Scoping indicators for impacts on freshwater biodiversity and ecosystem processes of rivers and streams

Prepared for Ministry for the Environment

May 2018

Prepared by:
R Storey
C Kilroy
F Matheson
M Neale, Puhoi Stour Ltd.
S Crow
A Whitehead

For any information regarding this report please contact:

Richard Storey
Freshwater Ecologist
Freshwater Ecology Group
+64-7-859 1880
richard.storey@niwa.co.nz

National Institute of Water & Atmospheric Research Ltd
PO Box 11115
Hamilton 3251

Phone +64 7 856 7026

NIWA CLIENT REPORT No:	20118118HN
Report date:	May 2018
NIWA Project:	MFE18202

Quality Assurance Statement		
	Reviewed by:	Clive Howard-Williams
	Formatting checked by:	Alison Bartley
	Approved for release by:	John Quinn

Contents

Tables

Figures

Executive summary

Under the Environmental Reporting Act (2015), the Government Statistician (Stats NZ) and the Secretary for the Environment (Ministry for the Environment) report on state of New Zealand's environment across five domains – air, atmosphere/climate, land, marine and freshwater. Each domain report must describe the state of the domain, the natural and human pressures that may be causing changes to the state, and the impacts (i.e., what the state of the domain means for us). Impacts must be reported using indicators in five impact categories, one of which is ecological integrity – defined as biodiversity and ecosystem processes. Currently the only indicator for impacts on biodiversity and ecosystem processes is conservation status of native freshwater plants, fish and macroinvertebrates. This is classed as a "case study indicator", meaning it is either less accurate or has less data coverage than a "national indicator". Ministry for the Environment (MfE) is seeking new indicators of impacts on biodiversity and ecosystem processes for use in the next freshwater domain report (due for release in 2020) that will meet the "Tier 1" criteria for national indicators.

In this report we discuss indicators of impacts on biodiversity of five organism groups and on three measures of ecosystem processes in fresh water. For reporting purposes, the fresh water domain relates to all the physical forms of unsalted water, including those in rivers, lakes, streams, ponds, wetlands, aquifers, and glaciers. However for this report, MfE has requested only indicators related to rivers and streams.

For most of the indicators presented, their responses to anthropogenic pressures are well documented and predictable. This means that a given value of an impact indicator can usually be understood as a response to particular pressures where those pressures are known and quantified. However, it may not be possible to infer or quantify the pressures from a given value of the impact indicator. This is because most environments involve multiple pressures which act simultaneously on the impact indicator individually and via complex interactions, thus it is usually not possible to determine which pressure or pressures an impact indicator is responding to.

Periphyton are the dominant primary producer in hard-bottomed streams and rivers. Biomass (as chlorophyll *a*) is established as a suitable periphyton indicator for which guideline values are available, and data that meet Tier 1 criteria are being collected across the country. Chlorophyll *a* is generally reflected by cover of periphyton on the stream bed. Therefore, measures of percentage cover can provide useful supplementary or alternative data to chlorophyll a. Periphyton biomass can be considered as an indicator of biodiversity, but also of ecosystem processes, as abundant growths can alter habitat for aquatic animals and cause fluctuations in water chemistry. In most streams and rivers, periphyton biomass responds predictably to anthropogenic pressures, but in some cases natural factors can have similar effects to human pressures. With further development a community composition index could usefully complement biomass or cover, allowing discrimination between human pressures and natural factors.

Macrophytes are the dominant primary producer in soft-bottomed streams and rivers. Biovolume of native and introduced species is recommended as a suitable macrophyte indicator, representing both biodiversity and ecosystem processes (as it affects habitat and water chemistry in a similar way to periphyton). Guideline values for biovolume are available. Data that meet Tier 1 criteria are being collected in some regions, and other regions are gradually incorporating macrophyte data in monitoring programmes. Native and introduced species biovolume responds in predictable ways to four important anthropogenic pressures associated with landuse change. A community composition index could usefully complement biovolume to detect incursions of high risk introduced species.

Macroinvertebrates represent the key trophic link between primary producers and vertebrate consumers (fish and birds). Their role in biological monitoring of streams and rivers is well established internationally, and a variety of indices has been developed overseas. In New Zealand, several indices, the MCI (Macroinvertebrate Community Index), EPT richness and % EPT richness are well established, and could be combined for greater discriminatory power. Condition bands have been determined for MCI, and could be refined by indexing against reference values which have been determined for each stream reach in the country. With modest extra investment, a more powerful index of macroinvertebrate biodiversity could be developed for New Zealand by building on existing information for reference condition.

Freshwater fish are undoubtedly the most highly valued of the aquatic biota, as they include food species (whitebait, tuna, lamprey) treasured by Māori and Pākeha. They are suitable as indicators of impact as they represent the top of the food chain. Indicators of fish biodiversity are not currently developed, but several models exist that could be developed with modest investment to enable development of suitable indicators. The NZ Freshwater Fish Database represents a rich source of data on which to base indicators, but biases associated with different fishing methods need to be carefully accounted for when developing indices. Natural spatial variability in fish biodiversity must also be carefully accounted for. Indices that can be calculated for a large number of sites cross New Zealand are likely to remain based on presence-absence data only, as the effort required to collect abundance data will limit the availability of indicators based on fish species abundance.

Freshwater-dependent birds are also highly valued, and therefore of particular interest for impact indicators. Like fish, they are at the top of the food chain, and therefore can be considered ecological “end-points”, making them suitable as indicators of impact. Despite their interest to the public, they have not been monitored in a systematic way as other organism groups have, and lack of data meeting Tier 1 criteria appears to limit their use as indicators of impact. Nevertheless, conservation status of individual bird species has been used successfully as an impact indicator in marine and land domain and biodiversity cross-domain reports. Conservation status of freshwater-dependent species could be used in a similar way in the freshwater domain report.

Measures of ecosystem function provide an assessment of stream health that complements indicators based on structural measures of biotic communities. Simultaneous use of both structural (biodiversity) and functional (ecosystem process) indicators would ensure that a more complete picture of stream health is being captured. The Stream Ecological Valuation is a composite index that combines fourteen ecological functions related to water flows, biogeochemical processes, habitat provision and biodiversity. As a broad measure of stream ecosystem health indexed against natural condition, it has several features that make it suitable as an indicator of human impact. Unfortunately it is used for routine State of Environment monitoring in only one region, so for reasons of data availability it may be disqualified as an indicator for national-scale environmental reporting until it becomes more widely used.

Gross primary productivity and ecosystem respiration are two processes that relate to the energy source for the stream food web. They are influenced by a range of natural and anthropogenic factors, some of which have opposite effects to others. Nevertheless, predictable relationships with catchment land use intensity have been established, and further research may improve understanding of the mechanisms of response. Both processes are relatively easily measured by deploying continuously recording dissolved oxygen sensors over 24-hour periods. In addition, in contrast to the biodiversity indicators, they can be applied in both large (non-wadeable) rivers and wadeable streams. Frameworks for interpreting results are established, but could be further refined to account for variation among stream and river types. An index more responsive to land use stress than simple values of GPP and ER could be developed with modest investment. Despite these advantages, the labour costs of deploying, retrieving and maintaining oxygen loggers *in situ* probably mean that datasets for these two ecosystem processes will remain limited to a relatively small proportion of council monitoring sites.

1 Introduction

1.1 Background

1.1.1 Environmental reporting act

Under the Environmental Reporting Act (2015), the Government Statistician (Stats NZ) and the Secretary for the Environment (Ministry for the Environment) report on state of New Zealand's environment across five domains – air, atmosphere/climate, land, marine and freshwater. One domain report is published every six months, with summary report across all domains once every three years. Thus for each domain one report is produced every three years. The first freshwater domain report was published in 2017 (MfE & Stats NZ 2017), and the next will be produced in 2020, based on data to 2019.

The freshwater domain covers the water that runs across and under land areas. Fresh water relates to all the physical forms of unsalted water, including those in rivers, lakes, streams, ponds, wetlands, aquifers, and glaciers. The freshwater domain excludes atmospheric water (included in the atmosphere and climate domain), sea water (included in the marine domain), and geothermal water (MfE & Stats NZ 2018a,b).

1.1.2 The Pressure-State-Impact framework.

According to the Environmental Reporting Act (2015), each domain report must describe:

(a) the **state** (or condition) of the domain, i.e., the physical, chemical, and biological characteristics of the environment, including biodiversity and ecosystems dependent on that domain. The report must also include changes to the state of the domain over time, and how the state of the domain measures against national or international standards

(b) the **pressures** (natural and human influences) that may be causing, or have the potential to cause, changes to the state of the domain

(c) the **impacts** that the state of the environment and changes to the state of the environment may be having, i.e., what the condition of the environment means for us.

According to the Environmental Reporting Regulations (MfE & Stats NZ 2016), impacts must be reported in each of the following **impact categories**:

(i) ecological integrity (biodiversity and ecosystem processes), and

(ii) public health, and

(iii) the economy, and

(iv) te ao Māori (Mātauranga Māori, tikanga Māori, and kaitiakitanga), and

(v) culture and recreation.

1.1.3 Indicators currently used in the freshwater domain

According to the 2017 Freshwater Domain Report (Ministry for the Environment & Stats NZ 2017),

> *"We currently have limited data on the pressures and impacts related to water quality, particularly monitored data at a national scale."*

In this report, the only indicators for impacts on biodiversity and ecosystem processes were conservation status of native freshwater plants (macrophytes and algae), fish and macroinvertebrates (see Appendix A). This indicator was classed as a "case study" statistic, which is defined as

> *"[relating] to areas that represent only part of the national situation, may not be as accurate as desired due to methodological reasons, or only provides partial information about a topic. A case study, at the least, is reasonably relevant to a particular topic." (MfE & Stats NZ 2018a)*

It is a lesser status than "National Indicator" which is defined as

> *"representative of the national situation and is highly accurate. A national indicator is directly relevant to a particular topic."*

Therefore, Ministry for the Environment is seeking new indicators of impacts on biodiversity and ecosystem processes for use in the next freshwater domain report.

1.1.4 Tier 1 statistics

Indicators used as "National Indicators" (see Appendix A) in environmental reporting must be of sufficient quality to be used as Tier 1 statistics, meaning they are based:

- on accurate and relevant data, that is nationally representative (or includes multiple regions)
- on nationally representative data or data that is collected in a consistent fashion to allow for comparisons across multiple regions
- on timely data, which is not outdated (i.e., more than 3 years old), and can be updated on a regular basis
- on data that can be easily communicated or interpreted, and
- where possible based on historical data that allows for trends to be assessed

1.2 Purpose of this report

The purpose of this report is to build on and improve the suite of "impact" indicators that Ministry for the Environment and Statistics NZ will use in future freshwater domain reports under the Environmental Reporting Act (2015). The aims are to recommend a small number of potential indicators for impacts on river and stream freshwater biodiversity and ecosystem processes and describe how they can be calculated and used in environmental reporting.

The report

- identifies the most useful indicators from the range of possible indicators
- describes how these indicators can be used and interpreted within the Pressure-State-Response framework
- describes the underlying data and where the data can be sourced
- comments on the quality of the data, and where appropriate, recommends how data quality could be improved in future
- describes the method for calculating the indicators from the data
- outlines the steps required to develop the indicators in cases where they are not yet ready for use in reporting.

1.3 Scope

Although impact indicators are needed for all types of freshwater ecosystem, the scope of this report is limited to rivers only, except in the case of birds, for which lake- and wetland-dependent species are also considered.

The authors have considered only indicators based on data that currently exist, or could realistically be collected in future, in New Zealand.

This report also excludes impact indicators relating to Mātauranga Māori because these are being developed within a different impact category. We would encourage interaction between the present work and the Mātauranga Māori work at a later date because the impact indicators outlined in the present report will overlap and align with some of the indicators relating to the Mātauranga Māori impact category.

1.4 Methods

In each section below the method used to develop the recommended indicators was essentially the same. The authors undertook a literature review and consulted relevant experts in the fields to identify and assess potential indicators. Then recommendations were made based on the authors' professional opinion.

2 Periphyton

2.1 What is periphyton?

Periphyton is the layer of biological material that grows attached to surfaces in freshwaters. The term periphyton usually refers to algae in rivers, including benthic cyanobacteria. Periphyton ranges from thin biofilms comprising mostly bacteria and/or diatoms to thick layers of macroalgae, variously described as mats, filaments, or sludge. Depending on the freshwater type and its physico-chemical condition, thick layers can include many different types of algae (including cyanobacteria) as well as bacteria and other heterotrophic organisms.

Periphyton is the main source of primary production in systems not dominated by macrophytes. As the base of the autotrophic (photosynthesis-based) food web, periphyton forms food for higher trophic levels and is consumed by macroinvertebrates which are then consumed by fish and birds.

Still-water bodies (ponds, lakes, reservoirs) also support periphyton in shallow areas where there is sufficient light, but this project is restricted to the riverine environment.

2.2 What does a periphyton indicator tell us?

Periphyton in rivers is both a component of biodiversity and a driver of biodiversity in other trophic levels. Periphyton is also an indicator of physico-chemical conditions in freshwaters.

2.2.1 Periphyton as a driver of biodiversity

The role of periphyton in influencing biodiversity in higher trophic levels can usually be linked to its total abundance. Abundance (i.e., biomass) can be conveniently measured as total chlorophyll *a*. All types of algae (including cyanobacteria) contain this pigment, which therefore reflects the total amount of live primary producing organisms in a sample. Chlorophyll *a* on the beds of rivers varies over time as river flows change (see below). Peak annual chlorophyll *a* concentrations greater than 200 mg/m^2 are generally considered high and < 50 mg/m^2 low (Biggs 2000; NZ Government 2017). High biomass is reflected by the appearance of periphyton, and comprises visible cover of river beds by thick algal mats or filamentous green algae or cyanobacteria.

High periphyton biomass can alter the quality and diversity of macroinvertebrate communities by modifying physical habitat conditions and altering water quality. For example, dominance of periphyton by filamentous algae creates pockets of slow-moving water, that may be poorly-oxygenated at night and favour populations of "tolerant" macroinvertebrates[1] (Suren et al. 2003a,b). High periphyton abundance also leads to large diurnal fluctuations in dissolved oxygen and pH, which can have negative effects on macroinvertebrates and fish (Klose et al. 2012), especially in the presence of high concentrations of ammonium, which converts to toxic ammonia at high pH (Hickey 2014). Proliferations of the introduced nuisance alga *Didymosphenia geminata* (didymo) apparently drives changes to macroinvertebrate communities in affected South Island rivers through changing habitat conditions (Kilroy et al. 2009).

Low periphyton abundance is usually seen as dominant cover by either thin films or no visible algae, with only occasional cover by thick mats and/or filamentous algae (e.g., in occasional patches). Thin algal films and consistently well-oxygenated water favour diverse communities of "high quality" native macroinvertebrates such as the larvae of mayflies, stoneflies and caddisflies (Suren et al. 2003a). The threshold of 50 mg/m^2 chlorophyll *a* in previous and current periphyton guidelines was based on deterioration of macroinvertebrate biodiversity above this level (on average) (Biggs 2000, Matheson et al. 2012).

[1] "Tolerant" invertebrates refer to those that tolerate conditions considered to represent poor water quality and poor habitat.

Figure 2-1: Examples of stream periphyton. From top left, clockwise: *Microspora* filaments overlying rocks within thin diatom film; mat of the cyanobacterium *Phormidium*; tufts of *Cladophora* attached to a rock; didymo mats.

2.2.2 Periphyton as a component of biodiversity

Periphyton usually comprises a rich diversity of different algal taxa. Taxa commonly found in New Zealand freshwaters represent at least 100 genera and many more species (Biggs & Kilroy 2000). Diatoms (Bacillariophyta) contribute most taxa to periphyton and many species are easily identifiable using microscopy. Green algae (Chlorophyta) and Cyanobacteria are more difficult to identify to species level and may require cultures and molecular techniques to confirm species identities (e.g., Novis 2003, 2004). A few genera of red algae (Rhodophyta) are widespread and probably include many different species (e.g., Entwisle & Foard 1997).

Most algal taxa in the periphyton of New Zealand rivers appear to be cosmopolitan species and are therefore not a focus of concern in the protection of biodiversity (e.g., Harper et al. 2012). Nevertheless, some common diatoms have been reported to be endemic to New Zealand. For example, morphological studies on the mucilage/sludge producing diatom *Cymbella kappii* (re-identified as *C. novaezelandiana*, Krammer 2002) and the stalked diatom *Gomphoneis minuta* var. *cassieae* (Kociolek & Stoermer 1988) suggest that these common species are restricted to New Zealand, although they are closely related to taxa from elsewhere. A few truly endemic and distinctive taxa do exist in undisturbed habitats such as oligotrophic, acidic peat bogs (e.g., Sabbe et al. 2001, Kilroy et al. 2006, 2007) and their protection will depend on protection of those habitats.

While cosmopolitan species, by definition, likely arrived from other locations, at this stage didymo is the only algal species in rivers that we are certain has been introduced to New Zealand in recent times (Kilroy & Novis 2018). In addition to driving changes in macroinvertebrate diversity, the introduction of didymo may have affected biodiversity of the periphyton itself. Analyses of periphyton data from a Southland River before and after the arrival of didymo suggested that didymo has created a more homogeneous community by reducing within-river patchiness caused by localised variability in hydraulic conditions (Kilroy et al. 2009).

2.2.3 Responses by periphyton to physico-chemical conditions

Increasing environmental pressure on landscapes (as indicated by the percentage of high-intensity agricultural land-cover in a catchment) leads to increasing concentrations of dissolved inorganic nitrogen (DIN) and dissolved reactive phosphorus (DRP) in rivers (Larned et al. 2016). An expected consequence of increased DIN and DRP concentrations is stimulation of algal growth rates, which potentially results in higher periphyton abundance. The effect of increasing DIN and DRP on increasing periphyton biomass has been demonstrated many times in controlled experiments (e.g., Biggs et al. 1999, Liess et al. 2009). However, the effect of enrichment by DIN and DRP on biomass is less clear in natural river habitats because periphyton abundance is driven by a combination of factors that lead to algal losses and algal growth (Biggs 2000) (Figure 1).

Natural catchment features can also influence nutrient availability. For example, naturally high DRP in the central North Island is attributable to the particular type of volcanic geology in the area, which results in water chemistry that facilitates release of P into the water column (Timperley 1983).

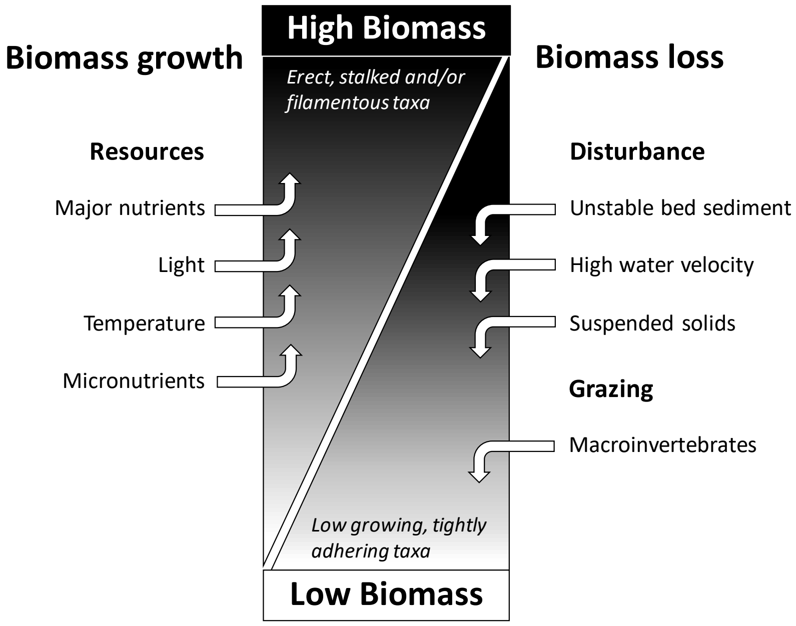

Figure 2-2: Conceptual model of processes that drive periphyton biomass growth and biomass loss, which together determine the rate of accrual. The relative strength of growth and loss processes may also influence the type of algae that dominates periphyton, shown at the low and high biomass extremes of the range (diagram adapted from Biggs 1996).

Periphyton losses are caused mainly by removal of biomass during high flows (through shear stresses and abrasion from mobile substrate) and by macroinvertebrate grazing. The effects of high flows on periphyton removal vary across rivers and are largely driven by the geomorphological characteristics at a site. In general, the threshold flow for removing periphyton is related to the magnitude of the flow that mobilises fine river bed sediments (Francouer & Biggs 2006), and the frequency of such events is a dominant control on potential biomass (Hoyle et al. 2016). At sites where removal events are infrequent (e.g., occurring for less than 10% of the time), growth-promoting factors such as nutrient enrichment become important (Hoyle et al. 2016).

The second cause of periphyton loss, macroinvertebrate grazing, almost always exerts a negative effect on periphyton biomass (Liess & Hillebrand 2004). Grazing occurs during periphyton accrual, and can be substantial. For example, in one experiment, excluding grazers resulted in increased periphyton biomass of >55% (Taylor et al. 2002). In a very productive stream, macroinvertebrate grazing reduced periphyton up to 60-fold over an accrual period of 16 days (Sturt et al. 2011). Higher trophic levels (fish) may also affect periphyton indirectly by reducing the density of macroinvertebrate grazers. For example, the presence of efficient predators of macroinvertebrates (trout) has been associated with higher periphyton biomass than the presence of less efficient predators (*Galaxias*) (Biggs et al. 2000). The New Zealand freshwater fish fauna lacks species that graze directly on periphyton, as occur commonly elsewhere (e.g., Power 1990). However, mullet (yellow-eyed and grey mullet) migrate upstream from their coastal habitat into the lower reaches of some rivers in summer (McDowall 1990) where they are likely to exert top-down control of periphyton biomass.

Periphyton growth is promoted by:

- adequate supplies of nutrients (higher DIN and DRP concentrations promote faster growth rates, up to the point where nutrients are saturating)

Saturating concentrations of DIN and DRP probably vary across rivers, regions and time because other factors can limit periphyton growth rates and standing crop. The effect of an increased supply of each nutrient also depends on an adequate (non-limiting) supply of the other. For DIN, saturation of nitrate-N uptake (from all sources, including periphyton) has been observed from 310 – 370 mg/m^3 (Matheson et al. 2012) to 760 mg/m^3 (Hoellein et al. 2007), though the true range may be wider.

For DRP, excessive chlorophyll *a* (e.g., > 200 mg/m^2) can occur when DRP concentrations are very low ~2 mg/m^3 (e.g., Kilroy & Wech 2016), possibly reflecting processes that enable periphyton to access phosphorus through internal recycling from phosphorus attached to sediment within periphyton mats (Dodds 2003; Wood et al. 2015). In experiments, a DRP concentration of ~28 mg/m^3 corresponded to peak chlorophyll *a* of approximately 350 mg/m^2 (Bothwell 1989), and this may be a realistic estimate of a saturating concentration for thick periphyton. However, cell division rates of diatoms in thin films appeared to saturate at very low concentrations (<2 mg/m^3) (Bothwell 1988).

Responses by periphyton to nutrient concentrations well in excess of saturating concentrations are unpredictable. Periphyton biomass has been observed to respond positively to point-source nutrient enrichment of DIN and DRP of, respectively, ~5500 and 50 mg/m^3 (Sturt et al. 2011). On the other hand, in experimental channels, total cell densities (a direct measure of algal biomass) responded to a nutrient gradient (from 36 to 6900 mg/m^3 N and 1.4 to 466 mg/m^3 DRP) with a subsidy-stress pattern, initially increasing, then declining at the highest concentrations (Wagenhoff et al. 2013). Lower than expected chlorophyll *a* at DIN > 1000 mg/m^3 has also been observed in Canterbury rivers (Kilroy et al. 2017).

Periphyton growth typically involves uptake of DIN and DRP from the water column and reduces the nutrient concentrations in the river water as it flows downstream, where eventually nutrients become growth-limiting and periphyton biomass declines (Chapra et al. 2014). Hence, both the periphyton biomass and the spatial extent of high biomass along rivers provide information on nutrient enrichment.

- adequate light (growth rates are usually lower under shaded conditions)

Most of the algae and cyanobacteria in periphyton are photoautotrophs (i.e., they depend on sunlight for photosynthesis and growth); thus, increasing light by removing shading vegetation increases periphyton growth potential. In a survey of North Island streams, periphyton standing crop exceeding 100 mg m^{-2} only occurred at sites where the incident light was >3% of unobstructed sky light (Davies-Colley & Quinn 1998). Incident light of less than 3% required heavy shading by tall trees, and such complete shading occurred only at stream sites where the waterway width was less than 4.5 to 5.5 metres (depending on tree height). Other work indicates that nuisance proliferations of periphyton can be controlled if average reach shading exceeds 60-65% of that in the open (Quinn et al. 1997b, Biggs 2000, Matheson et al. 2017b). The over three-fold variation in average daily solar radiation between mid-winter and mid-summer (Mitchell 1991) is also likely to contribute to seasonal variation in periphyton biomass.

- suitable temperatures (growth rates increase with temperature, within certain limits)

Over the course of a year, growth rates in periphyton are influenced strongly by seasonal variations in temperature. Growth rates of a diatom-dominated periphyton in summer (at 20 °C) were over three times those in winter (at 0.5 °C) (Bothwell 1988). Optimum temperatures for photosynthesis and growth in freshwater algae are species specific, ranging from 10 to 30 °C (Butterwick et al. 2005). Diatoms are generally favoured by cooler temperatures (5-20 °C), green algae by moderate temperatures (15-25 °C) and cyanobacteria by high temperatures (>30 °C) (He 2010). High temperatures can also influence periphyton biomass by reducing invertebrate grazing (Rutherford et al. 2000).

