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Biodiversity monitoring, ecological integrity, and the design of the New Zealand Biodiversity Assessment Framework

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Abstract: The New Zealand Department of Conservation is responsible for biodiversity management over approximately one-third of New Zealand's land area and a network of marine protected areas; it also has a more general role in managing protected species and biodiversity advocacy. In 2004 the Department of Conservation began the development of a national natural heritage monitoring framework known as the New Zealand Biodiversity Assessment Framework, which has been operational since 2011. 'Ecological integrity' is the integrating biodiversity concept underpinning the framework and is widely used in reports, policy and New Zealand legislation. However, the term has been criticised as being vague and difficult to operationalise. Here, we discuss ecological integrity and its application in a New Zealand context, and its relationship with the widely used related concepts of ecosystem health and mauri. An overview of the design principles behind the Biodiversity Assessment Framework is presented and the National Outcome Objectives, which collectively build a picture of the state of ecological integrity, discussed.

Keywords: biodiversity, conservation, ecological integrity, ecosystem health, mauri, monitoring

Introduction

The New Zealand Department of Conservation (DOC) manages public conservation land and waters (PCLW) that make up c. 32% of the land area of New Zealand and include protected marine areas, which make up 0.02–1% of each of the 14 coastal marine bioregions (Department of Conservation & Ministry of Fisheries 2011). DOC is responsible for biodiversity on PCLW but also has a broader requirement to 'advocate the conservation of natural and historic resources generally' (Conservation Act, 1987). DOC is therefore a lead agency for comprehensive, unbiased, up-to-date and accessible biodiversity information. These data are needed to document biodiversity status and trend, to guide, inform and report DOC's activities, to satisfy national oversight agencies such as the Auditor General's Office and Treasury, and to underpin international reporting (for instance to the United Nations Convention on Biological Diversity).

DOC was formed in 1987 from several public agencies and inherited from them a tradition of monitoring. DOC had 2300 active and archived projects in 2005, which formed a patchwork of disparate efforts without common goals or techniques and the data generated were often not analysed or even digitised (Lee et al. 2005). The earlier pre-DOC monitoring projects and inventory surveys focussed on natural resources such as merchantable timber, wild game bird and fish stocks, or perceived threats to these resources. As a result, there was a large national investment in monitoring the effects of introduced ungulates in forests and alpine ecosystems from the 1950s to the early 1980s because of perceived risks to forest resources and erosion (McKelvey 1995). In contrast, the prevailing wisdom in the middle of last century was that introduced predators and their indigenous bird prey were a self-adjusting component of the natural system (Williams 1962), and this did not encourage investment in monitoring them. Biodiversity data that were collected mainly took the form of occasional local surveys and taxonomic surveys carried out by a range of agencies and universities. Many monitoring programmes carried out by precursor agencies were scrapped after DOC was established, and monitoring focused on what were regarded as critical conservation issues.

Recognising the need for more systematic monitoring, in 2004 DOC formed a scientific working group with the brief of designing a biodiversity monitoring framework. This group adopted 'ecological integrity' (EI) as the most appropriate high-level goal for New Zealand conservation in light of our international and national obligations, and EI became the overarching goal of the resulting Biodiversity Assessment Framework (BAF; Lee et al. 2005). In the decade since this framework was initiated, measures and indicators it proposed have been researched (MacLeod et al. 2012; Peltzer et al. 2014), adopted, monitoring has begun, the first reports published (Allen et al. 2013; Bellingham et al. 2013), and results included

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in DOC annual reports from 2011. Ecological integrity has now been adopted, along with 'ecosystem health' (EH) as a central component of New Zealand biodiversity policy (see, for instance, the Environmental Reporting Act, 2015).

A comprehensive DOC Outcome Monitoring Framework (https://www.doc.govt.nz/omf) now sets out in detail how its four Intermediate Outcomes will contribute to the overall departmental outcome statement "New Zealanders gain environmental, social and economic benefits from healthy functioning ecosystems, recreation opportunities and living our history". The BAF forms part of this larger structure as the framework that addresses the Intermediate Outcome Goal 1: The diversity of our natural heritage is maintained and restored.

As with many biodiversity monitoring systems elsewhere, the BAF and its associated Biodiversity Monitoring and Reporting System has been much criticised and dismissed as a waste of conservation funding (see Gerry McSweeney, Sanderson Memorial Address, 2013). Therefore, while the BAF has lasted 15 years, and guided data collection since 2011, its survival depends on constant justification.

Here we discuss:

(1) ecological integrity (EI) as the overarching biodiversity goal for the BAF;

(2) the rationale for a systematic approach to biodiversity monitoring;

(3) design of the BAF, including the underlying principles; and

(4) the overall structure and high-level content of the BAF.

Ecological integrity

Despite EI being widely accepted as central to environmental outcomes, it has not been discussed in any detail in the New Zealand context. Part of that context are two other key concepts, mauri and EH, and we will discuss these first as they are closely linked to EI.

