, soils, and foliage

- Each vegetation type requires at least one plot, but preferably more, which will be determined by the size of each vegetation type within the wetland.
- Choose plot locations randomly, with back-up points in case locations prove unsuitable; all plots no less than 10 m apart.

- Using the GPS random point coordinates as the origin and south-west corner, set up a plot due north, east, etc. from that point using a compass, tape-measures and poles. Record the GPS location at the plot centre.
- Record mean canopy top height, elevation off Topomap, and disturbances (per Hurst & Allen 2007b), and make detailed notes and a diagram to relocate the plot.
- Establish permanent 5 m × 5 m plots in each vegetation type, with locations assigned in advance. For other woody vegetation types not assigned in Singers & Rogers 2014 (e.g. willow), establish multiple permanent 10 m × 10 m plots to reflect the greater stature of the vegetation, with a single 5 m × 5 m subplot nested within each plot (at its southwest sector) to facilitate comparisons among wetland plots. In exceptional circumstances where tall native forest (e.g. kahikatea) dominates the vegetation type, establish 20 m × 20 m plots per Hurst & Allen 2007a, with nested 10 m × 10 m and 5 m × 5 m plots at the southwestern sector of each plot.
- Record a continuous percentage estimate for all plant species identified in fixed height classes as a relevé (Clarkson & Bartlam 2017). The continuous percentage measure represents the vertical projection (spread) of the above-ground live biomass for each species, measured as percentage cover of the total area of the plot, irrespective of height or tier, or the position of other vegetation. Imagine each species is the only species in the plot and estimate its cover. Individual species cover cannot be more than 100%, but total vegetation cover will usually be >100%. This applies to all vascular species and *Sphagnum* moss. Bryophytes and lichens may also be recorded to species level if known, but must also be recorded collectively as bryophytes or lichens.
- Covariates relating to the nutrient status of wetlands should be collected if budget allows, following the protocols in Clarkson et al. (2004) and Clarkson & Bartlam (2017). These include soil samples and plant foliage samples. Plant foliage N:P ratios have been used to predict whether a plant community will respond to either nitrogen or phosphorus limitation (Burge et al. 2020). Collect soils from the southwest corner of the plot (e.g. 100 mm × 70 mm soil core) and foliage samples from the most common species within the whole plot (Clarkson et al. 2004). Determine bulk densities and pH for each soil sample. Send samples by courier to an ISO-accredited laboratory for analysis for total C, N, P and potassium. Measure the von Post index to assess rates of peat decomposition (if applicable; Taylor & Pohlen 1979).

Birds

- Use 5MBC to sample the bird community.
- Constrain counts to ≤100 m of the recording point.
- Conduct two counts at the southwestern corner of each plot, one on arrival. Conduct the second measurement on the following day when returning to collect chewcards (see below).
- Conduct bird counts at the same time of the year in repeated measures. Many councils conduct species-specific monitoring in wetlands (e.g. fernbirds, *Megalurus punctatus*; crakes, *Porzana* spp.; Australasian bittern, *Botaurus poiciloptilus*), but species-specific monitoring is beyond the scope of this report. Counts of birds in open bodies of water require additional methods that are also beyond the scope of this report.

Ungulates, lagomorphs

- Use the vegetation plots and record presence/absence of pellets by animal group (deer/goat, pig, horse, cattle, lagomorph).
- Regional councils can elect to count pellets and pellet groups, but we emphasise that a substantial amount of information is present in the presence/absence of pellets, and many field staff find presence/absence less ambiguous to measure than counts.

Possums

- Monitor possums using chewcards (9 cm × 18 cm cards made of 3 mm plastic corflute), baited with an aniseed-flavoured paste.
- Nail chewcards to stakes c. 30 cm above the ground at 20 m intervals along four 200 m transects extending diagonally from each corner of each permanent 5 m × 5 m vegetation plot, parallel to the 150 m transects used to monitor ungulates and lagomorphs, and separated by 3.5 m from them. Abandon or truncate transects that extend into deep water.
- Set chewcards for a single fine night along the transects (per DOC Tier 1 monitoring methods; DOC 2020). Collect them the following day, and label them according to each sample point, transect and position along them. If a chewcard has been chewed, then ensure the identity is possum only (Sweetapple & Nugent 2011).
- The data are presence of each animal species on each card, summarised as a proportion across all four transects for each plot, and then summarised at a site as the mean proportion across all plots.

5.5.3 Perpetual herbaceous communities (including some woody components)

Vegetation

- Multiple permanent 2 m × 2 m (turfs, short herbfields) or 5 m × 5 m (grasslands) plots per site, with locations assigned in advance.
- Use a GPS to record the location of the centre of each plot.
- Record slope, aspect, topographic index, mean canopy top height, elevation off Topomap, and disturbances (per Hurst & Allen 2007b), and detailed notes and a diagram to relocate the plot.
- If budget permits, collect a soil sample from the top 100 mm of mineral soil (i.e., scraping off litter and organic horizon) from each of four quarters of the plot, pool the samples, and analyse for pH, and total C, N, and P.
- Permanently mark the plot with metal rods that do not protrude significantly above the ground surface. If not locatable by visual inspection, a metal detector can be used. Do not use metal rods aboveground in grasslands with grazing animals, as they attract domestic grazers which may lead to destruction or to heavy trampling of the plot. Also, any rod or post may serve as perches for birds and thus may lead to atypical species recruitment in the plots or close by. The alternative is to place a stake standing aboveground and then use a compass bearing and distance to guide you to

the corner of the plot, which could be marked with a wooden pole, flush with the soil surface.

- Record all vascular plant species in each plot and allocate percentage live cover to each species as a relevé bounded by the plot edge. Use an ordinal cover score for each species (<1%, 1–5%, 5–25%, 25–50%, 50–75%, 75–100%) within fixed height tiers (<0.1 m; 0.1–0.3 m; 0.3–1 m; 1–2 m; 2–5 m; 5–12 m) and record presence, not cover, of epiphytes (Hurst & Allen 2007b). Note that the lower tiers are more finely divided than those used in forests.
- Photograph or preferably collect material of plants that cannot be readily determined in the field for later determination by experts. Collect flowering or fruiting material (or otherwise fertile material; e.g. fern fronds bearing sori) whenever possible to facilitate determination.
- Record site data for each plot to aid interpretation of differences within and between sites. The first group – altitude, slope, aspect, physiography – should not change and so only need to be recorded when the plot is established. At each census, record any human interference or natural disturbance that has affected the plot.
- See Hurst & Allen 2007b for further details of measurement protocols for each step above.

Birds

- Use 5MBC to sample the bird community.
- Constrain counts to ≤100 m of the recording point.
- Conduct two counts at the centre of each plot, one on arrival and one before departure.
- Conduct bird counts at the same time of the year in repeated measures.
- Independence among samples will be a challenge for mobile fauna at small sites. As much as practicable, aim for samples to be spatially independent.