- time (periphyton abundance increases during flood-free periods when algal losses from high flows are minimal)

Accrual of periphyton over time generally follows a predictable pattern of exponential growth (e.g., Bothwell 1988). Time to attain maximum biomass depends on the interacting effects of nutrient concentrations, temperature and light (see above) and potential for removal of biomass by macroinvertebrate grazing (see above), and has been reported to vary between 7 and 90 days (Biggs et al. 2005). A typical pattern after long accrual is that biomass can be lost through natural sloughing, presumably caused by weakening of the mat as the basal cells become deprived of resources by overgrowing material (Biggs & Stockseth 1996, Biggs 2000). The point at which this happens defines a maximum “carrying capacity” for periphyton biomass under the prevailing conditions (Biggs et al. 2005).

- micronutrients and major ions

Superimposed on the effects of DIN, DRP, light, temperature and time is the probable influence of other chemical solutes in rivers, which are essential for algal growth. Micronutrients include a wide range of metals (e.g., iron, cobalt). The major ions (calcium, magnesium, sodium, potassium, chloride, sulphate) may also affect growth and community composition. Major ion concentrations are reflected in measures of water conductivity. Recent analyses of Regional Council datasets (e.g., Kilroy et al. 2017) have reconfirmed a pattern recognised in New Zealand over 30 years ago (Biggs & Price 1987), that water conductivity is a better predictor of periphyton biomass (positive correlation) than nutrient concentrations. Where DIN concentrations are high (> 1000 mg/m^3), conductivity is generally correlated with DIN. However, at lower DIN, conductivity is independent of DIN, and the major ions and water alkalinity (concentrations of carbonate and bicarbonate) may drive differences in biomass through influencing community composition. For example, many diatoms species occur preferentially in calcium or sodium-dominated waters (Potapova & Charles 2003). Differences in conductivity between rivers are driven by the geological setting of the catchment. For example, limestone and soft marine sedimentary rocks tend to lead to higher conductivity than hard rocks such as granite and greywacke. Proximity to the coast can also increase conductivity through sodium inputs.

Periphyton community composition can also be influenced by DIN and DRP concentrations. Some taxa thrive only when nutrients are plentiful (e.g., the green filamentous alga *Stigeoclonium* is usually abundant only where DIN > ~400 mg/m^3 and DRP > 10 mg/m^3, such as below nutrient rich discharges, C. Kilroy (author) personal observations), and others do best in low nutrient-environments (e.g., the diatom *Didymosphenia geminata* is rarely recorded as visible biomass where DRP concentrations exceed 2 mg/m^3, Kilroy & Bothwell 2012). Benthic cyanobacteria are often nitrogen-fixers and some diatom genera (*Epithemia*, *Rhopalodia*) can access nitrogen through N-fixing symbiotic cyanobacteria. These N-fixing organisms are typical in rivers with low dissolved nitrogen concentrations (e.g., < 10 mg/m^3). Such associations with specific ranges of nutrient concentrations are the basis of algal indices used in other countries (e.g., Schneider & Lindstrøm 2011). Species – environment relationships specific to New Zealand have not been developed (see below).

Periphyton community composition is also influenced by substrate composition and local hydraulic conditions (i.e., water velocity and turbulence). For example, large stable boulders and fast flows favour growth of tightly attached algal species such as *Ulothrix* and *Cladophora* species that resist removal by all but the largest floods. More mobile substrates and deposits of fine sediment favour dominance by fast-growing motile diatoms such as *Navicula* and *Nitzschia*.

In summary, responses by periphyton biomass (chlorophyll *a*) to river state can be distilled down to a simple conceptual model (Figure 1). However, the catchment pressures that determine river state (i.e., nutrient enrichment, flow alteration) may overlap with natural processes so that the catchment pressure – biomass relationship is not always linear. Biomass at two sites can be similar even if physico-chemical conditions differ. In such cases, however, community composition usually differs, and can be used to distinguish between the two sites when biomass cannot. This suggests that a sensitive periphyton indicator of impacts on riverine freshwater biodiversity would incorporate both biomass and taxonomic composition.

2.3 Data availability

2.3.1 Periphyton abundance

Data on periphyton abundance are now available from most Regional Councils. To our knowledge, the following councils carry out ongoing monthly data collection at multiple sites as part of State of Environment monitoring in their regions: Northland, Bay of Plenty, Hawkes Bay, Horizons, Greater Wellington, Tasman, Canterbury, Southland. There is confidence that data collected by different councils are comparable to one another because standard methods for sampling periphyton and carry out visual assessments of cover have been available since 2000 (Biggs & Kilroy 2000). Regional Councils have largely adopted these methods directly, or have developed region-specific monitoring plans that are largely drawn from the 2000 manual (e.g., Kilroy et al. 2008).

In most cases, periphyton data collection comprises both estimation of mean chlorophyll *a* at a site, from collection of quantitative samples and visual estimates of periphyton cover on the river bed. Both measures are useful. Chlorophyll *a* provides a composite measure of the abundance of all algae combined and allows comparison with national guidelines. Percentage cover captures the visual impact of periphyton and is included in regional objectives by some Regional Councils. Percentage cover also provides an idea of community composition and allows identification of the extent of nuisance taxa such as green filamentous algae, didymo and the cyanobacterium *Phormidium*.

Periphyton data are generally not collected in Waikato and Auckland regions because most streams there are soft-bottomed and support macrophytes rather than periphyton. In the country as whole, soft-bottomed streams are estimated to make up over 25% of the stream length of New Zealand streams (as defined in the network of the River Environment Classification) (Snelder et al. 2013).

While national data coverage for periphyton abundance is quite broad, most datasets of monthly data are relatively short (four years or less). The longest is that collected by Horizons Regional Council, which commenced data collection in late 2008 (Kilroy et al. 2010).

2.3.2 Periphyton taxonomic composition

Data on the detailed taxonomic composition of periphyton in New Zealand are sparse compared to the coverage of abundance data. Annual community composition data were collected in some regions (e.g., Southland, Horizons) starting in the 1990s. However, as far as we are aware, the data were never analysed quantitatively (e.g., in analyses aiming to link species composition with nutrient concentrations or flow conditions). More recently samples from Canterbury (24 sites) and the Horizons region (> 60 sites) have been analysed for taxonomic composition for research purposes (e.g., Wagenhoff et al. 2017). Detailed taxonomic data also exist for 36 Northland rivers included in the current monthly monitoring programme.

There is currently no comprehensive, nationwide dataset of periphyton community composition. However, the samples to compile such a dataset, at least for the diatom component of periphyton, do exist. This sample collection was assembled in 2005, taking advantage of the nationwide delimiting survey to establish the distribution of didymo before it became widespread in the South Island. Every major catchment in New Zealand was sampled, with the exception of parts of Northland (Duncan et al. 2005). The survey included many pristine headwater locations as well as downstream reaches. Approximately 700 samples are archived in the NIWA diatom collection, Christchurch. This dataset (if developed) could serve as baseline data for future comparative studies. Limited environmental data were collected with the samples, but modelled data for a range of variables are available for all REC reaches.

2.4 Development level of periphyton indicators

2.4.1 Periphyton abundance

To be used as an indicator of impacts on biodiversity, periphyton abundance must be reported in relation to guideline values and/or reference condition and have a clear and known relationship with pressures. In addition, the indicator must be able to account for natural variability over time and among sites.

In the New Zealand Periphyton Guideline (Biggs 2000), thresholds of periphyton abundance (as chlorophyll *a*, ash-free dry mass[2], and percentage cover of the streambed) were proposed for the protection of a range of instream values. One of the values was benthic macroinvertebrate biodiversity, or "maintenance of clean-water benthic fauna and benthic [macroinvertebrate] biodiversity" (Biggs 2000). The proposed guideline was a maximum chlorophyll *a* of 50 mg/m^2, and mean monthly chlorophyll *a* of less than 15 mg/m^2 (presumably over a year).

While not explicitly stated by Biggs (2000), the guideline for benthic [macroinvertebrate] biodiversity is likely to define something close to a reference condition for periphyton applicable across New Zealand. In both the Manawatu-Whanganui and Canterbury regional datasets, sites with negligible intensive farming or exotic forestry (i.e., < 5% of the catchment) usually have mean chlorophyll *a* of less than about 15 mg/m^2, consistent with the Biggs (2000) guideline. However, exceptions to this occur in some circumstances. For example, lake-fed rivers tend to have relatively stable flows and stable bed sediments with little of the fine material that initiates periphyton removal (Hoyle et al. 2017) or negatively affects macroinvertebrates (Wagenhoff et al. 2013). Consequently, accrual periods are long and periphyton biomass can be persistently high even in a pristine catchment (e.g., winter biomass of > 70 mg/m^2 in the upper Mararoa River, Southland, downstream of Mavora Lakes (pre-didymo, Kilroy et al. 2006). Modification of river flows in hydro-electric power or irrigation schemes can also have the effect of extending the time between floods, leading to increased periphyton biomass (e.g., a long-term problem in the Lower Waiau River, Southland, Kilroy 2017). High periphyton biomass can also occur naturally in rivers draining catchments with calcareous geology (e.g., the Waipara River, Canterbury, Suren et al. 2003b), and in headwater streams enriched with phosphorus inputs originating from particular types of volcanic geology (Timperley 1983).

Positive correlations between periphyton chlorophyll *a* and a gradient of catchment land use impact (as indicated by the proportion of a catchment under pasture or urban development) have been repeatedly demonstrated in New Zealand and overseas (e.g., Busse et al. 2006, Niyogi et al. 2007). In the Manawatu-Whanganui region, highest mean chlorophyll *a* (annual mean of >50 mg/m^2, from monthly surveys) tends to occur at sites with more than 60% of the catchment under intensive agriculture, and these sites usually also have high percentage cover of nuisance cyanobacteria (*Phormidium*) and highest concentrations of DIN. In both Manawatu-Whanganui and Canterbury mean annual DIN is strongly correlated with % intensive agriculture (at least 50% of variance in DIN explained), and DIN is also a strong driver of periphyton biomass, although the effects of other environmental conditions must be accounted for to produce the strongest predictive relationships (Kilroy et al. 2016).

---

[2] Ash-free dry mass (AFDM) is a measure periphyton abundance that includes all organic material, not just living photosynthetic material. It includes structural parts of periphyton mats such as diatom stalks and mucilage, and the thick cell walls of some filamentous algae. It also includes dead cells and other organisms, such as small invertebrates, that live within the mat. Chlorophyll a and AFDM are often strongly correlated within and across sites, but the presence of taxa that produce a lot of non-photosynthetic organic material (such as didymo stalks) can weaken the correlation.

Periphyton abundance (as chlorophyll *a*) is already used as an indicator of stream ecosystem health (i.e., the periphyton attribute in the National Policy Statement for Freshwater Management (NPS-FM), New Zealand Government 2017). Definitions of the Bands in the periphyton attribute were based on the chlorophyll *a* thresholds recommended by Biggs (2000). The periphyton – macroinvertebrate relationships were re-confirmed using new data, and were found to be consistent with the existing recommendations (Matheson et al. 2012, 2015).

In the NPS-FM, variability of periphyton over time (due mostly to flow variability) is accounted for by using a metric that allows limited exceedances of thresholds over a three-year period. Assuming monthly sampling over a period of 36 months, a maximum of three exceedances of the thresholds (50, 120 or 200 mg/m^2 chlorophyll *a*, for no more than 8% of the time) places the site in attribute state bands A, B, or C, with band D defined as more than 8% exceedance of 200 mg/m^2. Naturally-productive rivers (defined by as certain source of flow and geology combinations in in the REC) are allowed six exceedances. Use of such metrics recognises that prolonged high abundances are more damaging to instream communities than occasional brief exceedances. Development of the thresholds and metrics is explained in detail by Snelder et al., (2013). Although chlorophyll *a* is specified as the biomass measure, MfE's guide to attributes in the NPS-FM allows for some monitoring to be carried out using visual estimates of cover (MfE 2017). Cover data provides basic information about the type of algae present, and it is often possible to estimate chlorophyll *a* from cover data (Kilroy et al. 2013). Weighted composite cover (WCC) combines cover by mats and filaments into a single index, and has been shown to be correlated with invertebrate community indices (Matheson et al. 2012). Targets for percentage cover by filaments and/or mats are also specified in regional plans (e.g., Horizons, Environment Canterbury).

The NPS-FM attribute does not specifically allow for the effects of upstream lakes, which may lead to high periphyton biomass through excessively high nutrients and relatively stable flows.

The discussion so far applies predominately to hard-bottomed systems (i.e., gravel-bed rivers and streams). In soft-bottomed streams, the main surfaces for periphyton growth are provided by macrophytes and wood. The ecological effects of periphyton growing on macrophytes (more correctly termed epiphyton) have received less attention than the effects of periphyton on rocks. Additionally, methods for the quantitative measurement of epiphytic biomass (as chlorophyll *a*) are not clearly defined (but see Matheson & Wells 2017). Because of the limitations in assessing epiphyton, the states for the NPS-FM periphyton attribute were developed from a dataset limited to hard-bottomed systems. Therefore, MfE (2018) recommends that the NPS-FM periphyton attribute should be applied only to hard-bottomed streams and rivers. More than 25% of stream length across New Zealand is soft-bottomed and likely to support macrophytes (Snelder et al. 2013). For these stream reaches, a periphyton impact indicator requires further work to develop quantitative assessment protocols and guideline values.

Except in soft-bottom and lake-fed streams and rivers and in naturally enriched rivers, periphyton abundance is well developed as an indicator of impact, as it can be reported in relation to guideline values or reference condition, has a known relationship with landuse pressures and can account for natural variability over time and among sites. Table 2-1 shows where periphyton biomass could be used effectively as an impact indicator in relation to various pressures.

2.4.2 Periphyton taxonomic composition

In other parts of the world, especially Europe, the use of periphytic algae in stream assessments has focussed almost entirely on the taxonomic composition rather than biomass as an indicator. Under the European Water Framework Directive (WFD: see Kelly 2013) all EU countries are obliged to assess river state using several indicators, including “phytobenthos” (i.e., periphyton). A further requirement is to report state relative to a reference condition. Most EU countries have fulfilled these requirements by adopting or developing diatom indices for detecting gradients of nutrients (usually phosphorus) or organic pollution (Besse-Lototskaya et al. 2011). A drawback of the use of indices is that links to pressures or state (other than the nutrient concentration used to develop the index) are unclear, and therefore the index results cannot easily be used as measures of ecological impact. For this reason, the use of diatom indices in isolation for assessing river health is now being questioned by EU scientists (Kelly 2013, Poikane et al. 2017). In fact, there is now a move to combine biomass indicators with taxonomic indicators (Kelly 2013), which could be a good approach in New Zealand in relation to an indicator of biodiversity.

In New Zealand the use of periphyton taxonomic composition as an indicator of stream health /integrity has not been tested, although localised studies have shown that diatom community composition accurately reflects water conductivity and pH (e.g., Kilroy et al. 2006, Schowe et al. 2013). More general observations suggest that confounding factors in the generally positive relationship between periphyton biomass and the consequences of catchment pressures such as nutrient enrichment and altered river flows could be resolved by taking community composition into account. A simple example is shown in Box 1.

Detailed lists of the tolerances of periphyton taxa to nutrient concentrations and other aspects of water quality have been compiled overseas (Van Dam et al. 1994, Kelly et al. 2008, Rott & Schneider 2014). These lists are likely to be largely applicable in New Zealand for the cosmopolitan taxa. Sufficient data currently exists to test them in at least two regions (Manawatu-Whanganui, Canterbury, and possibly Northland). The 2005 nationwide sample collection could also be used to look at national patterns, but this would require considerable sample analysis effort.

Another approach to using community composition as part of a biodiversity indicator would be to define indicator species rather than adopt the use of full indices (which require more effort in identifying many taxa). This approach was used recently in an analysis of diatom communities in wetlands (Kilroy et al. 2017). Several diatom species were identified as being characteristic of high or low conductivity, and high DRP.

Table 2-1 shows where periphyton taxonomic composition could be used effectively as an impact indicator in relation to various pressures.

BOX 1. Lake outlet versus lowland river periphyton

Periphyton biomass is often naturally elevated in lake outlet rivers, relative to equivalent rivers without lakes upstream. For example, mean chlorophyll *a* of 93 mg/m^2 was measured in April in the Gowan River, 3 km downstream from the outlet from Lake Rotoroa, Nelson Lakes National Park. Similar biomass (80 mg/m^2) was measured in the Opihi River, a partially regulated river in Canterbury, in March. The Opihi River is in a largely pastoral agricultural catchment.

Community composition at the two sites, however, was non-overlapping. The Gowan River community comprised taxa typical in oligotrophic lakes (the green alga *Bulbochaete*, branched cyanobacteria, *Epithemia* spp.) while the Opihi River sample comprised nutrient tolerant taxa such as *Stigeoclonium*, *Melosira*, *Diatoma vulgaris* and small motile diatoms.

The Gowan river taxa are characteristic of low nutrient, undisturbed environments, while the Opihi River taxa are tolerant of high nutrient environments, and also tolerate fine sediments.

Thus, the two sites cannot be separated by biomass, but can be separated by community composition.

Table 2-1: Pressure–State–Impact framework for recommended periphyton indicators.

Pressure	State	Impact on periphyton	Appropriate measure
Land use change leading to increased inputs of nutrients.	Enriched DIN and / or DRP.	Increased peak biomass, increased cover of the streambed by mats and / or filamentous algae, possibly including toxic cyanobacteria species. Accompanying effects on DO and pH. Shifts in species composition.	Periphyton chlorophyll *a.* Percentage cover of the streambed by mats and filaments, or WCC. Species composition to distinguish from natural high biomass.
Land use change leading to increased inputs of nutrients.	Highly enriched DIN.	Possibly declining peak biomass, shift in species composition. Increased downstream spatial extent of high biomass.	Periphyton chlorophyll a Species composition to distinguish from natural low biomass.
Land use change leading to increased inputs of fine sediment.	Inputs of fine sediment.	Shifts in species composition.	Species composition.

Pressure	State	Impact on periphyton	Appropriate measure
Flow alteration (loss of flood flows).	Increased time for periphyton accrual.	Increased peak biomass, as above.	Periphyton chlorophyll a Percentage cover of the streambed by mats and filaments, or WCC Species composition to distinguish impacted from naturally high biomass.
Flow alteration (longer periods of low flows).	Increased time for periphyton accrual and possibly increased water temperature.	Increased peak biomass, as above.	Periphyton chlorophyll a Percentage cover of the streambed by mats and filaments, or WCC Species composition to distinguish impacted from naturally high biomass.

2.5 Summary and recommendations

Periphyton biomass, in general, increases in response to changes caused by environmental pressures on catchments from altered catchment land use. These biomass increases have direct and indirect effects on macroinvertebrates and fish, usually leading to declining community quality and diversity. The thresholds in the periphyton attribute in the NPS-FM were defined based on the relationships between periphyton and macroinvertebrate community indices (with low biomass corresponding to high quality macroinvertebrate communities), and are relevant for an indicator of biodiversity. Reference state for periphyton usually corresponds to low biomass. However, exceptions are common because biomass can be naturally elevated in pristine locations such as lake outlets or in catchments with calcareous or high-phosphorus geology. The NPS-FM deals with naturally high biomass by defining productive classes in the REC. An alternative method - using data on community composition – could be more appropriate for a biodiversity indicator.

Recommendations are:

- periphyton biomass (as chlorophyll *a*) can provide the basis for an indicator of biodiversity in rivers
- measures of percentage cover of the stream bed by mats and filaments can be useful indices and are generally related to chlorophyll a
- the thresholds suggested in the NZ Periphyton Guideline (Biggs 2000) may be appropriate for defining a reference state (near pristine conditions) for biomass
- some analysis of existing data would be required to confirm that the NZ Periphyton Guideline threshold provides an appropriate reference state for all regions (as a range of biomass)

- community composition could be used to complement the biomass indicator, to identify sites that do not conform to a positive relationship between periphyton biomass and catchment pressure / state.

A method for incorporating community composition into a periphyton indicator for biodiversity in rivers would require development. As a first step, existing European indices could be applied to the available regional New Zealand data. Alternatively, indicator species could be identified.

3 Macrophytes

3.1 What are macrophytes?

Macrophytes are large aquatic plants often, but not always, vascular (Figure 3-1). They are usually the dominant primary producer in slow-flowing, soft-bottom streams and rivers. There are a variety of different freshwater macrophytes but they can generally be grouped into the following life-form types: erect emergent, sprawling emergent, free-floating, rooted floating-leaved and submerged. Both types of emergent lifeform tend to grow close to the stream bank but sprawlers can also spread out across the water surface as water velocity allows, often using adventitious roots to source nutrients and other elements from the water. Floating species, especially free-floaters, are typically found in slow-moving or back waters. Submerged species tend to grow in those parts of the channel where emergent or floating species are absent and consequently where there is sufficient light penetrating below the water surface to allow them to grow. Characeans (or stoneworts), the primitive ancestors of native land plants, and bryophytes (aquatic mosses and liverworts) are often included as a component of the New Zealand submerged macrophyte flora. Bryophytes require a stable substrate for attachment so, in contrast to other macrophytes, are most commonly found in hard-bottom streams with bedrock and boulders (Reeves et al. 2004).

Figure 3-1: Examples of river macrophytes. Clockwise from top left. Hard-bottom river with patches of aquatic moss adhering to boulders (R.Wells); The introduced sprawling emergent species, *Erythranthe guttate*, in Whangamata stream (F. Matheson); The introduced emergent sprawler, *Persicaria hydropiper* (J. Clayton); The native submerged species, *Potamogeton ochreatus* in flower (J. Clayton); The introduced submerged macrophyte, *Vallisneria australis* (R. Wells); A bed of native charophytes, *Nitella* spp. (R. Wells). The native floating leaved pondweed, *Potamogeton cheesemanii* (P. Champion); The introduced floating aquatic fern, *Azolla pinnata* (T. James).

3.2 What do macrophytes tell us?

The macrophyte communities that occur in rivers and streams reflect the influence of many factors but particularly light availability, flow regime, temperature, substrate type, water chemistry including nutrients, and colonization by introduced species (Booker & Snelder 2012, Matheson et al. 2012). Macrophytes are typically favoured by high light, stable flows, warm temperatures, “soft” sediments (fine particles such as sand and silt) and adequate nutrient supplies, either in water or sediment. Forest clearance and land development in New Zealand has resulted in many streams having reduced canopy cover and shade compared to pre-human times (Howard-Williams et al. 1987, Reeves et al. 2004), as well as increased sediment and nutrient inputs. The proportion of New Zealand’s total stream length with a bed dominated by fine sediments is thought to have increased from around 2% to 20% since forest clearance began (Clapcott et al. 2011, Leathwick et al. 2011). This suggests that the available habitat for macrophytes in our waterways may have increased; although this has likely been offset by reduced light penetration, due to increased water turbidity and possibly phytoplankton growth in larger rivers, limiting growth of submerged species. Prior to human habitation, New Zealand’s lowland streams and rivers with fine bed substrates were likely inhabited by native, and generally shade-tolerant, species of milfoils (*Myriophyllum* spp.,), pondweeds (*Potamogeton* spp.,) and charophytes (*Nitella* spp.,) (Reeves et al. 2004). However, introduced species are often better competitors than native species in well-lit, nutrient-enriched environments (see Ellenberg light and nutrient indicator values, Ellenberg et al. 1988). Together, these factors result in many lowland waterways now being dominated by introduced over native species (Champion & Tanner 2000, Reeves et al. 2004).

Macrophyte communities in rivers and streams can provide valuable habitat, cover and a food source for other stream biota (Sand-Jensen et al. 1989, Collier et al. 1999) and native macrophyte communities also provide biodiversity value in their own right. Furthermore, macrophytes are important regulators of flow and fluxes of carbon, oxygen and nutrients (Franklin et al. 2008, Howard-Williams & Pickmere 2010, O’Brien et al. 2014). At low to moderate macrophyte abundance, this regulation is seen as beneficial for the stream ecosystem. However, prolific growth of introduced species, is often viewed as problematic and they can be an attachment substrate for nuisance epiphytic algae (Matheson and Wells 2017). In particular, there are concerns about high biomass and metabolism of introduced macrophytes contributing to diurnal deficits of dissolved oxygen and elevated pH harmful to fish and macroinvertebrates (Collier et al. 1999), particularly in slow-flowing reaches with limited re-aeration and accumulations of fine sediment and organic detritus that can also contribute to biological oxygen demand (Wilcock et al. 1995). Emergent and floating life-forms may be especially problematic because as well as consuming oxygen during respiration, they do not release oxygen into the water during photosynthesis, as submerged life-forms do (Caraco and Cole 2002). Low night-time dissolved oxygen and high afternoon pH conditions could also conceivably promote phosphorus release from fine sediments trapped by macrophyte beds, contributing to eutrophication (see Matheson et al. 2017b), but this hypothesis is yet to be tested. Conversely, abundant growth of macrophytes can indicate a high potential for nutrient uptake during the growing season, altering the form and timing of downstream nutrient transport. However, the net effect of this uptake on dissolved nutrient concentrations will depend upon the overall flux of nutrients through the system and the fate of the assimilated nutrients when the plants undergo seasonal senescence (see Howard-Williams & Pickmere 2010, O’Brien et al. 2014, McKergow et al. 2016).