Mauri

Recent discussions of Māori approaches to the sustainable use and conservation of indigenous ecosystems have promoted the concept of 'mauri' (Harmsworth & Awatere 2013; Timoti et al. 2017; Lyver et al. 2018; Michel et al. 2019). Mauri, as defined by Harmsworth and Awatere is "…an internal energy or life force derived from whakapapa [the interconnectedness between all elements of the living and non-living realms], an essential essence or element sustaining all forms of life. Mauri provides life and energy to all living things, and is the binding force that links the physical to the spiritual worlds (e.g. wairua). It denotes a health and spirit, which permeates through all living and non-living things. All plants, animals, water and soil possess mauri. Damage or contamination to the environment is therefore damage to or loss of mauri". Timoti et al. (2017) add the notion that mauri "describes the representativeness and condition of the relationships and responsibilities between elements of whakapapa". Mauri encompasses elements of both EH and EI but, as wairua and whakapapa are essential parts of the concept (Morgan 2006), it extends far beyond either of them. As we discuss below, our concepts of EH and EI are based solely on ecological and sociological measures and adoption of mauri as an overall goal for BAF would be inappropriate. However, mauri is vitally important as a guiding principle in its implementation.

Ecosystem health

Ecosystem health is a high-level environmental goal and was originally formulated as: "a biological system… can be considered healthy when its inherent potential is realised, its condition is stable, its capacity for self-repair when perturbed is preserved, and minimal external support for management is needed" (Karr, 1991). Ecosystem health thus describes the fundamental physical and biological state of an ecosystem in relation to its ability to support services. It extends from individual organisms through landscape level processes to economic and societal impacts (Rapport et al. 1998) and incorporates aspects related to mauri. An ecosystem in good health is functionally appropriate for a given environment, generates biomass, exchanges gases, recycles nutrients, protects the land and water from erosion and pollutants and, unless managed for production, is self-sustaining. It is resilient to external threats, supports adequate functional diversity and all expected trophic levels are present and well interconnected (Tett et al. 2013). It provides an ecological norm by which sustainability of economic goals held by groups with differing cultural values and attitudes can be assessed (Callicott & Mumford 1997).

A healthy ecosystem may potentially take different forms in identical environments: for instance, in a temperate, fertile, well-watered region it could be productive exotic grassland, a production forestry plantation or an unmanaged indigenous forest. Even in an urban area, if the biological components are thriving, and pollutants minimised, the term 'healthy' still seems appropriate. Much of the New Zealand landscape, urban, rural and wild alike, is, by this definition, ecologically healthy: plants, fish, mammals, birds and invertebrates are abundant in self-sustaining communities; ecosystem services are maintained; citizens extract wealth and enjoyment.

Ecosystem health has been criticised as being too vague to operationalise (Larkin 1996) and of rhetorical value only (Jamieson 1995; Simberloff 1998). On the contrary, we argue that EH represents an important, quantifiable, underpinning goal in ecosystem management.

Ecological integrity

The EI concept was created to meet a specific legislative need with regard to biodiversity. The Water Pollution Control Act (1972) in the United States, and its requirement to 'restore and maintain' biotic integrity, spurred development of an operational definition of the concept (Karr 1991). Karr & Dudley (1981) defined biological integrity as: the ability to support and maintain "a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organisation comparable to that of the natural habitat of the region". Subsequent elaborations argued for more sociological factors such as intactness, wildness, and beauty (Andreasen et al. 2001). Although the original EI concept thus centred on the concept of 'naturalness', Woodley (2010), discussing the concept from a Canadian viewpoint, stated that: "..ecological integrity can and should be understood outside the context of whether or not humans are present in the system" and stated that EI is, in fact, used by Parks Canada as a replacement for the idea of 'natural' because of the long history of human occupancy. In this sense it overlaps with mauri, in which humanity is seen as an integral part of the system. Recognition of EI thus goes a long way to fulfilling the Māori imperative of incorporating the Treaty of Waitangi into conservation decisions. Ecological integrity therefore functions as a broader, more inclusive definition of an ideal ecological state than ‘natural’. The definition of EI in the New Zealand Environmental Reporting Act 2015, which establishes the mandate for regular national environmental reporting, is: “..the full potential of indigenous biotic and abiotic features and natural processes, functioning in sustainable communities, habitats, and landscapes”, which was adopted from Lee et al. (2005), who went on to say that: “…at larger scales, ecological integrity is achieved when ecosystems occupy their full environmental range.”

The critical distinction between EH and EI is that an ecosystem may have high EH (that is, be functioning well) but low EI (for instance through lacking representation or dominance of indigenous elements) but the reverse cannot be true. As with EH, EI has been criticised on the grounds that it is too vague and rhetorical to be operationalised (Larkin 1996; Manuel-Navarrete et al. 2004). We disagree, as we show later. The EI concept is now in widespread use, at a range of scales ranging from selected ecosystems, such as forests, where provision of economic services is also an important factor (Ghazoul et al. 2015), to national parks e.g. Canada National Parks Act (2000) where preservation of biodiversity is the over-riding aim, and as a high-level goal for a whole nation as in the New Zealand Environmental Reporting Act (2015).

Ecological integrity, as we employ it, describes a generalised ecological state but not any particular past state. It is an ideal constructed from a range of information sources including historical data, present occurrences, species and community models, climatic and soil data sets etc. The underpinning concept is that nothing important is missing, and ecosystem function is unimpaired. Over centuries to millennia, biodiversity is always in flux (Sprugel 1991), and biodiversity loss due to human activity in New Zealand has been underway since the 13th century. Where this loss is permanent and occurred before or during the earliest stages of European settlement – as is the case with moa and the huia, for instance it makes little practical sense to incorporate such loss in current depictions of EI. Historic baselines play an important but subsidiary role in countering a reduced sense of possibilities—for instance by establishing the former presence of a species or community in areas from which they have been extirpated—and thus providing an historic EI template (McGlone 2000; Wood et al. 2017). While the historic template is not neglected, EI is generally assessed by comparing the integrity of a site to extant, less modified or unmodified sites, whether for general guidance or a specific benchmark. With time, and continuing alteration of the New Zealand environment through human activity, exotic invaders and climate change, assessment of EI will increasingly rely on our understanding of species-level biology and community dynamics.

