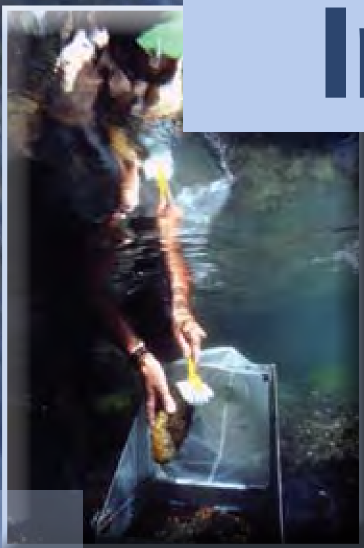
The Restoration Indicator Toolkit



Indicators for Monitoring the Ecological Success of Stream Restoration

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Book cover images

Front cover from top left:
Sampling macroinvertebrates (Chris Hickey)
Waitete Stream restoration site (Rob Davies-Colley)
Stream-bed particle size monitoring (Richard Storey)
Etherington family at Waitao Stream field day (Robyn Skelton)
Waitao Stream monitoring (John Quinn)
Back cover: Riparian vegetation at Avon River (Steph Parkyn).

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Part one:

Introduction

Purpose of the Indicator Toolkit

The purpose of this toolkit is to recommend and describe a range of indicators for monitoring improvement in stream restoration projects. We provide guidance on appropriate indicators depending on the goals of your restoration project and when to expect improvements.

Who is the Toolkit for?

The Toolkit has been developed primarily for the needs of regional councils with access to laboratories and technical equipment, but it should also be useful for community groups and resource users that are undertaking stream restoration without specialist equipment. It is based around the concept of identifying the important goals of the restoration and choosing appropriate indicators to measure the success of those goals. Some of the indicators require specialist equipment or technical training. However, there are several indicators for each type of goal, and when selecting from the Toolkit, a community group may simply avoid specialist indicators and choose others that match their goals and can be measured more easily. Alternatively, it may be possible for a community group to work with the regional council or research scientists in monitoring a restoration site.

Defining restoration success

Clear and measurable goals need to be established for your restoration project to design appropriate monitoring and evaluate whether the restoration has been successful. It is not the purpose of the Toolkit to dictate these goals, but the assumption is made that most restoration projects generally aim to return some or all of the following towards a more natural (pre-human) condition: biodiversity, physical habitat character, ecological processes, and water quality. Many projects do not begin with a clear statement of their goals and this hampers their ability to determine success (Hassett et al. 2007, Rumps et al. 2007).

GUIDING PRINCIPLES FOR THIS TOOLKIT

"Restoration" *is the actions taken to return stream ecosystems towards the natural condition. This can include actions to improve water quality, hydrology, physical habitat, connectivity, and/or key ecological processes to sustain native aquatic life. This definition of restoration may differ from stream management to enhance particular ecosystem services that support human-focused values (e.g., flood control and nutrient attenuation).*

"Restoration success" *can be measured in terms of the degree of movement towards a natural regime, typically defined by a comparable undisturbed (reference) site or by a guiding image of what the stream might have been like prior to human disturbance. Success doesn't necessarily mean achieving natural conditions (if catchment constraints make this impossible), but it does mean moving tangibly towards this goal. Success is best measured relative to a series of specific goals for your restoration project.*

Some common goals of restoration projects (typically management or human-focused goals) may conflict with the ecological goals and measures of success suggested in this document. Examples of goals that we have not designed indicators specifically for include:

- non-native fish or fisheries
- aquacultural practices that aim to maximise productivity of a food resource (when enhancement of one species is not the natural state of the stream and could impact other species)
- flood protection, infrastructure protection, land protection, or land drainage (when streams are managed to protect property)
- nutrient attenuation (when used to alleviate water quality issues downstream, e.g., growing watercress in channels to take up nutrients)
- hydro power generation (when unnatural flow regimes potentially override ecological benefits from other actions)
- aesthetics/recreation (when that differs from aesthetic and recreation values provided by the natural reference state).

We have followed the five criteria for judging restoration success put forward by Palmer et al. (2005) in the development of the Toolkit.

1. A dynamic ecological endpoint is identified beforehand and used to guide the restoration.
2. The ecological conditions of the river are measurably enhanced.
3. The river ecosystem is more self-sustaining than before restoration.
4. No lasting harm is done.
5. Both pre- and post- project assessment is completed.

To judge whether a stream has been measurably enhanced towards a predetermined dynamic endpoint depends upon measurements from the stream prior to impairment and some measure of reference conditions at a comparable undisturbed or minimally disturbed site.

In many developed nations, natural reference stream reaches no longer exist in geographic settings such as lowland areas (Woolsey et al. 2007). In this case a "guiding image" can be developed (based on historical information, undisturbed sites elsewhere, collective knowledge or theoretical models) which describes the restoration potential of a river under given circumstances and constraints (Palmer et al. 2005, Woolsey et al. 2007). Once the guiding image has been formulated, clear restoration goals can be defined and restoration success can be measured. The guiding image can be built using historical photographs or artwork, oral histories describing what the stream was like (e.g., was it silty or stony?), or visits to other streams in the area that you would like your stream to look like.

New Zealand has experienced human occupation relatively recently compared to many other countries, and in some regions, sites with minimal human intervention can still be found, although these are mostly in upland settings. Later in this document we give guidance on locating suitable reference sites or developing a guiding image that produces a measurable indicator of restoration success.

GENERAL RESTORATION APPROACHES

Stream restoration is a key activity promoted by regional councils and stream care groups throughout New Zealand. Guidelines for riparian management are available (e.g., Collier et al. 1995) and many regional councils provide guidelines for restoration tailored to their region, but the actual work is usually undertaken by landowners, members of the public, or resource users that are required to provide mitigation for their activities. Stream restoration can be a requirement of resource consents where streams may be damaged, piped or redirected.

Stock damage of farm stream banks.
Photo: Steph Parkyn

Typical examples of stream restoration actions include riparian planting, fencing of farm streams to exclude grazing stock, or re-engineering dams and road culverts to allow passage for migratory fish. In-channel activities, such as reinstatement of meanders or riffle and pool habitats, or reconnection of rivers with their floodplains, are far less common. In a workshop held at the NZ Freshwater Sciences Society Conference in 2006 (Appendix A), we identified the most common forms of restoration employed by councils, community groups, and regulatory agencies in New Zealand as: stock exclusion, riparian planting, bank stabilisation works, and fish passage enhancement.

A multi-tier riparian buffer with fencing and long grasses at the paddock edge and tall trees to shade the stream.
Photo: Thomas Wilding

The expectation of most stream restoration is that habitat rehabilitation will be sufficient to restore stream biodiversity and functioning. This expectation has been referred to as the Field of Dreams Hypothesis: "if we build it, they will come" (Palmer et al. 1997, Lake et al. 2007). However, there is *often insufficient (or no) testing of this hypothesis, in part because many restoration projects are not designed with scientific testing in mind (Lake et al. 2007). For example, there is often no sampling before restoration works are begun and no suitable reference site to monitor in conjunction with the restored site as a control. Brooks & Lake (2007) examined records for 2,247 restoration projects in Victoria, Australia, and found that riparian management projects were the most common, followed by bank stabilization and in-stream habitat improvement; but only 14% of the project records indicated that some form of monitoring was carried out. The length of stream over which restoration works are undertaken, the location of the restoration site in the catchment, and the presence of any constraints to colonisation (e.g., downstream barriers) all potentially influence the success of a particular restoration activity.*

Textured ropes installed in a stream culvert to aid fish passage.
Photo: Bruno David

Why monitor?

Simply put, we need to monitor so that we can learn from our successes and failures. Even projects that may initially appear to be failures can be turned into success stories by applying the knowledge gained from monitoring the project in an adaptive restoration approach (Palmer et al. 2007). Assessing the outcome of stream restoration projects is not only vital for adaptive management, but is also important for gaining public acceptance (Woolsey et al. 2007) and continued public funding. It is not necessary to monitor everything, but you should monitor something relevant to your goals.

MONITORING RESTORATION OUTCOMES

In the United States, billions of dollars are spent restoring streams and rivers, yet Palmer et al. (2005) report that there are no agreed upon standards for what constitutes ecologically beneficial stream and river restoration. According to a survey conducted by Bernhardt et al. (2007) of 317 stream restoration project managers across the United States, ecological degradation typically motivated restoration projects, but post-project appearance and positive public opinion were the most commonly used metrics of success. Less than half of all projects set measurable objectives for their projects, even though nearly two-thirds of all interviewees felt that their projects had been "completely successful".

In another survey of Pacific Northwest restoration practitioners, Rumps et al. (2007) found more than two-thirds (70%) of all respondents reported their projects were "successful", but 43% either had no success criteria or were unaware of any criteria for their project. Interviews revealed that many restoration practitioners were frustrated by the lack of funding for, and emphasis on, project monitoring (Bernhardt et al. 2007).

How to use this document

This document is not intended as a methodology guide for use in the field, but rather to provide a basis from which a field manual could be developed, tailored specifically for the goals of your restoration project.

Key components of this document are the predicted trajectories of successful restoration for each indicator, following typical best management practice. These predicted trajectories will be refined as research, monitoring, and the age of restoration projects increase. The trajectories can be used as a basis to compare actual data against. While they provide guidance on timescales of success, it must be stressed that they are merely a starting point and real data will improve this knowledge over time.

This document provides guidance on:

- designing a restoration monitoring programme
- choosing indicators to match project goals
- using appropriate methods and timeframes for monitoring the indicators
- understanding expected trajectories of improvement and when to expect success.

A pasture stream fenced to exclude stock and planted with native vegetation.
Photo: John Quinn

Part two:

Developing the Indicators

The Restoration Indicator Toolkit:
Indicators for monitoring the ecological success of stream restoration

Appendix A describes the methods we used to prioritise and develop the list of indicators. Our mandate was to focus on indicators to measure ecosystem function, aquatic biodiversity, and water quality. Table 2.1 shows the finalised list of indicators, and in Part 5 we describe each indicator in full (methodology and timescales for success).

We used three main ecological categories to ensure that the indicators covered a range of ecological functions:

1. Habitat, including flow regime and geomorphology
2. Water quality and biogeochemical functioning
3. Biota

To help you match the indicators to your project goals, we identified a range of potential goals and the specific type of restoration activity that each indicator would be most relevant for (Table 2.1). Although our focus was not on developing indicators for recreation, cultural, aesthetic or fisheries goals, several of the indicators can be used to measure restoration success for those goals. Further information on selecting appropriate indicators for your restoration goals is provided in Part 4.

To help you choose indicators that match the level of monitoring that your resources allow, we ranked each indicator to determine its level of general applicability. In Table 2.1:

1 = most commonly applicable to a wide range of restoration projects.

2 = likely to be relevant to projects with very specific goals.

3 = most likely to be measured for research or to understand constraints to restoration (diagnostic of problems).

The choice of whether to include an indicator depends on the goals of the project, but these rankings may assist to narrow down the list of indicators.

Stream monitoring.
Photo: Rob Davies-Colley

Table 2.1: *The list of restoration indicators described in this toolkit with criteria to help choose appropriate indicators for your restoration project.*

Codes for goals are: NH = Natural Habitat, WQ = Water Quality, EF = Ecosystem Functioning, AB = Aquatic Biodiversity, TB = Terrestrial Biodiversity, DH = Downstream Health, R = Recreation, C = Cultural, A = Aesthetics, F = Fisheries.

Scale of recovery approximates the time taken for the restored site to reach reference condition: Short term = 0—30 years, Medium term = 0—100 years, Long term = 0—400 years.

Applicability rankings describe general relevance of the indicator: 1 = almost all projects, 2 = specific types, 3 = specialised/research.

Indicator	Goals	Level of applicability	Type of restoration activity/ land use/setting most relevant to	Scale of recovery	Suggested minimum timescale of monitoring
Habitat					
Water and channel width	NH	1	All	Short	**Annually**
Bank erosion and condition	NH	1	All	Short	**Annually**
Longitudinal profile variability	NH	3	Channel reconstruction	Long	**Annually if channel recon., else 5-yearly**
Mesohabitats	NH	2	Channel reconstruction	Long	**Annually first 5 y if channel recon., else 5-yearly**
Residual pool depth	NH, F	2	Channel reconstruction	Medium	**Annually**
Water clarity	NH, WQ, A, F, DH	1	All	Short	**Monthly—annually**
Stream-bed particle size	NH	1	All	Medium	**Annually**
Organic matter abundance	NH	1	Riparian management	Medium	**Annually**
Leaf litter retention	NH, EF	2	All	Medium -Long	**2-yearly**
Rubbish	NH, WQ, A, R	2	Urban/farming	Short	**Annually or 2-yearly**
Shade of water surface	NH, AB	1	Riparian management	Medium	**Annually or 2-yearly**

Indicator	Goals	Level of applicability	Type of restoration activity/land use/setting most relevant to	Scale of recovery	Suggested minimum timescale of monitoring
Habitat					
Riparian microclimate	NH, AB, TB	2	Riparian management	Short	**Loggers summer periods—annually**
Water quality and biogeochemical functioning					
Water temperature	NH, WQ, AB	1	All	Short	**Loggers summer periods—annually**
Dissolved oxygen	WQ, EF, AB, F	2	All	Short	**Loggers/spot measures—annually**
Ecosystem metabolism	EF	2	Waterways >20cm depth	Short	**Seasonally—annually**
Organic matter processing	EF	2	Riparian management	Short	**Seasonally—annually**
Nutrients	WQ, DH	2	Farming, urban, point source	Short—Medium	**Monthly for 1 year then repeat at 5-yearly interval**
Faecal indicators	WQ, R, DH	2	Farming, urban, point source	Short	**Monthly for 1 year then repeat at 5-yearly interval**
Toxicants	WQ, DH	3	Urban/mining/geothermal input	Short—Medium	**Monthly for 1 year then repeat at 5-yearly interval**
pH	WQ	3	Urban/mining, where high plant biomass	Variable	**Monthly for 1 year then repeat at 5-yearly interval**
Biota					
Periphyton	AB, NH, A, R	2	All	Short	**Monthly during growing season.** Annually first **5 years then at 5-yearly intervals**
In-stream macrophytes	AB, NH	2	All	Medium	Annually first **5 years then at 5-yearly intervals**

Indicator	Goals	Level of applicability	Type of restoration activity/land use/setting most relevant to	Scale of recovery	Suggested minimum timescale of monitoring
Biota					
Benthic macro-invertebrates	AB, WQ	1	All	Medium	Annually first **5 years then at 5-yearly intervals**
Stream mega-invertebrates	AB, C	2	All	Medium	**Annually in summer**
Fish	AB, C, F	2	All	Medium	**Annually in summer (Dec–end Mar)**
Terrestrial plant biodiversity and survival of plantings	TB, NH	2	Riparian management	Medium	Annually first **5 years then at 5-yearly intervals**

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Part three:

Designing a monitoring programme

The Restoration Indicator Toolkit:
Indicators for monitoring the ecological success of stream restoration

The key steps in designing your monitoring programme begin with identifying project goals and catchment constraints, understanding your restoration site, and having a clear image or reference site to aim for. Figure 3.1 outlines how the parts of this document help you to form your monitoring programme.

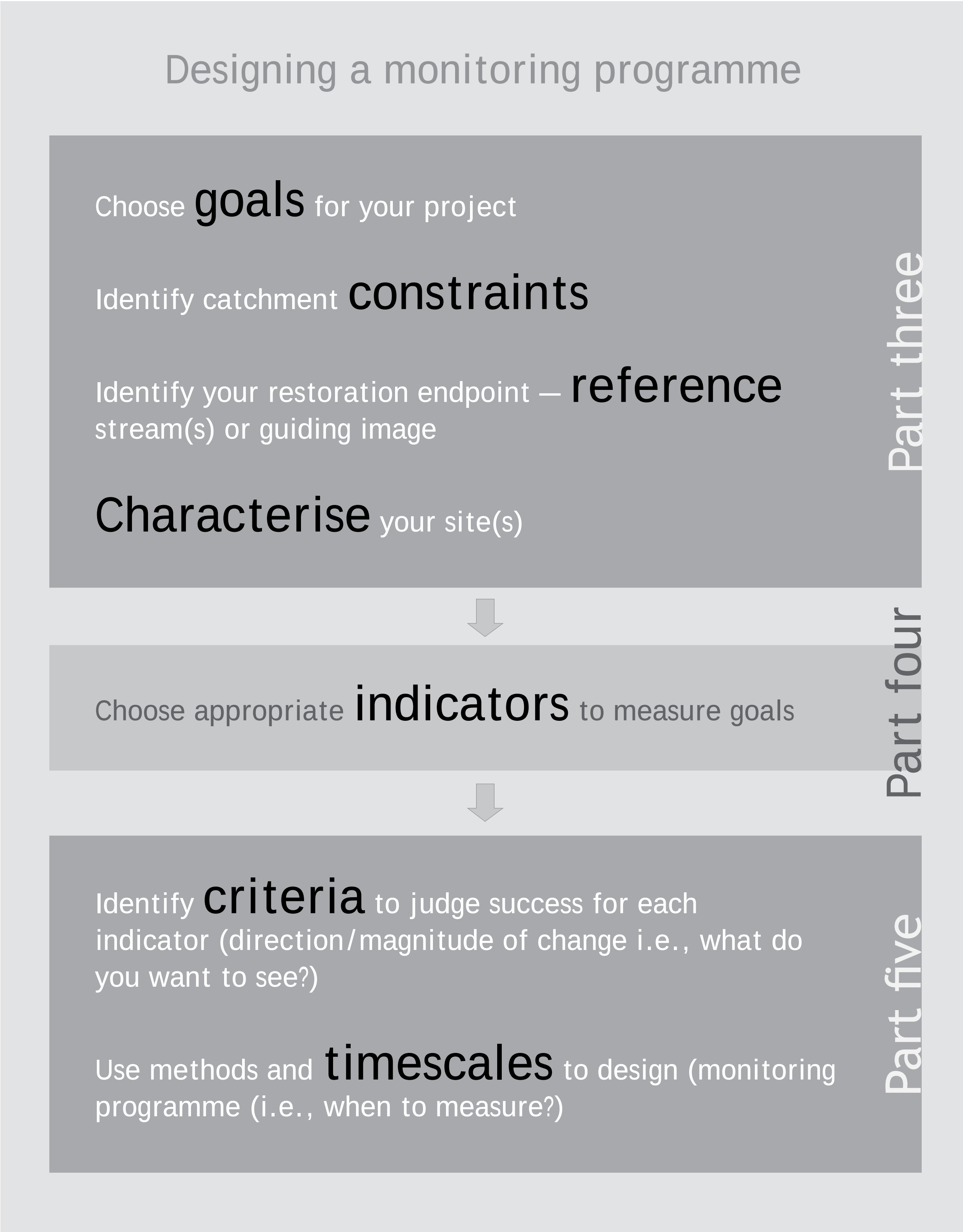

Figure 3.1: *Diagram of key steps for designing your monitoring programme and the parts of this document that can aid each step.*

Choose your project goals

It is essential to determine the primary goals of your restoration prior to beginning a monitoring programme. Ideally, these goals will have been decided before restoration activities begin at a site. You may need to keep in mind any catchment constraints (see below) that interact with goal setting.

The goals for your restoration may be diverse, and in some cases may even conflict (e.g., trout fisheries and aquatic biodiversity). In this document we provide guidance on indicators for measuring six main ecological goals. These are:

1. Natural Habitat (NH)
2. Aquatic Biodiversity (AB)
3. Ecosystem Functioning (EF)
4. Water Quality (WQ)
5. Downstream Health (DH)
6. Terrestrial Biodiversity (TB)

Additional goals that might be allied to these ecological restoration goals are:

- Cultural (C)
- Aesthetic (A)
- Fisheries (F)
- Recreation (R)

In Part 4 we help you choose indicators to match these goals. Try to identify the primary goal of restoration, as this will help you to prioritise what to monitor.

Identify constraints

There are a number of constraints that could affect restoration success and should be considered while setting goals for your restoration site. Some examples of constraints and the goals they affect are listed in Table 3.2. The condition of the wider catchment can override the rehabilitation of local habitat conditions, e.g., the alteration of natural flow regimes and high potential for chemical contamination common in urban catchments may mean that some biological objectives are slow (or impossible) to achieve under the current conditions. The hydrology of the stream could be a constraint to some goals, e.g., it can be difficult to reverse excess sedimentation in spring-fed streams that are not subject to floods. If the goal of your restoration is to restore fish communities, it will be important to establish whether there are downstream barriers to fish dispersal, namely free access to the sea, as many New Zealand fish species migrate upstream to find suitable adult habitat. Similarly, there may be dispersal barriers for many invertebrate species to return to the restored area, such as proximity of native forest in the catchment, which affect both biodiversity goals and integrated measurements of water quality (based on invertebrate community metrics).

The length of stream and type of restoration activity can also be a constraint. Generally, the longer the length of stream to be restored, the better the chance of achieving ecological goals. As a rule of thumb, restored stream lengths of <1 km may constrain restoration success.

Table 3.1: *Examples of constraints that can affect the achievability of project goals when riparian management of a stream reach is the only method of restoration.*

Constraint	Potentially unachievable goals	Achievable goals
Urban stream with high impervious area in catchment connected by stormwater	Water quality, aquatic biodiversity, natural habitat (e.g., flow regime)	Some ecosystem functions, terrestrial plant biodiversity, aesthetics, habitat for tolerant species
Pasture stream with extensive unrestored stream length upstream of restoration site and no native forest in headwaters	Aquatic biodiversity, water quality	Natural habitat, terrestrial plant biodiversity, aesthetics, ecosystem function
Pasture stream with a permanent barrier to migration of fish species downstream of restoration site but native forest headwaters	Aquatic biodiversity (fish)	Natural habitat, aquatic biodiversity (except fish), terrestrial plant biodiversity, bird diversity, water quality
Spring-fed stream with excess sedimentation	Natural habitat (substrate)	Terrestrial plant biodiversity, natural habitat (other than substrate), water quality
Stream receiving toxic leachates from a rubbish dump	Water quality, aquatic biodiversity	Terrestrial plant biodiversity, natural habitat

It is important to acknowledge these constraints, but not be put off by them. In many cases, complete restoration to pristine systems will be unachievable and the biotic communities that develop will potentially be regulated by new disturbance regimes, incomplete habitat requirements, barriers to effective dispersal, or introduced species. However, it will be possible to move degraded streams along a trajectory of improvement. Understanding constraints can inform goal selection and timescales of expected success, and aid adaptive management.

Identify your reference endpoint

In many cases, your choice of restoration site will be obvious; there may be only one stream for you to choose from and restoration will be occurring in one part of the stream. Where possible, monitor reference sites as a target for restoration that your site should be moving towards; You can also include one or more relevant control sites (unrestored, e.g., an unfenced pasture stream with no riparian plantings possibly upstream of your site or in a matched location nearby) in your survey design. A control site would give additional information about the degree to which your restored site has moved away from its degraded state. Monitoring at your restoration site, reference site, and/or control site should all be undertaken at the same time of year. Take baseline measures at all sites before restoration activities begin.

REFERENCE SITE OR GUIDING IMAGE?

A key component of being able to judge the ecological success of restoration is having an "endpoint" that the restoration is trying to reach. Universal ecological endpoints applied to all restoration sites are not possible because of regional differences in geology, climate, vegetation, land use history, and species distribution (Palmer et al. 2005). In natural systems, any endpoint can be expected to be dynamic within a range of conditions defined by commonly occurring environmental events, so the intention is to identify the "dynamic equilibrium" within which natural stream ecosystems function. Often the "endpoint" will be defined by a reference location, matched as closely as possible to the restoration site in terms of distance to sea, size of stream, substrate conditions, altitude, etc., or alternatively multiple sites that may define a general "reference condition" for your region. In the absence of a suitable reference site, we suggest developing a guiding image against which criteria of restoration success can be judged. This is a pragmatic approach to identifying restoration targets that move a stream towards the least degraded and most ecologically dynamic state practical given catchment constraints or the regional context.

Native forest reference stream.
Photo: Bruno David

Reference site selection

Reference sites should:

- be nearby restoration sites so that they experience similar climatic events at the same time
- have catchments with similar area (i.e., stream size), geology, soil types, and topography to restoration sites
- contain a range of habitats similar to those at the restoration sites
- not be downstream of restoration sites or other disturbances that could impact on the ecological integrity of the reference site.

A mixture of desktop and ground-truthing can be used to choose reference sites. You can use GIS-based stream classifications (e.g., River Environment Classification (REC) Snelder et al. 2004) available at **www.niwa.co.nz/our-science/freshwater/tools/rec)** to find appropriate reference sites with similar natural characteristics. For example, Collier et al. (2007a) used GIS and stream classes from the REC to identify sites with >85% of unmodified vegetation cover adjacent to the stream in the upstream catchment and then used land cover, amenity, and environmental impact databases to further classify streams with anthropogenic influences. Physically characterising your site using the descriptive variables described below will also help identify an appropriate reference stream in your region.

Developing a guiding image

As an alternative to a reference site, for example in lowland areas where undisturbed sites are rare, a guiding image can be developed to describe the dynamic, ecologically healthy waterway that could exist at a given site.

To develop a guiding image, use a combination of the following approaches.

- Collate historical information – aerial photographs, ground photography, oral histories, land and biological survey records.
- Visit relatively undisturbed or restored stream sites and take photos; choose sites as similar to your restoration site as possible, i.e., match lowland streams, geology, climate, etc. in the same way as you would choose a reference site.
- Use predictive or empirical models to assess what species or conditions should be at a site (e.g., Joy & Death 2004, Leathwick et al. 2009).
- Use recovery trajectories (like those supplied in Part 5 of this document) to develop an expectation of ecological endpoints.
- Use stream or riparian management classification systems (e.g., Brierley & Fryirs 2005, Quinn 2009) to help define expectations for particular stream types and predict the outcomes of restoration.

Characterise your sites

These descriptive variables can help you to characterise your restoration site, locate a suitable matched reference site (or develop your guiding image), and understand the landscape context.

Land use, restoration activities, and management history

To adequately describe a restoration site, it is important to gather as much information as possible about the current and past land use (e.g., upstream stocking densities, stream crossings, access for stock watering, condition of fencing, presence of rubbish dumps, forest harvesting) for both the stream reach and upstream and downstream of your project site. This will help you understand current and historical constraints to restoration potential. It can be done as a desktop exercise, but will most likely involve interviewing land owners or forest and farm managers. Record as much information as possible about the proposed or existing restoration activities, e.g., buffer width and length, channel reconstruction methods, planting plan, etc. Drawings of valley form, stream sinuosity, and mesohabitat types (runs, riffles, pools), and site photographs can all help to find a matching reference location and document baseline conditions.

Mapping barriers and connectivity

Structures such as dams, culverts, piping, fords, or high waterfalls can be barriers for fish that need to travel to and from the sea to complete their life cycle.

Identifying potential barriers can be a desktop exercise, such as noting that the stream that your restoration project is on flows through an urban area before reaching the sea, or it could be a practical exercise, where you trace the passage of your stream and note the size and type of culverts that could be barriers.

Example of potential barriers to fish passage: dam.
Photo: Richard Storey

The location of the restoration site within the landscape will influence potential source areas of recolonists that can readily get to the site once conditions become suitable. Proximity to the sea (for migratory fish and shrimps) or areas of remnant native bush will influence colonisation and should also be recorded. Fish and some invertebrates can only travel along streams, but freshwater insect species have aerial stages that allow them to travel overland. Proximity to source areas of recolonists will affect timescales of recovery.

Example of potential barriers to fish passage: perched culvert.
Photo: Richard Storey

Desktop GIS variables

Table 3.1 lists a range of segment scale parameters established from the River Environment Classification (REC; Snelder et al. 2004, Table 3.1). First you need to know the reach number (NZREACHID) for your site (a "reach" in the REC context is a segment of stream from where one tributary joins to the next). The CD supplied with the Stream Habitat Assessment Protocols (Harding et al. 2009) contains all REC reaches for New Zealand; these are the same reach numbers that are used in the Freshwater Environments of New Zealand (FWENZ, **www.ew.govt.nz/Environmental-information/REDI/1063385)** which contains a range of other underlying environmental variables tagged to NZREACH ID. The REC is available on the NIWA website: **www.niwa.co.nz/our-science/freshwater/tools/rec**. You will need access to GIS software, e.g., ArcView or ArcGIS (ESRI), to use the REC. If you do not have GIS, then topographic maps can be used to give an approximate elevation and distance along the stream to the sea or remnant bush.

REACH SLOPE

Approximations of reach slope can be obtained from the REC as described above and shown in Table 3.1. Channel slope can be measured in the field as the change in water surface elevation over the length of the reach using an inclinometer and two measurement poles. The water surface should be standardised at a point on both poles and the slope measured by sighting from the top of one pole to the other.

MEAN FLOW

The most accurate picture of the hydrological character of a stream is gained by collating flow variables from long-term data sets. Most often these data sets exist only for sites with permanent stage-height gauges. However, a gauging station close to the study site can be used to estimate flow variables by correlation or modelling. In addition, FWENZ and the REC can provide relatively coarse estimates of some hydrological statistics that are most reliable for streams and small rivers (e.g., mean annual low flow (MALF) and mean flow, Table 3.1). Simple measurements gathered in the field can be used to cross-validate these models, or more importantly, to provide information on the discharge and other flow variables at the time of habitat assessment (see SHAP, Harding et al. 2009).

Table 3.1: *Example of an output from the River Environment Classification with parameters that are particularly relevant for characterising restoration sites and finding matching reference locations (data from SHAP protocols; Harding et al. 2009).*

Parameter name	Variable
NZ Reach Number	9000495
Catchment Area m^2	4780800.00
Catchment Proportion Exotic Forest	0.00
Catchment Proportion Indigenous Forest	0.00
Catchment Proportion Pastoral Farming	0.60
Catchment Proportion Urban	0.36
Distance to Coast (m)	5028.30
Flow	0.15
Order	2.00
Rec Climate	Warm-Dry
Rec Geology	Alluvium
Rec Land cover	Urban
Rec Source of Flow	Low-Elevation
Rec Valley Landform	Low-Gradient
Segment Maximum Elevation (m)	13.80
Segment Minimum Elevation (m)	11.67
Segment Sinuosity	1.18
Segment Slope	0.00

Select your survey reach

If there are several streams within a catchment that are being restored and you are unable to monitor all sites, or if a significant length of stream is undergoing restoration, then you can select monitoring site locations (reaches) randomly or based on best judgement. Although random site selection provides an unbiased estimate of conditions within the stream section being restored, a judgemental approach may help ensure that the study reach is representative of the stream as a whole and that reference and restored reaches are more closely matched.

The aim of most restoration monitoring is to monitor temporal changes. Therefore, you will select potentially only one or a few restoration sites, but visit each site on multiple occasions, perhaps over a considerable time period. It is important to ensure that sites can be found again, particularly after substantial changes have occurred in the surrounding landscape or with changes in assessment personnel. Recording accurate grid references, noting prominent structures nearby, marking permanent photo-points, and drawing site diagrams will all aid in ensuring that the same reach is resampled on subsequent occasions.

Survey reach length

Reaches of 50–100 m length are usually practical for integrating representative information on small streams, but note that we recommend a minimum length of 150 m of wadeable stream for fish assessment. The rule of thumb for habitat assessment applied in the Stream Habitat Assessment Protocols (SHAP) manual is 20 times the average water width with a minimum of 50 m and maximum of 500 m (Harding et al. 2009). Sampling reaches should contain mesohabitats (runs, riffles, pools) that are representative of the larger stream length and broadly correspond to habitat types present at reference sites. Avoid confluences with other streams in the sampling reach if possible, and it is also important to have a buffer between upstream and downstream unrestored areas to avoid edge effects.

Monitoring timescales

To establish restoration success, you need to monitor both pre- and post-project. Ideally, you should obtain as much information as resources allow as a **baseline** before the restoration project begins, such as seasonal monitoring of relevant ecological indicators for 1–2 years beforehand. If seasonal monitoring is not feasible, we recommend at least 3 years of annual monitoring in summer to establish a pre-restoration baseline. It is best to start with as wide a range of indicators as practical because, even if all indicators are not used routinely after the restoration project is put in place, they may become important in later years and can be included in a monitoring programme if the appropriate baseline measures have been made.

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Part four:

Choosing indicators for your goals

The Restoration Indicator Toolkit:
Indicators for monitoring the ecological success of stream restoration

Choosing indicators

So far, you have chosen your primary goals for the restoration, identified any constraints to achieving those goals, and selected an appropriate reference site or developed a guiding image to judge restoration success. The next step is to choose a range of indicators to match your goals.

