Functional indicators of river ecosystem health – an interim guide for use in New Zealand

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Cover Photo: The lower reaches of the Motueka River – a healthy system? Roger Young, 2003

EXECUTIVE SUMMARY

1. A critical part of improving river health is accurate assessment of the current ecological state of river ecosystems so that causes of poor health, or the success of rehabilitation efforts, can be measured. River health monitoring, which has traditionally concentrated on the use of *structural measurements* (such as water quality or taxonomic composition of aquatic organisms), should be complemented in future by *functional indicators*, such as rates of primary productivity and community respiration or organic matter decomposition, to provide a more complete and accurate assessment of the state of these environments.

2. Of the various functional measures available, we have chosen to focus on two that are relatively straightforward to estimate and which describe fundamental aspects of ecosystem functional health, namely *leaf litter decomposition* and *river metabolism*. Data gathered in New Zealand and overseas indicate that both indicators show considerable differences between impacted and unimpacted sites and thus have potential to act as good indicators of ecosystem health.

3. *Leaf breakdown* is potentially an ideal measure because it links the characteristics of riparian vegetation with the activity of aquatic invertebrates and microbial organisms, and is affected by natural and human-induced variation in a wide range of environmental factors. Measurement of leaf breakdown is also relatively simple and has modest equipment requirements.

4. *Ecosystem metabolism* – the combination of primary productivity and ecosystem respiration – is a measure of how much organic carbon is produced and consumed in river ecosystems. Therefore it provides a direct estimate of the food base of river ecosystems and what determines their life supporting capacity.

5. We review the scientific literature relating to both of these functional health measures, discuss the factors that control their rates (such as position in the river hierarchy, stream bed characteristics, stream chemistry, riparian vegetation) and pay particular attention to the manner in which human-induced environmental stressors affect them. We suggest how measurements should be made and weigh up the advantages and disadvantages of each of the indicators. We also describe simple case studies to demonstrate the value of the two approaches.

6. We also suggest some preliminary criteria that could be used to interpret results from these indicators. We see these criteria as only a starting point. Information collected during the regional case studies carried out in the 2nd year of this project will be used to refine the criteria and provide the basis for improved monitoring of river health in the future.

TABLE OF CONTENTS

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GLOSSARY

Biomonitoring – ecosystem health monitoring using measurements of biological community composition or functioning.

Diffusion – the movement of a compound. In this report it refers to movement of oxygen between water and the atmosphere.

Ecosystem function – the rate of a specific process performed within an ecosystem (e.g. the rate of primary production).

Ecosystem metabolism – the combination of primary productivity and ecosystem respiration. Measures the balance between production and consumption of organic matter in an ecosystem.

Ecosystem respiration (ER) – the rate of oxygen uptake due to respiration of all the living organisms within an ecosystem.

Ecosystem structure – a description of the physical, chemical and biological composition of an ecosystem.

ER – ecosystem respiration.

Functional indicator – a measure of the rate of an ecosystem process that can be used to indicate ecosystem health.

GPP – gross primary production.

Heterogeneity – a measure of dissimilarity or variability (e.g. river with a mixture of large boulders, cobbles, fine gravel and sand has high substrate heterogeneity, while a river with just fine gravel has low substrate heterogeneity).

Hyporheic zone – a region of groundwater influenced by a nearby stream or river.

Invertebrates – animals with no backbone. Aquatic invertebrates include insects, molluscs, worms, crustaceans, sponges and spiders.

Keystone species – a species that has a major impact on an ecosystem. The impact is disproportionately large compared to their abundance.

Leaf processing – another term used for leaf breakdown, leaf decay and leaf decomposition.

Macrophytes – large aquatic plants.

MCI – macroinvertebrate community index, a biotic index commonly used in New Zealand to assess stream health. Taxa that are sensitive to pollution have high scores, while tolerant taxa have low scores.

Order – a classification system used to describe the position of a site within a river catchment. First order streams have no tributaries. Second order streams are the result of two first order streams joining, etc.

P:R – the ratio of gross primary production to ecosystem respiration.

Penetrometer – an instrument used to measure leaf toughness.

Primary production – the rate of photosynthesis. This is measured as either gross primary production (GPP), which is the total amount of photosynthesis, or net primary production, which is just the fraction used for growth.

Reaeration coefficient – a measure of the likelihood of a gas to move between the water and the atmosphere (e.g. a shallow turbulent stream will have a high reaeration coefficient, while a slow deep river will have a low reaeration coefficient).

Reference sites – sites that are not impacted by a particular disturbance and can be used to compare with sites that are impacted.

Riparian vegetation – vegetation growing along the banks of rivers and streams.

Structural indicator – a measure of patterns in the physical, chemical or biological composition of an ecosystem that are indicative of health.

Taxon richness – the total number of types of organisms found at a site.

1.   INTRODUCTION

Over the last century rivers throughout the world have come under increasing pressure from human activities, often affecting their ecological integrity or health. Awareness of this problem continues to grow among the general public. Managers of rivers and their catchments need to stop, or reverse, this trend in declining health. A critical part of improving river health is being able to accurately assess the current ecological state of river ecosystems so the causes of poor health, or the success of rehabilitation efforts, can be measured.

Sustainable river management aims to ensure that rivers should *function* in the same way as they do in unimpacted catchments, as well as supporting a similar range and abundance of organisms (i.e. have similar *structure*). River health monitoring has traditionally concentrated on the use of structural measurements like water quality, the composition of stream invertebrate communities and to a lesser extent the biomass of algae, macrophytes and fish to infer the health or integrity of river ecosystems (Boulton 1999). Functional indicators, such as the rates of primary productivity and organic matter decomposition, have been used only rarely in the past. Recent advances in technology make measurement of these functional indicators more simple, thus enabling them to be used alongside traditional approaches to biomonitoring. This will be a significant step forward since functional indicators provide direct measurements of the functions that river ecosystems perform and thus provide an alternative, but complementary, view of ecosystem health (Bunn 1995; Gessner & Chauvet 2002; Brooks et al. 2002).

This report is the first output from a three year project funded by the Minister of the Environment's Sustainable Management Fund and other stakeholders that aims to provide a framework for the use of functional indicators for assessing river ecosystem health in New Zealand. The report provides an overview of functional indicators, brings together information on the response of these functional indicators to a variety of impact types, and provides guidance on when and where these approaches may improve biomonitoring in New Zealand. We also provide detailed information on the methods that can be used to measure these indicators and a summary of the advantages and disadvantages of the approach. This report is primarily aimed at staff in regional councils and unitary authorities who have responsibility for environmental monitoring and management. However, people from other groups that are concerned with sustainable management of freshwater ecosystems will also potentially benefit from this information by having access to a wider variety of tools for assessing the health of rivers and streams. Feedback on the practicality of the approach is requested, along with suggestions for sites that could be used in regional case studies in the second and third years of the project. Information from this report, lessons learnt during the case studies, and proposed criteria for distinguishing healthy and unhealthy systems will be incorporated in a final report produced near the end of the project.

2.   OVERVIEW OF FUNCTIONAL INDICATORS

2.1   What are functional indicators?

Functional indicators are measures of the rate, or relative importance, of a particular process happening in an ecosystem, while *structural indicators* focus on patterns of abiotic resources or biological community composition (Matthews 1982). In other

words, *functional indicators* measure the services or functions provided by ecosystems, while *structural indicators* measure what lives in an ecosystem. As an analogy, the health of pasture on a farm (or a lawn) could be measured by looking at the species of grass growing there, the nutrient concentrations in the soil, and perhaps even the types of worms in the soil. All of these are examples of structural measurements. However, another good indicator of pasture health is the rate at which it grows – a functional measurement. Two contrasting sites may have the same types of grass, worms and nutrient concentrations but pasture growth may be much faster at one site than the other because of a difference in some causal factor such as rainfall or temperature. This difference between sites would be missed if only structural features were measured in each pasture.

There is considerable debate in the scientific literature about whether the numbers and types of species present (i.e. ecosystem structure) reflect the functioning of an ecosystem, or whether ecosystem structure controls ecosystem function (Naeem et al. 1994; Tilman 1999; Loreau et al. 2001; Cardinale et al. 2002). Either way it is clear that structural and functional components of river ecosystems are intricately linked and describe different aspects of the same entity. The number and types of organisms present at a site are dependent on functional processes. For example, mayflies that graze on algae will be uncommon if primary production is very low. In fact, studies have indicated that sites with high primary productivity have longer food chains than equivalent sites with lower productivity (Townsend et al. 1997). However, functional processes are also dependent on the types of species present. For example, leaf litter decomposition may be slow if leaf-eating invertebrates like koura (crayfish) are absent (Usio & Townsend 2001). Ideally, both types of measurements should be made to get a complete picture of ecosystem health (Bunn & Davies 2000; Gessner & Chauvet 2002).