Ecological integrity is scale-sensitive. At small spatial scales (signifying extent, as in reefs, dunes, peat bogs, ponds, reaches of rivers or forest fragments) it is improbable that every species that could be there, is there, as chance plays an important role. All that is necessary for good EI at small scales is that the indigenous biota typical of a region dominates sustainable, healthy ecosystems. If trophic representation, structural elements, absence of exotic dominants, and physico-chemical indicators score well, EI is maintained. However, the larger the spatial scale, the more important absences become. Ecological integrity at a regional level must be regarded as impaired if species that should be present are sparse or totally absent—an example is the almost complete extirpation of birds such as parakeets and mohua from South Island forests—or if once widespread ecosystems are dramatically reduced. A regional approach is also appropriate for the assessment of naturally rare ecosystems, such as thermal springs, or sand dunes (Holdaway et al. 2012). Global extinction of species or the complete loss of indigenous ecosystems from distinctive land environments need to be assessed at a national scale. In contrast, EH is not scale-sensitive: ecosystems can suffer poor ecological health at all scales and are assessed by the same criteria. Disturbed areas subject to nutrient enrichment, drainage, soil contamination, or soil loss may never recover their original EI status or take a long time to do so.

Over much of the wild New Zealand landscape, introduced plants, birds, and mammals are dominant. Mammalian browsing can stall tree regeneration (Richardson et al. 2014) or exotic predators can prevent re-establishment of native birds. Their influence is often compounded by lack of indigenous soil mycoflora (Williams et al. 2012) and distance from seed sources or communities of native animals. A widespread opinion among conservationists is that all exotic organisms are unwelcome in wild ecosystems and, with few exceptions, should be actively eliminated in conservation areas and restored ecosystems. Nevertheless, the idea that blended (that is quasi-permanent systems with a substantial presence of exotics as well as indigenous organisms) have a certain amount of legitimacy has been promoted in recent years (Hobbs et al. 2009). ‘Novel’ and ‘transformed’ are two well used terms for them although others are sometimes encountered (e.g. ‘recombinant’, ‘trashed’). Novel ecosystems have long been accepted in New Zealand. Introduced salmonids are protected species in some National Parks and encouraged in freshwater systems, and most exotic birds are tolerated in otherwise largely unmodified ecosystems. Recent legislation has expanded this de facto tolerance. The Game Animal Council Act (2013) has created a statutory body whose major role is to enhance the quality of game animal herds (deer, chamois, tahr, and wild pigs) which hitherto were solely regarded as pests in PCLW. As we see it, novel ecosystems have high EH but degraded EI.

Advocates of the novel ecosystem concept argue that for many wild ecosystems the tipping point has been passed and a new, stable situation has arisen they term the ‘new ecological world order’ (Hobbs et al. 2013). The counter argument is that the so-called tipping points and stable states are simply the result of accepting the current situation and there are few, if any ‘novel’ ecosystems that cannot be restored or at least set back on a trajectory towards EI (van Andel 2013; Murcia et al. 2014). A New Zealand example is the degraded short-tussock grasslands of the south-eastern South Island where invasion of mixed exotic–native woody species is occurring but there is reason to hope that, if managers work with successional trends, native-dominated, self-perpetuating plant communities will arise that are “as tall and biologically complex as a site can support” (Walker et al. 2009). This succession may take various unconventional pathways and have endpoints different from those of undisturbed systems (Sullivan et al. 2007), but may nevertheless have high EI status.

Ecological integrity in relationship to climate change will have to be addressed in the near future (Halloy & Mark 2003; McGlone & Walker 2011). To some extent, we are already familiar with the issue as indigenous plants and animals naturally confined to the north of New Zealand are spreading in the south due to human translocation. A well-studied example is the tree *Olearia lyallii* that has spread in subantarctic Auckland Island after having been brought south from the Snares Island in the early 19th century (Wilmshurst et al. 2015). Around Wellington and in the upper South Island, the strictly northern trees karaka (*Corynocarpus laevigatus*), pōhutukawa (*Metrosideros excelsa*), and karo (*Pittosporum crassifolium*) are all aggressively spreading in local coastal forests (Simpson 1997). Costall et al. (2006) recommended control of karaka in the southern North Island, outside its natural range, on the grounds that it reduces understory diversity. McGlone and Walker (2011) argued that unless range changes, unaided or anthropogenic, seem likely to degrade the local biodiversity (i.e. EI) of a site, they should be ignored. Wilmshurst et al. (2015), on this basis, argued that EI was not significantly threatened by the native 'alien' *Olearia lyallii*. The same argument could be used to support relocation of species to follow their climatic envelope when threatened by changing climates in their natural habitat, as has been suggested for hihi (*Notiomystis cincta*) (Chauvenet et al. 2013).