A study of a lowland Waikato stream dominated by introduced submerged species showed that macroinvertebrate diversity was reduced with high macrophyte biomass (Collier et al. 1999). An intermediate macrophyte biomass was therefore recommended to enhance macroinvertebrate diversity. This is consistent with overseas guidance suggesting that the abundance of macrophytes in a half-shaded channel (approximately half the abundance that occurs with no shade) was likely to be optimal in lowland streams and rivers (Dawson & Kern-Hansen 1979, Dawson & Haslam 1983). There are, as yet, few detailed studies of interrelationships between macrophytes, dissolved oxygen and other stream biota in lowland waterways. However, a compilation of data for lowland Waikato streams (from Collier et al. 1999 and Wilcock et al. 1999) suggested that a macrophyte cover >35% may be associated with daily dissolved oxygen minima below guidelines recommended for fish protection (Matheson et al. 2012). Considering all of the above findings, provisional macrophyte guidelines were recommended for New Zealand streams and rivers at no more than 50% channel volume and 50% water surface cover, as summer maxima, to protect a range of values including ecological condition (Matheson et al. 2012). The provisional status of these guidelines reflects the need for further testing and evaluation.

3.3 Data availability

Virtually all regions of New Zealand having at least some, and often many, soft-bottom rivers and streams. However, only a few regions presently have State of Environment (SoE) monitoring programmes that assess macrophytes beyond simple estimates of percent cover or volume using the stream habitat assessment (Harding et al. 2009) or Stream Ecological Valuation protocols (Storey et al. 2011). This is surprising given that macroinvertebrate communities are routinely sampled in soft-bottom rivers and streams (Stark et al. 2001), and in view of the contribution that macrophytes invariably make to the ecology, biodiversity and biogeochemistry of these systems. In a survey of Regional Councils, 84% of respondents indicated that lotic macrophyte growth was of concern in their region but only 8 of 14 Councils were undertaking some form of macrophyte monitoring (MfE 2016).

Waikato Regional Council presently have the most comprehensive monitoring programme for stream and river macrophytes. Their 10+ year monitoring programme, focusing on wadeable streams, quantifies the percent cover of different macrophyte life-forms (emergent, floating, submerged) and calculates a percent channel volume (clogginess) occupied by macrophytes and a percent cover by native species (Collier at al. 2006, Collier al. 2014). Sites are sampled once annually in summer. Percent cover by each species has also been recorded in the last five years. The Waikato dataset has records for nearly 300 sites across the region and analysis of factors regulating macrophyte abundance and community composition is in progress (Matheson et al. 2017a). Other regional councils that monitor macrophytes include Hawkes Bay and Canterbury. Hawkes Bay Regional Council has quantified percent channel volume, percent water surface cover and percent bed cover of macrophytes at least annually in summer at 32 sites, and more regularly (seasonally, or monthly through summer) at most of these sites since 2014 (S. Haidekker, pers. comm.). Environment Canterbury has quantified percent cover of emergent and submerged life-forms at 163 sites since 2004. Regional Councils that have expressed interest in monitoring macrophytes include Northland, Wellington, and Southland.

Although Regional Council monitoring datasets are currently limited, macrophyte research datasets are available which could potentially be combined to provide a more comprehensive database of macrophyte information for New Zealand streams and rivers (e.g., Wilcock et al. 1998, Collier et al.

1999, Wilcock et al. 1999, Champion & Tanner 2000, Riis & Biggs 2003, Riis et al. 2003, Matheson et al. 2009, O'Brien et al. 2014, Matheson & Wells 2017, K. Collins pers. comm.).

Lack of a comprehensive national monitoring protocol for lotic macrophytes is a recognized implementation barrier for councils to collect comparable SoE data on macrophytes (MfE 2016). Development of a National Environment Monitoring Standard (NEMS) for stream and river macrophytes would facilitate this process and align with NIWA's new Stream Habitat Monitoring and Assessment Kit (SHMAK) which includes relevant protocols for "citizen scientists" to collect useful macrophyte data.

3.4 Macrophytes as impact indicators

Under the pressure-state-impact (P-S-I) framework, macrophytes are useful indicators of impact in soft-bottom streams and rivers in a similar way to periphyton in hard-bottom systems. Under the NPS-FM, periphyton is used as an indicator of trophic state (MfE 2015). Applying the P-S-I framework to periphyton attribute as it is presented in the NPS-FM, the pressure is nutrient influx, the state indicator is stream nutrient concentration and the impact indicator is periphyton biomass (chlorophyll a, mg/m^2). As primary producers, macrophytes and periphyton both respond to a similar set of regulating factors (as described above), with some differences, i.e., macrophytes generally thrive in environments with finer substrates, deeper water and more stable flow regimes than periphyton. Apart from this, the responses of macrophytes and periphyton to the main anthropogenic pressures on river and streams have broadly similar trajectories. Key pressures on river and stream ecosystems are loss of riparian vegetation, sediment runoff, nutrient influx, water abstraction, and incursions of introduced macrophyte species (Tale 3.1). Related state indicators are stream shade & temperature, water clarity & deposited sediment, nutrient concentrations, water depth and frequency of flushing flows, and biotic community composition (Table 3.1).

Macrophytes could also possibly be viewed as state indicators under the P-S-I framework. This would be on the basis that macrophytes are known to impact on the biodiversity and ecosystem process components of streams and rivers. As discussed above, prolific growth of introduced macrophytes affects macroinvertebrate diversity and can exacerbate dissolved oxygen deficits. Under this scenario, macroinvertebrate diversity and dissolved oxygen deficit would be the impact indicators.

3.5 Development level of a macrophyte indicator

Simple adaptations of existing macrophyte metrics monitored by Regional Councils could be useful as impact indicators (Table 3.1). . In particular, biovolume (syn. volume, clogginess, blockage factor) and water surface cover of native and introduced macrophytes could be useful as impact indicators for the first four pressures: loss of riparian vegetation, sediment runoff, nutrient influx and water abstraction. A biovolume indicator, rather than the non-specific 'cover' determination currently used by most councils, is recommended because the former provides a better estimate of the abundance of macrophytes in the wetted channel of a river or stream and is a simple measure in common use by stream ecologists (e.g., Riis & Biggs 2003, O'Brien et al. 2014). The new SHMAK kit for citizen scientist monitoring includes protocols for biovolume and water surface cover assessments. For incursions of introduced macrophytes, particularly high-risk species, the best impact indicator is likely to be species presence/absence at a site through time. Both indicators require an ability to distinguish between native and introduced macrophyte species. The Waikato Regional Council macrophyte monitoring protocol includes a comprehensive pictorial species identification guide which assists field staff to do this.

Table 3-1: Pressure-State-Impact framework applied to macrophytes.

Pressure	State indicator	Impact	Impact indicator/s
Loss of riparian vegetation	Stream shade (lack of).	Nuisance growth of introduced macrophytes which are generally better competitors under high light conditions than native species. Floating and sprawling introduced species likely to dominate where flow velocity is low.	Macrophyte native vs. introduced biovolume and water surface cover.
Sediment runoff	Stream water clarity & deposited sediment.	Nuisance growth of predominantly emergent or floating introduced macrophytes. Growth of introduced submerged species constrained by light penetration into water. However, where higher flow velocities limit emergent and floating species, low light adapted submerged native macrophytes (e.g., *Nitella*) may persist.	Macrophyte native vs. introduced biovolume and water surface cover.
Nutrient influx	Stream nutrient concentrations.[3]	Nuisance growth of introduced macrophytes which are often better competitors under nutrient-enriched conditions than native species, except where light levels are low.	Macrophyte native vs. introduced biovolume and water surface cover.
Water abstraction	Stream water depth, velocity & frequency of flushing flows.	Nuisance growth of introduced species, particularly emergents because of reduced flow velocity and depth.	Macrophyte native vs. introduced biovolume and water surface cover.
Incursions of introduced species	Stream biotic community composition	Loss of native biodiversity. Spread of high-risk pest species.	Presence/absence of macrophyte species.

3.6 Recommendation

Macrophytes are the dominant primary producer in New Zealand's soft-bottom streams and rivers, and native species in particular are important contributors to biodiversity and ecosystem processes in these systems. There is growing interest in macrophyte monitoring and it is clear that macrophyte communities respond predictably to anthropogenic pressures in these vulnerable systems. Macrophytes should be included as impact indicators in soft-bottom streams and rivers in the way that periphyton are used as impact indicators in hard-bottom systems. Of the two macrophyte impact indicators discussed above, priority should be given to the native and introduced species biovolume and water surface cover of macrophytes which is responsive to four of the five main anthropogenic stressors identified.

[3] annual mean or median, not summer or instantaneous, concentrations because nutrient uptake by primary producers during high growth in summer can lower concentrations and therefore may not accurately reflect nutrient availability.

4 Macroinvertebrates

4.1 What are macroinvertebrates?

Macroinvertebrates are animals that do not have backbones, but are large enough to be seen by the naked eye. Stream/river macroinvertebrates include insects, crustaceans (such as shrimps and crayfish), molluscs (such as snails and clams), and various kinds of worms. Macroinvertebrates are very common on stream and river beds. Most river and stream macroinvertebrates crawl on the bottom of aquatic habitats, therefore are often referred to as "benthic macroinvertebrates".

4.2 What do macroinvertebrates tell us?

Macroinvertebrates have been used as indicators of the state of freshwater since the development of the Saprobien system in the early 1900s (Metcalfe 1989; Cairns & Pratt 1993). The ecology of macroinvertebrates is well suited to this role as a biological assessment tool for the following reasons;

1. Macroinvertebrates are ubiquitous and abundant in most freshwater habitats.
2. Macroinvertebrate communities are heterogeneous, providing a direct measure of biodiversity and a broad spectrum of potential responses to environmental factors.
3. Macroinvertebrates are relatively sedentary and are therefore representative of local conditions.
4. Macroinvertebrates are important processors of energy in food chain (linking primary production with consumers).
5. Macroinvertebrates have been established as surrogates for a wide range of values, including other biological groups, but also non-ecological values (e.g., cultural values (Harmsworth et al. 2011)).

The use of macroinvertebrates as indicators in freshwater has also been enabled by the development of methodological protocols and tools that enable broadly consistent data to be generated by different agencies. In New Zealand, this includes standard sampling procedures (Stark et al. 2001), reliable and comprehensive keys for identification (e.g., Winterbourn et al. 2006) and methods and indices for interpreting taxonomic information (e.g., Stark & Maxted 2007; Collier 2008)

Consequently, every Council in New Zealand currently includes macroinvertebrates in their State of the Environment monitoring programmes, with the use of the data in national scale projects made feasible by the data being generated using the protocols and tools described above.

4.3 Data availability

All councils collect macroinvertebrate data, hence there is national coverage for this type of data, but there remains some variability in geographic or river type coverage.

Nevertheless, there is a large body of information available. For example, recent comprehensive analyses of national-scale macroinvertebrate data used 1212 locations (Larned et al. 2017) and 1966 locations (Clapcott et al. 2017).

Whilst the macroinvertebrate data is collected according to standard methodologies, there are three caveats that should be recognised in relation to the data:

1. Quality assurance of the collection, analysis and reporting of macroinvertebrate data is rarely independent or transparent. The sorting and identification of macroinvertebrate samples is a labour-intensive activity and errors can affect the quality assessment of sites. Internationally, it has been shown that a routine, independent quality assurance has a strong positive effect on data quality, increasing the accuracy and precision of data and assessments based on it (Haase et al. 2010).

2. Much of the data is not fully quantitative, which may limit the use of more sensitive abundance-based indicators. For example, a presence-absence based index will not respond to changes in the abundance of individual taxa until they are no longer present (or appear where they were previously absent). However, we do note that some of the data is semi-quantitative (i.e., relative abundance categories).

3. Most of the data is generated at a taxonomic level suitable for calculating biological indices (e.g., EPT metrics or MCI[4]), which is typically at genus or above. Noting that it is not possible to identify all river macroinvertebrates in New Zealand to species level, this level of resolution creates some limitations in the use of the national data set for biodiversity purposes. For example, from a pure biodiversity perspective, identifying taxa of conservation interest may not be possible (Joy & Death 2013). Nevertheless, adoption of a consistent, pragmatic level of taxonomy would generate useful information indicating nominal (if not true) biodiversity state and trends.

4.4 Development level of a macroinvertebrate indicator

As stated above, the use of macroinvertebrates as indicators of rivers is well developed globally and in New Zealand the use of macroinvertebrates has been commonplace for at least 40 years, with the standard methodological tools enabling a degree of consistency in the data, within the limits of the above three caveats.

There is also a wide range of indices available, from simple richness and diversity measures, through indicators of ecological state (e.g., EPT metrics and MCI) to diagnostic metrics (e.g., LIFENZ[4] (Greenwood et al. 2016) and AMDI[4] (Gray & Harding 2012)). Individual metrics have also been combined into multi-metric indices (e.g., ASPM[4] (Collier 2008), MMI4[4] (Section 6.3 of Clapcott et al. 2017) and the invertebrate IBI[4] developed by the USEPA (Plafkin et al. 1989) and adapted for use in New Zealand (e.g., Quinn et al. 2009; Reid et al. 2010). Furthermore, a range of metrics have been developed based on trait information as opposed to taxonomic information (Phillips & Reid 2012).

Most of these indices were discussed in detail and calculated for a national macroinvertebrate dataset by Clapcott et al. (2017).

[4] Abbreviations for metrics:
EPT: Ephemeroptera, Plecoptera and Trichoptera, the three common and pollution-sensitive orders of stream insects. EPT metrics relate to the abundance or taxonomic richness of insects in these orders (as absolute numbers or percentage of the whole invertebrate community).
MCI: Macroinvertebrate Community Index, a commonly-used biotic index for assessing stream ecosystem health.
LIFENZ: Lotic Index for Flow Evaluation (New Zealand), an index sensitive to hydrological alteration of streams and rivers
AMDI: Acid Mine Drainage Index
ASPM: Average Score Per Metric, an index combining EPT richness, %EPT abundance and MCI
MMI4: Multimetric Index based on 4 metrics (EPT richness, % EPT richness, MCI and prevalence of a biological trait)
IBI: Index of Biotic Integrity

4.4.1 Pressure, State and Impact

The pressures that affect the macroinvertebrate community in New Zealand were reviewed by Collier et al. (2014) as part of the national freshwater policy development. In Figure 1 these pressures are summarised as orange boxes, while the pathways of effect are shown by arrows and grey boxes (which could be used as state indicators). Figure 1 demonstrates the wide ranging and interactive nature of these stressors and the pathways of effect.

Despite the macroinvertebrate data and indices meeting Statistics New Zealand's Tier 1 criteria (see Section 1.1.4), most of the indices used to communicate the data are state or diagnostic indicators. That is, they describe the state or the problem at a location, rather than an explicit assessment of the degree of impact.

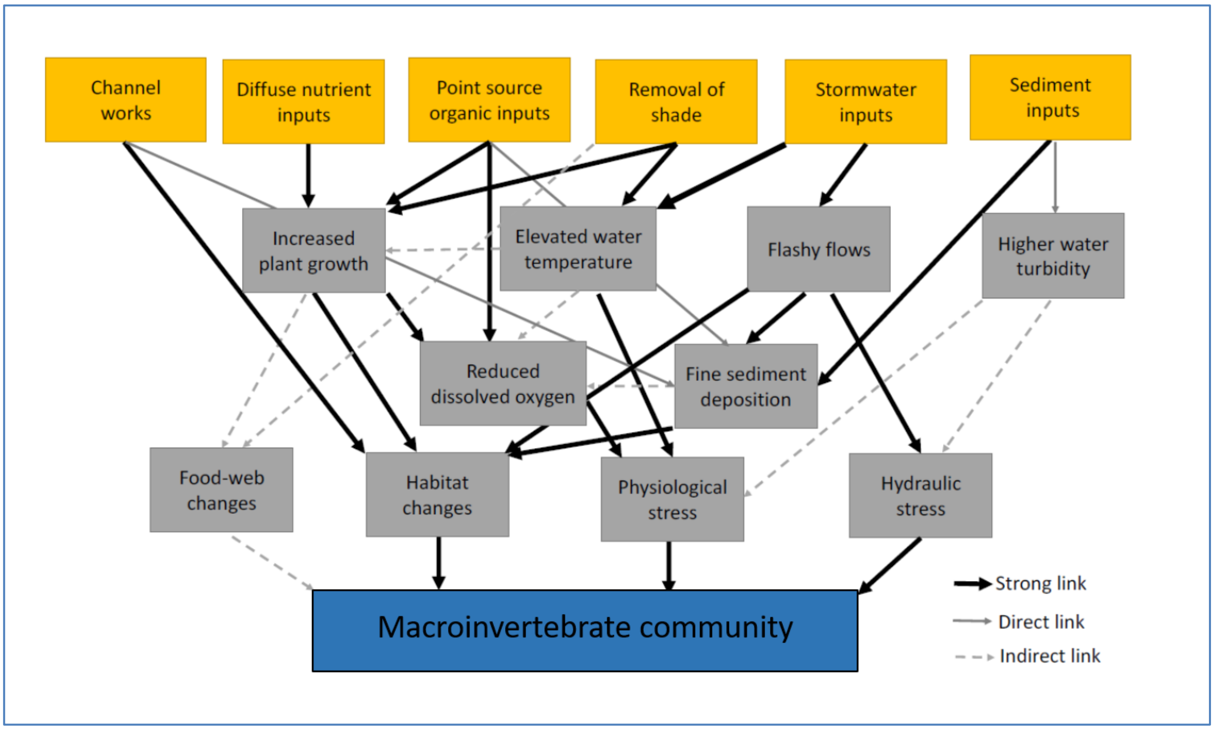

Figure 4-1: The pressures (orange boxes) that affect macroinvertebrate communities and pathways of effect (adapted from Collier et al. 2014).

To contextualise these indicators, banding systems have been developed that allow the results obtained from individual sites to be compared with the range of results obtained from a larger dataset (e.g., national scale). These banding systems typically set upper and lower bounds based on the results observed in the larger datasets (e.g., MCI quality bands (Stark & Maxted 2007)) with a site's quality graded according to this range.

Whilst such banding systems may be able to provide a broad indication of relative impact, the underlying assumption of such banding approaches is that all sites are created equal. That is, all sites can achieve the maximum value observed in the large dataset used to develop the banding system. However, not all sites may be able to achieve a result in the higher quality categories due to natural environmental limitations.

This is because the taxonomic composition of the macroinvertebrate community varies naturally from place to place, in addition to any impacts that might occur from human activities. Disentangling the effects of natural environmental variability and human impact remains a major challenge for biological assessment systems. An effective biological assessment should ensure than any index provides consistent meaning in different environmental settings, so that a given score from an index should indicate the same biological condition (state and/or impact) irrespective of geographic location or stream type (Mazor et al. 2016).

4.4.2 Reference condition

The above issue has commonly been addressed in biological assessment using a reference condition approach (Reynoldson et al. 1997). In other countries, models have been developed that predict the reference conditions for the macroinvertebrate community at a location, which is then compared with the observed macroinvertebrate community (or indices generated from it) at that location (Wright et al. 1993; Parsons & Norris 1996; Hawkins et al. 2000; Mazor et al. 2016). The deviation from reference is then considered to be the impact on the community arising from human activities. Hence, the reference condition approach offers the potential to explicitly assess the impact of human activities at a location.

The use of the reference condition approach is key to unlocking the use of macroinvertebrate metrics as 'impact' indicators, in contrast to 'state' indicators. The deviation from reference condition allows the assessment of the impact on the macroinvertebrate community, whereas using an indicator without comparison to reference condition allows only an assessment of state[5].

The challenge for New Zealand, as in other countries, is to develop models that accurately predict the macroinvertebrate community in streams for which reference conditions do not currently exist. Models can be developed to predict community composition in all stream reaches (see Section 4.5.2 below) but, because of the limited number of undisturbed streams for some stream types, model predictions may not be as robust for these streams types as for stream types with extant reference sites, and may be harder to validate.

4.4.3 Biodiversity indicators

All indices are based on the taxonomic composition of macroinvertebrate samples retrieved from a river, however this taxonomic information is typically summarised into measures of richness (e.g., number of taxa), diversity (e.g., number and evenness of taxa) or ecological condition, with the underlying taxonomic information largely unused. Most of these indices calculated from macroinvertebrate information are, at least in part, indicators of ecological state, rather than direct measures of biodiversity.

If we assume taxonomic composition is a core measure of biodiversity, then being able to assess biodiversity impacts requires the ability to assess changes in taxonomic composition directly. This is particularly important in using macroinvertebrate indicators for biological assessment as species replacements in response to environmental change are common. For example, in two studies the taxonomic composition at a location changed significantly in response to environmental changes, whereas most of the indices calculated from the community information did not change (Graham et

[5] One could make a case for trends over time being a special case – trend analysis allows the direction of impact (positive or negative) to be tracked over time, but does not allow the degree of impact to be assessed

al. 2015; Neale & Moffett 2016). This finding indicates rivers of different degrees of impact (with different macroinvertebrate communities) could have similar values for state indices.

Therefore, if the objective is to describe the impacts on biodiversity, it is important to incorporate taxonomic information into assessments of biodiversity impact. Sensitive macroinvertebrates that may produce high index scores, and indicate a high-quality river environment, are not the only taxa that contribute to biodiversity. A more effective measure of biodiversity would consider the macroinvertebrate fauna that is expected to occur in each location.

A relatively simple index that carries out this type of assessment was developed and implemented as part of the Stream Ecological Valuation (SEV) assessment tool (Storey et al. 2011). This index, called V_{invert} in the SEV, assesses the intactness of macroinvertebrate biodiversity by comparing taxa present at a location to a list of taxa that are expected (>50% probability) to be found in nearby reference streams (either "real" or modelled). This index is scaled from 0 to 1, with sites scoring close to 1 indicating high macroinvertebrate biodiversity and vice versa.

A similar approach was developed by Courtemanch & Davies (1987), which the US EPA included in its bioassessment protocols (i.e., Community Loss Index in Plafkin et al. 1989). Furthermore, such an approach was incorporated into an effects assessment of land use on benthic macroinvertebrate communities in Waikato (Quinn et al. 1997a).

This type of assessment is consistent with the reference condition approach, but would require the development of a nationally applicable index to directly assess macroinvertebrate biodiversity and the impacts upon it.

4.5 Recommendation

4.5.1 Immediate term

Notwithstanding the issues discussed above, in the immediate term, there is little alternative but to use existing indicators for river macroinvertebrates.

It would be logical to use MCI, EPT richness and % EPT richness at this time. This is because these indices can be calculated from all existing data (i.e., they do not require abundance information) and would also allow the calculation of multimetric indices that may provide greater power for differentiating ecological condition than individual indices (e.g., ASPM (Collier 2008), MMI4 (Clapcott et al. 2017) and Invertebrate IBI).

4.5.2 Medium term

In the medium term, it would be logical to build on the reference condition work that MfE have undertaken to develop operational tools for directly assessing impact on macroinvertebrate communities.

There are several ways in which a reference condition approach could be progressed, as described by Clapcott et al., (2017). However for directly assessing biodiversity impacts, the multivariate approach would likely provide the greatest potential. This is because it offers a site-specific prediction of the taxonomic composition of the macroinvertebrate community for each location. This would enable the use of the reference condition approach for assessing impacts on macroinvertebrate communities using existing indicators, but also provide the base information on which to develop a specific index of macroinvertebrate biodiversity in a similar manner to V_{invert} that is suitable for national application.

With sufficient resourcing, indicators of biodiversity impact using the reference condition approach could be developed to be used in the next Freshwater Domain report due for publication in April 2020 (recognising the June 2019 data deadline).

5 Fish

Freshwater fish are undoubtedly the highest-valued group of aquatic biota. They are already incorporated into the freshwater domain reporting, but the only impact indicator for freshwater fish biodiversity is change in Conservation Status (as defined by Department of Conservation). In this section we explore other impact indicators of freshwater fish biodiversity. Because Conservation Status is already clearly defined we do not cover this within the present chapter. Also, the indicators we consider (except the IBI) include only native fish, while exotic fish are considered only as pressures.

5.1 What do fish biodiversity indicators tell us?

No studies have directly determined the pressures driving changes in freshwater fish biodiversity metrics (e.g., species richness) in New Zealand. Therefore, we used the existing literature to explore what pressures drive changes in abundance of individual species. We then identified several key pressures that have been shown to affect multiple species, and assumed that these key pressures influence freshwater fish biodiversity. We describe each of these pressures and how they influence the structure of freshwater fish communities. The goal of this literature summary was not to provide an exhaustive summary of all the pressures that structure each fish species, but rather identify some of the common pressures that structure individual species.

In the following two sections, we consider both natural and anthropogenic processes. Anthropogenic pressures must be considered in the context of natural pressures because natural pressures are often the primary drivers of freshwater fish ecosystem structure (McDowall 1993; Leathwick et al. 2008; Crow et al. 2014). Failure to consider these natural processes may result in wrong conclusions about the drivers of changes in freshwater fish biodiversity.

In the Pressure-State-Impact framework, we consider State to refer to metrics of freshwater fish biodiversity, and impact indicators as reporting of these metrics relative to a baseline year (change over time) or to a modelled reference condition (Figure 5-1 and Section 5.3.2).