Assemble a group of people who will be involved in monitoring your stream restoration site. Review your project goals and have site photographs of the restoration site and reference site, and any data that you have collated at hand. Use the tables below (Table 4.1 and 4.2) and the detailed descriptions in Part 5 to assign indicators that you can use for each of your goals. Go through each of the indicators that match your goals and make a decision about whether they are relevant to your site. It will be helpful to keep these questions in mind:

- What are the key problems at your site that you want to resolve?
- What does your reference stream or guiding image look like? (I.e., what are you aiming for and what do you need to measure to prove that you achieved it?)
- What is going to change with the management methods used? (E.g., it will be pointless measuring shade in a wide stream if no trees have been planted.)
- What negative outcomes that might result from restoration should be monitored? (Remember: one of the intentions is to do no harm!)
- Are there any ecological constraints that will limit restoration outcomes?

In this Toolkit we have focused on ecological restoration goals, i.e., we are defining success as returning towards a natural reference state or a guiding image rather than other societal goals, such as improved fisheries or property protection. Therefore, we present indicators for six main **ecological goals**. If your goals differ from those, make sure that you have developed an appropriate indicator to measure the new goal(s).

Many of the indicators we describe will be *relevant to* a number of goals (see Table 2.1). A good way of thinking about whether to include an indicator is to ask – is it an *indicator of* the specific goal? For instance, water temperature and dissolved oxygen are relevant to aquatic biodiversity, but are not indicators of biodiversity. Several of the indicators appear in the tables below under two or more goals; e.g., aquatic invertebrates can be used to assess water quality and biodiversity, and some key water quality variables are also indicators of natural habitat. These are not measured in a different way; we have arranged them so that if your goal is natural habitat but not water quality, then the key water quality variables that contribute to natural habitat will still be included.

Table 4.1: *List of important indicators for each ecological restoration goal. See Part 5 for details of indicators and units of measurement.*

Natural habitat (NH)	Aquatic biodiversity (AB)	Ecosystem function (EF)	Water quality (WQ)	Down-stream health (DH)	Terrestrial biodiversity (TB)
Water temperature	Benthic macro-invertebrates	Organic matter processing	Water temperature	Nutrients	Terrestrial plant biodiversity and survival of plantings
Shade of water surface	Periphyton	Ecosystem metabolism	Nutrients	Faecal indicators	
Terrestrial plant biodiversity and survival of plantings	In-stream macrophytes	Dissolved oxygen	Faecal indicators	Water clarity	
Water and channel width	Stream mega-invertebrates	Leaf litter retention	Dissolved oxygen	Toxicants	
Stream-bed particle size	Fish		Water clarity	Water temperature	
Mesohabitats			Rubbish		
Bank erosion and condition			Benthic macro-invertebrates		
Water clarity			Periphyton		
Organic matter abundance			pH		
Longitudinal profile variability			Toxicants		
Residual pool depth					
Rubbish					
Periphyton					
In-stream macrophytes					
Riparian microclimate					

Table 4.2: *List of some additional goals that you may have for your restoration site and their relevant indicators. Although the main focus of this Toolkit is on goals for ecological restoration, several indicators can be applied to measure the success of these human or management focused goals.*

Cultural (C)	Aesthetic (A)	Fisheries (F)	Recreation (R)
Water clarity	Rubbish	Water temperature	Faecal indicators
Faecal indicators	Water clarity	Water clarity	Rubbish
Stream mega-invertebrates	Terrestrial plant biodiversity and survival of plantings	Residual pool depth	Terrestrial plant biodiversity and survival of plantings
Fish	In-stream macrophytes	Benthic macroinvertebrates	Water clarity
Cultural Health Index*¥	Periphyton	Fish	Periphyton
Cultural Opportunity Mapping and Assessment (COMA)*§	Bird diversity*	In-stream macrophytes	In-stream macrophytes
Traditional use plant species*		Traditionally harvested aquatic animal species*	Bird diversity*
		Commercial fish species*	

* *Indicators that have not been developed as part of the toolkit but could be included to address the goal.*
¥ *Tipa & Tierney (2006)*
§ *www.niwa.co.nz/_data/assets/pdf_file/0008/91997/Shallow-lakes-wetland.pdf*

In Appendix B we provide a range of hypothetical examples that describe how to assemble an appropriate list of indicators based on project goals.

Brainstorming project goals.
Photo: John Quinn

The scenarios in Appendix B describe a restoration activity for a stream, the management methods to be employed, the catchment context that the restoration site is in, who will be doing the monitoring, and their goals for undertaking restoration. We show the indicators that the project team chose given those hypothetical scenarios.

Riparian planting along an urban stream.
Photo: Steph Parkyn

Displaying monitoring data for project goals

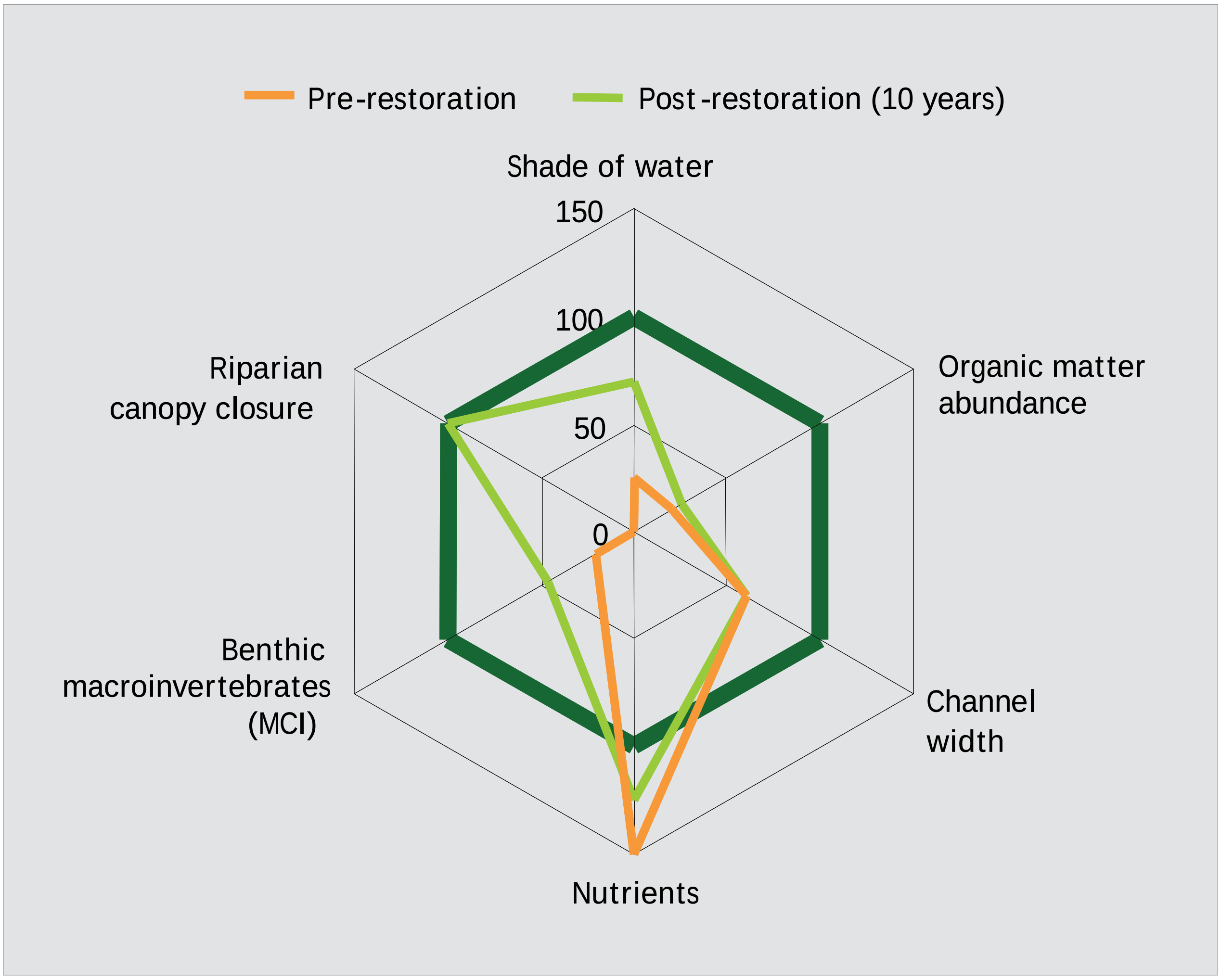
For each of the indicators that you have chosen to measure, data from the restoration site and reference site (and/or control, unrestored site) can be displayed in a graph showing changes over time in much the same way as we use to demonstrate timescales of change in Part 5. In cases where there is no reference site to measure, then the *a priori* level of success that you have assigned for each indicator based on the guiding image can be displayed on the graph as a target to reach.

An alternate way to display the results of monitoring relative to your project goals is to use a radar diagram of the key indicators for each goal (Figure 4.1). This is a simple and concise way to show the success (or failure) of a restoration project relative to a reference site (or guiding image) and to report a summary of project goals to stakeholders. Results over time can be shown in the same graph or in several graphs.

USING A GUIDING IMAGE TO JUDGE RESTORATION SUCCESS

To use the guiding image as an endpoint, you will need to make a list of the change you want to see at your site for each of the indicators. For example, if the substrate at the restoration site is predominantly silty, your criteria of success for the stream-bed particle size indicator might be returning the stream to predominantly stony substrate, according to your guiding image. Part 5 of this document should help you decide what the magnitude and direction of change for each indicator is likely to be.

Figure 4.1: *Example of a radar diagram (Microsoft Excel graph), which can be used to summarise the results of monitoring. This pasture stream has been fenced and planted with native vegetation and the goals were natural habitat and water quality. After 10 years, canopy closure of the plantings has been achieved (100% of reference), but shade over the stream is not yet at reference conditions (shown by solid dark green line), the channel has not started to widen, and organic matter in the stream is slowly increasing. Nutrients have decreased but are still above reference and the macroinvertebrate community index (MCI) has begun to improve (50% of reference).*



Part five:

The Indicators

The Restoration Indicator Toolkit:
Indicators for monitoring the ecological success of stream restoration

In this section we give guidance on:

- the importance of each indicator
- appropriate methods to measure the indicator
- when to take measurements.

In some cases, we suggest several methods that could be used to measure the indicator. You can choose the method most suitable to your situation, as long as the same method is consistently used for the length of the monitoring period at both restoration and reference (if applicable) sites. Guidance on when to measure the indicator assumes that pre-restoration and immediate post-restoration measures are taken in all cases and subsequent annual, 2- or 5-yearly measurements are taken depending on the timescales of change expected for each indicator. Typically, we suggest more frequent measurements during times that most change is expected, so these suggestions may need to be adjusted depending on the site-specific changes at your site.

For each of the indicators we have made predictions (graphs) of the likely timescales of success relative to a reference site in the same geology and matched in terms of stream size, etc. (see reference site selection in Part 3). In some cases, these predictions are informed by data from literature, but often the predictions are based on expert opinion and provided to give an indication of the hypothetical trajectory of stream restoration success. Predictions are generally based around two main scenarios: riparian management and channel reconstruction. However, alternative scenarios that are specific to some indicators have also been included when appropriate.

The **Riparian Management Scenario** has a hypothetical restored stream with the following features:

- 3–4 m wide channel
- canopy closure above the stream after 10 years
- fenced pasture stream
- replanted with natives at least 10 m buffer width either side of stream
- buffer along whole perennial stream length
- moderate gradient of 1–2%
- catchment dominated by pasture but some patches of remnant bush in headwaters
- water quality not limiting to biota, no toxins
- no barriers to fish/biota recruitment.

The **Channel Reconstruction Scenario** assumes the same conditions as above, but includes active channel modifications such as:

- remeandering of straightened channels
- placement of logs, boulders
- creation of pools

- removal of excess sediment from stream-bed
- bank remodelling.

In each of the timescale graphs we assume that 0 is the initial (pre-restoration) condition of the stream and that the time along the x-axis indicates years after restoration activity is initiated. The recovery timescales are expressed as a percentage of reference (100% = typical reference condition) on the y-axis and estimates of absolute values (where known) are shown on the right hand side secondary y-axis. A grey band on the graphs at close to100% of reference indicates that reference condition will be variable.

Habitat

Many of the habitat indicators suggested here are described in SHAP (Harding et al. 2009). We recommend that you undertake level P2 or P3 of the SHAP protocol in its entirety for each of your restoration sites and appropriate reference streams. The descriptions of these measures are included separately below for each of the indicators.

Water and channel width

Goal(s): NH

Background: The channel and wetted stream width is an indication of the amount of habitat available to stream life and an indication of flow or morphological changes to the stream. The conversion of forest to pasture is known to have narrowed channel widths (Davies-Colley 1997), at least in small hill-country streams, so we might expect that most stream restoration activities (e.g., both riparian management and channel reconstruction) will alter water and channel widths in similar settings.

Method: A tape measure or hip chain is used to measure water width perpendicular to stream flow (at base flow conditions) and bank-full channel width (to height of banks) at up to 20 evenly spaced points along the stream thalweg (deepest point). The reach surveyed should ideally be at least 20 times the average channel width (with a minimum reach of 50 m and maximum of 500 m).

When: Water and channel widths should be measured annually at low flow.

Timescales and measures of success: Figure 5.1 shows the expected trajectory of stream channel width after fencing and planting of the stream riparian zone. The channel initially narrows slightly due to removal of cattle access and encroachment of rank vegetation. However, after a decade or so the channel is expected to start widening owing to shading of pasture grasses (that armour the stream banks) by woody riparian vegetation, resulting in erosion of the banks (Davies-Colley 1997, Parkyn et al. 2005). The actual channel widening is step-wise, occurring mainly during large (bank-full or near bank-full) storm-flow events. Eventually the channel width is expected to approach that of a reference stream in an identical-sized catchment (identical flood flows) – a width approximately twice that of the original pasture stream. These changes may result in pulsed inputs of sediment to the stream as the channel re-adjusts to a shaded morphology (Collier et al. 2001).

The water width is not shown in Figure 5.1. It is expected to broadly follow channel width but be slightly smaller in magnitude. Changes in the ratio of channel width to stream width may be an additional indication of a shift to reference condition. Davies-Colley & Quinn (1998) found that base flow water width averaged about 83% of the channel bank-full width in forest streams but tended to be a higher proportion of bank-full width (average of 89%) in pasture streams in the Waikato.

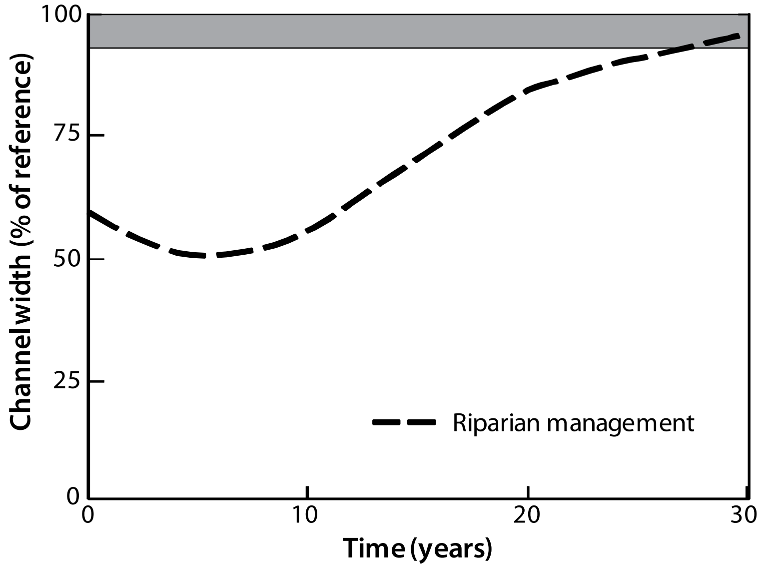


Figure 5.1: *Hypothetical trajectory of channel width in a pastoral stream after fencing to exclude livestock (primarily cattle) and riparian planting.*

Bank erosion and condition

Goal(s): NH

Background: The condition of stream banks may change considerably following stream restoration. Changes to flow regime from land use changes within the catchment or changes in management of dam release flows can also influence bank erosion. Fencing stock away from streams will reduce the amount of sediment released from stream banks, but shading by tall riparian vegetation will increase erosion during flood events and ultimately restore the stream to its previous width.

Method: A number of attributes related to stream bank condition are included in the P3 Riparian procedure of SHAP (Harding et al. 2009). For both sides of your stream reach:

1. Measure the stream bank length affected by gaps in the buffer (to the nearest 0.1 m).
2. Assess riparian **wetland soils** by measuring the length of stream bank with saturated or near saturated soils, i.e. soils that are soft/moist underfoot.
3. Measure the length of the stream bank with **stable undercuts**; often these are stabilised by vegetation roots.
4. Count (or measure) the number (or length) of **livestock access** points.
5. Measure the length of the site subject to active **bank slumping**. This category includes only obvious slips and erosion.
6. Measure the length of **raw bank** on the left and right banks indicated by exposed unvegetated banks, including an absence of moss, lichen, and small plants.
7. Measure the cross sectional area of eroded **rills and channels** along the length of the site.

8. We suggest that you select the measures that are most relevant to your site; it is likely that these would include **stable undercuts**, **livestock access**, **bank slumping**, and **raw bank**.

When: Assessment should be made annually.

Timescales and measures of success: Damage to stream banks at livestock access points and slumping caused by trampling are likely to heal within the first few years of livestock exclusion (Figure 5.2).

However, while slumped banks may not be active sources of sediment when grasses have grown over them, they will still be prone to erosion from flood events.

Channel widening (described above) will begin to occur after tall vegetation shades out stream-side grasses (after 10 years) and the amount of raw bank is expected to increase at this time. The erosion of banks will be episodic, so annual variation will be high. Figure 5.3 shows a generalised curve for what to expect in terms of bank slumping and amount of raw bank after riparian management of the pasture stream described in Scenario 1.

Figure 5.2: Bank damage of a pasture stream in pumice geology (top) and a downstream reach 3 years after fencing and planting (below).
Photo: Steph Parkyn

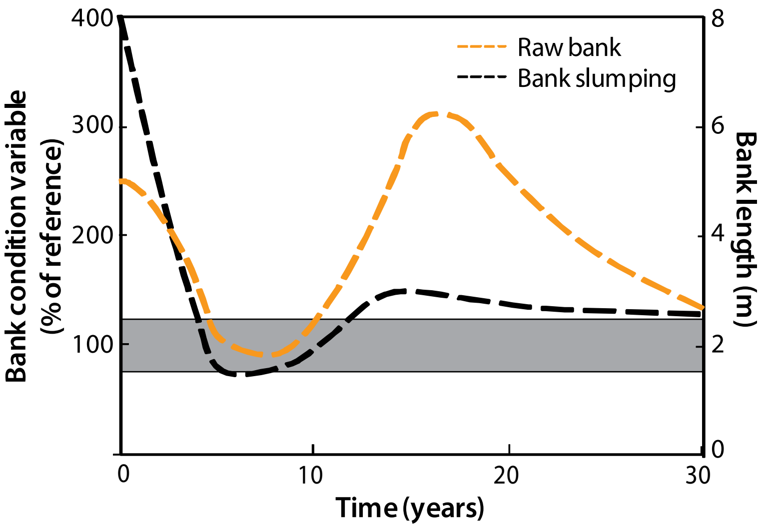


Figure 5.3: *Hypothetical timescales for the length of stream bank affected by bank slumping or raw banks following riparian management (fencing and planting) of a pasture stream reach (100 m long).*

Longitudinal profile variability

Goal(s): NH

Background: The channel longitudinal profile variability (LPV) provides a quantitative measure of changes in the variability of depth along a restored reach as a simple indicator of habitat variability.

Method: Measure the water depth along the channel thalweg (i.e., the deepest part of the channel cross-section) at 50 equally spaced distances along the channel (e.g., at 2 m intervals along a 100 m long reach). The data are used to calculate the standard deviation (SD) of depth for the reach.

When: After riparian management, assessments could be made at 5-yearly intervals. After channel reconstruction, assessments should be made annually for the first 5 years and then at 5-yearly intervals.

Timescales and measures of success: Channels that have been simplified by channelisation or lack of large wood input are expected to increase in longitudinal profile variability (LPV) after channel reconstruction or through time as wood is recruited into the channel from riparian reforestation. Channel reconstruction is expected to produce an abrupt step change in LPV to a new state, whereas riparian afforestation would be expected to have minimal effect until significant input of large wood occurs (after 70–400 years; Meleason & Hall 2005). Hypothetical responses of a previously straightened reach to restoration by riparian reforestation and channel reconstruction are shown in Figure 5.4.

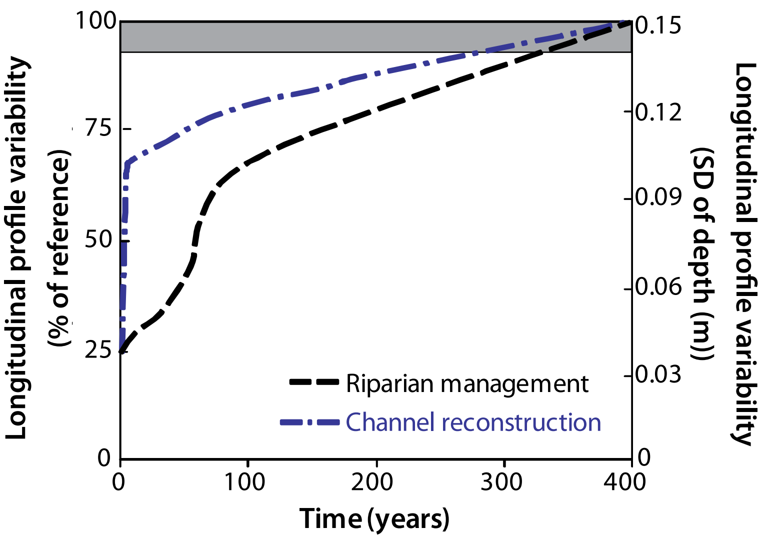


Figure 5.4: *Hypothesised restoration timescales for longitudinal profile variability (LPV) in response to two types of restoration action for a hypothetical stream with a mean depth of 0.42 m.*

Mesohabitats

Goal(s): NH

Background: Mesohabitats are defined here as the hydraulic habitats within a stream reach characterised by different mean water velocities and depths. The commonest habitat types are rapids, riffles, runs (or glides), pools, and backwaters (defined on page 34 of SHAP manual).

RIFFLE: shallow depth, moderate to fast water velocity, with mixed currents, surface rippled but unbroken.

RAPID: shallow to moderate depth, swift flow and strong currents, surface broken with white water.

RUN: character inbetween that of riffle/rapid and pool, slow to moderate depth and water velocity, uniform to slightly variable current, surface unbroken, smooth to rippled.

POOL: deep, slow flowing with a smooth water surface, usually where the stream widens and/or deepens.

BACKWATER: slow or zero flow zone away from the main flowing channel that is a surface flow dead-end; although flow could down-well to or up-well from groundwater.

Mesohabitats are often associated with different substrate types and have identifiable surface flow patterns (Figure 5.5). Stream biota have different hydraulic habitat preferences and species often benefit from a mix of different habitats for different activities (e.g., different feeding modes, resting, spawning) and life stages (Jowett et al. 2008, Jowett & Richardson 1994, Jowett et al. 1991). Increased mesohabitat diversity can result in greater biodiversity assuming no other constraints are present.

Figure 5.5: Run and riffle mesohabitats.
Photo: Rob Davies-Colley

The primary drivers of mesohabitat types along a reach are the channel slope, flow variability (at the annual–decadal scales), catchment geology, and sediment supply. Channelisation (straightening, widening and/or deepening) typically reduces the mesohabitat diversity and a high sediment supply can infill pools, reducing their volume and area of habitat. Reference or benchmark sites of comparable slope and catchment geology can provide the guiding image for proportions of mesohabitats. In some cases, the aim may be to increase the proportion of a missing or poorly represented habitat type that would be expected to occur in the reach setting (e.g., to increase the percentage of pools to enhance habitat for certain fish). The method below builds on the SHAP P2 mesohabitat assessment by providing a measure of mesohabitat diversity based on Simpson's Diversity Index (1 – D).

Method: Walk along the stream at the water's edge following the thalweg and record the dominant **mesohabitat length** in metres (from tape measure or hip chain) of each mesohabitat encountered as rapid, riffle, run, pool, backwater, and other. "Other" habitats may include cascades, chutes, or falls and should be measured separately. Sum the total length of each habitat type along the monitoring reach. Use these data to calculate Simpson's diversity (shown in Table 5.1) as follows:

$1 - D = 1 - (\sum n(n - 1)/(N (N - 1)))$

where n is the length of an individual mesohabitat type and N is the total length of all mesohabitats.

Table 5.1: *An example of the calculation of mesohabitat using Simpson's Diversity Index (1 — D) for a restoration reach (A) and a reference reach (B) of similar slope, catchment area, and geology.*

Mesohabitat	Lengths (m)	Lengths (m)	n*(n — 1)	n*(n — 1)
	Restoration (A)	Reference (B)	A	B
Rapid	0	5	0	20
Riffle	15	30	210	870
Run	85	30	7140	870
Pool	0	45	0	1980
Backwater	0	0	0	0
Other	0	0	0	0
Sum length (m)	100	110	7350	3740
Simpson's diversity (1 — D)			0.26	0.69

When: After riparian management, assessments could be made at 5-yearly intervals during summer base flow. After channel reconstruction, assessments should be made annually for the first 5 years and then at 5-yearly intervals.

Limitations: Two cautions are that:

1. mesohabitats are influenced by flow (e.g., riffles can become runs at high flow and deep runs can become pools at low flow), and
2. mesohabitats may vary at reference sites under standardised flow conditions in response to natural storm disturbances.

Consequently, assessments over time should be made under standardised flow conditions (e.g., summer base flows) and repeating measurements at both restoration and reference sites will enhance the reliability of the assessments by helping to account for natural variations.

Timescales and measures of success: The timescale for restoration will be similar to that for longitudinal profile variability (LPV). Channel reconstruction is expected to produce an abrupt step change in mesohabitat diversity after the new channel plan is engineered. Subsequently, a slow increase in mesohabitat diversity is predicted until large wood input commences at about 70 years and increases

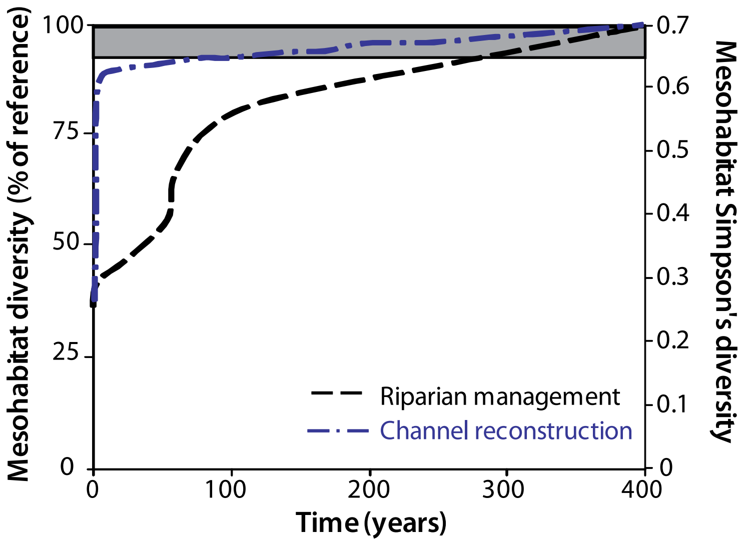


Figure 5.6: *Hypothesised restoration timescales for mesohabitat diversity (Simpsons 1 — D) in response to riparian management or channel reconstruction.*

thereafter (Meleason & Hall 2005). In contrast, riparian management of a pasture stream is predicted to be slower, with gradual deepening of pools as the supply of fines is reduced and greater change when significant input of large wood occurs. Hypothetical responses of a previously straightened reach to restoration by riparian reforestation and channel reconstruction are shown in Figure 5.6.

Residual pool depth

Goal(s): NH, F

Background: Residual pool depth (RPD) is the difference between the maximum water depth of a pool and the water depth at the riffle crest (hydraulic control) immediately downstream of the pool. Residual pool depth estimates the maximum depth of water that would remain in the pool when the stream ceases flowing and gives an indication of the remaining habitat available at these times, but not necessarily the quality of this habitat, i.e., reduced flow may change the suitability of habitat for certain biota. Residual pool depth can provide an indication of pool infilling due to increased sedimentation.

Method: We recommend the method outlined in P2 of the SHAP (Harding et al. 2009):

1. At each pool (maximum of 3) measure residual pool depth by measuring the maximum depth of water at the deepest part of the pool and the crest depth of water at the riffle crest immediately downstream of the pool. (An estimate of maximum pool depth is sufficient if it is too deep to measure, but note that it was estimated.)

2. Calculate average residual pool depth (maximum depth minus crest depth).

When: Once a year during base flow conditions when it is safe to enter the stream to perform measurements. If the focus is to assess the potential effects of a large sediment-carrying flow event, wait at least 7 days after the flow event for the stream-bed to stabilise.

Timescales and measures of success: Pool infilling will occur as a result of high sediment loads and the inability of flow to shift that sediment. Reduction in pool infilling and maintenance of residual pool depth requires a reduction in sediment delivery, i.e., by planting riparian vegetation and/or catchment vegetation. The length of time before a reduction of sedimentation and eventual decrease in pool infilling is realised will be highly site-specific, depending on restoration techniques and local stream conditions, especially slope, and the episodic nature of sediment and flow delivery (i.e., occurrence of event-based flows). Figure 5.7 shows a hypothetical recovery curve for residual pool depth in response to riparian management

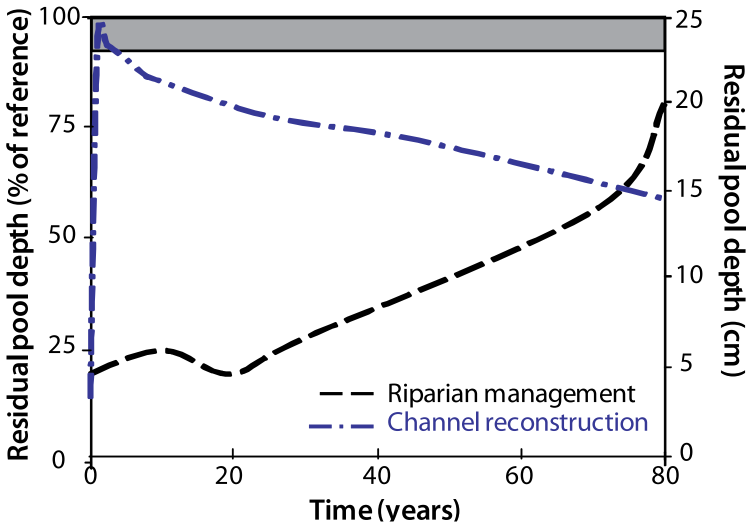


Figure 5.7: *Conceptual recovery curves for residual pool depth in response to riparian planting and channel reconstruction.*

and channel reconstruction. It is expected that riparian planting may result in a short-term increase in sediment delivery due to the shading of stream bank grasses (Parkyn et al. 2005), followed by a decrease in sediment delivery as tree roots restabilise banks. High flows may be required to "flush" pools in which case the timescale for recovery can be very long, but may accelerate by the delivery of large wood after approximately 70 years (Meleason & Hall 2005). In comparison, channel reconstruction by effectively "scooping out" excess sediment or introducing a hydraulic drop (e.g., weir, natural or artificial log) would result in the immediate increase in residual pool depth. Streams with low slopes and high sediment loads would naturally infill again with time. There are no recommended guidelines provided for residual pool depth; therefore, values should be compared to reference to evaluate stream condition.

Water clarity

Goal(s): NH, WQ, A, F, DH

Background: Visual clarity is such a fundamental attribute of waters (e.g., it is explicitly protected in the RMA1991) that its measurement should be strongly considered for monitoring response to all restoration efforts. Water clarity refers to light transmission through water, and has two important aspects:
visual clarity (sighting range for humans and aquatic animals) and **light penetration** for growth of aquatic plants (Davies-Colley & Smith 2001, Davies-Colley et al. 2003).

Visual clarity is an index of sighting ranges of practical importance in waters – for humans and for sighted aquatic animals such as fish and aquatic birds (Davies-Colley & Smith 2001, Davies-Colley et al. 2003). Visual clarity of waters is an important attribute affecting habitat for aquatic life as well as recreational safety and amenity value of waters. Light penetration is also fundamentally important because it controls light availability for growth of aquatic plants (Kirk 1994). There are existing guidelines for both visual clarity and light penetration (MfE 1994, ANZECC 2000).

Method:

BLACK DISC CLARITY OBSERVATION

Visual clarity of waters can be quantified by the maximum horizontal sighting distance (extinction distance) of a black target because this approximates sighting ranges of practical importance, such as fish reactive distance. The black disc method (Davies-Colley 1988) is well-proven and the method is well described in various publications, notably the MfE (1994) guidelines on colour and clarity of waters. An underwater periscope is used to observe (horizontally) under water, and a tape measure is used to measure the extinction distance of the black disc target (Figure 5.8). The extinction distance is recorded as the average of the disappearance distance and reappearance distance (see Davies-Colley 1988, Zanevald & Pegau 2003).