Ungulates, lagomorphs

- Use the vegetation plots and record presence/absence of pellets by animal group (deer/goat, pig, horse, cattle, lagomorph).
- Regional councils can elect to count pellets and pellet groups, but we emphasise that a substantial amount of information is present in the presence/absence of pellets and many field staff find presence/absence less ambiguous to measure than counts.

Possums

- Monitor possums using chewcards (9 cm × 18 cm cards made of 3 mm plastic corflute) baited with an aniseed-flavoured paste for a single night. Chewcards are suitable for monitoring possums and can yield data on many other pest animal species.
- Chewcards can be attached to 50 cm metal pegs. Pegs must be removed after this period as they can attract grazing animals and perching birds if left out for long periods

- Place 10 cards along transects at 50m intervals, with transects at least 200 m apart. This will not always be feasible on small sites, so many shorter transects may be a compromise.
- If a chewcard has been chewed, then the identity of the mammal that chewed it is recorded (brushtail possum, other mammal, or both: other mammals that chew the chewcards include rodents, stoats, and European hedgehogs; all of these can be distinguished, Sweetapple & Nugent 2011).
- The data are presence of each animal species on each card, summarised as a proportion across all four transects for each plot, and then summarised at a site as the mean proportion across all plots.

5.5.4 Herbaceous communities on stable ground undergoing succession

Vegetation

- Record as for forests, using multiple permanent 20 m × 20 m plots per site.
- Assign plot locations in advance. Map the vegetation according to height/stand development strata, then site-level statistics can be calculated so that if there is a strong height/stand development gradient in the vegetation, stratify sampling effort accordingly to more explicitly understand the nature of vegetation recovery over time.
- Use a GPS to record the location of the centre of each plot.
- Record slope, aspect, topographic index, mean canopy top height, elevation off Topomap, and disturbances (per Hurst & Allen 2007b), and notes to relocate the plot.
- If budget permits, collect a soil sample from the top 100 mm of mineral soil (i.e. after scraping off the litter and organic horizon) from each of four quarters of the 20 m × 20 m plot, pool the samples, and analyse for pH, and total C, N, and P.
- Record all vascular plant species in each plot and allocate percentage live cover to each species as a relevé (Hurst & Allen 2007b).
- Permanently tag and record all species with stems ≥2.5 cm diameter at 1.35 m height above ground (dbh), measure stem diameters to the nearest 0.1 cm, and record the 5 m × 5 m subplots in which they occur (Hurst & Allen 2007a). Record a tally of saplings by species in each subplot (saplings defined as woody stems >1.35 m tall and <2.5 cm dbh).
- Record seedlings and herbaceous species in 24 seedling subplots (0.75 m^2) set at regular points (equidistant 2.5 m along all plot interior edges of 5 m × 5 m subplots), as in forests. Record presence only for species <0.15 m, and presence in height tiers for herbaceous species, and tallies in height tiers ≥0.15 m tall for woody species.
- Photograph or preferably collect material of plants that cannot be readily determined in the field for later determination by experts.

Birds

- Use 5MBC to sample the bird community.
- Constrain counts to ≤100 m of the recording point.

- Conduct two counts at the centre of each plot, one on arrival and one before departure.
- Conduct bird counts at the same time of the year in repeated measures.

Ungulates, lagomorphs

- Collect data on presence of faecal pellets of ungulates and lagomorphs along four 150 m transects, as for forests.

Possums

- Monitor possums using chewcards as for forests, either nailing chewcards to trees c. 30 cm above the ground where possible or attached to 50 cm steel pegs. Remove pegs after use.

5.5.5 Skeletal ecosystems on stable substrates

Vegetation

- Record as for perpetual herbaceous communities, with locations assigned in advance, except that plots that are more than 90% unvegetated should be recorded as such, but not sampled. To obtain an adequate sample, more locations will need to be assigned in advance than will actually be sampled.
- Plot size will be determined based on the predominant vegetation structure on the site. Multiple permanent 2 m × 2 m (turfs, short herbfields), 5 m × 5 m (grasslands), and 10 m × 10 m (open shrublands) plots per site.
- Use a GPS to record the location of the centre of each plot.
- Record slope, aspect, topographic index, mean canopy top height, elevation off Topomap, and disturbances (per Hurst & Allen 2007b), and make detailed notes and a diagram to relocate the plot.
- If possible, permanently mark the plot with metal rods that do not protrude significantly above the ground surface. If not locatable by visual inspection, a metal detector can be used. Do not use sticks aboveground in grasslands with grazing animals, as they attract domestic grazers, which may lead to destruction or to heavy trampling of the plot. Also, sticks may serve as perches for birds and thus may lead to woody species recruitment in the plots or close by. For easier location of plots in the field, sticks – possibly painted – may be placed at fixed distances from the actual plots.
- If the surface is impenetrable (e.g. abundant bedrock near the surface), a battery-operated drill and 6 mm concrete screw can be used to attach a tag to a bedrock plot. Some plots may not be able to be permanently marked, in which case, each time the site is measured it will be a sample because strict repeated measures are not feasible (cf. section 5.5.6).
- Record all vascular plant species in each plot and allocate percentage live cover to each species as a relevé (or recce plot) bounded by the plot edge. Use an ordinal cover score for each species (<1%, 1–5%, 5–25%, 25–50%, 50–75%, 75–100%) within

fixed height tiers (<0.1 m; 0.1–0.3 m; 0.3–1 m; 1–2 m; 2–5 m; 5–12 m) and record presence, not cover, of epiphytes (Hurst & Allen 2007b).

- Photograph or preferably collect material of plants that cannot be readily determined in the field for later determination by experts. Collect flowering or fruiting material (or otherwise fertile material; e.g. fern fronds bearing sori) whenever possible to facilitate determination.
- Record site data for each plot to aid interpretation of differences within and between sites. The first group – altitude, slope, aspect, physiography – should not change and so only need to be recorded when the plot is established. At each census, record any human interference or natural disturbance that has affected the plot. Estimate the relative percentage (at ground level) of vegetation, bare ground and cryptogams (mosses and lichens) to the nearest 5% (sum of these three classes = 100). The area available to higher plants shows a strong relationship to composition and change in skeletal systems (e.g. Wiser & Buxton 2008; Kirkpatrick et al. 2002) and changes in these categories can strongly reflect human impacts (e.g. Cole 1995.)
- See Hurst & Allen 2007b for further details of measurement protocols for each step above.

Birds

- Use 5MBC to sample the bird community.
- Constrain counts to ≤100 m of the recording point.
- Conduct two counts at the centre of each plot, one on arrival and one before departure.
- Conduct bird counts at the same time of the year in repeated measures.
- Independence among samples will be a challenge for mobile fauna at small sites. As much as practicable, aim for samples to be spatially independent.

