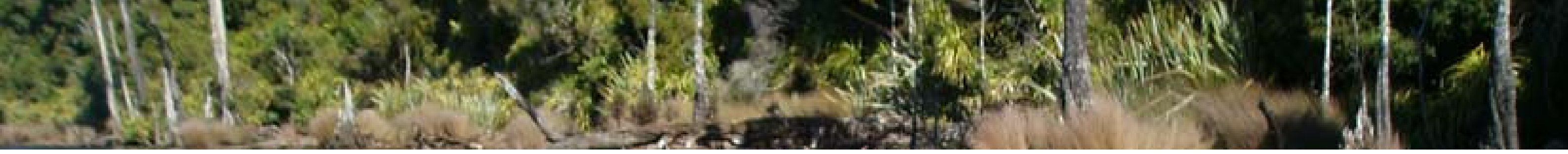

Approaches to assessing ecological integrity of New Zealand freshwaters

SCIENCE FOR CONSERVATION 307

Department of Conservation
Te Papa Atawhai

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Published by
Publishing Team
Department of Conservation
PO Box 10420, The Terrace
Wellington 6143, New Zealand

Cover: Dr Jean-Jacques Frenette assessing the optical properties and water chemistry of Lake Mahinapua (West Coast). Water clarity influences many aspects of the community structure (e.g. aquatic plants) and functional processes (e.g. thermal stratification) of lakes, and can be affected by a range of both natural and human-related factors, such as algal blooms from dissolved nutrient runoff. *Photo: D. Kelly.*

Science for Conservation is a scientific monograph series presenting research funded by New Zealand Department of Conservation (DOC). Manuscripts are internally and externally peer-reviewed; resulting publications are considered part of the formal international scientific literature.
This report is available from the departmental website in pdf form. Titles are listed in our catalogue on the website, refer www.doc.govt.nz under *Publications*, then *Science & technical*.

ISSN 1177-9241 (web PDF)
ISBN 978-0-478-14870-1 (web PDF)

This report was prepared for publication by the Publishing Team; editing and layout by Amanda Todd. Publication was approved by the General Manager, Research and Development Group, Department of Conservation, Wellington, New Zealand.

In the interest of forest conservation, we support paperless electronic publishing.

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ABSTRACT

This report critically assesses the concept of ecological integrity (EI) and its application in freshwater management. EI integrates a wide range of ecological values related to the structural and functional processes of ecosystems, making it particularly appealing for management application. EI is not inherently empirical, but it can be useful where its components are rigorously defined and appropriate ecological metrics are used to quantify these components. We analysed a number of published definitions of EI to arrive at our own working definition, which included four core components: nativeness, pristineness, diversity and resilience. We then developed a methodology for assessing EI based on the quantification of these components, recognising that this requires identifying ecosystems of interest and, if possible, establishing their reference conditions. Several indices that could be used to measure each of the four core components were then evaluated using a standard set of assessment criteria. For stream ecosystems, distributions of native species and habitat ecotyping can be used to reduce geographical variability in EI. In contrast, for lakes, ecotyping is not as useful and EI can instead be defined and assessed as the degree of departure of key metrics from threshold levels delimiting unfavourable conditions. We conclude with a list of potentially useful metrics for assessing EI in New Zealand streams and lakes, and provide information on how these might be combined in a multimetric index of EI.

Keywords: ecological integrity, biological integrity, structure, function, indicators, nativeness, pristineness, diversity, resilience, lake, river

This paper may be cited as:
Schallenberg, M.; Kelly, D.; Clapcott, J.; Death, R.; MacNeil, C.; Young, R.; Sorrell, B.; Scarsbrook, M. 2011: Approaches to assessing ecological integrity of New Zealand freshwaters. *Science for Conservation 307*. Department of Conservation, Wellington. 84 p.

1. Introduction

A major challenge facing the Department of Conservation (DOC) and other environment agencies is how to allocate scarce resources to best conserve the ecological values of New Zealand's diverse freshwater ecosystems. Although a range of ecological indicators have been developed to quantify ecological values, there currently exists no formalised way for defining ecosystem values in a holistic manner (whole-ecosystem values). This review investigates the potential utility of a concept known as ecological integrity (EI) for quantifying and integrating a wide range of ecological values for freshwater ecosystems.

For aquatic systems, it is important to look beyond individuals and populations to the context of the catchment (Fischman 2004). The concept of EI aims to quantify ecosystem structure and function in this context, taking into account physical, chemical and biological components (Mattson & Angermeier 2007). Integrity implies a state of being unimpaired or sound; a quality of being whole or complete. It has been argued that ecosystems subject to external anthropogenic disturbance will retain their integrity if all of their components and all of the relationships between them are preserved (De Leo & Levin 1997).

To quantify some human-induced pressures on freshwater values, DOC has assembled a series of human pressure data layers that are geo-referenced to specific surface water catchments (Leathwick & Julian 2007)[1]. These pressure layers include measures of the percentage of native vegetation in the catchment, percentage of impervious surface cover, predicted stream nitrogen loads, dam effects, mine and industrial discharge effects, and the probability of occurrence of a range of introduced fish. Unlike traditional site-specific measures of aquatic ecosystem integrity (e.g. biotic indices, water quality), these measures are both spatially explicit and scalable from stream reach to the nation. However, there are limited empirical data to define and quantify the relationships between these human pressure variables and indicators of EI. This multi-agency project funded by the Foundation for Research Science and Technology's Cross Departmental Research Pool (CDRP) has aimed to address this gap.

The purpose of this report is to define the EI of New Zealand's freshwaters by identifying key conceptual components of EI. Current definitions of EI in the scientific literature are evaluated to arrive at a definition of EI that is appropriate to the conservation of New Zealand's freshwater environments[2]. A number of indicators of EI in New Zealand lakes and rivers are then outlined. Finally, a suite of potentially suitable metrics for measuring EI are recommended.

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[1] Completed by the Department of Conservation as part of its Natural Heritage Management Strategy (NHMS) and its contribution to the New Zealand Government's Sustainable Water Programme of Action and assessment of Waterbodies of National Importance (WONI) (Chadderton et al. 2004).

[2] A parallel line of research was pursued on inland palustrine wetlands by Landcare Research and National Institute of Water & Atmospheric Research (NIWA) for assessing wetland condition (Clarkson et al. 2004).

2. What is ecological integrity?

Ecological integrity is a concept that is widely adopted by ecologists, policy makers, resource managers and the broader community. Like many ecological concepts that seem intuitive (e.g. biodiversity, resilience, stability), what appears to be reality is often illusory (Hurlbert 1971; Peters 1991). While there seems to be consensus on the characteristics of ecosystems with extremely high integrity (e.g. remote national parks) and extremely low integrity (e.g. areas with severe pollution), EI should define a measurable continuum. However, different interest groups often have quite disparate perceptions of how to weight diverse ecological characteristics to achieve a robust measure of EI. For example, salmonids have been introduced into many of New Zealand's lakes and rivers, including those in remote areas with relatively pristine catchments; these introduced predators can have significant effects on ecosystem structure and function (e.g. McIntosh & Townsend 1996), yet they are frequently seen as an important component of New Zealand's freshwater ecosystems, as evidenced by their protection under environmental legislation (e.g. Resource Management Act 1991). Since EI is a normative concept (subjective, relating to prescribed norms or human values), it is no surprise that there are many different definitions and interpretations of EI. Normative concepts, while commonly used in ecological science, are problematic from a strictly scientific point of view (Callicott et al. 1999; Hunter 2000). Therefore, we begin this report by examining what constitutes the complex concept of ecological integrity.

2.1 CONCEPTUAL DEFINITIONS

Fundamentally, most definitions of EI refer to whole or complete ecosystems, where all components are present and operating in an unimpaired fashion. Manuel-Navarrete et al. (2004) suggested that definitions of EI have generally taken one of four perspectives:

1. Wilderness-normative—referenced to a pristine state with an absence of human influence
2. Systematic-normative—dynamic self-organising systems that always have a human influence
3. Ecosystem-pluralistic—incorporates social values and does not prescribe how things should be
4. Transpersonal-collaborative—an organic concept evolving from social responsibility

Ecosystem health, while superficially similar to ecological integrity, is in fact distinguishable from it, defining the state of an ecosystem in terms of the stresses put on it, and its ability to keep providing products and processes (i.e. goods and services) for both economic and ecological means (Rapport et al. 1998a). In this context, ecosystem health is indicative of the preferred state of sites that have been modified by human activity, ensuring that their ongoing use does not degrade them for future use (Steedman 1994; Scrimgeour & Wicklum 1996).

By contrast, full integrity is attained when human actions have little or no influence on sites and when the biological community reflects the influence of ecological, rather than human, processes (Karr 1991; Angermeir & Karr 1994). For example, from a Māori perspective, the assessment of *mahinga kai*, or food resources, within a waterway could be interpreted as an indicator of ecosystem health, whereas *mauri*, or the essential life force, of a waterway could reflect EI (Tipa & Teirney 2003).

It has been argued that because human values are inherent in all definitions of EI, it is a useful and effective concept for engaging society in environmental debate and policy making (Robertson & Hull 2001; Tipa & Teirney 2003). However, while Manuel-Navarrete et al. (2004) argued for developing more transpersonal-collaborative approaches, the objectives of DOC's freshwater conservation management has been focussed principally on natural heritage values (Chadderton et al. 2004). Thus, we have aligned ourselves with most freshwater ecologists who have tended to take the wilderness-normative perspective that does not explicitly consider societal values, allowing for a more objective and rigorous assessment of EI. Although the wilderness-normative approach has generally been the theoretical basis of EI, there has been limited agreement on how to measure it.

2.2 PRACTICAL DEFINITIONS

Scientific attempts have been made to develop indices measuring components similar to EI. For example, Karr & Dudley (1981) developed indices of biotic integrity (IBI) based on a suite of indicators of anthropogenic impacts on biological communities, and defined biological integrity as:

> '... the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms.'

This definition was then later refined by Karr (1996) to:

> '... the ability to support and maintain a balanced, integrated, adaptive biologic system having the full range of elements and processes expected in the natural habitat of a region.'

This latter definition is a useful way of relating to natural heritage, where the focus is on the biological constituents of an ecosystem and the ability of the ecosystem to maintain processes that support the biology. However, this definition appears to neglect abiotic components that are important in structuring ecosystems. Barbour et al. (2000) suggested that 'the attainment of ecological integrity requires the attainment of three elements: physical, chemical, and biological integrity', and the concept of biological integrity is all encompassing, incorporating aspects of physical and chemical integrity. Furthermore, biogeographical and evolutionary processes are fundamental properties of an ecosystem's ability to organise, regenerate, reproduce, sustain, adapt, develop and evolve (Westra et al. 2000; Kutsch et al. 2001; Westra 2005; Oliveira & Cortes 2006).

Karr's and Dudley's definitions of biological integrity and their IBI approach have found many supporters, and this work has had a large impact on research in the field (e.g. Anon. 1990; Lyons 1992; Oberdorff & Hughes 1992) and led to the evolution of the concept of EI. However, despite the widespread uptake of such approaches, criticism has been raised over its subjectivity and lack of theoretical underpinnings (Peters 1991; Shrader-Fréchette & McCoy 1993; Sagoff 2000).

Partly in response to this, Ulanowicz and colleagues developed a thermodynamic approach for defining and measuring EI (Wulff & Ulanowicz 1989; Kay & Shneider 1992; Jørgensen 1995; Ulanowicz 2000), inspired by Odum's (1969) classic analysis of ecosystem development. Briefly, the Ulanowicz thermodynamic approach focuses on theoretically-based, generalisable measures of EI, combined into a composite index termed ecological ascendency. Ascendancy integrates an ecosystem's capacity for development, its performance, its resiliance and its developmental potential. These are only measurable via complex thermodynamic and trophic models. Hence, this approach is data intensive and requires sophisticated modelling skills. To date, few attempts to estimate ascendancy have been made. Kay & Schneider (1992), Jorgensen (1995), and Xu (2005) promoted a somewhat different thermodynamic approach whereby EI is defined as the ability of ecosystems to dissipate energy gradients (eco-exergy). While such thermodynamic models of EI may have a sound theoretical basis and may be generalisable, they have failed to capture the interest of practitioners and managers, and remain largely in the realms of theoretical and computational ecology. Consequently, most commonly used definitions of EI have been developed from a biological community approach rather than a thermodynamic approach.

Definitions of EI based on community ecology are related to the concept of biodiversity. Biodiversity, as defined by the Convention on Biological Diversity, is the variability among living organisms from all sources, including terrestrial, marine and other aquatic ecosystems, and the ecological complexes of which they are part, including diversity within species, between species and of ecosystems (Angemeir & Karr 1994). However, Miller (1991) provided some interesting discussion of EI with regards to considering genome-level components, community diversity and their relation to ecosystem function. Thus, EI could be viewed at several levels of organisation.

Bunn & Davies (2000) defined EI as 'minimal deviation from a natural reference condition'. This concept of the 'natural' state is present to some extent in most, if not all, definitions of EI. The implication is that an ecological state with high integrity will be more capable of supporting and maintaining a balanced, adaptive community of organisms with a diversity and organisation comparable to that which would be found in an unimpacted system (Karr & Dudley 1981). Therefore, it is desirable to determine the state of an ecosystem prior to it being impacted, recognising that systems change naturally, and ensuring that an appropriate time frame for assessing changes in EI is used. From this, it is considered that natural changes to EI (e.g. natural disasters) should not affect estimates of EI and, therefore, appropriate reference time frames should not be confounded by natural variation (as discussed in section 4.1).

2.3 OPERATIONAL DEFINITION

Ultimately, the common themes running through the definitions of EI outlined in the previous sections involve:

- The preservation of ecological structure and function, and their maintenance over time
- The recognition of a gradient of EI from high to low, such that natural or unimpacted ecosystems exhibit the highest integrity

In essence, EI can be defined as the preservation, to the greatest extent possible, of the condition that would be found if natural processes were allowed to predominate; an ecosystem has integrity when all of its components are maintained close to the natural condition. Therefore, for the purposes of this study, we define EI as:

> *The degree to which the physical, chemical and biological components (including composition, structure and process) of an ecosystem and their relationships are present, functioning and maintained close to a reference condition reflecting negligible of minimal anthropogenic impacts.*

The inclusion of minimally impacted, or near-reference, conditions in our definition is to address problems in ascertaining true reference conditions where human modification is pervasive in the landscape (e.g. many lowland water bodies).

3. What are the main components of ecological integrity?

Ecological integrity is a composite concept; it encompasses many ecological components. A review of 18 working definitions cited from the literature was undertaken to identify the range of components encompassed within EI. Although there was some degree of divergence in the terminologies used (e.g. naturalness, pristineness and wildness), four main components arose from most of the definitions that could be related to conservation of natural heritage. These were nativeness, pristineness, diversity, and resilience or adaptability. As such, EI can be viewed as a normative epi-phenomenon or emergent property of complex ecosystems. Below, we briefly address the four factors that have generally been identified as essential components of EI.

3.1 NATIVENESS

In a global context, New Zealand ecosystems contain a disproportionately large proportion of endemic species, and are also highly susceptible to invasion by exotic species. As such, nativeness was viewed as an important aspect of EI. Ecosystems are composed of a large number of constituent taxa, from microbes to higher organisms. The concept of nativeness refers to the degree to which an ecosystem's structural composition is dominated by the indigenous biota characteristic of the particular region. The concept relies on knowledge of the taxonomic makeup of the ecosystem as it exists in a reference state (Solheim 2005). A high degree of dominance of native species often equates to high ecological condition or integrity (Clayton & Edwards 2006a). However, in practice, nativeness criteria have generally applied only to conspicuous taxa that are readily taxonomically identifiable.

3.2 PRISTINENESS

Like nativeness, pristineness is a concept strongly linked to a reference state, requiring knowledge of the natural state of an ecosystem. This component of EI relates to a wide array of structural, functional and physico-chemical elements, but is not necessarily dependent on indigenous biota constituting structural and functional elements.

An important aspect of pristineness is connectivity, which refers to the dynamic interactions within and between habitats. Natural freshwater ecosystems can be viewed as complex mosaics of inter-connected habitats (Ward et al. 2002), characterised by intimate connections to their catchments (Hynes 1975). Longitudinal, lateral and vertical connections exist within and between ecosystems, and boundaries between ecosystems vary over time. Human activities in the landscape tend to alter these connections. Connectivity also relates to the maintenance of natural migration pathways, allowing recolonisation following perturbation (and, hence, also being a feature of resilience—see below), and the natural flow of genes between populations (Moss 2007).

3.3 DIVERSITY

The diversity of ecosystems is often a fundamental component considered when undertaking holistic management of ecosystems (Loreau et al. 2001; Duffy 2002). Biological diversity appears to be the most widely recognised ecological value and is covered by the United Nations Convention on Biological Diversity, to which New Zealand is a signatory. To implement the convention, New Zealand has enacted a National Biodiversity Strategy and Action Plan, which, among other goals, aims to halt the decline of indigenous biodiversity.

The classical definition of diversity in ecology comprises two components, richness (the number of taxa) and evenness (the distribution of individuals amongst taxa), which are often combined into diversity indices. Almost all measures of richness are influenced by variation in sampling effort, with the taxonomic richness of communities often increasing as sampling effort increases until a plateau is reached; this precludes comparisons between sites unless sampling effort was sufficient to reach the plateaus. Diversity is also dependent on geographical scale (see section 4.2.1).

There have been many reviews concerning diversity indices (e.g. Magurran 1988, 2004), and the list of indices, each with its own advantages and disadvantages, appears to be ever-growing (Lee et al. 2005). Abundance weighting is often used to normalise diversity measures (to assess diversity evenness across taxa).

Therefore, the inclusion of diversity as a component of EI requires clarification of the component in question (e.g. genes, communities and habitats); the intent to link to a reference condition or not; the intent to use abundance weighting or not; and geographical scale.

Spatial heterogeneity (or diversity) of habitats is important for maintaining biological diversity. Maximising diversity may be a desirable goal regardless of a reference state, as complexity is often seen to relate to maintenance and stability, although the strength of the general relationship between diversity and ecosystem function remains elusive (Bengtsson 1998; Loreau et al. 2001). Recent research has focussed on characterising direct linkages between structural diversity and functional integrity of ecosystems. For example, the diversity of plants has been strongly linked with both productivity and nutrient cycling (Tilman 1999). Within food webs, most evidence of the relationship between diversity and function is seen through the indirect effects of species losses at higher trophic levels, such as in trophic cascades (Carpenter & Kitchell 1987; Wooton 1994).

3.4 RESILIENCE

Resilience, which is sometimes referred to as 'self-renewal capacity' and 'long-term viability', is the ability of ecosystems to exist into the future despite natural environmental fluctuation, and can relate to both structural or functional ecosystem components (Holling 1973; Westman 1978). Adaptation, acclimation, and other forms of biotic and ecosystem change are essential for long-term viability, particularly in highly variable environments, indicating that resilience in structural and functional ecosystem attributes is integral to EI. As such, resilience extends the concept of EI to include ecosystem dynamics over time, and could potentially include longer-term natural changes such as climate change, sea level rise, and species colonisations and extinctions. Thus, our assessment of resilience is aimed at quantifying the probability of maintaining an ecosystem's structural and functional characteristics under varying degrees of human pressure.

4. Practical considerations for assessing ecological integrity

To create a workable and cost-effective EI assessment scheme, it is crucial that appropriate ecological indicators of EI are selected (Carignan & Villard 2002).

Several authors have cited attributes of ecological indicators that are effective in monitoring schemes. For example, Norris & Hawkins (2000) identified that for metrics to be effective indicators of ecological health, they must simplify ecological phenomena, provide interpretable outputs, respond predictably to pressure, be at an appropriate scale, relate to management goals and be scientifically defensible.