Systematic biodiversity monitoring

Definition of systematic biodiversity monitoring

We define systematic biodiversity monitoring as monitoring underpinned by statistically valid selection of monitoring plots, sites or locations; establishment of strict data collection protocols; and characterised by repeat measures. To be deployed at a national level, agreement has to be reached on common standards and coverage between a number of organisations and costs equitably shared. Because of the inherent difficulties in meeting these requirements, many other forms of monitoring are used or advocated in which some or all these requirements are weakened or abandoned. For instance, non-systematic collection of data by citizens (Citizen Science) is increasingly suggested as a cost-efficient alternative to professional efforts (e.g. Fairclough et al. 2015). Informal monitoring is of value (Tulloch et al. 2013), but there are clear limits to what can be achieved (McKinley et al. 2017).

Due to the cost and complexity of systematic biodiversity monitoring, much national reporting relies on independent studies and monitoring programmes of short duration or limited scope. Statistical techniques are used to amalgamate disparate data sets, collected at a range of scales and under different protocols to produce estimates of trends over time. While use of disparate data sets has its place, it has also resulted in "..a plethora of measures and indicators (based on existing data) that frequently fall short of their intended purpose" (Reyers et al. 2013). Moreover, statistical approaches can compensate for poorly structured data only to a limited extent and if whole groups of organisms are neglected, little useful can be said about them. Even reports of population trends in well studied organisms such as birds have relied on "…post hoc, serendipitous analyses…" (Donald et al. 2007).

Such data-mining approaches are usually responses to the need for monitoring of any sort to satisfy legislative or agency protocols or 'box ticking' in the words of the Parliamentary Commissioner for the Environment (Wright 2014). In a scathing commentary in 2019 on the current New Zealand environmental reporting system, the Parliamentary Commissioner for the Environment stated that "…there is no plan or commitment to gather new data. The system was designed to make do with whatever information happens to be available. This 'passive harvest' approach is inadequate to inform the stewardship of our environment." (see https://www.pce.parliament.nz/media/196936/focusing-aotearoa-new-zealands-environmental-reporting-system-faqs.pdf)

While a systematic approach to biodiversity monitoring thus seems highly desirable, the larger question remains as to what aspects of biodiversity should be included in its ambit. The Department of Conservation, charged with the responsibility for managing biodiversity across an entire nation, has a major challenge in developing a manageable biodiversity assessment programme. Public conservation land and waters harbour many thousands of indigenous species, hundreds of which are under extreme threat, and they form a large array of ecosystems, many of which themselves are at risk. As well, the presence of many exotic species in PCLW poses a complex set of problems. Some are invasive and pose a clear threat to indigenous species and ecosystems (e.g. cats, rats, stoats, possums) while others are tolerated (most birds), or managed for recreation, food provision, or commercial reasons (deer, thar, pigs, trout, salmon). Funding for monitoring and assessment will always be insufficient to address this complexity and thus careful choices, underpinned by a defensible rationale, must be made where to invest (Possingham et al. 2012).

Rationale for systematic monitoring

Some guidance to the scope of a national monitoring scheme is given in Article 197 of the official statement promulgated during the Rio+20 Conference in 2012, which affirms the intrinsic value of biological diversity including "the ecological, genetic, social, economic, scientific, educational, cultural, recreational and aesthetic values of biological diversity and its critical role in maintaining ecosystems that provide essential services". This wide brief is mirrored in the overall aims of the New Zealand Biodiversity Strategy (Goal Three: Halt the decline in New Zealand's indigenous biodiversity): "Maintain and restore a full range of remaining natural habitats and ecosystems to a healthy functioning state, enhance critically scarce habitats, and sustain the more modified ecosystems in production and urban environments; and do what else is necessary to maintain and restore viable populations of all indigenous species and subspecies across their natural range and maintain their genetic diversity", a goal confirmed by the adoption of EI as a goal in the Environmental Reporting Act 2015. The need to operationalise these broad, intentionally inclusive statements to a set of concrete biodiversity aspirations and goals has led to the twenty Aichi targets of the Convention of Biological Diversity, which state desired outcomes by 2020 (see http://www.cbe.int/sp/targets).

Threats to biodiversity or drivers of biodiversity change are now the organising principle for nearly all biodiversity reporting frameworks as, for instance, in the widely used Driver–Pressures–State–Impacts–Responses (DPSIR) framework (Svarstad et al. 2008; McGeoch et al. 2010), a version of which has been adopted by the New Zealand Ministry for the Environment (MfE) and endorsed by the Parliamentary Commissioner for the Environment (2019). Information collected under such frameworks is often presented as an environmental dashboard in which various high-level indicators are portrayed as either improving or declining, much as the dashboard of a vehicle will display its speed, fuel consumption, and distance travelled, etc. (Han et al. 2014). Such dashboards attempt to make broad policy sense of the dispersed, fragmented biodiversity information available. While relatively cheap to implement (leaving aside the fact that the cost of basic data collection and archiving is high and borne by the contributing organisations), such desktop exercises have the drawback of relying on an assortment of disconnected data series.

A broader approach is that of the Group on Earth Observations Biodiversity Observation Network (GEO BON, https//www.earthobservations.org/geobon.shtml) who have developed an initial list of Essential Biodiversity Variables (EBVs) (Pereira & Cooper 2006; Pereira et al. 2013). These are a suggested list of key variables for detecting major dimensions of biodiversity change and are intended to facilitate data integration by providing an intermediate layer bridging the gap between primary observations, indicators and assessments and the overall purpose of the system (Geijzendorffer et al. 2016).