Ungulates, lagomorphs

- Use the vegetation plots and record presence/absence of pellets by animal group (deer/goat, pig, horse, cattle, lagomorph).
- Regional councils can elect to count pellets and pellet groups, but we emphasise that a substantial amount of information is present in the presence/absence of pellets and many field staff find presence/absence less ambiguous to measure than counts.

Possums

- Monitor possums using chewcards (9 cm × 18 cm cards made of 3 mm plastic corflute) baited with an aniseed-flavoured paste for a single night. Chewcards are suitable for monitoring possums but can also yield data on many other pest animal species.
- Chewcards can be attached to 50 cm metal pegs. In habitats where the surface is impenetrable, such as many erosion pavements, add more sample points where it is possible to insert pegs. Pegs must be removed after this period as they can attract grazing animals and perching birds if left out for long periods.

- Ideally, cards would be along transects at least 200 m apart with 10 cards per transect, 50 m apart. This will not always be feasible on small sites, so many shorter transects may be a compromise.
- If a chewcard has been chewed, then the identity of the mammal that chewed it is recorded (brushtail possum, other mammal, or both: other mammals that chew the chewcards include rodents, stoats, and European hedgehogs; all of these can be distinguished, Sweetapple & Nugent 2011).
- The data are presence of each animal species on each card, summarised as a proportion across all four transects for each plot, and then summarised at a site as the mean proportion across all plots.

5.5.6 Skeletal ecosystems on mobile substrates

Vegetation

- Establish many small (1 m^2) plots using multiple transects to capture environmental gradients at each site (e.g. by placing transects perpendicular to the shoreline, river channel, or lake margin).
- Allocate transects and plots either randomly, regularly, or using a spatially balanced approach. The number of transects and plots required to sample these substrates (e.g. dunes, braided rivers) remains to be determined, but indicative numbers could be between 20 and 60 plots per site, based on a survey across New Zealand's gravel beaches, where larger beaches had more plots (Wiser et al. 2010).
- Transects should originate inland and extend to where vegetation is capable of surviving (e.g. on a beach this would be directly above the high-water mark, or on a braided river towards a likely flood mark).
- All points along each transect should have an equal probability of being sampled so avoid using a fixed value for the first plot (e.g. 2 m along from the origin); start at 0 and draw random numbers to sample the full width of the beach.
- Use a GPS to record the transect origins and centre of each plot. Ascertain whether permanent marking of transects is permissible (e.g. after consultation with mana whenua or private landowners). If permissible, mark the landward end of each transect using a waratah with labelled permolat, or a plastic electric fence post, but do not mark individual plots. Each time the site is measured it will be a sample, because strict repeated measures are not feasible in mobile landforms.
- If a plot supports no vegetation, this should be recorded. On sites where vegetation cover is sparse, it will be important to sample an adequate number of vegetation plots.
- Record all vascular plant species in each plot and record percentage live cover for each species. There is currently no consistency among regional councils in how cover is scored (i.e. ordinal cover scores in fixed height tiers vs a single percentage cover score), and we lack data to evaluate the performance of each approach. We recommend a single ordinal cover score for each species (<1%, 1–5%, 5–25%, 25–50%, 50–75%, 75–100%). At this point we do not know how much information we lose from not recording cover within fixed height tiers (as per the relevé or 'recce' method; Hurst & Allen 2007b); further work is needed to evaluate whether cover scores in

height tiers would be more sensitive to detecting changes in indigenous dominance over time in response to management. Regional councils can elect to record actual percentage cover, as these data can be readily converted to ordinal cover scores for comparisons with other data.

- Record the mean canopy top height of vegetation within each plot to the nearest 0.1 m. This information can be used for reporting on structural change (e.g. transitions from low-statured grasslands to shrublands).
- A high taxonomic standard is desirable, but where this is not possible (e.g. *Rytidosperma* spp. in coastal ecosystems), identify to genus or family. Aggregating species according to biostatus is undesirable because it is difficult to ascribe biostatus without knowing the species.
- Spring is optimal for identifying grasses and herbs that mature and dry out as summer progresses.
- Forested dunes (e.g. Ōreti river mouth, Southland) will require a forest method, but still use transects perpendicular to the coast. Small (1 m^2) plots can be nested within large (20 m × 20 m) plots to maintain comparability among other coastal and forested systems.
- Subjectively located plots can be added to determine composition associated with sparsely distributed species, or to capture distinctive combinations of species not otherwise captured on transects or plots. The goal of sampling is to measure the 'average' community, and unless sampling is very intensive at a site, some combinations of species will be not sampled. Prefix plot names with 'Subj', or something similar, to clearly indicate that these plots are not part of the unbiased sample.
- At the plot scale, record the percentage of bare ground for use in analyses as a covariate. At the site scale, record the percentage (to the nearest 5%) of different land cover types (definitions following the NZ Landcover Database) that occur within 50 m of the beach, braided river bed or lake margin for use in analyses as a covariate.
- Record mean canopy top height and disturbances (per Hurst & Allen 2007b), and make notes to relocate the plot.

Birds

- Use 5MBC to sample the bird community.
- Constrain counts to ≤100 m of the recording point.
- Conduct bird counts at the same time of the year in repeated measures.
- Independence among samples will be a challenge for mobile fauna at small sites. As much as practicable, aim for samples to be spatially independent. A suggested approach is to complete two 5MBCs at two of the plots on each transect (e.g. every second and fourth plot). Separate each 5MBC by 5 minutes.

Ungulates, lagomorphs

- Use the vegetation plots and record presence/absence of pellets by animal group (deer/goat, pig, horse, cattle, lagomorph).
- RCs can elect to count pellets and pellet groups, but we emphasise that a substantial amount of information is present in the presence/absence of pellets and many field staff find presence/absence less ambiguous to measure than counts.

Possums

- Chewcards are suitable for monitoring possums and can yield data on many other pest animal species.
- We recommend single night sampling for possums (DOC 2020).
- Ideally, chewcards would be along transects at least 200 m apart with 10 cards per transect, 50 m apart. This will not always be feasible in dune lands or on beaches, so many shorter transects may be a compromise.
- The data are presence of each animal species on each chewcard, summarised as a proportion on each transect, and then summarised at a site as the mean proportion across all transects.

5.5.7 Ecosystems that are dangerous to sample

Some ecosystems cannot be safely sampled using conventional field sampling that requires the presence of the data collector close to the biota. Tall cliffs, for example, may require use of climbing equipment, but the use of climbing routes can bias data (Strumia et al 2020); some geothermal systems may be unsafe to sample by any means.

Vegetation

Optical tools (binoculars, telescopes, telephoto lenses) have been used in such situations (e.g. Alfaro-Saiz et al. 2019 to monitor a rare herb; West 2002 to detect invasive plants on Rangitāhua/Raoul Island). In recent years, Unmanned aerial vehicles (UAVs, i.e. drones) have also been trialed (e.g. Strumia et al. 2020 for vegetation monitoring on coastal cliffs; Beadel et al. 2018 for mapping geothermal vegetation types). The obvious drawback is that only those taxa that are visible in the canopy and can be identified at this level of visual resolution can be assessed. Such an approach may be appropriate to assess change in canopy dominance and to identify the presence of large-statured exotic plants.