4.1 NORMALISATION TO REFERENCE CONDITION

Many of the components underpinning our operational definition of EI (see section 2.3) depend on the ability to quantify the degree of departure of a measured ecosystem state or condition from a reference ecological state or condition. Stoddard et al. (2006) outlined four possible types of reference condition: a minimally disturbed condition, a historic condition, the least disturbed condition and the best attainable condition. Thus, in the assessment of EI, the chosen reference state must also be clearly defined and justified.

Generally, the ideal reference state is the ecological state of the system immediately prior to the first anthropogenic impacts (e.g. Lee et al. 2005), which represents the minimally disturbed or historic condition. However, it is difficult to obtain data on the state or condition of freshwater ecosystems prior to human influence. Thus, palaeo-ecological reconstructions are increasingly being undertaken in New Zealand freshwaters (Woodward & Shulmeister 2005; Augustinus et al. 2006). These methods rely on datable signatures of past environmental states preserved in sediments and soils, as recovered by coring methods. However, palaeoecological methods are costly and time consuming, and reconstructions of many key variables are limited to the number of existing, New Zealand-specific transfer functions, which are the statistical models relating environmental conditions to biological communities upon which many paleoecological reconstructions are based (e.g. Reid 2005; Woodward & Shulmeister 2005; Cochrane et al. 2007). Though powerful, these techniques are not suited to widespread use due to cost.

As an alternative, a reference habitat or ecosystem that is deemed similar in critical factors, but which has not been influenced by humans or exotic species, can be nominated as a present example of conditions exemplifying the modified habitat prior to human modification (Brinson & Rheinhardt 1996). This often involves attempts to group habitats into similar ecotypes (see section 4.1.1) and may be subject to serious errors, as a deep understanding of key variables driving EI is a prerequisite for the successful selection of appropriate reference systems. Furthermore, the majority of New Zealand's freshwater ecosystems, especially those in areas amenable to human habitation, have already been perturbed by humans and/or exotic species, leaving few appropriate reference systems for some types of freshwater ecosystems. Therefore, more often than not, a least disturbed or best attainable state is adopted as the reference condition (Stoddard et al. 2006), which results in a compromised estimation of EI.

Reference states can also be inferred by expert assessment and interpretation of key environmental data presently describing a site (Moss 2007). Problems with this approach may include inconsistencies in expert opinions and interpretations, a lack of key environmental data to underpin assessments, and a lack of historical perspective or local knowledge concerning local environmental change.

Predictive modelling can also be used to infer reference conditions. This approach predicts reference conditions based on statistical relationships between ecological responses (e.g. invertebrate community structure) and environmental variables (e.g. dissolved oxygen). Environmental surrogate data are then used to predict faunal distribution patterns for areas where biological data are unavailable, and may be done in the context of developing an environmental classification (Snelder 2006; Leathwick et al. 2008b). This procedure has been applied to freshwater environments in New Zealand and is described in more detail in the next section.

Regardless of the approach used, it is important to recognise that unimpacted ecosystems do not maintain a static state; rather, they are dynamic systems, continuously changing in response to natural environmental and biological variability. As such, this dynamic variability, as well as rates of change, may be critical ecological attributes to identify and assess. Thus, the use of a reference state reflecting conditions prior to human influence to establish a condition of high EI does not acknowledge that reference environmental conditions (e.g. climate, hydrology, salinity, condition of catchment, species assemblages) might have varied naturally over the period of human influence. Consequently, normalising to a reference condition may be viewed as fundamentally flawed because it reflects a static view of ecosystem development over time. Furthermore, as all human (or human-mediated) influences can never be eliminated, the reference condition will never be achievable and may no longer be the optimal state to set as a goal for management or restoration. Thus, the tendency to normalise ecosystem assessments to a reference condition has been philosophically criticised for creating a fundamental separation between humans and ecosystems, whereby the presence of *Homo sapiens* and human-associated ecologies are strictly considered as perturbations that only decrease EI (see Holland 2000).

4.1.1 Ecotyping

Some approaches to measuring EI rely on habitat ecotyping for establishing reference condition. The assumption is made that pairs, or larger numbers, of ecosystems can be effectively treated as replicates. All natural ecosystems are distinct, but many have similarities with regard to specific ecological characteristics. Appropriate ecotyping requires that effective replicates be established, whereby ecosystems that share ecological characteristics relevant to EI are grouped into type classes. Variation in characteristics not relevant to EI can thus be ignored. However, it is essential that this is done correctly, otherwise variance due to mismatch of an ecosystem and its reference ecosystem can undermine the determination of EI.

Considerable effort has been expended in developing environmental classifications in New Zealand freshwaters. At present, two main classification systems exist: the River Environment Classification (REC), which is a rule-based hierarchical classification based predominantly on physical variables (Snelder et al. 2004), and the Freshwater Environments of New Zealand (FWENZ) classification, which is a multivariate classification based on physical ecotyping but optimised against a spatial database of biological community data (Leathwick et al. 2008b).

The process by which FWENZ environments were classified is based on a relatively new statistical method called Generalised Dissimilarity Modelling (GDM), which provides information on typology and also quantifies the degree of environmental similarity (Ferrier et al. 2007). Separate FWENZ classification systems were developed for river (Leathwick et al. 2008b) and lake (Snelder 2006) environments. Although much of the underlying catchment data were common to both classification systems, variables driving the classifications were selected and weighted according to what was considered to drive the underlying ecology of the two systems. Additionally, different biological databases were used to calibrate the physical environment typologies, with lakes optimised to a submerged macrophyte dataset, and rivers optimised to macroinvertebrate and fish datasets (Snelder 2006; Leathwick et al. 2008b).

4.2 VARIABILITY AND SCALING

At a lake or river site, there are many sources of variation that may influence the precision of any indicator, including spatial and temporal sampling variation (e.g. spatial heterogeneity, or diel, seasonal and annual variations), variation in the precision of sampling method (number of sampling units, habitats/ area sampled), and errors in sample processing and taxonomic identification (Bund & Solilmini 2006; Clarke & Hering 2006). For example, in a study analysing the spatio-temporal variability of 16 Swedish lakes, it was found that standardised effect sizes (the magnitude of change of indicator metrics with anthropogenic stress, expressed in standard deviation units) and estimates of statistical power varied markedly among habitats and indicators used (Johnson 1998). Therefore, it is important that extraneous variability be identified and minimised if signals due to anthropogenic pressures are to be quantified (Jackson et al. 2000; see also section 4.4). To this end, Lorenz & Clarke (2006) introduced the concept of 'sample coherence', which allowed estimates to be made of the relative importance of within-site, within-season and within-method variation. It is important to acknowledge issues of variability and error in ecological indicators because they emphasise the consideration of appropriate spatio-temporal scales and methods when measuring ecological attributes.

4.2.1 Spatial scales

When considering EI, it is important to delineate ecosystem boundaries. Although they are often implied, common definitions of 'ecosystem' tend not to define spatial boundaries. Freshwaters can be defined as the medium of liquid water that moves through the landscape from headwaters to the sea. Freshwater habitats are influenced by upstream conditions. Therefore, it can be argued that, at least from a biogeochemical point of view, the most appropriate spatial scale for delineating freshwater ecosystems is the catchment scale, including both surface water and interconnected terrestrial and groundwater habitats. Aquatic habitats within the ecosystem are themselves influenced by drivers at a hierarchy of spatial scales (Lodge et al. 1998). For example, within a stream reach, environmental drivers operate from the catchment scale to the scale of individual substrate particles.

The spatial classification systems that have been developed for New Zealand rivers operate at the reach scale (i.e. river segments between tributary confluences), with subsequent spatial aggregation at a range of levels. For example, FWENZ aggregates river reaches for the entire stream network into broad classes at the 20-group level, or at its finest level of classification to a 400-group level. Thus, in this report we also focus the development of indicators of river EI on these spatial scales.

Lakes also exhibit high spatial variability in habitats, ecological structure and functioning. Such variability should be accounted for when developing sampling strategies for the assessment of EI. We suggest partitioning lakes into open water and littoral / shallow water sampling stations. Sampling of thermally stratified lakes should also be vertically stratified to reflect the isolation of, and differences between, the hypolimnion and mixed layers.

4.2.2 Temporal scales

The consideration of appropriate temporal scales of ecosystem structure and function is important with respect to defining a reference condition, sampling to determine EI for particular ecosystems, and predicting long-term viability and resilience. Important temporal variability in freshwater ecosystems occurs over a variety of time scales. For example, floods can substantially alter the characteristics of riverine ecosystems in a matter of minutes to hours, with effects that last for weeks, months or even years; in lakes, significant variations in populations of algae and zooplankton can occur within days; temperature and solar radiation variations play important roles in structuring ecosystem biota and regulating key processes on a seasonal basis; significant climatic cycles (e.g. El Niño southern oscillation, interdecadal Pacific oscillation) and distant causes of climate change (e.g. vulcanism) can result in significant interannual variability; and on a broader scale, geomorphological and successional changes influence ecosystems on time scales of centuries and millennia. Long-term climatic and geological variations cause geographical distributions of organisms to shift with latitude and altitude, so that species immigrations and extirpations occur on time scales synchronous with climatic and geological variation.

Thus, one must consider the appropriate time scales over which EI is to be measured, while recognising the limitations that restrict sampling across multiple time scales. For example, an approach to the quantification of interannual variation could involve sampling only once or a few times per year whilst constraining sampling times to be consistent among years.

4.3 EASE OF SAMPLING AND ANALYSIS

Some of the most practical aspects considered by management agencies when assessing indicators to be incorporated into monitoring programmes are the ease of sampling and the costs of analysing samples. These aspects will often dictate the number of locations that can be monitored and the frequency at which monitoring can be conducted. Although these factors alone rarely determine the utility of a particular indicator, they do influence the likelihood of indicator use by management agencies in their monitoring programmes. ‘Ease of sampling’ and ‘ease of analysis’ can both quite differentially influence the resources required for a monitoring programme. For example, indicators of fish community composition usually require substantial resources in the form of field teams to collect sufficiently large sample sizes to cover the range of habitat types required to obtain an accurate assessment of the population. However, once the field data are collected, there are few additional costs associated with analysis of samples. In contrast, indicators such as water chemistry and benthic invertebrates are relatively easily collected from the field, but can require significant resources in terms of laboratory analysis and/or taxonomic identifications. Similarly, the use of certain indicators may be limited by the state of existing knowledge on taxonomy at some localities. Because resources available for monitoring are often a key consideration, we also considered this when assessing EI indicators.

4.3.1 Taxonomic resolution

By definition, EI should be established based on an analysis of a full ecosystem inventory. Unfortunately, this is impossible due to the inherent complexity of ecosystems. Alternatively, the assessment of EI should include indicators of ecological structure and functioning across the five kingdoms of organisms (bacteria, plants, fungi, protists and animals). Unfortunately, taxonomic difficulties have precluded the meaningful assessment of diversity and community structure for many microbial communities, although rapidly advancing genetic methods are remedying this to some degree (Lear et al. 2009).

Thus, assessments of EI, ecosystem health or condition often focus on conspicuous ‘indicator’ organisms, including key (or keystone) species that exert unusually large influences on ecosystem structure and function (see Lee et al. 2005). Furthermore, we suggest that good indicator organisms or assemblages are those that are most sensitive to pressures (Schindler 1987) and those that play a key role in providing valued ecosystem services.

An example of an indicator taxonomic assemblage is the percentage of aquatic invertebrate taxa collected at a stream site belonging to the insect orders Ephemeroptera, Trichoptera and Plecoptera (%EPT; Rosenberg & Resh 1993). An advantage of this indicator is that the taxonomy of these groups tends to be well characterised, whereas the taxonomy of other insect larvae (e.g. the Diptera) is more challenging. However, the accuracy of relatively simple taxonomic indicators still varies in relation to the taxonomic acuity of individual researchers, resulting in problems of comparison between studies. Therefore, a consistent taxonomic resolution should be employed in such analyses. Although species can be lumped into genera or families, important ecological information can be lost in this process. It has been shown that the discrimination of different samples and sites based on stream invertebrate communities is reduced when species are lumped into families (Lorenz & Clarke 2006).

4.4 SENSITIVITY TO PRESSURE GRADIENTS

When determining useful measures of EI, it is important to consider the nature of the expected relationships between the measures and the pressure gradients of interest. The most useful measures will be those that have a high sensitivity and show strong monotonic responses to the main pressure gradients. The determination of standardised effect sizes (i.e. the magnitude of response) may be a useful way to prioritise and/or statistically weight EI metrics and indices. Analyses of the strengths of relationships between various proposed measures of EI and selected pressure gradients for New Zealand freshwaters have been provided by Scarsbrook (2008), Drake et al. (2009) and Clapcott et al. (2010).

Ecological responses are not necessarily linearly correlated with pressures; relationships may be curvilinear, unimodal, or show distinct thresholds delineating two or more stable states. For example, taxonomic diversity often exhibits unimodal relationships along gradients of ecological disturbance (Connell 1978; Flöder & Sommer 1999), while water quality in some shallow lakes may alternate between two stable states along gradients of nutrient loading (Scheffer 1998).

The important work of Scheffer (1998) not only illustrates how ecosystem feedbacks can operate to keep shallow lakes in ecological homeostasis despite changes in pressures, but also illustrates the phenomenon of hysteresis (time lags) between pressures and responses in shallow lakes. These phenomena, which are common in ecosystems, may confound deterministic relationships between pressures and ecological responses, undermining attempts to develop simple EI indices (models) with high predictive power.

A good understanding of the sensitivities of various metrics and the nature of their relationships along pressure gradients is essential for the appropriate statistical treatment in the development of more complex, multi-metric measures of EI. For example, according to Connell's intermediate disturbance hypothesis (Connell 1978), low taxonomic diversity can be indicative of both low and high levels of ecological disturbance, such that other variables are required to be able to accurately relate diversity to a level of disturbance. In systems that have sharply defined thresholds between desirable and undesirable states, it may be useful to define an EI metric as the departure distance of its present value from the threshold value.

A theoretical definition of EI allows for the development of models linking EI to pressure gradients by identifying pressure gradients that correlate well with the measures of EI. In contrast, EI may also be defined by seeking the subset of ecosystem measures that correlate most strongly with the pressure gradients of interest. In the latter approach, EI can be empirically defined as an index comprising the ecosystem characteristics most sensitive to anthropogenic pressures.

Whether EI is theoretically or empirically defined depends both on the question driving the research and the level of confidence in the ecological and pressure gradient data at hand. However, it is important to consider the varieties of relationships that can exist between EI and anthropogenic pressures in order to maximise the amount of ecological information that can be extracted from EI assessments.

4.5 MAIN CRITERIA FOR ASSESSING INDICATORS OF ECOLOGICAL INTEGRITY

We suggest seven criteria for determining appropriate EI indicators for use in New Zealand river and lake ecosystems. Good indicators should clearly and effectively assess departure of structure and/or functionality from a minimally impacted reference condition; or, in the absence of such a minimally impacted reference condition, respond in a predictable manner to anthropogenic pressure gradients that are relevant to management goals. The criteria by which we assessed the suitability of a variety of EI indicators were subjectively ranked on a 1–3 point scale to allow an overall quantitative ranking of all the indicators evaluated.

The seven criteria were:

1. **Ease of sampling and analysis**
   This criterion takes into account the resources required for field sampling and laboratory analysis, whilst considering the taxonomic resolution for biologically based indicators. We numerically rated all indicators under this criterion on a 1–3 point scale, from easy (3) to difficult (1).
2. **Potential geographic coverage**
   This criterion takes into account the geographic coverage of the particular indicator, considering both its geographic spread as well as its application to the wide variety of environment types that occur nationally. We numerically rated all indicators under this criterion on a 1–3 point scale, from narrow (1) to wide (3) coverage.
3. **Relation to EI components (e.g. nativeness, pristineness)**
   This criterion takes into account whether the indicator is directly or indirectly linked to the EI component, and the strength of that linkage. We numerically rated all indicators under this criterion on a 1–3 point scale, from a weak (1) to strong (3) linkage to the particular EI component being assessed.
4. **Sensitivity to pressure gradients**
   This criterion takes into account present scientific understanding of the relationship between the indicator and various human pressures of concern, such as nutrient enrichment, sedimentation, invasive species, etc. We numerically rated all indicators under this criterion on a 1–3 point scale, from unknown or weak relationships (1) to well-documented strong relationships (3) to various human pressure gradients.
5. **Calibration to reference condition**
   This criterion takes into account the ability to predict reference condition for the indicator amongst a range of environment types. We numerically rated all indicators under this criterion on a 1–3 point scale, from unknown or difficult (1) to well known and easy (3) to predict reference condition for the indicator in a range of environment types.
6. **Temporal variability**
   This criterion takes into account the temporal variability of the indicator amongst a range of environment types. We numerically rated all indicators under this criterion on a 1–3 point scale, from high (1) to low (3) temporal variability.
7. **Use nationally and internationally**
   This criterion takes into account the current usage of this indicator by freshwater management agencies. We numerically rated all indicators under this criterion on a 1–3 point scale, from rare (1) to common (3) use in New Zealand and internationally. Although a high ranking for this criterion will generally mean good performance under other criteria categories (e.g. greater knowledge about the relationship with pressure gradients, and calibration to reference condition), we felt present use was an important consideration for uptake by management agencies.

The overall ranking of indicators was calculated from the sum of numeric scores for each of the seven criteria, and recommendations of indicators best suited for assessing the various EI components (section 6) was selected from the highest scoring indicators for each category.

5. Indices of ecological integrity for rivers and lakes

Although the definition and many of the components of EI utilised in this report are equally applicable to rivers and lakes in New Zealand, the degree of development of suitable metrics to evaluate these differ between the two habitat types. Furthermore, the management of rivers and lakes is often split within management agencies. We have therefore dealt with the habitats separately within the same general framework, although on occasion this has resulted in some repetition. We believe this will allow managers to more rapidly access the appropriate information.

5.1 RIVERS

The concepts of ecosystem health, pristineness and EI are generally well understood for rivers (Costanza et al. 1992; Cairns et al. 1993; Shrader-Fréchette 1994; Steedman 1994; Christensen et al. 1996; De Leo & Levin 1997; Rapport et al. 1998b; Boulton 1999). River managers, developers, environmental law makers and policy planners apparently understand and concur on what these terms mean. However, this is of little consequence if ecologists and those charged with monitoring the environment do not have precise definitions of what to measure and guidelines against which to judge the degree of departure from an acceptable state.

Our operational definition of EI encourages the use of what is generally termed the 'reference state' approach (Hughes et al. 1986; Boothroyd & Stark 2000; Boothroyd et al. 2002; Mazor et al. 2006). The advantages of using this approach are that there are established criteria for reference site selection in New Zealand (Boothroyd et al. 2002) and the statistical modelling of reference states for rivers has been used successfully in some regions of New Zealand (Joy & Death 2000, 2003b, 2004b). The disadvantages include difficulty in finding suitable reference sites, as discussed in section 4.1.

5.1.1 Nativeness

To date, riverine systems in New Zealand have generally been less impacted by exotic invasive species than terrestrial habitats, with most river macroinvertebrate species being endemic (Winterbourn 2000). However, exotic fish species, particularly introduced salmonids, have had detrimental effects on native fish, invertebrates and ecosystem processes (Townsend & Crowl 1991; Townsend 1996; McDowall 2003; Simon & Townsend 2003). The assessment of riverine EI in New Zealand is problematic, as salmonids prefer habitats of high water quality, meaning that the Macroinvertebrate Community Index (MCI; one of the often applied indicators of EI) is likely to be high in rivers where the fish community may be dominated by exotic species. This demonstrates the value of looking at different components of EI simultaneously—salmonids will have a detrimental effect on 'nativeness' components, but little or no effect on 'pristineness' components. Given that few river macroinvertebrates are exotic, 'nativeness' metrics for macroinvertebrates are not addressed below. However, macrophytes, bryophytes and fish are potentially important. The scoring of separate indicators of riverine nativeness against the seven assessment criteria are included in Table 1, with a brief discussion of some of the key points for the indicators in the following sections.