2.2   Why should functional indicators be included in routine biomonitoring?

The inclusion of functional indicators in regular biomonitoring may result in a variety of benefits. Firstly, an assessment of both the structural and functional components of an ecosystem gives *a broad, more complete picture of ecosystem health*. As Matthews et al. (1982) pointed out, an ecosystem can respond in three different ways to a human induced stressor:

- changes to ecosystem structure without changes in functional parameters, e.g. Nelson (2000) showed that macroinvertebrate community structure downstream of a source of metal pollution was quite different from that upstream, while differences in leaf breakdown rates above and below the source of pollution were not detected

- changes to ecosystem function without structural changes e.g. Bunn & Davies (2000) found that measurements of ecosystem metabolism responded strongly to differences in levels of nitrogen and turbidity among a group of small forested streams in southwestern Australia. However, assessments of invertebrate community composition at the same sites using the AusRivAS protocol (Norris & Hawkins 2000) were not related to spatial differences in water quality.

- changes to both structural and functional components of an ecosystem, e.g. Huryn et al. (2002) found differences in both leaf litter breakdown rates and leaf-eating

invertebrate biomass and diversity among 17 streams with contrasting land use in Maine, U.S.A.

A second advantage of including functional indicators in regular biomonitoring is the fact that these measurements provide a *truly integrated measure of stream health* because 1) they are affected by such a wide range of biotic and abiotic controlling variables, 2) they integrate environmental conditions over a moderate time period, and 3) they integrate across a variety of habitat types in a reach. For example, leaf litter breakdown is influenced by water temperature, nutrient concentrations, pollutants, flow fluctuations, acidity and involves bacteria, fungi, invertebrates and, in the tropics, fish. Leaf bags are typically deployed at sites for weeks to months at a time and therefore are exposed to the variety of conditions experienced at that site over the period of study. Measurements of ecosystem metabolism are usually made at a reach scale and therefore incorporate rates of primary productivity and ecosystem respiration in all the riffles, runs, pools and back-waters in a reach, and even include the processes occurring in the hyporheic zone beneath the river bed (Mulholland et al. 1997; Naegeli & Uehlinger 1997).

Functional indicators also allow flexibility in the type of habitats that can be assessed. For example, measurements of ecosystem metabolism are relatively easy in *large rivers* where traditional sampling of invertebrate communities is either dangerous or impossible. Although further studies are required, it appears that results from functional indicators may not be so heavily influenced by differences in geomorphology and substrate characteristics among sites, a problem which often confounds interpretation of invertebrate community composition data. It may even be possible to monitor ecosystem health between sites in transition zones, like the *tidal reaches* of lowland rivers, where again interpretations of invertebrate community composition results are often problematic and confounded by natural differences in species composition and abundance between sites.

Another advantage of functional indicators is that the ecosystem processes that are measured do not rely on a specific set of species, so comparisons can easily be made among *regions with different biogeography*. This is particularly advantageous in large land masses like Australia, U.S.A and Europe where there are major regional differences in community composition as a result of both current climatic differences and historical patterns of species evolution and dispersal (Bunn & Davies 2000; Gessner & Chauvet 2002). However, even in New Zealand there is significant regional endemicity, with some species commonly found in some areas but absent elsewhere (Boothroyd 2000).

Functional indicators may also be *more cost-effective and less time consuming* than some traditional structural approaches since taxonomic expertise for assessing community composition is not required. In addition, several researchers have reported *lower variability and higher sensitivity* for functional indicators than for structural indicators such as biomass or taxon richness (Crossey & LaPointe 1988; Niemi et al. 1993). This suggests that fewer replicate samples may be required to detect specified levels of difference from the mean (Niemi et al. 1993).

Gessner & Chauvet (2002) have also pointed out that functional measurements are usually *directly affected by the stressor* and therefore it is relatively easy to trace any concerns back to the likely cause of the problem. Traditional structural assessments of ecosystem health have tried to adopt a similar approach by splitting invertebrates into

functional feeding groups (i.e. algal grazers, predators, collectors, filterers) to infer the likely cause of changes to ecosystem health (Barbour et al. 1999). For example, an increase in the proportion of algal grazing invertebrates at a site suggests that algal growth has been stimulated in some way. However, direct measurements of algal productivity would provide a better indicator of whether algal growth has been altered. In addition, it is possible to use the measured relationship between light availability and production rates to assess whether light or some other variable, such as nutrient availability, is controlling algal growth (see Section 4.2 below for further details).

2.3 What types of functional indicators are likely to be most useful?

There are a variety of ecosystem processes that could potentially be used as indicators of river ecosystem health. These include rates of nutrient uptake (Sabater et al. 2000), benthic microbial respiration (Niyogi et al. 2001; Hill et al. 2002), nitrification (Bernhardt et al. 2002), fine particulate organic matter export (Wallace et al. 1996), invertebrate production (Buffagni & Comin 2000) and fish production. However, some of these involve large effort or sophisticated and expensive techniques. Rates of leaf litter decomposition (Young et al. 1994; Gessner & Chauvet 2002) and ecosystem metabolism (the combination of algal productivity and ecosystem respiration; Hornberger et al. 1977; Hickey 1985; Hill et al. 1997; Young & Huryn 1999; Bunn & Davies 2000) appear to have the most promise. All river and stream ecosystems are fuelled by a combination of terrestrially-derived organic material (including leaf litter) and algal material produced in-stream. Thus, measurements of the rate of leaf litter decay and ecosystem metabolism provide an indication of the food-base of the ecosystem and thus help determine its life-supporting capacity.

Existing information gathered in New Zealand and overseas indicates that both of these measurements show considerable differences between impacted and more natural sites and thus have potential to act as good indicators of ecosystem health. The following two sections provide:
- detailed information on typical techniques used for measuring both of these indicators
- options for data analysis
- advantages and disadvantages of using each indicator
- a summary of the environmental factors that influence them and likely responses to stressors
- a detailed review of the scientific literature on factors controlling these indicators

3. LEAF LITTER DECOMPOSITION

The rate of decomposition of terrestrial plant leaves in streams and rivers has been suggested for some time as an integrated measure of the effects of human disturbance (Webster & Benfield 1986). Leaf breakdown is potentially an ideal measure because it links the characteristics of riparian vegetation with the activity of invertebrates and microbial organisms, and is affected by natural and human-induced variation in a wide range of environmental factors. Measurement of leaf breakdown is also relatively simple and requires only a minimal amount of equipment.

3.1 Background on measurement techniques

3.1.1 Leaf bags

Leaf breakdown is simply measured by securing bunches of pre-weighed leaves to the stream bed at the sampling site and retrieving them after a certain period. Bunches of leaves are either tied together to form 'leaf packs' or contained within mesh bags ('leaf bags') (Boulton & Boon 1991). We recommend using leaves enclosed within mesh bags because they are easy to make and enable standardisation across sites and among different studies. Leaf bags also avoid abrasion-induced mass losses of large leaf fragments that have not been totally decomposed (Meyer 1980; Hicks & Laboyrie 1999). Their main potential disadvantages are underestimation of processing rates due to exclusion of macroinvertebrates, the potential for anoxic conditions to develop in the center of the bag (Boulton & Boon 1991), and the tendency to encourage accumulation of fine sediments within the bag (Dangles et al. 2001). However, these issues can be largely avoided by using appropriate mesh sizes, relatively small leaf bags, and by exposing leaf bags for relatively short periods of time (Boulton & Boon 1991).

If leaf bags with a single mesh size are to be used, we recommend using coarse-mesh bags (0.5 – 1 cm aperture) because they allow colonization by leaf-eating macroinvertebrates and thus simulate natural leaf breakdown more closely than fine-mesh bags (see e.g. Chergui & Pattee 1988; Stewart & Davies 1989; Gonzalez et al. 1998; Gessner & Chauvet 2002; Menendez et al. 2003). In addition, Webster et al. (2001) showed (using a computer simulation model to predict leaf standing stocks in a stream) that leaf breakdown rates determined with coarse-mesh bags were good estimates of the rates at which leaf material was actually disappearing from this stream.

If it is feasible to use more than one mesh size, then both coarse-mesh and fine-mesh bags (≤ 1 mm) should be used (see e.g. Gonzalez et al. 1998; Menendez et al. 2003; Pascoal et al. 2003) because this approach allows the estimation of relative contributions of macroinvertebrates and microorganisms (fungi and bacteria) to leaf breakdown rates (Gessner & Chauvet 2002). We advocate the use of both coarse-mesh and fine-mesh bags in trials to determine the usefulness of this approach in New Zealand streams, where leaf eating invertebrates are generally thought to be less important than elsewhere (see Young et al. 1994; Niyogi et al. 2003).

3.1.2 Choice of leaf species and pretreatment

Most research studies on leaf litter decomposition use leaves from the riparian zone surrounding the study sites to maximise the accuracy of measurements of natural leaf litter decay at that site (Boulton & Boon 1991). However, when rates of leaf breakdown are used as a functional measure of river ecosystem health, relative comparisons (i.e. between polluted and unpolluted reference sites) are often more important than absolute values for leaf breakdown rates. With this in mind, it is less important to mimic natural conditions and more important to standardize leaf type and treatment among sites.