Before a drone-based or binocular-based method is adopted, a more thorough literature review, followed by New Zealand-based trials, is required. Such a trial would need to quantify the correlation between assessments of canopy cover/species presence data using these technologies versus traditional ground-based approaches. This would need to be done at accessible locations where results acquired using alternative techniques can be directly compared.

Ungulates

Optical tools (binoculars, telescopes) have been used to determine densities of ungulates in steep terrain; for example, two observers, working independently, used binoculars and telescopes mounted on tripods looking across-valley to rock bluffs opposite (spanning 300 m elevation) to determine Himalayan tahr (*Hemitragus jemlahicus*) abundance (Tustin & Challies 1978; Tustin & Parkes 1988), and these data can be used as repeated measures, allowing assessment of changes in abundance after control (Forsyth & Hickling 1997).

5.6 Cross-ecosystem comparisons and change with time

Except for ecosystems that are dangerous to sample, our approach has been to recommend square plots as the basic unit to sample vegetation in all groups of ecosystems, on and around which birds and pest mammals can be sampled. The square plots are of variable size, but are scalable within ecosystems; for example, from 5 m × 5 m herbaceous wetland plots to 10 m × 10 m wetland plots sampling willows, to 20 m × 20 m forest plots in wetlands (e.g. for kahikatea). Scalability also applies across ecotones among broad ecosystem groups (e.g. from 1 m × 1 m plots for sampling dunes to 5 m × 5 m herbaceous wetland plots in inland dunes to 20 m × 20 m forest plots to sample dune kānuka forests). Scalability in plot size should enable transitions from one method to another because of successional change.

5.7 Issues of sampling

Sampling intensity within ecosystems and within managed sites is outside the remit of this report, but it is essential to determine an appropriate number of sample points for any given ecosystem that will give the power to detect change in measures of ecological integrity (Richardson et al. 2013). Initial reconnaissance measurements to determine within-site or within-ecosystem heterogeneity are valuable for determining sampling intensity and major environmental gradients, which might result in stratification of sites or ecosystems within them. This is an important precursor step before establishing permanent plots. Power analyses can be used to determine the number of plots needed to detect changes in measures of ecological integrity of a given magnitude with a given level of confidence (e.g. Mason & Bellingham 2018).

We acknowledge that replication of sample points in some managed sites will be difficult, or even impossible. Some ecosystems will be so small that replication of sample points or achieving independence among them will be impossible. Single plots in ecosystems are still worth installing to determine change among multiple small sites.

5.8 Best use of historical data

Successful biodiversity monitoring programmes build on past monitoring investments. It is axiomatic that if there are existing permanent sampling points in managed sites and ecosystems they should be incorporated to the greatest extent possible in new sampling schema. Some existing sampling points may be biased in placement, so new sample points would need to mitigate any biases.

Some of the methods we recommend in this report are departures from widely used standard approaches, especially for measuring vegetation change in perpetual herbaceous communities (including some woody components). Widely used methods in the past have focused on measuring structure at multiple points, typically along transects (e.g. Dickinson et al. 1992). If these methods are in use and provide historical data in prioritised ecosystems, then we recommend that these methods continue to be used.

Regional councils might weigh up the advantages and disadvantages of changing from past methods to new ones. Should they decide to change, during the transition phase they would do best to measure their plots with both methods for two cycles. An example that arose in discussions for this report concerned current methods for monitoring vegetation in sand dunes. Current methods differ among councils, and it would be wise to use the methods we recommend for them (section 5.5.6) alongside current methods during a transition phase to ascertain the potential loss of information and the degree to which data are consistent between the two methods.

5.9 Monitoring methods based on mātauranga Māori

Monitoring methods based on mātauranga incorporate a Māori world view that embraces human inter-relationships with the natural world (whakapapa) and spirituality, as well as human use of the natural world (e.g. mahinga kai) (Lyver, Timoti, Jones et al. 2017; Lyver, Timoti, Gormley et al. 2017). Since there is strong expression of local (iwi or hapū) identity in the inter-relationships, as well as distinct local tikanga, methods will not necessarily be widely applicable and may be restricted to a local area. In an assessment of the complementarity of monitoring methods described in this report for forests (section 5.5.1) and monitoring methods developed by the Tūhoe Tuawhenua Trust and by Ngāti Whare, there was a large amount of overlap (Lyver et al. 2018). This suggests that, for some ecosystems, there could be complementary methods for monitoring between those recommended in this report and those developed by Māori, and if both are applied then this may guarantee an enduring cross-cultural monitoring system for ecosystems of interest (Lyver et al. 2018).

5.10 Components of biodiversity considered out of scope for widespread standardised methods

The methods we recommend for ground-based monitoring to determine management effectiveness on prioritised ecosystem sites are a minimum that allows ready integration with national and regional data collected through Tier 1 systematic monitoring.

We recognise that this approach does not address some key pressures on ecosystems, especially most species of mammalian predators. Rodents and mustelids are key pressures on native biota, especially birds, reptiles, and invertebrates, and, in the case of rodents, they are also important seed predators. Strongly coupled predator relationships between rodents, mustelids, and birds (rodents as primary prey of mustelids, and birds as secondary prey of mustelids) are a basis for intensive management of rodents and mustelids in many ecosystems. We have ruled them out of scope for a standardised monitoring effort with maximum frequency of annual measurements because numbers of rodents and mustelids fluctuate during the year (e.g. Ruscoe et al. 2004), requiring a high frequency of measurement, so a single annual measurement will not necessarily be representative.

We recognise that an evidence base for management effectiveness will involve estimates of rodent and mustelid density in many ecosystems, including forests (Brown et al. 1996) and wetlands (Gillies & Brady 2018), and that regional councils may choose to employ well-developed methods to measure rodents and mustelids in prioritised ecosystems. Mammalian predator guilds are distinct in skeletal ecosystems on mobile substrates. For example, management of braided rivers to optimise the nesting success of birds requires managing mustelids, feral cats, and hedgehogs (Cameron et al. 2005), which councils may also choose to measure in addition to metrics we have outlined.

We have not included targeted approaches to measurements of rare native species (plants or birds), and we have not included methods to measure nocturnal bird communities (kiwi, owls, and, potentially, burrowing seabirds) and bats. For some ecosystems, a metric of management success could be enhanced abundances of some species that are sensitive to habitat degradation, or sensitive to pressures from non-native species (e.g. mistletoes in forests browsed by brushtail possums; Sweetapple et al. 2002; or several bird species confined to wetlands that are likely to be reduced in abundance by predatory mammals; O'Donnell et al. 2015). Councils may choose to include additional measurements of individual species of interest alongside the methods we recommend, and we suggest that, if they do, they should be coupled with the same sample points that quantify vegetation, bird communities, and measures of ungulates, possums, and lagomorphs to ensure maximum interpretability of results.