A fundamental assumption of the method is that the horizontal path of sight is uniformly lit; take care that shadows are not cast across the path of sight (under sunlit conditions). It is important to ensure that the observer's eyes are adapted to the underwater light (taking a minute or two). Finally the disc should be observed against the water background, not against the stream bank or rocks, for example.

Visibilities less than 100 mm are difficult to measure directly because the viewer itself may distort the light field in the water close to it. However, visibilities can be measured on a volumetrically diluted sample

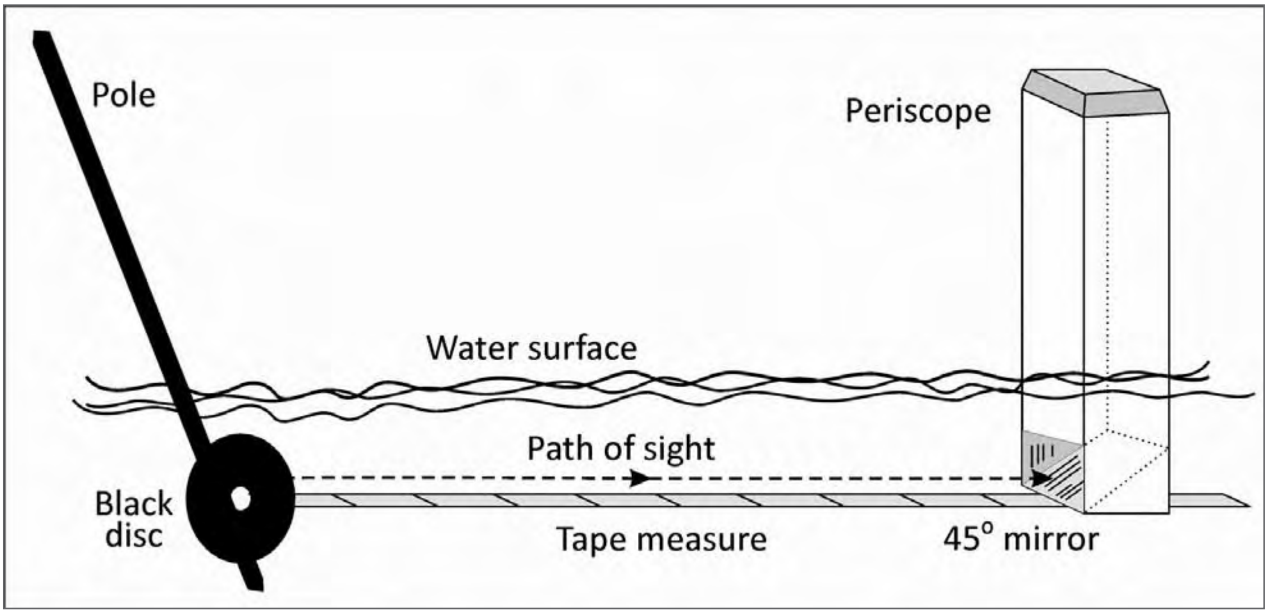


Figure 5.8: Schematic of black disc method of water clarity measurement.
Illustration: Rob Davies-Colley

contained in a trough (Davies-Colley & Smith 1992).

The state of flow of the stream or river should be noted (ideally as actual flow at a nearby hydrometric site) at the time of any visual clarity measurement.

SHMAK CLARITY TUBE

Various "clarity tubes" have been suggested for indexing visual water clarity and, although these are not recommended for robust scientific monitoring purposes, they can be used by community groups to monitor gross changes in turbid waters. One design that has scientific merit is the clarity tube from the Stream Health Monitoring and Assessment Kit (SHMAK). It consists of an optically clear acrylic tube for containing the water sample and an aquarium magnet pair with a small (20 mm diameter) black disc target attached to the magnet on the inside of the tube (Kilroy & Biggs 2002). The tube is filled with a water sample from the stream and held horizontally while observing the black disc target. (The position of this target is adjusted to the extinction point with the matching aquarium magnet.) The SHMAK tube visibility approximates the black disc visibility at low clarity (<0.5 m). However, visual ranges of importance in waters are often greater than can be measured accurately by the clarity tube (e.g., MfE 1994 recommend a minimum of 1.6 m black disc visibility for bathing safety), and we recommend that regional councils monitor visual clarity by the black disc method rather than by clarity tube.

TURBIDITY

Turbidity is an index of cloudiness of water due to light scattering; it is often measured in nephelometric turbidity units (NTU) as a way to quantify visual clarity. However, turbidity is a relative (instrument-specific) measurement versus arbitrary standards (Davies-Colley & Smith 2001), so is a poor substitute for visual clarity. Nevertheless, turbidity has some important virtues, notably that it can be measured continuously (including at night), and these attributes can be exploited to estimate visual clarity with suitable (local) calibration. We do not recommend reporting of turbidity in NTU; that is, turbidity data should always be calibrated (locally) to visual clarity (Davies-Colley & Smith 2001).

When: Monthly or at times when water samples for water quality analysis are taken, especially for the year(s) prior to and following restoration activities. Measurements should occur seasonally or annually after that.

Limitations: Because water clarity is strongly (inversely) related to state of flow in rivers (Smith et al. 1997), flow needs to be measured at the same time as clarity to interpret the visual clarity regime and the trend in visual clarity over time (Smith et al. 1996). If flow is not actually measured at the monitoring site, state-of-flow can be indexed to a nearby continuously recording hydrometric site, which ideally is on the same stream.

Visual clarity is not a measure of light penetration of waters, despite a broad overall correlation (Davies-Colley & Smith 2001). Light penetration can be difficult to measure and is best indexed by the diffuse light attenuation coefficient, which is measured by lowering a light sensor into water (Davies-Colley et al. 2003). However, Davies-Colley & Nagels (2008) recently reported a simpler, semi-empirical model for predicting light penetration in river waters from black disc visual clarity (or turbidity) measurements supplemented with measurements of coloured dissolved organic matter.

Timescales and measures of success: A rapid improvement in visual clarity may be expected after fencing that excludes cattle and some other livestock (deer) from channels, because these animals are very damaging to riparian areas and stream banks. Stock exclusion is expected to result in recovery of riparian vegetation and elimination of stock-induced mobilisation of sediments. However, after a decade or so visual clarity may actually worsen for a period of years owing to shading of pasture grasses (that armour the stream banks) by woody riparian vegetation, resulting in erosion of the banks and widening of the channel (Davies-Colley 1997, Parkyn et al. 2005). In this regard, visual clarity and the sediment regime are unusual, as conditions may actually get worse for a time before getting better (Figure 5.9). Note: MfE (1994) recommend a minimum of 1.6 m black disc visibility for bathing safety, which could be used as a secondary benchmark of success, as long as the natural processes of clarity reduction are also understood.

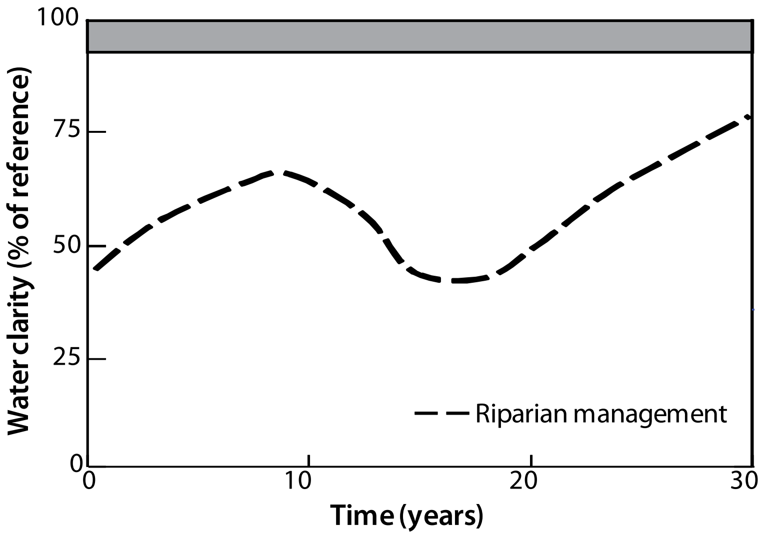


Figure 5.9: *Hypothetical trajectory of visual clarity after riparian restoration.*

Stream-bed particle size

Goal(s): NH

Background: Stream-bed particle size is a strong driver of the biological community in streams. Stream macroinvertebrate diversity and abundance are greatest on cobble- and boulder-sized particles in stream-beds (Death 2000). Fine sediments (sand and silt) are generally considered unsuitable for the majority of invertebrates (e.g., mayflies, stoneflies, and cased caddisflies), except for certain taxa such as worms, molluscs, some midges, and the burrowing mayfly *Ichthybotus hudsoni*. Most native fish are benthic in habit, using the stream-bed for shelter, foraging, and nesting, and thus benefit from large particles (cobbles and boulders). Loss of large particles, or increases in fine sediment, can cause a decrease in native fish abundance and diversity (Richardson & Jowett 2002) due to degradation of their habitat.

Stream-bed particle size varies naturally from one stream to another, and can be predicted with knowledge of geology, climate, topography, and position in the stream network (Harding et al. 2009). For example, boulders are more common in headwaters, whereas river mouths are typically composed of gravel, sand, and silt. Most Auckland streams, which drain sandstone or mudstone catchments, are naturally "soft-bottomed", whereas Hawkes Bay streams, which drain harder greywacke, are typically cobble "hard-bottomed" streams. However, changes in land use, such as urbanisation or replacement of native bush with pasture and increasing access by grazing animals to stream channels, usually lead to increased deposition of fine sediment on stream-beds. This increase in fine sediment can be reversed to some extent through appropriate riparian management. Fencing or revegetating riparian buffers can reduce input of fine sediment by:

- stabilising stream banks against erosion by stream flow
- physically trapping sediment runoff from the catchment
- keeping stock from trampling stream banks.

Method: The following method for stream-bed particle size evaluation, known as the Wolman walk, has been adapted from SHAP (Harding et al. 2009). Please also note that protocols for assessing sedimentation are currently being evaluated and new measures may be recommended for use instead of – or in addition – to the Wolman walk, particularly if your site is affected by excess silt and sand.

1. Lay tape measures across the stream at 6 positions including 2 riffles, 2 runs, and 2 pools.
2. At each cross section, randomly select 10 particles while wading across the stream. To achieve random selection, pick up the particle immediately in front of your boot at each step across the stream. If the particles are completely covered in a layer of fine sediment (i.e., the first

Figure 5.10: Using the "Wolman stick".
Photo: Richard Storey

particle touched is sediment and not the larger particle beneath), and if you are able to pick the sediment up without pinching finger tips together (to avoid overemphasising transient fine deposits of silt/sand), then record that particle as silt or sand.

3. Measure each particle using a gravelometer, or measure the length of its second-longest axis using a "Wolman stick" (Figure 5.10, 5.11), assigning it to one of the categories in Table 5.2.

4. Data can be reported in several ways: in the form of a cumulative frequency graph, as d_{50} (median particle size), as % fine sediment (<2 mm) or % cobble, or using a substrate index (e.g., the sum of the mid-point values of the size classes weighted by their proportional cover; Quinn & Hickey 1990). We recommend % fine sediment as it is conceptually simple and is ecologically relevant to benthic biota.

Table 5.2: *Size classes for the gravelometer or "Wolman stick".*

Size category	Category name		Size category	Category name
<0.063 mm	Silt, mud		16—64 mm	Large gravel
0.063—2 mm	Sand		64—128 mm	Small cobble
2—4 mm	Small gravel		128—256 mm	Large cobble
4—8 mm	Small medium gravel		256—4000 mm	Boulder
8—16 mm	Medium large gravel		>4000 mm	Bedrock

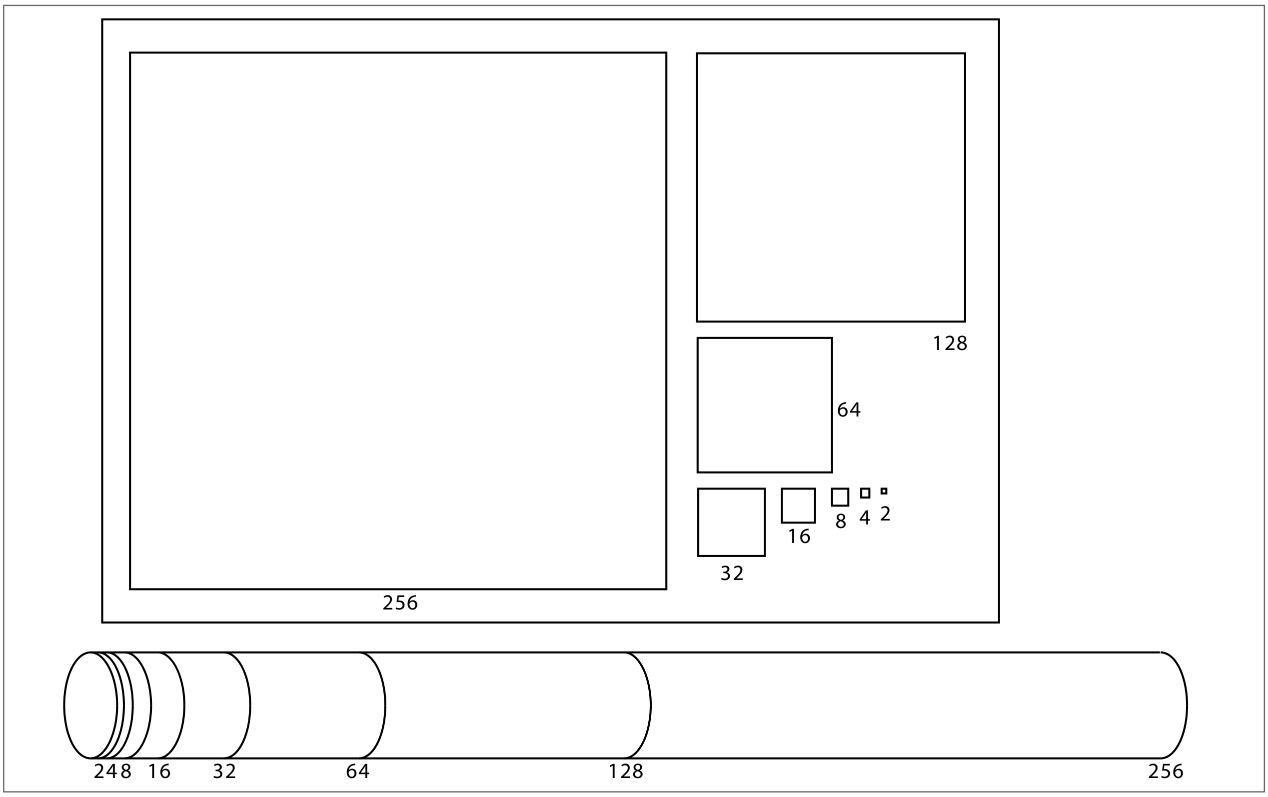


Figure 5.11: *Top: Gravelometer; a metal frame with square holes sized according to Wentworth classes. The "b" axis (second-longest axis) of a stream-bed particle is determined by the smallest hole it can be passed through. Bottom: Wolman stick, essentially a metal ruler marked at intervals according to the Wentworth size classes. The size of a stream-bed particle is determined by placing the stick across the "b" axis of the particle, with one end flush with the side of the particle and reading the size category at the other end of the particle.*

When: Annually during base flow conditions and within the same season each year.

Timescales and measures of success: Deposited fine sediment is expected to decrease significantly within the first 1 to 5 years after stock has been excluded from streams and stream banks (Figure 5.12). This assumes that deposited fine sediment will follow a similar trajectory of decrease as suspended sediment (Williamson et al. 1996, Owens et al. 1996, Line et al. 2000, McKergow et al. 2003, Parkyn et al. 2003, Carline & Walsh 2008). In these studies, fine sediment decreased because in the absence of stock, stream banks stabilised and soils near the stream recovered from treading compaction.

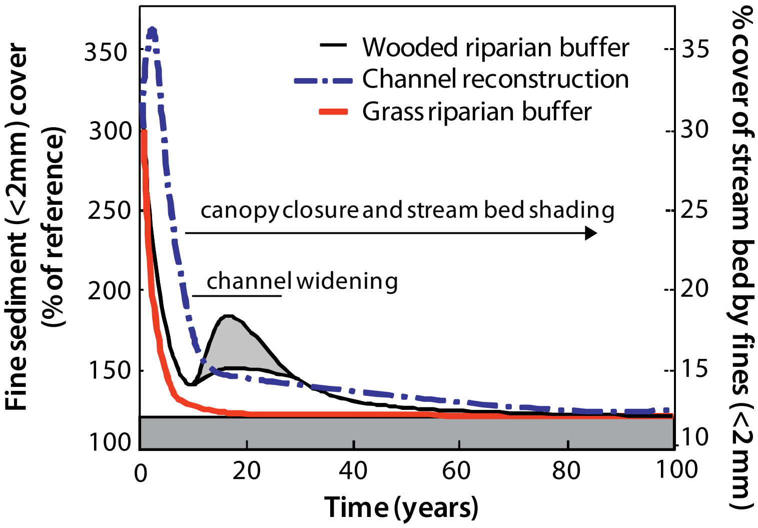


Figure 5.12: *Stream-bed particle size measured as % cover of fine sediment. Three scenarios considered here are: 1) Riparian fencing and planting (wooded), 2) Channel reconstruction + riparian planting and 3) Riparian fencing only (grass buffer). Actual values suggested by the right-hand axis are for scenario 1, derived from Quinn et al. (2009). The grey area under the curve indicates the range of values possible during channel widening. Actual values may change rapidly within this range in response to floods.*

With riparian management, planted riparian trees form a closed canopy and shade stream bank grasses after about 10 years. As this occurs, we expect a temporary increase of fine sediment due to erosion of the stream banks and channel widening to a previous size (Davies-Colley 1997, Parkyn et al. 2003, Carline & Walsh 2008). Observations suggest that during channel widening, the amount of fine sediment in the stream-bed may vary in response to floods. Large floods typically "clean" the stream-bed by washing out fine sediments. However, during channel widening floods will also cause stream banks to slump into the channel, leaving blocks of soil in contact with the stream water. These blocks of soil will be gradually eroded by the current, leading to a build-up of fine sediment over the stream-bed. At the next large flood, the process will be re-set, as the flood will both remove fine sediment and cause more blocks of bank soil to slump into the channel. Channel widening is expected to peak 15–20 years after the restoration work (Davies-Colley 1997), though the channel may still be adjusting after several decades (McBride et al. 2008). During channel widening, the stream morphology will adjust to forested conditions, and the fine sediment load in the stream-bed will gradually decrease (Davies-Colley 1997), leaving coarser stream substrates. As well as causing erosion of stream banks, shading will lead to a loss of aquatic macrophytes. Since macrophytes trap fine sediment around their roots and stems, loss of macrophytes may lead to a decrease in fine sediment. Between 30 and 100 years post-restoration, fine sediment in the stream-bed may continue to decline gradually as the forest buffer increases its capacity to trap silt in pasture runoff. This is expected to occur as leaf litter builds up on the riparian forest floor, and as the riparian soils become more permeable to water entering the buffer via overland flow.

Stream restoration often involves fencing to restrict stock access without planting of trees. In the short-term, such restoration measures typically result in lower fine sediment load than is achieved with riparian tree planting (e.g., Sovell et al. 2000). This is because buffer strips of long grass are usually more effective than an immature forest buffer in trapping the fine sediment in pasture runoff. When grass buffers are used instead of wooded buffers, channel widening and the associated variability in deposited sediment are not expected.

Channel reconstruction (Figure 5.12) is expected to result in some direct short-term changes. For the first few years, a rise in fine sediment is expected (Friberg et al. 1998, Tullos et al. 2009) due to loosening of stream channel sediments during reconstruction, and as the channel morphology adjusts after the engineering works. Riparian fencing and planting, instigated at the time of channel reconstruction, is predicted to reduce the supply of fine sediments to the stream. The total amount of fine sediment may decrease slowly due to retentive structures and the reduced stream power of the meandered reach (Kronvang et al. 1998, Roni et al. 2008). However, the distribution of fine sediment will change. Channel reconstruction and addition of in-stream features usually result in greater complexity in stream morphology, including depositional zones (e.g., pools, berms) and erosional zones (e.g., riffles) (Kronvang et al. 1998). Fine sediment becomes trapped in depositional zones and is cleared from erosional zones. We expect that channel reconstruction would involve widening the channel to the original channel width and will have removed much of the stored sediment from stream banks, but that there may be some short term deposition of fine sediments from channel widening upstream.

Organic matter abundance

Goal(s): NH

Background: The abundance of organic matter (leaves, wood) on the stream-bed provides an indication of the food and nutrients available to stream life. Leaf packs and wood also provide important habitat for fish and invertebrates. Large wood contributes to the retention of other organic matter and in-stream sediment as well as influencing localised hydraulics including hyporheic exchange.

Organic matter abundance is commonly used as part of a rapid assessment of stream condition (Barbour et al. 1999, Harding et al. 2009). Generally, streams that flow through forested catchments with intact riparian vegetation have greater amounts of large wood and coarse organic matter in-stream (Evans et al. 1993, Harding & Winterbourn 1995, Meleason et al. 2005). The amount of organic matter will be subject to seasonal litter inputs and the presence of retentive devices that minimise "flushing" of the system by high flows (Jones 1997, Quinn et al. 2007). Large wood replacement has been a strong focus of restoration efforts in rivers and streams (Bernhardt et al. 2005, Lester & Boulton 2008) where wood had previously been removed to aid navigation or due to logging activities.

Method: We recommend the use of the method provided in P2 of the SHAP (Harding et al. 2009). Select a representative riffle, run, and pool and at each mesohabitat:

- visually estimate the percentage of the wetted bed with wood and leaf packs, including trees, branches, and roots
- calculate the average proportion of the stream-bed where organic matter is present.

When: Annually during base flow conditions and within the same season each year.

Timescales and measures of success: The presence and abundance of leaves and wood in a stream reach will be related to the presence and abundance of current and historical riparian vegetation and the input of leaves and wood from upstream. The establishment of quick growing vegetation will shorten the recovery time. For example, native species such as mahoe, wineberry, and hoheria have soft, N-rich leaves and can be planted alongside unretentive streams to increase food sources (from leaf fall) during the early phases of restoration (Quinn et al. 2000a, b). Exotic deciduous plant species often have soft leaves but they have a different pattern of leaf delivery (high input in autumn) than native evergreen species, which provide a more natural continuous input of both soft and hard leaves.

Figure 5.13 gives a hypothetical recovery curve for organic matter abundance in response to riparian planting with native species. There are no established guidelines for organic matter abundance in streams; therefore, values need to be compared to a reference stream to evaluate success. After approximately 10 years, canopy closure will lead to an increase in leaf litter and small wood (twigs, roots) (Quinn et al. 2009) and after approximately 70 years large wood will increase in-stream (Meleason & Hall 2005). It is expected that organic matter restoration will follow wood jam evolution whereby a critical amount of material collects which then enhances the gradual accumulation of organic matter before the percentage of organic material in-stream stabilises due to the balance between delivery and breakdown (Manners & Doyle 2008). Coarse organic material will accumulate a lot faster than larger woody components, yet be subject to seasonal and hydrological variability, whereas large wood accumulation will take longer but be subject to less variability over time (Bilby & Ward 1991, Davies-Colley et al. 2009). For large wood recovery, the shape of the recovery curve will be the same as for coarse organic matter, but a wood jam and subsequent accumulation is likely to occur after 50 years rather than 10 years. This timescale may be shorter if wood is artificially added to the stream.

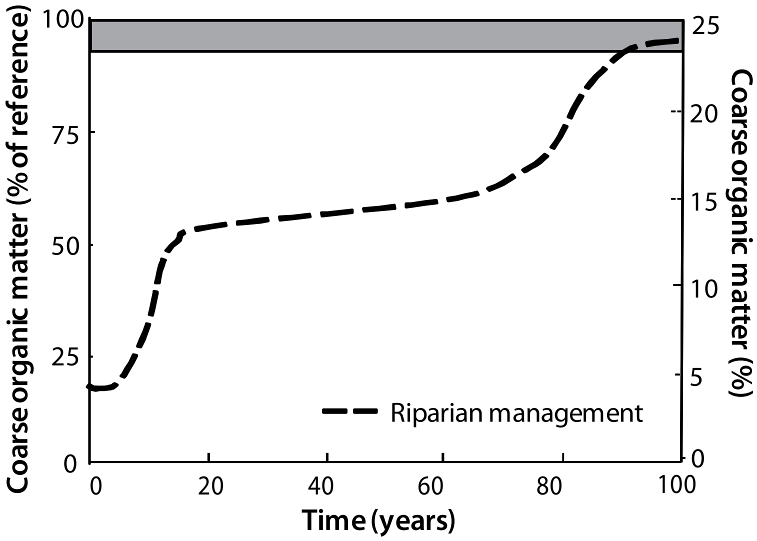


Figure 5.13: *Hypothetical recovery curve for coarse organic matter abundance in response to riparian restoration.*

Leaf litter retention

Goal(s): NH, EF

Background: The retentiveness for particulate matter (e.g., sticks, leaves, and finer particles) is a key functional attribute of streams. Retention influences:

- the available food resources for biota within a reach (and downstream)
- habitat complexity and refuges (leaf and wood packs)
- local and downstream nutrient and sediment retention, invertebrate community structure, and secondary production.

The goal is to restore *natural* levels of retention for a given stream size, rather than creating *high* levels of retention, because retention influences up–downstream linkages along river continua. For example, restoration may involve reducing retention in a headwater stream choked by macrophytes that trap all litter inputs and deprive downstream reaches of natural litter inputs. On the other hand, in an artificially straightened channel with low retention (James & Henderson 2005), the goal would be to increase retention to a natural level.

Stream habitat restoration can enhance retention by:

- restoring wider channels under forest lighting (Davies-Colley 1997) and hence reducing average depth, resulting in more opportunity for leaves/organic particles to settle or become trapped on projecting bedforms (Quinn et al. 2007)
- increasing in-stream wood debris dams that increase retention (Quinn et al. 2007)
- increasing encroaching riparian vegetation that interacts with the flow and acts as a filter
- increasing geomorphic variation resulting in backwaters and slow flow areas where leaves can settle out.

Stream habitat restoration can reduce retention by:

- reducing in-stream vegetation by shading out macrophytes or filamentous green algae, which can contribute to retention in some pasture streams (Quinn et al. 2007)
- restoring higher flows in a stream affected by abstraction (James & Henderson 2005)
- removing willows than encroach within stream channels (James & Henderson 2005).

Method: The method follows that of Quinn et al. (2007). It involves measuring the geometric mean travel distance (S_p = antilog of mean of log transformed distances to the points where leaves are trapped by the bed, vegetation, etc.) of 30 (depending on stream size) waterproof paper (plastic) triangles (4.4 cm sides) that act as freshly fallen leaf analogues when released under base flow conditions. These “leaves” should be released at equidistant points across the whole wetted width of the stream (Figure 5.14). At least two releases should be done from different standardised points (for repeat measurements over time, e.g., at the top and half way along the study reach). Retention distances should be measured in reference (minimally disturbed) stream reaches of similar catchment area and slope to the restored reach. Differences in catchment area between reference and restoration reaches can be accounted for by comparing catchment area specific S_p values (CS_p) (Quinn et al. 2007).

Figure 5.14: Release of waterproof paper triangles.
Photo: Richard Storey

When: Every second year at base flow conditions.

Timescales and measures of success: As discussed above, the restoration success measure should be the return to a natural retention rate. This would involve an increase in S_p where retention distance is unnaturally low (e.g., from willows or macrophytes choking a channel or from flow abstraction) but would involve a decrease in S_p where a straightened channel is "remeandered", or a pasture channel becomes wider and shallower after riparian afforestation, or widens and increases in roughness after introduction of engineered large wood jams or boulder substrates that protrude above the water surface.

Hard engineering interventions, such as willow removal, channel reconstruction (meandering) and engineered log jams, will have immediate effects, whereas soft engineering actions, such as riparian planting, will act at longer timescales (Figure 5.15). Retention by encroaching riparian vegetation (e.g., bankside sedges or shrubs with roots and branches extending into the channel) is likely to occur within 3–5 years of riparian fencing and replanting along streams with bank heights close to the water level. Effects of natural inputs of large wood derived from riparian forest regrowth are expected to begin at around 70 years after planting and increase over the next few centuries (Meleason & Hall 2005, Davies-Colley et al. 2009). Responses of S_p to restoration actions are predicted to be most rapid and marked in small streams that are naturally more retentive than larger streams (Quinn et al. 2007).

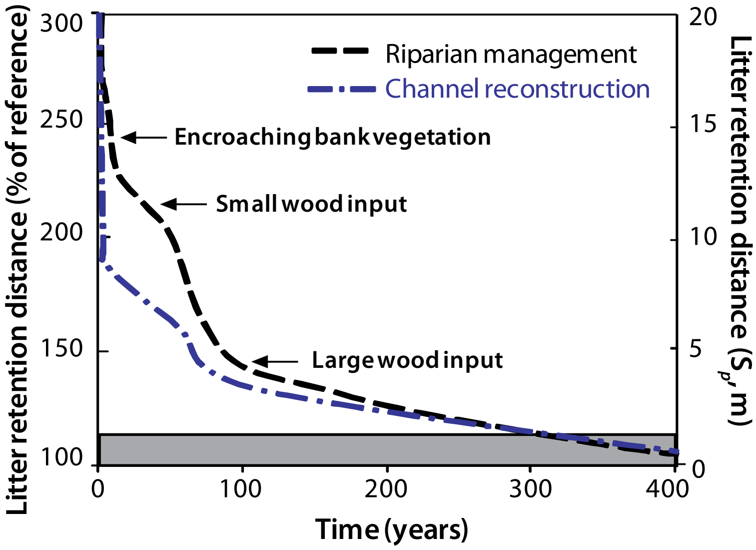


Figure 5.15: *Hypothetical retention distance of leaf litter inputs after riparian management and channel reconstruction, based on differences between S_p of pasture and forest stream reaches in straightened and natural channels (James & Henderson 2005, Quinn et al. 2007).*

Rubbish

Goal(s): NH, WQ, A, R

Background: Human-derived rubbish, either organic or inorganic, can be a major concern for the public and its removal a key measure of success in restoration for some projects (e.g., Gill 2005). While inorganic rubbish may provide substrate for in-stream organisms to grow on, it is generally unsightly, could break down and release chemical pollutants, and is indicative of an unnatural environment. Organic rubbish, such as garden waste, can spread noxious weeds through the restored area. Rubbish dumping may even increase after fencing and planting of a restored stream, particularly if the riparian area is regarded by some as "messy" and the vegetation can disguise the rubbish more effectively (Nassauer 1995, Parkyn & Quinn 2006).

Method: We propose a simple quantitative method of rubbish assessment (Table 5.3) where each piece of significant rubbish (large enough to be seen easily when walking alongside the stream) is counted over the length of a study reach (in-stream and within riparian buffer zone). Each piece of rubbish should be classified as organic (e.g., pile of garden clippings), inorganic (e.g., plastic bags, washing machines), or chemical (e.g., paint tins or other objects that contained chemical pollutants, presence of oil slicks).

Table 5.3: *Hypothetical rubbish count at a restored stream.*

	Organic	Inorganic	Chemical	Total
In-stream (sighted within the stream channel)		10	1	11
Riparian buffer zone (10 m either side of stream)	1	4		5
Total	1	14	1	16

The total count of rubbish per category could be tracked over time at the restored site and a measure of success would be when the amount had declined to virtually no rubbish. Alternatively, the total count could be expressed relative to a reference stream.

When: Annually or every second year.

Application of method: Particularly relevant for urban streams or pasture streams used as dumping areas, but should form part of a pre-assessment for all restoration projects.