We recommend a leaf species that decays relatively fast (because the exposure period can then be relatively short; see 3.1.3) and is commonly available throughout New Zealand. Mahoe (*Melyctus ramiflorus*) is one of the most promising native species because it fulfils both these criteria to some extent (see Linklater 1995; Hicks and Laboyrie 1999; Quinn et al. 2000). Furthermore, breakdown rates of mahoe leaves are similar to those of fast-decaying leaf species commonly used in the northern hemisphere (e.g. alder, *Alnus*

glutinosa; see e.g. Bärlocher et al. 1995; Haapala et al. 2001; Sampaio et al. 2001; Hieber and Gessner 2002; Pascoal et al. 2003). Therefore, the use of mahoe would also facilitate comparisons with processing of "fast-decaying leaves" in northern hemisphere streams. If it is feasible to use more than one leaf species, then using a slower-decaying species in addition to mahoe may prove useful (e.g. rewarewa [*Knightia excelsa*], Hicks & Laboyrie 1999; Quinn et al. 2000). This would be particularly important where regional councils only visit sites at 3-monthly intervals, a period over which mahoe leaf bags are likely to have lost more than 90% of their initial mass. Rewarewa leaves are likely to have lost less than 40% of their initial mass after 3 months (Hicks & Laboyrie 1999). The availability of leaves from these species may be limited in some regions of the country, so other species may also need to be considered.

Ideally, freshly fallen leaves should be used (Boulton & Boon 1991). However, logistic constraints are likely to preclude this approach if comparable studies are to be carried out in many locations. In this case, leaves should be picked from trees from a single location, to minimise variability among leaves. These should be air-dried prior to exposure in leaf bags (see Boulton & Boon 1991).

A promising alternative technique is the "cellulose decomposition potential" method (Hildrew et al. 1984; Boulton & Quinn 2000) first used in streams by Egglishaw (1972). Standard cotton strips are placed in the stream, and the extent of cellulose decomposition is measured as loss in tensile strength. This approach merits further trials to determine the efficacy of using standard cotton fabric as an assay of the impacts of environmental stressors on decomposition.

3.1.3   *Period of exposure and frequency of leaf batch recovery*

The majority of studies of leaf breakdown have retrieved leaf packs or bags repeatedly from a stream after various periods of exposure (e.g. Hill et al. 1992; Gonzalez et al. 1998; Haapala et al. 2001; Hieber & Gessner 2002; Menendez et al. 2003). We now know enough about the normal pattern of leaf weight loss that sequential retrieval is not vital when monitoring stream health, although some comparative studies of leaf loss in relation to certain stressors would still be worthwhile. In general, however, the aim should be to keep sampling effort to a minimum, and leaf bags should be exposed for a standard period of time and sampled once, at the end of this period. We recommend a period of one month. The fast-decomposing mahoe leaves (see 3.1.2) can be expected to lose between 30% (Parkyn & Winterbourn 1997) and 75% (Hicks & Laboyrie 1999) of their initial mass during this period. (Mass losses in Linklater [1995] were intermediate at 50% in 25 days.) These high mass losses and the considerable degree of variation between them in different streams imply that mahoe leaves exposed for one month should be well-suited for relative comparisons between stream sites subjected to different environmental stressors. As mentioned above, if it is more convenient for site visits to correspond with existing State of the Environment monitoring programs where sampling occurs less frequently than monthly, then consideration should be given to using a slower decaying leaf species such as rewarewa.

3.1.4   *Season of exposure*

If the intention is to use leaf packs as a functional measure of river ecosystem health, relative comparisons (i.e. between polluted and unpolluted reference sites) will generally be more important than absolute values of leaf breakdown rates. Therefore, leaf packs can be exposed at any time of the year. This differs from studies aimed at determining

absolute breakdown rates for certain leaf species, where the most realistic values are obtained when leaves are introduced at the time of their characteristic peak leaf fall (Boulton & Boon 1991). Thus, the preferred season of leaf bag exposure in northern hemisphere studies is the autumn because of the prevalence of deciduous tree species in many regions (see e.g. Garden & Davies 1988; Lopez et al. 2001; Menendez et al. 2003; Pascoal et al. 2003). By contrast, most native New Zealand tree species are evergreen and peaks of litter input are much less clearly defined (see e.g. Linklater 1995; Quinn et al. 2000). This provides further justification for exposure of leaves at any time of year in New Zealand.

3.1.5 *Where should leaf bags be positioned?*

Leaf bags should be anchored to the stream bottom using metal pegs (at least 15-20 cm long) driven beneath the surface of the bed so that hydraulic conditions are not dramatically altered. Leaf bags should be secured to the pegs using strong monofilament fishing line. Ideally leaf bags should be secured on or near the stream bed and not allowed to float up in the water column since this may alter decay rates (Mutch et al. 1983). We have found that it is helpful to place a rock over the fishing line to keep the leaf bags near the stream bed and stop them spinning in the current (Young et al. 1994).

To mimic the natural rate of leaf decomposition, leaf bags should be tethered in areas where leaves are likely to accumulate naturally (Boulton & Boon 1991). Breakdown rates have been shown to differ between habitat types within the same stream, with slowest decomposition typically for leaves buried in debris dams and pools, and higher decomposition rates in riffles (Meyer 1980; Casas 1996). Once again, since health measurements rely on relative comparisons among sites, it is important to ensure that the habitat types where leaves are positioned are as consistent as possible among sites. We recommend that riffles should ideally be used as a standard habitat type since leaves naturally accumulate in riffles (Speaker et al. 1984), invertebrate density and diversity are often highest in riffles (Brown & Brussock 1991), and sediment deposition and leaf pack burial is less likely in riffles. In larger rivers, riffles are also the shallowest areas making leaf bag deployment and recovery relatively easy.

3.1.6 *Response parameters*

The simplest way of assessing leaf breakdown is to use weight loss of leaves during the period that they have been deployed. As mentioned above, leaves should be air dried and accurately weighed prior to being secured at the sampling sites. After removal, they need to be air-dried again and re-weighed. Data normally are reported as the percentage of the initial weight remaining. The only equipment required for this assessment is an accurate balance capable of measuring leaf weight to the nearest 0.1 g. Sediment may accumulate on and among the leaves during the incubation period leading to an underestimate of decomposition rates. Sediment can be removed by gentle rinsing, however some leaf material may be lost during the washing process leading to overestimates of decomposition rates. One way to get around this problem is to report results in terms of ash free dry mass (i.e. organic matter only). To do this, subsamples of leaves, both prior to deployment and after collection, need to be burnt in a furnace to determine their inorganic (ash) content.

One of the problems associated with weight loss measurements is that some of the measured weight may result from loss of large leaf fragments through physical abrasion rather than 'real' decomposition. An alternative measure that can be conducted on any

remaining leaf material would be useful in this regard. The best example of an alternative measurement is leaf strength or toughness. Tensile strength could be used on some species of leaves and has been used with standard strips of cotton fabric as mentioned above (Hildrew et al. 1984; Boulton & Quinn 2000). However, a more versatile method is 'penetrometry' which measures leaf toughness via the force required to drive a blunt metal pin through a leaf (Suberkropp & Klug 1980). This force can be measured in Newtons with an expensive commercially available penetrometer, or more simply as the mass (lead shot or water added to a container directly above the pin) required to force the pin through the leaf. Leaf toughness measurements have been successfully used in several recent studies of leaf decomposition and appear to be a good measure of microbial decomposition of leaf material (Young et al. 1994; Quinn et al. 2000; Huryn et al. 2002; Chadwick & Huryn 2003; Niyogi et al. 2003).

Research studies often include a wide variety of other measurements to assess changes in the leaves during the decomposition process (Boulton & Boon 1991). These other measurements include the concentrations of polysaccharides, total nitrogen, protein, tannin, lignin, and ergosterol (an indicator of fungal biomass) within the leaves themselves (Suberkropp et al. 1976). Measurements of microbial growth and activity are also often made using radioactive thymidine and respiration rates, respectively. Although these measurements help determine the relative importance of role played by bacteria, fungi and invertebrates in the decomposition process, they are not necessary when using leaf litter breakdown as a routine measurement of river ecosystem health.

3.2   Data analysis options

The simplest method of reporting breakdown rates is to use the percentage of the initial weight of leaf material remaining after a certain time period (%R).

$$\%R = 100 \times \left(\frac{W(t_f)}{W(t_i)} \right) \tag{1}$$

where $W(t_i)$ is the initial weight of leaf material and $W(t_f)$ is the amount of material remaining after time (t). The percentage of the initial material lost per day can be calculated for comparison among sites. This method assumes that decomposition is linear and that a constant amount of material is lost throughout the decomposition process.

Research studies on leaf litter breakdown often observe exponential decay of the leaf material where a constant proportion of the material remaining at any time is lost throughout the decomposition process. In these situations it is more accurate to report breakdown rates in terms of an exponential decay coefficient (k, day^{-1}) (Petersen & Cummins 1974).

$$k = -\log_e \left(\frac{W(t_f)}{W(t_i)} \right) / (t_f - t_i) \tag{2}$$

If natural differences in water temperature are expected between sites, and temperature has been measured continuously throughout the study, then it is possible to factor out the effects of temperature on decomposition rates by using degree days, rather than days, as the measurement of time in the above calculations (Minshall et al. 1983). Degree days

can be calculated by summing all the daily average water temperature measurements during the period when leaves were decomposing. For example, if the average water temperature at a site was 15°C for a day then that site would have accumulated 15 degree days on that day. Degree days accumulate quickly at warm sites and slowly at cool sites.

Measurements of leaf breakdown in terms of changes in leaf toughness (see 3.1.6) can be calculated using the equations described here, by substituting toughness measurements for weight measurements.