Macrophytes and bryophytes

In Europe, macrophyte community composition has been used successfully to assess the ecological condition of rivers (Brabec & Szoszkiewicz 2006; Hering et al. 2006; Staniszewski et al. 2006; Szoszkiewicz et al. 2006); however, we are not aware of their use in New Zealand. Macrophytes are generally restricted to slower flowing lowland streams in New Zealand, limiting their widespread use in EI assessment. Furthermore, we know little about the pre-human condition of lowland streams in New Zealand, making it difficult to evaluate the expected flora under reference conditions. The advantages of macrophytes as ecological indicators includes their year-round presence (although they are subject to constant removal as part of flood management in many parts of New Zealand), relative longevity (months to years, meaning they can reflect environmental conditions impacting on plant growth over long time frames), and the relatively inexpensive costs associated with their sampling, monitoring and identification. Some of the metrics detailed in the lake section may prove useful as indicators, such as the diversity of native macrophytes and the percentage of exotic macrophytes. However, the lack of information on macrophyte distributions under reference conditions limits their usefulness as ecological indicators for rivers.

A number of studies have found strong relationships between environmental variables and bryophyte diversity patterns (e.g. Suren 1996; Heino et al. 2005). However, we are unaware of any studies that have used bryophytes as indicators of EI in rivers. As such, their limited use as indicators within New Zealand contributes to difficulties in taxonomic analysis, calibrating their distribution to reference condition, and understanding their overall relationship with nativeness in rivers.

TABLE 1. EVALUATION OF NATIVENESS METRICS FOR RIVERS AGAINST A SET OF ASSESSMENT CRITERIA. DETAILS OF THE ASSESSMENT CRITERIA ARE OUTLINED IN SECTION 4.5.

ASSESSMENT CRITERIA	MACROPHYTES	BRYOPHYTES	NUMBER OF INTRODUCED FISH SPECIES	PROPORTION OF EXPECTED NATIVE FISH SPECIES MISSING
Ease of sampling and analysis	Easy (3)	Moderate (2)	Moderate (2)	Moderate (2)
Potential geographic coverage	Narrow (1)	Moderate (2)	Wide (3)	Wide (3)
Relation to nativeness	Strong (3)	Unknown (1)	Strong (3)	Strong (3)
Sensitivity to pressure gradients	Unknown (1)	Unknown (1)	Moderate (2)	Moderate (2)
Normalisation to reference condition	Moderate (2)	Difficult (1)	Easy (3)	Moderate (2)
Temporal variability	Moderate (2)	Unknown (1)	Low (3)	Moderate (2)
Use in New Zealand	Common (3)	Rare (1)	Common (3)	Occasional (2)
Use in other countries	Common (3)	Occasional (2)	Common (3)	Common (3)
Rank [points]	3 [18]	4 [11]	1 [22]	2 [19]

Number of introduced fish species

In New Zealand, the fish fauna is arguably the component of the riverine biota that has been most affected by exotic introductions. However, fish have not been used extensively for monitoring EI in New Zealand rivers because of the marked altitudinal and biogeographical gradients in their distribution (McDowall 1990, 1993; Joy et al. 2000; Leathwick et al. 2008a), which contrasts with the situation in many other countries where altitude is not such a strong driver of fish community structure and fish are used more often as ecological indicators (Karr 1993; Angermeir & Karr 1994). The use of native fish metrics, such as native species richness, may be problematic due to high spatial variability. However, the number of introduced fish species could be used as a simple indicator of nativeness. As with macrophytes, fish offer the advantages that they are generally present year-round and are relatively simple and cost-effective to sample, monitor and identify, at least on a species presence/absence basis. There is also a considerable body of information on the distribution of fish in New Zealand within the NZ Freshwater Fisheries Database (http://fwdb.niwa.co.nz/; viewed June 2010), as well as the expected distributions of native species (Leathwick et al. 2008a). The disadvantage of this indicator is that the number of species counted can be strongly dependent on sampling effort and technique: the greater the effort and/or the more effective the technique, the more fish are observed and thus the more species are recorded (Ludwig & Reynolds 1988; Krebs 1999; Magurran 2004). Furthermore, the ecological effects of salmonids on 'nativeness' do not seem to be simply dependent on their presence or absence, but rather on a combination of their size and density distributions (Joy & Death 2004b).

Proportion of expected native species missing

The EI of rivers and streams of the lower North Island and Auckland has been assessed by several authors. McDowall & Taylor (2000) have evaluated differences in observed and expected fish faunas through direct observation, while other studies have used predictive modelling of fish (several variants of the RIVPACS observed/expected (O/E) approach; Joy & Death 2000, 2002; Olden et al. 2006). The O/E approach only uses species presence/absence in evaluation, although the modelled probability of expectation is abundance-dependent. One advantage of this method is that modelling reduces the sampling effort required and recent modelling by Leathwick et al. (2005, 2008a) offers potential to upscale this approach to a national scale. Therefore, the proportion of observed native fish in a river as a ratio of those predicted to occur in the absence of human alterations seems to offer good potential as a riverine indicator of 'nativeness'.

5.1.2 Pristineness

This component of EI includes structural, functional, physio-chemical and cultural elements. Hydrological connectedness for matter and energy flow and species dispersal is a key component of pristineness. We discuss a selection of measures for each of these components of pristineness.

Structural indicators

Structural indicators assess whether biota present at a site indicate pristineness. The scoring of separate indicators of riverine pristineness against the seven assessment criteria are included in Table 2, with a brief discussion of some of the key points for each of the indicators in the following sections.

%EPT

The relative abundance of macroinvertebrate taxa belonging to Ephemeroptera, Plecoptera and Trichoptera (%EPT) is a well-established metric for assessing water quality (Rosenberg & Resh 1993; Boothroyd & Stark 2000; Boothroyd et al. 2002). It is easy to measure and can facilitate the calculation of a macroinvertebrate index for datasets where taxonomic resolution is quite variable, since it uses only coarse taxonomic structure. However, since %EPT integrates information about several taxa, information about responses of individual taxa is lost. Whilst there are no established criteria for acceptable index values in New Zealand rivers, these could be derived with relatively little effort, with large datasets available through databases maintained for state of the environment monitoring. Furthermore, the relationship between %EPT and pressure gradients is relatively well known both in New Zealand (Quinn & Hickey 1990; Townsend et al. 1997a; Death & Joy 2004; Young & Collier 2009) and in other countries (Rosenberg & Resh 1993).

TABLE 2. EVALUATION OF STRUCTURAL PRISTINENESS METRICS FOR RIVERS AGAINST A SET OF ASSESSMENT CRITERIA. DETAILS OF THE ASSESSMENT CRITERIA ARE OUTLINED IN SECTION 4.5.

ASSESSMENT CRITERIA	%EPT	INVERT TRAITS	MCI	INVERT O/E	INVERT CMD[a]	FISH IBI	PERIPHYTON	CONNECT-EDNESS
Ease of sampling and analysis	Easy (3)	Difficult (1)	Easy (3)	Difficult (1)	Difficult (1)	Easy (3)	Moderate (2)	Difficult (1)
Potential geographic coverage	Wide (3)	Wide (3)	Wide (3)	Wide (3)	Wide (3)	Wide (3)	Wide (3)	Wide (3)
Relation to pristineness	Moderate (2)	Unknown (1)	Strong (3)	Strong (3)	Weak (1)	Strong (3)	Moderate (2)	Strong (3)
Sensitivity to pressure gradients	High[b] (3)	Moderate (2)	High[c] (3)	High[b] (3)	High[b] (3)	Moderate[d] (2)	Moderate[e] (2)	Moderate (2)
Normalisation to reference condition	Easy (3)	Difficult (1)	Easy (3)	Difficult (1)	Moderate (2)	Easy (1)	Difficult (1)	Moderate (2)
Temporal variability	Low (3)	Moderate (2)	Low (3)	High (1)	High (1)	Moderate (2)	High (1)	High (1)
Use in New Zealand	Common (3)	Rare (1)	Common (3)	Occasional (2)	Occasional (2)	Common (3)	Common (3)	Rare (1)
Use in other countries	Common (3)	Occasional (2)	Common[f] (3)	Common (3)	Common (3)	Common (3)	Common (3)	Rare (1)
Rank [points]	2 [23]	8 [13]	1 [24]	4 [17]	6 [16]	3 [20]	5 [17]	7 [14]

[a] Invertebrate community mulitvariate distance.
[b] Usually linear.
[c] Usually linear or threshold.
[d] Within regions developed.
[e] Usually log-linear.
[f] Similar indices.

TAXONOMIC TRAITS

The assessment of taxonomic traits is a relatively recent addition to the toolbox of biological measures of EI. This approach utilises a range of functional and structural information, such as feeding preferences and life-history characteristics, to group taxa with similar characteristics. Taxonomic trait indicators have several potential advantages over traditional indices and multivariate methods based on taxonomic composition (Doledec et al. 2006), with a key advantage being the removal of biogeography as a confounding effect when making assessments at large spatial scales. Taxonomic trait indicators may also provide insights into the mechanisms causing observed changes in functional and structural characteristics of communities, because rather than simply recording a loss or reduction in numbers of a species, it allows identification of the most sensitive life-history characteristics and can indicate potential causative mechanisms. However, in New Zealand, taxonomic trait indicators are a relatively new concept, and there has only been a limited amount of work to understand how they might relate to both natural (e.g. Townsend et al. 1997b) and anthropogenic (Doledec et al. 2006) perturbations, and how they could be calibrated to reference condition. Their usage in ecological monitoring also necessitates a high degree of taxonomic resolution because species within the same genera may have quite variable taxonomic traits.

MCI METRICS

MCI (Macroinvertebrate Community Index), QMCI (Quantitative MCI), SQMCI (Semi-quantitative MCI), and SBMCI (Soft Bottom MCI) (Stark 1985, 1998; Plafkin et al. 1989; Boothroyd & Stark 2000) are well established water quality assessment tools in New Zealand resource management, and an array of studies have linked these to land-use pressure gradients (e.g. Quinn et al. 1997; Townsend et al. 1997a; Doledec et al. 2006; Young & Collier 2009). The MCI sensitivity scoring system applied to taxa was developed to indicate levels of organic pollution in stony streams (Stark 1985), and so these scores may not be sensitive to other anthropogenic pressures, such as metals pollution (Stark & Maxted 2007). MCI is also reasonably strongly linked to other metrics of pristineness such as %EPT, making it intuitively simple to relate MCI to the composition of the macroinvertebrate community. Thresholds for different levels of water quality have been developed and are regularly used in resource management around New Zealand. However, Wright-Stow & Winterbourn (2003) suggested that threshold designation may not be consistent between MCI and QMCI. Recent work by Stark & Maxted (2007) has also identified that these threshold designations may vary considerably with river type (e.g. soft-bottomed v. hard-bottomed streams). Therefore, caution should be exercised when applying these thresholds. MCI, like other macroinvertebrate indices, can be influenced by seasonal trends or short-term disturbances such as floods, and thus necessitates some rules around the timing of sampling (e.g. not within 2 weeks of major flood, average of quarterly samples) to control for these factors.

OBSERVED/EXPECTED FISH AND/OR INVERTEBRATES

The observed v. expected (O/E) predictive modelling approach (RIVPACS, BEAST, AUSRIVAS) is well established throughout the world (Wright et al. 1984; Norris & Thoms 1999; Smith et al. 1999; Wright-Stow & Winterbourn 2003; Carlisle et al. 2008). Joy & Death (2000, 2002, 2004b, 2005) have used this approach in New Zealand to evaluate ecological condition. This approach is not sensitive to the particular type of pressure gradient, as the models are based on expected reference condition assemblage structure. This has the advantage of potentially being sensitive to a wide range of pressures, but has the disadvantage of not specifically relating to a particular pressure gradient of interest. There can also be difficulties in relating present condition to reference condition if sampled reference sites are also impacted by pressures (e.g. presence of introduced salmonids), thereby biasing the models. An advantage of this approach is that confounding natural environmental gradients will be minimised, as measured physical attributes are used to predict biotic assemblages using modelled relationships between the two at reference sites. Also, this approach retains species-specific information, although Van Sickle (2008) pointed out that information content is lost when data are collapsed to produce a simple O/E taxon richness index. The disadvantage of this approach is the intensity of sampling required, as the models require careful selection (Joy & Death 2003a) and sampling of between 20 and 200 reference sites depending on region size. At present, these models only assess what is absent from a test site at the time(s) of sampling compared to taxa that would be present given an estimated reference condition, so temporal variability must also be accounted for.

PERCENTAGE SIMILARITY OF COMMUNITY COMPOSITION

The percentage similarity of community composition uses a multivariate statistical method (e.g. Bray-Curtis distance) to assess the departure of a system from its mean reference condition (Van Sickle 2008). This is similar to the O/E approach outlined above, but considers the entire community rather than simply the absence of key taxa. The advantage of this approach is that it is not sensitive to the type of pressure gradient (i.e. it measures how much the biota have changed in response to a disturbance) or to a specific measure of EI (i.e. pristineness v. nativeness), as long as the reference community applied is appropriate to the natural physico-chemical conditions of the test sites. Thus, careful selection of reference sites is important in calibrating the model, and can be problematic for environment types where pristine reference sites are either unavailable or difficult to access. Statistical methods can sometimes be used to eliminate the effects of confounding natural environmental gradients if these are known and, thus, such statistical models may measure pressure gradients more accurately. Species identities are retained and most weight is given to the presence of key taxa, rather than their absence. The disadvantages are that this approach is not intuitive, requires significant effort sampling reference sites (c. 20–200 depending on region size) and the common taxa tend to carry more weight. As with other macroinvertebrate indices, sampling strategies need to account for seasonal and short-term variability of the communities.

FISH INDEX OF BIOTIC INTEGRITY

An Index of Biotic Integrity (IBI) is the standard international measure used when fish are desired as indicators for biomonitoring (Karr 1993; Boothroyd et al. 2002). It was once widely believed that this index could not be adapted for use in New Zealand because of the cosmopolitan niche habits of our native fish and the relatively low species richness (McDowall & Taylor 2000). However, Joy & Death (2004a) have demonstrated the potential of a fish IBI for New Zealand. The approach links the native fish fauna with components of EI, such as pristineness and nativeness. However, there are some issues with sensitivity to sampling method and the potentially confounding effects of natural environmental gradients (e.g. geology). The dominance of diadromous fish species in New Zealand also means that communities naturally change with distance from the coast and, thus, accessibility to the coast must be accounted for when comparisons are made between sites. Although sampling of the fish community using electric fishing is reasonably cost effective, diadromy presents some challenges for sampling strategies, which must take into account the fact that both the community composition and population structure will vary over seasons. Calibration of the fish IBI data to reference condition is still a relatively unexplored area, and there is some indication that communities in pristine headwater portions of catchments could be affected by degradation further downstream (Joy et al. 2000).

PERIPHYTON BIOMASS AND COMPOSITION

Periphyton biomass and composition is a well-established metric, and has been used to assess the effects of eutrophication (high nutrient concentration) on rivers in New Zealand (Biggs 2000b; Quinn & Raaphorst 2009). Guidelines for acceptable levels of periphyton cover provide a clear relationship to pristineness, and are widely used by regional management authorities to assess the effects of nutrient enrichment (Biggs 2000a). These standards may also be used to determine the nutrient status of rivers, but must take into account that some streams do experience natural proliferations of periphyton during some periods of the year. There are well-established protocols for the sampling of periphyton in the Stream Periphyton Monitoring Manual (Biggs & Kilroy 2000), and sampling is relatively simple and cost effective. However, a disadvantage is that periphyton biomass can be highly variable through time, so sampling frequency is an important consideration for monitoring. In addition, there can be complex interactions between periphyton composition and biomass, hydrology (flood history/frequency) and grazing pressure (Biggs 2000b).

BIOLOGICAL INDICATORS OF CONNECTEDNESS

Aquatic ecosystems are characterised by strong linkages laterally, vertically and longitudinally. Many of the anthropogenic pressures on freshwaters alter the natural connectedness in one or more of these dimensions. These include riparian connections (e.g. Sanzone et al. 2003; Baxter et al. 2005), terrestrial insect contributions to fish diet (Edwards & Huryn 1996), riparian bird communities (Feck & Hall 2004), and groundwater–surface water interactions (Boulton 2000). However, research on these in New Zealand is at a very early stage, limiting our ability to incorporate such measures into this review.

Functional indicators

Our operational definition of EI proposes that rivers should function in the same way as they do in unimpacted catchments. The functional components of a river ecosystem relate to rates, patterns and relative importance of different ecosystem processes. River monitoring techniques have traditionally concentrated on the use of structural measurements to infer the health or integrity of river ecosystems, making the assumption that the presence of an appropriate assemblage automatically implies appropriate functioning (Biggs et al. 2000). However, the structural and functional components of river EI are not always directly related (Boulton 1999; Bunn & Davies 2000). Therefore, information on both structure and function is required for the adequate assessment of ecosystems, because stressors may cause changes to structure but not function, to function but not structure, to both simultaneously, or to both but on different time scales (e.g. Matthews et al. 1982; Gessner & Chauvet 2002). There is growing awareness of the need to include ecosystem process measures in assessments of river EI, since they provide an alternative, complementary and sometimes independent appraisal (Gessner & Chauvet 2002; Young et al. 2008). The scoring of separate functional indicators of riverine pristineness against the seven assessment criteria is included in Table 3. A brief discussion of some of the key points for each of the indicators is presented in the following sections.

TABLE 3. EVALUATION OF FUNCTIONAL PRISTINENESS METRICS FOR RIVERS AGAINST A SET OF ASSESSMENT CRITERIA. DETAILS OF THE ASSESSMENT CRITERIA ARE OUTLINED IN SECTION 4.5.

ASSESSMENT	METAB-	DECOMPOSITION			LEAF/STICK	NUTRIENT	Δ^{15}N[a]	SECOND.	BOD
CRITERIA	OLISM	LEAF	WOOD	COTTON	RETENTION	UPTAKE		PROD.	
Ease of sampling and analysis	Moderate (2)	Moderate (2)	Easy–mod (3)	Easy–mod (3)	Moderate (2)	Difficult (1)	Moderate (2)	Difficult (1)	Moderate (2)
Potential geographic coverage	Wide (3)	Wide (3)	Wide (3)	Wide (3)	Wide (3)	Wide (3)	Wide (3)	Wide (3)	Wide (3)
Relation to pristineness	Moderate (2)	Unknown (1)	Unknown (1)	Unknown (1)	Unknown (1)	Unknown (1)	Unknown (1)	Unknown (1)	Moderate (2)
Sensitivity to pressure gradients	High[b] (3)	Low[c] (1)	Moderate[d] (2)	Moderate[d] (2)	High[e] (3)	Low[e] (1)	High[f] (3)	Low[e] (1)	Low[d] (1)
Normalisation to to reference condition	Moderate (2)	Difficult (1)	Difficult (1)	Difficult (1)	Difficult (1)	Difficult (1)	Moderate (2)	Difficult (1)	Moderate (2)
Temporal variability	High (1)	Moderate (2)	Low (3)	High (1)	Moderate (2)	High (1)	Moderate (2)	Moderate (2)	High (1)
Use in New Zealand	Occasional (2)	Rare (1)	Occasional (2)	Occasional (2)	Rare (1)	Rare (1)	Rare (1)	Rare (1)	Occasional (2)
Use in other countries	Occasional (2)	Occasional (2)	Occasional (2)	Occasional (2)	Rare (1)	Rare (1)	Occasional (2)	Rare (1)	Common (3)
Rank [points]	1 [17]	7 [13]	1 [17]	5 [15]	6 [14]	9 [10]	4 [16]	8 [11]	1 [17]

[a] Of consumer.

[b] Non-linear.

[c] Probably non-linear.

[d] Usually non-linear.

[e] Unknown.

[f] Linear.