Timescales and measures of success: If upstream sources of rubbish are not able to be controlled, then these may wash down into the restored site regardless of activities onsite. Therefore, it is useful to monitor the presence of rubbish in both the stream and riparian areas. If the riparian rubbish count is declining but in-stream rubbish remains high, then this could indicate that local rubbish dumping is reduced but further work is needed upstream. Of particular concern is any evidence of chemical pollution or items that contained chemical pollutants. A reasonable measure of success would be the absence of these types of rubbish. We expect that rubbish dumping may increase during the early stages of riparian plant establishment, when the diversity and proliferation of ground cover plants is high and the area may

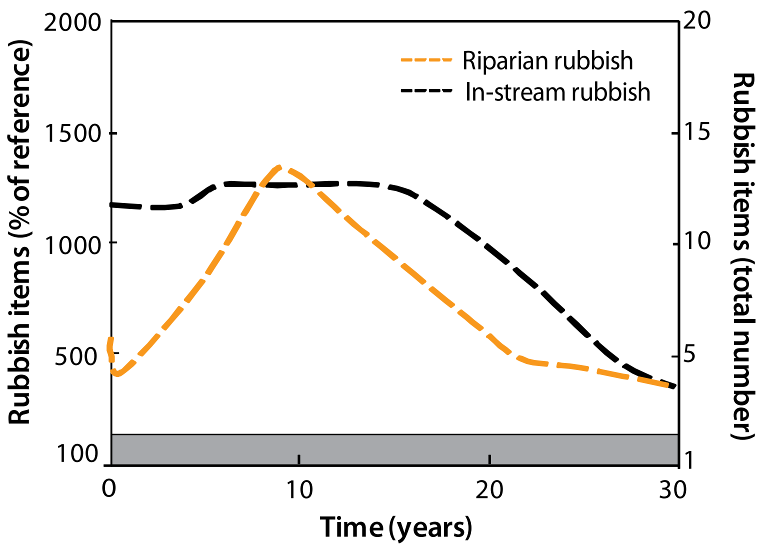


Figure 5.16: *Hypothetical trajectory of total rubbish input relative to a reference site (with 1 rather than 0 rubbish items) in the riparian area and in the stream channel of an urban stream.*

be considered "untidy" to some. Weed control and adequate signage declaring the site to be a restored area may alleviate these problems.

Hypothetical timescales of total rubbish counts for in-stream and riparian areas are shown below (Figure 5.16). These assume that the restoration is of an urban stream where upstream sources of litter could still be present resulting in a lag in improvement for in-stream rubbish counts behind that of terrestrial litter. High variation is likely, given the sporadic nature of litter inputs, differences in retention times in the environment, and variability of cleanup efforts by stream care groups. Nevertheless, we might expect initial improvements particularly of rubbish in the riparian area immediately after planting, but as vegetation begins to grow more "wild" there is the potential for additional rubbish dumping. The steady decline after vegetation becomes established assumes that there is further education and restoration work upstream in the catchment as well as better appreciation of the area by the public.

Shade of water surface

Goal(s): NH, AB

Background: Shade plays an important role in the regulation of stream light and temperature, with profound effects on in-stream plant growth, ecosystem metabolism, and the relative suitability of the habitat for differing biota. Shade provides an indication of the level of natural organic litter input by vegetation over the stream (Scarsbrook et al. 2001). Shade varies naturally along river systems as stream size increases, which opens a canopy gap between the riparian vegetation on each bank (Davies-Colley & Quinn 1998, Vannote et al. 1980).

Method: Quantitative shade assessments should be made using either a spherical densiometer (see SHAP Figure 20, page 49), or paired light meters or canopy analysers. When light or canopy meters are used, measure simultaneously at points over the stream and at a nearby unshaded location (e.g., hilltop) and calculate the average shade by difference (Davies-Colley & Payne 1998). We recommend measurements at 20 randomly selected points across the full width of the water surface, e.g., by randomly assigning positions across the stream (10, 20, 80% of stream width, etc.) at regular transects along the reach, so that the influence of stream banks, stream bank vegetation, and hill slopes is included.

Qualitative assessments are less reliable but may be made using photographs of reaches where shade has been quantitatively measured (Figure 19 in SHAP) as a guide.

When: Annually or every second year.

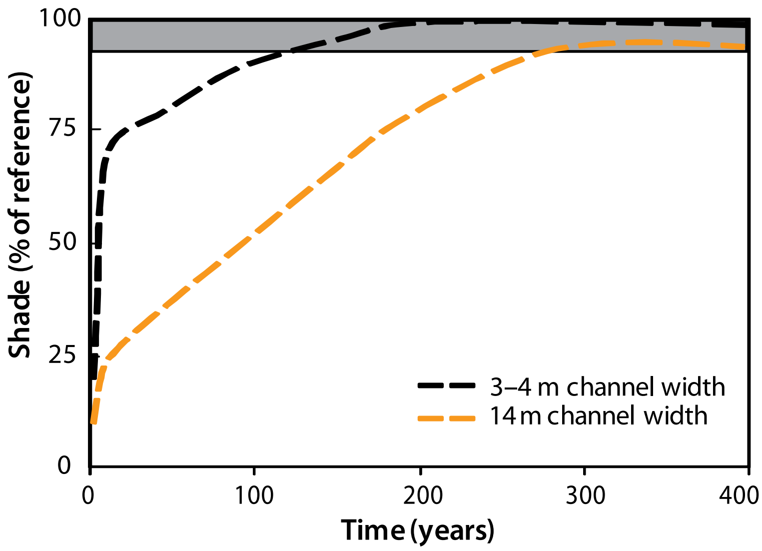


Figure 5.17: *Predicted timeframe and success of riparian management along pasture streams with 3.3 m and 14 m wide channels (based on Davies-Colley et al. (2009)).*

Timescales and measures of success: The timescale for restoration of shade and the shading goal vary with stream channel width and the type of riparian vegetation (Davies-Colley et al. 2009, Davies-Colley & Quinn 1998). Shade develops rapidly over small streams with actively planted riparian areas (Quinn et al. 2009), whereas shade develops slowly over wide streams and/or without active planting (Davies-Colley et al. 2009). Riparian reforestation of a 3–4 m wide stream is expected to result in rapid restoration of shade (Figure 5.17), whereas shade development will be much slower for a 14 m wide channel. Please note the absolute level of shade would be lower in the wide channel than the narrow channel, but the reference situation for a wide stream would also have a lower level of shade. The channel reconstruction scenario is not expected to have significant additional effects on shade development.

Riparian microclimate

Goal(s): NH, AB, TB

Background: Restoring riparian forest benefits both stream and terrestrial biodiversity by creating habitat conditions favourable to indigenous species. Microclimate is a key aspect of riparian habitat that affects ecosystem processes and the success of both terrestrial biota (Murcia 1995) and the adult (aerial, reproductive) phase of stream insects (e.g., caddisflies and stoneflies) that spend time in the riparian area and are sensitive to hot/dry conditions (Collier & Scarsbrook 2000; Collier & Smith 1998, Collier & Smith 2000, Smith & Collier 2005). Microclimate can refer to a range of meteorological variables, including air and soil temperature and wind strength and direction (Davies-Colley et al. 2000). We suggest using air temperature and relative humidity as key indicators of riparian microclimate that are simple to monitor continuously using relatively inexpensive data loggers.

Method: Measure air temperature and relative humidity continuously (30-minute intervals) using a data logger mounted at 1.8 m above ground level in a slotted conical cover to provide shade with good air movement (Figure 5.18). Mount the logger on a stake in the middle of the riparian buffer (e.g., at 5 m from the outer edges in a 10 m wide buffer). A variety of loggers that measure both temperature and humidity are available. The conical covers are available from NIWA Instruments. Data loggers can store almost 3 years of data recordings made at 30-minute intervals, but we recommend

Figure 5.18: Microclimate data loggers mounted in a slotted conical cover to provide shade and relatively unimpeded air flow.
Photo: John Quinn

downloading the data every 1–2 months to avoid loss of all data in the event of instrument loss or failure.

A second set of simultaneous measurements should be made at a reference site (long established forest of at least the dimensions of the riparian buffer) and possibly also at a control site (a nearby open canopy site) to allow progress towards the restored state to be determined by difference between the restoration site and reference/control, without need to account for variations in climate. If it is not practicable to run a reference station, information from a single riparian site can be used to measure long-term trends, but the data will have more "noise" due to the influence of natural variations in climate.

Trends in temperature and relative humidity can be measured using the daily mean, minimum, and maximum temperature. Where paired reference and riparian site measures are made, the differences in these values between the sites is used to monitor change.

When: Deploy loggers all year round and download every 1–2 months. If it is impractical to maintain the climate site continuously, then measurements over several weeks around the annual extremes (mid-summer and mid-winter) will still provide a useful means of tracking long-term trends.

Timescales and measures of success: The trajectory of microclimate recovery will be controlled mainly by the growth rate of riparian vegetation. This is, in turn, determined by the species planted and environmental conditions that control their growth rate, such as soil nutrients, rainfall, general climate, and disturbance (e.g., by grazing animals and floods). The trajectories in Figure 5.19 are for a climate station in the middle of a 10-m wide riparian buffer with a mix of native shrub and tree seedlings (0.2–0.5 m high) planted at 2–3 m density. The height of these plantings is expected to be above the data logger mounting level (1.8 m) about 5 years after planting. The endpoints for daily maximum and minimum air temperature are based on observations at the mid-point of a 30 m wide, established (15 m tall) native forest riparian buffer on the Coromandel Peninsula, North Island (Meleason & Quinn 2004). Relative humidity is mainly a function of air temperature and consequently it will follow a very similar trajectory to that of air temperature (Figure 5.19).

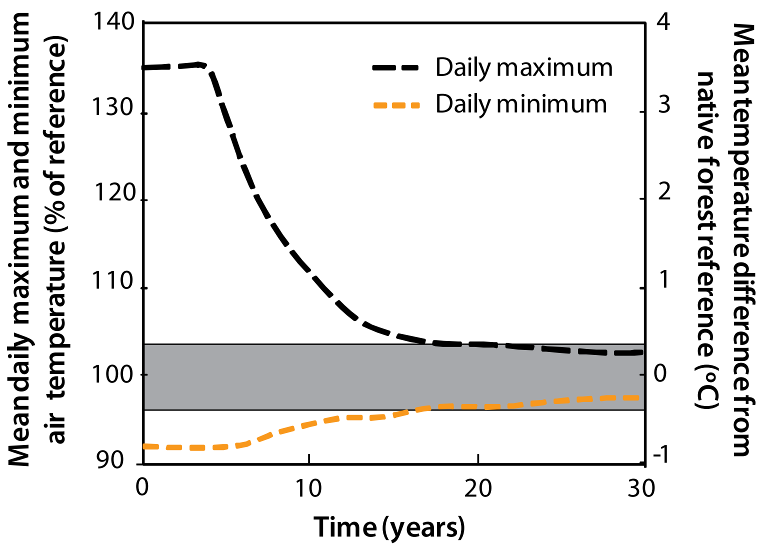


Figure 5.19: *Hypothetical annual average daily maximum and daily minimum air temperature after riparian management.*

Biogeochemistry and Water Quality

Water temperature

Goal(s): WQ, NH, AB

Background: Water temperature has a fundamental influence on biogeochemical reactions in aquatic environments, with rates of reactions increasing strongly with increasing temperature. Water temperature also influences the equilibrium point of competing reactions in water; for example, increasing water temperature decreases equilibrium dissolved oxygen (DO) solubility in water (Hauer & Hill 1996). High water temperatures, occurring in plumes of hot wastewater, or during the afternoon of hot days in mid-summer in streams lacking riparian shade, can be stressful to aquatic animals, including native fish (Richardson et al. 1994) and invertebrates (Quinn et al. 1994). Stream restoration projects are often concerned with restoring riparian shade to streams to reduce high temperature excursions, as well as to reduce aquatic plant growth (Rutherford et al. 1999).

Cox & Rutherford (2000a, b) suggest that temperature criteria determined for constant water temperatures, such as those of Quinn et al. (1994) for New Zealand aquatic invertebrate animals, may be interpreted for diurnally varying temperatures mid-way between the daily mean and daily maximum temperature. Therefore, recording diurnal temperature variation is most useful in restoration studies – to estimate the mean of the daily mean and daily maximum (the "Cox-Rutherford index") for comparison with thermal stress criteria.

Method:

SPOT TEMPERATURE READINGS

Because stream water temperature is strongly diurnally variable (Figure 5.20), reflecting the diurnal pattern of heating from sunlight, temperature usually reaches a maximum in the mid-to-late afternoon. A spot temperature reading of the maximum temperature on a clear sunny day is therefore most useful, and a value close to the maximum can usually be recorded in the mid-afternoon. The only reason for measuring spot water temperature at other times would normally be to interpret simultaneous DO measurements (refer DO protocol).

Spot temperature readings are simple to make with thermometers or thermisters (temperature-sensitive electronic resistors) (Hauer & Hill 1996). Alcohol (rather than mercury) thermometers are most suitable for field use. To avoid localised high-temperature bias, take the temperature reading in the shade and in flowing water.

The Cox-Rutherford index can be estimated from a single mid-afternoon mid-summer point reading of water temperature on a sunny, clear day, combined with the daily mean water temperature. Ideally, water temperature should be monitored continuously so as to calculate the daily mean water temperature (see the next sub-section on continuous temperature logging) and to establish the daily maximum.

TEMPERATURE LOGGER DEPLOYMENT

Small temperature sensors with integral loggers (e.g., Onset Stowaway, Onset TidbiT) are extremely useful and convenient for stream temperature studies. These have sufficient memory to be left in place in a study stream for several months, logging temperature every 15 or 30 minutes through the warmest days of summer, so as to define thermal regime during the season when thermal stress is most likely. For example, when monitoring restoration sites, Environment Waikato (EW) deploys stream water temperature loggers attached to steel stakes below residual pool depth at the downstream end of riparian restoration reaches and compares results with those installed at nearby forested reference sites (installation is in December; retrieval the following April).

Few protocols have been published for interpreting temperature logger data. However, in our opinion, daily maxima, daily amplitude (the difference between daily maxima and minima), and the Cox-Rutherford index (mean of daily mean and daily maximum temperature, Cox & Rutherford 2000a, b) are most useful. Daily maxima indicate the most extreme conditions encountered and are useful for interpreting critical thermal stress, whereas the Cox-Rutherford index can be compared directly with constant temperature criteria for thermal stress on stream animals determined under laboratory conditions.

Figure 5.20 shows typical water temperature data for a poorly shaded pasture stream and a shaded reference stream on a clear day in mid-summer. The Cox-Rutherford index, calculated for a period of low flow in late summer (February), is about 23 °C. Not surprisingly, this stream lacked stoneflies and other temperature-sensitive stream insects, for which constant temperature criteria may be as low as 19 °C.

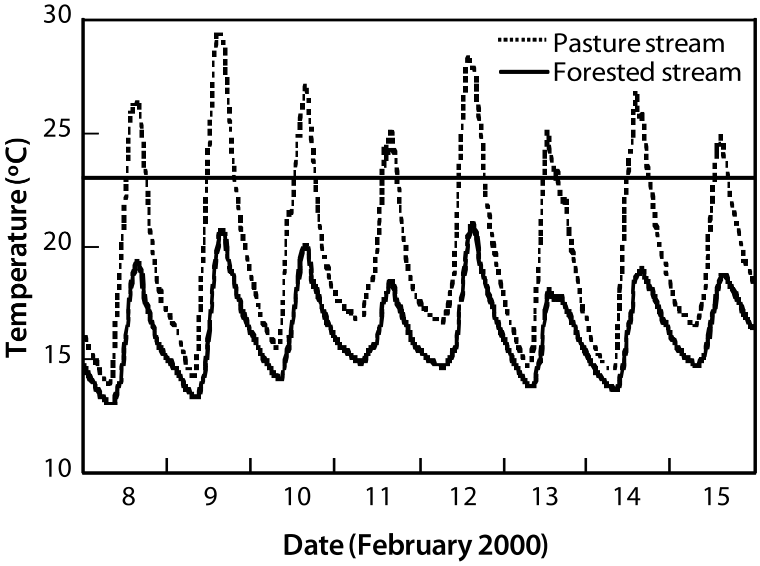


Figure 5.20: *Stream water temperature recorded in a poorly shaded pasture stream and nearby forested reference stream in the Gisborne District (RDC unpublished data). The Cox-Rutherford index calculated during summer (February) low flow conditions is shown as a bar at 23°C. This temperature index can be compared with constant temperature criteria for thermal tolerance.*

When: Temperature monitoring should be part of any water quality monitoring programme at restoration sites. Spot measurements should be done monthly at the same time as water quality monitoring, with mid-afternoon measurements required to record daily maxima. Ideally, temperatures should be monitored continuously at intervals of an hour or less through at least one month of mid-to-late summer (target month February).

Timescales and measures of success: Timescales of recovery of thermal regimes are strongly dependent on development and extent of riparian shade, which, in turn, depends crucially on tree growth (canopy height) in relation to stream size (channel width) (Davies-Colley et al. 2009). Small (narrow) streams (in which thermal stress is most severe because such streams are also shallow) will recover much more quickly than large streams or rivers, because shading is greatest when channel width is small. For instance, Quinn & Wright-Stow (2008) found that recovery rate of stream thermal regimes after plantation forest harvest and replanting (both daily mean and daily maximum temperature recovery rates measured as °C/year), were a strong decreasing function of stream size (channel width range: 2–12 m) in streams on the Coromandel Peninsula.

A valuable index of thermal recovery (and hence restoration success) is when late-summer, low-flow stream water temperatures fall below laboratory derived thresholds for thermal stress at constant temperature – often around 21°C for a range of species. Davies-Colley et al. (2009) used simulations of shade and water temperature to predict recovery trajectories for the thermal regime of streams in the Pureora Forest Park, central North Island. The Cox-Rutherford thermal index dropped below a threshold of 21°C at the following times after hypothetical native tree planting along all upstream channels (channel widths in parentheses): 4 years (1.3 m), 5 years (1.6 m), 12 years (3.3 m), 110 years (6.6 m), 240 years (14 m). The short time frames of thermal recovery of the three small streams reflect rapid shading by planted native pioneer species, while the slow thermal recovery of the two large (6.6-m and 14-m wide) streams reflects growth of tall native trees that would eventually take over from the plantings.

Figure 5.21 shows the simulated thermal recovery (Cox-Rutherford index) of a 3.3-m wide stream in the central North Island originally in pasture after planting of the riparian zone (all upstream length) with common pioneer native trees as described by Davies-Colley et al. (2009). Recovery of shade, and therefore temperature regime, is expected to be very dependent on stream size; very small streams (of 2 m or less channel width) would recover rapidly (within a few years) while streams of 6 m width or wider may take more than a century to shade sufficiently.

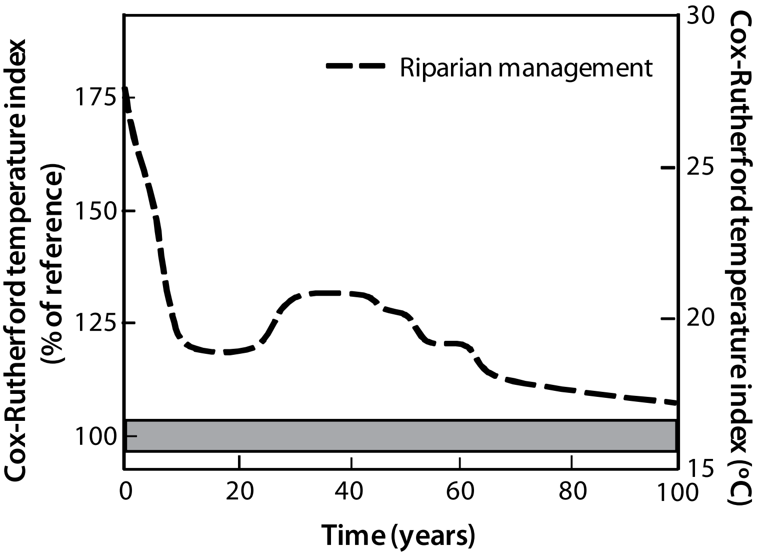


Figure 5.21: *Recovery of thermal regime of a 3.3 m wide stream in the central North Island, planted with native trees that progressively re-shade the channel (model predictions from Davies-Colley et al. 2009). The Cox-Rutherford temperature index value of a reference stream (about 16°C) is not approached until the climax forest grows to its full height at about 200 years (not shown).*

Dissolved oxygen

Goal(s): WQ, AB, EF, F

Background: Dissolved oxygen (DO) is vital for life in freshwater environments. A limitation or over abundance of DO may indicate anoxic (too much respiration) or eutrophic (too much productivity) conditions that can be harmful for stream life and the healthy functioning of streams. Furthermore, low DO concentrations have the potential to kill fish and other aquatic life (Dean & Richardson 1999). As such, DO concentrations can provide a good measure of restoration success when the aim is to enhance the life-supporting capacity of a stream experiencing low minimum DO levels.

Method: Place a dissolved oxygen (DO) probe in gently flowing water – at the downstream end of a pool is best. Ensure the DO meter is calibrated before use according to manufacturer's instructions. Allow the value to stabilise and record DO percent saturation and water temperature.

When: A datasonde that logs DO continuously can be deployed in-stream, or two spot DO readings can be made using a hand-held DO logger. A reading as close to dawn as possible provides a measure of the minimum DO saturation in-stream, and a reading in the late afternoon provides a measure of the maximum DO concentration in-stream.

Timescales and measures of success: Daily DO minima, maxima, and range are important metrics for assessing the availability of dissolved oxygen in-stream. There are numerous factors that affect DO dynamics and any attempt to restore DO values will be subject to multiple response pathways. For example, excessive vegetation in-stream can lead to a wide daily range in DO values; hence, removing invasive vegetation may be one means of restoring natural DO dynamics. Excessive nutrients in-stream can lead to elevated biological oxygen demand and associated low DO minima; hence a reduction in nutrient delivery can improve DO minima values. Altered channel form can affect the residence time of water in-stream in as well as reduce or increase shallow water habitats where greater reaeration of water occurs. Thus, channel reconstruction can increase or decrease DO minima and maxima.

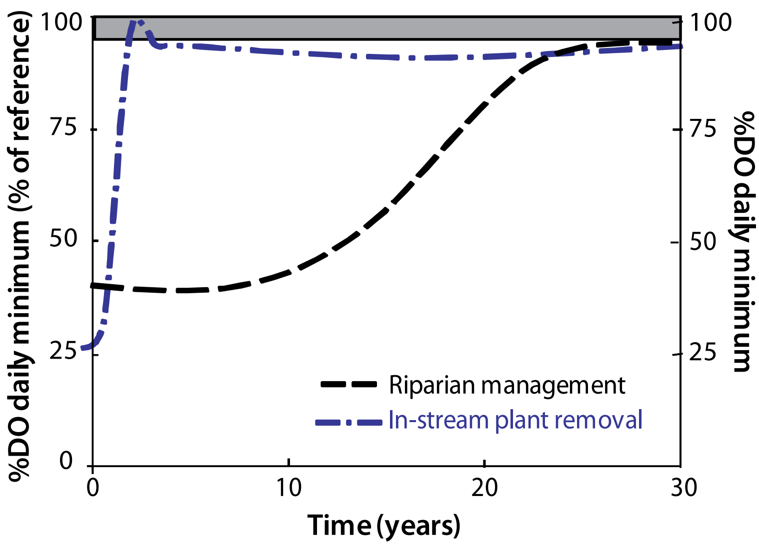


Figure 5.22: *Conceptual recovery curves for dissolved oxygen daily minimum in response to riparian restoration and invasive vegetation removal. The farm stream is assumed to have significant macrophyte growth under both management scenarios.*

In general, an improvement in DO dynamics can be expected to occur in response to riparian replanting from the associated reduction in water temperature and/or nutrient delivery in-stream. Channel reconstruction can also improve DO dynamics by introducing a hydraulic drop to increase turbulence and hyporheic exchange or by the physical removal of excessive aquatic vegetation.

Dissolved oxygen minima are expected to recover in response to reduced biological oxygen demand due to reduced in-stream productivity associated with lower temperature and light availability (Figure 5.22). However, if deciduous trees are used in riparian planting, then this curve will be suppressed by the additional biological oxygen demand of falling leaf litter. A similarly shaped recovery curve would be expected in response to invasive vegetation removal, although recovery would be much quicker (Figure 5.22).

Ecosystem metabolism

Goal(s): EF

Background: Ecosystem metabolism is a measure of how much organic carbon is produced and consumed in river ecosystems. The balance between organic carbon production (primary productivity) and consumption (ecosystem respiration) provides information on the relative importance of the two key sources of energy that fuel production of higher life-forms in river ecosystems – algae or terrestrial organic matter. 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 organic matter from upstream or the surrounding catchment is being used to maintain the system. Therefore, ecosystem metabolism provides a direct measurement of the food base of river ecosystems and is a good indicator of river ecosystem health (Young et al. 2008). Ecosystem metabolism is a direct measure of ecosystem services such as primary production and carbon cycling.

Ecosystem metabolism responds to a wide variety of factors including light intensity, water temperature, nutrient concentrations, organic pollution, chemical contaminants, flow fluctuations, and shading by riparian vegetation. As these factors change with restoration activities, ecosystem metabolism can be a sensitive indicator of ecosystem change for assessing river restoration.

The ecosystem metabolism of a river can be estimated by measuring the daily changes in dissolved oxygen concentration. Oxygen is constantly being removed from the water by respiration, however, during the day algae and aquatic plants photosynthesise and release oxygen back 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.

Method: To measure **gross primary productivity (GPP)** and **ecosystem respiration (ER)** we recommend the use of the single station open system metabolism protocol below based on diurnal measurements at the downstream end of the restored area as outlined in Young et al. (2006).

EQUIPMENT: Dissolved oxygen logger/sonde, warratah and/or chains or rope to secure sonde, ruled rod or similar, light logger (optional).

1. Ensure that the batteries in the sonde have sufficient power and replace the membrane on dissolved oxygen probe (if using electrochemical probe). The oxygen probe needs to be carefully calibrated according to the manufacturer's instructions. The probe should be calibrated at similar temperatures to those it will experience during the deployment. Altitude also needs to be considered, so either calibrate all probes at one altitude and correct the data later for differences in site altitude, or calibrate at each site.

2. Program the sonde to measure at regular intervals (between 5 and 15 minutes) for at least 24 hours and preferably for a few days in case anything unusual occurs during one day.

3. Before deployment, run the probe in water-saturated air at the sampling site long enough to allow at least 3–4 recordings. Make sure that the air temperature is similar to ambient water temperature. Any difference between the measured value and 100% saturation can be used to correct the data later.

4. Deploy the sonde in a location as close as possible to the thalweg (deepest and usually central part of the flow) and preferably out of sight to reduce the likelihood of vandalism. Use a chain or strong rope to secure the instrument to the bank or other suitable solid substrates (e.g., tree, bridge pile).

5. Consider measuring light intensity throughout the deployment periods. Light data are helpful (but not essential) for accurately determining the timing of dusk, which is required for the metabolism calculations.

6. Estimate the average depth of each site by taking at least 5 measurements of depth at each of 5 cross-sections spaced out at regular intervals upstream of the sonde sufficient to cover local variation in channel morphology.

7. After at least 24 hours remove the sonde. Run the sonde again at the site in water-saturated air for a period sufficient to get 3–4 readings. Again, any differences between measured values and 100% saturation can be used to correct the data.

8. Download the data and check them. Discard any data that are affected by significant rainfall. Use data smoothing techniques (e.g., running mean) to remove any instrument "noise" that is present in the data. Check the data collected in water saturated air just before and just after deployment and correct the data if necessary.

9. Calculate metabolism values using the RiverMetabolismEstimator spreadsheet model (available from **www.cawthron.org.nz/coastal-freshwater-resources/downloads.html**). It works best to have an uninterrupted night of data (i.e., use 24-hour data sets starting around midday, rather than data sets starting during the night).

WHERE*:* The sonde should be deployed at the downstream end of the stream reach being studied.

Limitations: This method relies on the in-stream plant community being uniform for a distance upstream of the sensor equivalent to $3U/k_2$ (U = mean velocity (m d^{-1}) and k_2 = reaeration rate (d^{-1})) (Chapra & Di Toro 1991). If the restored reach is shorter or contains point source discharges that cause abrupt changes in benthic metabolism, use two-station diel DO curve analysis methods (Odum 1956) (e.g., at the upstream and downstream ends of the restored reach) to measure respiration and photosynthesis.

When: Ecosystem metabolism responds to seasonal changes in temperature, light, nutrient availability, and discharge. Therefore, to ensure measurements of ecosystem metabolism are comparable between sites and over time, sampling should occur on fine days at base flow, and at the same time each year, preferably at the end of summer when worst-case conditions are most likely. If multiple measures are possible each year, then at the end of each season is preferable.

Timescales and measures of success: The trajectory of recovery will vary for ecosystem metabolism depending on the nature of the impairment and the nature of the restoration effort. Different stream types are also likely to have different recovery curves due to variability in natural reference states as well as the potential of some restoration efforts to be more successful in some streams compared to others. For example, riparian restoration in a small agricultural stream is likely to be successful at reducing elevated ecosystem metabolism to rates indicative of healthy conditions due to canopy closure. However, in a large river, shading due to riparian restoration is unlikely to be sufficient to fully reduce in-stream metabolism. Similarly, ecosystem respiration may be suppressed due to a contaminant or pollutant in a system or elevated due to excess nutrients in another. Thus, the starting point for recovery will differ in these streams. These examples of variability are important to consider when using ecosystem metabolism as a measure of restoration success. The easiest way to account for this variability is the comparison to relevant reference streams.

Furthermore, it is likely that recovery curves will vary for ER and GPP as they are influenced by different environmental drivers (i.e., light and temperature for GPP versus temperature and nutrients for ER).

As such, these measures are considered separately, and two scenarios where measures of ecosystem metabolism are most likely to provide indicators of restoration success are discussed below.

GROSS PRIMARY PRODUCTIVITY

The recovery of GPP can be assessed as "healthy", "satisfactory", or "poor" by comparison to reference condition as outlined in Young et al. (2006), based on the ratio of GPP at a test site to GPP at a reference site (Table 5.4). In the absence of a reference site, absolute values may be used, but this is a less sensitive assessment than comparison to an appropriate reference site (Table 5.4).

Theoretically, the difference between the GPP values of reference streams and treatment streams will converge with similar levels of riparian shading, secondarily influenced by temperature, bed substrate stability (and/or mesohabitat abundance), and nutrient availability (McTammany et al. 2007). Light has been shown to limit GPP when riparian shading is greater than 70% (Bunn et al. 1999, Rutherford et al. 1999). Therefore, the primary restoration approach to reducing GPP is increasing riparian shading (Figure 5.23).

Channel reconstruction or flow restoration (e.g., reinstating flushing flows in a dammed river) could reduce the standing stock of algae and subsequently reduce ecosystem metabolism quicker than shading alone (Figure 5.23). Resetting minimum flows and/or seasonal or annual high flows can maintain rates of ecosystem metabolism at levels indicative of healthy systems. Channel reconstruction to create pools could result in increased growth of algae in low flow areas and increased GPP before full shade develops.

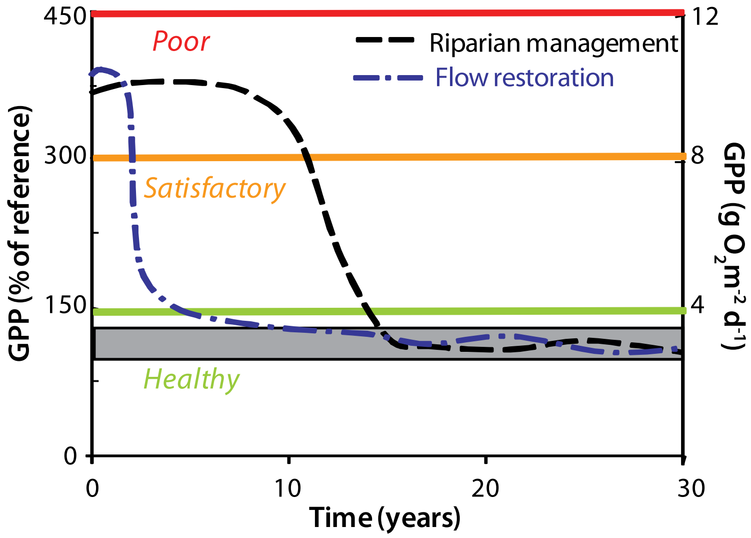


Figure 5.23: *Hypothetical recovery curves for GPP in response to riparian management and flow restoration. The colour bands represent "healthy" (green), "satisfactory" (orange), and "poor" (red) conditions as suggested by Young et al. (2008).*

ECOSYSTEM RESPIRATION

Ecosystem respiration can be either greater or less than reference values depending on the nature of impairment. Therefore, recovery of ER values toward reference can involve stimulating or suppressing ER by increasing healthy nutrient transformations (e.g., denitrification) or decreasing excess nutrient inputs and stream temperature. As with GPP,

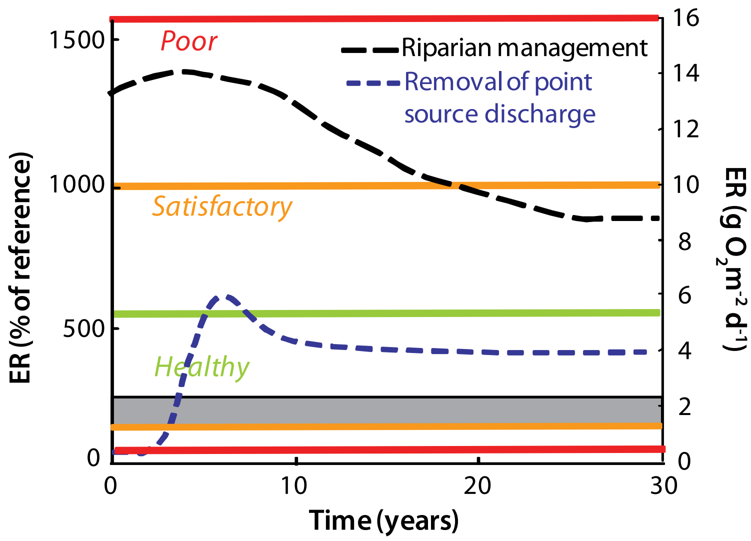


Figure 5.24: *Hypothetical recovery curves for ecosystem respiration (ER) in response to riparian management and the removal of a contaminant. The colour bands represent "healthy' (green), "satisfactory" (orange) and "poor" (red) conditions as suggested by Young et al. (2008).*

ER can be assessed in comparison to absolute values (derived from national reference data) or as the ratio of a local reference site (preferable) (Table 5.4).