As mentioned in 3.1.3 above, sequential recovery of leaf bags from study sites is not necessarily required when using the technique to measure ecosystem health. However, if only the initial and final weights (or toughness) of leaf material are measured then it is impossible to determine if decomposition was linear or followed the exponential model. Breakdown rates based on the exponential model are the standard measure reported in the scientific literature.   Therefore, we recommend that exponential decomposition is assumed and thus Equation 2 should be used to describe decomposition rates.

3.3   Advantages and disadvantages of using leaf litter decomposition

Along with the general advantages of being a functional indicator (as listed in Section 2.2), leaf litter breakdown has several specific advantages that make it a particularly good indicator. These include the facts that:

- measurement of leaf breakdown is relatively simple and only requires inexpensive equipment that is probably already available in most regional councils and unitary authorities.

- a large number of scientific studies have examined the factors that control leaf breakdown (see Sections 3.4 & 3.5). Therefore, responses to natural variation and most stressors can be predicted with confidence.

- leaf breakdown can be measured anywhere from tiny streams to large rivers. It could also be used in lakes and estuaries.

- Gessner & Chauvet (2002) have proposed interim criteria for linking breakdown rates with ecosystem health. These authors (and others) are currently involved in a European Union project (http://www.ladybio.ups-tlse.fr/rivfunction/index.html) that aims to develop and disseminate methods for using leaf breakdown to assess the functional health of river ecosystems. Lessons learnt in Europe may be useful here in New Zealand.

Nothing is ever perfect, and there are some potential disadvantages of using leaf breakdown as an indicator. These include the facts that:

- a wide variety of factors influence leaf litter breakdown. Ideally, indicators should respond predictably to damage caused by humans, while being insensitive to natural spatial or temporal variation (Norris & Hawkins 2000). However, this is a difficult goal to achieve since sensitive indicators, like leaf litter breakdown, will generally respond to both natural variation and human induced changes to ecosystems. Inevitably, tradeoffs among generality, sensitivity and robustness

need to be made in the choice of appropriate indicators (Gessner & Chauvet 2002).

- leaf litter breakdown is measured at a specific location in a stream and is only indicative of conditions at that location, rather than over the entire reach. In some ways this is an advantage, in that any differences among sites that are observed can be related back to conditions at specific sites. However, a stressor may have a strong impact on one habitat type (e.g. pools), but little impact on another habitat type (e.g. riffles). Therefore, if breakdown was only measured in riffles then the impact on pools would not be detected.

- there may be some difficulty explaining the measurement and its meaning to a wider audience. Concepts behind some structural indicators (e.g. pollution-sensitive species are rarely found at polluted sites) can be more easily grasped by the general public. This should not stop leaf litter breakdown being used as an indicator, but it is recognized that simple, clear explanations are required to demonstrate how leaf breakdown rates measure stream health. Hopefully, material presented earlier in this report (e.g. Section 2.1) will be useful in this regard.

3.4 Summary of controlling factors and likely responses to stressors

Many factors control leaf litter decomposition and range from those that vary naturally from site to site (e.g. climate, position in river hierarchy) through to those that are strongly influenced by human-induced disturbance to ecosystems (e.g. toxic chemicals, organic pollution). However, most controlling factors lie between these extremes and vary due to both natural and anthropogenic causes (e.g. nutrients, pH, sediment, riparian vegetation, temperature). The importance of terrestrial leaf litter for fuelling stream ecosystems became widely recognised in the 1970's. Since then there has been a large amount of research published on leaf litter breakdown and the factors that control it. A detailed review of this information is provided in Section 3.5 for those who are interested in the specifics. However for ease of reading a summary of the likely responses to various environmental stressors found in New Zealand is shown in Table 1.

Table 1. Likely responses to a range of stressors commonly occurring in New Zealand.

Stressor	Change	Response	Comments
Water temperature	Warmer water	Faster leaf breakdown	
Sediment	More fine sediment	Slower breakdown	Often linked with other stressors
pH	Acid condition	Slower breakdown	
pH	Alkaline conditions	???	
Nutrients	Nutrient enrichment	Faster breakdown	If sediment is a problem too – response may be cancelled out by sediment effect
Organic pollution	Increased pollution	Faster breakdown	Response due to nutrient stimulation but may be cancelled out by changes to invertebrate community
Toxic chemicals	Toxic chemical inputs	Slower breakdown	Via reduced invertebrate and microbial activity

Riparian vegetation	Loss of stream-side vegetation	Faster breakdown	Via warmer temperatures, although effect may be cancelled out by increased sedimentation
River regulation	Damming of a river	Faster breakdown	Response due to warmer winter temperatures, but may be different in other seasons and if invertebrate community also changes
Channelisation	Simplification of habitat	Slower breakdown	Via loss of natural leaf accumulations and leaf-eating invertebrates
Water abstraction	Reduced flows	???	Would probably depend on magnitude of flow reduction

3.5   Literature review of factors controlling leaf litter decomposition

As mentioned above, many factors control leaf litter decomposition and there has been a large amount of research published on leaf litter breakdown and the factors that control it. In this section we review this research, initially focussing on factors that primarily vary naturally, then moving along the continuum to factors influenced primarily by human disturbance. There is considerable detail in this section and readers are encouraged to focus on Table 1 for an initial overview. If more information is required then this section is the place to look.

3.5.1   *Climatic zone*

Graca (2001) noted that most leaf breakdown studies have been carried out in temperate locations and questioned whether results can safely be generalised to non-temperate situations. In this context, Chergui & Pattee (1991) studied breakdown of willow (*Salix* sp.) and oleander (*Nerium oleander*) leaves at four sites in a Moroccan river. Despite an annual mean water temperature of 20°C, the breakdown of willow leaves proceeded no faster than in more temperate climates. Mathuriau & Chauvet (2002), who studied the breakdown of *Croton gossypifolius* and *Clidemia* sp. leaves in a 4th-order neotropical stream (Andean Mountains, Colombia), came to a rather different conclusion. They recorded high rates of breakdown of the two leaf species, probably related to strong biological activity (of both fungi and invertebrates in general, but not leaf eating invertebrates in particular) under the stable and moderately high water temperatures (19°C), and concluded that high fungal activity associated with rapid leaf breakdown may be characteristic of tropical streams.

Given the lower temperatures encountered in alpine as opposed to lowland zones, leaf breakdown would be expected to be lower in alpine systems. Gessner et al. (1998) studied breakdown of green alder (*Alnus viridis*) in four Swiss glacial alpine streams that do not normally receive notable inputs of leaf litter. Leaf breakdown was relatively slow but fungal biomass in decomposing leaves attained levels as high as those found in temperate woodland streams. Robinson et al. (1998) studied the relationship between macroinvertebrate assemblages and the breakdown of alder leaves in the same streams. Although glacially fed, the four sites (pro-glacial, glacial lake outlet, main channel, and a side-channel with a mix of water sources) differed physically and contained different benthic communities. Leaf breakdown and associated fungal properties differed widely

among sites. Leaf breakdown rate was fastest at the lake outlet, probably reflecting the presence of *Acrophylax zerberus*, a leaf-eating caddisfly, and slowest at the pro-glacial site, where invertebrate taxa were few and leaf breakdown could be attributed primarily to micro-organisms.

3.5.2   *Position in the river hierarchy*

Differences in breakdown rates can be large when streams of different sizes or at different altitudes are compared, although occasional studies have recorded no influence of stream order on breakdown rates (e.g. Graca et al. 2001). Some studies have reported higher breakdown rates in small, low order streams than high order rivers. For example, Jonsson et al. (2001) studied 23 boreal streams of varying size (1st-7th order) in central and northern Sweden and found that leaf breakdown was fastest in smaller streams, where leaf-eating invertebrates were more abundant. Similarly, Baldy et al. (1995) found that breakdown of a variety of leaf species in a large French river (7th order) was slow compared to low-order streams.

In contrast, Pozo (1993) recorded faster breakdown rates at downstream than at headwater sites in a Spanish stream, possibly because of higher N and P concentrations downstream. Paul & Meyer (1996) also found that breakdown rates of tulip-poplar (*Liriodendron tulipifera*) and rhododendron (*Rhododendron maximum*) were faster in a 4th order than a 1st order site on an Appalachian stream, reflecting differences in stream geomorphology. Fabre & Chauvet (1998) compared breakdown of alder (*Alnus glutinosa*) leaves at 14 sites along a 1400 m altitudinal gradient (1860-460 a.s.l.) of a 1st to 3rd-order stream in Southern France. Breakdown rates almost doubled from the upper to the lower sites, in association with increases in the abundance of leaf-eating invertebrates and fungal biomass. Fleituch (2001) also found faster rates of breakdown (of beech leaves) at more downstream sites along a 295 m altitudinal gradient of a 1st to 5th order Polish stream, attributing this increase to increased physical fragmentation or faster microbial activity downstream.

3.5.3   *Stream bed characteristics*

Differences in leaf breakdown rates among different habitats within streams have been observed in several studies. Meyer (1980) found that leaf decomposition was highest in rapids and lowest in pools where sediment deposition affected decomposition rates. Casas (1996) studied the breakdown of *Acer pseudoplatanus* leaves in different patch types in a backwater of a subalpine stream in Austria and found a gradient of processing rates from slowest for leaves buried in debris dams to highest in riffles. In a study of a tallgrass prairie stream in Kansas, Smith (1986) came to the same conclusion and noted that leaf aggregations from riffles contained more invertebrates than those from pools.