While the high-level Aichi targets supported by DPSIR and EBV data integration approaches have a role to play, in particular for communication of monitoring results, as the organising principle for a data acquisition system they are inadequate. Their concentration on drivers such as exotic predators, contaminants, anthropogenic activities etc, and the status of rare and endangered taxa and ecosystems, in practice means neglect of common and abundant organisms and ecosystems. These latter are vital to the provision of all manner of ecosystem services (Gaston 2008, 2010). Therefore, unless underpinned by a comprehensive scheme for collection and analysis of biodiversity data, most schemes will default to an exclusive focus on immediate threats.

The conclusion reached by Lee et al. (2005) was that there was a need for a systematic monitoring and reporting system, supported by an outcome assessment framework, that would provide comprehensive data on biodiversity and its relation to environmental factors that would underpin a range of different needs from policy through to local conservation actions. The BAF is broadly structured around high-level outcomes that are based on the actions needed if high levels of EI are to be maintained but include provision for collection of a wide range of data essential for understanding ecosystem function, provision of services and establishment of baselines. While this would seem to be a non-controversial aim, it has proved not to be.

Opposition to systematic monitoring

Despite the clear need for systematic monitoring, and the dangers of relying on ad hoc data collection, formal, centralised monitoring systems are often poorly supported or actively opposed, and thus short-lived (Watson & Novelly 2004). They divert, or are assumed to divert, funding from other important activities or personal projects (Possingham et al. 2012). Many academic scientists are indifferent or opposed to long-term biodiversity monitoring programmes, characterising them as inefficient or as simply collection of data for data's sake with no explicit rationale (Nichols & Williams 2006; Nichols 2010; Wintle et al. 2010). They argue that such programmes lack specific management or scientific questions, are not optimised to a particular purpose, and lack a priori hypotheses (Lindenmayer & Likens 2010). However, some of the most useful and skilfully deployed environmental monitoring is status and trend or base-line monitoring. Examples include atmosphere and ocean monitoring of chemical composition, radiation, temperature, salinity and pressure (e.g. the Mauna Loa Observatory in Hawai'i). Base-line monitoring is just as important for biodiversity.

Opposition to monitoring also arises because existing monitoring data is often neglected in development of policy or management and most conservation decisions are not guided by formal scientific evidence at all (Pullin et al. 2004; Sutherland et al. 2004). The Department of Conservation has not been free of this tendency: an Office of the Auditor-General review concluded that "…decisions were made without confirming the condition of biodiversity values that would be affected." (Office of the Auditor-General 2012). Expenditure on monitoring will be often resisted by managers as they rarely use data derived from it.

Due to this deep-seated resistance to monitoring, monitoring schemes themselves are often ephemeral creations arising out of theoretical explorations by academic researchers or bureaucratic responses to a legislative imperative but have weak organisational commitment and attract few resources to sustain them. A report commissioned by MfE into state or national monitoring frameworks (Gurnsey 2012) included no framework created before 2005, reflecting their rapid obsolescence. For instance, the LUCID programme of United States Department of Agriculture Forest Service (Wright & Colby 2002) which was designed to guide comprehensive forest environmental monitoring, and which provided the inspiration and high-level structure for the BAF, was never implemented. The BAF development team was aware of this problem and a great deal of effort was put into design, consultation, testing and implementation of the framework.

Design of the Biodiversity Assessment Framework

"Designing a monitoring project is like getting a tattoo: you want to get it right the first time because making major changes later can be messy and painful." (Oakley et al. 2003).

Development of the Biodiversity Assessment Framework

Given the rather gloomy prognosis for monitoring schemes in general (Watson & Novelly 2004), the DOC development team took care to develop a carefully argued and principled case (Lee et al. 2005). The first issue was what was the overall goal of systematic monitoring, and EI, as discussed, was selected as the most appropriate high-level formulation of this goal.

The second issue was scope, and it was here that the team had its most vigorous debates. Some argued for a tight focus on threatened species and ecosystems, with monitoring closely connected to conservation activity, and others for a broader concept of relevance supported by a national-level permanent monitoring network. It was decided that the BAF would include a range of monitoring approaches, supported in the first instance by a pre-existing national-level permanent plot system, but with the aim of extending to more local and focused initiatives. However, everything that current understanding suggests could be important to the achievement of EI, and about which it would be useful to have monitoring data, is included in the framework even though it is unlikely that all these aspects of biodiversity will actually be monitored. The rationale for a broader scope than is currently feasible with existing budgets is that, while there may be insufficient resources to monitor all important aspects of biodiversity, we should be aware of what is missing, and alert to possibilities of including them should resources or new techniques become available, or the need apparent.

The final issue regarding the framework per se was how to organise the various components. The LUCID programme of United States Department of Agriculture Forest Service, which outlined how ecological and socio-economic indicators for forests and grasslands could be incorporated in a nation-wide system and which had extensive documentation available, was taken as a template. LUCID, in turn, was based on the Criteria and Indicator Framework that underpins international agreements such as the Montreal Process (international monitoring of forest health). In the next section we discuss how this framework informed BAF.

Structure of the Biodiversity Assessment Framework

The BAF is a hierarchical scheme in which higher-level, more general statements are supported by more specific lower levels which are underpinned in turn by concrete data (Tables 1, 2).