Table 5.4: *Framework for assessing functional stream integrity using metabolism data. From Young et al. (2008).*

Method	Assessment parameter	Criterion	Assessment
Absolute value	GPP at test site (g O_2/m^2/day)	GPP <4.0	Healthy
		GPP = 4.0—8.0	Satisfactory
		GPP >8.0	Poor
	ER at test site (g O_2/m^2/day)	ER = 1.5—5.5	Healthy
		ER = 0.7—1.5 or 5.5—10.0	Satisfactory
		ER <0.7 or >10.0	Poor
	P/R at test site	P/R <1.3	Healthy
		P/R = 1.3—2.5	Satisfactory
		P/R >2.5	Poor
Comparison with reference	Ratio of GPP at test (GPPt) and reference (GPPr) sites	GPPt:GPPr = 0.4—1.5	Healthy
		GPPt:GPPr = 0.1—0.4 or 1.5—3.0	Satisfactory
		GPPt:GPPr <0.1 or >3.0	Poor
	Ratio of ER at test (ERt) and reference (ERr) sites	ERt:ERr = 0.4—1.4	Healthy
		ERt:ERr = 0.2—0.4 or 1.4—2.5	Satisfactory
		ERt:ERr <0.2 or >2.5	Poor

Decreasing the stream temperature and the reduction of algal/macrophyte biomass via riparian planting is one valuable restoration goal for improving ER. Riparian plantings can also decrease excess diffuse nutrient inputs into streams, further improving ER. A hypothetical recovery curve for ER based on decreased temperature/plant biomass as a function of increased riparian shading is given in Figure 5.24. If riparian planting is the only form of restoration, then recovery of ER may be inhibited from reaching "healthy" status if other pressures are in play, such as elevated nutrient status or high sediment loading. It is likely ER will decrease in part with shading but may stay elevated in comparison to reference due to such additional pressures.

Contaminants that can suppress ER include toxic substances, natural acidic waters, organic pollutants, and sediments. Whilst sediments are natural, excess sediments can block the connection between groundwater and surface water (i.e., the hyporheic zone) and lead to anoxic conditions. Sediment delivery can be reduced through riparian planting and good catchment management. Alternatively, channel manipulation or high release flows may be used to flush sediments from a stream. In that case, the shape of the recovery curve for ER would be similar to that observed for GPP, in respect to flow restoration (Figure 5.24). Most other contaminants enter the stream through point-source discharges and the only way to restore ER relevant to reference is by removing those contaminant pathways. A hypothetical recovery curve for ER based on reducing pollutant loadings is given in Figure 5.24. Following the reduction of a contaminant, natural respiration processes will be enhanced and may exhibit a rebound effect before stabilising at healthy rates.

Organic matter processing

Goal(s): EF

Background: Terrestrial organic matter provides a valuable energy source to streams. The rate that this matter breaks down depends on a combination of environmental and biological factors subject to natural variability and human influences, such as temperature, nutrient availability, pH, and sedimentation. However, one factor that most influences break down is the *type* of organic matter. Therefore, in terms of assessing stream restoration, it can be useful to standardise the type of organic matter used to measure breakdown rates. There are two standardised assays recommended for estimating organic matter processing in-stream: the cotton strip assay and the wooden stick assay (Figure 5.25). The breakdown of both materials can provide an indication of the health of in-stream components that contribute to organic matter processing. Measuring rates of organic matter breakdown provides a direct measure of carbon cycling.

Cotton strips are made up of cellulose, a major building block in leaves and woody debris. Measuring cellulose decomposition using a cotton strip assay specifically evaluates the contribution of microbial decomposition to organic matter processing. Cotton strips are more expensive than wooden sticks to analyse but they are deployed for only 7 days so there is less chance of sample loss.

Figure 5.25: Wooden sticks and cotton strips deployed in-stream
Photo: Joanne Clapcott

In comparison, the breakdown of wooden sticks is due to microbial decomposition, physical abrasion, and in some cases consumption by freshwater organisms. Therefore, the wooden stick assay provides a measure of how well a stream is processing energy and nutrients for localised food webs and to provide resources for downstream. Wooden sticks are relatively cheap and easy to analyse compared to cotton strips but are more prone to loss and increased variability in measurements because they are deployed for 3 months.

Method:

COTTON STRIP

Deploy cotton strips in riffle habitat as outlined below. If riffles are not present then use runs.

1. Use a standardised test material such as that available from EMPA (Switzerland) or Shirley burial fabric (UK). Supplies can be obtained from the Cawthron Institute. An alternative unbleached cotton fabric (e.g., artist canvas) could also be used, with a similar warp and weft, following validation with a standardised fabric.
2. Tie nylon string to the end of the cotton strip and secure at least 5 strips at each of your sampling sites using short warratahs, lengths of reinforcing rod (40–50 cm), or similar stakes. Drive the stakes down into the substrate, leaving only the top exposed so that flow conditions are not altered and the stakes do not catch passing debris. Use weights or rocks to keep the cotton strips submerged near the river bed.
3. If making comparisons among sites, try to standardise the types of habitat where the cotton strips are deployed. Riffles are preferable.
4. Use flagging tape, detailed drawings, or, preferably, site photos to indicate the exact location of the cotton strips after they have been deployed.
5. If possible, deploy a water temperature logger at each site so that the effects of any differences in temperature among sites can be compensated for.
6. Retrieve the cotton strips after 7 days.
7. Keep the cotton strips cool and freeze flat if there is any delay before processing.
8. Gently rinse any adhered material from the strips and dry at 60 °C for at least 24 hours in a draught oven.
9. Remove cotton threads from the side of cotton strips so that strips are exactly 100 threads wide and cut into three strips of equal length. Discard the length where the nylon string was attached.
10. Break the replicate lengths using a tensometer and record the breaking strain.
11. Break at least 10 non-deployed cotton strips as a procedural control (soaked in stream water and then treated as above (steps 7–10).
12. Report cotton decomposition potential in terms of % cotton tensile strength loss:
    % CTSL = 100 – breaking strain/control breaking strain * 100
    Or calculate the exponential decay coefficient:
    Cotton decay coefficient (k) = -LN (breaking strain/control breaking strain).
13. If temperature data were collected concurrently with strip deployment (recommended practice), degree days can be used to compensate for any differences in temperature among sites. Correct % CTSL or decay coefficients by dividing by the sum of average daily temperature for every day over the period when cotton strips were deployed.

WOODEN STICKS

We recommend the use of the wooden stick protocol outlined below from Young et al. (2006). Deploy wooden sticks in riffle habitat. If riffles are not present then use runs.

1. Use birch wood coffee stirrer sticks (114 x 10 x 2 mm, Figure 5.25), which can be purchased from most supermarkets.
2. Drill a hole at one end of each stick and weigh each stick to the nearest 0.1 mg to get the start weight.
3. Tie 5 sticks onto nylon string and include a label tag. Use a cut length of plastic drinking straw (c. 1 cm) between each stick to keep them separated. Expect any labels marked on the sticks themselves to disappear during deployment. Make a distinctive mark on at least 1 stick (prior to weighing) and use the order of the sticks around the nylon string to identify each stick after deployment.
4. Keep at least 5 sticks aside to determine the correction factor for the difference between the start weight and oven dry weight:
   Correction factor = oven dry weight/start weight.
5. Secure at least 3 sets of 5 sticks at each of your sampling sites using robust cord and short warratahs, lengths of reinforcing rod (40–50 cm), or similar stakes. Drive the stakes down into the substrate leaving only the top exposed so that flow conditions are not altered and that the stakes do not catch passing debris. Use weights or rocks to keep the sticks submerged near the river bed and not spinning in the current.
6. If making comparisons among sites, try to standardise the types of habitat where the sticks are deployed. Riffles are preferable.
7. Use flagging tape, detailed drawings, or, preferably, site photos to indicate the exact location of the sticks after they have been deployed. In-stream conditions can change considerably over 3 months making stick recovery difficult if their location is not accurately recorded.
8. If possible, deploy a water temperature logger at each site so that the effects of any differences in temperature among sites can be compensated for.
9. Retrieve the sticks after 3 months.
10. Keep the sticks cool after retrieval and freeze if there is a delay before processing.
11. Wash any loosely adhering material from the sticks and then dry in a 60°C drafted oven for at least 24 hours.
12. Weigh each stick to establish its end weight and correct this based on its estimated original oven dry weight (using the correction factor determined from the subset of sticks that were put aside):
    Oven dry weight = end weight * correction factor.
13. Report the mass loss in terms of either % mass loss:
    % mass loss = (start weight – oven dry weight)/oven dry weight * 100
    Or calculate the exponential decay coefficient:
    Stick decay coefficient (k) = -LN(oven dry weight/start weight)

14. If temperature data were collected concurrently with stick deployment (recommended practice), degree days can be used to compensate for any differences in temperature among sites. Correct % mass loss or decay coefficients by dividing by the sum of average daily temperature for every day sticks were deployed.

When: Sample annually at the same time each year, preferably at the end of summer when worst-case conditions are most likely and to avoid the possibility of high flows restricting stream access. If multiple measures are possible each year, then at the end of each season is preferable. Note: this method requires 2 site visits – day 1 when cotton strips/wooden sticks are deployed and again 7 days later when retrieved.

Timescales and measures of success: Cotton breakdown has been shown to be faster at higher temperatures (Tiegs et al. 2007) and at sites with elevated nutrients (Boulton & Quinn 2000), and inhibited by high pH (Hildrew et al. 1984). Breakdown is also affected by the abundance of invertebrate shredders in streams (Hildrew et al. 1984, Clapcott & Barmuta 2009). Similarly, wood decay increases in response to increasing temperature, nutrients, and physical abrasion as a function of flow (Petersen & Cummins 1974, Young 2006, McTammany et al. 2008). Some wetting and drying regimes can even accelerate breakdown by enhancing biofilm development (Ryder et al. 2006). Both cotton and wood breakdown are influenced by many environmental factors, but the most dominant are temperature and nutrient availability.

Restoration effort that reduces stream temperature and nutrient inputs are likely to improve both cotton and wood breakdown rates relative to a reference. A hypothetical recovery curve for organic matter processing in response to riparian planting is provided in Figure 5.26. It is expected that elevated organic matter breakdown rates in warmer pasture streams will decline following riparian restoration due to a reduction in nutrient input and a reduction in temperature due to shading.

Eventually, shredding invertebrate abundances may recover if organic matter retention is sufficient, leading to a subsequent increase in breakdown rates of wooden sticks that may differ from that of cotton strips. Organic matter processing can also be temporarily elevated by abrasion associated with high flow events and high nutrients from point-source inputs, so the cotton strips or wood sticks should be redeployed if an unusual event occurs during the monitoring period. Absolute values for wood and cotton breakdown are guided by previous studies (Young 2006, Young RG, Cawthron Institute, unpublished data), but it is recommended that wood breakdown be assessed by comparison to reference.

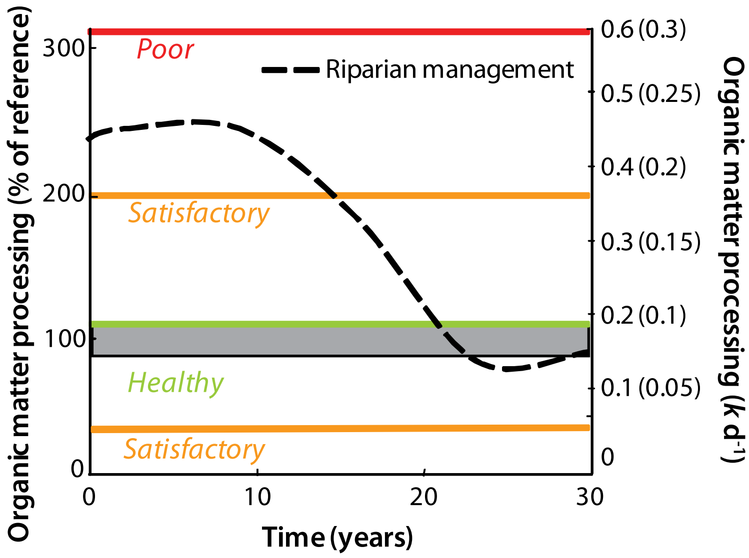


Figure 5.26: *Hypothetical recovery curve for organic matter processing in response to riparian management. Y-axis values for cotton k (with wood k in parentheses). The colour bands represent "healthy" (green), "satisfactory" (orange), and "poor" (red) conditions as suggested by Young et al. (2008).*

Nutrients

Goal(s): WQ, DH

Background: Nutrients are used by aquatic plants, periphyton, and microbes (bacteria, fungi) for growth. The two primary nutrients are nitrogen (N) and phosphorus (P), which occur both as dissolved ionic forms (NO_3^-, NH_4^+ and PO_4^{3-}) and bound in dissolved or particulate organic molecules (e.g., amino acids). The dissolved ionic forms are most biologically available and readily assimilated by stream plants. In pristine stream waters, nutrient concentrations are normally low and at growth-limiting levels (Steinman & Mulholland 2006, Tank et al. 2006). Nutrients enter streams naturally in rainwater, from soil and rock weathering and from biological processes (e.g., nitrogen fixation by cyanobacteria). Nutrients also enter streams as a result of human activities (e.g., fertiliser leachate/runoff, wastewater and effluent discharges). These inputs can be point sources (e.g., pipes, drains) or diffuse sources (e.g., groundwater seepage, runoff). Nutrients entering streams as a result of human activities are often at high concentrations and can stimulate nuisance growths of aquatic plants, periphyton, and microbes.

Stream restoration activities that fence stock out of streams and/or establish planted riparian buffer zones are expected to reduce nutrient inputs to streams, particularly where this restoration effort is applied to a large proportion of the catchment. Keeping stock out of streams prevents direct input of nutrients in manure and urine into the stream. Riparian buffer zones or filter strips can trap and filter nutrients in overland runoff and groundwater. Riparian vegetation also helps to stabilise eroding banks that can be a major source of particulate P.

Method: Collect water samples for nutrient analysis from the centre of the stream channel during base flow (i.e., during a period of stable river flow with minimal rainfall). Store the water sample chilled (e.g., in an ice-filled chilli bin) during transport to the laboratory. Samples should be processed within 24 hours of collection or frozen to be analysed later. Follow the directions of your analytical laboratory regarding the choice of bottle and specific methods for filling the bottle as these can differ depending on the analysis required.

Nutrient concentrations in stream water can be measured and monitored over time to detect changes in the degree of stream nutrient enrichment and the relative availability of N and P (which may indicate greater sensitivity to one nutrient) in response to stream restoration efforts. A molar N:P ratio >32 indicates likely P deficiency (and thus greater sensitivity to P than N inputs) in freshwater benthic algae (Kahlert 1998).

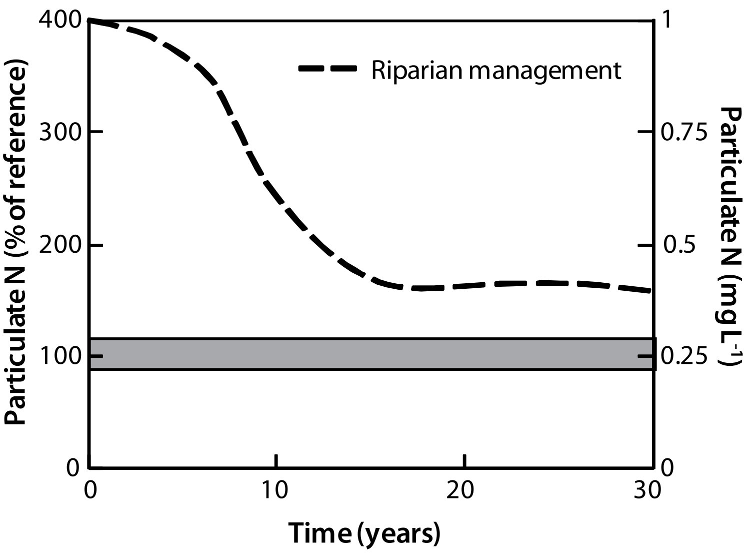


Figure 5.27: *Hypothetical concentrations of particulate nitrogen after riparian management. Particulate phosphorus is likely to respond in a similar manner but concentrations will be about 10 times lower.*

Typical nutrients to measure are total nitrogen (TN) and total phosphorus (TP), which incorporate both dissolved and particulate nutrients (including those bound to sediment particles), and the dissolved nutrients: nitrate (NO_3^-), ammoniacal-N (NH_4^+), and dissolved reactive phosphorus (DRP). Total dissolved nitrogen (TDN) and total dissolved phosphorus (TDP) may also be measured to determine whether organically bound nutrients are present primarily in particulate or dissolved form in the stream water.

When: Monitor sites monthly (preferably under stable flow conditions) for 1 year prior to restoration to establish a robust baseline dataset, then repeat this monthly monitoring for 1 year at 5-yearly intervals.

Timescales and measures of success: It may take many years before stream nutrient concentrations (particulate (Figure 5.27) or dissolved (Figure 5.28)) decline in response to riparian restoration efforts. McKergow et al. (2003) found minimal change in TN and TP concentrations 6 years after riparian planting of a stream catchment in Western Australia. However, they did find that maximum suspended sediment concentrations had decreased by an order of magnitude over this period. In Ngongotaha stream, Williamson et al. (1996) observed particulate N and P loads decrease by 30–40% in a period of 10–12 years after riparian retirement.

However, both of the studies noted increases in some soluble nutrient forms over these periods. McKergow et al. (2003) found that the filterable (soluble) fraction of P had increased, and although Williamson et al. (1996) found that soluble P loads decreased by 25%, soluble N loads (dominated by NO_3^-) increased by 25%. Possible explanations for higher soluble nutrient loads following riparian restoration include reduced assimilation by in-stream plants following shading of the stream, and groundwater lag times (e.g., 70–80 years for some groundwaters in the Lake Taupo catchment).

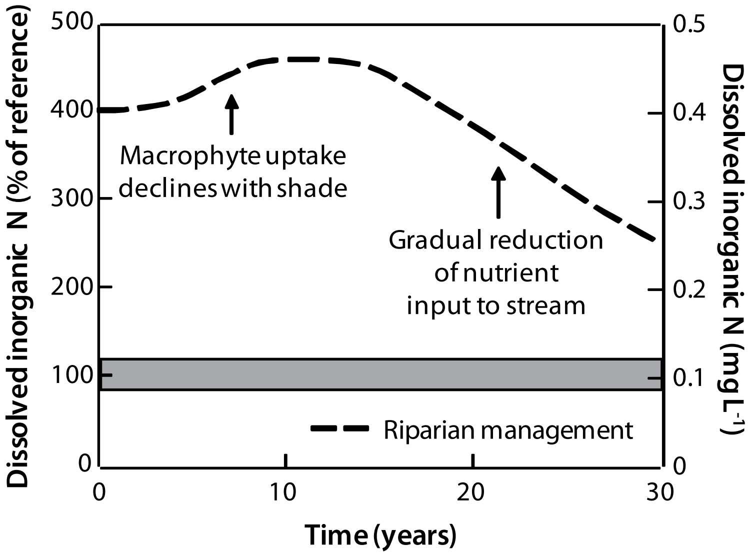


Figure 5.28: *Hypothetical concentrations of dissolved inorganic nitrogen after riparian management. Dissolved reactive phosphorus is likely to respond in a similar manner but concentrations will be approximately 10 times lower.*

It is unlikely that restored streams will ever achieve the nutrient concentrations of a reference stream, particularly where the reference stream is in a pristine forested catchment and where a large proportion of the catchment of the restored stream remains in developed land use (e.g., farm land, plantation forestry, urban).

Faecal indicators

Goal(s): WQ, DH, R

Background: Faecal microbes that can cause disease in humans ("pathogens") are of concern in natural waters primarily because of the threat they pose to people engaged in contact recreation (MfE/MoH 2003). Microbial pathogens can also be concentrated by filter-feeding bivalve shellfish, posing a risk to people consuming shellfish. Faecal contamination is normally monitored using "indicator" microbes that are universally present in the faeces of warm-blooded animals (including people), notably the bacterium *Escherichia coli* (in freshwaters) and the Enterococci group (in marine waters) (MfE/MoH 2003). Monitoring for all pathogens that are potentially present in waters (of which there are a very large number) would be hugely expensive and is currently impractical (MfE/MoH 2003).

In streams draining forested catchments, faecal microbial concentrations are typically much lower than in pastoral streams (e.g., Davies-Colley & Nagels 2002), but even near-pristine reference streams are never totally free of faecal contamination because of faecal deposition by birds and feral mammals in their catchments. *E. coli* concentration in streams is highly dependent on state of flow (e.g., Davies-Colley et al. 2008) with concentrations two or three orders of magnitude higher during storm flows (particularly on rising limbs of hydrographs) than during base flows. Forested streams can have surprisingly high faecal contamination during rainstorms, although still much lower than equivalent events in pastoral streams. The high faecal microbial contamination of pastoral streams in New Zealand can be attributed predominantly to livestock (e.g., Till et al. 2008), particularly cattle and deer due to their wallowing behaviour (de Klein et al. 2003). Sheep, in contrast, do not usually spend time in water, so faecal contamination in streams draining sheep country may be lower – at least during base flows, although similarly elevated during and after rainstorms when land runoff occurs.

Method: Because state of flow strongly affects faecal microbial concentrations, it is important to index faecal microbial testing of stream waters to flow. Therefore, the faecal microbial test results should be recorded with date and time of sampling, weather, and stream flow conditions, including whether samples were taken on a stream rising or falling stage. Ideally, a stream flow measurement should be made (see methods in SHAP Harding et al. 2009), or, less satisfactorily, the state-of-flow in the study stream may be indexed by reference to flow percentile at a nearby flow-monitoring site.

STANDARD METHOD

There are several standard methods of testing for *E. coli* specified in "Standard Methods" (APHA 1998), and most commercial water-testing laboratories with microbial facilities should be capable of offering at least one of these tests. Recently the Colilert™ (Coliform/*E. coli*) method has become very popular for its robust simplicity. This method requires only a 35°C oven and no specialist microbiological laboratory facilities other than the Colilert reagents and sealer unit (for sealing multi-well trays containing the water sample and added nutrient medium). Readers are referred to APHA (1998) for guidance on faecal contamination testing and MfE/MoH (2003) for guidance on interpretation.

Application of method: The standard methods for enumeration of *E. coli* or faecal coliforms, involving testing of water samples by an accredited laboratory with microbial expertise, are almost universally applicable in small (wadeable) streams and would normally be used by regional council staff. Community groups, however, might wish to consider setting up their own gear for the Colilert method if faecal bacteria are of concern in their waterways. A DIY method for *E.coli* detection and enumeration using cheap Petrifilm™ count plates is being developed by NIWA (Stott 2005) for use by community groups.

When: Monitor sites monthly (preferably under stable flow conditions) for 1 year prior to restoration to establish a robust baseline dataset, then repeat this monthly monitoring for 1 year following restoration and then at 5-yearly intervals.

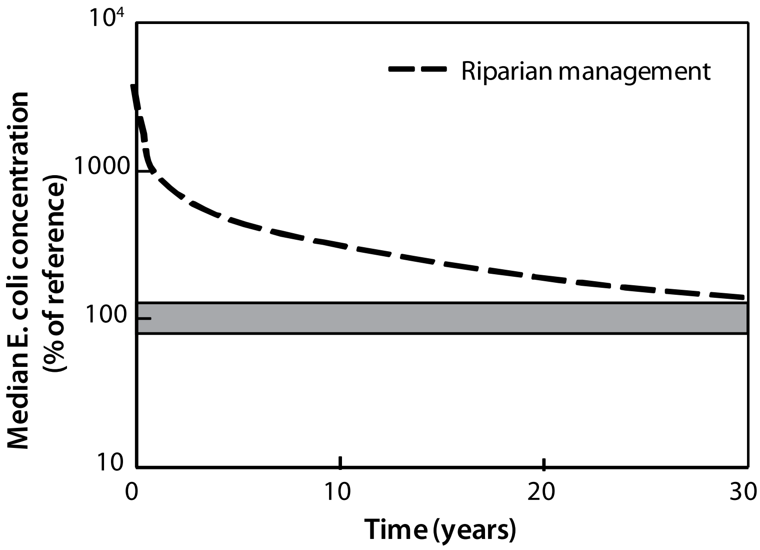


Figure 5.29: *Hypothetical trajectory of recovery from faecal pollution of a pastoral stream following fencing to exclude livestock (primarily cattle) from channels and riparian zones. (Note the logarithmic scale on the y-axis.)*

Timescales and measures of success: Recovery from faecal contamination in streams draining livestock farming areas is strongly dependent on exclusion of livestock – particularly cattle – access to stream channels and riparian zones by fencing. The faecal microbes stored in the riparian and channel areas are expected to be quickly (weeks to months) depleted after livestock exclusion by both microbial die-off and flushing by storm-flows. A slow further improvement in faecal microbial status towards that of reference sites is then expected over many years to decades as the riparian soils recover their infiltration (and thus microbe-entrapping) capacity (Figure 5.29).

Toxicants

Goal(s): WQ, DH

Background: A toxicant is defined as an agent that can produce an adverse response (effect) in a biological system, seriously damaging structure or function, or producing death (Rand et al. 1995). Discharges to the environment from industry, agriculture, or urban environments usually contain a complex mixture of toxicants that can cause adverse effects on aquatic communities. Examples of toxicants include heavy metals (mining, urban stormwater, geothermal, timber treatment), pesticides and herbicides (agriculture, horticulture), and polycyclic aromatic hydrocarbons (PAHs) (urban stormwater).

Stormwater runoff to urban streams can contain high concentrations of heavy metals, hydrocarbons, and toxic chemicals such as organochlorines. Vehicles and runoff from some unpainted roofs can be major sources of some metals, especially copper (wear and tear of brake linings) and zinc (tyre wear, unpainted zinc-coated roofing). Typical metals that are indicators of heavy metal pollution in urban streams are **copper, zinc, and lead**. The toxicity of these, and some other heavy metals, is hardness-dependent and therefore some knowledge of water hardness (obtained through one or two background samples) will be needed to assess the results against guidelines (e.g., ANZECC 2000).

Monitoring the dissolved form of these metals should be sufficient for assessing general stormwater contamination and potential for toxic ecological effects. Contaminants often exceed recommended toxicity levels for many aquatic invertebrates; however, total concentration of toxic materials may not be a good indicator of toxicity as the bioavailability of heavy metals and other compounds can be reduced when there is abundant organic matter in the stream. We suggest that direct measures of chemicals in water and/or sediments only be used in combination with an assessment of ecological effects (either ecotoxicological assessment (specialist) or macroinvertebrate biomonitoring (protocols provided)). An exception to that is if the measures are solely relating to downstream health of another waterbody.

Method: There are several ways to measure the impact of toxicants:

- direct measurement of chemicals in water or sediments and relate to reference condition or trigger values
- ecotoxicological field or laboratory assays of the direct effect of sediments or water on test species
- biological assessment using macroinvertebrate community metrics as an integrated measure of effects (see Biota section – note microbial indicators are also currently under investigation).

Ecotoxicological measures are a specialist area not covered by this document. Direct measurement of chemicals can be determined from a water or sediment sample, but must be analysed by a chemistry laboratory. Follow the instructions of your analytical laboratory for collection of water and sediment samples. Care must be taken to obtain a representative sample of stream water or sediment and to avoid contamination from airborne particles, residue on containers, or changes due to temperature or pH during sampling and subsequent transport to the laboratory.

Application of the method: Measurement of toxicants is likely to be relevant to restoration in special circumstances only, such as restoration of mine sites or streams in urban or industrial areas. In some instances, you may want to measure metals or other toxicants to diagnose the constraints within a catchment (e.g., to help understand why recovery of biological communities is not happening after riparian planting of an urban stream). Targeted measurement of toxicants could help to isolate important sources of water contaminants to the stream (e.g., sampling water above and below inflow pipes). We suggest that specialist advice be sought if you are intending to restore a mine-affected site as actions to mitigate effects and measure improvement may be more specialised than this Toolkit can provide. The types of toxicants to measure will depend on the mining activity (e.g., coal, gold etc.). Further information on the effects of different types of mining on New Zealand stream communities can be found in Harding et al. (2000) and the response of invertebrates to toxicants in Hickey (2000). Similarly, if your restoration site is a farm stream that has been exposed to chemicals through dumping of pesticides or herbicides or a stream below an industrial plant, then your choice of toxicant to measure will be influenced by the suspected active ingredients of those products (e.g., DDT, chlordane, glyphosphate herbicides).

When: Frequency of sampling could depend on the level of initial contamination. Ideally, water or sediment samples should be collected monthly (or at least seasonally) for a year prior to restoration work (or longer if possible), and monthly (or seasonally) each year for 5 years following restoration work. Sampling years may be at 5-yearly intervals after that.

Timescales and measures of success: The ANZECC (2000) guidelines provide trigger values that have been determined by multiple laboratory tests of single-species responses to a range of toxicant concentrations (i.e., from ecotoxicological tests). A measure of success could be when your restored site scores below these trigger values for a key contaminant or suite of contaminants. Please note the ANZECC guidelines are under review and subject to change after 2013. Alternatively, measures of toxicants could be compared to a reference location.

The rate of recovery from mine drainage or stormwater input can have widely differing recovery timescales depending on the type of activity and the remediation action. Major factors contributing to a long recovery period include accumulation of fine sediments that are contaminated with heavy metals or continued seepage of toxic minerals from disused mine sites. Some types of mining activities produce lower concentrations of heavy metals than others and take less time to recover (e.g., alluvial mining would have a faster recovery time than that of hardrock tunnelling; Harding et al. 2000). Urban stormwater can be treated by detention ponds and constructed wetlands (Suren 2000), and if these devices function efficiently, you might expect that removal of toxicants would be rapid at least until the efficiency of the ponds begins to decline. Changes in industrial practices, fuels, and housing products in future decades may also reduce the supply of heavy metals and chemicals to urban streams.

The illustrated hypothetical example (Figure 5.30) shows two response curves of total metal concentrations in stream water relative to a reference location under two potential contamination scenarios. In the urban stormwater scenario, stormwater detention ponds and constructed wetlands have been installed so that inflow stormwater to the urban stream is treated. The initial reduction is rapid and assumes that there are no sources of metals being released (i.e., from sediments) within the restored reach. The concentration of metals in the stream is proportional to the efficiency of the stormwater treatment methods. In the mine restoration scenario, a forested stream is monitored below an abandoned mine site and the initial reduction in total metals following the closure of the mine is rapid, but continued seepage from mine work sites and from heavy metals bound in sediments causes elevated metal concentrations at the restoration site over several decades.