Reice (1974) compared breakdown rates of leaves placed on four different substrate types; fine silt, sand, gravel and rock. Breakdown rates were consistently lower on fine silt than on the other substrate types. Low invertebrate community diversity and possible anaerobic conditions within the silty substrate were given as the likely reasons for these differences among substrate types.

Rounick & Winterbourn (1983) compared leaf breakdown between two streams with different stability. Breakdown rates in coarse mesh leaf bags were higher in the stable stream than in the unstable one, while breakdown rates in fine mesh bags were equivalent between sites. Increased stream-bed stability appears to result in higher densities of the

leaf-eating caddisfly, *Zelandopsyche ingens*, which were responsible for the higher breakdown rates in the leaf bags they were able to access in the stable stream.

3.5.4   *Presence/absence of keystone species*

We have already mentioned many cases where the abundance or richness of the leaf-eating invertebrates seem to be positively associated with leaf breakdown rate. In this context, Schofield et al. (2001) assessed whether crayfish influenced rhododendron leaf breakdown in a forested Appalachian stream in both summer (when leaves other than rhododendron are relatively scarce) and autumn (when other leaves are relatively abundant). Rhododendron is considered a low quality food resource for leaf-eating invertebrates. Crayfish were excluded from the benthos using electric 'fences'. As predicted, excluding crayfish reduced the breakdown rate of rhododendron leaves in both summer and autumn. Crayfish accounted for 33 and 54% of rhododendron breakdown in summer and autumn, respectively, probably due to direct consumption of rhododendron by crayfish. Biomass of leaf-eating insects, insect predators and fungi did not differ between control and exclusion treatments, indicating that insectivorous sculpins (*Cottus bairdi*) had no effect on rhododendron breakdown and that omnivorous crayfish did not exert an indirect effect via alteration of insect or fungal biomass. In summer (when other, more palatable leaf types were not available), rhododendron leaf packs appeared to provide 'resource islands' for leaf-eating insects. There was a significant inverse relationship between leaf-eating insects and leaf pack mass in the summer exclusion treatment: insects were the only organisms eating leaves in this treatment and, as leaf-eating insect biomass increased, remaining leaf pack mass decreased. In the control treatment, however, this relationship was not observed; here, the effect of leaf-eating insects was presumably swamped by the impact of crayfish.

Vertebrate consumers have also been reported to indirectly influence breakdown rates. Thus, Konishi et al. (2001) conducted a field experiment to test for trophic cascading effects of predatory fish on detritus processing by benthic invertebrates in stream channels running through a wetland forest in Japan. They found that the fish had an indirect but significant effect on leaf litter processing. Two treatments (fish-present and fish-absent) were established for 4 weeks, with two common predatory fish, rainbow trout (*Oncorhynchus mykiss*) and freshwater sculpin (*Cottus nozawae*), being introduced into and excluded from stream cages. At the end of the experiment, biomass of the dominant leaf-eater (the amphipod *Jesogammarus jezoensis*) and loss of oak leaves (*Quercus crispla*) from litter bags were both significantly less in the fish-present treatment than in the fish-absent treatment. The authors concluded that predator-induced lower biomass and likely lowered foraging activities of *J. jezoensis* were responsible for the suppression of litter processing efficiency. In a similar experiment in Costa Rica, Rosemond et al. (1998) also found that exclusion of top consumers (fishes and shrimps) resulted in significantly higher densities of small invertebrates inhabiting leaf packs, but none of these were leaf-eaters. Despite the increase in invertebrate density, in this case breakdown rates of leaves were not statistically affected by exclusion of top consumers.

Usio (2000) performed a field experiment in a forested Otago stream to test the hypotheses that omnivorous crayfish both promote breakdown of leaves (basal resources) and decouple any potential trophic cascade by simultaneously affecting intermediate consumers as well as their basal resource. Leaf packs were placed inside artificial channels, which excluded or allowed access by crayfish. During a 4-week period, crayfish greatly promoted leaf processing, with breakdown rates among the fastest ever

recorded from temperate streams. Crayfish also affected invertebrate abundance in the leaf packs. As a result of resource consumption, predation and sediment removal, crayfish treatments contained significantly lower densities of invertebrates. In contrast, exclusion of crayfish did not promote leaf breakdown via increased colonisation by leaf-eating invertebrates, primarily because of the conspicuous lack of leaf-eating invertebrates in this (and in other) New Zealand streams (references in Winterbourn 1995).

The distribution of species, whether leaf-eating insects, crayfish or fish, is patchy at a range of scales; species may be present or absent at the catchment scale, in individual tributaries or even in reaches within a tributary. Because certain keystone species can affect leaf breakdown rate, some of the variation in this functional process can be expected to be due to patchy distributions of the keystone consumers.

3.5.5 *Conductivity*

Rosset et al. (1982) compared breakdown rates of various leaf species in fine or coarse-mesh bags in two hard water (Swiss Jura; conductivity 273-320 µS; pH 8.36-8.44) and two soft water streams (Black Forest; conductivity 36-38 µS; pH 6.5-7.3). In fine-mesh bags, where decomposition is presumed to be performed mainly by microorganisms, breakdown rates of oak leaves (*Quercus robur*) were significantly faster in the hard water streams; in coarse-mesh bags breakdown of both oak leaves and larch needles (*Larix decidua*) was faster in hard water, where gammarid crustaceans (leaf-eaters) were particularly prominent.

3.5.6 *pH*

A consistent result when acid and circumneutral streams have been compared is for breakdown to be reduced under acid conditions. For example, Griffith and Perry (1994) found leaf breakdown rates (and to a lesser extent fungal biomass associated with the leaves) reduced with pH in four streams with different underlying geology and water pH. Similarly, Dangles & Guerold (2001) studied beech leaf breakdown and associated fauna along a forested stream showing a gradient in acidification level (4 stations, pH 5.3-7.2) and found that breakdown rate was severely depressed under acidic conditions. They compared the shredding efficiency of the main invertebrate species present at each station, the acid-tolerant *Protonemura* sp. and the acid-sensitive *Gammarus fossarum*. Not only did the acidic station show lower biomass of leaf-eaters, but *Protonemura* sp. showed significantly lower feeding activity than *G. fossarum* in the non-acidic stations. At the other three stations, feeding activities of *G. fossarum* were similar and differences in litter breakdown rates appeared to be related to *G. fossarum* densities.

Leaf breakdown is consistently slower in acidic than in circumneutral conditions, regardless of whether the low pH is caused by simulated or natural acid rain (e.g. Allard & Moreau 1986; Garden & Davies 1989; Dangles & Guerold 1998, 2001; Dangles & Chauvet 2003) or mining drainage waters (e.g. Siefert & Mutz 2001). However, minor decreases in pH may not be strong enough to affect leaf decomposition (e.g. pH 5.5 in Kirby 1992). Although New Zealand is not troubled by acid rain, nevertheless some streams are naturally more or less acid. Thus, Collier & Winterbourn (1987) studied breakdown of kamahi (*Weinmannia racemosa*) leaves enclosed in 1 mm mesh bags in two naturally acidic, brown-water streams and in two circumneutral, clear-water streams in South Westland. Leaves broke down more slowly in the acidic brown water streams. Fungi were more common on leaves from brown-water sites while bacteria were more

common on leaves from clear-water sites. Invertebrate faunas colonising leaf bags were dominated by (non-leaf-eating) chironomids at the brown-water sites, whereas obligate (*Triplectides* sp.) and facultative (*Austroperla cyrene*, *Oeconesus* sp. and *Olinga feredayi*) leaf-eaters were common at the clear-water sites. The authors speculated that these leaf-eating invertebrates were largely responsible for the faster breakdown rates recorded there.

3.5.7 *Riparian vegetation*

Different leaf species break down at different rates, and this reflects leaf chemistry (e.g. concentrations of essential nutrients, chemical inhibitors, fibre content; Webster & Benfield 1986; Enriquez et al. 1993; Ostrofsky 1993, 1997; Campbell & Fuchshuber 1995; Royer & Minshall 2001). Even the same species of leaf from plants grown under different circumstances may differ in N content and decompose at different rates (Chadwick & Huryn 2003). Hutchens & Benfield (2000) investigated whether changes caused by gypsy moth (*Lymantria dispar*) attack on chestnut oak (*Quercus prinus*) leaves affected leaf breakdown rates in six Appalachian streams. Breakdown rates of second-flush leaves produced after defoliation were compared to those of natural spring-flush leaves shed in autumn. Second-flush leaves, which had lower fibre contents, generally broke down faster than spring-flush leaves, indicating that insect attack could accelerate detritus processing in these streams.