RIVER METABOLISM

Measurements of river metabolism indicate how much organic matter (the energy currency of food webs) is produced and consumed in rivers. River metabolism incorporates two components: gross primary productivity (P), which measures photosynthetic rates of algae and other aquatic plants; and respiration (R), which measures the rates of respiration of all living organisms, including fish, invertebrates, algae, aquatic plants and microbes. The ratio of these two parameters (P:R) is very informative, providing information on the relative importance of the two key sources of energy that fuel river ecosystems (e.g. algae v. terrestrial organic matter). If carbon fixation through photosynthesis equals or exceeds respiration, there is likely to be sufficient organic matter produced within the system to support the food chain. In contrast, if respiration greatly exceeds carbon fixation, organic matter from upstream or the surrounding catchment maintains the system. Metabolism can be estimated through either open- or closed-system methods. Open-system methods measure natural changes in oxygen concentration within rivers, while closed-system methods measure oxygen concentration changes with an air-tight chamber that encloses part of the river. Open-system methods have the advantage that they include the whole river ecosystem and integrate the processes occurring in all the different habitats present. Open-system field measurements are also relatively simple to make, but do require an estimate of the amount of oxygen diffusing between the atmosphere and the water (reaeration). This can easily be estimated in most rivers and streams; however, more complicated techniques are required in small, very turbulent streams with low primary productivity (Marzolf et al. 1994; Young & Huryn 1998, 1999). Closed cell systems enable metabolism to be measured in separate components of an ecosystem, but chamber measurements require a large amount of equipment for effective replication. Disadvantages of chamber methods include ‘chamber effects’ such as artificially high water temperatures, the induction of nutrient limitation and the disturbance of substrates during deployment, all of which limit the realism of the measurements (Young et al. 2008).

There is a growing amount of information available on how river metabolism measurements may respond to different pressure gradients (Young & Huryn 1996; Mulholland et al. 2001; Fellows et al. 2006a; Young et al. 2008), although it has been found that the responses may be non-linear, making the calibration of data to reference condition more difficult (Young & Collier 2009). Metabolism measurements also vary over time, with GPP particularly responsive to the amount of sunlight available and the recent flooding history of the site (Young & Huryn 1996; Acuña et al. 2004). Therefore, to obtain a complete picture of the energy balance of a site, multiple seasonal measurements of river metabolism would be required. Guidelines on how to interpret river metabolism results have recently been published (Young et al. 2008).

LEAF DECOMPOSITION RATES

Rates of organic matter decomposition have been considered as a potential indicator of river EI for some time (Young et al. 1994; Bunn et al. 1999; Gessner & Chauvet 2002). The decomposition of organic matter is a key ecosystem process in rivers, particularly in forested headwaters where leaf litter and other terrestrial organic matter is the major source of energy for the ecosystem. Leaves are a 'natural' substrate found regularly in rivers and, therefore, their decomposition reflects a natural process occurring in the system. Leaf decomposition is promoted through the activities of leaf-shredding stream invertebrates, bacteria and fungi, so measurements also provide an indication of the activity of all these groups (Boulton & Boon 1991).

The importance of leaf litter to headwater streams in particular has led to a large number of studies being conducted on the factors controlling leaf decomposition. As a result, it is possible to predict how leaf decomposition will respond to various pressure gradients, although these responses will not necessarily be linear (Webster & Benfield 1986; Young & Collier 2009). Although general patterns of expected changes in decomposition rates in relation to stressors have been reported (Young et al. 2008), the availability of quantitative information is limited in New Zealand, making it difficult to calibrate data to expected reference condition or to accurately predict relationships with particular stressor gradients.

Leaf decomposition rates can be measured in terms of mass loss or toughness loss of leaves deployed in the river for a period of time or, alternatively, through changes in the chemical composition of the leaves (e.g. concentrations of ergosterol, which relates to the biomass of fungal decomposers; Gessner & Chauvet 2002). However, comparisons of decomposition rates between sites or among different collection methods can be complicated due to a number of factors. Firstly, decomposition rates of leaves not only vary widely among plant species but also within the same species, with differences in decay rates expected between leaves grown on different trees of the same species, or even on the same tree (Gessner & Chauvet 2002); for example, new leaves decay more quickly than older, senescent leaves. Secondly, because of the complex structure of leaves, they decompose at a non-linear rate, with rapid initial leaching of soluble compounds, moderate decomposition of the soft leaf flesh and then slow decomposition of the remaining hard veins (Boulton & Boon 1991). This non-linear response means that the length of deployment time is critical when trying to compare decay rates among sites, especially if all leaves are recovered after a single deployment period (Boulton & Boon 1991). Leaf decomposition measurements will also vary according to the different habitats in which the leaves are deployed. Therefore, to allow effective comparisons among sites, leaf packs need to be deployed in consistent habitat types.

WOOD DECOMPOSITION RATES

As wood is commonly found in many river systems, wood decomposition can be an important natural process linked to EI. Measures of wood decomposition rates potentially overcome some of the limitations that surround the measurement of leaf decomposition rates. For example, large numbers of standardised wooden sticks can easily be deployed into a river. Due to their robusteness, they can be deployed for longer periods to provide a longer term, integrated measure of decomposition. However, the long deployment period required to result in measurable mass loss also increases the probability that sticks will be buried or lost during floods.

Existing knowledge of how leaf decomposition rate responds to pressure gradients could potentially be used to also predict how wood decomposition rate may respond. However, the decomposition of wood is likely to reflect different processes than leaf decomposition (e.g. fungi may play a more important role in decomposing wood than bacteria do), which may explain the differences in patterns of leaf and wood decomposition (Tank & Winterbourn 1996; Young & Collier 2009). Wood decomposition rates are unlikely to be linearly related to pressure gradients and depend on the types of habitat in which the wood is deployed (Young et al. 2006). Because of the limited amount of wood decomposition field data collected in New Zealand rivers, calibrating measurements to reference condition and understanding its quantitative relationships with pressure gradients is difficult.

COTTON DECOMPOSITION RATES

The rate of decomposition of cotton cloth provides a specific measure of the cellulose decomposition potential at a site (Young et al. 2006). Cotton decomposition in the water column appears to be primarily mediated via bacterial activity, so is likely to reflect changes in the bacterial community. Cotton is not found naturally in rivers and streams, so cotton decomposition rates should be seen as an index of microbial activity rather than a direct measure of a natural process.

Cotton is highly labile, so significant decomposition can occur over a short period (e.g. 7 days), reducing the likelihood of the material being disturbed while deployed. Cotton cloth is a cheap and standardised material, allowing accurate comparisons among sites. However, like leaf and wood decomposition, the response to pressure gradients is likely to be non-linear. Decomposition rates are best measured using changes in the tensile strength of the material, because measures of mass loss have been shown to be relatively insensitive (Tiegs et al. 2007). Measurement of tensile strength requires specialised equipment (tensometer), but the measurement is relatively quick and cost effective. There are a few studies that have examined cotton decomposition in river systems (Hildrew et al. 1984; Boulton & Quinn 2000; Tiegs et al. 2007; Young & Collier 2009; Clapcott et al. 2010), but the use of cotton as a tool for the assessment of river EI is relatively unexplored.

LEAF/STICK RETENTION RATES

Coarse particulate organic matter (CPOM) can be an important source of energy for some stream and river ecosystems. CPOM must settle or be trapped within the river channel to allow processing to occur. Therefore, CPOM retention rates provide a potentially useful measure of the physical features/conditions that could be related to river EI (Scarsbrook & Townsend 1994; James & Henderson 2005; Dewson et al. 2007; Quinn et al. 2007).

Retention is measured by releasing marked CPOM particles, or analogues, and recording the distance each particle moves downstream. Retention is then reported as the exponential rate of loss of particles in transport with distance downstream. CPOM analogues that have been used include wooden dowels, soaked ginkgo leaves and triangles cut from waterproof paper (Quinn et al. 2007). Retention rates vary between the types of particles used, so a standard particle (or set of particles) is required for inter-site comparisons. This measure is also highly dependent on stream size, channel slope and flow, so alternative measures, such as deposition velocity (Brookshire & Dwire 2003) and stream width- or catchment area-specific retention distances (Quinn et al. 2007), have been proposed to facilitate comparisons among sites.

However, there has been only a limited amount of work conducted to understand how this indicator may relate to specific pressure gradients and how reference condition might be determined for specific rivers. As such, further work would be required for this indicator to be considered a useful measure of EI.

NUTRIENT UPTAKE RATES

Measurement of nutrient uptake and spiralling is potentially an important component of EI assessment, especially if the measure can be related to the importance of a particular site in the processing of nutrient inputs (Alexander et al. 2007). There is a relatively large body of scientific literature on factors affecting nutrient uptake (Dodds et al. 2002) and some of these happen to be other key indicators of EI (e.g. rates of ecosystem metabolism; Hall & Tank 2003; Fellows et al. 2006b). Nutrient uptake rates are closely linked to river size and flow but, as mentioned above for CPOM retention, there are various metrics that have been designed to compensate for these effects and allow comparisons among diverse sites (Earl et al. 2006). Such compensation, however, adds complexity when relating this metric to pressure gradients and calibrating it to expected reference condition. The biggest disadvantage in using nutrient uptake rates is the effort involved in making the measurements. The simplest approach involves releasing a pulse of nutrients into a stream and sampling its passage at several downstream sites, but this approach has been heavily criticised (Earl et al. 2007). Instead, the preferred approach is to make continuous injections of isotope-labelled nutrients (Earl et al. 2007), but this is only possible in small streams and is well beyond the scope of standard river EI assessment. Thus, it has not been used extensively either in New Zealand or internationally for assessing EI.

STABLE ISOTOPE ANALYSIS

As an alternative to direct measures of nutrient cycling, stable isotope analysis has proven an effective tool in evaluating in-stream nitrogen processing, as well as the source of nitrogen entering, and being retained, in-stream (Mayer et al. 2002; Sebilo et al. 2003; Anderson & Cabana 2005). Specifically, changing land use in catchments has been shown to correlate strongly with the δ^{15}N signal of dissolved forms of inorganic nitrogen in water (Mayer et al. 2002; Cole et al. 2006; Voss et al. 2006), aquatic plants (Kaushal et al. 2006; Voss et al. 2006), sediment (Udy & Bunn 2001; Bunting et al. 2007), invertebrates and fish (Fry & Allan 2003; Anderson & Cabana 2005; Udy et al. 2006). While the causal link in these relationships can be difficult to interpret, the δ^{15}N of particulate and/or dissolved nitrogen is likely to be a good indicator of disturbance gradients, such as predicted N-loading, percentage of native vegetation remaining in the catchment or percentage of impervious area in the catchment. Additionally, the δ^{15}N of primary consumers incorporates the nitrogen signal over a longer time period than water or sediments and, as such, may provide a good surrogate measure of nitrogen processing in relation to EI. However, while we know that these indicators may potentially be sensitive to such pressures, there has been a limited amount of work completed in New Zealand to understand the nature of the relationship with EI or to calibrate it to expected reference condition (but see Clapcott et al. 2010). Thus, it has only been used to a limited extent in EI assessment.

SECONDARY PRODUCTIVITY

The productivity of invertebrate and fish populations would be a useful indicator of EI because it combines biomass and growth rates (e.g. Buffagni & Comin 2000). Unfortunately, measurements of invertebrate and fish population productivity are time-consuming and are, therefore, not included in most standard river EI assessments. There is limited information in the literature about how secondary productivity may respond to various pressure gradients, and non-linear responses are likely. This also limits the ability to calibrate measurements of secondary productivity to expected reference condition. For these reasons, imeasurements of secondary productivity are not widely used as an indicator in EI assessment.

BIOLOGICAL OXYGEN DEMAND

The measurement of the biological oxygen demand (BOD) of water samples is the only functional measure that has been widely included in previous river EI assessments (Young et al. 2008). This measure is relevant for sites that are strongly influenced by waste water discharges, where pollutants can stimulate substantial biological oxygen demand in the water. It is routinely measured as part of consent monitoring requirements for discharges to freshwater. However, in most rivers the majority of the oxygen demand occurs on or within the riverbed and so is not accounted for in standard BOD measurements.

Physico-chemical indicators

Physico-chemical indicators are routinely used in relation to the assessment of stream EI, where they may assist in the interpretation of biological indicators. Moreover, they tend to be one of the few groups of indicators for which regional and national standards are set for the protection of both human use and ecological values of freshwaters (e.g. ANZECC 2000). The scoring of separate physico-chemical indicators of riverine pristineness against the seven assessment criteria are included in Table 4 and a brief discussion of some of the key points for each of the indicators follows in the sections below.

WATER CLARITY, TURBIDITY AND SUSPENDED SOLIDS CONCENTRATION

Measurements of water clarity, turbidity and the concentration of suspended solids can indicate the effects of suspended sediment and coloured dissolved organic matter on EI. Small increases in suspended sediment concentrations may influence fish feeding and distribution, while larger increases can result in changes in macrophyte production rates, and in the abrasion of gills and other sensitive organs of fish and invertebrates (e.g. Hicks & Barrier 1996; Granqvist & Mattila 2004). Thus, suspended sediment concentrations and water clarity can be linked with various components of EI and pristineness. These indicators also reflect the likelihood that fine sediment will be deposited on the river bed, along with associated effects on food and habitat quality. These indicators are widely used throughout New Zealand and respond to various stressors in predictable ways. Because there are natural sources of suspended sediments in water, such as glacial flour and dissolved organic matter (Kirk 1983), this can complicate calibration to reference condition. Water clarity and turbidity are closely related and complement each other well because water clarity measures are highly sensitive and turbidity measures give the most accurate readings in waters of low clarity.

TABLE 4. EVALUATION OF PHYSICO-CHEMICAL PRISTINENESS METRICS FOR RIVERS AGAINST A SET OF ASSESSMENT CRITERIA. DETAILS OF THE ASSESSMENT CRITERIA ARE OUTLINED IN SECTION 4.5.

ASSESSMENT CRITERIA	WATER CLARITY	WATER TEMP	DISSOLVED OXYGEN	CONCENTRATION	
				NUTRIENTS	CONTAMINANTS
Ease of sampling and analysis	Easy (3)	Easy (3)	Easy (3)	Easy (3)	Moderate (2)
Potential geographic coverage	Wide (3)	Wide (3)	Wide (3)	Wide (3)	Wide (3)
Relation to pristineness	Strong (3)	Moderate (2)	Strong (3)	Moderate (2)	Strong (3)
Sensitivity to pressure gradients	High[a] (3)	High[a] (3)	High[a] (3)	High[a] (3)	High[a] (3)
Normalisation to reference condition	Moderate (2)	Easy (3)	Moderate (2)	Moderate (2)	Moderate (2)
Temporal variability	Moderate (2)	High (1)	High (1)	Moderate (2)	High (1)
Use in New Zealand	Common (3)	Common (3)	Common (3)	Common (3)	Common (3)
Use in other countries	Common (3)	Common (3)	Common (3)	Common (3)	Common (3)
Rank [points]	1 [22]	2[21]	2 [21]	2 [21]	5 [20]

[a] Usually linear.

WATER TEMPERATURE

Water temperature is an important parameter affecting ecosystem processes, growth rates and survival of all river organisms. Because it is important for the interpretation of so many other ecological indicators (e.g. all functional processes, dissolved oxygen), it is regularly measured in monitoring programmes. Temperature preferences and thresholds are known for some species, particularly fish and plant species, making interpretation of biological data easier. Spot measurements of temperature are not particularly useful because they are so highly variable; however, inexpensive data loggers are now readily available, allowing comprehensive monitoring of thermal conditions, both spatially and temporally. The response of water temperatures to pressure gradients, such as canopy removal, is predictable and potentially linear, thus making calibration of the data to reference condition reasonably easy (Rutherford et al. 1999). Water temperature measurements are also directly relevant to the issue of global warming.

DISSOLVED OXYGEN

Dissolved oxygen (DO) is a critical parameter affecting the abundance and diversity of organisms within a river ecosystem. Similar to temperature, DO is important for the interpretation of many other ecological indicators (e.g. fish communities, metabolism); thus, it is regularly measured in monitoring programmes. However, oxygen concentrations vary widely on a daily basis and, therefore, single measurements are of limited value. Reliable oxygen loggers are now available, allowing continuous monitoring of oxygen concentrations and accurate assessments of daily oxygen minima, but these loggers are considerably more expensive than temperature loggers. Measurements of daily changes in dissolved oxygen are useful for determining rates of primary production and ecosystem respiration in river systems, if collected regularly over at least a 24-hour period.

NUTRIENT CONCENTRATIONS

Nutrient concentrations give an important indication of the trophic status of a site and provide guidance on the likelihood of algal proliferations, which can affect other constituents of the food web (Biggs et al 2000). Thus, there is a strong linkage with pristineness and other components of EI. Nutrients are monitored extensively, with most freshwater monitoring programmes in New Zealand now including regular nutrient sampling, so there is a large amount of existing data on various nutrients. The effects of pressures such as agriculture, forestry and urbanisation on nutrient concentrations has been the focus of a great deal of scientific study both nationally and internationally, and nutrient concentrations can be reasonably accurately predicted using land use models (Elliot et al. 2005; Alexander et al. 2007). There are well-established relationships with these pressures, and determination of reference condition is possible for many environment types. As with other physico-chemical indicators, there is moderate temporal variability in concentrations of most nutrients, and this is particularly related to flow variations. Therefore, sampling frequency should account for this variability.

CONTAMINANT CONCENTRATIONS

Contaminants (e.g. faecal bacteria, heavy metals) have a major impact on diversity and ecosystem processes in situations where urban, industrial or mining developments lead to substantial inputs of toxic contaminants to streams. The presence and concentration of faecal bacteria have important implications for human and stock use of waterways, and could also be seen as a sensitive indicator of land-use impacts on EI. Contaminant sampling is often included in specific council monitoring programmes, so there is a substantial amount of existing data and the effects of different pressures on contaminant concentrations are well understood. One disadvantage with measuring contaminant concentrations is the wide variety of compounds potentially involved, making an assessment of all possible contaminants very expensive.

Cultural health indicators

Tipa & Teirney (2003) quantified Māori freshwater values and combined these into an index of 'stream health' called a 'cultural health index' (CHI). The main components of CHI are:

1. Was the site traditionally used for food gathering and would it be useful in the future?
2. What was the past, present and likely future state of wild food (mahinga kai) resources at the site?
3. What is the health of the stream at the site as evaluated by selected members of the Māori community?

The stream health component (question 3) involves scoring the stream site according to a range of factors, including the condition of the riverbanks, flow characteristics, the sound of flowing water, state of riparian vegetation, the uses made of the river banks, catchment land use, the smell of the water, substrate type on the stream bed, whether the fish present are safe to eat, whether the water is safe to drink, and the uses of the river—takes and discharges.

An evaluation of CHI against the seven assessment criteria is included in Table 5. Whilst CHI assesses physical and biological integrity, it clearly focuses on the value of rivers for human services (i.e. food provision) and as such has limited value in the context of relating it to the different components of EI. However, CHI has been shown to correlate significantly with other measures of stream pristineness, such as the Stark (1985) macroinvertebrate community index (Townsend et al. 2004), suggesting that CHI captures stream integrity or 'health' values in a similar manner to other indicators. There is relatively little known about the quantitative nature of the relationship between CHI and human pressure gradients, and it would be expected that collection of traditional knowledge would be required to inform this relationship for use in management decisions.

TABLE 5. EVALUATION OF CULTURAL HEALTH INDICES FOR RIVERS AGAINST A SET OF ASSESSMENT CRITERIA. DETAILS OF THE ASSESSMENT CRITERIA ARE OUTLINED IN SECTION 4.5.

ASSESSMENT CRITERIA	TRADITIONAL/ FUTURE USE	MAHINGA KAI	CULTURAL STREAM HEALTH MEASURE
Ease of sampling and analysis	Easy[a] (3)	Easy[a] (3)	Easy[a] (3)
Potential geographic coverage	Wide (3)	Wide (3)	Wide (3)
Relation to EI components	Low (1)	Moderate (2)	High (3)
Sensitivity to pressure gradients	Unknown (1)	Moderate[b] (2)	Moderate[b] (2)
Normalisation to reference condition	Easy (3)	Difficult (1)	Difficult (1)
Temporal variability	Low (3)	Moderate (2)	Moderate (2)
Use in New Zealand	Occasional (2)	Occasional (2)	Occasional (2)
Use in other countries	Rare (1)	Rare (1)	Rare (1)
Total Points	1 [17]	3 [16]	1 [17]

[a] But need specialised expertise.