While enhancing diversity and abundance of reptiles is often a desired management outcome in coastal ecosystems, we do not recommend including measuring reptile abundance in standardised monitoring protocols for four reasons:

- handling reptiles requires Wildlife Act Authorisation
- lizard data can be challenging to analyse because of the very high numbers of data that are zeros (Handford et al. 2018)
- the results, when obtained, are known to be difficult to interpret because changes in abundance reflect not only true abundance at the site, but also weather, time of day and year
- behavioural changes can themselves be induced by monitoring.

Even studies comparing intensive mark-recapture methods from pitfall traps with artificial structure counts have found complex relationships between estimates of total population size and daily counts at a site (Lettink et al. 2011). Our experience is that many organisations collect small amounts of data on reptiles that cannot be interpreted, and given the potential stress to the animals, we suggest using a specialist herpetologist if there are specific questions related to reptiles. The same applies to methods to assess populations of native amphibians, at least one species of which faces further pressures from the non-native fungal pathogen *Batrachochytrium dendrobatidis* (Bell et al. 2004).

Insects play a critical role in ecosystem processes and services (e.g. pollination), but we have not advocated for their measurement. Assessing taxonomic diversity across all insects is an unrealistic goal across most regions because New Zealand's insect fauna is 90–100% endemic in most orders (McGlone 2006) with large proportions (e.g. in Coleoptera, Diptera) undescribed. Therefore, the capacity issues in skills to deal with insect samples are substantial. Monitoring insects would also substantially increase field costs and the amount of time spent at a site, since, for example, diurnal vs. nocturnal species, or volant vs. soil-dwelling, species all require different sampling techniques.

However, some components of insect communities have been measured in response to management at some sites in New Zealand (e.g. Watts et al. 2014). Simple metrics, such as changes in biomass of insects, may be informative (Hallmann et al. 2017), but equivalent suitable methods remain undeveloped in New Zealand. Other metrics may be suitable for use for pest insects, notably non-native wasps. Recently, 5-minute wasp counts (total counts for *Vespula* spp. and ordinal abundance categories for *Polistes* spp. within 75 m of a central sampling location) were used in forests on northern New Zealand islands (Schmack et al. 2020) and may be suitable for more widespread use. Soil microbial communities, including fungi and prokaryotes (bacteria & Archaea), play vital roles in ecosystems and provide critical ecosystem services, but we have not recommended their measurement.

5.11 Issues of capacity and skills needed to monitor change

In discussions during the formulation of this report, some regional councils identified lack of capacity in their regions (either staff or contractors resident in the region) as a barrier to being able to implement some of the methods, especially those that require species identification (vegetation and bird methods).

Within regions, training of those implementing the methods is an essential first step to ensure standardisation of implementation. Observer error is a pervasive component of all sampling, and a portion of this error is due to characteristics of the plant or bird community and the environment associated with the sampling, and is unavoidable (Morrison 2016). Other components of observer error associated with species identification and species concepts can be reduced through initial reconnaissance. For any given prioritised sites, or groups of sites, field teams would benefit from initial reconnaissance before establishing the permanent sampling points. Using available information (e.g. species lists, any data from extant plots), field teams might compile and add to species lists during the reconnaissance, familiarising themselves with the species present, and working out 'difficult' groups of species (e.g. grasses and sedges, finches) together.

Morrison (2016) recommended that the degree of observer error associated with vegetation sampling should be quantified and reported along with the results; while it may not be possible to determine the accuracy of observer estimates, precision can and should be documented. Double sampling 10% of plots may represent a reasonable trade-off between resources and the need for error reporting. Morrison (2016) concluded that, in the absence of error validation, any differences <25% should be viewed with scepticism, as all of the documented change could simply be due to observer error. Mason et al. (2018)

quantified observer error in the measurement of vegetation in the same 20 × 20 m plots in New Zealand forests by different field teams, weeks apart, and found that the greatest discrepancies occurred in species composition among teams, but there was much less for measurements of stem diameters.

5.12 Archiving monitoring data

Archiving vegetation monitoring data in the National Vegetation Survey databank (NVS) https://nvs.landcareresearch.co.nz/Home/Index ensures long-term data security and accessibility and provides a ready mechanism to support data sharing, if desired. The software package NVS-Express, or a NVS-specific EXCEL data entry template, can be downloaded from https://nvs.landcareresearch.co.nz/Data/Tools. Validation and mechanisms to ensure data integrity are more rigorous in NVS-Express, but some may find an EXCEL template easier to use. Wetland data can be sent to Neil Fitzgerald (Manaaki Whenua – Landcare Research, Hamilton; fitzgeraldn@landcareresearch.co.nz).

Bird count data can be entered into eBird (https://ebird.org/newzealand/home). There is currently no centralised repository for ungulate, possum, and lagomorph data. Templates for consistent formatting of these data need to be developed among regional councils, in collaboration with DOC. We recommend storing data for now in csv files with continuous columns (Broman & Woo 2018).

6 Discussion

This report recommends methods that can be adopted readily, and by using them, benefits of management should be evident within 5 years in some ecosystems, depending on how effective the management was, pest density–impact relationships, etc. In other ecosystems it will take longer to assess the benefits of management (>10 years) because of the inherently slow growth rates of dominant species. For some ecosystems, research and development of appropriate methods may take up to 5 years, and a further 5 years to report management effectiveness (Watson & Novelly 2004). Benefits will be sustained if the advice is adopted widely, and if consistency across councils is maintained.

The recommendations in this report are intended to provide the harmonised evidence base needed to report the effectiveness of councils' management of ecosystems they have prioritised. An evidence base will expose ineffective management practices and highlight effective ones. Compelling evidence from consistent monitoring maintained over time will benefit the environment, because transparent data should trump the anecdotal evidence that is often the basis of conflict in current circumstances. The advice will ensure a flow of comparable information within and among councils, which will improve environmental management widely by enabling other managers (e.g. DOC, NGOs, private landowners) to contribute to, and learn from, this evidence base.

Positive changes in how councils operate are most likely to be achieved when quantitative data derived from monitoring are comparable, enabling councils to compare outcomes, and understand local departures from regional and national trends. This is likely to result in entrenchment of a culture of defensible monitoring activities and a decline in *ad hoc* methods and reliance on anecdote.

The methods in this report are suitable for use in experimental treatments implemented on prioritised sites (e.g. fenced exclosure plots to determine the effects of ungulates in ecosystems, which could be assigned at random to sample points; e.g. Bellingham, Richardson et al. 2016).