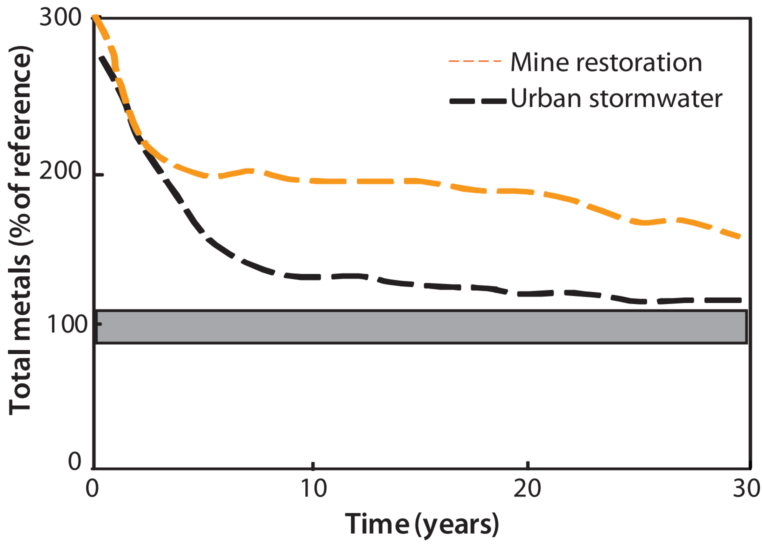


Figure 5.30: *Hypothetical response curves for total metal concentrations under two restoration scenarios: 1) treatment of inflow stormwater to an urban stream with detention ponds and constructed wetlands, and 2) restoration of a forested stream below an abandoned mine that still has sources of heavy metal contamination.*

pH

Goal(s): WQ

Background: pH (defined as the log base ten of the inverse of the hydrogen ion activity; $-\log_{10}\{H^+\}$) is an index of the activity of the hydrated proton (H^+; "hydronium" ion) in water. pH is important because it affects the equilibrium point of various important chemical reactions in waters, notably the ratio of free ammonia to the ammonium ion. pH affects aquatic life by a variety of mechanisms including via ammonia toxicity and influence on enzymatic reactions (Davies-Colley & Wilcock 2004).

pH in waters is buffered by the carbonate system, and is usually in the slightly alkaline range between about pH 7 and pH 8 (Davies-Colley & Wilcock 2004) because of the (conservative) property of alkalinity (or acid-neutralising capacity) of waters associated (mainly) with bicarbonate. Higher (alkaline) pH (up to pH 9 or even higher) is encountered in highly eutrophic waters owing to the metabolism of aquatic plants which fix aqueous CO_2 in the dark reactions associated with photosynthesis (Davies-Colley & Wilcock 2004). Lower (acidic) pH is generally only found in waters affected by:

- high organic acids, such as some wetland waters or where mineral acidity occurs naturally
- water draining certain rock and soil types (e.g., in New Zealand some soils in Northland naturally contain pyrite – as does overburden from some coal mines)
- industrial wastes.

Because of carbonate buffering, pH is not expected to be much affected in most stream restoration scenarios. The main exception would be restoration of a stream affected by acid mine drainage in which pH might initially be very low, in the vicinity of 3 or 4, and effects may be compounded by deposition of metal precipitates on the stream-bed.

Method: pH is routinely measured in natural waters with a glass electrode (APHA 1998). pH buffers (standards) can be prepared from laboratory reagents (e.g., potassium hydrogen phthalate) but commercially available buffers are convenient. Such buffers (typically near pH 4 and pH 10, bracketing the pH values expected in most natural waters) should be used for standardising the glass electrode on each and every use. Great care must be taken in rather weakly buffered fresh waters to prevent carryover of buffer to the natural water sample, and considerable time (5 minutes or more) should be allowed for the glass electrode potential to stabilise in natural waters which are low in ionic strength. Measurements should preferably be made in the stream rather than in water samples transported back to a laboratory.

The SHMAK kit (**www.niwa.co.nz/our-science/aquatic-biodiversity-and-biosecurity/tools/shmak**) includes universal indicator paper for roughly assessing pH according to colour, and may be useful for coarse-level pH screening.

When: pH of stream water need only be monitored routinely (e.g., monthly as part of general water quality survey) where pH has been a historical issue, as with some (industrially polluted) urban streams or mining-impacted streams. The glass electrode method is recommended, and consideration should be given to characterising the diurnal cycling of pH if any significant periphyton or other aquatic plant biomass is present.

Timescales and measures of success: Timescales of pH recovery in a stream historically affected by acid conditions could vary vastly depending on the source of acid, so no general guidance can be given. For example, recovery of a stream historically subjected to industrial acid discharge could be extremely quick (hours to days – although return of acid-sensitive stream fauna might be much slower), but recovery of a stream subjected to acid mine drainage could be very slow (many decades) depending on the time taken for acid minerals in mine overburden to be oxidised and the resulting acid to be flushed or neutralised.

Biota

Periphyton

Goal(s): AB, NH, A, R

Background: Periphyton is a complex assemblage of predominantly benthic algae, but also bacteria and fungi, that grows on surfaces (i.e., rocks, sand, wood, macrophytes) in streams. Periphyton can take two general forms:

1. microscopic, unicellular algae forming thin layers on stream substrates (i.e., diatoms)
2. macroalgae that develop as filaments, sheets or mats.

The macroalgae often dominate in low-gradient open streams while diatoms tend to dominate in higher-gradient, shaded streams (Murphy 2001).

Periphyton is an important food source for stream biota (e.g., snails, mayflies, caddisflies, midges) which in turn are fed upon by fish and birds. However, nuisance growths of periphyton can develop, generally when there is a high availability of light and nutrients for growth, and make the stream-bed habitat unsuitable for many sensitive invertebrate species (Figure 5.31). Nuisance growths of periphyton that make the stream unattractive for swimming and angling are considered to occur when cover of the visible stream-bed is >30% for filamentous algae and/or >60% for diatom/cyanobacterial mats (Biggs 2000).

Figure 5.31: Filamentous green algae.
Photo: Steph Parkyn

Periphyton growth in streams is regulated by light availability, flow and scour regime, nutrients, temperature, and grazing. Light is the primary factor controlling periphyton abundance since algae require light for photosynthesis and growth. Periphyton growth is restricted in highly shaded or turbid streams where only shade-adapted species may persist (e.g., diatoms). Small forest streams often receive less than 5% of full sunlight. In “lighter” open-canopy streams, green algae often dominate. Light saturation for most benthic algae occurs between 30–60% of full sunlight (Murphy 2001). While light is a primary factor controlling periphyton abundance, stream flow and bed scouring can slough periphyton from stream substrates under high current velocities (e.g., during floods) and where suspended sediment loads cause abrasion (Horner et al. 1989). Filamentous algae are susceptible to dislodgement if currents exceed 0.5 m s^{-1} (Horner & Welch 1981). Periphyton growth is also regulated by nutrient availability and can be low in open-canopy streams where dissolved nutrient concentrations are low. However, when nutrient enrichment occurs in open-canopy streams (e.g., as a result of fertiliser runoff, wastewater discharges) periphyton blooms can develop. Blooms are particularly common during the summer months when warmer temperatures enhance algal growth rates but stress some invertebrate grazers (e.g., mayflies).

Method: To monitor periphyton in stream restoration projects, you should use the periphyton cover rapid assessment method of Collier et al. (2007b). A measurement form is supplied in Appendix C. This method assesses periphyton cover across transects at the reach scale (50–100 m), and we recommend calculation of four of the indices: Periphyton Filamentous Index (PFI), Periphyton Mat Index (PMI), Periphyton Proliferation Index (PPI) and Periphyton Sliminess Index (PSI).

1. Within the reach, select a minimum of 5 evenly spaced transects across the wetted width of the stream. (These could coincide with transects used for substrate assessments.) Begin at the downstream transect.

2. For each transect, assess periphyton cover in a 10 cm diameter circle at 5 evenly spaced sampling points across the transect (i.e., at 10, 30, 50, 70 and 90% of the width).

3. Assess periphyton on whatever substrate occurs at each point (i.e., cobbles, sand, wood, macrophytes). Record the percentage cover of upper surface for the different periphyton categories (which are based on mat thickness or filament length and colour; Table 5.5).

4. Repeat for all five transects. Calculate a mean % cover for each periphyton category across the 5 transects. Calculate a periphyton total % cover by summing the mean % cover for each category type.

Table 5.5: *Periphyton thickness and colour categories.*

Thickness/filament category	Colour category		
Thin mat/film (0.5 mm thick)	NA	–	–
Medium mat (0.5–3 mm thick)	Green	Light brown	Black/dark brown
Thick mat (>3mm thick)	Green/light brown	Black/dark brown	–
Short filaments (<2cm long)	Green	Brown/reddish	–
Long filaments (>2cm long)	Green	Brown/reddish	–

Periphyton Filamentous Index (PFI) = Σmean%cover long filaments (green+brown/reddish)
Periphyton Mat Index (PMI) = Σmean%cover thick mats (green/light brown + black/dark brown)
Periphyton Proliferation Index (PPI) = PFI+PMI
Periphyton Sliminess Index (PSI) = [(%Thin/mat film) + (%Short filaments * 2) + (Medium mat * 3) + (Long filaments * 4) + (%Thick mat * 5)]/5

The PFI and PMI indices reflect the percent of stream-bed cover by long filaments and thick mats, respectively. The PPI index reflects the percent of total cover by both long filaments and thick mats.

When: Assess periphyton during the growing season (summer and autumn when plant biomass peaks) under base flow conditions and at least 3 weeks after the last significant spate (stream flows that have removed algae). We recommend monthly sampling during the growing season from January–April. An initial assessment should be undertaken before stream restoration activities begin. This should then be repeated annually during January–April. After 5 years, these assessments can be repeated at 5-yearly intervals.

Timescales and measures of success: Stream restoration activities that involve riparian planting will reduce the amount of light reaching the stream channel and consequently have a dominant effect on periphyton growth. Stream restoration activities that involve channel modifications to create more areas of slower river flow (e.g., reinstatement of meanders and natural rock dams, additions of wood) would be expected to increase the habitat available for periphyton growth, particularly filamentous algal growths, except in highly shaded or turbid streams where light is limiting or where stream nutrient concentrations are very low and limiting. Wood additions may also provide a more stable substrate for periphyton colonisation in soft-bottomed streams (Coe et al. 2009). Conversely, activities that seek to restore natural flow variations (e.g., removal of artificial dams) will increase periphyton susceptibility to disturbance by flood flows and scour. Where a restoration stream site has high light and nutrient availability, restoration activities that slow river flows should be carried out in conjunction with riparian planting to shade the stream and limit the development of nuisance periphyton growths.

Figure 5.32 shows an example of hypothetical response trajectory for the PPI anticipated to result from riparian management and channel reconstruction. Periphyton communities are likely to be impacted rapidly (i.e., within 1 year) by engineering works that have disturbed the channel bed, but periphyton may also proliferate following disturbance and when modifications to create pools have produced areas of slow flow. Stream periphyton responses to riparian planting will be more gradual, with the community shifting slowly as the vegetation canopy develops and shading increases. Both scenarios show complete canopy cover over the stream after 10 years, which has the greatest influence on periphyton proliferation. Periphyton sliminess and mat type may also change with shading as community structure changes.

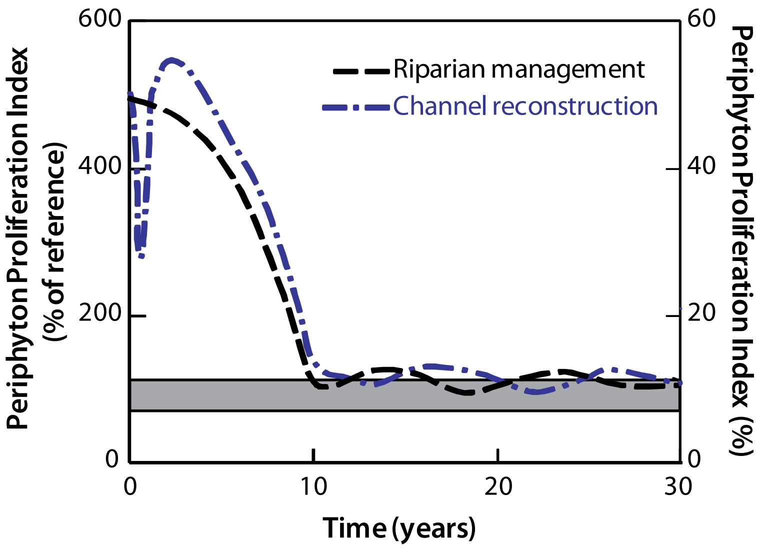


Figure 5.32: *Hypothetical Periphyton Proliferation Index scores after riparian management or channel reconstruction (with plantings that shade the stream channel).*

In-stream macrophytes

Goal(s): AB, NH

Background: In-stream macrophytes (plants) include tall-growing vascular species, bryophytes (i.e., liverworts, mosses) and charophytes. While charophytes such as *Nitella* are often regarded as an unusual form of macroalgae, they are not true algae as they develop complex reproductive structures (Clayton 2002). They also strongly resemble higher plants and are an important component of the native flora of New Zealand streams.

Macrophytes are important components of stream ecosystems. They provide habitat and cover for invertebrates and fish and a surface for colonisation by algae and bacteria. They also reduce water velocity and encourage the deposition of fine particles, and their roots help to stabilise the stream-bed. However, it is possible to have too much of a good thing. Dense growths of macrophytes in streams, particularly of invasive introduced species, can smother benthic habitats, reduce stream biodiversity, and impede water flow, and their photosynthesis–respiration cycle can cause wide fluctuations in dissolved oxygen and pH. In-stream macrophyte growth in streams is strongly controlled by light availability. All plants require light for photosynthesis, so highly shaded streams (e.g., small streams in native forest) typically have a low abundance of in-stream plants and support only those species able to tolerate light-limiting conditions (e.g., bryophytes).

In contrast, streams with minimal shading (e.g., large and/or open streams in farmland) and low channel gradients often support dense growths of macrophytes, particularly fast-growing species, many of which are introduced (Figure 5.33). Light can also be limiting for submerged species in turbid streams (e.g., polluted lowland streams). Light availability primarily controls seasonal changes in macrophyte abundance, with peak abundances when day lengths are longer and light intensities higher in the spring and summer seasons. This period also coincides with fewer floods and more stable flows that can enhance macrophyte growth. The abundance and composition of the in-stream plant community is strongly dependant on flow regime. Most macrophyte species, with the exception of some mosses, are unable to colonise and grow in fast-flowing waters or in those streams regularly

Figure 5.33: Excessive growth of in-stream macrophytes.
Photo: Steph Parkyn

disturbed by flood flows. Tall-growing macrophytes are generally absent from streams with a mean water velocity >0.9–1.0 m s^{-1} (Henriques 1987, Chambers et al. 1991, Bowden et al. 2006), and/or with more than 13 high-flow disturbances per year (i.e., flows 7 times greater than mean annual flow) (Riis & Biggs 2003). Mosses are sensitive to substrate instability. They are often most abundant in fast-flowing waters on stable boulders and bedrock (Murphy 2001).

The availability of nutrients and inorganic carbon is also important for in-stream macrophyte growth. However, these are not usually regarded as major factors regulating plant distribution and abundance in streams, due to the constant replenishment of supply from upstream sources and the ability of many rooted macrophyte species to take up nutrients from stream-bed sediments (Bowden et al. 2006). Competition and herbivory are other factors that can structure in-stream plant communities. Introduced species are often vigorous competitors and may displace smaller stature, slower-growing native species in sites where space is limited. Worldwide, herbivory is generally not considered to be a major factor controlling in-stream plant distribution and abundance (Bowden et al. 2006), and in New Zealand specifically there are few stream invertebrates that graze directly on in-stream plants (with the exception of the freshwater crayfish, *Paranephrops,* and the moth larva, *Hygraula nitens*) (Collier et al. 2007). Although some waterfowl, notably black swans, are also grazers of New Zealand's freshwater plants (especially *Egeria densa* and *Elodea canadensis*), these birds mostly reside in slow-flowing larger rivers, lakes, and ponds. Some pest fish, e.g., grass carp and rudd, can eat macrophytes in slow-flowing lowland streams and rivers.

Method: To monitor changes in in-stream macrophyte communities you can use three simple indices: Macrophyte Total Cover (MTC), Macrophyte Channel Clogginess (MCC) and Macrophyte Native Cover (MNC). These indices form the basis of the macrophyte cover rapid assessment method of Collier et al. (2007b). This method assesses macrophyte cover at the reach scale (50–100 m); a measurement form is supplied in Appendix D. For larger-scale restoration projects (i.e., where whole catchments or sub-catchments are subject to restoration activities), you should apply the assessment method to multiple reaches. A summary of the method is provided below (see Collier et al. 2007b for further details).

1. Within the selected reach, select at least 5 evenly spaced transects across the wetted width of the stream.

2. For a 1 m wide band upstream of each transect, estimate the total cover of macrophytes.

3. Estimate the total of cover of emergent (with parts above water) and submerged (below water surface) macrophytes. Identify each emergent species and estimate a percentage cover (if less than 5% cover these can be categorised as "other"). For quick guides for identifying aquatic species see **www.niwa.co.nz/our-science/aquatic-biodiversity-and-biosecurity/tools/quickguides**

4. Categorise submerged species as either "below surface" or "surface reaching" (at or very near the surface of the water) and estimate cover for each category.

5. Identify each submerged species and estimate a percentage cover.

6. Calculate indices:
   Macrophyte Total Cover (MTC) = {∑(%emergent + %submerged)}/no. of transects
   Macrophyte Channel Clogginess (MCC) = {∑(%emergent + %surface reaching) + (%below surface*0.5)}/ no. of transects
   Macrophyte Native Cover (MNC) = {∑(%native species)}/ no. of transects

When: Macrophyte assessments should be performed during the growing season (summer to early autumn for maximum abundance) under base flow conditions. Avoid sampling in late autumn as macrophytes naturally die off over winter and note that large floods can dislodge macrophytes which will affect your results. An initial assessment prior to stream restoration activities should be repeated annually, preferably during the same month each year as the pre-restoration assessment was done. Macrophyte cover and abundance won't change as rapidly in response to conditions as periphyton. After 5 years, assessments can be carried out on a 5-yearly basis.

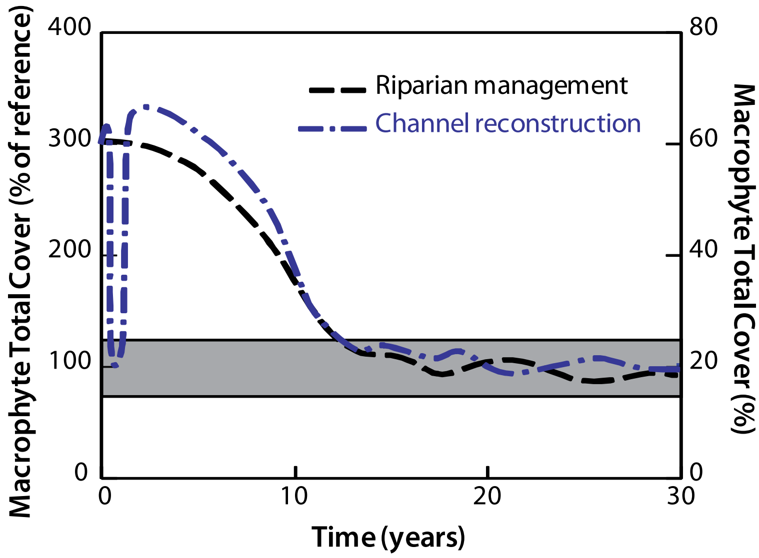


Figure 5.34: *Hypothetical Macrophyte Total Cover (MTC) after riparian management and channel reconstruction (with plantings that shade the stream channel).*

Timescales and measures of success: Stream channel shading is likely to reduce in-stream Macrophyte Total Cover (MTC) and Macrophyte Channel Clogginess (MCC), particularly where introduced macrophytes are present. It is also likely to alter species composition, favouring species better adapted to low light levels (e.g., native bryophytes, charophytes, *Potamogeton ochreatus*). Shading is therefore likely to increase Macrophyte Native Cover (MNC) in some situations, although small stony streams typically do not support macrophytes in general.

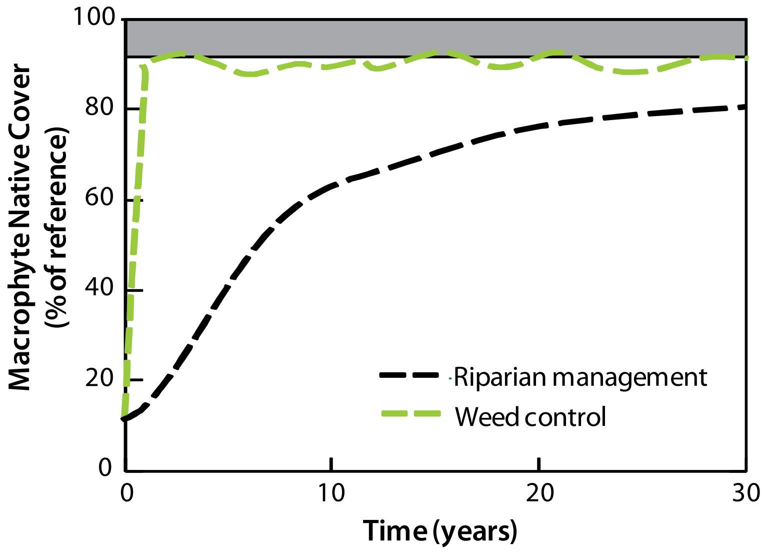


Figure 5.35: *Hypothetical Macrophyte Native Cover after riparian management or weed control.*

Stream restoration activities that involve channel modifications to create pools (e.g., remeandering, placement of logs and boulders, creation of pools) would be expected to increase the habitat available for macrophyte growth, and thus lead to increases in MTC and MCC indices, except in highly shaded streams where light is limiting. Conversely, activities that seek to restore natural flow variations (e.g., removal of artificial dams) will increase macrophyte susceptibility to disturbance by flood flows and scour, resulting in likely decreases in MTC and MCC. Where a restoration site has exotic invasive species already present (see Table 5.6), restoration activities that slow river flows should be carried out in conjunction with riparian planting to shade the stream and ensure that these species do not develop into nuisance growths. Macrophyte communities may be damaged during channel modifications such as wood addition or pool formation, but are likely to recover relatively quickly (i.e., within 1 year) to changes that have immediate effects on stream flow regime and turbidity. Stream macrophyte responses to riparian planting will be more gradual, with the community shifting slowly as the vegetation canopy develops and shading increases. Figure 5.34 illustrates the hypothetical response trajectories for MTC (MCC will likely follow a similar trajectory) anticipated to result from riparian management and channel reconstruction.

Weed control activities (e.g., hand-weeding and use of selective herbicides) can potentially be used alongside other restoration activities to speed the recovery of Macrophyte Native Cover (MNC) in streams, as it is possible that exotic invasive weeds will restrict native species recovery (Figure 5.35). For example, diquat (a selective herbicide that is effective against many of the introduced oxygen-weeds) has minimal effect on native charophytes and other freshwater biota (e.g., fish) if used in accordance with label application guidelines. However, local council requirements for the use of herbicides in waterways should be checked, as consents may be required.

Table 5.6: *Macrophyte species commonly found in New Zealand streams. * = invasive and can develop into nuisance growths in slow-flowing streams and rivers (compiled from Coffey & Clayton 1988, Clayton 2002, Collier et al. 2007b).*

Native species	
Submerged	
Pondweeds	*Potamogeton ochreatus* (blunt pondweed)
Charophytes	*Nitella hookeri/cristata*
	Nitella stuartii
Turfs	*Glossostigma* sp.
Milfoils	*Myriophyllum triphyllum*
	Myriophyllum propinquum
Emergent	
Persicaria decipiens (swamp willow weed)	
Floating	
Azolla filliculoides	
Lemna minor (duckweed)	

Introduced species	
Submerged	
Callitriche stagnalis (starwort)	
Ranunculus trichophyllus (water buttercup)	
Oxygen weeds	*Ceratophyllum demersum* (hornwort)*
	*Elodea canadensis**
	*Egeria densa**
	*Lagarosiphon major**
Pondweeds	*Potamogeton crispus* (curly pondweed)*
Emergent	
Apium nodiflorum (water celery)*	
Glyceria maxima (reed sweet grass)*	
Ludwigia palustris (water purslane)	
Mimulus guttatus (monkey musk)*	
Myriophyllum aquaticum (parrots feather)*	
Nasturtium officinale (water cress)*	
Persecaria hydropiper (water pepper)	
Veronica americana/anagallis-aquatica (water speedwell)*	
Floating	
Azolla pinnata	

Benthic macroinvertebrates

Goal(s): AB, WQ

Background: Benthic macroinvertebrates are animals without backbones; they range in length between about 0.25 mm and 15 cm for a full-grown kōura (i.e., they are visible to the naked eye) and they live on or a short distance beneath the stream-bed surface. Macroinvertebrates play a central role in stream ecosystems and in monitoring of stream ecological health. In most streams, the majority of macroinvertebrates are likely to be insects, but other groups such as molluscs (snails, clams), worms (several types including oligochaetes, flat worms, nemerteans), crustaceans, mites, hydroids and springtails are all common and may be very abundant. Macroinvertebrates occupy a niche as primary consumers, feeding mostly on periphyton (algae and associated microflora), dead leaves and wood, or each other. In turn, they are eaten by fish and other vertebrates, such as blue duck. Adult aquatic insects leave the water and become food for birds, bats, spiders, etc. Therefore, benthic macroinvertebrates are extremely important for processing terrestrial organic matter and primary productivity in streams, and passing it on to higher levels of the food chain. Benthic macroinvertebrate communities are widely used as indicators of stream ecosystem health, because they:

- include a wide range of species, each with relatively well-known sensitivity or tolerance to stream environmental conditions (e.g., Macroinvertebrate Community Index (MCI); Stark & Maxted 2007)
- are visible, easily caught, and fairly easily identified without the need for specialised equipment (though a microscope is helpful)
- live for a moderately long time (up to 1–2 years) as aquatic forms, and so their presence or absence reflects average environmental conditions over many months
- are not highly mobile (except by drifting downstream), so they tend to reflect environmental conditions at the place they are found.

The benthic macroinvertebrate community typical of pristine conditions has a high diversity of species or "taxa" (taxa are groups of species, often used when the species themselves are undescribed or hard to identify). The invertebrate community is usually dominated by three orders of insects: the mayflies, stoneflies, and caddisflies (together known as EPT, referring to their scientific names Ephemeroptera, Plecoptera and Trichoptera, respectively).

Method: We recommend using the protocols for sampling wadeable streams described in Stark et al. (2001), which has extensive guidelines on selecting the most appropriate method for your purposes. For this document see website: **www.mfe.govt.nz/publications/water/macroinvertebrate-protocols-wadeable-streams-nov01.html**

A number of guides for identification of New Zealand benthic macroinvertebrates are available. We recommend the booklet "Auckland Stream Invertebrates" available from Wai Care in Auckland (**www.waicare.org.nz**) for use by community groups. More detailed guides to genera are also available from NIWA online at **www.niwa.co.nz/our-science/aquatic-biodiversity-and-biosecurity/tools** To assess the health of the macroinvertebrate community, a number of indices are available. These indices rely on the different sensitivities of different invertebrate groups to degraded conditions. One index is the richness of EPT taxa, another is the proportion of invertebrates in a community belonging to EPT. The most commonly used index in New Zealand is the Macroinvertebrate Community Index (MCI; Stark and Maxted 2007). MCI assigns a score to each species or taxon, based on its tolerance or sensitivity to organic pollution, then calculates the average score of all taxa present at a site.

For community groups, we recommend using % EPT taxa (excluding Hydroptilidae – these are Trichoptera that consume algae and can be commonly found in open pastoral streams) as your measure of invertebrate community change and restoration success. For regional councils that are able to process samples with a higher taxonomic level, we suggest MCI, % EPT, and EPT richness may be more suitable. If local or regional reference site data are available, these indices can be combined simply into a single index following Collier (2008).

Other useful measures that can be applied using macroinvertebrates are **adult insect monitoring** *and* **invertebrate species traits analysis.** *Both these measures currently require specialist skills so have not been included as indicators in this document, but they may still be useful to monitor in some situations.*

Monitoring the adult, flying stage of macroinvertebrates can give you a better idea of the total biodiversity of your stream and riparian area. Seepages, small tributaries, backwaters, and pools are all habitats from which adult aquatic insects emerge and these are not commonly sampled by usual methods of benthic macroinvertebrate collection. The taxonomy is better known for adult insects and species can be identified more readily (usually by experts) leading to more accurate assessments of biodiversity. In some cases where access to streams is difficult, or where the water is too deep to sample, sampling by trapping adult insects in the riparian zone can be a substitute for benthic invertebrate sampling.

Macroinvertebrate species traits can be used to assess aspects of ecosystem function. Species traits, such as life history characteristics (mode of reproduction, number of eggs produced), resilience or resistance characteristics (body form and flexibility, dispersal capacity), and general physiological and biological features that enable organisms to live where they do (mode of respiration, feeding habits), provide a means of linking the functional roles of organisms with environmental processes (e.g., invertebrate breakdown of organic matter contributes to nutrient cycling). Further information on New Zealand macroinvertebrate species traits can be obtained by searching the website :

FBIS (FRESHWATER BIODATA INFORMATION SYSTEM)
secure.niwa.co.nz/fbis/

When: The presence or absence of macroinvertebrate taxa in New Zealand is not strongly seasonal, so communities can be sampled at any time of year if using indices such as EPT taxa richness and MCI. However, we recommend sampling during the same season each year to maximise consistency between years and for % EPT calculations as composition can be more variable than presence/absence. Sampling should not be done within 2–3 weeks after a flood (a flood being a flow event high enough that it occurs on average only once per year). Sample on several occasions before restoration (preferably for a few years to get a good baseline) and annually for the first 5 years and 5-yearly after that.

Timescales and measures of success: Very few studies have examined the timescales over which macroinvertebrates recover after stream restoration activities. Two New Zealand studies in small pastoral hill-country streams with upstream sources of colonists (Quinn et al. 2009, Jowett et al. 2009) found that MCI and several other invertebrate metrics showed significant improvement within 6–8 years after riparian fencing and planting. However, Parkyn et al. (2003), examining 9 streams with riparian plantings from 2–24 years old, found QMCI (a quantitative version of MCI) had significantly improved in only 3 streams, with riparian buffers 8, 20, and 24 years old, respectively. The streams with younger plantings showed some changes in the proportions of pollution-sensitive and pollution-tolerant species (in particular, mayflies, stoneflies, and chironomids) and some decrease in invertebrate density, which the authors interpreted as indicating transitional communities moving towards a "pristine" condition. Extrapolating from their data, we could expect a stream invertebrate community to reach a "clean water" MCI score after about 30 years, though the community may still be somewhat different to the reference stream at that time. Carline & Walsh (2008) found differences in invertebrate density – but not diversity – in restored streams compared to control streams 3–5 years after restoration measures were implemented.

Figure 5.36 shows the expected improvement in Macroinvertebrate Community Index (other macroinvertebrate indices should follow a similar trajectory) under three scenarios. With riparian fencing and planting, significant reductions in fine sediment may occur as early as 1–2 years after stream restoration begins and may continue for the next few years as stream banks stabilise. Peak summer water temperatures and in-stream plant/algal growth will reduce as the stream channel becomes shaded by riparian vegetation, which may begin 5 years after riparian trees are planted (Parkyn et al. 2003), and will be largely complete by 10 years for a small stream. The macroinvertebrate community is expected to continue to change towards that of reference communities as inputs of terrestrial leaves and wood increase (providing food and habitat). However, input of large wood will take more than 400 years to return to natural levels (Meleason & Hall 2005) unless addition of wood becomes part of the restoration project.

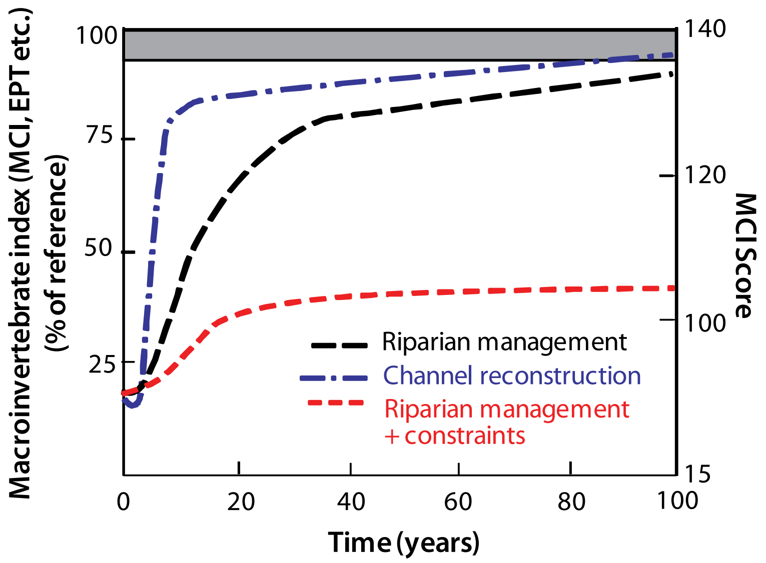


Figure 5.36: *Trajectory for improvement of benthic macroinvertebrates (as measured by the Macroinvertebrate Community Index) with riparian management, channel reconstruction, or when restoration is subject to constraints (see text for explanation).*

Channel reconstruction is a common form of stream restoration in Europe and North America, and many studies have monitored the responses of macroinvertebrates to such habitat engineering (see Roni et al. 2008). Reconstructing channels or adding in-stream features (e.g., logs) to increase habitat complexity can lead to increased diversity of invertebrates (Tullos et al. 2009), but the actual response of invertebrates has varied widely (Roni et al. 2008). Some studies have shown a rapid improvement in invertebrate diversity and density to a new stable level, in some cases within 4 years (Wallace et al. 1995, Ebrahimnazhad & Harper 1997, Friberg et al. 1998). Adding habitat features such as logs or artificial riffles can result in an almost immediate improvement in the fauna (Wallace et al. 1995, Ebrahimnazhad & Harper 1997, Haapala et al. 2003), whereas reconfiguring the stream channel may cause an initial decrease in macroinvertebrate diversity and density (lasting 1–4 years) due to temporary disruption of the physical habitat (Friberg et al. 1998, Lorenz et al. 2009, Tullos et al. 2009). While those studies have shown a rapid response in the invertebrate fauna, others have shown no detectable difference between restored and unrestored reaches (Roni et al. 2008). Channel reconstruction may not benefit the invertebrate fauna if it is designed solely for fish habitat enhancement, or if factors other than habitat complexity are limiting the invertebrate fauna (Roni et al. 2008).