A hierarchical structure ensures that the system is robust, transparent, and flexible. The system is robust because a concept at a given level must be supported by a range of components in the underlying levels, which in turn are supported by a discrete data set. Transparency is ensured because it is clear how an upper-level concept will be quantified by lower-level measures and elements, and likewise how an individual data element contributes upwards to the larger picture. The system is flexible because a large number of components can be inactive or new ones slotted in with minimal disruption to the framework as a whole. Should a new threat arise or an existing situation demand more monitoring, the most suitable thread in the BAF will be located and either existing levels within it activated, or new ones created.

The original Lee et al. (2005) framework did not include marine areas nor did it specifically separate out freshwater concerns. After further investigation of the applicability of EI to marine (Thrush et al. 2011) and freshwater systems (Schallenberg et al. 2011), it was decided that they could be accommodated in an expanded framework using the domain terminology of the New Zealand Environmental Reporting Act (2015). Domains are recognised at the measure level by the suffixes L (Land), F (Freshwater) and M (Marine), which show the scope of that particular measure.

The domains differ in fundamental ways that greatly affect how they are assessed. Temporal turnover of biotic communities in the marine domain for instance is often rapid. Substrate stability is generally high in the land domain but low in many freshwater systems. In New Zealand, exotic invaders are not as critical an issue in the marine domain that they are

Table 1. Description and rationale for the hierarchical levels in the New Zealand national Biodiversity Assessment Framework.

Hierarchical level	Description	Rationale
DOC(National) Outcome	Departmental goal	Statement of the organisation's fundamental purpose and obligations
Intermediate Outcome	Critical components of the organisational goal	*Aspirational level*: a statement of the desired outcome
Outcome objective	Key factors contributing to intermediate outcomes	*Imperative level*: it encapsulates an essential feature of biodiversity needed to ensure EI is being maintained and states directly what must happen for aspirations to be achieved Addresses the question of why are we concerned about this issue.
Indicator	Quantitative or qualitative aspects that should be assessed in relation to an objective	*Definable*: whereas the levels above are general, this is where decisions begin to be made as to what will be measured. Addresses the question of what sort of information is needed.
Measure	Concrete factors with their methodology and source of information for assessing indicator performance	*Quantifiable*: Measures can be complex entities, but each is directly underpinned by systematically collected data. Choices made here reflect the degree to which managers and scientists believe some aspects of natural heritage are more important than others. Addresses the question of how we are going to obtain relevant information.
Element	Discrete underpinning data or data layers. Sometimes Measures effectively are the element.	*Data layer*: the bedrock of the system. These data layers may be used to support more than one measure.

Table 2: Specific lines of influence and support within the framework we term 'threads'. An example of a single thread (Reducing spread and dominance of exotic species) in the New Zealand national Biodiversity Assessment Framework.

Hierarchical level	Description
DOC(National) Outcome	New Zealanders gain environmental, social and economic benefits from healthy functioning ecosystems from recreational opportunities, and from living our history
Intermediate Outcome 01	The diversity of our natural heritage is maintained and restored
Outcome objective 1.3	Reducing spread and dominance of exotic species
Indicator 1.3.1	Exotic species occurrence
Measure 1.3.1.1-LFM (L = Land domain; F = Freshwater domain; M = Marine domain)	Occurrence of self-maintaining populations of exotic species
Element	Lists by orders of all self-maintaining exotic species in the country Mapped distribution of species regarded as potential or actual risks

in land and freshwater domains but marine noise pollution is far more serious than the intermittent and localised issue it is on land. The nutrient enrichment problem that affects many freshwater systems is less severe in most terrestrial ecosystems.

A result of this incorporation of marine and freshwater concerns was that the original high-level components (indigenous dominance, species occupancy and environmental representation) of the Lee et al. (2005) framework had to be abandoned, although these concepts inform lower levels. For instance, water quality is an immensely important factor in freshwater systems but it was unclear how it could be incorporated within the ambit of the original three components. Even in the original formulation, the outcome objectives had to be shoehorned into these categories and expansion of the framework made them unworkable as high-level components.

A number of concepts such as 'sustainability' and 'resilience', and 'ecosystem services' are prominent as high-level goals in national and international declarations (e.g. Campbell et al. 2012). The BAF will provide data that can be used or re-purposed to address these and many other issues, but is not structured around them.

If monitoring is focussed only on the results of conservation activity, it loses context and becomes vulnerable to capture by managers anxious to show the effectiveness of their interventions (Newton 2011). Monitoring data is needed just as urgently to assess the overall state of biodiversity. To take a parallel example from the health system, it is as though only the success rate of surgical interventions were used to assess the health of the nation. Monitoring must therefore have a broader ambit than simply documenting conservation successes.

The BAF is not a reporting framework but delivers information that can be incorporated in reports. Moreover, it needs support from dedicated monitoring programmes such as the Biodiversity Monitoring and Reporting System programmes (for instance Tier 1 national monitoring) that are already delivering information used in many different applications as well as reporting.

National Outcome Objectives in the Biodiversity Assessment Framework

There are eight National Outcome Objectives in the BAF. Collectively, they are intended to give a comprehensive overview of the state of EI in New Zealand. We discuss them here, but further details of indicators and measures for the entire DOC Outcome Monitoring Framework, of which the BAF forms part, are given in McGlone & Dalley (2015).