The methods recommended in this report have not embraced newer technologies such as cameras and drones, or acoustic recorders for birds. We note that a recent New Zealand study compared camera traps with chewcards and found that, while the two methods produced comparable data (chewcards produced more accurate assessments of hedgehog detections), chewcards were cheaper, easier to use, and had much less risk of theft than cameras (Nottingham et al. in press). Nonetheless we expect there will be a growing role for new technologies improve, and the capacity to automate and have confidence in the data derived from them (e.g. from bird acoustic recorders) increases.

In all cases, we recommend evaluating a range of new technologies over a trial period alongside existing methods. This is not just to ensure comparability of data during a transition period (section 5.8), but also to determine the best new technology. For example, chewcards replaced leg-hold traps in DOC Tier 1 after evaluation alongside alternatives (e.g. wax tags) and were found to calibrate best with existing data from trap-catch indices (Forsyth et al. 2018).

Resources

During the development of this report several regional councils identified a lack of funding as one barrier to increased biodiversity monitoring, and all councils stressed the need to choose monitoring techniques that are as cost-effective as possible. While we have borne those concerns in mind, a full cost–benefit analysis of the biodiversity monitoring recommended here is outside the scope of this report. Nevertheless, we believe that the value of monitoring biodiversity on private land extends beyond regional councils and should be considered in any discussion of how to fund such monitoring.

A think piece commissioned by regional councils concluded that the benefits of Tier 1 monitoring of terrestrial biodiversity extend beyond regional councils: ‘Hence some contribution from central government may be justifiable’ (Willis 2016). We agree, and believe this view should be extended to the benefits of Tier 2 monitoring.

Often the costs of biodiversity monitoring are all too real and quantifiable, while the benefits cannot be given a reliable dollar figure. We suggest, instead, that it is more useful initially to discuss principles that would help assign value and benefits as opposed to quantifying costs. For example, DOC’s monitoring on public land is designed to meet their responsibilities; however, it also benefits regional councils, because that information can contribute to their own management decisions and regional policy-making.

Similarly, having standardised and readily accessible data about biodiversity on private land from regional councils would have value for DOC and other Crown agencies (e.g. MfE). For both Crown agencies and regional councils, then, an initial question is, ‘How do we estimate the value of the additional work necessary to standardise Tier 2 monitoring on private land?' We do not attempt to answer this question here. However, we do endorse the principle that because the benefits of doing this work are spread across multiple parties, costs should be too.

7 Recommendations

We recommend that a consortium of regional councils work with DOC and researchers to undertake the following activities.

1. Conduct widespread field trials of the methods used for skeletal ecosystems on mobile substrates (especially on dunes and braided rivers) at a range of scales. These should span environmental gradients, and should consider constrained or truncated gradients (e.g. dunes truncated by roads) vs. less constrained, spatially extensive ecosystems). The approach should seek to integrate with and compare among existing methods used in these ecosystems.
2. Further develop appropriate means to quantify changes in ecological integrity of sites that are dangerous to sample, especially using remote technology (e.g. UAVs, fixed camera points), which is also potentially useful in other low-stature ecosystems such as wetlands.
3. Evaluate suitable levels of replication and spatial independence for sampling birds and ungulates. Johnson (1995) recommended a minimum distance between sample points for birds of 200 m, but this may differ across ecosystems (forested habitat vs. non-woody ecosystems).
4. Evaluate the advantages and disadvantages and information loss of changing from the existing standard methods to the methods we recommend. In particular, we recommend evaluating the advantages and disadvantages of changing from the methods used to evaluate changes in structure in herbaceous communities (especially perennial grasslands; Wraight 1962; Dickinson et al. 1992; Wiser & Rose 1997) to the fixed-area square plots we recommend. In all cases, we recommend maintaining existing and recommended methods for a transition period to allow the evaluation of advantages, disadvantages, and information loss.
5. Evaluate the information loss between 3-night sampling for rodents on dunes (as currently conducted by Greater Wellington Regional Council) compared with the single-night sampling we recommend on the basis of possums only (cf. DOC 2020).
6. Evaluate existing forest plot data to assess the information gained from the current seedling plot method or from altering minimum dbh in permanent plots. This would need to be collaborative with DOC and MfE to ensure that any recommendations from the work aligns with existing monitoring efforts.
7. Evaluate the areal extents of ecosystems of interest across all regional councils to determine optimal sampling in ecosystems that are very small (e.g. <1 ha), especially those in which human disturbance caused by establishing the plots may compromise their management. If the methods in this report are unsuitable for very small sites, then further method development will be necessary.

8 Determine optimal sampling intensities and placement of samples for prioritised ecosystems. Ideally similar sampling methods should be developed between DOC and regional councils that are easily comprehended and straightforward to use. DOC has recently developed an algorithm for assigning plot locations within managed sites (van Dam-Bates et al. 2018). It is technically difficult to incorporate underlying abiotic gradients (which may result in ecotones or strata according to underlying geology) into this algorithm. Other methods (e.g. generation of random locations and transects) can achieve spatial balance and an unbiased sample. Ideally, similar sampling methods that are technically accessible should be developed between DOC and regional councils.

9 Evaluate the opportunities to implement the methods in this report alongside cultural indicators developed by mana whenua. For example, implementing wetland methods as outlined in this report might be conducted best by working alongside iwi and hapū partners who are implementing methods to assess wetland health and cultural wetland values (Taura et al. 2017). To achieve concurrent implementation of cultural indicators and methods in this report, early engagement with iwi and hapū is vital.

10 Determine opportunities to integrate specific measures of mammalian herbivory (e.g. foliar browse scores) into plots, as recommended in this report, rather than running this as a separate exercise on transects.

11 Work with DOC to determine options for chewcards made of materials other than corflute to minimise plastic waste on site.

12 Work with DOC to investigate the suitability of 5-minute wasp counts (Schmack et al. 2020) as a technique to determine pest wasp abundance across ecosystem types and in response to management.

13 Develop and maintain databases suitable for use among all agencies to contain standardised data for pest mammals and birds.

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Appendix 1 – Ecosystem types according to Singers & Rogers 2014 and suggested plot-based methods for each.

This table provides *suggestions* for which method to apply for each ecosystem. Because the descriptions in Singers & Rogers (2014) are necessarily generalised to work across many sites, they often do not state vegetation stature, the extent of bare ground, or the presence of hazards so while the suggested methods below should work in many instances, the user must determine the most appropriate method for a site.