[b] But of unknown nature.

5.1.3 Diversity

Diversity is a biological measure intuitively associated with EI. Diversity indices can be applied to fish, invertebrates, macrophytes and periphyton, although more data are currently available for fish and invertebrates than for other groups. However, the quantitative assessment of diversity is not always straightforward (Magurran 2004). Diversity indices are subject to the issues of taxonomic resolution discussed in section 4.3.1. The scoring of separate diversity indicators against the seven assessment criteria is included in Table 6, and a brief discussion of some of the key points for each of the indicators follows in the sections below.

TABLE 6. EVALUATION OF DIVERSITY INDICES FOR RIVERS AGAINST A SET OF ASSESSMENT CRITERIA. DETAILS OF THE ASSESSMENT CRITERIA ARE OUTLINED IN SECTION 4.5.

ASSESSMENT CRITERIA	TAXONOMIC RICHNESS	DIVERSITY INDICES	TAXONOMIC RICHNESS		TAXONOMIC D&D[a]	ABIOTIC STRUCTURE
			RAREFIED	O/E		
Ease of sampling and analysis	Easy (3)	Easy (3)	Easy (3)	Moderate (2)	Easy (3)	Easy (3)
Potential geographic coverage	Wide (3)	Wide (3)	Wide (3)	Wide (3)	Wide (3)	Wide (3)
Relation to diversity	Strong (3)	Strong (3)	Strong (3)	Strong (3)	Strong (3)	Weak (1)
Sensitivity to pressure gradients	Moderate[b] (2)	Moderate[b] (2)	Moderate[c] (2)	Strong[d] (3)	Weak[e] (1)	Moderate[c] (2)
Normalisation to reference condition	Moderate- (2)	Moderate (2)	Difficult (1)	Moderate (2)	Difficult (1)	Moderate (2)
Temporal variability	Moderate (2)	Moderate (2)	Low (3)	Low (3)	Low (3)	Low (3)
Use in New Zealand	Common (3)	Common (3)	Rare (1)	Rare (1)	Common (3)	Rare (1)
Use in other countries	Common (3)	Common (3)	Rare (1)	Common (3)	Common (3)	Common (3)
Rank [points]	1 [21]	1 [21]	6 [17]	3 [20]	3 [20]	5 [18]

[a] Taxonomic diversity and distinctiveness.

[b] Unimodal.

[c] Probably linear.

[d] Linear.

[e] Unknown.

Taxonomic richness and diversity indices

Taxonomic richness is the simplest and easiest indicator to measure of all the diversity indices, and is widely reported for benthic invertebrate and fish communities. It intuitively links with overall site diversity; however, it is also more sensitive to sampling effort than some other diversity indices, such as Simpson's index and Fisher's α index (Ludwig & Reynolds 1988; Magurran 2004; Moore & Suthers 2005). The usefulness of taxonomic richness for interpreting EI may be compromised by its tendency to exhibit unimodal relationships with some pressure gradients and by its tendency to be strongly correlated with natural environmental gradients, which may confound the process of determining reference condition. Taxonomic diversity comprises two components: species richness (the number of species) and species evenness (how individuals are distributed amongst those species). Thus, measures of diversity incorporating evenness, such as Shannon or Simpson diversity indices, may relate more strongly to pressure gradients than simple taxonomic richness (Krebs 1999; Magurran 2004).

Rarified taxonomic richness

Rarified taxonomic richness (sometimes called species density) is a measure of species diversity that corrects for the number of individuals collected (Magurran 2004). It has been argued that it is a useful index of diversity because it is less sensitive to variation in sampling effort and is independent of abundance, which may be an important consideration in assessing EI. However, it does suffer from many of the same disadvantages as simple measures of taxonomic richness. In our experience, it provides a relatively poor indicator of environmental gradients such as stability and forest cover in comparison to the more classical diversity measures such as taxonomic richness (Gotelli & Colwell 2001).

Observed/expected taxa richness

Observed/expected taxa richness is the ratio of the number of taxa collected to the number of taxa that would be present in that type of river in the absence of human pressures. This index can be calculated based on a predictive modelling approach, as discussed in section 4.1 (Joy & Death 2003b, 2004b; Death & Zimmermann 2005), or by direct measurement at both impacted and reference sites (e.g. Wright et al. 2000). The former approach has been used successfully as a conservation tool in English waterways (Plafkin et al. 1989) and both approaches are generally applied as biological indices of water quality elsewhere in the world. However, in contrast to more simple diversity indices, it requires intensive sampling effort at a number of reference sites. This approach retains species identity in the assessment of richness. It can also account for the effect of confounding natural environmental gradients, which can affect diversity measures, and is thus more likely to link directly with pressure gradients. In New Zealand, Death & Collier (2009) found a good relationship between a crudely constructed O/E richness index and the percentage of Waikato catchments in native forest, and Joy & Death (2003b) developed a predictive model for invertebrates for the Manawatu-Whanganui region. However, the predictive modelling approach has not been explicitly used to generate indices of diversity in New Zealand rivers.

Taxonomic diversity and distinctness

Indices of taxonomic diversity and distinctness, as developed by Warwick & Clarke (1995, 1998), allow both the phylogenetic relatedness and the richness of the biota to be considered so that, for example, a stream with three species of chironomids would be considered less diverse than a stream with one species of mayfly, stonefly and chironomid, even though both streams have three species. These indices have the advantage that some of the distinctness measures are independent of sample size/effort (Warwick & Clarke 1998; Death & Collier 2009) and so may be easier to statistically relate to stressor gradients. Initial examination of the response of this indicator to a gradient of native vegetation cover has proved promising (Death & Collier 2009), although the pattern found was opposite to that of marine environments, where taxonomic distinctiveness has been shown to decline with increasing stress from oil exploration activities (Clarke & Warwick 1994). Although there has been only a limited amount of work carried out relating this variable to pressure gradients and calibrating it to reference condition, it may contribute useful information to assessments of EI.

Abiotic structure and ecological processes

The previous measures of biological diversity discussed are often used to infer the presence of a diverse range of habitats and ecological processes. However, it may be useful to assess the diversity of abiotic structure directly (Muhar & Jungwirth 1998; Petts 2000). Worldwide, habitat assessment has been an important component of assessing the EI of rivers (e.g. the Index of Stream Condition (Ladson et al. 1999), River Habitat Audit Procedure (Anderson 1993), River Styles (Brierley & Fryirs 2000), and AusRivAS (Parsons et al. 2004)). Recently, habitat assessment protocols have been developed for wadeable rivers and streams in New Zealand (Harding et al. 2009). However, to date, information collected using these protocols has not been specifically tested in relation to known pressure gradients. Although the method generates a great deal of quantitative data on habitat attributes, it does not specifically generate a habitat quality score, so statistically relating it to pressure gradients and reference condition may be complicated.

5.1.4 Resilience

Resilience is a normative concept, the definition and measurement of which has been debated extensively in the ecological literature (DeAngelis & Waterhouse 1987; Warwick & Clarke 1995). In river systems, ecological resilience is generally defined as the return to a pre-existing condition following a perturbation; it incorporates concepts of recovery and stability, and is often discussed in terms of persistence (Gunderson 2000; Lake 2000; Ives 2005). This contrasts somewhat with the lakes view (see section 5.2.4). The resilience of macroinvertebrates in particular to flood events is a fundamental characteristic of river systems worldwide (Boulton & Lake 1992; Lake 2000); however, we still know very little about many of the mechanisms and drivers of this resilience or how to measure them in natural situations (Gunderson 2000). There have been some attempts at measuring resilience in rivers and these are discussed below, but it must be noted that knowledge about the resilience of river biota to anthropogenic disturbances remains limited. Furthermore, to measure resilience requires knowledge of temporal variability, which in turn requires long-term datasets for each site. Consequently, resilience is more often inferred than measured (but see Uehlinger 2000, 2006; Scarsbrook 2002; Collier 2008). The scoring of separate resilience indicators against the seven assessment criteria is included in Table 7. A brief discussion of some of the key points for each of the indicators is discussed in the following sections.

Presence/absence

The presence of species has been used to measure resilience and recovery following disturbance in macroinvertebrate (Bradt et al. 1999; Melo et al. 2003; Collier 2008), fish (Martin-Smith et al. 1999; Bond & Lake 2005) and algal communities (Peterson & Stevenson 1992) in rivers. These studies suggest that the presence of a 'reference' condition community infers both resistance and resilience to future disturbance, although there is also recognition that reference communities can be highly temporally variable, which in itself may infer resilience (Metzeling et al. 2002; Collier 2008).

Taxonomic traits

The examination of specific traits may offer some insight into the resilience of specific species to disturbance. As discussed in section 5.1.2 ('Taxonomic traits'), this is a relatively new area of research, but studies have inferred a link between life-history characteristics and community resilience (Townsend et al. 1997b; Doledec et al. 2006). For example, Death (2003) showed that flow disturbance had greater effects on invertebrates that were reliant on allochthonous (organic material generated outside the stream) energy sources; therefore, the relative abundance of certain functional feeding groups may give some insight into flow variability and resilience. Similarly, Doledec et al. (2006) showed that intense agricultural development resulted in invertebrates with traits associated with population resilience (short generation time, asexual reproduction, protection of eggs) becoming more common. Dewson et al. (2007) has also shown that taxa with specific trophic traits are more sensitive to flow reduction than others. However, generally less is known about how the composition of taxonomic traits of various communities responds to pressure gradients, and there is relatively little known about reference condition for these indicators.

TABLE 7. EVALUATION OF RESILIENCE INDICES FOR RIVERS AGAINST A SET OF ASSESSMENT CRITERIA. DETAILS OF THE ASSESSMENT CRITERIA ARE OUTLINED IN SECTION 4.5.

ASSESSMENT CRITERIA	INDICATOR SPECIES PRESENCE/ABSENCE	SPECIES TRAITS	ECOSYSTEM FUNCTION
Ease of sampling and analysis	Easy (3)	Easy (3)	Moderate (2)
Potential geographic coverage	Moderate (2)	Wide (3)	Wide (3)
Relation to resilience	Low (1)	Low (1)	Moderate (2)
Sensitivity to pressure gradients	High (3)	Unknown (1)	High[a] (3)
Normalisation to reference condition	Difficult (1)	Difficult (1)	Moderate (2)
Temporal variability	High (1)	Moderate (2)	Moderate (2)
Use in New Zealand	Common (3)	Occasional (2)	Occasional (2)
Use in other countries	Common (3)	Occasional (2)	Common (3)
Rank [points]	2 [17]	3 [15]	1 [19]

[a] Probably non-linear.

River function

Several studies have used measures of river metabolism and organic matter processing (see section 5.1.2 'Functional indicators') to infer resilience by focussing on ecosystem efficiency (Uehlinger & Naegeli 1998; Uehlinger 2000; Acuña et al. 2007). For example, Acuña et al. (2007) showed that streams were functionally more resilient to flood disturbances during summer because they were efficiently processing carbon entering the stream at this time. Ecosystem functions that occur in the hyporheic zone (e.g. hydrological exchange) have also been suggested as important indicators of river resilience (Valett et al. 1994), as they characterise the condition of an important refuge habitat. To assess functional resilience, knowledge of the pre-disturbance state (or of the functioning of reference sites) is also required, necessitating sampling of some reference sites. Some functional indices, such as the balance between rates of P and R, could also directly imply resilience, with a balanced system ($P/R = 1$) being more resilient than one relying entirely on external inputs of energy ($P/R << 1$). Thus, the use of functional attributes to infer ecosystem resilience has some attraction, and these are probably the best documented indicators related to resilience.

5.2 LAKES

Karr's (1996) definition of EI (see section 2.2) has influenced the development of the majority of current classification schemes for lakes that attempt in one form or another to measure EI (US EPA 1998; European Union 2000; Murphy et al. 2002; Solimni et al. 2006). The term 'integrity' has been criticised because it implies a definitive state rather than a gradient of states, with the ecosystem either having integrity or not (Westra et al. 2000). In response to this, the terms 'ecological condition' or 'ecological status' have been adopted by some freshwater lake classification schemes, notably in the European Water Framework Directive (WFD), where the ecological status of surface waters is defined as '… an expression of the quality of the structure and functioning of aquatic ecosystems associated with surface waters …' (European Union 2000). WFD uses primarily biological indicators to assign water bodies, including lakes, to one of five ecological classes ranging from high to low quality (European Union 2000). Our definition of EI also implies a gradient of condition or status.

Difficulties persist in how to define and determine the EI of specific lakes because integrity has often been defined in terms of deviation from a reference condition (Søndergaard et al. 2005), and it has been argued that the establishment of a realistic and appropriate reference condition is essential in lake ecological monitoring programmes (Moss et al. 1996; Wallin & Solheim 2005). Various approaches have been used to infer lake reference conditions, including palaeolimnological analyses (Laird & Cumming 2001), historical data combined with a modelling approach (Nielsen et al. 2003), survey data from existing reference sites, and expert judgement (US EPA 1998; Wallin & Solheim 2005).

In the reference condition guidance (REFCOND) produced by the European WFD, it was ultimately concluded that, for practicality's sake, reference conditions of lakes need not necessarily equate to totally undisturbed, pristine conditions, but instead could include the condition of very minor disturbance, as long as there is no, or minimal, ecological effect (Anon. 2003). This is a practical compromise because of the difficulty in obtaining appropriate, pristine reference lakes (Leira et al. 2006). For example, historical and current datasets may be limited or of little relevance (Wallin & Solheim 2005), and palaeolimnological datasets with transfer functions capable of describing pre-human lake conditions are not yet common in New Zealand (but see Reid 2005; Cochrane et al. 2007).

It has been suggested that the use of data from limnological surveys is the most transparent, practical and defensible methodology for establishing reference conditions (US EPA 1998; Wallin & Solheim 2005). This approach usually takes one of two forms:

1. When a sufficient number of non- (or minimally) impacted lakes exist, a reference lake scheme approach can be used, whereby such lakes can define reference conditions
2. If there are insufficient minimally impacted reference lakes available, a lake population distribution approach can be used, whereby reference conditions are selected from the best quartile of the entire population of lakes.

Karr et al. (1986) developed the latter approach for the Index of Biotic Integrity (IBI) and it has been suggested that this approach may be useful for highly impacted areas, such as lowland agricultural areas, where unimpacted reference sites do not exist (US EPA 1998; Wallin & Solheim 2005). However, this approach is relativistic and substantially impacted lakes could be attributed with a high EI.

Reference conditions must reflect not only individual lake types but also regional variation, because lakes in different regions may respond differently to anthropogenic pressure (US EPA 1998; Søndergaard et al. 2005; Wallin & Solheim 2005). For example, lakes in different ecoregions of the USA have significantly different levels of total phosphorus and chlorophyll because of differences in soils, vegetation, land form and land use (US EPA 1998). Similarly, lakes of the Central Volcanic Plateau of New Zealand tend to have higher natural inputs of phosphorus than many other New Zealand lakes due to the higher phosphorus content of the underlying volcanic geology (Burns et al. 1997).

The use of ecological classification systems to define lake types and regions may assist in establishing appropriate reference conditions. However, type- and region-specific assessments of reference condition may be problematic because individual lakes may have unique communities and system dynamics influenced by lake-specific factors or conditions, which are not accounted for in the reference lake approach. Long hydraulic residence times in lakes dampens variability and strengthens biological interactions, which may promote stochastic dynamics driven by species interactions. Such systems with strong, specific biological interactions may respond uniquely to pressure gradients.

The above arguments indicate that the approach of establishing lake EI via the normalisation of current New Zealand lake conditions to appropriate reference conditions is problematic.

The literature on EI in lakes reflects the fact that the concept of EI, and indicators for assessing it, are relatively under-developed in lakes compared with rivers and streams. Below, we present a number of potential indicators for measuring each of the four components of EI. Many of the indicators, excluding those relating to the measurement of nativeness, do not explicitly require a knowledge of reference conditions and have been previously used by researchers as indicators of the degree of degradation of aquatic ecosystems. Although the list of indicators we discuss is not exhaustive, our aim is to present and discuss some common and cost-effective indicators that may be useful for the determination of EI in New Zealand lakes.

5.2.1 Nativeness

Nativeness refers to the taxonomic makeup of the ecosystem as it existed in a reference state (Solheim 2005) and, therefore, high nativeness equates to high ecological condition or integrity (Harig & Bain 1998; Clayton & Edwards 2006b). New Zealand's aquatic macrophyte, fish and zooplankton communities have been impacted by introduced species, whereas the spread and impacts of introduced benthic macroinvertebrates in New Zealand are less well understood at present (Closs et al. 2004; Duggan et al. 2006). The scoring of separate nativeness indicators for lakes against the seven assessment criteria is included in Table 8 and a brief discussion of some of the key points for each of the indicators follows below.

Macrophyte community composition

Several European and North American schemes have used macrophyte community composition to monitor lake condition (e.g. Palmer et al. 1992; Melzer 1999; Nichols et al. 2000). In New Zealand, LakeSPI or 'Lake submerged plant indicators' has been developed as a management tool, integrating three separate indices: a native condition index (extent and diversity of native macrophytes), an invasive condition index (extent and impact of invasive macrophytes), and an integrated LakeSPI Index, derived from components of the previous two indices. Macrophyte community composition has been linked to various pressures in New Zealand lakes (Clayton & Edwards 2006a, b). Advantages of using macrophyte species composition as a measure of nativeness include their year-round presence, their

TABLE 8. EVALUATION OF NATIVENESS INDICES FOR LAKES AGAINST A SET OF ASSESSMENT CRITERIA. DETAILS OF THE ASSESSMENT CRITERIA ARE OUTLINED IN SECTION 4.5.

ASSESSMENT CRITERIA	MACROPHYTES	FISH	ZOOPLANKTON
Ease of sampling and analysis	Moderate (2)	Easy (3)	Moderate (2)
Potential geographic coverage	Moderate (2)	Wide (3)	Wide (3)
Relation to nativeness	Strong (3)	Strong (3)	Moderate (2)
Sensitivity to pressure gradients	High[a] (3)	Intermediate[b] (2)	Intermediate[c] (2)
Calibration to reference condition	Moderate (2)	Easy–Moderate[d] (3)	Difficult (1)
Temporal variability	Low (3)	Low (3)	High (1)
Use in New Zealand	Common (3)	Common (3)	Rare (1)
Use in other countries	Common (3)	Common (3)	Occasional (2)
Rank [points]	2 [21]	1 [23]	3 [14]

[a] Unimodal.

[b] Probably not unimodal.

[c] Poorly known.

[d] Probably non-linear.

relative longevity (allowing an integration of environmental impacts over time), and the moderate costs associated with macrophyte sampling, monitoring and identification. The nativeness of a lake can be assessed as the ratio of the number of native macrophyte species present to the number of macrophyte species expected to have inhabited the lake prior to human influence (observed/expected or O/E). However, this measure of nativeness requires robust predictions of the reference macrophyte community specific to each lake or lake type (Solheim 2005).

Invasive macrophytes are defined as a key ecosystem pressure in the LakeSPI system (Clayton & Edwards 2006a). Invasive condition assesses the invasiveness and relative abundances of invasive macrophytes, as well as their depth and coverage of the lake bed or shoreline. The proportion of the lake shoreline colonised by native or invasive macrophytes was selected by Moss (2007) as one of the most robust variables for assessing lake ecological condition. The major disadvantage of this system is that it is, by definition, indicative of only one pressure (plant invasion), although invasive species composition could in some circumstances also act as an indicator of oligotrophic or eutrophic conditions (e.g. *Lagarosiphon major* tends to colonise the former, while *Ceratophyllum demersum* tends to colonise the latter). In many freshwater lakes in New Zealand, Europe and North America, the geographical distribution of invasive macrophytes is spreading, indicating that many lakes have yet to be exposed to the colonists. Thus, mechanisms of spread, as well as biogeography, must be considered when using EI indicators that are based on the presence of exotic taxa.