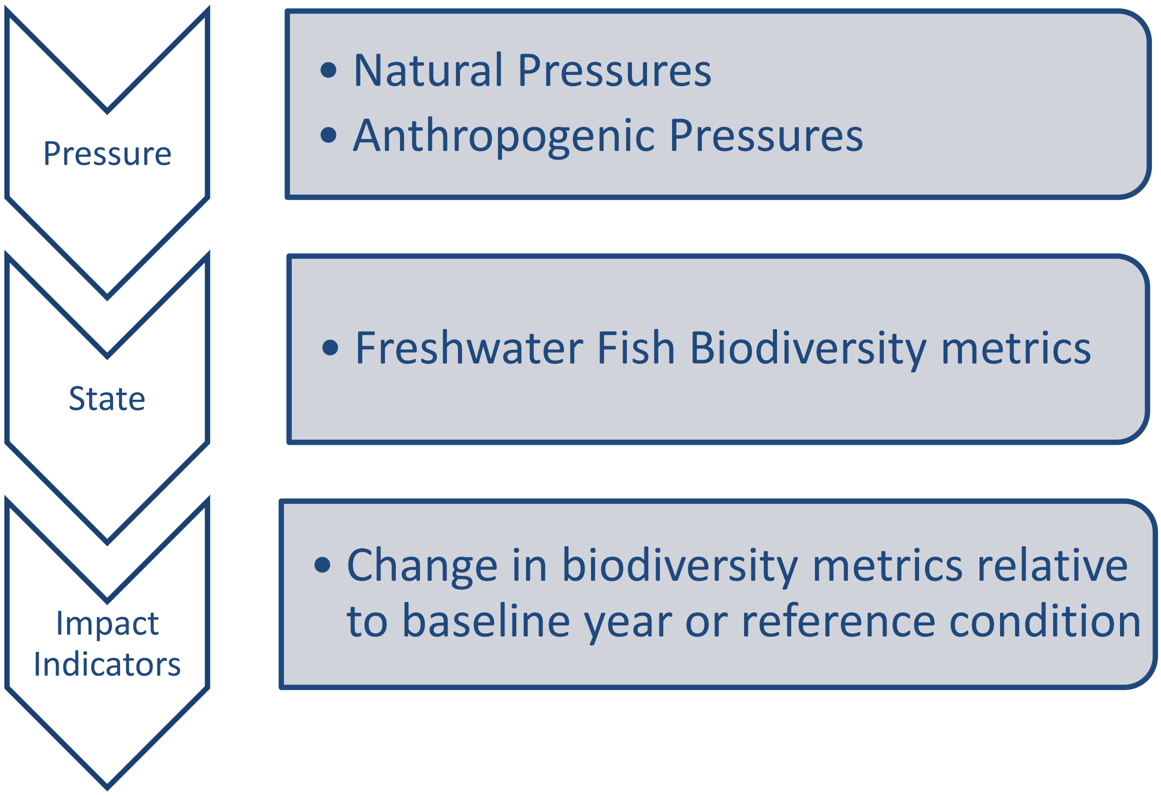

Figure 5-1: Overview of the how the sections discussed within the present chapter fits within the Pressure-State-Impact framework use by MfE.

5.1.1 Natural processes structuring the state of riverine freshwater fish biodiversity

Diadromy and altitudinal gradients

A high proportion of New Zealand's native freshwater fish are diadromous (i.e., they have a marine phase in their lifecycle), therefore migration is one of the primary processes structuring fish biodiversity and community structure (McDowall 1993; McDowall 2009; McDowall 2010a). Several species grow to adulthood in freshwater then migrate downriver to breed in the sea (e.g., eels, mullet, freshwater flounder), but most breed in freshwater with their juveniles travelling downriver to develop at sea (e.g., galaxiids, smelt and bullies). Only the lamprey breeds and develops in freshwater streams and spends its entire (non-breeding) adult life at sea. Because all diadromous species need to migrate upriver from the sea at some stage, freshwater fish biodiversity is partially structured by geography. Specifically, altitude and distance from the sea interact with species' life histories to produce longitudinal changes in biodiversity within river systems (Leathwick et al. 2008; McDowall 2010a; McDowall 2010b; Clapcott et al. 2011; Crow et al. 2014) (Figure 5-2).

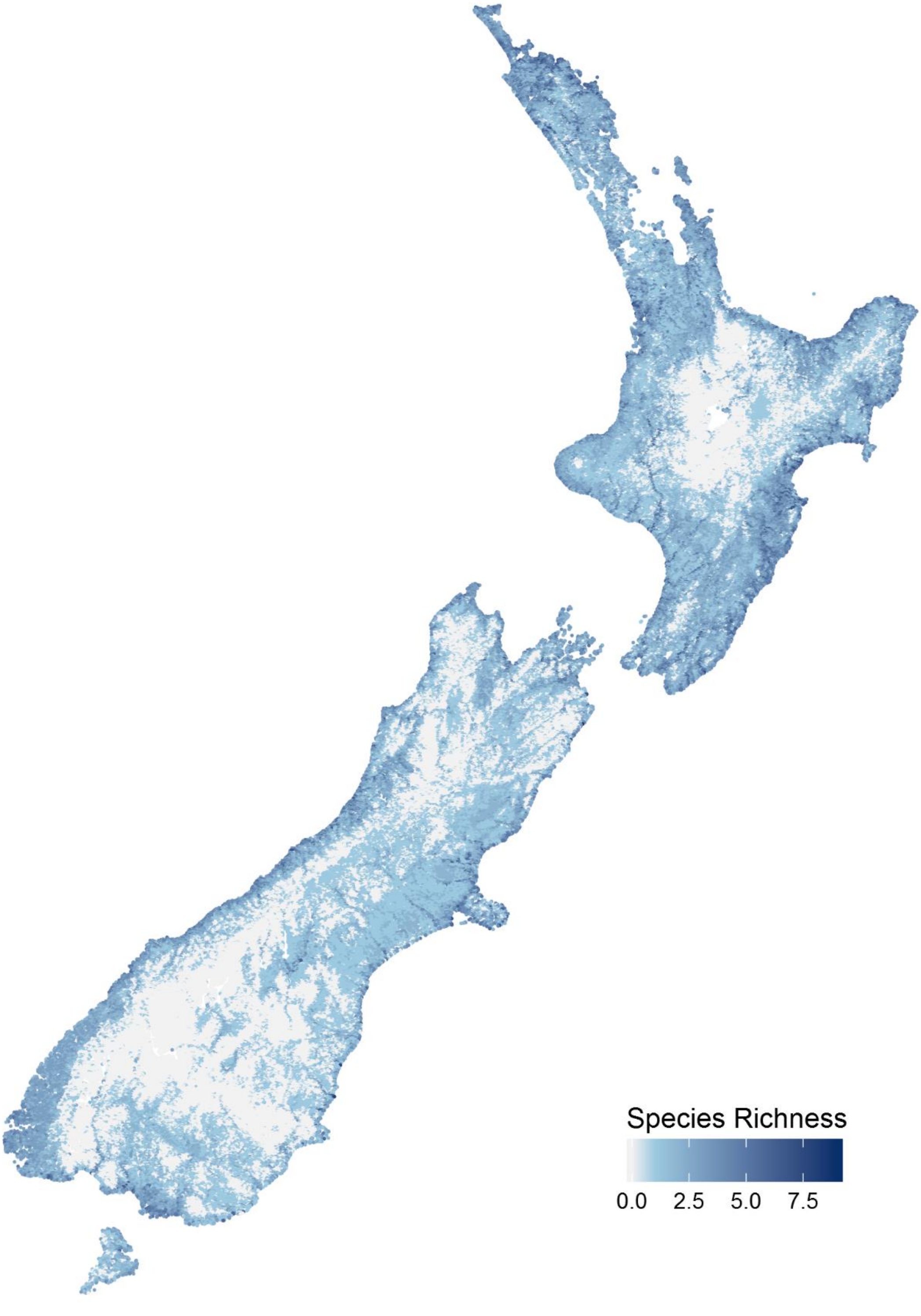

Figure 5-2: Biodiversity of freshwater fish in New Zealand. Biodiversity values are taken from the predicted occurrence of 27 freshwater fish species across the river network (from Crow et al. 2014).

Temperature

Temperature is one of the key environmental gradients that structures fish communities in New Zealand. National models show that, out of 15 variables, water temperature is on average the second most important variable to predict occurrence of 16 species (Crow et al. 2014). Similarly, the national scale occurrence models of Leathwick et al. (2008) also showed temperature to be one of the most important predictors of diadromous and non-diadromous fishes. Temperature is influenced by both natural (e.g., elevation, groundwater inflows) and anthropogenic factors (e.g., riparian shade alteration, flow abstraction, climate change).

Biogeographical patterns

In addition to patterns driven by migration dynamics and environmental gradients, freshwater fish biodiversity varies naturally among areas because of the limited distributional ranges of many non-diadromous species (McDowall 2010a). For example, several non-diadromous species present in Otago are not found outside of this region. In areas with high levels of non-diadromous fish diversity (mostly in Otago and Southland), spatial patterns in biodiversity will be driven by the natural geographical limitations of the fauna.

5.1.2 Anthropogenic pressures structuring the state of freshwater fish biodiversity

Habitat degradation

Habitat degradation is one of the key pressures that has been implicated in threat ranking assessments of many freshwater fish species (Allibone et al. 2010; Goodman et al. 2014). Since European settlement, there have been many changes in land use in New Zealand, with large areas cleared for human habitation and agriculture. For example, it is estimated that wetlands once covered at least 670,000 ha before European settlement, but have now been reduced to about 10 % or their original extent (MfE & Stats NZ 2015). Within the Waikato catchment alone, the loss of wetlands was estimated to be 84% between 1840 and 1976 (McDowall 1990). These wetland areas are key habitats for eels, kokopu species and spawning areas for īnanga (McDowall 1990), so the loss of this habitat has undoubtedly driven changes in the community composition of freshwater fishes.

The effects of land-use change on freshwater fish biodiversity are multi-layered, affecting the temperature, light, water quality and geomorphology of ecosystems. The biomass of large eels is directly related to the amount of suitable cover (Burnet 1952), so the loss of cover by large wood and macrophyte removal, channelisation of waterways, together with siltation, reduces the quality of habitat available to longfin and shortfin eel species. The removal of macrophytes for the purposes of drainage management in suburban areas has been shown to displace native fish and reduce catch-per-unit-effort by 60% (Greer et al. 2012). In contrast, Hicks and McCaughan (1997) found that the conversion of land from forest to pasture resulted in an increase in the abundance of shortfins, probably because of greater primary production in the pasture streams (Quinn et al. 1997a) boosting secondary production of invertebrates. Access to pasture may provide eels with an additional food source of terrestrial invertebrates (Chisnall 1987; Chisnall & Hicks 1993).

Increased sedimentation rates associated with deforestation may also reduce food availability to fish in pasture sites by smothering streambed substrates that are habitat for macroinvertebrates (Hanchet 1990). Jowett and Richardson (1990) found higher invertebrate biomass was associated with coarse sediments, confirming that food resources for freshwater fishes were reduced with increased sedimentation rates. Sedimentation may also impact juvenile eels who are known to seek refuge in subterranean substrates (Cairns 1950). High suspended sediment rates have also been shown to impact on drift feeding rates of freshwater fish (Greer et al. 2015), and increasing sedimentation rates are negatively associated with occurrence of several freshwater fish species (Depree et al. 2017).

Habitat degradation has also impacted on spawning areas for lamprey (*Geotria australis*). Spawning and nesting occurs in large boulder substrates (Baker et al. 2017), but this habitat type is thought to have been reduced because of landuse conversions from forest to farmland (Closs et al. 2014). This is believed to have greatly reduced the distribution and abundance of this species, as it has for other freshwater fish species (Closs et al. 2014).

Shortjaw, giant and banded kōkopu are found in areas with plenty of overhead vegetation providing cover. Therefore the loss of forest cover in New Zealand (from 85% before human arrival to just 28% in the 1990s (Taylor & Smith 1997)) is considered to be one of the most significant pressures influencing the abundance and distributions of these species.

Connectivity and barriers to fish passage

Barriers to fish passage control biodiversity of diadromous fishes by restricting movements between freshwater and marine environments. Barriers to migration can restrict access to habitats required for foraging and feeding, predator avoidance, shelter, and spawning (Gibson et al. 2005). Lack of access to these habitats, particularly for obligate migratory species, can ultimately lead to a reduction in recruitment, population decline, and a loss of biodiversity (Jellyman & Harding 2012). Maintaining connectivity between habitats can be critical to ensuring the long-term success of fish populations (Fullerton et al. 2010)

Large hydro-electric dams represent major barriers to migration in some river networks, though many have had upstream passage facilities installed. At a national scale, however, smaller-scale obstructions, such as weirs and culverts, are the most problematic artificial barriers because there are many of them, and provisions are often not made for fish passage. For example, of the estimated 3.6 culverts per 100 hectares in the Waikato Region, 36% or 1.3 per 100 hectares were barriers to all fish at all flows. As the catchment area for the lower Waikato River below Karāpiro (excluding the major lakes) is approximately 6,500 km^2, approximately 8,500 culverts could be limiting fish movement. Some of these culverts are more serious barriers than others because they restrict access to larger amounts and/or quality of habitat upstream and/or are not passable at any state of flow (Watene-Rawiri et al., in press).

Some barriers, such as hydro-electric dams and pump stations associated with flood and tide-gates, also impact on outgoing fish passage as fish are often seriously injured or killed (Franklin et al. 2018). Since longfin eels are the species that penetrate farthest inland, the installation of hydroelectric dams has impacted this species the most by compromising their upstream access and also causing the death of outgoing migrant adults (Figure 5-3).

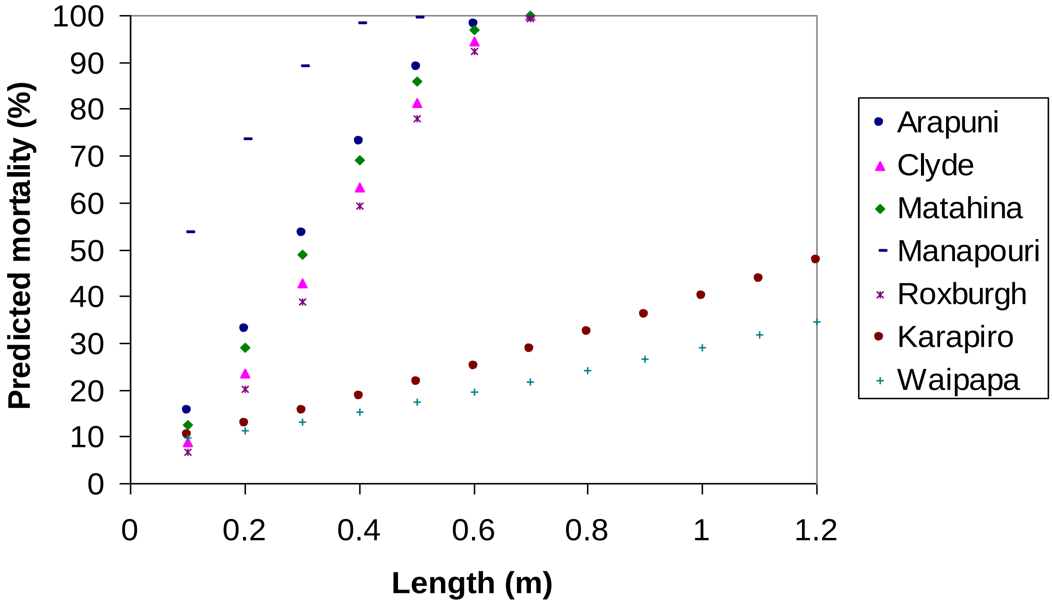

Figure 5-3: Estimated mortality of migrating eels of varying lengths, for different hydroelectric stations in New Zealand. The lower mortalities of Karāpiro and Waipapa are due to different turbine types compared to other stations. Figure taken from Jellyman (2013).

Exotic fish introductions

Exotic fish species in New Zealand compete with and predate on galaxiids and juvenile eels, altering community composition and reducing indigenous species richness. Most of the predation pressure placed on galaxiids (both diadromous and non-diadromous) is from brown trout (McDowall 2006; McIntosh et al. 2010). Glova (2003) presented evidence, from behavioural studies in a small stream simulator, that the number of īnanga declined when they shared stream habitats with brown trout (255–390 mm long), and also that the galaxiids shifted their microhabitat use when trout were present. Presumably, this resulted in the galaxiids occupying less favourable microhabitats for drift feeding on invertebrates. In the South Island, some non-diadromous galaxiids appear to have been almost completely displaced from areas where brown trout are present (Townsend & Crowl 1991; McIntosh et al. 1992). Other introduced pest fish (perch, koi carp, tench, gambusia, rudd, and catfish, though the latter three are mainly restricted to lakes) can also indirectly impact on native fish biodiversity through competition for food resources and by degrading habitat (Rowe 2004).

5.2 Data availability

The New Zealand Freshwater Fish database (NZFFD) is the main dataset for reporting freshwater fish impact indicators at the national scale. The NZFFD (http://www.niwa.co.nz/our-services/online-services/freshwater-fish-database) is an open resource where freshwater fish data are entered into a predefined data sheet and then submitted to the database. The data are checked for accuracy by the NZFFD administration team before records are loaded into the database. This is an extensive temporal and spatial dataset on freshwater fish distributions and abundance that dates back to 1901. The NZFFD is therefore the largest dataset available for examining impact indicators, but it has some limitations. Although the accuracy of the data is quality controlled, the sampling methodology used to collect the fish data is not standardised. Differences in sampling methodology between NZFFD observations result in sampling bias, which would confound temporal comparisons of impact indicators. Crow et al. (2016) presented a methodology for minimising the influence of sampling bias in the NZFFD, which should be followed for impact indicators generated using the NZFFD. A new database input format is currently being developed for the NZFFD that will identify records collected using a consistent methodology (Joy et al. 2013). This new format will improve the robustness of the database for calculating impact indicators because it will allow the impact indicators to be calculated from observations collected using the same methodology. Regional Councils have recently started collecting information using the sampling protocols of Joy et al. (2013) as part of their State of the Environment (SoE) monitoring programs, which means the NZFFD will contain much more consistently collected data over the coming years.

impact indicators based on species abundances require more labour intensive sampling protocols than are normally used in New Zealand. Most fish sampling data in New Zealand are collected using a single-pass electric-fishing approach outlined in Joy et al. (2013). This protocol is used frequently because it can be quickly used to assess species occurrence and generate an index of relative abundance. The alternative and more labour-intensive multi-pass electric fishing method has an additional benefit of being able to generate an estimate of population size. Population size estimates are preferable to single-pass catches as an index of abundance because they account for differences in sampling efficiency (Graynoth et al. 2012). However, the current and historic preference of regional councils for single-pass data means that there is limited ability to monitor changes in population estimates. One possible solution is to develop a model that could adjust single-pass catches to population size estimates based on factors that influence capture efficiency (e.g., habitat type, substrate, species present, size classes). A model that adjusts single pass catches to population estimates is currently being developed at NIWA for shortfin eel (NIWA unpubl. data), but this would need to be tested for other fish fauna at a national scale. Developing these models would likely take two-three years, and it is difficult to predict how successful these models would be. There are additional multi-pass data (e.g., universities, DOC) available within the NZFFD, but they are spatially restricted and typically only represent a few areas. Alternatively, Regional Councils (key data providers for the NZFFD with their SoE monitoring protocols) could be encouraged by MfE to collect multi-pass data at a sub-set of their SoE sites. If it were feasible for Regional Councils to collect multi-pass data at a sub-set of their sites, this would result in a large dataset of abundance information across the country.

5.3 Development level of impact indicators for changes in freshwater fish biodiversity

5.3.1 Index of Biotic Integrity (IBI)

The freshwater fish IBI is an indicator developed in the Northern United States (Karr et al. 1986), which is a unitless metric calculated across a range of up to 12 fish attributes relating to species richness (six metrics), trophic composition (three metrics), fish abundance and condition (three metrics). The IBI represents a holistic score that describes how fish communities respond to anthropogenic processes. Not all 12 metrics can be used in countries other than the USA, but the IBI framework has been modified for the use in streams throughout the world (Hughes et al. 1998) and has been trialled in New Zealand (Joy & Death 2004). The international use of this metric makes it an attractive option to use as an impact indicator, but the limitations outlined by Joy & Death (2004) suggests that further work is required before it could be used as a robust impact indicator in New Zealand.

Because the IBI is calculated across multiple attributes, it is difficult to identify exactly what pressures drive changes in an IBI value. Arguments have been presented overseas discussing how the IBI responds to multiple interacting anthropogenic process (Karr et al. 1986). In New Zealand, Joy & Death (2004) reported that IBI scores respond to landuse. However, as they themselves noted, Joy & Death (2004) did not account for natural processes (see Section 5.1.1) or the sampling bias within the NZFFD (see Section 5.2 and Crow et al., (2017)), and this omission undermines the reported link between the IBI and pressures associated with land use. The lack of a link to specific pressures makes it difficult at present to use the IBI within the Pressure-State-Impact framework of the Environmental Reporting Act and the requirement to show quantitative links between indicators and pressures. Nevertheless, in their original paper, Karr et al. (1986) suggest that the IBI responds to various pressures, implying that with further work to conclusively identify these responses (i.e., account for sampling bias and natural processes) in the New Zealand context, the IBI would fit within the Press-State-Impact framework. This work would likely require a workshop with several freshwater fish ecologists to refine the method for calculating some of the 12 individual metrics. Once the IBI is updated, an analysis could then examine spatial variability in the metric and determine the pressures that drive the IBI and/or its 12 individual components.

The promising aspect of utilising the IBI as an impact indicator is the availability of ongoing data. The primary data source used to monitor changes in the IBI scores would be the NZFFD, which is updated each year with approximately 1000-3000 observations.

5.3.2 Indigenous species richness

Indigenous Species Richness (ISR) would be an informative indicator of freshwater fish biodiversity. As a simple measure of number of species present, ISR would be a state indicator. However, if presented as a change relative to a point in time, ISR would represent the impact on the fish community resulting from human activities since that time. This can be done relative to 1977 using information in the NZFFD and following a similar data selection and analysis to that in Crow et al. (2016). Potentially in future, ISR could be expressed as deviation from reference condition (as recommended for macroinvertebrates, Section 4.4), if reference condition can be defined through further work.

As an impact indicator, ISR can be interpreted in relation to human activities because individual species have already been shown to respond to key pressures, as outlined in Section 5.1. ISR can be measured using a variety of different metrics, but two potential metrics are species richness (i.e., total number of species) and the more complex Shannon's Diversity Index. The Shannon's diversity index is preferable to the simple species richness because it incorporates abundance of species. However the fish abundance data require multi-pass electric fishing data (as outlined above), whereas simple species richness can be calculated from presence-absence data.

Like the IBI, ISR would need to account for natural spatial patterns (geographic and environmental gradients and species biogeographical patterns). This could be done by calculating species richness and Shannon's diversity between areas that have similar community composition and environmental conditions. Defining the rationale for identifying these areas with comparable environmental conditions would require further work and input from several fish ecologists in a workshop setting. A rationale for defining reference conditions for New Zealand freshwater fish biodiversity could be developed at the same time. As for the IBI, the primary data source for ISR would be the NZFFD.

5.3.3 Abundance/biomass of indicator species

The abundance of key individual species could be used as impact indicators if they are associated with changes in abundance of other species (Jowett & Richardson 2003). Jowett & Richardson (2003) utilised a Two-way Indicator Species Analysis (TWINSPAN) to identified 12[6] fish communities that were characterised by a dominant species that occurred with the other species. The dominant species could be used as indicator species, with assumption that changes in the abundance of the dominant species would be associated with changes in the other fish species. It is unlikely that there will be sufficient abundance data available to monitor changes in abundance of all 12 species in the near future. Moreover, there may not be established literature identifying how the individual species will respond to the pressures outlined above, which means this type of indicator would not fit within the Pressure-State-Impact framework. Therefore, identification of the key indicator species would need to be further refined. This could include repeating the TWINSPAN analysis (or similar approach) to see if fewer key species can be identified, now that there are more fish observations available within the NZFFD (approximately twice the number of observations used by Jowett & Richardson (2003)). Exploring the NZFFD would also help identify which of the potential indicator species have sufficient abundance data available to make this approach workable.

Longfin eels should be developed as an indicator species because of the large amount of abundance information available and because abundance of this species responds to most of the pressures driving biodiversity (e.g., habitat loss, presence of barriers). A longfin eel impact indicator would fit within the Pressure-State-Impact framework because of the multiple proven links between longfin abundance and anthropogenic pressures. Furthermore, ongoing research on this species by MPI, DOC, NIWA and universities will ensure that information on this species distribution and abundance is collected in the future. One current project that could form a basis for ongoing monitoring in the future is the MPI funded programme to develop an index of longfin eel biomass for New Zealand. There are also some new spatiotemporal modelling approaches being developed for longfin eels (collaboration between NIWA, Victoria University and The University of Washington) that could be used in the future to display how the spatial occurrence and relative abundance of longfin eels and other indicator species change with time (Figure 5-4), (similar to the temporal analysis from Crow et al. (2016)).

[6] Shortjaw kokopu, banded kokopu, redfin bully, crans bully, shortfin eel, īnanga, torrentfish, upland bully, brown trout, rainbow trout, kōaro, longfin eel.