Constraints to biota recolonisation can affect the rate at which the macroinvertebrate community recovers, or the endpoint that is achieved (Figure 5.36). For example, if the restored stream reach is:

- a long distance from a high quality stream habitat, stream invertebrates may be unable to reach the stream despite habitat conditions being suitable
- short (e.g., <300 m), or the riparian buffer strip is narrow (e.g., <10 m), aspects of the stream environment such as fine sediment or water temperature may not improve sufficiently to allow the return of sensitive fauna
- subject to catchment-scale constraints such as upstream land use intensification (Nerbonne & Vondracek 2001; Harrison et al. 2004).

If riparian restoration does not involve planting trees but only fencing a grass buffer strip, grasses and stream banks may not provide the shading needed to reduce stream temperatures and inhibit aquatic plant or algal growth. In this case, temperature-sensitive mayflies and stoneflies may not return, and leaf-shredding insects are unlikely to replace algal-piercing ones. Under any of these constraints the improvement trajectory would be unlikely to reach reference condition (Figure 5.36).

Stream mega-invertebrates

Goal(s): AB, C

Background: Stream mega-invertebrates are defined here as large invertebrates that are not commonly collected during macroinvertebrate sampling. The public may have particular interest in these invertebrates for cultural reasons or they may form part of mahinga kai (traditional food for Māori), and their presence may be an indicator of restoration success.

Examples of mega-invertebrates include:

- the freshwater crayfish or kōura (*Paranephrops*)
- the freshwater mussel or kākahi/kaeo (*Echyridella* and *Cucumerunio*)
- the freshwater shrimp (*Paratya*).

These large invertebrates have ecological roles that influence in-stream functions and other members of the stream community. For example, kōura process leaf litter into smaller pieces, creating faster decomposition (Parkyn et al. 1997), kākahi/kaeo filter phytoplankton from the water column (Phillips 2007), and shrimps and kōura may help clean the stream-bed of sediments when they occur in large numbers (e.g., Pringle et al. 1993; Parkyn et al. 1997). *Paratya* predominantly occur in streams near the coast, sometimes in such large numbers that they can be difficult to catch and count. Further inland, however, their numbers may be lower (e.g., Hicks 2003).

Method: The simplest measure of restoration success for each of these species is to note their presence or absence, although abundance estimates are more informative. Size can also be measured to assess recruitment of juveniles.

Kōura abundance can be assessed at the same time as fish surveys (number caught over a standard reach length) using electric fishing or spot-lighting, and if the method is kept standard over time, this can be an adequate relative measure of change. Kōura can be monitored annually, preferably in late summer when numbers are likely to be highest and a full range of size classes will be present. It is important to monitor at the same time of year each year, as the numbers caught can vary seasonally. If kōura are the target of restoration, then size ranges should be determined either by measuring each individual (OCL = length from orbit (behind eye) to end of carapace) or by grouping into approximate year classes (Young-of-the-year: <8 mm OCL, Year 1: 8–20 mm OCL, Year 2+ : >20 mm OCL), so that recruitment can be assessed over time.

A quick quantitative measure of **kākahi/kaeo** abundance would be to count the number occurring across transects (metre-wide band) which are also monitored for substrate particle size analysis. If the stream is too deep for this assessment, then qualitative counts by observers snorkelling or using an underwater periscope/viewer in a longitudinal transect may also be used if safe to do so (e.g., Jones et al. 2001, Kusabs & Emery 2006). If kākahi/kaeo are the target of restoration and numbers are low, then more intensive searching may be required. Rainforth (2008) used timed searches of sites in the Whanganui River (a standard unit of 1 hour searching using snorkelling and wading). To indicate whether new ones are re-establishing, note the approximate size classes of mussels (<30 mm (<3 years), 30–60 mm (3–12 years), <60 mm (>12 years); N. Phillips, NIWA, pers. comm.), but try and avoid removing them from the substrate if possible.

Paratya occasionally occur in benthic samples, but are more likely to be sampled by sweep net through macrophytes (e.g., Carpenter 1983). Numbers in sweep net samples should be categorised according to a coded abundance (e.g., Stark (1998):

- Rare = 1–4
- Common = 5–19
- Abundant = 20–99
- Very abundant = 100–499
- Very very abundant = 500+.

When: Annually in summer.

Limitations: Large invertebrates may be unable to recolonise a restoration site naturally because of dispersal constraints. Kākahi/kaeo rely on a fish host to be present to carry the juvenile life stages to new areas or may be impacted by high levels of sedimentation (Phillips et al. 2007, Rainforth 2008). *Paratya* require access to estuaries to complete their life cycle (planktonic zoae are washed downstream to develop into larvae in estuaries and eventually move back upstream; Carpenter 1983), and while kōura are mobile, they do not appear to move overland to new catchments (Smith & Smith 2009). Therefore, the absence of these mega-invertebrates may not indicate that the restoration has failed, but may suggest that there are other limiting factors present, and active translocation may be necessary.

Timescales and measures of success: Very little information is available on the recovery of these species after restoration. Studies of mussel populations in Virginia, USA, eradicated after a toxic spill event, suggest that mussel recovery is likely to take decades and may require successive reintroductions for some species (Jones et al. 2001). Mussels can live for up to 40 years and require a fish host for dispersal of their larval stage, and so an absence of fish may limit re-establishment. Parkyn & Collier (2004) found that kōura populations took 5 years to recover after a major flood in a Waikato pastoral stream when numbers were reduced to very low levels.

An obvious indicator of success is the natural reoccurrence of a target species within the restored area, either as absolute presence/absence or as the presence of smaller size classes indicating recruitment. Ultimately, the gradual increase in abundance towards that of a reference site would be considered an indicator of success (Figure 5.37). For the hypothetical example of a pasture stream that has been fenced and planted, there may only be slight improvements in kōura numbers over time as good populations of kōura can exist in pasture streams (Parkyn et al. 2002). Typically, kōura numbers and longevity may increase once the restored site reaches forested reference conditions, but growth and secondary production will decrease as water temperature and food sources change.

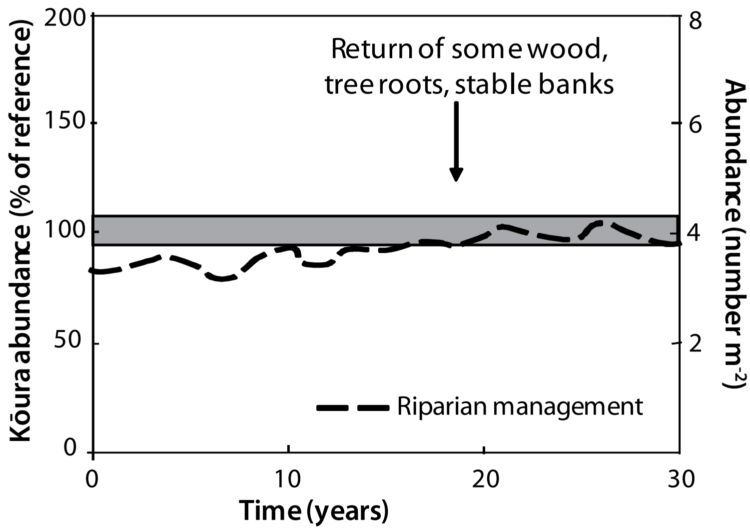


Figure 5.37: *Hypothetical timescale of kōura (Paranephrops planifrons) recovery with riparian management of a pasture stream. Abundance was measured in March based on data from small, Waikato hill-country streams.*

In cases where there are barriers to natural dispersal, translocation of mega-invertebrate species may be required. If the target of your restoration is the return of these species and you suspect there are constraints to their natural recolonisation, then separate studies detailing the habitat suitability, continuing threats to reestablishment, and connectivity to existing populations need to be conducted.

Fish

Goal(s): AB, C, F

Background: Freshwater fish are an important part of stream ecosystems, both for their intrinsic value and as an indicator of overall ecosystem function. Many fish are long-lived and relatively easily captured, and the range of species resident in a stream and their abundance can reflect long-term conditions at a site. The existence of healthy fish communities in streams is often highly valued by the community, thus re-establishing fish populations could be an important goal of your restoration. Approximately one third of fish species in New Zealand are capable of diadromy (able to migrate between fresh and salt waters to complete their life cycle), and consequently their distributions are strongly influenced by distance inland, gradient, and altitude. For non-migratory species, natural historical events including glaciations, volcanism, and geological uplifting have influenced their present-day distributions. All these factors and processes need to be considered when assessing potential restoration or rehabilitation of fish communities in New Zealand.

Method: Standard national protocols for sampling freshwater fish communities are still under development, and here we advise using the Environment Waikato protocols (David & Hamer 2010) for wadeable streams (1st – 3rd order streams). Detailed procedures are available at **www.ew.govt.nz/Publications/Technical-Reports/TR-201009/** and summarised below.

Define the characteristics of the restoration site(s). In particular it will be necessary to determine the:

- elevation (metres above sea level) for the site(s)
- stream order (1st, 2nd, or 3rd order)
- stream segment gradient
- distance inland from the coast (km inland)
- region in which the site(s) lie.

All of these attributes can be obtained off GIS database layers. For more information on initial desktop analyses, see SHAP (Harding et al. 2009) or Part 3 of this document.

These site attributes should then be used to identify a comparable reference site(s) for setting meaningful restoration targets for fish. In selecting an appropriate reference site(s) for fish, the attributes of the restoration site(s) should be as similar as possible to those listed above and, most importantly, meet the following minimal human disturbance criteria (if possible):

- no artificial impediments/ barriers to fish migration
- >70% native riparian vegetation from headwaters to the site of interest
- contain <10% total composition of introduced fish species in the assemblage
- have no mines within catchment
- be subjected to <10% alteration of natural flow regime (e.g., through water abstraction or hydro dams).

Figure 5.38: Electrofishing.
Photo: Bruno David

Choose whether electrofishing (Figure 5.38) or spotlighting is most suitable for your sites or situation (e.g., access to equipment). Refer to Appendix E to help make this decision.

IF ELECTROFISHING:

- Sample a 150-m reach, separated into 10 subsections of 15-m length.
- Identify, count, and measure (total length) all fish species caught in each subsection. An online key to identifying New Zealand fish species is available at: **www.niwa.co.nz/our-science/freshwater/tools-old/fishatlas/key**

IF SPOTLIGHTING:

- Mark a 150-m reach, separated into 10 subsections of 15-m length. Mark sections during the day and begin spotlighting 45 minutes after dark.
- Walk beside the stream if possible, shine the spotlight 1–2 m ahead, and sweep from bank to bank. Count and identify all fish. An online key to identifying New Zealand fish species is available at **www.niwa.co.nz/our-science/freshwater/tools-old/fishatlas/key**

Important metrics to evaluate include:

- total fish species diversity
- relative species abundance and total community abundance (fish/100m^2)
- assemblage balance (proportion of each species within the assemblage)
- size structure of a target species population or community
- ratio of exotic to native species.

Identifying fish species by presence/absence can give you a ratio of **exotic to native** species, but fish **diversity** alone is unlikely to be a sufficiently sensitive indicator to enable confident reporting of change (improvements/successes/failures), particularly over short timescales (less than 3 years). This is mainly because New Zealand's native fish fauna has relatively few species, particularly at inland sites where non-migratory species predominate. An assessment of the New Zealand Freshwater Fish Database records by Richardson & Jowett (1996) indicated that at high elevation sites (above 150 m asl) an average of 2–3 native species is typically found, and at lower elevations (below 150 m asl) 3–5 species could be expected, although expected numbers can vary regionally, especially where recently discovered galaxid complexes occur. Thus, even with a robust sampling methodology designed to detect changes in diversity at a reach scale, the generally depauperate fauna presents difficulties for confidently assessing change.

We recommend monitoring **relative abundance** and **assemblage balance** (proportion) of each species within the community. For establishing recruitment patterns, fish should be measured to determine population or community **size structure**.

Some fish species can be targeted as **indicator species** and other fish species can be ignored as they do not make good indicators of restoration success. For instance, small, seasonal species that shoal and are generally pelagic, highly mobile, and difficult to monitor successfully with electrofishing methods (e.g., īnanga, smelt) can produce highly variable results (presence/absence, relative abundance) through time, and are unlikely to be good indicators of restoration success or failure. Eels are also unlikely to make good indicators for a variety of reasons: they can be quite mobile, tend to be tolerant of degraded conditions, are widespread, and adult numbers may be significantly impacted by local recreational and/or commercial harvesting.

Species that are likely to make good indicators include shortjaw, giant, and banded kōkopu. If the focus is to be on these species only, the recommended survey method is spotlighting (Appendix E). If more detailed data on these species are warranted (e.g., individual turnover and growth rates), the Department of Conservation's large galaxiid monitoring guidelines may be more appropriate (contact the Large Galaxiid Recovery Group Leader, Department of Conservation, Head Office, Wellington). Other species likely to make good indicators are kōaro, torrentfish, and the various bully species. For these species, the recommended survey method is electrofishing (Appendix E).

When: Monitor both the rehabilitated site and reference site at the same time of year. Sampling should be conducted anywhere between December to end of March in any year. This is the period when maximum species diversity will be present within New Zealand rivers (Hayes et al.1989), and when fish are most active and accessible for capture. Note that the number of young-of-the-year (YoY) fish within the population can vary markedly over this period based on recruitment peaks that can occur variably during this period. While many of these fish can come and go, fish that are a year or more old will better represent the local population. Sampling 1–2 times per year should be sufficient to detect any trends because most recruitment to New Zealand streams (particularly coastal streams) happens between September–March. Sampling during elevated flows or within 14 days of a bed-moving flood should also be avoided as some fish will seek deeper and or lateral refuge or be more mobile during these periods (Jowett & Richardson 1989, David & Closs 2002, McEwan 2009), influencing their probability of capture at the reach scale.

Timescales and measures of success: Defining success of stream restoration for fish communities could be achieved by:

1. the use of predictive models that allow for site specific predictions and comparisons of the likelihood of species being present
2. an index that has multiple metrics and is calibrated to local conditions
3. directly measuring what is present at reference versus restoration sites.

In this document, we focus primarily on the third method. Direct measurement of a fish community at a reference site allows you to track natural changes and compare this against the community at your restoration site.

Once habitat conditions are suitable for fish, indications of recovery can be expected within 2 years if there are no downstream constraints to colonisation. Full recovery in a best-case scenario (inferred from population size structure and other metrics discussed above) could be possible within 10–15 years (excluding eels, which could take 30–50+ years to re-establish unimpacted population size structure). The potentially rapid recovery response is mainly due to the availability of new recruits arriving annually as part of the whitebait run (note recruitment occurs locally for non-migratory species e.g., Crans bully). If new recruits are known to have arrived (e.g., high number of young-of-the-year fish in sample), but have not established in the restored reach the following year (and are expected to be found there based on similar reference site), then it is possible that local conditions (e.g., water quality, in-stream cover) are not yet suitable to accommodate these species.

As an example, Jowett et al. (2009) studied the fish community response to restoration (planting riparian vegetation and preventing stock access) of pastoral sections in two small streams draining from native forest catchments with unimpeded passage into the Waikato River and about 80 km from the sea. After 10 years, the restoration efforts had more than doubled the numbers of giant kōkopu and redfin bullies, slightly increased numbers of banded kōkopu, and decreased shortfin eel numbers by about 40%. Riparian restoration was most effective for the fish species that use cover and pool habitat.

Figure 5.39 shows the hypothetical example of riparian management of a pasture stream with the additional assumptions of low elevation (0–50 m asl) and less than 5 km from the coast. In this case, eels (red line) are more abundant than the reference initially due to higher productivity caused by minimal shading. We predict that as productivity decreases with canopy closure after 10 years, eel numbers also decrease toward reference condition. Torrentfish numbers (green line) at the restoration site would not change greatly from reference as local physical geology (steep gradient) of the bed is more important for determining their relative abundance in this scenario.

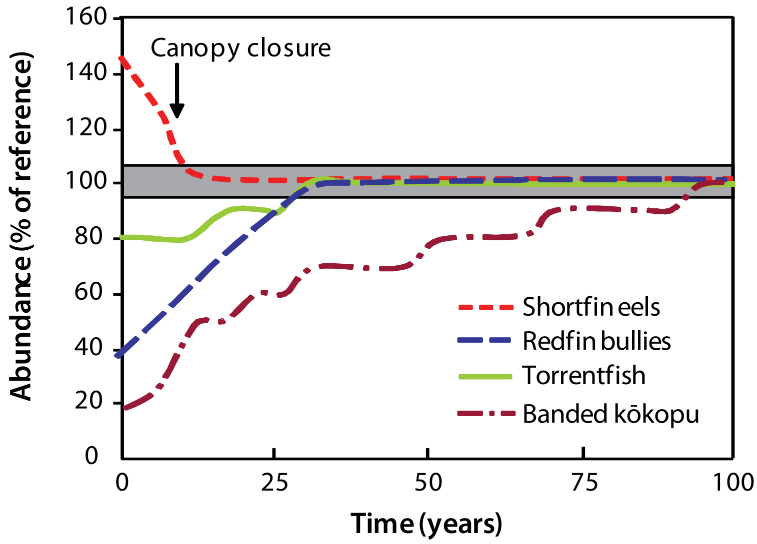


Figure 5.39: *Hypothetical recovery of fish species abundance relative to a reference stream after riparian management.*

Nevertheless, as finer substrates reduce with increasing bank integrity (after 20 years) and stream-bed heterogeneity increases, the available habitat for torrentfish will improve and result in a slight increase in numbers over time. We predict a similar pattern for redfin bullies (blue line), although this species is generally more ubiquitous at the reach scale and, therefore, increased substrate heterogeneity is likely to have a more pronounced effect (increase) on their numbers relative to reference. Banded kōkopu (maroon line) are known to respond to shade and are hypothesised to show an initial rapid increase in response to canopy closure, but numbers then stabilise and increase slightly as more large wood (a favoured habitat type) increases over the longer term.

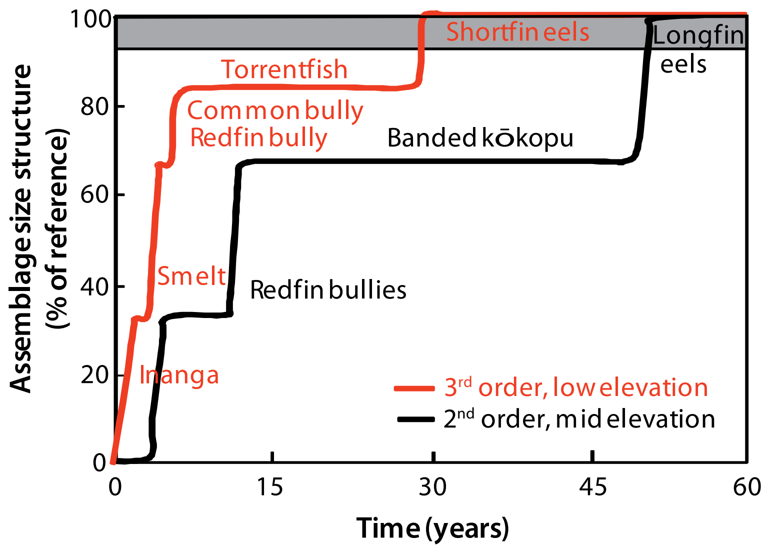


Figure 5.40: *Hypothetical time for fish populations to achieve normal size structure based on maximum fish ages and assuming that habitat is suitable for recolonisation in two coastal stream examples.*

The time for each species to reach normal **size structure** within a restored reach can be estimated by the maximum age of each species (Figure 5.40). In the case of a 3rd order, low elevation, coastal stream, more than 80% of the reference assemblage size structure could be reached within 8 years but it may take 30 or more years to reach 100% because shortfin eels are long-lived relative to other members in the assemblage. In the case of a 2nd order, mid elevation, coastal stream, over 60% of the reference assemblage size structure could be reached within 13 years but it may take over 50 years to reach 100% as longfin eels are very long-lived relative to other members in the assemblage. These estimates assume that the habitat is suitable for recolonisation.

BIRDS

The return of bird species can be an indicator that the riparian buffer zone has developed sufficiently to provide habitat and ecosystem functioning to terrestrial vertebrate species. Below is a description of the standard 5-minute bird count methodology and the survey form from the Department of Conservation (Appendix F). If your restoration site is in an urban area you might like to use the online Garden Bird survey method: **www.landcareresearch.co.nz/research/biocons/gardenbird/index.asp.** *This website also has a bird identification guide.*

Five-minute Bird Count:

- *Identify the location of the count station, either with a marker or GPS coordinates, or both. Record time of day and conditions. Obtain at least three replicates of counts over three consecutive days if possible.*
- *Count stations should be spaced apart so that an individual bird is unlikely to be recorded more than once.*
- *If possible, establish counts in adjacent habitats that are comparable with the restoration site, as a control (initial condition) or reference (target condition).*
- *Record numbers of all birds seen and heard over a timed five minute period. Avoid double counting. Remain stationary at the count station.*

When:

- *Undertake counts in each season (summer, autumn, winter, spring).*
- *Don't undertake counts at start or end of day, or in very windy conditions or heavy rain.*

Terrestrial plant biodiversity and survival of plantings

Goal(s): TB, NH

Background: The successful establishment of riparian vegetation is vital for stream restoration. Most indicator measures rely on or are influenced by the development of shade and other attributes (e.g., provision of microclimate, food, and habitat resources) that comes from tall, woody vegetation. Therefore, the success of riparian plantings is a key measure to include as many of the other indicators and expectations of success are dependent on the ultimate canopy closure above the stream. Successful establishment of plantings can be assessed using techniques such as:

- representative photographs
- survival and growth of planted species
- changes in vegetation composition (biodiversity) and canopy closure.

Depending on the aims and objectives of riparian plantings, representative photographs should be the minimum level of monitoring undertaken, and information should also be collected on the survival of planted species and the degree of riparian canopy closure. Survival and canopy closure are key indicators of the relative success of plantings and the level of development attained at a planted site.

Assessments of changes in vegetation composition over time provide very useful information on rates of vegetation development and changes in species composition and total species richness. There are well-established techniques for the assessment of vegetation composition over time. The most rigorous approaches involve quantitative measurement of marked vegetation plots using standard techniques such as 20 × 20 m plots (forest, Hurst & Allen 2007a, c; grassland, Wiser & Rose 1997). However, 20 × 20 m vegetation plots are often not suitable for narrow riparian margins or take too long to install and measure. (An experienced team of two will only measure 1–2 plots/day.)

A simpler and much quicker method is to use Reconnaissance (Recce) plots (Hurst & Allen 2007b), which can be applied within 20 × 20 m plots or within smaller defined areas. Recce plots involve recording all plant species within defined height tiers and then assigning cover scores to each species within each tier.

Representative photographs (Photopoints)

Photographs can provide an important visual indicator of the success of stream bank planting. Although not a quantitative measure, they can be a great tool for conveying information. Select a range of representative sites with good views across the site, ensuring that views will continue to be available as the vegetation increases in height. Record reference information for each photograph (see example photopoint sheet in Appendix G) .

When: *At least annually at the same time each year.*

Survival and growth of planted species

Good records should be kept of the species planted (including plant grades, numbers, and/or proportions), the plant spacings, and any other relevant information, such as the site preparation undertaken, use of fertiliser (type and amount), pest control undertaken and required, and any other relevant management issues or interventions required. Various techniques can be used to assess survival and growth, including:

- *sample plot (e.g., 5 x 5 m) counts and measurements of species, cover, and heights*
- *walk-through assessments on marked transects, recording numbers of each species, their condition and height*
- *repeated measurements of a sample of marked plants within a site.*

Method:

1. *Select and mark plot locations, transect alignments, or species to be measured through representative parts of the site.*
2. *Record the following: species present, score each plant for condition (1 = excellent to 5 = dead), plant heights.*
3. *Within each sample plot, assess and record overall cover and degree of cover provided by each species.*
4. *Photograph sample plot(s) or transects.*

When: *Immediately after planting, and then at least annually.*

Method: For each predetermined plot location, assess the canopy cover class for each of the vegetation tiers using the standard height tiers and standard cover classes (Table 5.7 & 5.8).

Data sheets are available from the Recce manual (Hurst & Allen 2007b).
See website: nvs.landcareresearch.co.nz/html/NVSmanual.aspx

Table 5.7: *Tier classes used in Reconnaissance (Recce) plot analysis (Hurst & Allen 2007b).*

Tier	Standard tiers used when vegetation is predominantly non-woody	Standard tiers used when vegetation is predominantly woody
1	–	>25 m
2	–	12–25 m

Tier	Standard tiers used when vegetation is predominantly non-woody	Standard tiers used when vegetation is predominantly woody
3	5–12 m	5–12 m
4	2–5 m	2–5 m
5	–	0.3–2 m
5A	1–2 m	–
5B	0.2–1 m	–
6		<0.3 m
6A	0.1–0.3 m	–
6B	<0.1 m	–
7	Epiphytes (at any height)	Epiphytes (at any height)

Table 5.8: *Cover classes applied to species present in each height tier on the Recce vegetation description (source: Hurst & Allen 2007b).*

Cover Class	Percent (%) Canopy Cover
1	<1
2	1–5
3	6–25
4	26–50
5	51–75
6	76–100

A canopy cover scale (Figure 5.41 from Hurst & Allen 2007b) can be used to assess overall cover or the degree of canopy closure on a planting site and the degree of cover provided by each species.

When: Annually for first 5 years then at 5-yearly intervals after that.

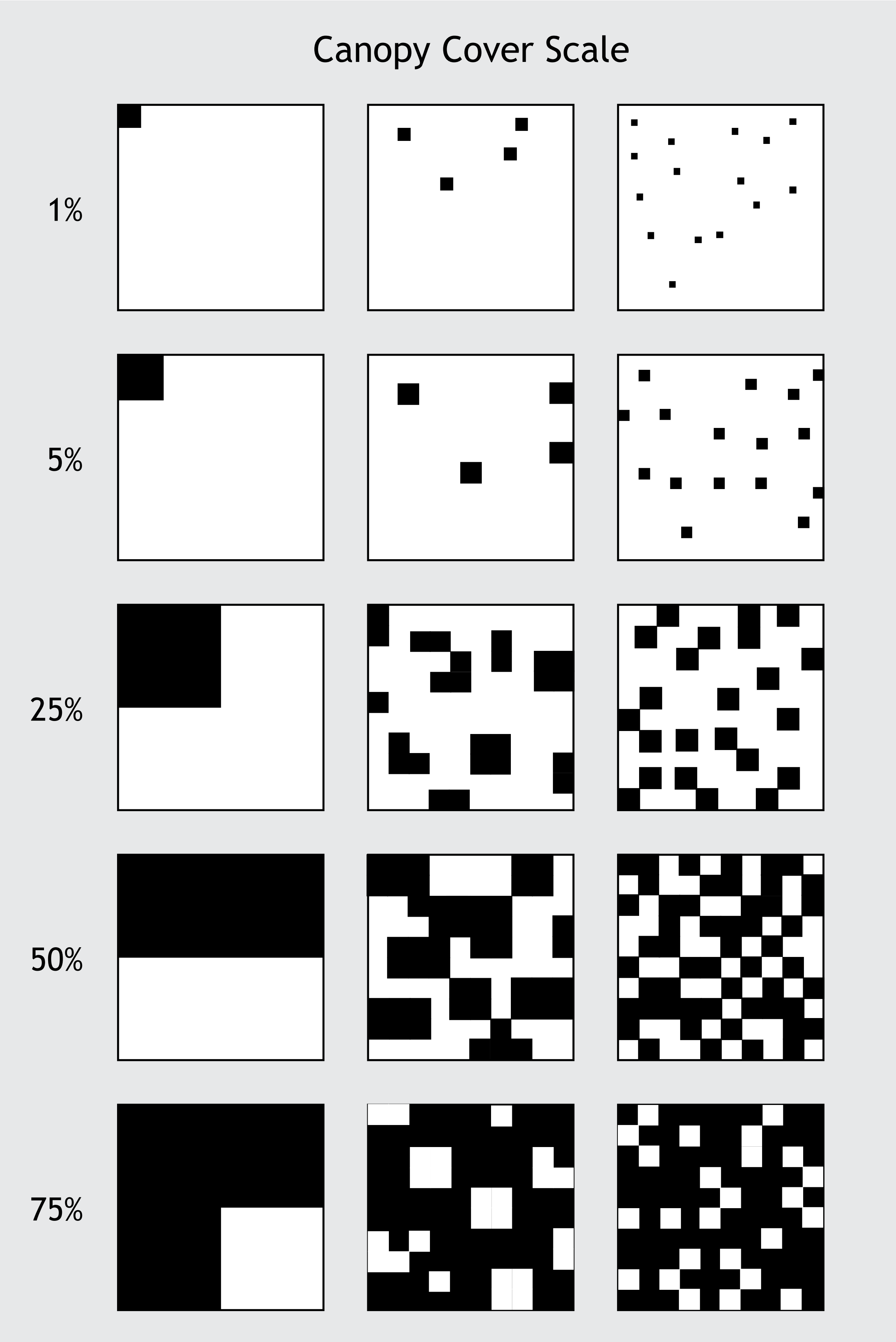


Figure 5.41: *Divisions of the standard cover-abundance scale (showing the proportion of the Recce area represented by each division). Use this scale when assigning cover-classes for the Recce vegetation description.Canopy cover guides (source Hurst & Allen 2007b).*

The Recce method described above can also be used to give an estimate of native plant species richness that can be compared against a reference location. **Biodiversity** could be compared at each tier, to give an indication of natural recruitment and self-sustaining populations. The simplest measure would be total species richness.

Timescales and measures of success:

Canopy closure of plantings in the riparian zone (note: this is not over the stream channel) can be achieved within 5 years if the plants were planted densely and the site is subject to good growing conditions, particularly sufficient rainfall (Figure 5.42). With widely-spaced plantings or in drier conditions, canopy closure may take up to 10 years. The time to reach riparian canopy closure is faster than that of canopy closure above the stream. Riparian canopy closure after 5 years is roughly equivalent to canopy closure above a small stream at 10 years.

Increases in native plant species richness are expected to take much longer to reach a reference state. Planting native species will sharply increase the proportion of native species from that of a pasture riparian area, but natural recolonisation will be slower depending on weed control, soil and microclimate conditions, and bird or wind delivery of seed propagules (Figure 5.43). Because riparian buffer zones are much narrower than typical forest remnants, we expect that total species richness may never reach that of mature primary native forest.