Several New Zealand studies have compared breakdown rates of native and introduced tree leaves. In a first order tributary of the Waimakariri River in Canterbury, Parkyn & Winterbourn (1997) found no difference in the breakdown rates of exotic and native tree leaves; of six species tested, the introduced elm (*Ulmus procera*), the native red beech (*Nothofagus fusca*) and the introduced willow (*Salix babylonica*) had the fastest breakdown rates. Hicks & Laboyrie (1999) also examined comparative mass-loss rates of native evergreen and exotic deciduous trees in a Waikato stream, using fallen leaves incubated in bags with 2 x 3 mm mesh openings for 28 days. The native trees were mahoe (*Melicytus ramiflorus*), kahikatea (*Dacrycarpus dacrydioides*), silver beech (*Nothofagus menziesii*), rewarewa (*Knightia excelsa*) and tawa (*Beilschmiedia tawa*); the introduced trees were silver birch (*Betula pendula*) and alder (*Alnus glutinosa*). Rates of mass loss followed the sequence: mahoe > silver birch > alder > kahikatea > silver beech > rewarewa > tawa. Quinn et al. (2000) investigated factors influencing in-stream disintegration and uptake of dissolved nutrients by leaves to improve the basis for selecting plants for riparian management. Leaves from five introduced (black walnut, "grass" [*Lolium perenne*], eucalypt, poplar and pine) and five New Zealand native plants (wineberry, mahoe, hoheria, tawa and rewarewa) were studied in 12 streamside channels and a natural stream. Rates of mass loss in streamside channels followed the sequence: black walnut > wineberry > mahoe > hoheria > grass > eucalypt > poplar > pine > tawa > rewarewa.

The type of natural inputs of organic matter to streams can also influence leaf breakdown processes. Thus, Eggert & Wallace (2003) measured breakdown rates of leaves of maple (*Acer rubrum*) and *Rhododendron maxima* for the first three years in a stream in which detrital inputs were excluded for 7 years. Processing rates of maple leaves (but not rhododendron) were slower in the exclusion stream than in the reference stream, in association with lower leaf-eating invertebrate production and consumption rates. It seems that leaf-eating invertebrates tracked high quality organic matter resources

(leaves), illustrating the interdependence of stream organisms and terrestrial organic matter input.

Changes in riparian vegetation can also have indirect effects on leaf breakdown rates. For example, Whiles & Wallace (1997) reported that breakdown rates of both pine and red maple litter were faster in a stream flowing through pine forest than one flowing through an adjacent hardwood catchment. They attributed this to the lower stream temperatures of the pine stream than the hardwood stream. Other things being equal, it seems likely that, streams flowing through landscapes with more open canopies (deciduous vs non-deciduous trees, grassland biomes compared to forest biomes) will experience higher temperatures and faster leaf breakdown rates.

Logging of stream catchments generally increases leaf processing rates. Benfield et al. (2001) found breakdown rates in a North Carolina stream were consistently faster after logging than before (or in an adjacent unlogged stream), possibly partly due to a 3-10 fold increase in nitrate concentrations and partly to a potential increase in leaf-eating invertebrates. Similarly, Hutchens & Benfield (2000) reported that leaf breakdown was more rapid in three streams draining a recovering 14 yr-old clear-cut catchment than in three streams draining a reference catchment. They attributed this increase in leaf breakdown to higher abundance and density of leaf-eating insects and greater microbial conditioning in leaf packs in the streams of the recovering clear-cut catchment. Griffith & Perry (1991) also found that a stream logged 20 years before had higher breakdown rates (associated with higher invertebrate densities) than one undisturbed for 80 years. Webster & Waide (1982) found a slightly different pattern when comparing breakdown rates before, during and after clearcutting of a second order stream in North Carolina. Breakdown rates were slowed during and immediately after clear-cutting, but then accelerated later. The slow breakdown rates during logging activities were attributed to burial of leaf packs by sediment.

Bird & Kaushik (1992) compared leaf litter processing in a forested and agricultural reach of a stream. Breakdown rates were similar between reaches but there were major differences in the mechanism of breakdown. Physical abrasion and microbial activity were most important in the agricultural reach, while microbial and invertebrate activity were primarily responsible for breakdown in the forested reach.

3.5.8 *Sediment*

Increased levels of fine sediment appear to slow down leaf processing rates. Thus, Meyer (1980) followed leaf breakdown at six sites in a forest stream in New Hampshire and found breakdown was slowest in sites where sediment deposition was high. Similarly, Rader et al. (1994) found that breakdown rates of sweet gum leaves (*Liquidambar styraciflua*) in a South Carolina stream were inversely related to sediment accumulation (both fine particulate organic and inorganic particles). Niyogi et al. (2003) found that increased fine sediment associated with agricultural development reduced breakdown and seemed to counteract the positive effect of elevated nutrient levels on leaf breakdown rates. Reice (1974), Triska & Buckley (1978), Herbst (1980), Chauvet (1988) and Chergui & Pattee (1990) all found that siltation or burying of leaves delayed leaf litter breakdown.

3.5.9   *Water temperature*

While there are exceptions, studies involving a wide variety of leaf species have shown faster leaf breakdown when warmer streams have been compared with cooler ones and when warm periods have been compared with cold periods in seasonal studies (Webster & Benfield 1986). For example, seasonal comparisons of the processing of box elder (*Acer negundo*) in a Rocky Mountain stream (McArthur et al. 1988), hackberry (*Celtis laevigata*) leaves in a Texas stream (Short & Smith 1989) and various deciduous leaf species in a second order stream in Spain (Lopez et al. 2001) all showed more rapid weight loss during the warmer months of the year than during winter. The results of studies such as these, although inevitably confounded by variation in factors other than temperature, are supported by controlled laboratory studies (Webster & Benfield 1986). However, the effects of temperature are not uniformly strong, and may be swamped by variation in other factors. For example, in a study of three Appalachian streams that differed slightly in mean temperature and in stream pH, leaf weight loss was not related to average temperature (Rowe et al. 1996) but the most acid stream had the lowest rate of leaf weight loss. Similarly, Menendez et al. (2003) found a temperature-related faster rate of leaf breakdown in summer in a eutrophic Spanish river but not in a neighbouring oligotrophic stream.

Temperature appears to have its strongest effects on microbial processes and seems to exert less influence on invertebrates feeding on leaf material (Webster & Benfield 1986). Thus, in situations where invertebrates play a dominant role in decomposition, breakdown rates can be high even at very cold temperatures (Short et al. 1980). For example, Bunn (1988) found that jarrah (*Eucalyptus marginata*) leaves in a small forest stream in Western Australia were processed at a faster rate in winter than summer, attributing the difference to the increased density of leaf-eating invertebrates in that season. Garden and Davies (1988, 1989) also found faster breakdown rates of balsam poplar (*Populus balsamifera*) in a Canadian stream in autumn than in spring, despite the higher initial quality of leaves (in terms of nutrient content) in spring. Reduced spring/summer breakdown rates were the result of decreased microbial activity and lower numbers, kinds and biomass of leaf-eating invertebrates.

Few studies have investigated the impact of increased water temperatures (e.g. due to climate change) on leaf processing. Buzby and Perry (2000) used a computer simulation model to evaluate some of the projected impacts of climate change, including elevated temperatures and increased frequency and magnitude of floods and droughts, on leaf pack processing. The model simulated microbial processing, invertebrate consumption, and transport along a 1-km second-order stream. The effects of wet and dry years with and without a 2°C temperature increase were examined. Both invertebrates and microbes processed more leaf material under the elevated temperature scenarios; however, the invertebrate response was greater than the microbial response. Invertebrates processed a greater percentage of the inputs in wet than in dry years, while microbial processing rates were unaffected. All climate change scenarios resulted in decreases, sometimes substantial, in coarse particulate organic matter availability to leaf-eating invertebrates during the summer months.

3.5.10   *Nutrients*

Many studies have found that high concentrations of dissolved nutrients, mainly N and P, stimulate leaf breakdown in aquatic environments (see Menendez et al. 2003). For example, Suberkropp and Chauvet (1995) found that nitrate concentration best explained

variation in breakdown in headwater streams in Alabama. Furthermore, Rosemond et al. (2002) examined effects of landscape-scale natural variation in streamwater phosphorus at 16 stream sites in Costa Rica and found that breakdown rate was controlled by phosphorus concentration. Microbial processes appeared to be most important in driving differences in breakdown among sites, but invertebrates also contributed to elevated rates at high-P sites.

The majority of studies of the effects of nutrient enrichment on leaf processing found that enrichment increased processing rates. Gulis and Suberkropp (2003) reported that artificially elevating ammonium, nitrate and phosphate significantly increased leaf breakdown rate (and fungal and bacterial biomass). Huryn et al. (2002) found reduced breakdown rates in agricultural and urban streams compared with forest and wetland (suburban) streams, and that leaf breakdown rates were positively correlated with increased concentrations of nitrate and phosphate. Elwood et al. (1981), Meyer and Johnson (1983), Pozo et al. (1998), Robinson and Gessner (2000), Graca et al. (2001) all found that high concentrations of dissolved nutrients (mainly N and P) in stream environments stimulated leaf breakdown. In New Zealand, Young et al. (1994) and Niyogi et al. (2003) reported elevated breakdown rates of tussock leaves in more agriculturally developed catchments in New Zealand (related most closely to oxidised nitrogen in Young et al. 1994, and to both nitrate and phosphate in Niyogi et al. 2003).