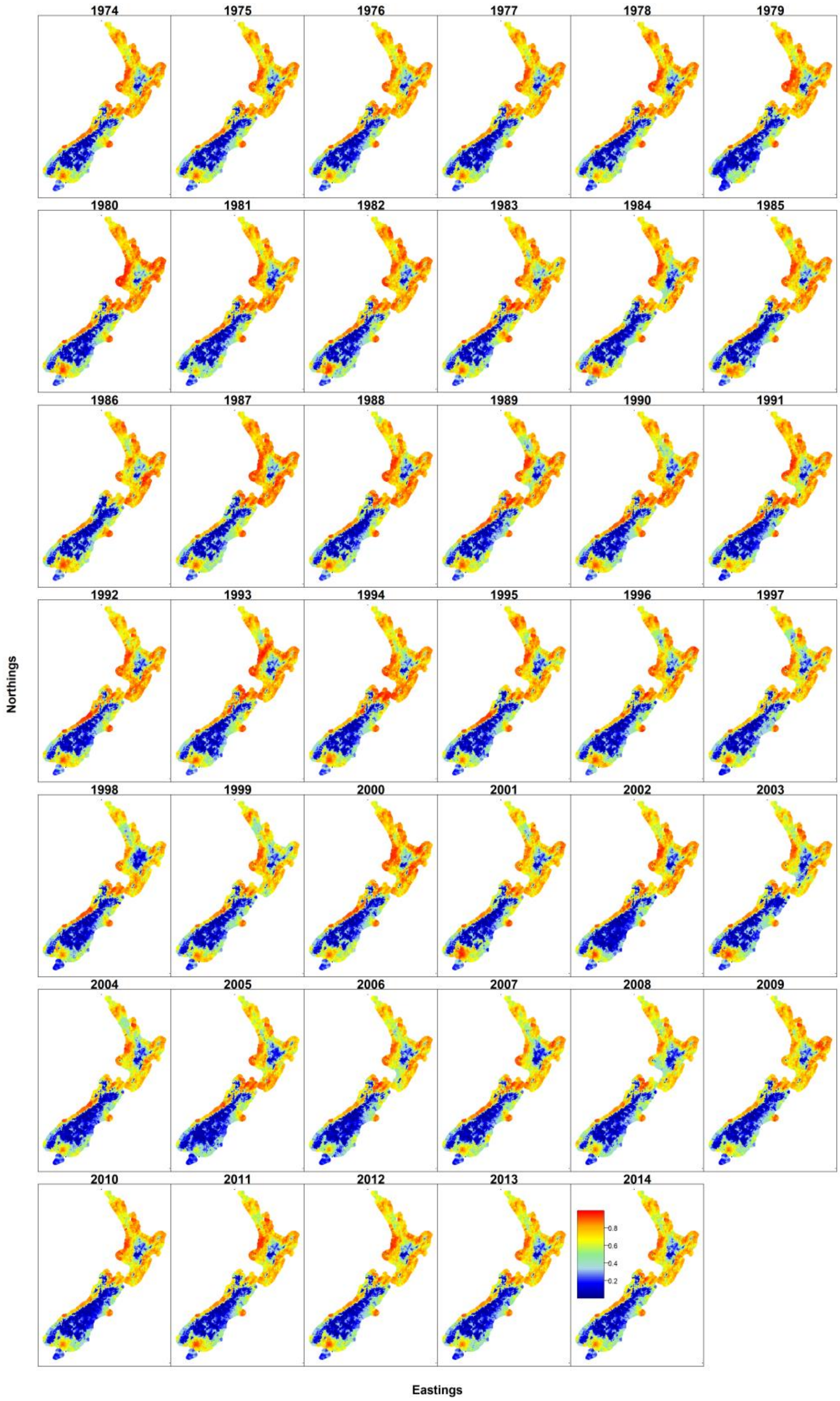

Figure 5-4: Spatial changes in the predicted capture of longfin eels through time. Predictions are made using data from the NZFFD and a modelling approach outlined in Thorson et al., (2015).

5.3.4 Connectivity and barriers to fish passage

Connectivity for migrations is an aspect that MfE has already identified it wants to monitor in SoE Reporting. Here we recommend diadromous species richness (or Shannan Diversity Index) as the indicator for monitoring the impacts of connectivity loss. We recommend reporting this impact indicator in conjunction with the proportion of the national river network that is impacted by fish barriers, which is a pressure indicator.

An impact indicator summarising the richness (or Shannon Diversity Index) of diadromous species would reflect connectivity because stream reaches below dams have higher richness of diadromous species than reaches above (Jellyman & Harding 2012). Information on spatial changes to diadromous species richness through space and time would reflect the level of connectivity. This information is currently available within the NZFFD, which means there will be plenty of data available for periodically updating this impact indicator. The indicator could be further developed by incorporating individual species' climbing abilities. For example, eels and kōaro are exceptional climbers and are able to negotiate considerable barriers, but īnanga are much poorer climbers. The different climbing abilities of the taxa would suggest that areas with īnanga have very good connectivity, while areas with only kōaro and eels may have poorer connectivity. Developing an assessment of climbing abilities (Figure 5-5) would require more information and input from various fish experts across the country.

A pressure indicator summarising the proportion of the national river network that is impacted by downstream fish passage barriers would require data on the locations of all existing barriers in New Zealand. This dataset would also need to include corresponding information on whether each barrier has fish passage structures associated with it or not, and which species may be able to pass the barrier. This information could then be used to estimate the proportion of the river network that is inaccessible to fish migrations. Fortunately, information on the locations of large (>2 m high) barriers within the River Environment Classification (Snelder & Biggs 2002) could form a useful data framework for recording this information. NIWA is also currently collating information on locations of additional smaller barriers and associated fish passage provisions which would form a useful starting point for calculating this impact indicator. This pressure indicator could be updated periodically as new barriers and/or new fish passage structures are installed to display how connectivity has decreased (e.g., more barriers installed without fish passage structures) or increased (e.g., more existing barriers have fish passage structures installed) through time. The dataset could also be used to identify which barriers block the largest area of the stream network, potentially helping resource managers prioritise sites for fish passage restoration. Data on the installation of new fish passage structures at barriers is often recorded by Regional Councils and Department of Conservation, so information for the ongoing changes to the proportion of the upstream network should be accessible.

5.4 General Discussion

The relationships between pressures and the state of freshwater fish biodiversity could be strengthened by developing a biodiversity model for New Zealand freshwater fishes. The predictions of Crow et al. (2014) and Leathwick et al. (2008) could be modified and used to produce these relationships, with native species richness as the metric being predicted across the country (Shannon's Diversity Index could also be predicted across the country, but the paucity of abundance data within the NZFFD may make predictions highly uncertain). This would provide a prediction of native fish biodiversity for New Zealand along with a list of important processes driving variation in biodiversity, directly linking pressures to biodiversity. The models to provide these predictions already exist and could be re-used.

An index of Observed/Expected (O/E) fish biodiversity would be an informative metric to use as an impact indicator, but it is not included in the present chapter because it is yet to be conclusively developed. Development of an O/E index for fish has been attempted in New Zealand (Joy 2013), but failed because of the rationale associated with the calculations of the observed and expected values. Another attempt could be made, but it will require considerable development of the rationale behind the approach before proceeding. This would likely require a workshop with several freshwater scientists to develop the justification and approach. Following development of the rationale, spatial modelling of the metric across the REC would be required.

New Zealand freshwater fish ecologists have discussed each of the impact indicators suggested above. However, final development of indicators should proceed with input from a wider expert panel of fish ecologists, and should include Māori researchers if synergies are to be explored between the current section and the Mātauranga Māori category currently being developed. Incorporating wider feedback from more experts will ensure that a robust list of metrics is developed and implemented for monitoring the changes in state of New Zealand's freshwater fish biodiversity. We envisage a series of workshops where each of the suggested Indicators are discussed.

Figure 5-5: Four common species of native freshwater fish with different climbing abilities. The species at top (īnanga *Galaxias maculatus*, left; common bully *Gobiomorphus cotidianus*, right) are very poor climbers. The species at bottom (kōaro *Galaxias brevipinnis*, left; shortfin eel *Anguilla australis*, right) have the best climbing ability of any New Zealand freshwater fish. Photo credits: Bob McDowall (kōaro) and Nelson Boustead (shortfin eel).

5.5 Summary and recommendations

Table 5-1 below summarises the connections between each of the impact indicators to the Pressure-State-Impact framework. The table of impact indicators summarises the pressures that the impact indicators respond to, as well as available data and any suggestions for further development. Overall we provide an outline in regards to four impact indicators that could be used to monitor changes in the state of New Zealand's freshwater fish biodiversity. As discussed, all four of these indicators require more development but most of this work could be completed in time for the 2020 freshwater domain report.

We recommend that Indigenous Species Richness (ISR) is likely to be the most informative and easily developed impact indicator for fish biodiversity, provided both ISR indices (native species richness and Shannon's Diversity) can be used. The ISR could potentially also incorporate changes in community composition (i.e., replacement of one species by another). This would allow species displacements to be monitored). The abundance information included within Shannon's Diversity gives this index greater sensitivity than ISR, but requires multi-pass electric fishing to collect the data. If abundance data for whole fish communities cannot be collected, then abundance/biomass of key indicator species would likely be the next most robust impact indicator. This also requires abundance data (collected by multi-pass electric-fishing), but only for the indicator species. If there are insufficient multi-pass data to monitor changes in indicator species abundance, then the two remaining impact indicators are connectivity (as diadromous species richness) and IBI. Of these, connectivity will be quicker to develop than the IBI, but the IBI potentially provides a more holistic score that describes how fish communities respond to anthropogenic processes. The holistic nature of the IBI makes it a preferred option, but it will require more work to develop and test than the connectivity indicator.

Table 5-1: Potential indicators for monitoring impacts on freshwater fish biodiversity, with links to pressures and information availability.

Pressures driving changes in the impact indicator	Impact Indicator	Available Data	Suggested Further Development	Ongoing data availability
Habitat loss, landuse changes	Index of Biotic Integrity	NZFFD, RC, DOC	*Approach to account for natural processes *Identify links between IBI and pressures	NZFFD, Regional Council (RC), DOC
Habitat loss, landuse changes, presences of dams, exotic fishes	Native species richness	NZFFD, RC, DOC	*Approach to accounting for natural processes * Defining spatial areas with consistent environmental conditions	NZFFD, RC, DOC
Habitat loss, landuse changes, presences of dams, exotic fishes	Abundance of key indicator species	NZFFD, RC, MPI	*Approach to accounting for natural processes *TWINSPAN analysis	NZFFD, RC, MPI
Presence of dams/connectivity	Richness or diversity of diadromous species	NZFFD, RC	*Approach to accounting for natural processes *Index of climbing ability	NZFFD

6 Birds

6.1 What are “freshwater-dependent” birds?

The focus of this report is on riverine freshwater-dependent flora and fauna. Many of New Zealand’s birds are associated with fresh waters to a greater or lesser extent. In this chapter, the focus is on birds that depend on fresh waters for at least one aspect of their life, including feeding, breeding and nesting. Although other chapters are limited to river flora and fauna, MfE has requested that this chapter include all types of freshwaters, including rivers, lakes and wetlands.

6.2 What do birds tell us?

Birds are often considered to be good indicators of ecological health (Gregory et al. 2005) because they occupy a broad range of habitat types, are moderately abundant and have moderate body sizes and lifespans, making them likely to show population responses to environmental changes at moderate spatial and temporal scales. In addition, they are often near the top of the food chain, making them susceptible to bioaccumulation of potential toxins.

Changes in land use and increased demand for water are rapidly altering freshwater environments, with negative impacts on native species occupying these habitats. Many of New Zealand’s native birds regularly use freshwater environments, making them vulnerable to population declines through freshwater habitat degradation and increased predation. For example, poor water quality can reduce food availability, while changes to flow regimes (e.g., due to pressures such as water abstraction, flood control, impoundments) alter the availability and quality of instream freshwater habitats for birds and their prey (Hughey et al. 1989; Jowett & Biggs 2006; Gray & Harding 2007). Altered flow regimes also impact adjacent terrestrial habitats (Greenwood & Booker 2016), changing the distributions of invasive weeds and mammalian predators (Keedwell & Brown 2001; Caruso et al. 2013; Pickerell 2015; Brummer et al. 2016). Many of New Zealand’s native freshwater birds are threatened, partly due to degradation of the freshwater environment (O’Donnell et al. 2016), and further degradation through declining water quality, loss of habitat or continuing predation will likely result in declining populations and increased risk of extinction.

Birds are generally easy to detect, identify and survey, meaning that they are of interest to communities and, as such, are often well-surveyed, with good long-term historical data and spatial cover. For these reasons, birds are used as indicators of ecological health in several jurisdictions, including the United Kingdom, European Union and North America (Canterbury et al. 2000; Gregory et al. 2005; Melles 2005; Everard & Noble 2010; Gregory & Strien 2010) – see below.

6.3 Data availability

Unlike other jurisdictions, New Zealand has not had a systematic long-term approach to monitoring birds, making regular reporting on the state of bird populations, and potential drivers of change, difficult. Monitoring has largely focussed on a subset of threatened species, often over short timeframes or small spatial scales, to identify drivers of declines or the effects of management interventions. There are some potential datasets available that could inform an impact indicator for birds, but these have some limitations. The Department of Conservation’s Biodiversity Reporting and Monitoring System includes 5-minute bird counts to identify the species present at randomly located forested and non-forested sites on public conservation land (Lee et al. 2005). While this data is systematically collected at a national-scale, it is not appropriate for developing an impact indicator for freshwater birds as freshwater habitats are not specifically targeted. Fish & Game may have long-term records for game birds (including some native freshwater species) that could be appropriate for assessing population trends over time, but these data are limited to a small number of lake and wetland species. Birds New Zealand and local community groups also undertake monitoring of some freshwater bird species, although these often have limited spatial coverage and may not be representative of national trends. In addition, much of NZ's bird observation data is now recorded in online databases, such as eBird and NatureWatch, that allow researchers and citizen scientists to record sightings and access data collected by other people. Combining information from sources like those mentioned may provide sufficient information to assess trends in freshwater bird populations across New Zealand. However, the use of inconsistent collection methodologies may make it difficult to robustly assess long-term population trends and, therefore, limit the use of bird indicators that have been used overseas (e.g., Gregory et al. 2005; Everard & Noble 2010; Gregory & Strien 2010).

6.4 Development level of bird indicators in NZ

In the United Kingdom, European Union and North America long-term systematically-collected datasets allow a range of bird indicators to be used for assessing ecological health. These indicators include metrics of long-term trends in abundance for individual species of importance and composite indicators that combine individual species indices into a single indicator (Canterbury et al. 2000; Gregory et al. 2005; Melles 2005; Everard & Noble 2010; Gregory & Strien 2010). For example, the United Kingdom and European Union have developed composite bird indicators that combine data for select species within different habitats to assess long-term population trends (Gregory et al. 2005; Everard & Noble 2010; Gregory & Strien 2010).

Previous work in NZ using birds as biodiversity indicators of impacts has been somewhat limited. Walker & Monks (2004) used observational data collected by the Ornithological Society of New Zealand (OSNZ) in two time periods from 1969-1979 and 1999-2004 (OSNZ; Bull et al. 1985; Robertson et al. 2007) to estimate the status and change of species' ranges. They identified that forest and inland wading species showed range declines over this time-period, with these trends very likely due to pressure from mammalian predators and habitat loss due to intensive land use. These data represent two discrete national-scale surveys for all NZ's native birds. Repeating similar, systematic, national-scale surveys would be required to build on them for the purposes of environmental reporting. Other indicator work in New Zealand with birds has largely focussed on population trends in forest species with respect to predator control (e.g., Hoare et al. 2013).

Any indicator representing a change in abundance must be in relation to field data collected within the period of human habitation, with comparison to a reference point representing natural condition (e.g., pre-human state). To date, there has been no attempt to define a reference point for the pre-human abundance of different bird species in New Zealand, meaning that it is currently not possible to develop an indicator of change from natural condition.

Previous domain reports (Land and Marine) have included birds as case studies, using both conservation status and changes in conservation status as indicators of state and impact, respectively. The reports used the New Zealand Threat Classification System (Robertson et al. 2016) to identify the number of birds in each threat status category (state indicator – assessed by habitat guilds) and assess how many species have changed their conservation status over time (impact indicator – assessed alongside all indigenous species). A change in a species' conservation status reflects a change in its risk of extinction and may indicate that the ecosystem is degrading. The New Zealand Threat Classification System is updated every five years for each taxonomic group, with birds last updated in 2016 (Robertson et al. 2016).

The Marine Domain Report also included bycatch of protected seabirds as an impact indicator reflecting pressure from current fishing practices. The Land Domain Report assessed the number of bird species present on public conservation land as a state indicator (Bellingham et al. 2013), using data from Tier 1 of the Department of Conservation's Biodiversity Reporting and Monitoring System (Lee et al. 2005). This monitoring program uses 5-minute bird counts to identify the species present at randomly located forested and non-forested sites on public conservation land. While this data is systematically collected at a national-scale, it is not appropriate for developing an impact indicator for freshwater birds as there is currently insufficient data to assess trends over time and freshwater habitats are not specifically targeted.

6.5 Recommendations

We recommend that birds be included as impact indicators in the Freshwater Domain Report by assessing changes in the conservation status of native freshwater-associated species (see proposed list in Table 6-1). Changes can be assessed by comparing the number of species whose conservation status has changed between the 2012 and 2016 New Zealand Threat Classification System reports for birds (Robertson et al. 2013, 2017). This method would be comparable to the impact indicator used in the Land Domain Report for indigenous species. It is important to note that wide-ranging or migratory bird species can provide some challenges in developing impact indicators if their population dynamics are being driven by changes outside of New Zealand. However, this should not be a problem for most of New Zealand's freshwater-dependent bird species.

Table 6-1: List of New Zealand bird species associated with freshwater environments, ordered by family. The species list was compiled using published literature (Walker and Monks 2004; O'Donnell et al. 2015; O'Donnell et al. 2016; Robertson et al. 2017) and expert opinion (author, Colin O'Donnell Department of Conservation, John Whitehead Southland Conservation Board). The 2016 conservation status is based on Robertson et al. (2017).

Scientific Name	Common Name	Family	Conservation status
Circus approximans	Swamp harrier	Accipitridae	Not threatened
Todiramphus sanctus vagans	New Zealand kingfisher	Alcedinidae	Not threatened
Anas aucklandica	Auckland Island teal	Anatidae	Nationally vulnerable
Anas chlorotis	Brown teal	Anatidae	Recovering
Anas gracilis	Grey teal	Anatidae	Not threatened
Anas nesiotis	Campbell Island teal	Anatidae	Nationally vulnerable
Anas rhynchotis	Australasian shoveler	Anatidae	Not threatened
Anas superciliosa	Grey duck	Anatidae	Critically endangered
Anas superciliosa x platyrhynchus	Grey duck – mallard hybrid	Anatidae	Not threatened
Aythya novaeseelandiae	New Zealand scaup	Anatidae	Not threatened
Cygnus atratus	Black swan	Anatidae	Not threatened
Hymenolaimus malachorhynchos	Blue duck, whio	Anatidae	Nationally vulnerable
Tadorna variegata	Paradise shelduck	Anatidae	Not threatened

Scientific Name	Common Name	Family	Conservation status
Ardea modesta	White heron	Ardeidae	Critically endangered
Botaurus poiciloptilus	Australasian bittern	Ardeidae	Critically endangered
Egretta novaehollandiae	White-faced heron	Ardeidae	Not threatened
Anarhynchus frontalis	Wrybill	Charadriidae	Nationally vulnerable
Charadrius bicinctus bicinctus	Banded dotterel	Charadriidae	Nationally vulnerable
Charadrius obscurus obscurus	Southern New Zealand dotterel	Charadriidae	Critically endangered
Elseyornis melanops	Black-fronted dotterel	Charadriidae	Naturally uncommon
Vanellus miles novaehollandiae	Spur-winged plover	Charadriidae	Not threatened
Haematopus chathamensis	Chatham Island oystercatcher	Haematopodidae	Critically endangered
Haematopus finschi	South Island pied oystercatcher	Haematopodidae	Declining
Hirundo neoxena neoxena	Welcome swallow	Hirundinidae	Not threatened
Larus bulleri	Black-billed gull	Laridae	Critically endangered
Larus dominicanus dominicanus	Southern black-backed gull	Laridae	Not threatened
Larus novaehollandiae scopulinus	Red-billed gull	Laridae	Declining
Bowdleria punctata punctata	South Island fernbird	Megaluridae	Declining
Bowdleria punctata stewartiana	Stewart Island fernbird	Megaluridae	Nationally vulnerable
Bowdleria punctata vealeae	North Island fernbird	Megaluridae	Declining
Anthus novaeseelandiae novaeseelandiae	New Zealand pipit	Motacillidae	Declining
Phalacrocorax carbo novaehollandiae	Black shag	Phalacrocoracidae	Naturally uncommon
Phalacrocorax melanoleucos brevirostris	Little shag	Phalacrocoracidae	Not threatened
Phalacrocorax sulcirostris	Little black shag	Phalacrocoracidae	Naturally uncommon
Phalacrocorax varius varius	Pied shag	Phalacrocoracidae	Recovering
Podiceps cristatus australis	Southern crested grebe	Podicipedidae	Nationally vulnerable
Poliocephalus rufopectus	New Zealand dabchick	Podicipedidae	Recovering
Fulica atra australis	Australian coot	Rallidae	Naturally uncommon
Gallirallus australis australis	Western weka	Rallidae	Not threatened

Scientific Name	Common Name	Family	Conservation status
Gallirallus philippensis assimilis	Banded rail	Rallidae	Declining
Porphyrio melanotus melanotus	Pukeko	Rallidae	Not threatened
Porzana pusilla affinis	Marsh crake	Rallidae	Declining
Porzana tabuensis tabuensis	Spotless crake	Rallidae	Declining
Himantopus himantopus leucocephalus	Pied stilt	Recurvirostridae	Not threatened
Himantopus novaezelandiae	Black stilt	Recurvirostridae	Critically endangered
Chlidonias albostriatus	Black-fronted tern	Sternidae	Nationally endangered
Hydroprogne caspia	Caspian tern	Sternidae	Nationally vulnerable
Sterna striata aucklandorna	Southern white-fronted tern	Sternidae	Nationally vulnerable
Sterna striata striata	White-fronted tern	Sternidae	Declining
Platalea regia	Royal spoonbill	Threskiornithidae	Naturally uncommon

7 Ecosystem process indicators

The indicators discussed above mostly represent biodiversity, though some (especially periphyton and macrophytes) also have implications for ecosystem processes. Several other indicators of ecosystem processes were also considered. These included the Stream Ecological Valuation and ecosystem metabolism, which is composed of gross primary productivity and ecosystem respiration.

7.1 Stream Ecological Valuation (SEV)

7.1.1 What is the SEV?

The Stream Ecological Valuation (SEV; Storey et al. 2011) is an index that summarises fourteen ecological functions that occur in streams (Table 7-1). It is designed for quantifying the "contribution" a discrete stream reach makes to the ecological functioning of the whole stream network in a catchment. The fourteen functions, which may be considered as ecosystem processes, belong to four categories – hydraulic, biogeochemical, habitat provision and biodiversity. Each function is quantified by measuring the state of observable features in the stream reach that are known to perform that function. Most of the functions are not a direct measure of an ecosystem process. Rather, performance of the processes is inferred by the state of the stream features that perform or facilitate the process. Some of these features are measured using simple equipment such as tape measures or rulers. Others are assessed visually, by the observer selecting one of several narrative categories that best describes the stream reach, or the proportion of the reach that matches each category (e.g., Table 7-2). Functions are based on between one and four measured variables, which are combined in a mathematical equation giving greater weighting to variables that are more influential in the performance of the function. For example,

$$\text{natural flow regime} = (((2V_{chann} + V_{lining})/3) \times V_{pipe}$$

where V_{chann}, V_{lining} and V_{pipe} are variables related to channel modification, channel lining and stormwater pipes draining to the study reach.

The overall SEV score is an average of the fourteen function scores, all given equal weighting.

Its original purpose was to determine the compensation required for loss or degradation of that reach, though it has been used more generally to assess the condition of stream reaches, for example it has been used for State of Environment reporting by Auckland Council since 2009.

Table 7-1: Fourteen functions in the Stream Ecological Valuation.

Function type	Function	what does it measure?	what is it based on?
Hydraulic	Natural flow regime	the way water moves through a stream channel	channel modification, connected stormwater
	Floodplain effectiveness	connectivity between a channel and its floodplain	bank modification, state of riparian zone
	Connectivity for migrations	connectivity for fish and shrimp migrations	number and severity of migration barriers
	Connectivity to groundwater	extent of water exchange between channel surface and subsurface	channel lining and shape
Biogeochemical	Water temperature control	protection of water from warming	amount of channel shading
	Dissolved oxygen maintained	processes that remove oxygen from water	evidence of oxygen-reducing processes
	Organic matter input	amount of leaves and wood entering the channel	riparian vegetation type
	In-stream particle retention	retention of organic matter within the reach	macrophyte growth and channel straightening
	Decontamination of pollutants	potential to remove contaminants from water	surfaces that removal processes occur on
Habitat provision	Fish spawning habitat	spawning habitat for galaxiids and bullies	suitable riparian habitat and large rocks/wood in channel
	Habitat for aquatic fauna	water quality, physical habitat and hydrology for macroinvertebrates	water temperature and oxygen, impervious surface, in-channel habitat
Biodiversity	Fish fauna intact	native and invasive fish fauna present	Fish Index of Biological Integrity
	Invertebrate fauna intact	macroinvertebrate fauna	MCI, EPT richness, invertebrate loss index
	Riparian vegetation intact	condition and extent of riparian vegetation	riparian vegetation condition, connectivity between channel and riparian zone

Table 7-2: Example scoring table for V_{chann} (extent of channel modification). From Neale et al., (2011).

Type of channel modification	Proportion of channel (0-1)
Natural channel with no modification.	
Natural channel, but flow patterns affected by a reduction in roughness elements (e.g., woody debris, or boulders).	
Channel not straightened or deepened, but upper banks widened to increase flood flow capacity.	
Natural channel, but evidence of channel incision from flood flows.	
Natural channel, but flow patterns affected by increase in roughness elements (e.g., excessive macrophyte growth).	
Flow patterns affected by artificial in-stream structure (e.g., ponding due to culvert, weir or unnatural debris).	
Channel straightened and/or deepened.	