Native fish species composition

Some advantages of using fish species composition as an indicator of nativeness include the relatively low number of species that must be considered, the relatively simple taxonomy involved and the high sensitivity of some fish taxa to human pressures (Søndergaard et al. 2005). Measurements of fish assemblage composition and abundance have been incorporated into the Index of Biotic Integrity (IBI) to assess pressures on river systems (Karr et al. 1986; Karr 1991) and attempts have been made to apply similar systems to lakes (Jeppesen et al. 2003). However, the high variability in lake morphology, bottom type and sampling efficiency (US EPA 1998) render the method costly. The sampling effort required to obtain representative samples of fish assemblages in lakes is high because thorough surveys usually necessitate using a combination of sampling methods, such as gillnetting, seining, trawling and electrofishing (Wanzenböck et al. 2002). Furthermore, some of these methods are unable to catch fish quantitatively, making it difficult to obtain useful abundance data. Carol et al. (2006) found that for fish assemblages with low richness, catch per unit effort (CPUE) and species composition were better indicators of pressures, including eutrophication, than commonly used species richness and diversity indices. Yonekura et al. (2004) found that the CPUE data showed clear impacts of two invasive species in 14 Japanese farm ponds, with negative relationships between the total abundances of native and exotic fish. However, both of these studies cautioned against comparing abundances among species when standard fish sampling methods are used because most methods have different capture efficiencies for different species. Fish species can exhibit seasonal and diel changes in abundance and behaviour, and such temporal variability should be considered when planning fish sampling programmes. However, if carefully quantified, the relative contribution of native fish species to the fish community is a useful indicator of nativeness that can be applied directly and quantitatively to lakes regardless of their geographical locations.

Zooplankton

New Zealand is now home to at least three exotic crustacean zooplankters (C.W. Burns, University of Otago, unpubl. data). In addition, the exotic freshwater jellyfish *Craspedacusta sowerbyi* has been observed from numerous lakes (Boothroyd et al. 2004; M. Schallenberg, unpubl. data). While zooplankton are easily quantitatively sampled, the appearance of species in a lake and their abundances can be episodic, so that the absence of zooplankter species from a lake should be confirmed by repeated sampling over an annual cycle. Furthermore, it has been suggested that a number of native taxa may represent species complexes, indicating that taxonomic differentiation based on morphometry can be problematic. Thus, genetic taxonomic differentiation is increasingly used to identify certain species. Biogeographic patterns structure the distributions of some native zooplankters, as was shown by Jamieson (1998) for native copepod species. Thus, biogeography and dispersal should also be taken into account when interpreting the geographical distributions of some zooplankton taxa and when inferring potential reference conditions related to zooplankton abundance and distribution.

As exotic zooplankters appear to be relatively new invaders to New Zealand lakes, little is known about relationships between these new species and New Zealand ecological pressure gradients. However, anecdotal evidence suggests exotic *Daphnia* species may achieve higher densities than the native *D. carinata* and, thus, may improve water clarity in some lakes (M. Schallenberg, unpubl. data).

5.2.2 Pristineness

Pristineness may refer to various key components and processes of the ecosystem. Therefore, we have included structural, functional and physio-chemical elements for this component of EI. We discuss potentially useful indicators of New Zealand lake pristineness under the three elements below.

Structural indicators

Structural indicators relate either to taxonomic community structure or to the abundance and/or distribution of taxa within and among lakes. Here we will emphasise communities that are not covered under the nativeness component of EI. The scoring of separate structural indicators of pristineness for lakes against the seven assessment criteria is included in Table 9 and a brief discussion of some of the key points for each of the indicators follows below.

TABLE 9. EVALUATION OF STRUCTURAL INDICES OF PRISTINENESS FOR LAKES AGAINST A SET OF ASSESSMENT CRITERIA. DETAILS OF THE ASSESSMENT CRITERIA ARE OUTLINED IN SECTION 4.5.

ASSESSMENT CRITERIA	PHYTO-PLANKTON	ZOO-PLANKTON	MACRO-INVERTS	MACROPHYTES	EPIPHYTON/ PERIPHYTON
Ease of sampling and analysis	Difficult (1)	Difficult (1)	Moderate (2)	Easy (3)	Difficult (1)
Potential geographic coverage	Wide (3)	Wide (3)	Wide (3)	Moderate (2)	Moderate (2)
Relation to pristineness	Moderate (2)	Moderate (2)	Moderate (2)	Strong (3)	Limited (1)
Sensitivity to pressure gradients	Moderate[a] (2)	Moderate[b] (2)	Unknown[c] (1)	High[d] (3)	Unknown (1)
Normalisation to reference condition	Difficult (1)	Difficult (1)	Difficult (1)	Easy (3)	Difficult (1)
Temporal variability	High (1)	Moderate (2)	Low (3)	Low (3)	High (1)
Use in New Zealand	Common (3)	Occasional (2)	Rare (1)	Common (3)	Rare (1)
Use in other countries	Common (3)	Rare (1)	Common (3)	Common (3)	Occasional (2)
Rank [points]	2 [16]	4 [14]	2 [6]	1 [23]	5 [10]

[a] Cyanobacteria.

[b] Rotifers sensitive to trophic state.

[c] Some sensitive taxa.

[d] Some indices.

PLANKTON COMMUNITY COMPOSITION

Phytoplankters are important primary producers in lake food webs. The abundances of individual phytoplankton taxa may be positively or negatively correlated with nutrient inputs and, thus, the ratio of positive to negative species can be used as a metric of ecological status (Söndergaard et al. 2003). Algal community composition has been shown to be more sensitive to some pressure gradients than functional indicators such as primary productivity (Schindler 1987, 1990).

Zooplankton assemblage composition can be controlled by both higher (planktivores) and lower (phytoplankton) trophic levels and, therefore, provides a useful trophic link to fish predators. However, for this reason, interpretation of responses may be difficult and temporal variability in zooplankton community composition and abundance can be high (US EPA 1998). Chapman & Green (1999) found little seasonal regularity in crustacean zooplankton composition in Lake Rotorua/Te Rotorua nui ā Kahu-matamomoe and suggested that stochastic events such as storms may be the most important factor related to community structure. In contrast, a study examining the spatial and temporal variation of zooplankton community structure in a coastal lake/wetland showed that zooplankton community structure and biomass were highly sensitive to small changes in salinity and moderately sensitive to variables related to trophic state (Schallenberg & Burns 2003; Schallenberg et al. 2003). Ratios, such as the number of large species of cladocerans to total number of cladocerans, have sometimes proven more useful than measures of overall community composition for determining the ecological condition of lakes in Europe (Moss et al. 2003).

Jamieson (1998) showed that strong biogeographical drivers underpinned the distributions of native copepod species in New Zealand and, thus, restrictions on dispersal should also be taken into account when interpreting zooplankton community data and when inferring potential reference conditions of the zooplankton community.

In New Zealand (Duggan et al. 2001) and elsewhere (Gannon & Stemberger 1978), it has been shown that planktonic rotifer community composition is a good indicator of nutrient enrichment in lakes.

Malthus & Mitchell (1989) found that the ratio of phytoplankton biomass to zooplankton biomass was higher in a number of New Zealand lakes than the ratio typical of lakes elsewhere. Although the ratio was not suggested as a metric of pristineness, the authors did suggest that it could indicate the degree of balance between planktonic primary producers and grazers in lakes. This ratio has subsequently been used as an index of EI (Xu 2005).

It can be difficult to relate plankton community composition to pristineness when no appropriate reference lakes are available. Palaeolimnological methods may be used to reconstruct past plankton communities, and a transfer function has been developed linking diatom community structure with trophic state (Reid 2005), allowing the reconstruction of past trophic states in lakes. Fossil pigment analysis in sediment cores can also be used to reconstruct historical phytoplankton community structure (e.g. Leavitt et al. 1994), although problems of pigment preservation in older sediments cast some doubt on interpretations. Remains of body parts of zooplankters are sometimes well preserved in sediments, and work in other countries has resulted in the development of ecological transfer functions linking zooplankton community structure with lake trophic status and fish community structure (Jeppesen et al. 1996). Thus, it may be feasible and fruitful to determine linkages between plankton community structure and reference conditions via palaeolimnological approaches.

MACROINVERTEBRATE COMMUNITY COMPOSITION

Many macroinvertebrates have widespread geographical distributions, making them potentially good indicators of pristineness among lakes. For example, Timms (1982) showed that macrobenthic community composition changed along a gradient of lake trophic state. However, the community structure tended to respond more strongly at higher trophic states. Benthic macroinvertebrates are fed upon by fish and are, thus, important for the maintenance of energy transfer to higher trophic levels in lakes (Kelly & McDowall 2004; Rowe & Schallenberg 2004). While macrobenthos community composition is considered to exhibit moderate temporal variability in some countries (e.g. US EPA 1998), New Zealand's mild temperate climate results in low temporal variability (Talbot & Ward 1987). However, there are some disadvantages to their use as indicators of pristineness in lakes. Considerable within-lake spatial heterogeneity is common, necessitating considerable sampling effort (US EPA 1998; Carter & Griffiths 2001; MacNeil et al. 2001); for example, in New Zealand, depth preferences and substrate type strongly influence species composition (Forsyth & McCallum 1981; Weatherhead & James 2001). In addition, while extensively used in New Zealand streams and upland rivers, macroinvertebrate metrics are poorly developed for New Zealand lakes; however, macrobenthos communities are used as indicators of lake condition overseas (e.g. Anon. 1990; European Union 2000).

In Canterbury, reference conditions of Lakes Forsyth (Wairewa) (Woodward & Shulmeister 2006), Taylor (Schakau 1986) and Grasmere (Schakau 1991, 1993) have been reconstructed using chironomid (sub) fossils preserved in lake sediments, showing that the community structure of chironomids is a useful indicator of changes in trophic state and temperature in lakes.

MACROPHYTE DEPTH LIMITS

There is a strong relationship between the maximum depth to which macrophytes can grow and water clarity, across a wide range of lakes (Schwarz et al. 2000), indicating a strong sensitivity to pressure gradients affecting water clarity. The depth to which macrophytes can grow defines the littoral zone of lakes, which is generally the zone of highest productivity and biodiversity. Therefore, substantial reductions in macrophyte depth limits and water clarity have direct impacts on lake pristineness. Macrophyte depth limits are resilient to short-term variation in water clarity (Schwarz et al. 2000). Thus, this indicator may be preferable to more direct measures of water clarity, which may vary substantially over short time scales, particularly in shallow lakes (e.g. Schallenberg & Burns 2004). Macrophyte depth limits can be determined by SCUBA diving and should be assessed along a number of transects within a lake to account for potential spatial variability. In New Zealand, the maximum depth of submerged plants is used in the LakeSPI system as an indicator of ecological condition in lakes (Clayton & Edwards 2006b). In Europe, the depth limit of submerged plants has been found to be a better indicator of the ecological condition of small lakes than total percentage coverage of macrophytes (Mäemetes & Freiberg 2007). However, macrophyte depth limits are not applicable to shallow lakes in which macrophytes either inhabit the entire lake bed or are completely absent from the lake.

EPIPHYTON AND PERIPHYTON COMPOSITION

Changes in the composition of epiphyton and periphyton have been shown to be useful indicators of eutrophication (Danilov & Ekelund 2000). The organisms that make up these communities generally have widespread geographical distributions, making them suitable for use as indicators of pristineness throughout New Zealand. Benthic diatoms can readily be referenced to pristine conditions using palaeolimnological techniques because their silicate frustrules are generally well preserved in lake sediments (Battarbee et al. 2001). However, monitoring epiphyton and periphyton can be difficult and time consuming due to high spatial variability and the challenges of representative sampling and taxonomic identification (King et al. 2006). In addition, because light attenuation affects taxonomic composition, this should be taken into account when interpreting the composition of these groups as indicators of pristineness (Vis et al. 2006). The use of these groups as indicators is in its infancy in Europe (King et al. 2006), and has only been developed in New Zealand for subalpine pools (Kilroy et al. 2006).

Functional indicators

Functional indicators relate specifically to ecosystem processes that underpin ecosystem functionality. In response to increasing pressures, functional indicators such as rates of productivity or metabolism are generally conserved by species replacement within communities, where new species better adapted to new conditions replace species adapted to prior conditions (Schindler 1987, 1990). Therefore, while functional indicators may not be good early warning indicators of pressure changes, changes to ecosystem function can be catastrophic and, thus, indicators of ecosystem function should also be monitored. The scoring of separate functional indicators of pristineness for lakes against the seven assessment criteria is included in Table 10 and a brief discussion of some of the key points for each of the indicators follows below.

COMMUNITY METABOLISM

The metabolic rate of microbes in lake water and sediments has been studied for almost a century because of the strong link between this and higher level biological productivity. Between the 1950s and 1970s, there was great interest in these methods for assessing the ecological effects of nutrient enrichment in lakes. Phytoplankton primary productivity (PPr) is a key limnological variable for measurement because it represents a large proportion of the total energy available to the food webs of many lakes, particularly lakes in which macrophytes cover only a small proportion of the lake bed. PPr measurements of the water column are usually carried out in bottles filled with lake water, incubated for relatively short periods (e.g. hours to days). A variety of methods have been used to measure rates of phytoplankton production, including radio-labelled inorganic carbon uptake, oxygen production or changes in chlorophyll *a* (i.e. phytoplankton biomass) over time (Schallenberg 2004). Such bottle methods have also been used extensively to determine whether phytoplankton growth is limited by particular nutrients (e.g. nitrogen or phosphorus). While these methods can be relatively easily employed in any lake, studies indicate that rates of productivity can vary greatly on a seasonal basis and some lakes (e.g. Lake Taupo (Taupomoana), Lake Wakatipu) exhibit peaks in phytoplankton productivity in winter (Vincent 1983; Schallenberg & Burns 1997).

TABLE 10. EVALUATION OF FUNCTIONAL INDICES OF PRISTINENESS FOR LAKES AGAINST A SET OF ASSESSMENT CRITERIA. DETAILS OF THE ASSESSMENT CRITERIA ARE OUTLINED IN SECTION 4.5.

ASSESSMENT CRITERIA	METABOLISM	DECOMPOSITION	SEDIMENT ANOXIA	CONNECTEDNESS
Ease of sampling and analysis	Moderate (2)	Difficult (1)	Easy–moderate (2)	Easy (3)
Potential geographic coverage	Wide (3)	Wide (3)	Moderate (2)	Wide (3)
Relation to pristineness	Moderate (2)	Limited (1)	Moderate (2)	High (3)
Sensitivity to pressure gradients	Moderate[a] (2)	Unknown (1)	Moderate[b] (2)	Strong[c] (3)
Calibration to reference condition	Difficult (1)	Difficult (1)	Difficult (1)	Easy (3)
Temporal variability	Moderate (2)	High (1)	Moderate (2)	Low (3)
Use in New Zealand	Rare (1)	Rare (1)	Rare (1)	Occasional (2)
Use in other countries	Occasional (2)	Rare (1)	Rare (1)	Common (3)
Rank [points]	2 [15]	4 [10]	3 [13]	1 [23]

[a] Probably non-unimodal.

[b] In most cases, pristine sediments are aerobic.

[c] For certain pressures, e.g. dams.

The assessment of whole lake rates of PPr and respiration (R) using inorganic carbon or oxygen budgets is difficult because of the equilibration of the gas molecules with the atmosphere, across the water–atmosphere interface. Consequently, the hypolimnion of stratified lakes has been a fruitful setting for research on carbon cycling because thermal stratification virtually prevents gases in this region from equilibrating with the atmosphere. Most research on hypolimnetic metabolism has focussed on oxygen depletion rates (Burns 1995; Schallenberg & Burns 1999), but some researchers have also examined carbon dioxide and methane ($CO_2 + CH_4$) accumulation rates, which is more appropriate if/when hypolimnia become anoxic (Carignan & Lean 1991). Rates of change of hypolimnetic gaseous carbon and oxygen have been expressed as volumetric and areal rates, and much debate has ensued as to which is more informative.

As in river systems, the relative rates of PPr and R in lakes has been extensively studied (e.g. Del Giorgio & Peters 1993; Carignan et al. 2000), though not specifically as measures of pristineness. Whole lake PPr:R ratios provide an indication of the balance between primary productivity and respiration, which could theoretically be an important indicator of pristineness. For example, it can be argued that primary productivity and community respiration should be balanced in pristine ecosystems. However, PPr:R is also strongly driven by the amount of allochthonous organic matter input to lakes (that exported from the catchment mainly as dissolved organic matter), as well as by lake morphology and rates of organic carbon sequestration into the sediments. Therefore, the link between PPr:R and pristineness is also often correlated with other natural environmental gradients.

DECOMPOSITION RATES

The rate of decomposition of particulate organic matter (POC) is influenced by a wide range of environmental factors, which are themselves subject to natural and anthropogenic change (Allan 1995). Although studies on POC decomposition have been common in rivers and streams (e.g. Young et al. 2008), they are relatively rare in lakes. Pope et al. (1999) examined litter colonisation and leaf processing rates in the littoral zone of an oligotrophic lake in Canada and found that litter was used primarily as invertebrate habitat rather than a food source. Talbot & Ward (1987) found that benthic invertebrates specialised in shredding POC were relatively uncommon in Lake Alexandrina. Little is also known about how lake depth, substrate type, proximity to the shoreline and the slope of the lake bed influence POC processing rates in lakes (Pope et al. 1999). Lake sediments are highly spatially heterogeneous with respect to variables such as organic matter content, grain size and water content (Häkanson & Jansson 1983). Therefore, the assessment of sediment POC decomposition rates on a lake-wide basis would require a large number of sampling sites. Decomposition rates are also temperature dependent and, therefore, vary seasonally, not only in relation to thermal stratification.

Hoeniger (1985, 1986) and Ahn (1991) pioneered the use of dyed, regenerated cellulose strips to measure the decomposition rate of cellulose by bacteria in the water columns and sediments of lakes. When measuring the decomposition rates of any introduced substrates, it is important to note that the rates may be non-linear over time because the processing of new substrates integrates different processes, including microbial colonisation, growth and community succession. Microbial enzyme activities on plant substrates (such as cellulose or lignocellulose) have been shown to be potentially useful indicators of organic matter processing rates in lentic environments (Jackson et al. 1995). Hill et al. (2006) found that sediment microbial enzyme activity in the Laurentian Great Lakes was directly related to nutrient availability and may be indicative of regional-scale anthropogenic stressors.

Perhaps the major disadvantages with the use of decomposition measures as indices of pristineness is that little is known about how decomposition relates to pristineness or to pressure gradients. On a broad scale, decomposition rates are positively related to rates of organic matter production and, hence, trophic state, but there can be time lags and spatial decoupling of organic matter production and decomposition within lakes (e.g. Viner 1989), and the magnitude and timing of allochthonous organic matter inputs may play a major role in the metabolism of some lakes (Wetzel 2001).

SEDIMENT ANOXIA

Anoxia (the depletion of dissolved oxygen) in lake sediments often accompanies eutrophication, especially in lakes that exhibit vertical thermal or density stratification of the water column. Anoxia can influence the abundance and composition of the sediment macroinvertebrate assemblage, the rate of organic matter decomposition, and the chemistry and bio-availability of phosphate in sediments. The oxygen status of the sediments can be determined directly using oxygen electrodes or indirectly by measuring either the sediment reduction-oxidation (redox) potential or the depth in the sediment below which an iron rod or wire is inhibited from rusting due to the lack of free oxygen (Schultz 1999). Redox measurements, though indicative of oxygen status, are also influenced by the concentrations of other redox couples (e.g. NO_3^-/NH_4^+, SO_4^{2-}/H_2S, CO_2/CH_4, Fe^{3+}/Fe^{2+}) in the sediment. These methods can be applied generally across lakes, but the oxygen content of sediment is strongly influenced by a number of site-specific factors like temperature, substrate organic matter content, substrate porosity, the light climate at the lake bed and the presence of macrophytes. Therefore, the oxygen content of the sediment exhibits high spatial and temporal variability in lakes.