Maintaining ecosystem processes

Description

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.

Commentary

This broad objective deals mainly with EH and includes a number of indicators and measures dealing with the status of ecosystems and the influence of abiotic factors that tend to degrade them (such as sedimentation, fragmentation of habitat, fire, etc). Climate change is dealt with in a separate outcome objective. The identity of the species involved is not the issue here, but whether ecosystems as a whole are functioning well in the sense of processing nutrients and energy, providing quality habitat and ecosystem services. A wide range of states and processes are potentially included. Some can be captured by remote sensing and are already monitored (e.g. indigenous vegetation cover, net ecosystem primary productivity, fish catches) others are the focus of existing monitoring networks (e.g. water quality and quantity, seeding of key indigenous trees, fire) but others are not yet systematically monitored (e.g. substrate alteration, disease outbreaks). This outcome objective is a key component for aquatic ecosystems.

Limiting environmental contaminants

Description

Presence and concentration in the environment of non-nutrient contaminants including faecal bacteria from mammalian sources, vertebrate toxins, pesticide residues and heavy metals, hormones or hormone mimics as a result of human activities. Persistent litter and disruptive noise in the aquatic environment.

Commentary

Incorporation of chemical residues and heavy metals in food chains may be toxic for some life forms and create developmental problems for some species; they can affect the suitability of game for human consumption. The conceptual basis for interpreting both nutrient and toxin content of soils and sediments is well understood, and highly sensitive measurement techniques are available. Non-nutrient contaminants may severely disrupt species and communities, have long-term impacts and persist in the environment for decades or more; freshwater systems are particularly at risk (Fleeger et al. 2003).

Two different contaminants are included. Litter, especially plastic litter in the marine environment endangers aquatic life. Noise in the marine environment has become a matter of great concern because sound carries well in water and the underwater marine soundscapes of vital importance to many species, including cetaceans, many fish, and reef crustaceans (Putland et al. 2017). Anthropogenic noise masks marine biological signals, can pose a potential threat to marine assemblages, affecting behaviour and communication of a wide array of marine animals (Pieretti et al. 2017).

Reducing spread and dominance of exotic species

Description

Documentation of the presence, dominance and rate of increase of exotic species in the natural environment.

Commentary

Few New Zealand ecosystems are free of exotic organisms, but these are not regarded as degrading indigenous dominance per se unless they are abundant or play a critically negative role. Invasive pests or weeds are here defined as exotic species with the potential to become widespread or abundant and have disproportionate effects on native species and ecosystems. Invasive pests and weeds have permanently altered New Zealand's EI through elimination of a suite of birds, reptiles, fish, invertebrates and possibly a few plant species. Along with vegetation clearance, waterway pollution and, in the near future, climate change, they compromise the major threats to EI.

Recruitment of new pests and weeds from recent arrivals and long-established populations is an ever-present threat and taking steps to prevent further enrichment of the biota with exotics a sensible precautionary activity (Tomiolo et al. 2016).

A broad surveillance of all exotic species in the wild thus appears prudent. The promotion of 'predator-free New Zealand' as an aspirational goal for New Zealand by 2050 (see https://www.beehive.govt.nz/release/new-zealand-be-predator-free-2050) has significantly changed the political landscape. While many pest species are not part of the 2050 goal (deer, goats, thar, chamois, pigs, hedgehogs, cats, dogs, mice, trout, salmon, wasps, ants), it will still require a substantial ramp up in New Zealand's capacity to track populations, assess reinvasion probabilities and confirm clearance (Goldson et al. 2015; Peltzer et al. 2019).

Preventing declines and extinctions

Description

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.

Commentary

The conservation status and security of indigenous taxa is one of the few systematically, comprehensively and regularly assessed factors of biodiversity both in New Zealand and globally. This attention is warranted because of the finality of extinction, and the high prominence accorded to species threat status in public debates over biodiversity issues. Threat Status classifications are not without their problems and the methodology suffers from its reliance on expert opinion and a pervasive lack of quantitative information (Breen & Middleton 2013). This outcome objective will remedy this lack. Critically threatened taxa invariably have small or small and sparse populations. Intensive management often involves breeding or nursery programmes and translocations. Inbreeding depression, loss of genetic diversity and mutation accumulation in these reduced populations can theoretically lead to an extinction vortex in which chance events have a devastating effect on weakened populations (Lacy & Lindenmayer 1995). While genetic factors have tended to be underutilised in New Zealand, there is a compelling case for routinely analysing them in all intensively managed taxa (Jamieson et al. 2006).

Maintaining ecosystem composition

Description

Demography of functional groups, their representation, abundance of common and widespread taxa and changes in species diversity.

Commentary

Ecosystems may be put at risk through loss of functional groups (such as bird fruit dispersers) or loss of key species - such as occurred in North America with the loss of American chestnut (*Castanea dentata*) and other trees (Evans & Finkral 2010), or the Southern Ocean with over-harvesting of whales. It is therefore important that secure background information is available on the most common and abundant taxa. For example, 50% of the basal area of New Zealand forests is made up by just five tree species (*Weinmannia racemosa*, *Metrosideros umbellata*, *Lophozonia menziesii*, *Fuscospora fusca* and *F. cliffortioides*) (unpublished inventory data, National Vegetation Survey databank; https://nvs.landcareresearch.co.nz/). Moreover, that a given species is abundant now is no guarantee that that it will not come under threat, as has happened repeatedly in the past. Red-fronted parakeets (*Cyanoramphus novae-zelandiae*) were once numerous enough to be crop pests (Oliver 1955) and the once dominant and still common iconic kauri (*Agathis australis*) is now under threat from Phytophthora infection (Lewis et al. 2019).