Code	Short Name	Most Likely Method	Other Possible Method(s)
SF1	Kermadec pōhutukawa forest	Forest	
WF1	Tītoki, ngaio forest	Forest	
WF2	Tōtara, mataī, ribbonwood forest	Forest	
WF3	Tawa, tīitoki, podocarp forest	Forest	
WF4	Pōhutukawa, pūriri, broadleaved forest [Coastal broadleaved forest]	Forest	
WF5	Tōtara, kānuka, broadleaved forest [Dune forest]	Forest	
WF6	Tōtara, mataī, broadleaved forest [Dune Forest]	Forest	
WF7	Pūriri forest	Forest	
WF8	Kahikatea, pukatea forest	Forest	Wetland
WF9	Taraire, tawa, podocarp forest	Forest	
WF10	Kauri forest	Forest	
WF11	Kauri, podocarp, broadleaved forest	Forest	
WF12	Kauri, podocarp, broadleaved, beech forest	Forest	
WF13	Tawa, kohekohe, rewarewa, hīnau, podocarp forest	Forest	
WF14	Kāmahi, tawa, podocarp, hard beech forest	Forest	
MF1	Tōtara, tītoki forest	Forest	
MF2	Rimu, mataī, hīnau forest	Forest	
MF3	Mataī, tōtara, kahikatea, broadleaved forest	Forest	
MF4	Kahikatea forest	Forest	Wetland
MF5	Black beech forest	Forest	
MF6	Kohekohe, tawa forest	Forest	
MF7	Tawa, kāmahi, podocarp forest	Forest	
MF8	Kāmahi, broadleaved, podocarp forest	Forest	
MF9	Tānekaha forest, locally with hard beech	Forest	
MF10	Tōtara, mataī, kahikatea forest	Forest	
MF11	Rimu forest	Forest	
MF12	Rātā, hard beech, kāmahi forest	Forest	
MF13	Kahikatea, northern rātā, kāmahi forest	Forest	

Code	Short Name	Most Likely Method	Other Possible Method(s)
MF14	Kahikatea, silver pine, kāmahi forest	Forest	Wetland
MF15	Kahikatea, mataī, Westland tōtara, rimu forest	Forest	
MF16	Rimu forest	Forest	
MF17	Rimu, kāmahi, tāwheowheo forest	Forest	
MF18	Yellow silver pine, mānuka forest	Forest	
MF19	Kāmahi, rimu, miro, southern rātā forest	Forest	
MF20	Hard beech forest	Forest	
MF21	Tawa, kāmahi, rimu, northern rātā, black beech forest	Forest	
MF22	Tawa, rimu, northern rātā, beech forest	Forest	
MF23	Chatham Island akeake, karamū, māhoe, ribbonwood forest	Forest	
MF24	Rimu, tōwai forest	Forest	
MF25	Kauri, tōwai, rātā, montane podocarp forest	Forest	
CLF1	Hall's tōtara, mountain celery pine, broadleaf forest	Forest	
CLF2	Hall's tōtara forest [Dune forest]	Forest	
CLF3	Podocarp, ribbonwood, kōwhai forest	Forest	
CLF4	Kahikatea, tōtara, mataī forest	Forest	
CLF5	Mataī, Hall's tōtara, kāmahi forest	Forest	
CLF6	Kāmahi, southern rātā, podocarp forest	Forest	
CLF7	Rimu, kāmahi, beech forest	Forest	
CLF8	Silver beech, kāmahi, southern rātā forest	Forest	
CLF9	Red beech, podocarp forest	Forest	
CLF10	Red beech, silver beech forest	Forest	
CLF11	Silver beech forest	Forest	
CLF12	Silver beech, mountain beech forest	Forest	
CDF1	Pāhautea, Hall's tōtara, mountain celery pine, broadleaf forest	Forest	
CDF2	Dracophyllum, mountain celery pine, Olearia, Hebe scrub [Subalpine scrub]	Forest	Herbaceous or Stable skeletal
CDF3	Mountain beech forest	Forest	
CDF4	Hall's tōtara, pāhautea,	Forest	
CDF5	Hall's tōtara, pāhautea, kāmahi, southern rātā forest	Forest	
CDF6	Olearia, Pseudopanax, Dracophyllum scrub [Subalpine scrub]	Forest	Herbaceous or Stable skeletal
CDF7	Mountain beech, silver beech, montane podocarp forest	Forest	

Code	Short Name	Most Likely Method	Other Possible Method(s)
AL1	Narrow-leaved and slim snow tussock tussockland/shrubland	Herbaceous	Stable skeletal
AL2	Slim and mid-ribbed snow	Herbaceous	Stable skeletal
AL3	Red tussock tussockland/shrubland	Herbaceous	Stable skeletal
AL4	Mid-ribbed and broad-leaved snow tussock tussockland/shrubland	Herbaceous	Stable skeletal
AL5	Mid-ribbed, broad-leaved, red and carpet grass tussockland/shrubland	Herbaceous	Stable skeletal
AL6	Mid-ribbed and narrow-leaved snow tussock tussockland/shrubland	Herbaceous	Stable skeletal
AL7	Pungent snow tussock tussockland/shrubland	Herbaceous	Stable skeletal
AL8	Stewart Island snow tussock tussockland/shrubland	Herbaceous	Stable skeletal
AL9	Subantarctic snow tussock tussockland/shrubland	Herbaceous	Stable skeletal
AH1	Gravelfield/stonefield [Fellfield]	Stable skeletal	Herbaceous
AH2	Dracophyllum muscoides cushionfield	Stable skeletal	Herbaceous
AH3	Gravelfield/stonefield, mixed species cushionfield	Stable skeletal	Herbaceous
AH4	Woolly moss, bristle tussock, blue tussock mossfield/tussockland/stonefield	Stable skeletal	Herbaceous
TI1	Bog pine, mountain celery pine scrub/forest	Forest	Herbaceous or Stable skeletal
TI2	Kānuka, Olearia	Forest	Herbaceous or Stable skeletal
TI3	Monoao scrub/lichenfield	Herbaceous	Stable skeletal
TI4	Coprosma, Olearia scrub [Grey scrub]	Forest	Herbaceous or Stable skeletal
TI5	Bog pine, mountain celery pine, silver pine scrub/forest	Forest	Herbaceous or Stable skeletal
TI6	Red tussock tussockland	Wetland	Herbaceous or Successional
WL1	Mānuka, gumland grass tree, Machaerina scrub/sedgeland [Gumland]	Wetland	Forest or Herbaceous or Stable skeletal
WL2	Mānuka, greater wire rush restiad rushland	Wetland	Forest or Herbaceous or Stable skeletal
WL3	Bamboo rush, greater wire rush restiad rushland	Wetland	Herbaceous or Stable skeletal
WL4	Mānuka, lesser wire rush, tangle	Wetland	Forest or Herbaceous or Stable skeletal
WL5	Chatham Island bamboo rush restiad rushland	Wetland	Herbaceous or Stable skeletal
WL6	Lesser wire rush, tangle fern restiad rushland/fernland	Wetland	Herbaceous or Stable skeletal
WL7	Tall tussock tussockland	Wetland	Herbaceous or Successional
WL8	Herbfield/mossfield/sedgeland	Wetland	Herbaceous or Stable skeletal
WL9	Cushionfield	Wetland	Herbaceous or Stable skeletal