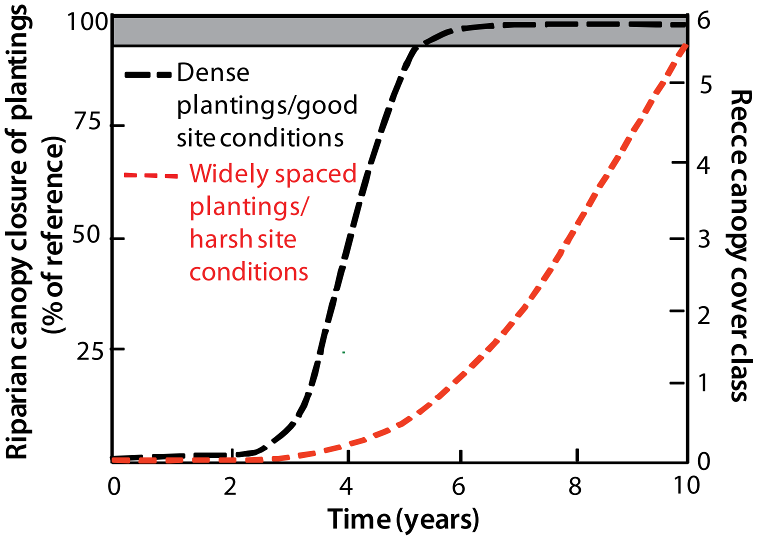


Figure 5.42: *Hypothetical rates of canopy closure of riparian plantings (on banks and not over the stream) over a 10-year period (subject to monitoring, pest control, and other post-planting maintenance). Estimated Recce cover class for a tier of 0.3–2 m.*

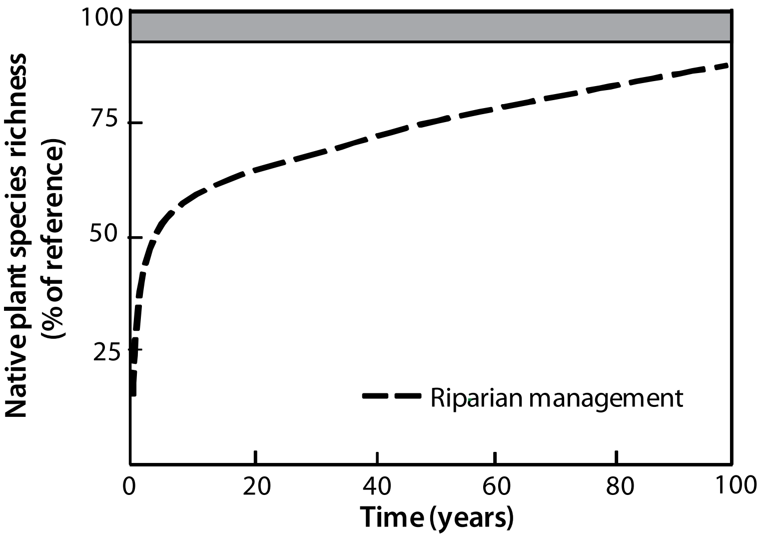


Figure 5.43: *Hypothetical trajectory of native plant species diversity after planting with native species in a 15–20 m buffer alongside a pasture stream.*

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Part six:

References

The Restoration Indicator Toolkit:
Indicators for monitoring the ecological success of stream restoration

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Part seven:

Appendices

The Restoration Indicator Toolkit:
Indicators for monitoring the ecological success of stream restoration

Appendix A: Developing the Toolkit

Setting priorities for indicators

To determine the key priorities for these restoration indicators, we held a workshop at the New Zealand Freshwater Sciences Society conference in November 2006, which was well attended by regional council staff, representatives from many other agencies (DoC, Fish & Game etc.), and scientists involved in stream restoration. At this workshop we surveyed the specific types of restoration occurring in regions, and the perceived need for monitoring tools for each type of restoration (Table A.1). The responses indicated a clear need for monitoring tools almost across the board for all the restoration types we outlined, with particular emphasis on in-stream habitat enhancement, riparian planting, riparian filter strips, and stormwater controls. In developing the Toolkit, we focused on those restoration activities that were the most common and that scored highly in terms of need for monitoring tools, namely: riparian planting, stock exclusion, fish passage enhancement, and bank stabilisation.

Table A.1: *Types of restoration activities and need for monitoring tools across New Zealand (based on 13 responses from a workshop at the New Zealand Freshwater Sciences Society conference in November 2006). The activities that are commonly used and most in need of monitoring tools are highlighted in bold.*

Restoration Activity	***How common?*** Please rank from 1–5 (1 = nil–minimal, 2 = a few, 3 = common, 4 = many, 5 = very frequent)		***Is there a need for monitoring tools for this type?*** (1 = nil–low, 2 = low, 3 = medium, 4 = high, 5 = very high)	
	Most Common Rank	**Average Rank**	**Most Common Rank**	**Average Rank**
Riparian planting	**2**	**2.6**	**4**	**3.9**
In-stream habitat enhancement	1	1.5	5	4.0
Restoration of large-scale gravel extraction	1	1.2	2	2.7
Fish passage enhancement	**3**	**2.3**	**4**	**3.2**
Dam removal	1	1.0	1	2.0
Bank stabilisation/erosion control	**2**	**2.8**	**3**	**3.2**
Stock exclusion	**3**	**3.0**	**4**	**3.5**
Daylighting piped streams	1	1.0	1	1.7

Restoration Activity	***How common?*** Please rank from 1–5 (1 = nil–minimal, 2 = a few, 3 = common, 4 = many, 5 = very frequent)		***Is there a need for monitoring tools for this type?*** (1 = nil–low, 2 = low, 3 = medium, 4 = high, 5 = very high)	
Riparian filter strips	1	1.7	4	3.5
Bridging for stock	1	2.1	1	2.1
Pest management	1	1.8	4	2.8
Stormwater controls	1	1.9	4	3.7
Meanderisation	1	1.1	1	2.3
Flow regime enhancement	1	1.2	4	2.8
Sediment regime enhancement	1	1.3	4	2.9
Reintroduction of biota	1	1.1	5	2.9

We asked the workshop attendees (50–60 people) to give 5 votes to the types of goals that were most important for them to be able to monitor and show the success of their restoration efforts. The top three goals scored well above all others and these were aquatic biodiversity, ecosystem function, and water quality. The remaining goals, in decreasing order of importance, were terrestrial biodiversity, education, health of downstream receiving environments, cultural, aesthetic, fisheries, and recreation. This response provided a clear imperative to focus on indicators relating to aquatic biodiversity, ecosystem function, and water quality.

List of potential indicators

A list of potential indicators was developed and discussed by an expert panel (Table A.2) over several workshops. The project team also sought additional advice and input from experts in their relative fields (Table A.3). An initial list of indicators (Table A.4) was evaluated according to the likelihood of use by regional councils or community groups; the method of measurement, stream size that they were applicable to, and whether there was information available to judge a level of success by trajectory, threshold, or reference endpoint. This working table was used to narrow down the list of indicators. Some indicators were later excluded because they were expensive or difficult to measure, useful for specific types of pollution only, or methods were not sufficiently well developed. Because these indicators are still potentially useful for monitoring restoration, we have included them in Table A.4 along with the expert panel's reasons for their exclusion.

Once the list of indicators had been finalised (Table 2.1), members of the expert panel were assigned indicators in their specialist areas of expertise to develop appropriate protocols for measurement and suggested frequency of monitoring. The protocols were generally taken from established methodology, but in some cases were developed for this project. The expert panel met to discuss and agree on each of the protocols after they were developed.

Table A.2: *Members of the expert panel.*

Name	Organisation	Expertise
Joanne Clapcott	Cawthron Institute	Ecosystem processes
Kevin Collier	Environment Waikato/ The University of Waikato	Stream ecology
Bruno David	Environment Waikato	Fish ecology
Rob Davies-Colley	NIWA	Water quality, stream shade, and geomorphology
Fleur Matheson	NIWA	Aquatic plants and nutrients
Stephanie Parkyn	Freshwater consultant	Stream ecology
John Quinn	NIWA	Stream ecology
William Shaw	Wildland Consultants Ltd	Terrestrial ecology
Richard Storey	NIWA	Stream ecology

Table A.3: *Experts that contributed to the development of the Restoration Indicator Toolkit.*

Name	Organisation	Expertise
Paul Champion	NIWA	Aquatic plants
John Clayton	NIWA	Aquatic plants
John Leathwick	NIWA	Predictive fish models
Juliet Milne	Greater Wellington	Water quality
Ngaire Phillips	NIWA	Invertebrate species traits
Brian Smith	NIWA	Adult insects
Rebecca Stott	NIWA	Faecal indicators
Summer Warr	Greater Wellington	Overview review
Roger Young	Cawthron Institute	Functional indicators

Table A.4: *Working table of draft indicators including those discussed but rejected (shaded and in italics). RC = regional council, CG = community group, Ref = reference site.*

Indicator	Stream size	RC	CG	Trajectory of recovery known	Threshold of success known	Endpoint known	Measure/tool	Likely to use
Bank erosion	All	Y	Y	Y	N	Ref	Visual measure off tape	Y
Bank undercuts	*All*	*Y*	*Y*	*?*	*?*	*Ref*		*N — difficult to measure accurately*
Bank overhanging vegetation	*All*	*Y*	*Y*	*?*	*?*	*Ref*		*N — difficult to measure accurately*
Bedload								*N — too difficult to measure*
Bed particle size	All	Y	Y	? stream site specific	?	Ref	Wolman walk or % fines	Y
Biological micro assay								*N — not developed in NZ*
BOD	*All*	*Y*	*N*	*Y*	*Y*	*Ref*	*BOD bottles*	*Only in organic point pollution*
Community metabolism	*All*	*Y*	*N*	*Y*	*N*	*Ref*	*Chambers*	*N — specialist equipment*
Conductivity	*All*	*Y*	*Y*	*N*	*N*	*Ref*	*Meter in SHMAK*	*N — relatively insensitive for restoration in NZ*
Cotton strip decay	All	Y	Y	Y	N	Ref	Cotton strips	Y

Indicator	Stream size	RC	CG	Trajectory of recovery known	Threshold of success known	Endpoint known	Measure/tool	Likely to use
Denitrification								*N — too difficult to measure*
DO (diurnal range)	All	Y	N	Y	Y	Ref	DO meter spot measures or loggers	Y
Ecosystem metabolism	2nd order+	Y	N	Y	N	Ref	DO logger	*Y in large scale restoration*
Ecotoxicological field/lab assays	*All*	*Y*	*N*	*Y*	*Y survival*	*Ref/survival — but sub-lethal impacts*	*Caged species*	*Y urban/mining — used for finding why species are absent*
Embeddedness	*All*	*Y*	*Y*	*Y*	*?*	*Ref*	*Visual key*	*N — uncertainty in method*
Faecal indicators	All	Y	?	Y	Y	Ref/guide-lines	Petrifilm being developed for CG	Y
Fish (biodiversity, PA, guilds (QIBI))	All	Y	Y	Y?	Y?	Ref	FWENZ, QIBI, spotlight, EF, PA, trapping	Y
Fish size class (recruits)	All	Y	Y	Y	Y — Normal distribution	Ref	Spotlight, EF, trapping	Y
Fish density, biomass	All	Y	N?	Y?	N	Ref	EF, spotlight, trapping, repeated pass	Y
Fish — ratio of exotic fish to native species	All	Y	Y	Y	Y?	Ref	Methods above	Y

Indicator	Stream size	RC	CG	Trajectory of recovery known	Threshold of success known	Endpoint known	Measure/tool	Likely to use
Fisheries mahinga kai	*All*	*Y*	*Y*	*Y*	*Y*	*Ref*	*CPUE fishing data*	*N — not focus of ecological restoration to enhance fisheries*
Leaf fall	*All*	*?*	*?*	*Y*	*Y*	*Ref*	*Litter traps*	*N — Assume that overhead vegetation will provide input*
Light/dark bottle metabolism	*All*	*?*	*N*	*Y*	*?*	*Ref*		*N — unlikely to measure*
Litter retention	All	Y	Y	Y	N	Ref	Leaf analogue	Y
Macroinvertebrates	All	Y	Y	Y	Y	Ref	SHMAK or protocols	Y
Macroinvertebrate species traits, functional groups	*All*	*Y*	*N*	*N*	*N*	*Ref*	*Web tool, data collected as above*	*N — tool still under development, could be used in future*
Macroinvertebrates — ratio of exotic to native inverts	*All*	*Y*	*Y*	*N*	*N*	*Ref*		*N — not enough information*
Macrophyte cover and clogginess	All	Y	Y	Y — bell shaped	?	Ref	Visual + scoring	Y
Macrophyte species diversity (+ mosses, bryophytes)	All	Y	Y	Y related to ref	Y related to ref	Ref	Visual + scoring	Y

Indicator	Stream size	RC	CG	Trajectory of recovery known	Threshold of success known	Endpoint known	Measure/tool	Likely to use
Mega-invertebrates	All	Y	Y	?	Y — P/A	Ref	EF, spotlight, taukowa	Y
Metals	All	Y	N	?	Y	Ref	Water and sediment sample	Y — but specialist, site specific, urban
Mesohabitats	All	Y	Y	Depends on channel reconstruction	N	Ref	Visual	Y
N & P	All	Y	N	Y?	Y	Ref/guidelines	Water sample	Y — but expensive
Organic matter	All	Y	Y	Y	N	Ref	Wolman walk org. or visual%	Y
PAH	All	Y	N	?	Y	Ref	Water sediment samples	Y — but specialist, site specific, urban
Periphyton growth	*All*	*Y*	*Y*	*Y*	*Y*	*Y*	*SHMAK, tiles, natural substrate*	*N — state variable better not confounded by sediment*
Periphyton biomass community structure	All	Y	Y	Y	Y (e.g., no FGA)	Ref	Visual + scoring	Y
Periphyton nutrient bioassay	*All*	*Y*	*N*	*Y*	*N*	*Ref*	*Agar diffusers*	*N — unlikely to measure*

Indicator	Stream size	RC	CG	Trajectory of recovery known	Threshold of success known	Endpoint known	Measure/tool	Likely to use
pH	All	Y	Y	?	?	Ref	Meter or SHMAK litmus	Y — site specific, e.g., mine polltn.
Pool depth	All	Y	Y	Depends on channel reconstruction	?	Ref	Mean depth, residual pool depth	Y
Shade	All	Y	Y	Y	Y	Ref	Light meter, visual est., densiometer, canopy analyser	Y
Stored fines	*All*	*Y*	*N*	*Y*	*N*	*Ref*	*Quorer or method based on settled volume*	*N — expensive, new method needs to be added*
Substrate stability	*All*	*Y*	*Y*	*N*	*N*	*Ref*	*Compactness*	*N — may not be influenced by restoration. Geology governs looseness or armouring of bed.*
Suspended sediment	*All*	*Y*	*N*	*Y*	*Y*	*Ref*	*Water sample*	*N — expensive and surrogates avail*
Targeted threatened species	All	Y	Y	Y	Y	Ref	Depends on species	Y

Indicator	Stream size	RC	CG	Trajectory of recovery known	Threshold of success known	Endpoint known	Measure/tool	Likely to use
Temperature	All	Y	Y	Y	Y	Ref	Temp loggers or spot	Y
Terrestrial invertebrate input	*All*	?	?	?	?	*Ref*	*Flag for research*	*N — hard to get meaningful information, variable. Assume that overhead vegetation will provide input.*
Terrestrial vegetation	All	Y	Y	Y	Y	Ref	Recce plots	Y
Turbidity/clarity	All	Y	Y	Y	Y	Ref	Clarity tube or black disk or turbidity sensor	Y
Water and channel width	All	Y	Y	Y	?	Ref of similar size	Tape	Y
Wood decay	All	Y	Y	Y	N	Ref	Ice-cream sticks	Y

Appendix B: Choosing indicators to match your goals – examples

Scenario 1: Riparian management on a farm stream (regional council)

Activity: Restored stream in pastoral land use
Management method(s): Riparian buffer zone 10 m on either side of stream, fenced and planted with native trees/plants
Catchment context: Drystock farm with sheep and beef access to stream prior to fencing, headwater streams have some remnant forest
Catchment constraints: Some impairment of species dispersal due to pastoral land use
Monitored by: Regional council
Primary Goal(s): NH, WQ, EF, AB
Secondary Goals: N/A

Table B.1: *Goals and indicators to monitor riparian management on a farm stream chosen by a hypothetical regional council (NH = natural habitat, WQ = water quality, EF = ecosystem functioning, AB = aquatic biodiversity, TB = terrestrial biodiversity).*

Goals		Indicator
	NH	Shade of water surface
	NH	Water and channel width
	NH	Stream-bed particle size
	NH	Mesohabitats
	NH	Bank erosion and condition
	NH	Organic matter abundance
	NH	Longitudinal profile variability
	NH	Residual pool depth
	NH	Periphyton
	NH	Macrophyte cover and clogginess
WQ	NH	Water temperature
WQ	NH	Water clarity
WQ		Faecal indicators
WQ		Nutrients

Goals		Indicator
WQ	EF	Dissolved oxygen
	EF	Ecosystem metabolism
	EF	Organic matter processing
	EF	Leaf litter retention
AB		In-stream macrophytes
AB		Benthic macroinvertebrates
AB		Stream mega-invertebrates
AB		Fish
TB	NH	Terrestrial plant biodiversity and survival of plantings

In this hypothetical example, the regional council has decided that natural habitat, water quality, ecosystem function, and aquatic biodiversity are the primary goals of the restoration. They have chosen several indicators to address each goal and have a suitable comparable reference site against which to assess restoration success. They have established that the site does not have a rubbish problem, so have not included that as an indicator of success. The site is unlikely to have contamination from toxicants so they have excluded those indicators that are more commonly associated with urban or point-source contamination. The success of the management method (riparian planting) is dependent on the survival of plantings, so they have decided to monitor the growth of plantings and record canopy cover. They have chosen not to include riparian microclimate at this stage, but they may measure this as a baseline and incorporate in the monitoring at a later date as resources allow.

Scenario 2: Riparian management on a farm stream (community group)

Activity: Restored stream in pastoral land use
Management methods: Riparian buffer zone 10 m on either side of stream, fenced and planted with native trees/plants
Catchment context: Drystock farm with sheep and beef access to stream prior to fencing, headwater streams have some remnant forest
Catchment constraints: Some impairment of species dispersal due to pastoral land use
Monitored by: Stream care group
Primary Goal(s): NH, WQ
Secondary Goals: AB, EF, TB

Table B.2: Goals and indicators to monitor riparian management on a farm stream chosen by a hypothetical community group (NH = natural habitat, WQ = water quality, EF = ecosystem functioning, AB = aquatic biodiversity, TB = terrestrial biodiversity).

Goals		Indicator
	NH	Shade of water surface
	NH	Water and channel width
	NH	Stream-bed particle size
	NH	Mesohabitats
	NH	Bank erosion and condition
	NH	Organic matter abundance
	NH	Longitudinal profile variability
	NH	Periphyton
WQ	NH	Water temperature
WQ	NH	Water clarity
WQ		Faecal indicators
	EF	Organic matter processing
	EF	Leaf litter retention
AB		In-stream macrophytes
AB		Benthic macroinvertebrates
AB		Fish — targeted indicator species
	TB	Terrestrial plant biodiversity and survival of plantings (photopoint survey)

This example has the same scenario of a drystock pasture stream with fencing and planting as the one before, but it is going to be monitored by a stream care group. Their goals are primarily based on a guiding image developed from similar streams around the area, and are focused on returning natural habitat and water quality while hoping that improvements in habitat will bring about increased biodiversity and ecosystem function. They choose to include most of the same measures of natural habitat as the regional council because these are easily measured, but exclude expensive measures such as monitoring nutrients. In-stream community metabolism is technically difficult to measure and dissolved oxygen requires expensive equipment, so these have been excluded. Simpler measures of ecosystem function, such as cotton strip decay and leaf litter retention, have been included. The care group does not have access to electrofishing equipment, so they have chosen to monitor a target fish species by night spotlighting. They are confident that they will be able to monitor aquatic invertebrates, periphyton, and macrophyte biodiversity, given the guides developed for community groups, and will use a photopoint survey to monitor the growth and survival of plantings.

Scenario 3: Fish passage in a native bush catchment

Activity: Fish passage reinstatement
Management method: Adding a fish ladder to a road culvert to enhance passage for climbing species (those present >50 m above sea level)
Catchment context: Native bush catchment, upland (>50 m asl), close to the sea
Catchment constraint: None identified
Monitored by: Regional council
Primary Goal(s): AB (Native fish)

Table B.3: Goals and indicators to monitor fish passage enhancement (AB = aquatic biodiversity).

Goals	Indicator
AB	Fish

Because this fish pass is built in a native bush stream, no change in the natural habitat is expected. The focus is entirely on aquatic biodiversity and, in particular, restoring the native fish populations. The regional council will monitor all the fish metrics described in Part 5, including the ratio of exotic fish to native species in case the fish ladder inadvertently allows access to unwanted exotic species (restoration should do no lasting harm). If the site were not in shaded native bush, then periphyton would also be monitored in case of top-down changes to the base of the food web (i.e., in case increased fish predation reduces invertebrate grazing of periphyton leading to increased periphyton biomass). An adaptive management solution in that case would be to plant shade trees.

Scenario 4: Willow removal along a pasture stream

Activity: Willow removal
Management method: Following the best practice manual of Environment Waikato – willow removal prior to replanting, some in-stream log placement, fencing, then replanting with native plants
Catchment context: 3rd order pasture stream; 1 km section of stream having willows removed because they were choking stream flow. Native trees planted for improved natural habitat and for stream flow.
Catchment constraint: Large area of upstream catchment in pasture with no native forest
Monitored by: Regional council
Primary Goal(s): Water conveyance (WC)†, NH
Secondary Goals: AB, EF

Table B.4: Goals and indicators to monitor willow removal and native replanting on a farm stream (NH = natural habitat, EF = ecosystem functioning, AB = aquatic biodiversity, TB = terrestrial biodiversity, WC† = water conveyance).

Goals		Indicator
(WC)†	NH	Continuous flow recording*
(WC)†	NH	Longitudinal profile variability
(WC)†	NH	Residual pool depth
(WC)†	NH	Water and channel width
	NH	Mesohabitats
	NH	Bank erosion and condition
	NH	Stream-bed particle size
	NH	Shade of water surface
TB	NH	Terrestrial plant biodiversity and survival of plantings
	NH	Organic matter abundance
	NH	Water temperature
	NH	Water clarity
	EF	Dissolved oxygen
	EF	Ecosystem metabolism
	EF	Organic matter processing
	EF	Leaf litter retention
AB		Periphyton
AB		In-stream macrophytes
AB		Benthic macroinvertebrates

†Goals that are management focused rather than ecological, and therefore, indicators have not been developed as part of the toolkit.

*Examples of indicators that could be used to address the management focused goal whose descriptions are not included in the toolkit.

In this example of a 3rd order stream receiving 1 km of willow removal and replanting, most changes are expected to occur in the stream bed and banks. Water chemistry is not likely to change and improving water quality was not a goal of the restoration. Therefore, nutrients and faecal indicators have not been included by the regional council. Water temperature and clarity may decline during the willow-removal/replanting phase of the restoration, so these need to be monitored to ensure that the functions that were maintained by the willows are ultimately restored. Dissolved oxygen would be expected to change with better stream flow. Many of the flow-related natural habitat indicators can be used to assess the goal of improved water conveyance, but continuous flow recording could also be added.

This restoration involves a considerable phase change from shaded to open to shaded, so monitoring of community metabolism and biota that are influenced by light (periphyton and macrophytes) is included. Aquatic macroinvertebrates have been added as an indicator of biodiversity improvements, but not fish in this instance, as aquatic biodiversity is a secondary goal and the resources required to assess fish communities are greater. However, because the removal of willows and damage to bank habitat could impact fish populations, if rare fish species are known to be present in the catchment (e.g., giant kōkopu), then we would advise monitoring of fish species.

Scenario 5: Channel reconstruction of an urban stream

Activity: Reconfigure straightened channel in an urban park

Management method: Reinstating meanders and adding wood and large inorganic substrate to create channel complexity, particularly for fish. Limited amount of revegetation, mainly for stabilisation, stream remains open to maintain water views.

Catchment context: Small urban stream. Fish passage downstream is unimpeded but upstream impervious area may constrain some ecological outcomes.

Catchment constraint: Urban land use has hydrological and potential contaminant effects beyond control of restoration

Monitored by: Regional council

Primary Goal(s): NH, AB (Native fish)

Secondary Goals: A†

Table B.5: Goals and indicators to monitor riparian management on an urban stream (NH = natural habitat, AB = aquatic biodiversity, A† = aesthetics).

Goals		Indicator
	NH	Sinuosity*
	NH	Success of channel installations*
	NH	Cross-sectional shape*
	NH	Longitudinal profile variability
	NH	Water and channel width

Goals		Indicator
	NH	Stream-bed particle size
		Organic matter abundance
A		Mesohabitats
A		Bank erosion and condition
A	NH	Residual pool depth
A	NH	Terrestrial plant biodiversity and survival of plantings
A	NH	Water clarity
A	NH	Rubbish
A	NH	In-stream macrophytes
A	AB	Periphyton
	AB	Fish

†Goals that are management focused rather than ecological, and therefore, indicators have not been developed as part of the toolkit.

*Examples of indicators that could be used to address the management focused goal but whose descriptions are not included in the toolkit.

The important indicators to measure in this example are those that achieve natural habitat and provide refugia from stormwater hydrology for some native fish species. Some of these indicators can also be used to measure the secondary aesthetic goals in relation to channel reconstruction. Additional specific measures of sinuosity, channel cross-section (see SHAP, Harding et al. 2009), and monitoring the stability of management structures have been added in this instance. Because there is limited revegetation in the restoration plan, indicators such as shade and water temperature are not included. Wood has been placed within the stream to enhance fish communities, so the full range of fish metrics will be measured. Aquatic invertebrate community metrics are not included because stormwater quality issues are not being addressed (a catchment constraint). The management actions will not address water quality, so toxicants and nutrients are not included.

However, almost all urban restoration sites will be exposed to stormwater inputs unless the stream is on the urban fringe or the stream is not strongly connected to the stormwater system. Therefore, if the physical restoration measures are not working, then indicators of toxicants (metals, PAHs), water temperature, benthic macroinvertebrates, or continuous flow monitoring could be added. The regional council will take baseline measures of these indicators in case they are added to the monitoring programme in the future.

Scenario 6: Restoring stream flow below a dam

Activity: Restoring minimum flows below a dam

Management method: Release more water from base of dam to reinstate flushing flows and imitate a more natural flow regime including base flow increases

Catchment context: Montane forest on volcanic geology. The release water comes from the base of a dam in a geothermally influenced area. No impediments to fish passage are present downstream. The stream is visited by recreational trampers and kayakers. Rare blue duck (whio) are present.

Monitored by: Department of Conservation

Primary Goal(s): AB, NH

Secondary Goals: A†, R†

Table B.6: Goals and indicators to monitor restoration of minimum flows (NH = natural habitat, WQ = water quality, AB = aquatic biodiversity, A† = aesthetics, R† = recreation).

Goals		Indicator
WQ	NH	Water temperature
WQ	NH	Dissolved oxygen
WQ	NH	Toxicants (specifically arsenic)
WQ	NH	pH
	NH	Continuous flow monitoring*
	NH	Cross-sectional shape*
	NH	Longitudinal profile variability
	NH	Water and channel width
	NH	Stream-bed particle size
	NH	Organic matter abundance
A†	NH	Mesohabitats
A†	NH	Bank erosion and condition
A†	NH	Residual pool depth
A†, R†	NH	Water clarity
A†, R†	NH	In-stream macrophytes
A†, R†	AB	Periphyton
	AB	Targeted threatened species (blue duck)*

Goals		Indicator
	AB	Stream mega-invertebrates
	AB	Benthic macroinvertebrates
	AB	Fish

†Goals that are management focused rather than ecological, and therefore, indicators have not been developed as part of the toolkit.

*Examples of indicators that could be used to address the management focused goal but whose descriptions are not included in the toolkit.

The aim of this restoration is to improve habitat for aquatic species including the rare blue duck, so a full complement of aquatic biodiversity measures has been selected. Specialist information on surveying blue duck (not provided here) has been included as an indicator by DoC. The geothermal influence could mean that the water is low in oxygen and contains metals, so these indicators should be monitored in case of adverse effects. The habitat indicators chosen are those that would be most influenced by changes in flow. Additional measures of channel cross-section shape and continuous flow monitoring have also been added (see SHAP, Harding et al. 2009). Because the area is naturally in forest and this will not change, shade has not been included as an indicator. Water temperature, however, could be influenced by the release of dammed water.

Appendix C: Datasheet for periphyton rapid assessment

Stream: ______________________________ Date: ______________

Thickness category	Colour category	A	B	C	D	E	Mean cover
Thin mat/film (<0.5 mm thick)	NA						
Medium mat (0.5–3 mm thick)	Green						
	Light brown						
	Black/dark brown						
Thick mat (>3 mm thick)	Green/light brown						
	Black/dark brown						
Short filaments (≤2 mm long)	Green						
	Brown/reddish						
Long filaments (>2 cm long)	Green						
	Brown/reddish						
Submerged bryophytes	NA						
Iron bacteria growths	NA						

Bryophytes and iron bacterial growths are recorded here for convenience (NA = not applicable)

Notes: ______________________________

Appendix D: Datasheet for macrophyte rapid assessment

Stream: ______________________________________ Date: ____________________

<table>
<tr><th rowspan="4">Transect</th><th colspan="8">Vegetation cover (% wetted area)</th></tr>
<tr><th rowspan="3">Total cover</th><th colspan="5">Submerged plants</th><th colspan="2">Emergent plants</th></tr>
<tr><th rowspan="2">Total submerged</th><th colspan="2">Surface-reaching</th><th colspan="2">Below surface</th><th colspan="2"></th></tr>
<tr><th>Sub-total</th><th>Species[§]</th><th>Sub-total</th><th>Species[§]</th><th>Total emergent</th><th>Species[§]</th></tr>
<tr><td>1</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>2</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>3</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
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<tr><td>5</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
</table>

[§] Use codes for species names

Notes: ____

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Appendix E: Fish sampling for wadeable streams

Detailed protocols for sampling fish by electrofishing and spotlighting (including data forms) have been developed by Environment Waikato (David & Hamer 2010) and are available at **www.ew.govt.nz/Publications/Technical-Reports/TR-201009/**. Comparison of the two methods is included here.

Spotlighting vs. electrofishing

Choice of method will depend on the suitability of the site and availability of equipment. Occasionally the use of both methods may be valid. Comparing sites where the two methods have been used suggests that there are some consistent species differences with regards to detection, such as:

1. Eels – electrofishing tends to detect higher numbers than spotlighting (particularly smaller eels). Note: detection of eels with either method appears to decline rapidly once water temperatures fall below 12 ^{o}C.
2. Banded kōkopu – electrofishing tends to detect lower numbers than spotlighting (particularly fish <70 mm).
3. Redfin bullies – electrofishing tends to detect higher numbers than spotlighting (results more similar when riffles less abundant).
4. Kōura – electrofishing tends to detect higher numbers than spotlighting (results more similar in fishless streams).
5. Trout – similar numbers spotlighting vs. electrofishing in small wadeable rivers.

Advantages of spotlighting

- non invasive
- rapid (approximately 4–6 times faster than electrofishing)
- not affected by salinity or conductivity
- works well in deep pools providing good water clarity
- requires only 2 people
- minimal equipment required.

Disadvantages of spotlighting

- not effective in streams with abundant riffles (suggest electrofishing if riffle habitat >50%)
- capturing fish may be more time consuming relative to electric fishing
- not effective in turbid conditions
- conducted outside normal working hours
- identification of species may be more difficult without experience.

Appendix F: Five-minute bird count standard data field form

Observer		**Date**		**General Location**		**Specific Location**	

Line number					
Station number					
Start time (24 hour)					
Temperature (1—6)					
Grid ref — Easting, 7 digit					
Grid ref — Northing, 7 digit					
Wind (0—3)					
Other noise (0—2)					
Sun (minutes)					
Precip type (N,M,R,H,S)					
Precip value (0—5)					

Species	Seen	Heard	Seen	Heard	Seen	Heard	Seen	Heard	Seen	Heard

Sun (0—5)
Record approximate duration, in minutes, of bright sun on the canopy immediately overhead

Seen and Heard
Birds that are first heard should be entered under **H** (even if they are later seen); birds that are first seen should be entered under **S**. Adding H and S should give the total number of birds observed

Time 24 hour clock, at the beginning of each count

Unbounded Counts are unbounded

Temperature
1 freezing <0°C
2 cold 0—5°C
3 cool 6—10°C
4 mild 11—15°C
5 warm 16—22°C
6 hot >22°C

Wind
The average for each five-minute count on a modified Beaufort scale:
0 Leaves still or move without noise (Beaufort 0 and 1)
1 Leaves rustle (Beaufort 2)
2 Leaves and branches in constant motion (Beaufort 3 and 4)
3 Branches or trees sway (Beaufort 5, 6 and 7)

Other Noise
i.e.**,** Other than wind, the average for the five minutes
0 not important
1 moderate
2 loud

Precipitation type
Average for each count
N None
M Mist
R Rain
H Hail
S Snow

Precipitation value
0 None
1 Dripping foliage
2 Drizzle
3 Light
4 Moderate
5 Heavy

Appendix G: Photopoint record sheet

Photopoint record sheet

Site Name and No. ____

Date:		Recorder:		Photographer:	
GPS co-ordinates:		Grid ref:			
Date established:		Aerial photo no:			
Camera details:					
Photo no.				Light:	
Time:		Compass bearing (mag):		Altitude:	
Route to photopoint:					
Vegetation description (photographed site):					
Notes:					
Sketch-map of photopoint location:					

The Restoration Indicator Toolkit

This book provides a range of indicators for monitoring improvement in stream condition after restoration. Stream restoration is a key activity for enhancing water quality, ecosystem function, and aquatic biodiversity. Often there is insufficient monitoring of the ecological success of restoration projects, in part because of confusion over what indicators should be monitored and how to measure them. Monitoring is vital for adaptive management of restored streams and in order to judge success it needs to be targeted to project goals. This Toolkit provides the necessary steps and tools to effectively monitor the ecological success of stream restoration.

The Toolkit provides guidance on:

- *designing a restoration monitoring programme*
- *choosing indicators to match project goals*
- *using appropriate methods and timeframes for monitoring the indicators*
- *understanding trajectories of improvement and when to expect success.*

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