7.1.2 What does the SEV tell us?

The SEV function scores are designed to describe how well each ecological function is being performed relative to the natural, undisturbed (reference) condition. Reference condition can be defined by conducting an SEV assessment at a set of reference sites that are comparable to test sites in terms of stream type and geographic location. In cases where reference streams are not available, the SEV manual (Storey et al. 2011) gives guidance for developing a hypothetical reference condition. For most functions this is simply a matter of applying logic or common knowledge, whereas for a few (especially the biodiversity functions) it may require modelling, expert judgment or interpolation from known sites. All SEV functions, variables, and the overall SEV score, are scaled between 0 and 1, with 1 representing no significant deviation from natural condition and 0 representing complete loss or non-performance.

7.1.3 Data availability

SEV data are collected by developers and consultants for resource consents routinely in Auckland, Wellington, and sometimes in Whanganui-Manawatu and Hawke's Bay. However, the only extensive SEV data set suitable for reporting under the Environmental Reporting Act is for Auckland, as this is the only region where SEV is used by a council for State of Environment (SoE) monitoring.

A few of the 14 SEV functions could be calculated using the data currently available in regional council SoE data sets. For example, "invertebrate fauna intact" could be calculated from existing monitoring data, and "fish fauna intact" could be calculated using the same methods that are currently used in SoE monitoring. However, most of the other functions require data that are not part of most regional council protocols. These data, which are mostly visual observations related to physical habitat, are not difficult to collect, and could be collected in place of other protocols for physical habitat assessment with only brief training. However, this would require a shift in protocols.

In addition, calculating some SEV functions requires reference site data. Reference site data have been collected in Auckland, Wellington, and is being collected gradually in Whanganui-Manawatu. Other regions would have to build up a database of reference site data.

7.1.4 Development level of the SEV as an impact indicator

SEV was developed by consensus of an expert panel of 11 freshwater ecologists from NIWA, Auckland Council, Massey University, Landcare Research, Waikato Regional Council (Rowe et al. 2006, 2008). Where possible, the experts based the algorithms for the functions on published literature. SEV was revised after five years of use, based on user experience and recorded data over this period (Storey et al. 2011). The original method and the revised method have both been internationally peer-reviewed and published in international journals (Rowe et al. 2009; Neale et al. 2017).

SEV was originally designed for headwater urban streams in Auckland, but it has been validated for a variety of stream types in several regions, including Hawke's Bay, Wellington, Southland and Whanganui-Manawatu (Parkyn 2008, Storey 2009, Storey 2011, Storey 2013, respectively). It is considered applicable to streams up to fourth order in any region of NZ, including hard- and soft-bottomed, lake-fed, spring-fed, and intermittent streams (Storey et al. 2011, Neale et al. 2016). It is not appropriate for saltwater-influenced stream reaches.

7.1.5 Summary and recommendations

The following features make SEV suitable as an impact indicator:

- It neatly summarises a range of ecological functions or ecosystem processes.
- It is a composite index that can be reported as a single number for brevity or broken into its component parts for in-depth analysis.
- It inherently measures deviation from natural, undisturbed (i.e., reference) condition
- The ecological functions are intuitive, so are relatively easily interpreted and communicated.
- It uses methods that would be easily learned by regional council field staff. It is not very different to current protocols for physical habitat and biological monitoring.

However, the SEV has the following drawbacks:

- Some of the variables measured by observer judgment rather than measurements, meaning they are potentially vulnerable to observer bias or differences between observers. This problem also exists for other physical habitat assessment methods. It can be reduced somewhat by "cross-calibrating" different observers, but this would be a major exercise at a national scale, particularly as much regional council monitoring data is collected by seasonally-employed staff.
- Currently, there is a lack of an extensive monitoring network for regularly-collected data in all regions except Auckland.

For this last reason, despite its suitability, SEV is unlikely to be a suitable indicator of impact on ecosystem processes for the 2020 Water Domain report. Future use of the SEV in this way could be achieved, but would require councils to adopt the SEV method for their regular SoE monitoring to ensure a coordinated, national programme of data collection.

7.2 Ecosystem metabolism

7.2.1 What is ecosystem metabolism?

Ecosystem metabolism is a measure of how much organic carbon is produced and consumed in river ecosystems. Organic carbon is produced by photosynthesis in aquatic plants (periphyton and macrophytes). The plants are referred to as "primary producers", since they create organic carbon "food" for other organisms using inorganic compounds and sunlight, and the process is referred to as gross primary production, or GPP. Organic matter is consumed by a variety of animals, fungi, bacteria and plants via the cellular process of respiration. "Ecosystem respiration" (ER) is the total respiration from all organisms within a given area. Ecosystem metabolism is the balance between GPP and ER and can be formally measured as the ratio of GPP/ER.

The ecosystem metabolism of a river can be estimated by measuring the daily changes in dissolved oxygen concentration and estimating or measuring reaeration (i.e., gas exchange across the water surface). Oxygen is constantly being removed from the water by respiration, however, during the day algae and aquatic plants photosynthesise and offset consumption by releasing oxygen into the water. This leads to a characteristic daily pattern in dissolved oxygen concentration, which can be substantial in rivers with a high biomass of aquatic plants.

Metabolism can be estimated by measuring natural changes in dissolved oxygen concentration in the river (Bernhardt et al. 2018). These measurements are relatively simple and require just one data-logging dissolved oxygen meter in stream or river reaches lacking significant upstream inputs of organic pollutants (e.g., BOD from sewage treatment plants) and that are relatively uniform upstream for a distance equivalent to many hours of water travel time. More complex situations require two-station methods to measure metabolism over a reach (length that gives sufficient travel time to detect changes in DO between sites, c. 2-4 hours) between paired dissolved oxygen meters. Oxygen concentrations are measured with the oxygen logger at regular intervals over at least one 24-hour period, and changes in concentration are related to oxygen input via photosynthesis and oxygen removal via respiration. Determining ecosystem metabolism from these measurements requires an estimate of reaeration rate (the amount of oxygen diffusing between the air and water; Appling et al. 2018) in the stream, but tools are available for making these adjustments.

7.2.2 What does ecosystem metabolism tell us?

Organic matter in streams is either produced within the stream as the biomass of periphyton and macrophytes (primary producers), or imported from the surrounding terrestrial environment as leaves and wood. Aquatic plant biomass and terrestrial leaves and wood represent the two main sources of energy that fuel production of higher life-forms in river ecosystems.

The balance between organic carbon production (GPP) and consumption (ER) provides information on the relative importance of these two key sources of energy. If organic carbon production equals or exceeds carbon consumption then organic matter produced within the system is probably supporting the food web, whereas if carbon consumption greatly exceeds carbon production then either organic matter from upstream or the surrounding catchment is being used to maintain the system, or the system is gradually consuming the organic matter stock.

Metabolism measurements are representative of the entire reach and cover the range of habitat types present, even though the oxygen concentrations used to calculate metabolism are only measured at one or two specific locations, because of the natural movement and mixing of water in a river (Young et al. 2008).

Ecosystem metabolism is influenced by a wide range of factors. Some factors vary naturally (e.g., longitudinal position in the river, climate), whereas others are influenced primarily by anthropogenic disturbance to ecosystems (e.g., organic pollution, river regulation, toxic chemicals, aquatic plant management). However, many of the factors that control ecosystem metabolism vary as a result of both natural and anthropogenic causes (e.g., light, substrate composition, turbidity, nutrients, pH, riparian vegetation, flow fluctuations).

The amount of light reaching primary producers on the streambed appears to be the main factor influencing GPP in rivers (Young et al. 2008; Bernhardt et al. 2018). Factors influencing the amount of light at the stream bed include natural factors such as the orientation of the valley, slope of the banks amount and factors that are often strongly affected by human impacts such as water clarity and the type of riparian vegetation.

The probable effects of natural variation on rates of metabolism must be understood and taken into account when designing monitoring programs or when interpreting data. For example, GPP/ER typically changes in a predictable way from the headwaters to the lower reaches of natural river systems, particularly as light first increases with widening river channels then decreases with increasing turbidity (Young et al. 2008; Table 7-3). Thus, metabolic rates at potentially impacted sites should be compared with rates measured at (more) pristine sites that are characterised by similar stream order and size.

Table 7-3: Expected patterns in gross primary productivity (GPP) and ecosystem respiration (ER) and in the ratio of photosynthesis (gross primary production) to ecosystem respiration (GPP/ER) in relation to natural variation and responses to environmental stressors (adapted from Young et al. 2008).

Factor	Change	Response	Comments
Position from headwaters to river mouth	Forested headwaters: dense shade	Decrease GPP (GPP/ER << 1)	GPP light limited, but not in grassland (GPP/ER high)
	Middle section: more light	Increase GPP (GPP/ER ~ 1)	GPP/ER decreases downstream in grassland
	Lower river: deep, turbid	Decrease GPP (GPP/ER < 1)	Where strong floodplain connection, high organic input and ER (GPP/ER<<1)
Influential species	Trout reduce insect grazing, increase algae	Increase GPP and GPP/ER	Algal biomass higher, GPP sometimes higher

Factor	Change	Response	Comments
Light	More sunlight	Increase GPP and GPP/ER	Mainly based on the amount of light reaching the riverbed; affected by season, cloud cover, canopy cover, and turbidity
Temperature	Warmer water	Increase ER, possibly GPP	Only weak evidence
Nature of substrate	More fine sediment	Increase ER, decrease GPP/ER	More organic matter
	Less stable or more heterogeneous substrate	Decrease GPP, decrease GPP/ER	Algal production higher on large stable particles
	Impaired connection with hyporheic zone	Decrease ER, increase GPP/ER	Large proportion of ER occurs in hyporheic zone
Turbidity	More suspended sediment	Decrease GPP, decrease GPP/ER	If river depth is sufficient to limit light
Nutrients	Nutrient enrichment	Increase GPP and ER	GPP and ER are useful predictors of dissolved nutrient attenuation rates
Organic pollution	Input of organic waste	Increase ER, decrease GPP/ER	Possible increase in GPP too, if nutrients released
Toxic chemicals	Toxic inputs	Decrease GPP and ER	May be offset by nutrients in toxic discharge
Riparian vegetation	Lose stream-side vegetation, increase light	Increase GPP and GPP/ER	ER may also increase if system is dominated by algal respiration
	Increase organic matter inputs	Increase ER, decrease GPP/ER	Some trees (especially deciduous) drop more leaves or have seasonal pulses
Channelization	Loss of habitat heterogeneity	Increase GPP, increase GPP/ER	Loss of riparian cover partly responsible
Flow fluctuations	Floods	Decrease GPP, ER (a little), decrease GPP/ER	High flows and abrasion reduce algal biomass
	River regulation	Increase GPP and ER	Loss of flushing flows enhances periphyton and macrophyte accrual
Aquatic plant management	Plant removal	Decrease GPP and ER	Only if macrophytes contribute strongly to metabolism

7.2.3 Data availability

Almost all regional councils deploy dissolved oxygen loggers at a few State of Environment or other monitoring sites, thus have the data required to calculate ecosystem metabolism at these sites. However, a few regions do not deploy any oxygen loggers, and in no region are loggers deployed at more than about 10 sites (Juliet Milne, NIWA, pers. comm.). This total number is comprised of a few permanent deployments (<5 sites in each region except Auckland, which has 10) and a few seasonal deployments (up to 9 sites, varying by region). Therefore, there is currently insufficient data coverage to use measures of ecosystem metabolism for national scale reporting. The main reason that oxygen loggers are limited is the cost of staff time and/or maintenance. For short deployments, deploying and retrieving the loggers is labour-intensive, whereas for long deployments, loggers must be serviced to reduce fouling. These costs may continue to limit the use of oxygen loggers to a small proportion of monitoring sites. Continuous measurements of dissolved oxygen are likely to increase somewhat in future, partly due to legal requirements and expected declining costs of probes. Under the NPS-FM (2017), councils are required to measure continuous dissolved oxygen downstream of point source discharges. However, such data collected in response to this policy are clearly biased towards more degraded sites and by themselves are not useful for State of Environment reporting.

In addition to the costs of data collection, metabolism is difficult to measure in small, turbulent streams with low productivity, and this difficulty limits use of this method in such streams (Young et al. 2008).

7.2.4 Development level of ecosystem metabolism indicators

The method for calculating GPP and ER is well established, and tools are available to account for re-aeration rates in streams. However, use of GPP and ER as impact indicators at a national scale also requires that results can be interpreted relative to pressures and that effects of natural factors can be accounted for.

At a national scale, the main pressures on aquatic ecosystems are related to catchment land use intensity. A typical gradient of land use intensity includes replacement of forest with pastoral farming, and increasing urban intensity represented by % cover of impervious surface. Several factors that directly influence GPP and ER change together along this land use gradient, but as Table 7-3 indicates, some of these have opposite effects on GPP and ER. For example, greater % pastoral land cover (reduced forest cover) is typically associated with increased light, nutrient concentrations, turbidity, temperature and flood flows in streams. While increased light and nutrient concentrations increase GPP, increased turbidity and flood flows decrease it.

Despite these compensatory effects, Clapcott et al. (2010) were able to show positive correlations between GPP and each of % impervious cover, % vegetation cover removal and predicted nitrogen concentration. Each of these correlations was non-linear, with apparent thresholds at <10% impervious cover, between 40 and 80% loss of native vegetation cover, and at 0.5 and 3.2 mg L^{-1} N. ER had an overall positive relationship with % vegetation cover removal, but showed curvilinear responses to % impervious cover and nitrogen concentration. The mechanisms for the curvilinear responses were not clear. However, in general ER can show a curvilinear response across a land use gradient as respiration is low both in natural streams with low nutrient concentrations and water temperature (Acuna et al. 2008), and also in highly impacted streams where sedimentation may block the connection between surface waters and the hyporheic zone (Wilson & Dodds 2009).

Clapcott et al. (2010) were also able to show that the responses of GPP and ER to the three measures of land use intensity varied little among three bioregions of New Zealand. Although the relationships between GPP, ER and the three land use measures showed high variability, they concluded that GPP and ER exhibit predictable relationships with land use.

A framework for interpreting values of GPP and ER in terms of ecosystem health has been established, based on international data (Young et al. 2008; Young et al. 2016; Table 7-4). The framework includes condition bands for absolute values (GPP alone, ER alone and GPP/ER ratio) and values of "test sites" relative to reference sites. Because the dataset for basing these condition bands comes from various countries and a wide range of stream types, the framework is very broad, and could be tightened considerably by developing criteria based on appropriate local reference sites.

For example, light input and shading affect GPP, and patterns of GPP vs ER differed markedly between reference sites with closed canopies (smaller or forested sites) and open canopies (larger or grassland sites) (Young et al. 2008). Values of GPP and (especially) ER also differ between small streams and large non-wadeable rivers (Clapcott et al. 2015). Therefore, if using absolute values, condition bands for GPP and ER should be determined separately for closed- vs. open-canopy streams and for small streams vs. large rivers (Clapcott et al. 2015). If using ratios of test site to reference site values, reference sites must be of a similar stream type to the test site.

An alternative to using single values of GPP and ER is an index of temporal variability in GPP and ER. Clapcott et al., (2016) found that sites with high land use stress show greater day-to-day and between-season variability than sites with lower stress. Variability in both GPP and ER was positively correlated with total nitrogen and negatively correlated with Macroinvertebrate Community Index, % riparian shade and riparian condition. GPP was also correlated with mean monthly water temperature, and ER with correlated with *E. coli* concentration. An index for reporting based on variability would require more development, but this could be done using existing data (J. Clapcott, Cawthron Institute, pers. comm). It would also require loggers to be deployed multiple times during summer and among seasons.

Table 7-4: Framework for assessing stream health using gross primary productivity (GPP) and ecosystem respiration (ER) data. The scores indicate the health of the test site: 2 = no evidence of an impact on ecosystem functioning, 1 = mild effect on ecosystem functioning, 0 = severely impaired ecosystem functioning. Subscript t = test site, subscript r = reference site. From Young et al. (2008) and Parkyn et al. (2010).

Method	Assessment parameter	Criterion	Score
Comparison with reference	GPP_t/GPP_r	GPP_t/GPP_r <2	2
		GPP_t/GPP_r = 2.5–5.0	1
		GPP_t/GPP_r > 5.0	0
	ER_t/ER_r	ER_t/ER_r = 0.4-1.6	2
		ER_t/ER_r = 0.2–0.4 or 1.6–2.7	1
		ER_t/ER_r < 0.2 or ER_t/ER_r > 2.7	0

Method	Assessment parameter	Criterion	Score
Absolute value	GPP_t (g $O_2/m^2/d$)	$GPP_t < 3.5$	2
		$GPP_t = 3.5–7.0$	1
		$GPP_t > 7.0$	0
	ER_t (g $O_2/m^2/d$)	$ER_t = 0.8–1.6$ or $ER_t = 5.8–9.5$	1
		$ER_t < 0.8$ or $ER_t > 9.5$	0
	GPPt/ERt	$GPP_t/ER_t < 1.3$	2
		$GPP_t/ER_t = 1.3-2.5$	1
		$GPP_t/ER_t > 2.5$	0

7.2.5 Summary and recommendations

Gross Primary Productivity and Ecosystem Respiration are potentially suitable indicators of impact on ecosystem processes because:

- they are direct measures ecologically-relevant ecosystem processes (the balance between energy supply and demand in river ecosystems)
- their meaning and interpretation are relatively easily explained to non-experts
- measurements are representative of an entire reach, even though measurements are made at one or two specific locations
- field data can be collected in as little as two days from a single deployment.

Methods for calculating GPP and ER are well established, and condition bands have been determined for New Zealand streams. However, further refinement of condition bands is desirable for different stream types. This could be done with only modest additional investment. Responses to land use stresses are well understood for GPP, and but less so for ER. A more sensitive indicator (temporal variability of GPP and ER) could be developed with modest investment.

Nevertheless, the use of GPP and ER as national indicators is limited because of the labour costs of deploying, retrieving and maintaining the oxygen loggers. Although data coverage is likely to increase in future, it is unlikely to reach the same level of coverage as the biodiversity indicators described in earlier sections, and unless continuous oxygen monitoring is mandated in future, measures of ecosystem metabolism are likely to remain as case studies for places where data are available.

8 Summary and Recommendations

In this report we discuss indicators of impacts on biodiversity of five organism groups and on three measures of ecosystem processes, with the discussion limited to stream and river ecosystems. A further restriction is that most of these indicators and the protocols for data collection have been developed primarily for wadeable (i.e., shallower than chest-deep) streams. The indicators and/or field protocols for periphyton, macrophytes, macroinvertebrates and Stream Ecological Valuation may require further validation to be used in deep rivers.

For periphyton, three potential indicators are discussed: biomass as surface density of chlorophyll *a*, "weighted composite cover" of periphyton mats and filaments, and an index of community composition. Of these, chlorophyll *a* is currently the most suitable as a national indicator as it requires no further development, has well established guideline values and extensive data coverage.

For macrophytes we discuss two indicators: presence/absence of macrophyte species and biovolume of native vs. introduced species. The latter provide more useful and robust information as a national indicator. Data that meet Tier 1 criteria are being collected in some regions, and other regions are gradually incorporating macrophyte data in monitoring programmes.

For macroinvertebrates we discuss three widely used indices (Macroinvertebrate Community Index, EPT richness and % EPT richness), which could be combined for greater discriminatory power. Condition bands (to interpret values in terms of stream ecosystem health) have been determined for MCI, and could be refined by indexing against reference values which have been determined for each stream reach in the country. With modest extra investment, a more powerful index of macroinvertebrate biodiversity could be developed for New Zealand by building on existing information for reference condition.

For freshwater fish we discuss four impact indicators that are in various stages of development. Each of these could be developed for use as a national indicator with modest investment. Indices based on abundance data are more sensitive and informative than those based on presence-absence. However, abundance data in the NZ Freshwater Fish Database (NZFFD) (the main data source) are currently limited, and are likely to remain so due to the effort required to collect abundance data. We recommend temporal changes in Indigenous Species Richness (ISR), relative to a baseline year, as the most informative and easily developed impact indicator for fish biodiversity. Ideally, ISR requires both native species richness and Shannon's Diversity to be calculated, and the latter require abundance data of the entire fish community at each site. Relevant data are available in the NZFFD, sufficient (we believe) to determine annual changes since 1977. In future, reporting ISR as deviation from reference condition may be possible. Abundance/biomass of key indicator species is the next most-preferred index but this also requires abundance data for the indicator species. If abundance-based indicators cannot be used as national indicators, the Index of Biotic Integrity potentially provides a holistic score that describes how fish communities respond to anthropogenic processes, but requires more work to develop and test than the connectivity indicator.

For freshwater birds, only one indicator is possible because regular monitoring is not done in a systematic way across the country. This indicator is conservation status of individual freshwater-dependent bird species. This indicator is already used in the Land and Marine domain reports.

Three indicators of ecosystem processes were discussed. The Stream Ecological Valuation is framework for indirectly measuring ecological functions (processes) using state indicators. The way it is constructed and the breadth of its scope make it a good candidate for an indicator of impact, however, it is currently used for routine monitoring only in Auckland region. Therefore, at present it could be used only as a case study indicator, though data collection methods are not difficult for councils to adopt if it were regarded as a useful national indicator for the future.

Gross Primary Production and Ecosystem Respiration are direct measures of energy production and consumption in stream food webs. They have been shown to vary in predictable ways with land use stress and the individual stressors associated with land use change. Data interpretation is possible by comparing values to established condition bands, though further work is required to refine these for different stream and river types. The main limitation with using these as indicators of impact, however, is the extent of data coverage. Data are likely to remain limited to a small proportion of council monitoring sites for the near future, thus these indicators may be most suitable as case studies.

Organism group or ecosystem process	Recommended indicators	Data availability	Development level of indicator	Work needed
Periphyton	Biomass (surface density of chlorophyll a)	Comparable monthly data from most regions; annual data from some regions	Fully developed for hard-bottomed streams; sampling methods and guideline values needed for epiphyton in soft-bottomed streams	None
	"Weighted composite cover" of periphyton mats and filaments	Comparable monthly data from most regions	Fully developed for hard-bottomed streams	None
	Index of community composition	Data from some regions from some years; nationwide sample set exists and could be analysed to provide baseline dataset	Limited development	Test applicability of European indices or identify NZ indicator species
Macrophytes	Presence/absence of macrophyte species	Only one region collecting all required data	Fully developed	Most councils to begin monitoring macrophytes; some councils to modify existing protocols. Both could be achieved by a National Environment Monitoring Standard (NEMS)
	Biovolume of native vs. introduced species	Only one region collecting all required data	Fully developed	Most councils to begin monitoring macrophytes; some councils to modify existing protocols. Both could be achieved by a National Environment Monitoring Standard (NEMS)
Macroinvertebrates	Combination of MCI, EPT richness and % EPT richness and IBI	Comparable annual data from all regions	Indices and reference condition values fully developed	None

Organism group or ecosystem process	Recommended indicators	Data availability	Development level of indicator	Work needed
	Index of macroinvertebrate biodiversity	Comparable annual data from all regions	Overseas examples can be adopted in NZ	Define reference species assemblages for different stream reaches
Freshwater fish	Index of Biotic Integrity	NZ Freshwater Fish Database	Developed but with issues	1. Account for sampling bias and natural processes; 2. Determine the pressures that drive the IBI and/or its 12 individual components
	Indigenous Species Richness	NZ Freshwater Fish Database; one component requires abundance data, which is sparse in NZFFD	Direct field measure; but defining reference condition requires determining natural spatial variations	Account for natural spatial variability
	Abundance or biomass of indicator species	NZ Freshwater Fish Database; but abundance data sparse	Requires identifying suitable indicator species. Longfin eels recommended	Identifying other indicator species require statistical analysis of NZFFD
	Diadromous species richness (or Shannon Diversity Index)	NZ Freshwater Fish Database	Direct field measure; but defining reference condition requires determining natural spatial variations; indicator could be further developed by incorporating individual species' climbing abilities	Incorporate individual species' climbing abilities
Freshwater-dependent birds	Change in the conservation status of native freshwater-associated species	2012 and 2016 New Zealand Threat Classification System reports for birds	Fully developed	None
Stream Ecological Valuation	SEV	Comprehensive for Auckland; some data in Wellington	Fully developed	Adopt SEV protocols in all regions

Organism group or ecosystem process	Recommended indicators	Data availability	Development level of indicator	Work needed
Ecosystem Metabolism	GPP	10 or fewer sites per region; none in some regions	Index calculation developed; guidelines for interpretation needed	Guideline values need developing, based on analysis of larger nationwide dataset
	ER	10 or fewer sites per region; none in some regions	Index calculation developed; guidelines for interpretation needed	Guideline values need developing, based on analysis of larger nationwide dataset

9 References

Acuna, V., Giorgi, A., Muñoz, I., Uehlinger, U., Sabater, S. (2004) Flow extremes and benthic organic matter shape the metabolism of a headwater Mediterranean stream. Freshwater Biology, 49(7): 960-971.

Allibone, R.M., David, B.O., Hitchmough, R., Jellyman, D.J., Ling, N., Ravenscroft, P., Waters, J.M. (2010.) Conservation status of New Zealand freshwater fish, 2009. New Zealand Journal of Marine and Freshwater Research, 44: 271-287.

Appling, A., Hall, R., Yackulic, C., Arroita, M. (2018) Overcoming equifinality: Leveraging long time series for stream metabolism estimation. Journal of Geophysical Research: Biogeosciences, 123 (2): 624-645. https://doi.org/10.1002/2017JG004140

Baker, C.F., Jellyman, D.J., Reeve, K., Crow, S.K., Stewart, M., Buchinger, T., Li, W. (2017) First observations of spawning nests in the pouched lamprey (*Geotria australis*). Canadian Journal of Fisheries and Aquatic Science, 74: 1603-1611.

Bellingham, P.J., Richardson, S.J., Gormley, A.M., Macleod, C.J., Forsyth, D.M. (2013) Department of Conservation biodiversity indicators: 2013 assessment. Landcare Research report prepared for Department of Conservation. Lincoln.