Under anoxic conditions, sulphate in the sediments can be microbially converted to hydrogen sulphide (H_2S), which may diffuse into the water column and reach toxic concentrations. When the rate of oxygen uptake in sediment is high, the oxic/anoxic boundary can migrate above the sediment into the water column facilitating the release of the plant nutrient phosphate into the water column. In addition, sediment and water column anoxia results in the loss of habitat for multicellular organisms except for those that are most tolerant of anoxia (e.g. many insect larvae of the Chironomidae).

There is no commonly used palaeolimnological method to establish reference sediment oxygen concentrations for a lake. However, the presence and/or abundance of some fossil benthic organisms such as chironomids, benthic diatoms or macrophytes may be used to infer the historical extent of oxygen conditions at the sediment surface at specific sites.

CONNECTEDNESS

The hydrological connectivity of a lake to the associated aquatic environments within its catchment is important for maintaining migrating organisms within lakes, as well as maintaining natural flows of water, energy and matter (e.g. nutrients, dissolved organic matter) through lakes. By definition, a lake with an intact hydrological regime should not receive water diverted from outside its catchment and it should not have its inflow discharges reduced by the diversion or abstraction of water (Moss et al. 2003). As such, the relationship between hydrological connectedness and pristineness is clearly defined and relatively easily determined for any lake. In coastal lakes, the lack of predicted or expected diadromous fish species provides an indication of barriers to hydrological connectedness. An example of managed connectivity in coastal lakes is the artificial opening of the seaward barrier bars of Lake Ellesmere (Te Waihora) and Waituna Lagoon to reduce flooding of surrounding farmland (Duggan & White 2010; Schallenberg et al. 2010). Given current elevated levels of nutrient and sediment loading to these systems, managed openings may now actively enhance EI. Such managed opening regimes highlight the fact that temporal variability may influence the connectivity of some systems. Many hydrological barriers (e.g. dams) and diversions can be assessed from maps and databases, but small-scale abstractions may be more difficult to account for. The New Zealand Water Bodies of National Importance (WONI) database currently contains information on dams and an additional spatial layer is being added for all consented abstractions; thus it may allow adequate assessment of hydrological connectedness/pristineness.

Physico-chemical indicators

Physico-chemical conditions of lakes can be indicators of pristineness. Below, we discuss three commonly used approaches for monitoring the physico-chemical condition of lakes. The scoring of separate physico-chemical indicators of pristineness for lakes against the seven assessment criteria is included in Table 11 and a brief discussion of some of the key points for each of the indicators follows below.

TABLE 11. EVALUATION OF PHYSICO-CHEMICAL INDICES OF PRISTINENESS FOR LAKES AGAINST A SET OF ASSESSMENT CRITERIA. DETAILS OF THE ASSESSMENT CRITERIA ARE OUTLINED IN SECTION 4.5.

ASSESSMENT CRITERIA	TLI AND COMPONENTS	DISSOLVED OXYGEN	NON-NUTRIENT CONTAMINANTS	DISSOLVED ORGANIC MATTER
Ease of sampling and analysis	Easy (3)	Moderate (2)	Moderate (2)	Easy (3)
Potential geographic coverage	Wide (3)	Wide (3)	Wide (3)	Wide (3)
Relation to pristineness	High (3)	High (3)	High (3)	Low (1)
Sensitivity to pressure gradients	High[a] (3)	Moderate[b] (2)	Moderate[c] (2)	Low[d] (1)
Normalisation to reference condition	Moderate (2)	Difficult (1)	Easy (3)	Difficult (1)
Temporal variability	Moderate (2)	Moderate (2)	High (1)	Moderate (2)
Use in New Zealand	Common (3)	Occasional (2)	Common (3)	Rare (1)
Use in other countries	Common (3)	Common (3)	Common (3)	Rare (1)
Rank [points]	1 [22]	3 [18]	2 [20]	4 [13]

[a] Linear.

[b] Complex.

[c] Depends on contaminant.

[d] Some relations with vegetation.

TROPHIC LEVEL INDEX (TLI) AND ITS COMPONENTS

The Trophic-Level Index (TLI) integrates measures of key nutrients, algal biomass and water clarity to indicate lake water quality (Burns et al. 2000). Partly because of its ease of measurement and widespread applicability, it has been widely used in New Zealand lakes, and has been linked to nutrient loading and catchment land use (Hamill 2006). Thus, it is related to the concept of pristineness. TLI has four component parts: total nitrogen (TN), total phosphorus (TP), Secchi depth and chlorophyll *a*.

TN and TP are used as indicators of nutrient availability in many lake quality assessment schemes (Moss 2007), and they are strong predictors of lake productivity in temperate northern hemisphere regions (Vollenweider 1975).

The Secchi depth (or Secchi disk transparency) is a measure of water clarity that integrates factors such as suspended sediment, phytoplankton biomass and water colour caused by dissolved organic matter (known as chromophoric dissolved organic matter or CDOM).

Chlorophyll *a* concentration in the water is a measure of phytoplankton biomass and can be a robust indicator of nutrient enrichment (Moss 2007). However, variation in biomass-specific pigment content (e.g. chlorophyll *a* content per cell) and community composition may, to some extent, decouple chlorophyll *a* concentrations from phytoplankton biomass.

When using TLI as an indicator of pristineness, one must keep in mind that pristine lakes may exhibit high TLI scores due to high natural nutrient loads (shallow lakes situated on fertile plains, waterfowl nutrient loading, etc.) or naturally low water clarity (high levels of suspended sediments, peat staining of water, etc.). TLI variables often vary seasonally and, therefore, TLI measurements are often made on a monthly basis and then statistically adjusted for seasonality. Shallow lakes can exhibit high temporal variability in the components of TLI due to the entrainment of bottom sediment into the water column in windy conditions. Concentrations of inorganic suspended sediments can be used to correct TLI for sediment resuspension effects (Burns et al. 2000).

DISSOLVED OXYGEN

The dissolved oxygen (DO) content of water is a key component of water quality because of its metabolic importance to organisms and its important role in biogeochemical element cycling. DO measurements are particularly useful in lakes that undergo vertical thermal/density stratification because this virtually isolates the bottom waters (hypolimnion) from gas exchange with the atmosphere (reaeration). As a result, rates of DO consumption in the hypolimnion are more directly linked to metabolic processes than rates of change of DO in the mixed layer. Interannual changes in the DO content of the hypolimnion may indicate a trend in trophic state, as oxygen content may decline before other indicators of state (Carlson & Simpson 1996). It is generally considered that most fish cannot survive extended periods of time at oxygen levels below 2.0 mg/L (Wetzel 2001). If DO levels in the water column are below this threshold, it is highly likely that bottom sediments are anoxic, which has numerous implications for lake functioning (see section 5.2.2 ‘Sediment anoxia’). Thus, periods of anoxia in lake waters and surficial sediments would indicate an undesirable, and potentially non-pristine, state. However, the important influences of lake morphometry, temperature and organic detritus can facilitate hypolimnetic anoxia in lakes, even if nutrient loading and phytoplankton biomass are at pristine levels.

In the mixed layer, single measurements of DO are usually not very indicative of water quality, due to typically large spatial, diel and seasonal variations (Moss et al. 2003). However, the use of data-logging DO sensors reduces the need for labour-intensive manual oxygen profiling and allows for measurements at high temporal resolution over long time scales. In shallow lakes, the degree of diel variation in DO concentration of the mixed layer can potentially indicate the level of metabolic activity occurring in lakes, although high-frequency measurements and corrections for oxygen equilibration with the atmosphere are required (Young & Huryn 1999).

NON-NUTRIENT CONTAMINANTS

Anthropogenic, non-nutrient contaminants often influence the public's perception about the ecological state of the environment (Mason 1997) and their presence in freshwater ecosystems is a clear indication of departure from a pristine condition. For example, in non-geothermal areas, heavy metal loading to freshwaters tends to be linked to point source contamination (e.g. mines, smelters) and trace element concentrations in lakes can be influenced by fertiliser use (Downs et al. 2008). The prevalence of organic pesticides in New Zealand freshwaters has not been assessed in detail. These contaminants tend to result from their applications on land by farmers, and by the horticultural and forestry industries. However, both heavy metals and organic pesticides are sometimes directly applied to freshwaters to manage nuisance growths of algae/cyanobacteria (e.g. $CuSO_4$, chelated copper), macrophytes (e.g. Diquat) and riparian vegetation (e.g. glyphosate). Furthermore, alum (an aluminium-based, P-chelating chemical) has been added to lakes to attempt to reduce available phosphorus in the water column (e.g. Paul et al. 2008).

Faecal bacteria are routinely monitored as indicators of the presence of manure and sewage in freshwaters. Their occurrence in freshwaters either tends to be episodic and related to precipitation and resulting runoff from farms, or is related to sewage discharges.

The above indicators are often specific to certain anthropogenic activities and, therefore, their application is generally most useful when targeted to those specific activities or certain point sources of pollution. Contamination events are often episodic and, therefore, sampling to assess the degree of contamination can be difficult. For this reason, sampling sediment contaminant concentrations can be a useful way of determining time-integrated contaminant loads. Some of these indicators can be quite complex and costly to analyse (e.g. specific organic chemicals), whereas others are measured routinely and are not expensive to analyse (e.g. faecal bacteria).

For contaminants that naturally occur in low concentrations, determining pristine reference concentrations in lakes can be challenging and requires careful palaeolimnological procedures and exacting analytical methods.

DISSOLVED ORGANIC MATTER

Organic matter dissolved in lake waters is composed of a wide variety of organic solutes, from transparent, low-molecular-weight molecules to coloured, highly polymerised molecules. The latter are collectively referred to as chromophoric dissolve organic matter (CDOM), but have also been referred to as water colour, dissolved humic substances and gelbstoff. CDOM is responsible for a wide range of ecological functions in lakes, including chemical chelation (e.g. ions and a range of toxic contaminants; Steinberg 2004), the absorption of harmful UV radiation (Rae et al. 2001) and the provision of substrates for microbial food webs (Steinberg 2004). CDOM is typically supplied to lakes from soil organic matter and from wetland vegetation in the catchment (Rasmussen et al. 1989; Steinberg 2004). While CDOM is mainly sourced from the catchment, it is degraded in lakes by photo-oxidation and microbial degradation (Steinberg 2004) and, consequently, its concentration in lakes has been shown to decline with increasing water residence time (Rasmussen et al. 1989). Schindler et al. (1997) have also shown that the CDOM concentration of lakes in the Canadian boreal forest region is sensitive to climatic variations. Thus, while CDOM is an easily measured key physico-chemical component of lake water (Cuthbert & Del Giorgio 1992) reflecting the abundance and condition of soils and wetlands in the catchment, its concentration in lake water is also affected by *in situ* degradation processes and regional climate. Furthermore, little work has been done to characterise reference CDOM concentrations in lakes or CDOMs contribution to EI and, hence, its utility as an indicator of EI is questionable at this time.

5.2.3 Diversity

Biological diversity is usually a community-specific indicator that refers simply to species richness (number of species) or to any of a wide range of diversity indices that incorporate other data to account for the distribution of biomass across the taxa that make up the community (see section 5.1.3 for discussion about diversity indices). For example, a variety of diversity indices have been commonly used to summarise New Zealand stream invertebrate and fish community structure (e.g. Death & Winterbourn 1995). However, less work has been done on lake biodiversity in New Zealand. It has been suggested that diversity in ecosystems should impart ecological resilience (McCann 2000). The scoring of separate community diversity indicators for lakes against the seven assessment criteria is included in Table 12 and a brief discussion of some of the key points for each of the indicators follows below.

Native aquatic macrophyte diversity is probably the best-documented indicator, and is included as a component of the commonly used LakeSPI methodology of lake condition assessment (e.g. Clayton & Edwards 2006b). The diversity of low-growing aquatic plants was found to be positively correlated to intra-annual water level variation, while negatively correlated to inter-annual water level variation in 21 New Zealand lakes (Riis & Hawes 2002). This work indicates that macrophyte diversity is sensitive to lake condition, the presence of certain invasive aquatic macrophytes and physical disturbance regimes. While macrophyte biomass can vary greatly depending on season, macrophyte species richness is relatively unaffected by seasonal variation (De Winton & Schwarz 2004).

TABLE 12. EVALUATION OF DIVERSITY OF DIFFERENT TAXONOMIC GROUPS FOR LAKES AGAINST A SET OF ASSESSMENT CRITERIA. DETAILS OF THE ASSESSMENT CRITERIA ARE OUTLINED IN SECTION 4.5.

ASSESSMENT CRITERIA	PHYTO-PLANKTON	ZOO-PLANKTON	MACRO-INVERTS	FISH	MACRO-PHYTES	EPIPHYTON/PERIPHYTON
Ease of sampling and analysis	Difficult (1)	Difficult (1)	Moderate (2)	Moderate (2)	Moderate (2)	Difficult (1)
Potential geographic coverage	Wide (3)	Wide (3)	Wide (3)	Moderate (2)	Moderate (2)	Moderate (2)
Relation to diversity	N/A	N/A	N/A	N/A	N/A	N/A
Sensitivity to pressure gradients	Low[a] (1)	Low[a] (1)	Moderate[b] (2)	Moderate[c] (2)	High[d] (3)	Low[a] (1)
Normalisation to reference condition	Difficult (1)	Difficult (1)	Difficult (1)	Moderate (2)	Moderate (2)	Difficult (1)
Temporal variability	High (1)	Moderate (2)	Low (3)	Moderate (2)	Low (3)	High (1)
Use in New Zealand	Rare (1)	Occasional (2)	Rare (1)	Rare (1)	Common (3)	Rare (1)
Use in other countries	Rare (1)	Rare (2)	Rare (1)	Occasional (2)	Common (3)	Occasional (2)
Rank [points]	5 [9]	4 [12]	2 [13]	2 [13]	1 [18]	6 [9]

[a] Poorly known.

[b] Some sensitive taxa.

[c] Sensitivity to invasive species.

[d] Some indices.

The negative impacts of introduced trout on native fish diversity have been clearly demonstrated in streams (Townsend 1996). However, evidence for similar effects in lakes remains largely circumstantial (Deans et al. 2004; Wissinger et al. 2006). It has also been suggested that, due to the generally low freshwater fish diversity in New Zealand, fish species diversity is a poor indicator of lake condition, despite its popularity as an indicator of lake condition in other countries (Rowe & Graynoth 2002). Freshwater fish diversity can vary seasonally due to migrations.

Macroinvertebrate diversity has been shown to relate to the nutrient status of lakes, but the relationship was quite non-linear, with the main response occurring at the super-eutrophic end of the nutrient gradient (Timms 1982). Weatherhead & James (2001) also showed that littoral invertebrates were strongly influenced by physical gradients of depth and exposure, and Kelly & Hawes (2005) demonstrated that the community composition of macroinvertebrates was related to invasive macrophytes. Generally, the diversity of sub-littoral and littoral zone macroinvertebrates remains quite stable throughout the year (Talbot & Ward 1987; Kelly & McDowall 2004).

Zooplankton exhibit sufficient levels of diversity in New Zealand lakes to indicate that zooplankton diversity could be a useful indicator of EI. This has not been extensively explored in New Zealand, but Drake et al. (2010) showed that rotifer diversity was negatively correlated with land-use pressures in shallow lakes.

The measurement of diversity is sensitive to sampling effort because the ability to collect rare species in a sample can be strongly dependent on sampling effort (Hughes 1978). Thus, when comparing diversity measures among samples, the diversity values should be standardised for sampling effort, although there is no universally accepted method for doing this. One approach may be to count an equal number of individuals in each sample. Thus, a large number of samples are required to effectively quantify the diversity in habitats with high diversity density (e.g. littoral zones of lakes; MacNeil et al. 2001). The taxonomic acuity of the researcher identifying the organisms also influences diversity assessments, making it difficult to compare diversity assessments made by different researchers. The ability to resolve taxa to genus or species varies among groups of taxa (e.g. fish are easier to identify than phytoplankton). Therefore, diversity is more likely to be quantified in communities in which accurate taxonomic identification and discrimination is easier.

Across wide geographical scales, the diversity of communities is related to a number of natural gradients such as altitude, latitude, evapotranspiration, and spatial and temporal habitat variability (Mason 1997). Furthermore, biogeographical influences related to immigration rates (especially on islands), extinction rates and species refugia influence the diversity of communities in ways that do not reflect anthropogenic pressures. The human-assisted introduction of non-indigenous species into ecosystems compromises the nativeness and pristineness components of EI. Therefore, the measurement of the diversity of native species aligns more closely to our definition of EI than does the total species diversity of a given freshwater community.

Currently, little is known about how levels of native diversity under pristine conditions compares with diversity in lakes experiencing various levels and types of anthropogenic pressures.

5.2.4 Resilience

Ecological resilience can be assessed by measuring the amount of perturbation required to modify the set of mutually reinforcing processes and structures that characterise a lake ecosystem (Petersen et al. 1998). Thus, the concept of ecological resilience relates to factors affecting stability and vulnerability to change in lake ecosystems. Below, we discuss how light limitation and macrophyte collapse, nutrient limitation and cyanobacterial blooms, and food web complexity may be potentially useful indicators of ecological resilience of lakes. In Table 13, we assess the utility of these resilience indicators against the seven assessment criteria for assessing lake EI.

TABLE 13. EVALUATION OF INDICES OF RESILIENCE FOR LAKES AGAINST A SET OF ASSESSMENT CRITERIA. DETAILS OF THE ASSESSMENT CRITERIA ARE OUTLINED IN SECTION 4.5.

ASSESSMENT CRITERIA	LIGHT LIMITATION	NUTRIENT LIMITATION	FOOD WEBS
Ease of sampling and analysis	Easy–moderate (3)	Moderate (2)	Difficult (1)
Potential geographic coverage	Wide (3)	Wide (3)	Wide (3)
Relation to resilience	Complex (1)	Complex (1)	Complex (1)
Sensitivity to pressure gradients	Low[a] (1)	Low[a] (1)	Low[a] (1)
Normalisation to reference condition	Difficult (1)	Difficult (1)	Difficult (1)
Temporal variability	High (1)	High (1)	Low (3)
Use in New Zealand	Common (3)	Common (3)	Occasional (2)
Use in other countries	Common (3)	Common (3)	Common (3)
Rank [points]	2 [16]	3 [15]	1 [17]

[a] Poorly known.

Light limitation and macrophyte collapse

The growth of aquatic macrophytes is strongly linked to water quality in lakes. Jeppesen et al. (1990) found significantly higher water transparencies in lakes with a large coverage of macrophytes compared with lakes where macrophytes were absent. In addition, Scheffer et al. (1993) argued that macrophyte growth leads to self-stabilisation of a clear water, macrophyte-dominated state that is ecologically valuable because macrophytes increase water clarity by reducing algal biomass and the resuspension of sediments. Scheffer (1998) described a theory of alternative stable states in shallow lakes, whereby either a macrophyte-dominated clear water state or a plankton-dominated turbid state can be maintained by ecological feedbacks, despite pressures to change from these states. If pressures exceed a resilience threshold, a lake can undergo rapid change to the alternate stable state.

Light limitation of macrophytes can be caused by increased turbidity and excessive growth of periphyton and/or deposition of fine particulate material on macrophyte fronds. The euphotic depth (depth in the lake to which 1% of surface irradiance can penetrate) is indicative of the maximum depth to which macrophytes can grow. Euphotic depth is assessed by measuring depth profiles of PAR (photosynthetically active radiation) with an underwater PAR sensor. The euphotic depth can be assessed in any lake but can vary markedly over time in response to seasonality, floods, algal blooms and wind. Excessive variation in, or consistent reductions in, the euphotic depth over time can indicate that any macrophyte communities that are present may be become light-limited, leading to the collapse of these communities. Thus, where historical water clarity data are lacking, the ratio of the euphotic depth to the maximum depth of macrophyte growth is likely to be a useful indicator of recent change in water clarity. Water level variations and the effect of the wave wash zone also help determine the potential area of lake bed available for macrophyte growth.