Ensuring ecosystem representation

Description

The extent, protection status and ecological condition of indigenous ecosystems.

Commentary

In biodiversity strategies, ecosystems are treated as worthy of preservation in their own right and are of particular importance because, if in good condition, they may ensure the survival of many taxa which cannot be directly managed or of which we may not be aware. Some ecosystems are clearly defined (e.g. lakes) while others transition gradually into other ecosystems and boundaries may be difficult to define (e.g. sand dune systems). After years of subjective, expert opinion-based classifications of ecosystems which have proven difficult to implement, systematic quantitative approaches are now available (Snelder et al. 2007; Williams et al. 2007; Leathwick et al. 2010; Wiser et al. 2011; Holdaway et al. 2012) and these will form the basis for assessment.

Adapting to climate change

Description

Documentation of changing climates, and the biological responses.

Commentary

Climate change is a defining environmental issue of the 21st century (Oreskes 2004). New Zealand is in the somewhat anomalous position, relative to many regions, of having had little discernible environmental effect from climate change on land so far, despite a warming of nearly 1°C since the turn of the 20th century with a marked downward trend in frost frequency (McGlone & Walker 2011). Effects on marine biodiversity in the oceans surrounding New Zealand as a result of recent climate change are challenging to detect and assess in time and space (Willis et al. 2007). Nevertheless, highly significant negative effects in marine, terrestrial and freshwater systems are predicted.

By the end of this century, mean annual temperature will be significantly higher, rainfall patterns will have shifted markedly, and seasonality will be greatly altered (McGlone & Walker 2011). For this reason, a system for monitoring climate-related effects is needed as a watching brief to inform potential mitigation. Waterways and indigenous vegetation patches are likely to suffer in the drier east as this region is set to become even drier in coming years; alpine and subalpine environments will be affected by warmer temperatures and ultimately endure more pressure from mammalian predators and loose snow patch and bare ground habitats (Bannister et al. 2005); coastal wetlands and dune fields will be impacted by rising seas and increased coastal development; and marine areas by warmer currents and ocean acidity (Christie 2014).

Human use and interaction with natural heritage

Description

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 wellbeing.

Commentary
New Zealand's population and overseas tourists interact with the natural world in many ways producing both negative and positive outcomes for ecosystems and people alike. Recreation and some aspects of marine and freshwater harvesting have been studied but the wider benefits and negative effects of experiencing and interacting with the natural world remain largely undocumented but are critical to public support for conservation (Blaschke 2013). Among benefits, a review of recreational hunting (Woods & Kerr 2010) showed that both in New Zealand and elsewhere, experiencing nature and social interaction with family and friends were two of the top motivations and equally important as the actual hunting outcomes. Among negative effects, large or unregulated numbers of tourists can reduce local populations of dolphins (Lusseau et al. 2006). Without such insights, motivations that promote public support for EI could remain misinterpreted by policy analysts and managers.

Issues pertaining specifically to Māori, in particular those connected with tino rangatiratanga and with giving effect to the Tiriti O Waitangi, are comprehensively covered elsewhere in the broader Outcome Monitoring Framework (McGlone & Dalley 2015) in particular in Outcome 4, New Zealanders connect and contribute to conservation.

Final comments

Although there are other ways that a national biodiversity monitoring framework could be constructed, we have opted for a comprehensive approach. The monitoring programmes arising out of the BAF will deliver data of immediate relevance and use, and also will provide fundamental information on how biological systems function. Options will not be foreclosed through a sole focus on current perceptions of threats. The essential features of the BAF are its transparent hierarchical structure, comprehensiveness and robustness. The hierarchical structure ensures that broad, higher level biodiversity issues are articulated and supported by lower levels that address them. It is thus possible to discover easily what data address a particular concern and, as importantly, why it is necessary to collect a certain type of data. Comprehensiveness ensures that elements that may be regarded as being of lesser importance or too costly to implement immediately are retained in the scheme. While cost and perceived importance will play a large role in determining what elements of the BAF are activated and when, the framework will ensure that absences will be noted. Robustness comes from the modular structure: elements can be inserted or deleted without requiring anything but modest rearrangements. Stability is of paramount important in a biodiversity framework and we believe the BAF has already demonstrated this.

With regard to EI as the overarching goal, it addresses the concerns that have been expressed from time to time about biodiversity conservation being unduly focussed on the unattainable ideal of restoring a long-gone past. Ecological integrity reflects an ideal state and one that is of course informed by history, but also by EH, the potential of the constituent species in the biota and the reality of current ecosystem dynamics. Our concept of EI will develop over time but it will form a readily understood and quantified goal.

Finally, regardless of the approach, biodiversity monitoring needs systematic, principled development rather than being assembled solely out of pre-existing data sets with convenience and cost the dominant considerations. While short-term savings can undoubtedly be made through such repurposing, in the long term the extraordinary statistical power of a well-designed monitoring system will pay for itself many times over through better quality data and greater flexibility.

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