Bernhardt, E.S., Heffernan, J.B., Grimm, N.B., Stanley, E.H., Harvey, J.W., Arroita, M., Appling, A.P., Cohen, M.J., McDowell, W.H., Hall, R.O., Read, J.S., Roberts, B.J., Stets, E.G., Yackulic, C. (2018) The metabolic regimes of flowing waters. Limnology and Oceanography, 63: S99-S118.

Besse-Lototskaya, A., Verdonschot, P.F.M., Coste, M., Van de Vijver, B. (2011) Evaluation of European diatom trophic indices. Ecological Indicators, 11: 456-467.

Biggs, B., Price, G. (1987) A survey of filamentous algal proliferations in New Zealand rivers. New Zealand Journal of Marine and Freshwater Research, 21(2): 175-191.

Biggs, B.J.F. (2000) New Zealand periphyton guideline: detecting, monitoring and managing enrichment of streams. Ministry for the Environment.

Biggs, B.J.F., Francoeur, S.N., Huryn, A.D., Young, R., Arbuckle, C.J., Townsend, C.R. (2000) Trophic cascades in streams: effects of nutrient enrichment on autotrophic and consumer benthic communities under two different fish predation regimes. Canadian Journal of Fisheries and Aquatic Sciences, 57(7): 1380-1394.

Biggs, B.J.F., Kilroy, C. (2000) Stream Periphyton Monitoring Manual. Published by NIWA for Ministry for the Environment.

Biggs, B.J.F., Nikora, V.I., Snelder, T.H. (2005) Linking scales of flow variability to lotic ecosystem structure and function. River Research and Applications, 21(2-3): 283-298.

Biggs, B.J.F., Stockseth, S. (1996) Hydraulic habitat suitability for periphyton in rivers. Regulated Rivers: Research & Management, 12: 251-261.

Biggs, B.J.F., Tuchman, N.C., Lowe, R.L., Stevenson, R.J. (1999) Resource stress alters hydrological disturbance effects in a stream periphyton community. Oikos, 85(1): 95-108.

Bonnett, M.L., Crow, S.K., Jellyman, P.G. (2013) A review of freshwater fish resources of the Waitaki River catchment. NIWA Client Report CHC2013-078 :63.

Booker, D., Snelder, T. (2012) Development of nutrient criteria for managing macrophytes in lowland and spring-fed Canterbury streams. NIWA report for Environment Canterbury. January 2012.

Booker, D.J., Whitehead, A.L., Henderson, R.D., Yang, J. (2016) National environmental indicators for water allocation. NIWA client report to Ministry for the Environment.

Bothwell, M.L. (1988) Growth rate responses of lotic periphytic diatoms to experimental phosphorus enrichment: The influence of temperature and light. Canadian Journal of Fisheries and Aquatic Sciences, 45: 261 - 270.

Brummer, T.J., Byrom, A.E., Sullivan, J., Hulme, P.E. (2016) Alien and native plant richness and abundance respond to different environmental drivers across multiple gravel floodplain ecosystems. Diversity and Distributions, 22:1–13.

Bull, P.C., Gaze, P.D., Roberston, C., Rutherford, J. (1985) The atlas of bird distribution in New Zealand. Ornithological Society of New Zealand, Wellington, NZ.

Burnet, A.M.R. (1952) Studies on the Ecology of the New Zealand Long-finned Eel, *Anguilla dieffenbachii* Gray. Marine and Freshwater Research, 3: 32-63.

Busse, L.B., Simpson, J.C., Cooper, S.D. (2006) Relationships among nutrients, algae, and land use in urbanized southern California streams. Canadian Journal of Fisheries and Aquatic Sciences, 63(12): 2621-2638.

Butterwick, C., Heaney, S.I., Talling, J.F. (2005) Diversity in the influence of temperature on the growth rates of freshwater algae, and its ecological relevance. Freshwater Biology, 50(2): 291-300.

Cairns, J., Pratt, J.R. (1993) A history of biological monitoring using benthic macroinvertebrates. In: Rosenberg, D.M. & Resh, V.H. (eds.). Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, London.

Canterbury, G.E., Martin, T.E., Petit, D.R., Petit, L.J., Bradford, D.F. (2000) Birds communities and habitats as ecological indicators of forest condition in regional monitoring. Conservation Biology, 14:544–558.

Caraco, N.F., Cole, J.J. (2002). Contrasting impacts of a native and alien macrophyte on dissolved oxygen in a large river. Hydrobiologia, 441: 1-12.

Caruso, B.S. (2006) Project River Recovery: restoration of braided gravel-bed river habitat in New Zealand’s high country. Environmental management, 37: 840–61.

Caruso, B.S., Pithie, C., Edmondson, L. (2013) Invasive riparian vegetation response to flow regimes and flood pulses in a braided river floodplain. Journal of Environmental Management, 125: 156–168. Elsevier Ltd.

Champion, P.D., Tanner, C.C. (2000). Seasonality of macrophytes and interaction with flow in a New Zealand lowland stream. Hydrobiologia, 441: 1–12.

Chapra, S.C., Flyle, K.F., Rutherford, J.C. (2014) Parsimonious model for assessing nutrient impacts on periphyton-dominated streams. Journal of Environmental Engineering, 140(6): 04014014.

Chisnall, B.L. (1987) Juvenile Eel Biology in the Backwaters of the Waikato River. Master of Science thesis, University of Waikato, New Zealand: 152.

Chisnall, B.L., Hicks, B.J. (1993) Age and growth of longfinned eels (*Anguilla dieffenbachii*) in pastoral and forested streams in the Waikato River basin, and in two hydro-electric lakes in the North Island, New Zealand. New Zealand Journal of Marine and Freshwater Research, 27: 317-332.

Clapcott, J., Collier, K.J., Death, R.G., Goodwin, E.O., Harding, J.S., Kelly, D., Leathwick, J.R., Young, R.G. (2011) Quantifying relationships between land-use gradients and structural and functional indicators of stream ecological integrity. Freshwater Biology, 57: 74-90.

Clapcott, J., Wagenhoff, A., Neale, M., Storey, R., Smith, B., Death, R., Harding, J., Matthaei, C., Quinn, J., Collier, K., Atalah, J., Goodwin, E., Rabel, H., Mackman, J., Young, R. (2017) Macroinvertebrate metrics for the National Policy Statement for Freshwater Management. Prepared for the Ministry for the Environment. Cawthron Report No. 3073.

Clapcott, J., Young, R., Neale, M., Doehring, K., Barmuta, L. (2016) Land use affects temporal variation in stream metabolism. Freshwater Science, 35(4): 1164-1175.

Clapcott, J.E. (2015) Development of management bands for ecosystem metabolism in non-wadeable rivers. Cawthron Report, No. 2770 prepared for Waikato Regional Council: 21 p. plus appendix.

Clapcott, J.E., Young, R.G., Goodwin, E.O., Leathwick, J.R. (2010) Applied Issues: Exploring the response of functional indicators of stream health to land-use gradients. Freshwater Biology, 55(10): 2181-2199.

Clapcott, J.E., Young, R.G., Harding, J.S., Matthaei, C.D., Quinn, J.M., Death, R.G. (2011) Sediment Assessment Methods: Protocols and guidelines for assessing the effects of deposited fine sediment on in-stream values. Cawthron Institute, Nelson, New Zealand.

Closs, G.P., Franklin, P., David, B.O., West, D., Crow, S.K., Allibone, R.M., Ling, N., Hitchmough, R. (2014.) *Geotria australis*. The IUCN Red List of Threatened Species 2014. http://dx.doi.org/10.2305/IUCN.UK.2014-3.RLTS.T197275A2480788.en. Downloaded on 18 May 2018

Collier, K.J., Champion, P.D., Croker, G.F. (1999) Patch and reach scale dynamics of a macrophyte invertebrate system in a New Zealand lowland stream. Hydrobiologia, 392: 89-97.

Collier, K.J., Hamer, M., Champion P. (2014). Regional guidelines for ecological assessments of freshwater environments: Aquatic plant cover in wadeable streams - version 2. Waikato Regional Council Technical Report, 2014/03: 33.

Collier, K.J., Kelly, J., Champion, P. (2006) Regional guidelines for ecological assessments of freshwater environments: Aquatic plant cover in wadeable streams. Waikato Regional Council Technical Report, 2006/47.

Collier, KJ., Clapcott, J., Neale, M. (2014) A macroinvertebrate attribute to assess ecosystem health for New Zealand waterways for the national objectives framework – Issues and options. Environmental Research Institute report, 36, University of Waikato, Hamilton.

Courtemanch, D.L., Davies, S.P. (1987) A coefficient of community loss to assess detrimental change in aquatic communities. Water Research, 21(2): 217-222.

Crow, S.K., Booker, D.J., Sykes, J.R.E., Unwin, M., Shankar, U. (2014) Predicting distributions of New Zealand freshwater fishes. NIWA Client Report CHC2014-145: 94.

Davies-Colley, R.J., Quinn, J. (1998) Stream lighting in five regions of North Island, New Zealand: control by channel size and riparian vegetation. New Zealand Journal of Marine and Freshwater Research, 32: 591-605.

Dawson, F.H., Haslam, S.M. (1983) The management of river vegetation with particular reference to shading effects of marginal vegetation. Landscape Planning, 10: 147-169.

Dawson, F.H., Kern-Hansen, U. (1979) The effect of natural and artificial shade on the macrophytes of lowland streams and the use of shade as a management technique. Internationale Revue Gesamten Hydrobiologie, 64: 437-455.

Depree, C.et al., (2017) Development of ecosystem health thresholds for suspended and deposited sediment in New Zealand rivers and streams. NIWA Client Report 2017076HN: 315.

Dodds, W.K. (2003) The role of periphyton in phosphorus retention in shallow freshwater aquatic systems. Journal of Phycology, 39(5): 840-849.

Duncan, M.J., Kilroy, C., McBride, G., Burnip, G., Bullians, M. (2005) (2005) A New Zealand-wide survey in October 2005 for the presence of the non-indigenous freshwater diatom *Didymosphenia geminata* in sites most likely for its introduction and establishment. NIWA Consultancy Report CHC 2005-141. For Biosecurity New Zealand: 112.

Ellenberg, H. (1988) Vegetation ecology of Central Europe, 4th edition. Cambridge: Cambridge University Press.

Entwisle, T., Foard, H. (1997) Batrachospermum (Batrachospermales, Rhodophyta) in Australia and New Zealand: new taxa and emended circumscriptions in Sections Aristata, Batrachospermum, Turfosa and Virescentia. Australian Systematic Botany, 10: 331-380.

Everard, M., Noble, D. (2010) The development of bird indicators for British fresh waters and wetlands. Aquatic Conservation: Marine and Freshwater Ecosystems, 20: S117–S124.

Francoeur, S.N., Biggs, B.J.F. (2006) Short-term effects of elevated velocity and sediment abrasion on benthic algal communities. Hydrobiologia, 561: 59-69.

Franklin, P., Dunbars, M., Whitehead, P. (2008) Flow controls on lowland river macrophytes: a review. Science of the Total Environment, 400: 369-378.

Fullerton, A.H., Burnett, K.M., Steel, E.A., Fitcroft, R.L., Pess, G.R., Feist, B.E., Torgersen, C.E., Miller, D.J., Sanderson, B.L. (2010) Hydrological connectivity for riverine fish: measurement challenges and research opportunities. Freshwater Biology, 55: 2215-2237.

Gibson, R.J., Haedrich, R.L., Wernerheim, C.M. (2005) Loss of fish habitat as a consequence of inappropriately constructed stream crossings. Fisheries, 30: 10-17.

Glova, G.J. (2003) A test for interaction between brown trout (*Salmo trutta*) and inanga (*Galaxias maculatus*) in an artificial stream. Ecology of Freshwater Fish, 12: 247-253.

Goodman, J.M. et al., (2014) Conservation status of New Zealand freshwater fish. Wellington, New Zealand, Department of Conservation.

Graham, E., Quinn, J. (2015) Is beta better? beta-diversity as an alternative indicator of restoration success. Oral presentation to the New Zealand Freshwater Sciences Society, Wellington, New Zealand.

Gray, D.P., Harding, J.S. (2007) Braided river ecology: A literature review of physical habitats and aquatic invertebrate communities. Science for Conservation: 1–50.

Gray, D. P., Harding, J. S. (2012). Acid Mine Drainage Index (AMDI): a benthic invertebrate biotic index for assessing coal mining impacts in New Zealand streams. New Zealand Journal of Marine and Freshwater Research, 46(3): 335-352.

Graynoth, E., Bonnett, M.L., Jellyman, D.J. (2012) Estimation of native fish density in lowland streams by repeated electric fishing during the day and following night. New Zealand Journal of Marine and Freshwater Research, 46: 243-261.

Greenwood, M.J., Booker, D.J. (2016) Influence of hydrological regime and land cover on traits and potential export capacity of adult aquatic insects from river channels. Oecologia, 180: 551–566. Springer Berlin Heidelberg.

Greenwood, M. J., Booker, D. J., Smith, B. J., Winterbourn, M. J. (2016). A hydrologically sensitive invertebrate community index for New Zealand rivers. Ecological indicators, 61: 1000-1010.

Greer, M.J.C., Closs, G.P., Crow, S.K., Hicks, A.S. (2012) Complete versus partial macrophyte removal: the impacts of two drain management strategies on freshwater fish in lowland New Zealand streams. Ecology of Freshwater Fish, 21: 510-520.

Greer, M.J.C., Crow, S.K., Hicks, A.S., Closs, G.P. (2015) The effects of suspended sediment on brown trout (*Salmo trutta*) feeding and respiration after macrophyte control. New Zealand Journal of Marine and Freshwater Research, 49: 278-285.

Gregory, R.D., Strien, A. van. (2010) Wild Bird Indicators: Using Composite Population Trends of Birds as Measures of Environmental Health. Ornithological Science, 9: 3–22

Gregory, R.D., van Strien, A., Vorisek, P., Gmelig Meyling, A.W., Noble, D.G., Foppen, R.P.B., Gibbons, D.W. (2005) Developing indicators for European birds. Philosophical Transactions of the Royal Society B: Biological Sciences, 360: 269–288.

Haase, P.S.U., Pauls, K., Schindehütte, Sundermann, A. (2010) First audit of macroinvertebrate samples from an EU Water Framework Directive monitoring program: human error greatly lowers precision of assessment results. Journal of the North American Benthological Society, 29(4): 1279-1291.

Hanchet, S.M. (1990) Effect of landuse on the distribution and abundance of native fish in tributaries of the Waikato River in the Hakarimata Range, North Island, New Zealand. New Zealand Journal of Marine and Freshwater Research, 24: 159-171.

Harmsworth, G.R., Young, R.G., Walker, D., Clapcott, J.E., James, T. (2011) Linkages between cultural and scientific indicators of river and stream health. New Zealand Journal of Marine and Freshwater Research, 45: 3, 423-436.

Harper, M. A., Cassie-Cooper, V., Chang, F.H., Nelson, W.A., Broady, W.A. (2012). Phylum Ochrophyta. Brown and golden-brown algae, diatoms, silicoflagellates, and kin. In: Gordon, D. P. (ed.), The New Zealand inventory of biodiversity, Vol. 3, Kingdoms Bacteria, Protozoa, Chromista, Plantae and Fungi. Canterbury Univ. Press, Christchurch, NZ.

Hawkins, C.P., Norris, R.H., Hogue, J.N., Feminella, J.W. (2000) Development and evaluation of predictive models for measuring the biological integrity of streams. Ecological Applications, 10: 1456–1477.

He, N. (2010) Temperature elevation drives biomass and community shifts in periphyton. M.Sc. thesis. Department of Zoology. Michigan State University: 40.

Hickey, C. (2014) Derivation of indicative ammoniacal nitrogen guidelines for the National Objectives Framework. Memorandum to Ministry for the Environment. https://www.mfe.govt.nz/sites/default/files/media/Fresh%20water/derivation-of-indicative-ammoniacal.pdf

Hicks, B.J., McCaughan, H.M.C. (1997) Landuse, associated eel production and abundance of fish and crayfish in streams in Waikato. New Zealand. New Zealand Journal of Marine and Freshwater Research, 31: 635-650.

Hoare, J.M., Monks, A., O'Donnell, C.F.J. (2013) Do population indicators work? Investigating correlated responses of bird populations in relation to predator management. Ecological Indicators, 25: 23–34. Elsevier Ltd. Available from http://linkinghub.elsevier.com/retrieve/pii/S1470160X1200324X (accessed October 18, 2012).

Howard-Williams, C., Pickmere, S. (2010) Thirty years of stream protection: long-term nutrient and vegetation changes in a retired pasture stream. Science for Conservation, 300: 1–30.

Howard-Williams, C., Clayton, J.S., Coffey, B.T., Johnstone, I. (1987) Macrophyte invasions. In: (Viner, A.B.) Inland Waters of New Zealand. DSIR Science Information Publishing Centre Wellington, 307-332.

Hughes, R.M., Kaufman, M, Herlihy, A.T., Kincaid, T.M., Reynolds, L., Larsen, D.P. (1998) A process for developing and evaluating indices of fish assemblage integrity. Canadian Journal of Fisheries and Aquatic Science, 55: 1618-1631.

Hughey, K.F.D. (1998) Nesting home range sizes of wrybill (*Anarhynchus frontalis*) and banded dotterel (*Charadrius bicinctus*) in relation to braided riverbed characteristics. Notornis, 45: 103–111.

Hughey, K.F.D., Fraser, B.R., Hudson, L.G. (1989) Aquatic invertebrates in two Canterbury rivers - related to bird feeding and water development impacts. Department of Conservation Science and Research Series, No. 2.

Jellyman, P.G., Harding, J.S. (2012) The role of dams in altering the freshwater fish communities in New Zealand. New Zealand Journal of Marine and Freshwater Research, 46: 475-489.

Jowett, I.G., Biggs, B.J.F. (2006) Flow regime requirements and the biological effectiveness of habitat-based minimum flow assessments for six rivers. International Journal of River Basin Management, 4: 179–189.

Jowett, I.G., Richardson, J. (2003) Fish communities in New Zealand rivers and their relationship to environmental variables. New Zealand Journal of Marine and Freshwater Research, 37: 347-366.

Joy, M.K. (2013) Freshwater fish predictive modelling for bioassessment: a scoping study into fish bioassessment models as national indicators in New Zealand. Report for Wairesearch Limited: 23.

Joy, M.K., David, B.O., Lake, M. (2013) New Zealand Freshwater Fish Sampling Protocols Part 1: wadeable streams. In: ed. Palmerston North, Massey University: 58.

Joy, M.K., Death, R.G. (2004) Application of the Index of Biotic Integrity methodology to New Zealand freshwater fish communities. Environmental Management, 34: 415-428.

Joy, M.K., Death, R.G. (2013) Freshwater Biodiversity. In: Dymond et al., ed. Ecosystem services in New Zealand – conditions and trends. Manaaki Whenua Press, Lincoln, New Zealand.

Karr, J.R., Fausch, K.D., Angermeier, P.L., Yant, P.R., Schlosser, I.J. (1986) Assessing biological integrity in running waters: a method and its rationale. Illinois National History Survey, Champaign, Illinois: 28.

Keedwell, R.J., Brown, K.P. (2001) Relative abundance of mammalian predators in the upper Waitaki Basin, South Island, New Zealand. New Zealand Journal of Zoology, 28: 31–38.

Kelly, M. (2013) Data rich, information poor? Phytobenthos assessment and the Water Framework Directive. European Journal of Phycology, 48(4): 437-450.

Kelly, M., Juggins, S., Guthrie, R., Pritchard, S., Jamieson, J., Rippey, B., Hirst, H., Yalloop, M. (2008) Assessment of ecological status in UK rivers using diatoms. Freshwater Biology, 53: 403-422.

Kilroy, C. (2017) Managing nuisance periphyton in the Lower Waiau River. Monitoring and management 2016-17. For Meridian Energy. NIWA Client Report No: 2017248CH: 51.

Kilroy, C., Biggs, B.J.F., Blair, N., Lambert, P., Jarvie, B., Dey, K., Robinson, K., Smale, D. (2006) Ecological studies on *Didymosphenia geminata*. NIWA Client Report CHC2005-128. For Biosecurity New Zealand: 77.

Kilroy, C., Biggs, B.J.F., Death, D. (2008) A periphyton monitoring plan for the Manawatu-Whanganui region. NIWA Client Report CHC2008-03: 43.

Kilroy, C., Biggs, B.J.F., Vyverman, W. (2007) Rules for macroorganisms applied to microorganisms: patterns of endemism in benthic freshwater diatoms. Oikos, 116(4): 550-564.

Kilroy, C., Biggs, B.J.F., Vyverman, W., Broady, P. (2006) Benthic diatom communities in subalpine pools in New Zealand: relationships to environmental variables. Hydrobiologia, 561: 95-110.

Kilroy, C., Booker, D.J., Drummond, L., Wech, J.A., Snelder, T.H. (2013) Estimating periphyton standing crop in streams: a comparison of chlorophyll a sampling and visual assessments. New Zealand Journal of Marine and Freshwater Research, 47(2): 208-224.

Kilroy, C., Bothwell, M.L. (2012) *Didymosphenia geminata* growth rates and bloom formation in relation to ambient dissolved phosphorus concentration. Freshwater Biology, 57(4): 641-653.

Kilroy, C., Larned, S.T., Biggs, B.J.F. (2009) The non-indigenous diatom *Didymosphenia geminata* alters benthic communities in New Zealand rivers. Freshwater Biology, 54(9): 1990-2002.

Kilroy, C., Novis, P. (2018) Is *Didymosphenia geminata* an introduced species in New Zealand? Evidence from trends in water chemistry, and chloroplast DNA. Ecology and Evolution, 8(2): 904-919.

Kilroy, C., Suren, A.M., Wech, J.A., Lambert, P., Sorrell, B.K. (2017) Epiphytic diatoms as indicators of ecological condition in New Zealand's lowland wetlands. New Zealand Journal of Marine and Freshwater Research, 51(4): 505-527.

Kilroy, C., Wech, J. (2015) Spatial and temporal patterns in nutrient concentrations and periphyton in the Hurunui River. NIWA Client Report CHC2015-086. For Ngai Tahu Forest Estates Ltd: 96.

Kilroy, C., Wech, J., Chakraborty, M., Brown, L., Watson, B., Patterson, M., Patterson, M., Shell, T. (2016) Periphyton in the Manawatu - Whanganui region. State, trends and seasonality, 2009 - 2015. NIWA Client Report CHC2016-013. For Horizons Regional Council: 160.

Kilroy, C., Wech, J.A., Kelly, D., Clarke, G. (2017) Analysis of a three-year dataset of periphyton biomass and cover in Canterbury Rivers. For Environment Canterbury. NIWA Client Report No: 2017085CH: 112.

Klose, K., Cooper, S.D., Leydecker, A.D., Kreitler, J. (2012) Relationships among catchment land use and concentrations of nutrients, algae, and dissolved oxygen in a southern California river. Freshwater Science, 31(3): 908-927.

Kociolek, J.P., Stoermer, E.F. (1988) Taxonomy, ultrastructure and distribution of *Gomphoneis herculeana*, *G. eriense* and closely related species (Naviculales: Gomphonemataceae). Proceedings of the Academy of Natural Sciences of Philadelphia, 140 (2): 24 - 97.

Krammer, K. (2002) Cymbella. Vol 3. of Lange-Bertalot, H. (ed.) Diatoms of Europe. A.R.G. Gantner Verlag K.G.

Larned, S.T. (2010) A prospectus for periphyton: recent and future ecological research. Journal of the North American Benthological Society, 29(1): 182-206.

Larned, S.T., Snelder, T., Unwin, M.J., McBride, G.B. (2016) Water quality in New Zealand rivers: current state and trends. New Zealand Journal of Marine and Freshwater Research, 50(3): 389-417.

Larned, S.T., Snelder, T., Unwin, M.J., McBride, G.B. (2016) Water quality in New Zealand rivers: current state and trends. New Zealand Journal of Marine and Freshwater Research, DOI: 10.1080/00288330.2016.1150309.

Leathwick, J.R., Elith, J., Chadderton, W.L., Rowe, D., Hastie, T. (2008) Dispersal, disturbance and the contrasting biogeographies of New Zealand's diadromous fish species. Journal of Biogeography, 35: 1481-1497.

Leathwick, J.R., Snelder, T., Chadderton, W.L., Elith, J., Julian, K., Ferrier, S. (2011) Use of generalised dissimilarity modelling to improve the biological discrimination of river and stream classifications. Freshwater Biology, 56: 21-38.

Lee, W., McGlone, M., Wright, E. (2005) Biodiversity Inventory and Monitoring: a review of national and international systems and a proposed framework for future biodiversity monitoring by the Department of Conservation. Landcare Research Contract Report, LC0405/122: 218.

Liess, A., Hillebrand, H. (2004) Invited review: Direct and indirect effects in herbivore periphyton interactions. Archiv Fur Hydrobiologie, 159(4): 433-453.

Liess, A., Lange, K., Schulz, F., Piggott, J.J., Matthaei, C.D., Townsend, C.R. (2009) Light, nutrients and grazing interact to determine diatom species richness via changes to productivity, nutrient state and grazer activity. Journal of Ecology, 97(2): 326-336.

Ligon, F.K., Dietrich, W.E., Trush, W.J. (1995) Downstream ecological effects of dams. BioScience, 45: 183–192. American Institute of Biological Sciences, Oxford University Press.

Matheson, F., Hickey, C., Costley, K., Smith, J., Taumoepeau, A., Nagels, J., Croker, G., Safi, K. (2009) Morrinsville wastewater treatment plant outfall effects on the Piako river. NIWA Report HAM2009-031, for Matamata Piako District Council: 44.