Code	Short Name	Most Likely Method	Other Possible Method(s)
WL10	Oioi restiad rushland/reedland	Wetland	Herbaceous or Stable skeletal
WL11	Machaerina	Wetland	Herbaceous or Stable skeletal
WL12	Mānuka, tangle fern scrub/fernland	Wetland	Forest or Herbaceous or Stable skeletal
WL13	Sphagnum mossfield	Wetland	Herbaceous or Stable skeletal
WL14	Herbfield [Ephemeral wetland]	Wetland	Herbaceous or Stable skeletal or Mobile skeletal
WL15	Herbfield [Lakeshore turf]	Wetland	Herbaceous or Stable skeletal or Mobile skeletal
WL16	Red tussock, Schoenus pauciflorus tussockland	Wetland	Herbaceous or Stable skeletal
WL17	Schoenus pauciflorus sedgeland [Alpine seepages/flushes]	Wetland	Herbaceous or Stable skeletal
WL18	Flaxland	Wetland	Herbaceous or Stable skeletal
WL19	Raupō reedland	Wetland	Herbaceous or Stable skeletal
WL20	Coprosma, twiggy tree daisy scrub	Wetland	Forest or Herbaceous or Stable skeletal
WL21	Swamp akeake, Chatham Island karamū, Coprosma propinqua var. martinii short forest/flaxland	Wetland	Forest or Herbaceous or Stable skeletal
WL22	Carex, Schoenus pauciflorus sedgeland	Wetland	Herbaceous or Stable skeletal
DN1	Beach morning glory, knobby clubrush vineland/sedgeland	Mobile skeletal	
DN2	Spinifex, pīngao grassland/sedgeland	Mobile skeletal	
DN3	Pīngao sedgeland	Mobile skeletal	
DN4	Pīngao sedgeland, mega-herbfield	Mobile skeletal	
DN5	Oioi, knobby clubrush sedgeland	Wetland	Herbaceous or Stable skeletal or Mobile skeletal
EP1	Rockland	Stable skeletal	Herbaceous or Dangerous
CL1	Pōhutukawa treeland/flaxland/rockland	Dangerous	Forest or Stable Skeletal
CL2	Ngaio, taupata treeland/herbfield/rockland	Dangerous	Forest or Stable Skeletal
CL3	Coprosma, Muehlenbeckia shrubland/herbfield/rockland	Dangerous	Herbaceous or Stable skeletal
CL4	Chatham Island akeake, Hebe, māhoe treeland/herbfield/rockland	Dangerous	Forest or Stable Skeletal
CL5	Harakeke, Hebe elliptica flaxland/rockland	Dangerous	Herbaceous or Stable skeletal
CL6	Hebe, wharariki flaxland/rockland	Dangerous	Herbaceous or Stable skeletal
CL7	Pachystegia, Carmichaelia shrubland/tussockland/rockland	Dangerous	Herbaceous or Stable skeletal
CL8	Helichrysum, Melicytus shrubland/tussockland/rockland	Dangerous	Herbaceous or Stable skeletal
CL9	Parataniwha, Machaerina sinclarii herbfield/sedgeland	Dangerous	Herbaceous or Stable skeletal

Code	Short Name	Most Likely Method	Other Possible Method(s)
CL10	Kiokio fernland/rockland	Dangerous	Herbaceous or Stable skeletal
CL11	Mountain tutu, Hebe, wharariki, Chionochloa shrubland/tussockland/rockland	Dangerous	Herbaceous or Stable skeletal
SC1	Gravelfield [Screes and boulderfields]	Mobile skeletal	Dangerous
BR1	Hard tussock, scabweed	Mobile skeletal	
BR2	Scabweed gravelfield/stonefield	Mobile skeletal	
BR3	Bristle tussock, Raoulia, Muehlenbeckia gravelfield/sandfield	Mobile skeletal	
SA1	Mangrove forest and scrub	Wetland	Forest or Herbaceous or Stable skeletal
SA2	Searush, oioi, glasswort, sea primrose rushland/herbfield [Saltmarsh]	Wetland	Herbaceous or Stable skeletal or Mobile skeletal
SA3	Glasswort, sea primrose herbfield [Saltmarsh]	Wetland	Herbaceous or Stable skeletal or Mobile skeletal
SA4	Shore bindweed, knobby clubrush gravelfield/stonefield	Mobile skeletal	
SA5	Herbfield [Coastal turf]	Herbaceous	Stable skeletal
SA6	Kermadec ngaio scrub, mixed herbfield/loamfield	Herbaceous	Stable skeletal
SA7	Ice plant, glasswort herbfield/loamfield	Mobile skeletal	Herbaceous or Stable skeletal
SA8	Mutton bird sedge, Senecio radiolatus sedgeland/herbfield/loamfield	Mobile skeletal	Herbaceous or Stable skeletal or Dangerous
SA9	Olearia, Brachyglottis, Dracophyllum scrub/herbfield/loamfield [Mutton-bird scrub]	Forest	Herbaceous or Stable skeletal or Dangerous
SA10	Poa litorosa grassland/herbfield/loamfield	Herbaceous	Stable skeletal
SA11	Kirk's scurvy grass herbfield/loamfield	Stable skeletal	Herbaceous
UM1	Pōhutukawa, tānekaha forest/scrub/rockland	Forest	
UM2	Conifer, beech, mānuka forest/scrub/rockland	Forest	
UM3	Tussockland/stonefield/rockland	Stable skeletal	Herbaceous
GT1	Geothermal kānuka scrub	Dangerous	Herbaceous or Stable skeletal
GT2	Geothermally-heated water	Dangerous	Herbaceous or Mobile skeletal
CV1	Subterranean rockland, stonefield [Caves]	Dangerous	Mobile skeletal
VS1	Pōhutukawa scrub/forest	Forest	
VS2	Kānuka scrub/forest	Forest	Herbaceous or Stable skeletal
VS3	Mānuka, kānuka scrub	Forest	Herbaceous or Stable skeletal
VS4	Mānuka scrub	Forest	Wetland
VS5	Broadleaved species scrub/forest	Forest	Wetland
VS6	Matagouri, Coprosma propinqua, kōwhai scrub [Grey scrub]	Forest	Herbaceous or Stable skeletal
VS7	Mountain tauhinu, Dracophyllum rosmarinifolium scrub	Forest	Herbaceous or Stable skeletal

Code	Short Name	Most Likely Method	Other Possible Method(s)
VS8	Monoao scrub	Forest	Herbaceous or Stable skeletal
VS9	Inaka scrub	Forest	Herbaceous or Stable skeletal
VS10	Bracken fernland	Herbaceous	Stable skeletal
VS11	Short tussock tussockland	Successional	Herbaceous or Stable Skeletal
VS12	Sward grassland	Successional	Herbaceous or Stable Skeletal
VS13	Red or copper tussock tussockland	Successional	Herbaceous or Stable Skeletal
VS14	Tall tussock tussockland	Herbaceous	Stable skeletal