On the other hand, several studies have reported that enrichment with N or P (not both together) did not affect leaf processing rates. In most of these cases, however, this was because enrichment was performed with the non-limiting nutrient (e.g. Triska & Sedell 1976; Newbold et al. 1983; Chadwick & Huryn 2003), or because of co-limitation by both N and P (e.g. Howarth and Fisher 1976; Grattan and Suberkropp 2001). Only Royer and Minshall (2001) found that the simultaneous addition of N and P fertiliser pellets to leaf bags did not increase breakdown rates and concluded that breakdown rate was not nutrient limited in their study stream. However, N and P concentrations in this stream were fairly high even without the experimental enrichment.

3.5.11 Organic pollution

Overall, organic pollution seems to have a strong influence on the diversity of aquatic organisms and the relative contributions of invertebrates and microorganisms to leaf breakdown, thus resulting in altered breakdown rates. Pascoal et al. (2003) assessed the effects of organic pollution in a Portuguese river and found that downstream enrichment with organic and inorganic nutrients was correlated with accelerated leaf breakdown rates. Higher nutrient concentrations were associated with an increase in density but a decrease in richness of macroinvertebrates, as well as a strong decline in spore production of aquatic fungi but no major change in fungal richness. However, the ratio of leaf breakdown rates in coarse-mesh bags (which exclude invertebrates) and fine-mesh bags indicated that ecosystem functioning was compromised at all organically-polluted sites. In another Portuguese river, Pascoal et al. (2001) also found that breakdown rates and abundance of invertebrates associated with leaf packs increased with organic pollution but invertebrate richness declined. On the other hand, Raviraja et al. (1998) reported that leaf breakdown rates in an organically polluted river in India were not statistically different from an earlier study in a neighbouring but unpolluted stream (Raviraja et al. 1996), even though spore production from stream-exposed leaves by aquatic hyphomycetes (aquatic fungi) was dramatically lower in the polluted river.

3.5.12   *Toxic chemicals*

Elevated concentrations of heavy metals and other toxic chemicals can have negative effects on leaf breakdown rates, although apparently mainly at very high levels. For example, Sridhar et al. (2001) studied leaf breakdown in two heavy metal-polluted streams (one with a high and one with a moderate load) in a former mining district in Germany. They showed that decomposition was slowed relative to unpolluted streams only in the stream with the high heavy metal load, possibly because high N and P concentrations in the moderately polluted stream stimulated fungal activity. Similarly, Niyogi et al. (2001) found a significant relationships between litter breakdown rates and the concentrations of zinc and deposition rates of metal oxides in 27 sites in the Rocky Mountains, some of which were affected by mine drainage. Sites with high zinc concentrations had low biomasses of leaf-eating invertebrates, while deposition rates of metal oxides were correlated with microbial respiration rates, suggesting that mine drainage influenced leaf breakdown rates via changes to invertebrate and microbial communities and activity. Schultheis and Hendricks (1999) also reported that leaf breakdown was inhibited by high in-stream copper concentrations. In contrast, Nelson (2000) reported no effect of breakdown downstream of metal pollution from point sources in another former mining area.

Kreutzweiser et al. (1998) investigated whether residues of a herbicide had adverse effects on leaf-eating invertebrates inhabiting and using natural leaf packs. Despite considerable accumulations of the herbicide in the packs in systems treated at or near expected environmental concentrations, there was no significant mortality of leaf-eating insects and no significant reductions in leaf consumption. Significant mortality and reduced feeding occurred only in systems treated at concentrations well above (up to 10 times) expected environmental concentrations. In an even more extreme case, Wallace et al. (1982) treated a small forest stream with high doses of an insecticide, which caused massive downstream insect drift and reduced aquatic insect densities to less than 10% of an untreated reference stream. Leaf breakdown rates were reduced signficantly, probably due to a lack of macroinvertebrate consumers.

Forrow and Maltby (2000) studied the mechanistic basis for reduced leaf processing in a stream contaminated with superhighway runoff, using in situ and laboratory studies on *Gammarus pulex*, the dominant leaf-eater. The in situ feeding rate of *G. pulex* was significantly reduced downstream of the motorway discharge. Laboratory studies demonstrated that the main mechanism responsible for the reduction in feeding was direct toxicity and that this was most severe when animals were in direct contact with contaminated sediments. Stout & Coburn (1989) also found reduced litter processing rates downstream of the impact of highway construction. However, they attributed this to the absence of natural leaf accumulations and leaf-eating invertebrates at impacted sites, rather than to inputs of any toxic chemicals.

3.5.13   *River regulation*

The few studies of effects of dams on leaf breakdown downstream have yielded contrasting results. Short & Ward (1980) reported faster leaf litter processing in the Colorado River downstream of a dam than in a nearby unregulated tributary. They initially expected to see the opposite result due to lower numbers of leaf-eating invertebrates downstream of the dam. However, buffered winter temperatures downstream of the dam apparently allowed faster microbial breakdown of the leaves than in the tributary. Nelson & Roline (2000) found increased weight loss below a reservoir

in coarse-mesh bags when compared to an unimpacted stream. Nevertheless, they found no effect on decomposition rates in fine mesh bags, suggesting that differences in the activity of invertebrates rather than microbial organisms was responsible for the contrast between sites. Casas et al. (2000) reported no effect of a headwater dam on breakdown rate – it may be that the effects of higher nutrient concentrations below the dam were counteracted by higher peak flows breaking up packs above the dam.

Little information is available on the influence on leaf breakdown of other forms of river regulation, such as channelisation or water abstraction. Gelroth & Marzolf (1978) reported that leaf litter breakdown was faster in a natural reach than a channelised reach of a Kansas stream. The reasons for this difference were unclear but the authors reported that natural leaf packs were absent from the channelised reach, suggesting that there were insufficient natural resources of leaf material to support leaf-eating invertebrates in this reach. Further research is needed on the effects of river channelisation and water abstraction.

3.5.14 *Multiple stressor effects*

Human activities in a catchment area will often be reflected by changes in the operation of more than one stressor. For example, when riparian vegetation shading a stream is removed for agricultural or urban development, there are likely to be increases in temperature, nutrient concentrations and fine sediment input into the stream. Sometimes the multiple stressors can be expected to operate in concert to increase leaf breakdown rates, as would be the case when temperature and nutrient concentrations are simultaneously increased. On the other hand, Niyogi et al. (2003) reported that positive effects of higher nutrient concentrations were counteracted by negative effects of more fine sediment in streams in catchments developed for grazing. Similarly contrasting effects of multiple stressors have been reported by Shridhar et al. (2001) who found the negative effects of increased heavy metal concentrations were counteracted by the positive effects of high nutrient concentrations. Moreover, Pascoal et al. (2001) recorded a decline in invertebrates as a result of treated sewage effluent but this was more than compensated by an increase in microbial activity.

3.6   **Case Study A – a problem landfill?**

Consider the following example, which demonstrates how leaf litter processing rates might be used to improve biomonitoring.

> *Concerns have been raised about the potential for contaminated water from an old landfill to affect the health of an adjacent stream. Invertebrate samples were collected in the stream from sites above and below the landfill. Significant reductions in taxon richness and biotic index values were found downstream suggesting an effect of the landfill. However, the stream channel upstream of the landfill is relatively steep and dominated by shallow riffles, whereas the valley flattens out below the landfill and the habitat is dominated by slow pools. Therefore it was difficult to say whether the change in invertebrate communities was due to the landfill or just related to changes in habitat.*
>
> *Five leaf packs each containing approximately 5 g of pre-weighed mahoe leaves were deployed in the stream in three pools above and 3 pools below the landfill to tease out the effects of habitat versus potential leachates from the landfill. After one month, all the leaf packs were removed from the stream and dried at 60°C until they reached a constant weight. Leaf strength of a subset of leaves from each*

bag was assessed using a penetrometer. No differences in the rate of leaf weight loss or decline in leaf strength were found between sites upstream and downstream of the landfill, suggesting that microbial activity was not affected by the landfill. This result suggests that the landfill was not having an effect on ecosystem health and that the change in the invertebrate community that was observed downstream of the landfill may have been caused by habitat differences alone.

4. ECOSYSTEM METABOLISM

Ecosystem metabolism – the combination of primary productivity (photosynthesis) and ecosystem respiration – is a measure of how much organic carbon is produced and consumed in river ecosystems. Algae and other aquatic plants are responsible for primary productivity, while ecosystem respiration measures the rates of respiration of all life, including fish, invertebrates, algae, aquatic plants, and microbes. The balance between organic carbon production and consumption provides information on the relative importance of the two key sources of energy that fuel 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 chain, 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 thus helps to determine their life supporting capacity. Ecosystem metabolism has also recently been linked with nutrient uptake, another important function of river ecosystems (Hall & Tank 2003). Therefore, information on ecosystem metabolism gives an indication of the likely capability of rivers to remove and transform nutrients from the water.