Matheson, F., Mouton, T., Haidekker, S., Wilding, T., Hamer, M., Catlin, A. (2017a) Riparian shading as a tool to manage nuisance instream plants: testing the concept in Hawkes Bay and Waikato streams and rivers. Proceedings of the 5th Biennial Symposium of the International Society for River Science, Hamilton 19-24 November.

Matheson, F., Quinn, J., Haidekker, S., Madarasz-Smith, A., Wilding, T., Clapcott, J., Harding, J., Wilcock, B., Young, R., Baker, M., Clode, G., Heath, N., Hicks, A., Rutherford, K. (2017b) Ecosystem health in disturbed lowland catchments: Karamū catchment, Hawkes Bay. NIWA Report for the Ministry of Science and Innovation Envirolink Fund: 49. http://www.envirolink.govt.nz/assets/Uploads/1733-HBRC223-Ecosystem-health-in-highly-modified-lowland-catchments-Karamu-catchment-Hawkes-Bay.pdf

Matheson, F., Quinn, J., Hickey, C. (2012) Review of the New Zealand instream plant and nutrient guidelines and development of an extended decision-making framework: Phases 1 and 2 final report. Prepared for the Ministry of Science & Innovation Envirolink Fund. NIWA Client Report HAM2012-081.

Matheson, F., Quinn, J., Unwin, M. (2015) Review of the New Zealand instream plant and nutrient guidelines and development of an extended decision-making framework: Phase 3. Prepared for the Ministry of Science & Innovation Envirolink Fund. NIWA Client Report HAM2015-064. http://www.envirolink.govt.nz/assets/Envirolink/R9-3-Instream-plant-and-nutrient-guidelines-Phase-3-report.pdf

Matheson, F., Wells, R. (2017) Periphyton and macrophytes in seven Hauraki-Coromandel rivers. An initial assessment of seasonal abundance, provisional attribute state and management options. NIWA Client Report 2017243HN for Waikato Regional Council: 70.

Mazor, R.D., Rehn, A.C., Ode, P.R., Engeln, M., Schiff, K.C., Stein, E.D., Gillett, D.J., Herbst, D.B., Hawkins, C.P. (2016) Bioassessment in complex environments: designing an index for consistent meaning in different settings. Freshwater Science, 35: 249-271.

McDowall, R.M. (1990) New Zealand freshwater fishes: a natural history and guide. Auckland, Heinemann Reed: 553.

McDowall, R.M. (1993) Implications of diadromy for the structuring and modelling of riverine fish communities in New Zealand. New Zealand Journal of Marine and Freshwater Research, 27: 453-462.

McDowall, R.M. (2006) Crying wolf, crying foul, or crying shame: alien salmonids and a biodiversity crisis in the southern cool-temperate galaxiid fishes? Reviews in Fish Biology and Fisheries, 16: 233-422.

McDowall, R.M. (2009) 'Making the best of two worlds': diadromy in the ecology and evolution of aquatic organisms. In: Haro AJ, Smith KL, Rulifson RA, Moffitt CM, Klauda RJ, Dadswell R, Cunjak A, Cooper JE, Beal KL, Avery TS ed. Challenges for diadromous fishes in a changing environment. American Fisheries Society, Symposium 69, Bethesda, Maryland: 1-22.

McDowall, R.M. (2010a) New Zealand freshwater fishes: an historical and ecological biogeography. London, Springer.

McDowall, R.M. (2010b) Why be amphidromous: expatrial dispersal and the place of source and sink population dynamics? Reviews in Fish Biology and Fisheries, 20: 87-100.

McIntosh, A.R., McHugh, P.A., Dunn, N.R., Goodman, J.M., Howard, S.W., Jellyman, P.G., O'Brien, L.K., Nystrom, P., Woodford, D.J. (2010) The impact of trout on galaxiid fishes in New Zealand. New Zealand Journal of Ecology, 34: 195-206.

McIntosh, A.R., Townsend, C.R., Crowl, T.A. (1992) Competition for space between introduced brown trout (*Salmo trutta* L.) and a native galaxiid (*Galaxias vulgaris* Stokell) in a New-Zealand stream. Journal of Fish Biology, 41: 63-81.

McKergow, L.A., Matheson, F.E., Quinn, J.M. (2016). Riparian management: a restoration tool for New Zealand streams. Ecological Management and Restoration, 17: 218-227.

Melles, S.J. (2005) Urban bird diversity as an indicator of human social diversity and economic inequality in Vancouver, British Columbia. Urban Habitats, 3:25–48.

Metcalfe, J.L. (1989) Biological water quality assessment of running waters based on macroinvertebrate communities: history and present status in Europe. Environmental Pollution, 60: 101-139.

Ministry for the Environment (1997) The state of New Zealand's environment 1997: 655.

Ministry for the Environment (MfE) (2015) A Draft Guide to Attributes in Appendix 2 of the National Policy Statement for Freshwater Management 2014. Wellington: Ministry for the Environment. http://www.mfe.govt.nz/publications/fresh-water/draft-guide-attributes-appendix-2-national-policy-statement-freshwater

Ministry for the Environment (MfE) (2016) Minutes of the Dissolved Oxygen/Temperature Technical Workshop. 11 March 2016.

Ministry for the Environment (2017) Draft Guide to Attributes in Appendix 2 of the National Policy Statement for Freshwater Management 2014 (as amended 2017). Wellington: 43.

Ministry for the Environment (2018). A draft technical guide to the Periphyton Attribute Note. Ministry for the Environment, Wellington: 80 (draft).

Ministry for the Environment and Statistics NZ (2015) New Zealand's Environmental Reporting Series: Environment Aotearoa 2015. Available from www.mfe.govt.nz and www.stats.govt.nz. http://www.mfe.govt.nz/sites/default/files/media/Environmental%20reporting/Environment-Aotearoa-2015.pdf

Ministry for the Environment and Statistics NZ (2016) Topics for Environmental Reporting, ME1246: 11.

Ministry for the Environment and Statistics NZ (2017) New Zealand's Environmental Reporting Series: Our fresh water 2017, ME1305: 98.

Ministry for the Environment and Statistics NZ (2018a) New Zealand's Environmental Reporting Series Environmental Indicators. Ministry for the Environment and Statistics NZ. http://archive.stats.govt.nz/browse_for_stats/environment/environmental-reporting-series/environmental-indicators/Home/About.aspx#abouttopics

Ministry for the Environment and Statistics NZ (2018b) New Zealand's Environmental Reporting Series Environmental Indicators. Ministry for the Environment and Statistics NZ. http://archive.stats.govt.nz/browse_for_stats/environment/environmental-reporting-series/environmental-indicators/Home/Fresh%20water.aspx

Mitchell, N. (1991) The derivation of climate surfaces for New Zealand and their application to the bioclimatic analysis of the distribution of Kauri (*Agathis australis*). Journal of the Royal Society of New Zealand, 22(1): 13-24.

Neale, M., Storey, R., Quinn, J. (2016) Stream Ecological Valuation: Application to Intermittent Streams. Prepared by Golder Associates (NZ) Limited for Auckland Council. Golder Report No. 1529657-001-R-Rev0. Auckland Council Technical Report 20XX.

Neale, M.W., Moffett, E.R. (2016) Re-engineering buried urban streams: daylighting results in rapid changes in stream invertebrate communities. Ecological Engineering, 87: 175-184.

Neale, M.W., Storey, R.G., Rowe, D.K. (2017) Stream Ecological Valuation: revisions to the method for assessing the ecological functions of New Zealand streams. Australasian Journal of Environmental Management, 24(4): 392-405. 10.1080/14486563.2017.1369463

Neale, M.W., Storey, R.G., Rowe, D.K., Collier, K.J., Hatton, C., Joy, M.K., Parkyn, S.M., Maxted, J.R., Moore, S., Phillips, N.R., Quinn, J.M. (2011) Stream Ecological Valuation (SEV): a user guide. GD2011/001: 71.

New Zealand Government (2017) National Policy Statement for Freshwater Management 2014 (Amended 2017). http://www.mfe.govt.nz/publications/fresh-water/national-policy-statement-freshwater-management-2014-amended-2017.

Niyogi, D.K., Koren, M., Arbuckle, C.J., Townsend, C.R. (2007) Longitudinal changes in biota along four New Zealand streams: declines and improvements in stream health related to land use. New Zealand Journal of Marine and Freshwater Research, 41(1): 63-75.

Novis, P. (2003) A taxonomic survey of Oedogonium (Oedogoniales, Chlorophyta) in the South Island and Chatham Islands, New Zealand. New Zealand Journal of Botany, 41: 335-358.

Novis, P. (2004) New records of Spirogyra and Zygnema (Charophyceae, Chlorophyta) in New Zealand. New Zealand Journal of Botany, 42: 139-152.

O'Brien, J.M., Lessard, J.L., Plew, D., Graham, S.E., McIntosh, A.R. (2014) Aquatic macrophytes alter metabolism and nutrient cycling in lowland streams. Ecosystems, 17: 405–417.

O'Donnell, C.F.J., Clapperton, B.K., Monks, J.M. (2015) Impacts of introduced mammalian predators on indigenous birds of freshwater wetlands in New Zealand. New Zealand Journal of Ecology, 39: 19–33.

O'Donnell CFJ, Sanders MD, Woolmore C, Maloney RF 2016. Management and research priorities for conserving biodiversity on New Zealand's braided rivers. Wellington, New Zealand, 46.

Parkyn, S. (2008) Modification of the SEV method for Hawke's Bay streams. Letter to Hawke's Bay Regional Council: 5.

Parsons, M., Norris, R.H. (1996) The effect of habitat-specific sampling on biological assessment of water quality using a predictive model. Freshwater Biology, 36: 419–434. DOI: 10.1046/j.1365-2427.1996.00088.x.

Pickerell, G.A. (2015) Braided-river islands as refuges from introduced mammalian predators: characteristics influencing predator presence, and consequences of reduced flow. University of Otago.

Plafkin, J.L., Barbour, M.T., Porter, K.D., Gross, S.K., Hughes, R.M. (1989) Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish. US EPA Report EPA/440/4-89/001.

Poikane, S., Kelly, M., Cantonati, M. (2016) Benthic algal assessment of ecological status in European lakes and rivers: Challenges and opportunities. Science of the Total Environment, 568: 603-613.

Potapova, M., Charles, D. (2003) Distribution of benthic diatoms in US rivers in relation to conductivity and ionic composition. Freshwater Biology, 48: 1311-1328.

Power, M.E. (1990) Resource enhancement by indirect effects of grazers: armored catfish, algae and sediment. Ecology, 71(3): 897-904.

Quinn, J.M., Cooper, A.B., Davies-Colley, R.J., Rutherford, C., Williamson, R.B. (1997a) Land use effects on habitat, water quality, periphyton, and benthic invertebrates in Waikato, New Zealand, hill-country streams. New Zealand Journal of Marine and Freshwater Research, 31(5): 579-597.

Quinn, J.M., Cooper, A.B., Stroud, M.J., Burrell G.P. (1997b) Shade effects on stream periphyton and invertebrates: an experiment in streamside channels. New Zealand Journal of Marine and Freshwater Research, 31: 665-683.

Quinn, J.M., Croker, G.F., Smith, B.J., Bellingham, M.A. (2009) Integrated catchment management effects on runoff, habitat, instream vegetation and macroinvertebrates in Waikato, New Zealand, hill-country streams. New Zealand Journal of Marine and Freshwater Research, 43(3): 775-802.

Reeves, P., Collier, K., Suren, A. (2004) Aquatic and riparian vegetation of rivers and streams. Chapter 14 In: Freshwaters of New Zealand (Eds. P. Mosley and J. Harding), New Zealand Limnological Society and New Zealand Hydrological Society, Caxton Press Christchurch, 14.1-14.16.

Reid, D.J., Quinn, J.M., Wright-Stow, A.E. (2010) Responses of stream macroinvertebrate communities to progressive forest harvesting: Influences of harvest intensity, stream size and riparian buffers. Forest Ecology and Management, 260 1804-1815.

Riis, T., Biggs, B.J.F. (2003). Hydrological and hydraulic control of macrophyte establishment and performance in streams. Limnology and Oceanography, 48: 1488-1497.

Riis, T., Biggs, B.J.F., Flanagan, M. (2003) Seasonal changes in macrophyte biomass in South Island lowland streams, New Zealand. New Zealand Journal of Marine and Freshwater Research, 37: 381-388

Robertson, C.J.R., Hyvönen, P., Fraser, M.J., Pickard, C.R. (2007) Atlas of bird distribution in New Zealand 1999-2004. Ornithological Society of New Zealand, Inc., Wellington.

Robertson, H., Dowding, J., Elliott, G., Hitchmough, R., Miskelly, C., O'Donnell, C., Powlesland, R., Sagar, P., Scofield, R., Taylor, G. (2013) Conservation Status of New Zealand Birds, 2012. Page New Zealand Threat Classification Series 4.

Robertson, H.A. et al., (2017) Conservation status of New Zealand birds, 2016. New Zealand Threat Classification Series, Wellington: 26.

Rott, E., Schneider, S.C. (2014) A comparison of ecological optima of soft-bodied benthic algae in Norwegian and Austrian rivers and consequences for river monitoring in Europe. Science of the Total Environment, 475: 180-186.

Rowe, D., Collier, K., Hatton, C., Joy, M., Maxted, J., Moore, S., Neale, M.W., Parkyn, S., Phillips, N., Quinn, J. (2008) Stream ecological valuation (SEV): a method for scoring the ecological performance of Auckland streams and for quantifying environmental compensation. TP302 second edition. NIWA, Hamilton: 85.

Rowe, D., Quinn, J., Parkyn, S., Collier, K., Hatton, C., Joy, M., Maxted, J., Moore, S. (2006) Stream ecological valuation (SEV): a method for scoring the ecological performance of Auckland streams and quantifying mitigation. NIWA, Hamilton: 85.

Rowe, D.K., Parkyn, S., Quinn, J., Collier, K., Hatton, C., Joy, M.K., Maxted, J., Moore, S. (2009) A rapid method to score stream reaches based on the overall performance of their main ecological functions. Environmental management, 43(6): 1287-1300.

Rowe, D.L. (2004) Exotic fish: a growing problem in New Zealand fresh waters. Water & Atmosphere, 12: Available via: https://www.niwa.co.nz/publications/wa/vol12-no3-september-2004/exotic-fish-a-growing-problem-in-new-zealand-fresh-waters.

Rutherford, J.C., Scarsbrook, M.R., Broekhuizen, N. (2000) Grazer control of stream algae: Modeling temperature and flood effects. Journal of Environmental Engineering ASCE, 126(4): 331-339.

Sabbe, K., Vanhoutte, K., Lowe, R., Bergey, E., Biggs, B., Francoeur, S., Hodgson, D., Vyverman, W. (2001) Six new Actinella (Bacillariophyta) species from Papua New Guinea, Australia and New Zealand: further evidence for widespread diatom endemism in the Australasian region. European Journal of Phycology, 36: 321-340.

Sand-Jensen, K., Jeppesen, E., Nielsen, K., Van der Bijl, Hjermind, L. Wiggers Nielsen, L., Moth Iversen, T. (1989) Growth of macrophytes and ecosystem consequences in a lowland Danish stream. Freshwater Biology, 22: 15–32.

Schneider, S.C., Lindstrøm, E.-A. (2011) The periphyton index of trophic status PIT: a new eutrophication metric based on non-diatomaceous benthic algae in Nordic rivers. Hydrobiologia, 665: 143-155. DOI 10.1007/s10750-011-0614-7

Schowe, K.A., Harding, J.S., Broady, P.A. (2013) Diatom community response to an acid mine drainage gradient. Hydrobiologia, 705(1): 147-158.

Snelder, T., Biggs, B.J.F., Kilroy, C., Booker, D. (2013) National Objective Framework for periphyton. NIWA Client Report CHC2013_122. For Ministry for the Environment: 39.

Snelder, T.H., Biggs, B.J.F. (2002) Multiscale river environmental classification for water resources management. Journal of the American Water Resources Association, 38: 1225-1239.

Stark, J.D., Boothroyd, I.K.G, Harding, J. S., Maxted, J.R., Scarsbrook, M.R. (2001) Protocols for sampling macroinvertebrates in wadeable streams. New Zealand Macroinvertebrate Working Group Report No. 1. Prepared for the Ministry for the Environment. Sustainable Management Fund Project, No. 5103: 57.

Stark, J.D., Maxted, J.R. (2007) A User Guide for the Macroinvertebrate Community Index, Cawthron Report, No. 1166. Ministry for the Environment, New Zealand.

Storey, R. (2009) Applicability of the SEV method for streams in the Wellington region. Letter to Greater Wellington Regional Council: 3.

Storey, R. (2011) Suitability of the Stream Ecological valuation (SEV) method for assessing stream values in Southland. NIWA Client Report HAM2011-025: 17.

Storey, R. (2013) Adapting the Stream Ecological Valuation for use in Horizons region. Letter to Horizons Regional Council: 3.

Storey, R.G., Neale, M.W., Rowe, D.K., Collier, K.J., Hatton, C., Joy, M., Maxted, J., Moore, S., Parkyn, S., Phillips, N., Quinn, J. (2011) Stream Ecological Valuation (SEV): a method for assessing the ecological functions of Auckland streams. Auckland Council Technical Report, 2011/009: 66.

Sturt, M.M., Jansen, M.A.K., Harrison, S.S.C. (2011) Invertebrate grazing and riparian shade as controllers of nuisance algae in a eutrophic river. Freshwater Biology, 56(12): 2580-2593.

Suren, A., Biggs, B., Duncan, M., Bergey, E., Lambert, P. (2003a) Benthic community dynamics during summer low -flows in two rivers of contrasting enrichment 2. Invertebrates. New Zealand Journal of Marine and Freshwater Research, 37: 71-83.

Suren, A., Biggs, B., Kilroy, C., Bergey, E. (2003b) Benthic community dynamics during summer low-flows in two rivers of contrasting enrichment 1. Periphyton. New Zealand Journal of Marine and Freshwater Research, (37): 53-70.

Taylor, B., McIntosh, A., Peckarsky, B. (2002) Reach-scale manipulations show invertebrate grazers depress algal resources in streams. Limnology and Oceanography, 47: 893-899.

Taylor, R., Smith, I. (1997) The stat of New Zealand's environment 1997. Wellington, Ministry for the Environment: 176.

Thorson, J.T., Scheuerell, M.D., Shelton, A.O., See, K.E., Skaug, H.J., Kristensen, K. (2015) Spatial factor analysis: a new tool for estimating joint species distributions and correlations in species range. Methods in Ecology and Evolution, 6: 627-637.

Timperley, M.H. (1983) Phosphorus in spring water of the Taupo volcanic zone, North Island, New Zealand. Chemical Geology, 38: 287-306.

Tockner, K., Stanford, J.A. (2002) Riverine flood plains: present state and future trends. Environmental Conservation, 29: 308–330.

Townsend, C.R., Crowl, T.A. (1991) Fragmented population structure in a native New Zealand fish: an effect of introduced brown trout? Oikos, 61: 347-354.

Van Dam, H., Mertens, A., Sinkeldam, J. (1994) A coded checklist and ecological indicator values of freshwater diatoms from the Netherlands. Netherlands Journal of Aquatic Ecology, 28(1): 117-133.

Vörösmarty, C.J., Green, P., Salisbury, J., Lammers, R.B. (2000) Global water resources: vulnerability from climate change and population growth. Science, 289: 284–288. American Association for the Advancement of Science.

Wagenhoff, A., Clapcott, J.E., Lau, K.E.M., Lewis, G.D., Young, R.G. (2017) Identifying congruence in stream assemblage thresholds in response to nutrient and sediment gradients for limit setting. Ecological Applications, 27(2): 469-48

Wagenhoff, A., Lange, K., Townsend, C.R., Matthaei, C.D. (2013) Patterns of benthic algae and cyanobacteria along twin-stressor gradients of nutrients and fine sediment: a stream mesocosm experiment. Freshwater Biology, 58(9): 1849-1863.

Wagenhoff, A., Townsend, C.R., Phillips, N., Matthaei, C.D. (2011) Subsidy-stress and multiple-stressor effects along gradients of deposited fine sediment and dissolved nutrients in a regional set of streams and rivers. Freshwater Biology, 56(9): 1916-1936.

Walker, S., Monks, A. (2004) New Zealand’s native land birds: status and change on the mainland from estimates of occupancy for 1969–1979 and 1999–2004. Landcare Research report prepared for the Parlimentary Commissioner for the Environment. Dunedin: 68.

Watene-Rawiri, E., Boubée, J.A., Williams, E.K. (in press) Waikato Tainui Tuna Restoration Plan. Waikato Tainui College for Research & Development Report

Wilcock, R.J., Champion, P.D., Nagels, J.W., Croker, G.F. (1999) The influence of aquatic macrophytes on the hydraulic and physico-chemical properties of a New Zealand lowland stream. Hydrobiologia, 416: 203–214.

Wilcock, R.J., McBride, G.B., Nagels, J.W., Northcott, G.L. (1995) Water quality in a polluted lowland stream with chronically depressed dissolved oxygen: causes and effects. New Zealand Journal of Marine and Freshwater Research, 29: 277-289.

Wilcock, R.J., Nagels, J.W., McBride, G.B., Collier, K.J., Wilson, B.T., Huser, B.A. (1998) Characterisation of lowland streams using a single-station diurnal curve analysis model with continuous monitoring data for dissolved oxygen and temperature. New Zealand Journal of Marine and Freshwater Research, 32: 67-79.

Winterbourn, M.J., Gregson, K.L.D., Dolphin, C.H. (2006) Guide to the Aquatic Insects of New Zealand Bulletin of the Entomological Society of New Zealand, Number 14.

Wood, S.A., Depree, C., Brown, L., McAllister, T., Hawes, I. (2015) Entrapped sediments as a source of phosphorus in epilithic cyanobacterial proliferations in low nutrient rivers. PLoS One, 10(10), e0141063.

Wright, J.F., Furse, M.T., Armitage, P.D. (1993) 3 RIVPACS: a technique for evaluating the biological quality of rivers in the U.K. European Water Pollution Control: 15–25.

Young, R., Clapcott, J., Simon, K. (2016) Ecosystem functions and stream health. Chapter 8. In: P. Jellyman, T. Davie, C. Pearson & J. Harding (Eds). Advances in New Zealand freshwater science. New Zealand Freshwater Sciences Society and New Zealand Hydrological Society: 696.

Young, R.G., Matthaei, C.D., Townsend, C.R. (2008) Organic matter breakdown and ecosystem metabolism: functional indicators for assessing river ecosystem health. Journal of the North American Benthological Society, 27(3): 605-625.

Appendix A State, pressure and impact indicators used in the Freshwater Domain report (2016)

Table A-1: State, pressure and impact indicators used in the 2016 Freshwater Domain report. Abbreviations: NI = national indicator; CS = case study; SI = supporting information. See below for definitions of these.

Topic (2016 topic list)	Indicator	Indicator type
State		
Freshwater ecosystems and habitats	Wetland extent	CS
Freshwater species, taonga species, and genetic diversity	Conservation status of native freshwater fish and invertebrates	CS
	Lake submerged plant index	CS
	Trends in freshwater fish	CS
Freshwater quality, quantity, and flows	Geographic pattern of natural river flows	NI
	Groundwater quality	CS
	Groundwater pesticides	CS
	Groundwater physical stocks	CS
	Lake water quality	CS
	Location and extent of New Zealand's aquifers	SI
	River water quality: clarity	CS
	River water quality: Escherichia coli	CS
	River water quality: nitrogen	CS
	River water quality: macroinvertebrate community index	CS
	River water quality: phosphorus	CS
	Streambed sedimentation	CS
	Urban stream water quality	CS
	Water physical stocks: precipitation and evapotranspiration	NI
Pressure		
Pests, diseases and exotic species	Freshwater pests	SI

Topic (2016 topic list)	Indicator	Indicator type
Resource use and management and other human activities	Consented freshwater takes	NI
	Selected barriers to freshwater fish in Hawke's Bay	SI
	Land cover	NI
	Land use	CS
	Livestock numbers	NI
Discharge and waste	Estimated highly erodible land in the North Island	NI
	Estimated long-term soil erosion	NI
	Geographic pattern of agricultural nitrate leaching	CS
	Trends in nitrogen leaching from agriculture	NI
Physical form of the land and freshwater environments	*	
Climate and natural processes	Annual rainfall	CS
	Annual maximum three-day rainfall	NI
Impact		
Impacts on biodiversity and ecosystem processes	Conservation status of native freshwater fish and invertebrates	CS
Impacts on public health	*	
Economic impacts	Value of water resources used for hydroelectric generation	NI
Mātauranga Māori, tikanga Māori, and kaitiakitanga	Cultural health index for freshwater bodies	CS
Mātauranga Māori, tikanga Māori, and kaitiakitanga	Kaitiakitanga of the Waikouaiti catchment	CS
Customary use and mahinga kai	Tau kōura: freshwater crayfish traditional fishing method	SI
Sites of significance, including wāhi taonga and wāhi tapū	*	
Impacts on culture and recreation	Participation in recreational fishing	SI

Statistical measures

Each indicator reports on an analysis of datasets (or statistic or set of statistics) about a specific topic. These statistics are assessed as either a national indicator, case study, or supporting information.

National indicator: the statistic is representative of the national situation and is highly accurate. A national indicator is directly relevant to a particular topic.

Case study: the statistic relates to areas that represent only part of the national situation, may not be as accurate as desired due to methodological reasons, or only provides partial information about a topic. A case study, at the least, is reasonably relevant to a particular topic.

Supporting information: the statistic provides only contextual information, or may only inform a topic indirectly. Supporting information are still reasonably accurate.