The collapse of macrophytes has been found to accompany declines in water quality in many New Zealand lakes (Hayes & Rutledge 1991; Hamill 2006; Kelly & Jellyman 2007; Schallenberg & Sorrell 2009). While the consumption of macrophytes by birds, fish or freshwater crayfish can contribute to macrophyte collapse (Hamill 2006), the reduction in light penetration as a result of excessive algal growth (usually the result of nutrient enrichment from the catchment) and the introduction of the invasive exotic macrophyte *Egeria densa* may also be important drivers (Schallenberg & Sorrell 2009). Other factors include the erosion of lake margins due to water level fluctuation, increased levels of suspended solids contributed from tributaries and the introduction of bottom-feeding exotic fish (Rowe 2004; Schallenberg & Sorrell 2009). Therefore, regime shifts from clear water to turbid states and the past frequency of such shifts are strong indicators of vulnerability to abrupt changes in EI. Furthermore, the determination of the proximity of a lake's condition to its resilience threshold (demarcating the onset of macrophyte collapse) is a key indicator of the resilience of a lake to macrophyte collapse and the potential to switch states (Schallenberg & Sorrell 2009). To assess whether specific lakes exhibited regime shifts prior to anthropogenic influence necessitates the use of palaeolimnological techniques, but these rarely provide the high temporal resolution required to determine the historical frequencies and/or durations of regime shifts.

Nutrient limitation and cyanobacteria blooms

Phytoplankton blooms are generally related to lower water quality and they often have serious consequences for lake ecosystems. The growth of phytoplankton can be limited by the availability of a number of factors in lakes. While it is often assumed that phytoplankton will respond positively to nutrient additions, this may not be true if they are nutrient saturated and are limited by other factors such as light (Schallenberg 2004). On the other hand, where the availability of nutrients does limit phytoplankton growth, nutrient management may be a useful tool for controlling algal proliferation. Therefore, understanding the factor(s) limiting algal proliferation in lakes is important if we are to understand the potential resilience of lakes to anthropogenic pressures such as nutrient loading.

In shallow lakes that do not thermally stratify, phytoplankton light limitation is related to the ambient light levels, the euphotic depth, the maximum lake depth and physiological characteristics of the phytoplankton community. In thermally stratified lakes, light limitation is also related to the mixed layer depth. Some phytoplankters are able to overcome light stress passively by regulating their buoyancy using gas vesicles (e.g. cyanobacteria) or by active movement (e.g. dinoflagellates).

In lakes in which phytoplankton receive sufficient light, nutrient availability may limit the growth of phytoplankton at times (Schallenberg 2004). Nutrient limitation can be determined by carrying out nutrient enhancement bioassays, and may be inferred by examining nutrient concentrations (i.e. availability) in the mixed layer (White et al. 1985). Nutrients that commonly regulate phytoplankton growth in lakes include inorganic nitrogen, inorganic phosphorus and trace elements (Schallenberg 2004; Downs et al. 2008). Phytoplankters that can utilise alternative sources of nutrients, either by fixing atmospheric nitrogen (e.g. some cyanobacteria) or by consuming bacteria (e.g. some dinoflagellates), are able to overcome a lack of inorganic nutrient availability that can limit the growth of other phytoplankters. Thus, nitrogen fixing cyanobacteria often dominate the phytoplankton in lakes in which other phytoplankters are nitrogen limited. Such conditions may be inferred by examining the ratios of dissolved inorganic nitrogen to total phosphorus in the mixed layer (Morris & Lewis 1988).

Under certain conditions, some cyanobacteria can achieve very high biomasses. Such 'blooms' can trigger anoxia in bottom waters as the blooms decompose, facilitating the anaerobic release of phosphate previously bound in the lake sediments. This can then fuel further phytoplankton growth, establishing a positive ecological feedback. Furthermore, some cyanobacteria taxa can produce toxins that can injure or kill wildlife, pets and humans that come into contact with lake water (Wood et al. 2006). The frequency of occurrence of visible cyanobacterial blooms has been suggested as one of the few variables that can be used robustly and inexpensively to assess the ecological condition of European lakes (Moss 2007). Blooms and periods of toxin production can be highly episodic and, therefore, monitoring should be undertaken at a relatively high frequency to determine whether cyanobacteria occur and produce toxins in a given lake.

As light and nutrients are often the main drivers of phytoplankton biomass and growth in lakes, knowledge of light and nutrient conditions in lakes is important for assessing the potential for phytoplankton blooms to occur, particularly cyanobacterial blooms. Whether New Zealand lakes in their pristine condition experienced cyanobacterial blooms is not yet understood; however, the resting stages (akinetes) produced by some cyanobacteria have been recovered from lake sediments up to 100 years old (Wood et al. 2009). Thus, palaeolimnological methods appear to be able to provide information on the historical presence of cyanobacteria in lakes. Determining both the resilience of lakes to phytoplankton blooms and the relationships between resilience and anthropogenic pressures are important in assessing EI.

Food web analysis

Studies investigating the impact of biodiversity loss on ecosystem function have become widespread as a result of concern over species extinctions (Loreau et al. 2001; Duffy 2002). Examination of food web complexity or food chain length is a useful means of integrating information on potential changes to the functioning of aquatic ecosystems (Vander Zanden et al. 1999b). Sampling of food web components can be time consuming and costly, especially if whole food webs are examined. However, costs can be reduced by focusing on portions of the food web (e.g. top trophic levels). Food web studies can be conducted in any system; however, biogeography and lake ecotype considerations may be relevant when comparing results between lakes or regions, or calibrating results to reference lakes. Food web studies can help develop an understanding of the complex effects of introduced species (Vander Zanden et al. 1999a; Rowe & Schallenberg 2004; Kelly & Hawes 2005) and eutrophication (Fry 1991; Jones et al. 2004) on aquatic systems. Stable isotopes of carbon and nitrogen have often been used to provide time-integrated information about food web dynamics and energy flow through the food web (Vander Zanden et al. 1999b; James et al. 2000; Kelly & Jellyman 2007; Pace et al. 2007). Stable isotope data have been collected in a variety of New Zealand lakes and may provide insights into how lake food webs respond to disturbance, serving as a suitable structural indicator of ecosystem resilience.

6. Discussion

Ecological integrity continues to be a controversial concept, both in definition and measurement. Our working definition follows a long line of previous attempts worldwide. Although the precise definition may vary, EI is generally considered to be a composite of at least some aspects of nativeness, pristineness, diversity and resilience. To exhibit maximum integrity, a river or lake would maintain itself independently of any human influence, exhibiting resilience to natural changes.

Problems persist in defining appropriate reference conditions against which to assess the current EI of New Zealand's rivers and lakes. Many water quality classification systems have opted for using minimally impacted sites to represent reference conditions. Around the world, classification systems are increasingly being used to classify lakes and streams into types and type-specific biological targets, as a result of many government agencies adopting a holistic, ecoregion approach. Such geographical classifications are useful for dealing with natural geographical variability, as they acknowledge that differences between regions may influence how pressures impact on biota (Hawkins et al. 2000; Sandin & Verdonschot 2006). The European Water Framework Directive (WFD) uses abiotic variables to classify streams and rivers into types, defined by ecoregions, catchment area, geology and altitude, whereas the UK RIVPACS predictive model and its many derivatives use biotic variables. Sandin & Verdonschot (2006) suggested that multivariate models were more effective at predicting reference conditions for macroinvertebrates than WFD physical typology categories, as they made use of continuous rather than categorical variables. However, FWENZ (Freshwater Environments Classification of New Zealand rivers), the GIS-based multivariate environmental classification of the New Zealand river network, seems to be an advance on the WFD methodology. It uses a unique combination of biological (fish and macroinvertebrate) and environmental inputs to classify New Zealand's rivers and streams into general habitat groups at a range of scales, from national to local (Leathwick et al. 2008b). The detailed nature of local and regional type classes offers great potential for EI assessment in specific river habitats or ecotypes, allowing focussed use of metrics (especially if major pollution threats have already been identified) and providing environmental contexts for interpretation.

Rivers and lakes have some key differences with regard to the factors governing their physico-chemical conditions and structuring their biotic communities. However, both are subject to an array of anthropogenic pressures, impacting at a variety of spatial and temporal scales (Allan et al. 1997; Danz et al. 2007). In general, analytical tools for the evaluation of EI in lakes are less developed than those for rivers. This may be a reflection of the great ecological variability that seems to exist within lake ecotypes, necessitating the establishment of reference conditions on a case-by-case basis. Useful indicators of EI should be relatively robust to natural spatio-temporal variability, but must be sensitive to the anthropogenic pressures they attempt to assess. We have discussed some of the sources of variability and uncertainty related to many potential EI indicators and, ideally, sampling error, variability in taxonomic resolution and other extraneous sources of variability must be accounted for when using them. This has also been highlighted as a concern in a number of current European and North American indicator schemes.

6.1 RECOMMENDED INDICATORS

The careful selection of structural and functional indicators is important when developing a practical scheme for assessing EI. We have evaluated a range of common indices that can be used to measure the four core components of EI: nativeness, pristineness, diversity and resilience. In Tables 14 & 15, we present a list of the indicators for rivers and lakes that we determined as being the most useful and robust for measuring the components of EI. Our assessment was guided by seven assessment criteria.

For the purpose of this review, there was no weighting of the assessment criteria, so each contributed equally to the indicator's rank. If the objectives of the monitoring programme were focussed around specific criteria, such as sensitivity to particular pressure gradients, the assessment criteria could be weighted accordingly. Within the tables, we have also cited the predominant pressure gradients associated with each of the indicators; thus, a suite of indicators could be selected to align with known pressures. Because there is some redundancy in the indicators identified in the tables (typically three indicators per EI component), the use of all the suggested indicators is not essential to obtaining a robust assessment of EI. Thus, ideally, the combination of indicators selected should reflect the range of pressures expected over the monitoring network and should cover the four EI components.

TABLE 14. SUGGESTED LIST OF INDICATORS FOR THE ASSESSMENT OF ECOLOGICAL INTEGRITY IN RIVERS AND STREAMS.

The indicators identified under each category were taken from the highest ranking indicators as tested against the seven assessment criteria (Tables 1–7). Indicators are thereby thought to be universally applicable, robust, relatively inexpensive, require minimal taxonomic skill and are likely to be the most responsive to anthropogenic stressors. The main stressors that can be detected by the indicators are also reported.

GENERAL PROPERTY OF EI	INDICATOR	EXAMPLES OF MAIN STRESSORS THAT MAY BE DETECTED
Nativeness	Native fish (% native, no. of introduced species, O/E)	Invasion and introduction
	Presence of invasive macrophytes/algae	Invasion and introduction
Pristineness		
Structural	Macroinvertebrate community composition (MCI, %EPT)	Multiple disturbances
	Fish IBI	Invasion and introduction
Functional	Ecosystem metabolism	Eutrophication, habitat degradation, flow abstraction
	Wood decomposition rates	Eutrophication, change in land use
	BOD	Organic enrichment
	δ15N of primary consumers	Specific N and P enrichment
Physico-chemical	Water clarity, turbidity	Eutrophication, sedimentation
	Nutrient concentrations	Eutrophication, sediementation
	Water temperature, dissolved oxygen	Riparian and catchment clearance, abstraction
Diversity	Macroinvertebrate taxonomic richness, diversity, O/E richness	Multiple disturbances
	Abiotic structure (habitat template)	Change in physical template, abstraction, irrigation
Resilience	Presence/absence of key indicator taxa	Multiple disturbances
	Ecosystem function	Change in physical template

TABLE 15. SUGGESTED LIST OF INDICATORS FOR THE ASSESSMENT OF ECOLOGICAL INTEGRITY IN LAKES.

The indicators identified under each category were taken from the highest ranking indicators as tested against the seven assessment criteria (Tables 8–13). Indicators are thereby thought to be universally applicable, robust, relatively inexpensive, require minimal taxonomic skill and are likely to be the most responsive to anthropogenic stressors. The main stressors that can be detected by the indicators are also reported.

GENERAL PROPERTY OF EI	INDICATOR	EXAMPLES OF MAIN STRESSORS THAT MAY BE DETECTED
Nativeness	Catch per unit effort (CPUE) of native fish	Invasion by / introduction of exotic species
	Percentage of species native (macrophytes, fish)	Invasion by / introduction of exotic species
	Absence of invasive fish and macrophytes	Invasion by / introduction of exotic species
	Proportion of shoreline occupied by native macrophytes	Invasion by / introduction of exotic species
Pristineness		
Structural	Depth of lower limit of macrophyte distribution	Eutrophication (benthic effects)
	Phytoplankton community composition	Eutrophication
Functional	Intactness of hydrological regime	Connectedness, abstraction, irrigation, artificial human barriers
	Continuity of passage to sea for migrating fish (potentially indicated by diadromous fish)	Connectedness, artificial human barriers
	Water column DO fluctuation	Eutrophication
	Sediment anoxia (rate of redox potential change in sediments)	Anoxia, eutrophication (benthic effects)
Physico-chemical	TLI (or its components)	Eutrophication
	Non-nutrient contaminants	Depends on pressures
Diversity	Macrophyte, fish, invertebrate diversity indices	Loss of biodiversity
Resilience	Number of trophic levels	Loss of top predators
	Euphotic depth compared to macrophyte depth limit	Macrophyte collapse
	Instance/frequency of macrophyte collapse or recorded regime shifts between clear water and turbid states	Macrophyte collapse
	Compensation depth compared to depth of mixed layer	Potential for light or nutrient limitation of phytoplankton growth
	DIN:TP ratio	Risk of cyanobacterial blooms
	Presence of potentially bloom-forming cyanobacteria	Risk of cyanobacterial blooms

EI indicators could also be used to assess the outcomes of restoration or site rehabilitation. In such cases, the selection of indicators would reflect the management actions employed at the site. For example, if nutrient attenuation (e.g. riparian enhancement) was the primary focus of the restoration, indicators could reflect the physico-chemical components of pristineness, such as nutrient concentrations, and the structural and functional components of the ecosystem that could respond to such changes in physico-chemistry, such as native plants and macroinvertebrates, and ecosystem metabolism.

6.2 COMBINING METRICS TO MEASURE ECOLOGICAL INTEGRITY

Diverse anthropogenic pressures, such as agriculture, point source pollution and changes in land use, overlap in space and time, and can have independent, synergistic or antagonistic effects on ecosystems (Niemi & McDonald 2004). This review has evaluated a range of potentially useful indicators of EI and grouped them into the four components of EI. From a management perspective, it would be useful to combine these individual component metrics into a single, comprehensive measure of EI. This has previously been attempted in different ways, but predominantly by either combining or averaging indicator values, informed by expert opinion, into a single multimetric index, or by developing a multimetric model from a range of indicators using multivariate statistical methods.

Karr (1981) was the first to suggest combining fish metrics to create an index of biological integrity (IBI). His approach formed the cornerstone of multi-metric index (MMI) development and has been widely adopted in river assessment (e.g. Lyons 1992; Hering et al. 2006) and to some degree in lakes (Reavie et al. 2008). When they work, these can be powerful tools for making informed management decisions about EI. However, they can also be over-simplistic and misleading if insufficient attention is paid to assumptions, or if it is difficult to measure or evaluate the present departure of the system from expected reference conditions (Niemi & McDonald 2004). Recent comparative studies of multiple indicators have shown how different groups of organisms provide complementary information about ecological condition. For example, a parallel investigation of fish, macroinvertebrates and diatom assemblages showed that assessments based on one group were less likely to indicate degraded conditions than assessments based on several groups. Individual component metrics should be carefully weighted, as the discriminatory power and relative importance of different metrics have been shown to vary markedly in rivers and streams. For example, in a study of over 150 streams, Johnson et al. (2006) found that benthic diatom and macroinvertebrate metrics showed high discriminatory power and low error for a pressure gradient describing nutrient enrichment, but performed poorly for a gradient describing habitat alteration and hydromorphology; the converse was true for fish and macrophyte metrics. Snelder et al. (2006) highlighted the problems of weighting and transforming variables when designing multivariate environmental classifications. Similarly, Leathwick et al. (2008b) recognised the problems in the development of FWENZ, which used environmental variables for which selection, weighting and transformation had been defined via analysis of fish and macroinvertebrate distribution data.

Multivariate statistical methods such as canonical correspondence analysis (CCA) and redundancy analysis (RA) (Ter Braak & Verdonschot 1995) have been used to derive statistical models for estimating and predicting ecological condition (e.g. Danz et al. 2007), and informative comparisons have been made between these methods and multimetric methods (e.g. Reynoldson et al. 1997; Karr 2000). Such methods include axis reduction procedures, which can help simplify and focus models on key variables and relationships. Other advantages include objective model selection, statistical optimisation and validation, and the modelling of multivariate response variables (e.g. EI).

Multivariate regression tree modelling is a type of multivariate analysis related to RA and CCA. However, rather than analysing environmental gradients as RA and CCA do, the regression tree approach produces a classification of discreet environment types (De'Ath 2002). Boosted regression tree modelling (BRT; Elith et al. 2008) was applied to a national New Zealand dataset to examine the responses of multiple ecological indicators to a range of environmental stressors (e.g. agricultural land use, urbanisation), yielding some useful information on the performance of various ecological indicators against known stressor gradients (Clapcott et al. 2010; J. Clapcott, unpubl. data). The BRT models provided a comparative measure of strength of association (% deviance explained) and predictive error (cross validation coefficient), as well as an indication of the form of the relationship (i.e. linear, curvilinear, unimodal). The strongest indicators, based on model sensitivity and precision, were NOx, the $\delta^{15}N$ of primary consumers and the macroinvertebrate community index, while the weakest overall were gross primary productivity, fish taxon richness and invertebrate taxon richness. This information was then used in a multimetric indicator of EI, with individual metrics weighted based on their contribution to the model (% deviance explained) and the proportion of the river network for which the indicator data were available (i.e. certainty). Similarly, Drake et al. (2010) used BRT modelling to examine a range of lake metrics to land-use stressor gradients, but due to the limited sample size (45 shallow lowland lakes), were unable to develop a multimetric lake EI index from the dataset.

The next stage of this project will quantitatively examine the performance of indicators against a number of human pressure gradients as defined in the Waterbodies of National Importance (WONI) model (Leathwick & Julian 2007). This work, which is partially underway, will provide a quantitative evaluation of the recommended EI indicators for lakes (Drake et al. 2010) and rivers (Clapcott et al. 2010), and will also provide further guidance on measuring the EI of freshwaters.

7. Acknowledgements

This research was funded by a grant from the Foundation for Science Research and Technology, Cross Departmental Research Pool fund to the Department of Conservation (Investigations 3947 and 3948). We would like to thank two anonymous reviewers for their helpful comments on improving the manuscript. Reviews on earlier versions of the manuscript were also provided by Vera Power and Clive Howard-Williams. We would also like to acknowledge input into discussions on defining and measuring ecological integrity in freshwaters with members of the project's technical advisory group, including Chris Arbuckle, Kevin Collier, Bruno David, Deanne Drake, David Hamilton, Jon Harding, John Leathwick, Dean Olsen and Dennis Trolle.

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How can ecological integrity be used to manage New Zealand freshwaters?

Ecological integrity (EI) can be used to quantify ecosystem structure and function. However, there are many definitions of EI and, consequently, several different methods are used to measure it. In this report, we arrive at a working definition of EI, which includes four components: nativeness, pristineness, diversity and resilience. We then develop a methodology for assessing EI based on quantification of these components. We conclude with a list of suitable indicators for determining EI in New Zealand rivers and lakes.

Schallenberg, M.; Kelly, D.; Clapcott, J.; Death, R.; MacNeil, C.; Young, R.; Sorrell, B.; Scarsbrook, M. 2011: Approaches to assessing ecological integrity of New Zealand freshwaters. *Science for Conservation 307.* 84 p.