4.1 Background on measurement techniques

4.1.1 *Oxygen, carbon or pH*

Primary production (or photosynthesis) involves uptake of carbon dioxide and release of oxygen into the water, while ecosystem respiration is essentially the reverse of this with oxygen uptake and carbon dioxide release. It is therefore feasible to measure rates of ecosystem metabolism using either changes in dissolved oxygen or carbon dioxide concentrations (Bott et al. 1978). It is relatively difficult to measure carbon dioxide concentrations directly in water, so some researchers have used measurements of pH, which closely correspond with carbon dioxide concentrations, to measure metabolism (Simonsen & Harremoes 1978; Cushing & Wolf 1984). It is also possible to use radioactive $^{14}CO_2$ uptake to measure photosynthesis rates (Bott & Ritter 1981). However, we recommend that changes in dissolved oxygen are used to measure metabolism because this is relatively easy to measure, the magnitude of oxygen change is typically large, and this is the most commonly used technique in the scientific literature.

4.1.2 *Open-system measurements or chambers*

Metabolism can be estimated by measuring natural changes in oxygen concentration within river systems, or alternatively by enclosing part of the ecosystem within an air-tight chamber and measuring oxygen changes within the chamber. Open-system methods have the advantage that they include the whole ecosystem and in many situations measurements are relatively simple and require just one oxygen logger (Young & Huryn 1996). Oxygen concentrations are measured at regular intervals over at least

one 24 hour period and changes in concentration are related to oxygen inputs due to photosynthesis and removal via respiration. The main difficulty with open-system measurements is that they require an estimate of the amount of oxygen diffusing between the air and the water. Diffusion can be easily estimated in most rivers and streams (see Section 4.1.3). However, more complicated techniques are required in small, very turbulent streams with low primary productivity (Marzolf et al. 1994, 1998; Young & Huryn 1998, 1999). The extra equipment and effort required to estimate diffusion probably limits the feasibility of open-system measurements for routine stream health monitoring in these types of streams.

Measurements of metabolism made within chambers usually also use changes in oxygen concentration in chambers over at least a 24 hour period (Bott et al. 1978). However, respiration rates and maximum photosynthesis rates can be estimated over shorter periods by comparing oxygen changes in chambers exposed to high light intensities with those in artificially darkened chambers (Hickey 1988). Chamber measurements have been useful to assess the contribution of different components of river ecosystems to overall metabolism (Naiman 1983; Mulholland et al. 1997; Naegeli & Uehlinger 1997). Since the oxygen changes are measured within an airtight chamber, estimates of metabolism can be made without measurements of diffusion. However, there are many disadvantages of using chambers which include:

- material placed within the chamber is invariably disturbed during the process

- water velocity, light and temperature within the chamber will differ from natural conditions experienced in the river

- errors may occur when trying to relate measurements from different components of the ecosystem determined at a small spatial scale to what is occurring at the scale of a whole reach

- nutrients can become depleted within chambers resulting in artificially low metabolism measurements

- during periods of peak photosynthesis, oxygen may diffuse out of the water within the chamber forming bubbles which are not included in subsequent measurements of dissolved oxygen.

- metabolic processes within important components of river ecosystems, such as the hyporheic zone (the habitat connected to the river but beneath the stream bed), are difficult or impossible to measure using chambers

- chamber design and construction is not easy and is relatively expensive

- A large amount of equipment (chambers, pumps, hoses, power supply) is required, especially because at least 3 separate chambers are required for replicate measurements at each site.

Considerable effort has been made to overcome some of these disadvantages (Bott et al. 1997; Dodds & Brock 1998; Bunn et al. 1999; Uzarski et al. 2001). However, many of the problems cannot be solved simply by adjustments to chamber design. Therefore, we do not recommend chambers for use in routine measurements of river ecosystem health.

4.1.3   *Diffusion: How to measure it*

As mentioned above, open-system measurements of metabolism require an estimate of the amount of oxygen diffusing between the water and the atmosphere. The easiest way to measure diffusion is to use measurements of changes in oxygen concentration, which are required for the metabolism measurements anyway. Once again, there are two techniques for using the oxygen record to estimate diffusion rates. The first uses information on changes in oxygen concentration through the night and also simultaneously calculates the respiration rate. During the dark, photosynthesis stops and so any changes in oxygen concentration are due to either uptake by respiration within the river, or diffusion of oxygen through the river surface, as represented in the following equation:

$$\frac{dO}{dt} = -R \pm kD \tag{3}$$

where *dO/dt* is the rate of change of oxygen concentration, *R* is the rate of oxygen uptake (respiration), and *kD* represents the rate of oxygen diffusion through the river surface. *k* is the reaeration coefficient, and *D* is the oxygen deficit or difference between the measured oxygen concentration and the concentration if the water was fully saturated with oxygen. *R* has a negative sign in front of it because respiration always involves oxygen uptake, while *kD* can be either positive (oxygen diffusing into the river) or negative (oxygen diffusing out of the river) depending on whether the water is less than fully saturated with oxygen or more than fully saturated. The rate of change of oxygen concentration (*dO/dt*) and *D* are known from the oxygen record. Therefore, using a simple linear regression on data points collected throughout the night it is possible to estimate *R* and *k* as the y-intercept and slope of the regression line, respectively (Figure 1). We recommend using this approach when using ecosystem metabolism to assess river ecosystem health.
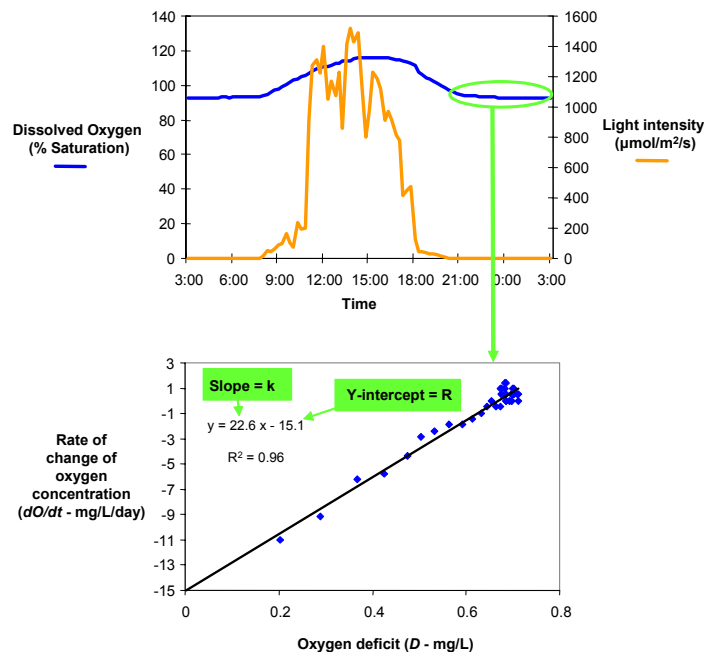
Figure 1.   Typical changes in dissolved oxygen over a 24-hour period and a demonstration of how the oxygen record is used to calculate the respiration rate and the reaeration coefficient.

A second method to calculate the reaeration coefficient using the oxygen record relies on a mathematical solution of the equation describing changes in dissolved oxygen concentration at a site over a 24 hour period. This mathematical solution shows that the reaeration coefficient is dependent on only two variables; the time lag between solar noon and when the minimum dissolved oxygen deficit occurs, and the length of the daylight period (Chapra & DiToro 1991). McBride (2002) has provided a logistic equation to describe this relationship.

$$k = 7.5 \left(\frac{5.3\eta - \phi}{\eta\phi} \right)^{0.85} : \eta = \left(\frac{f}{14} \right)^{0.75}$$ (4)

where k is the reaeration coefficient at 20°C, ϕ is the time lag between solar noon and maximum DO, η is the photoperiod correction factor, and f is the photoperiod or day length. This procedure seems to work well in rivers with a relatively low reaeration coefficient, but is very sensitive to error in the calculation of the time lag in streams with high reaeration coefficients, which will have short time lags. Using the example presented in Figure 1, the peak light intensity occurred at 13:45. The highest oxygen saturation was 116% and was constant at this level from 14:30 to 16:00. Therefore the time lag was somewhere between 0.75 and 2.25 hours. Assuming a day length (photoperiod) of 12 hours, the reaeration coefficient estimated from equation (4) varied from 9 - 33 day^{-1}, which is substantial variability. If an average time lag of 1.5 hours is assumed, the reaeration coefficient is estimated to be 15.8 day^{-1}, which is reasonably close to the value (22.6 day^{-1}) calculated using the method presented in Figure 1. This technique to estimate the reaeration coefficient is used in WAIORA, a low flow decision support system developed by NIWA (McBride et al. 1998).

An alternative approach to measuring the reaeration coefficient is to use empirical equations from the literature which use mean reach depth and mean velocity to estimate reaeration coefficients. A variety of equations have been suggested in the past (Table 2; Wilcock 1982). If this approach is used then we recommend the O'Connor-Dobbins, Owens-Edwards-Gibbs, or Bennett-Rathbun equations; these appear to perform reasonably well at sites with relatively low reaeration coefficients (<50 day^{-1}), but not so well in small turbulent streams (Young & Huryn 1999)

Table 2. Empirical velocity depth equations for calculating k (base e, day^{-1}). U is mean stream velocity (m/s), H is mean stream depth (m). (modified from Wilcock 1982).

Authors	Formulae
O'Connor-Dobbins	$3.74\, U^{0.5} / H^{1.5}$
Churchill-Elmore-Buckingham	$5.01\, U^{0.969} / H^{1.673}$
Isaacs-Gaudy	$4.75\, U / H^{1.5}$
Langbein-Durum	$5.13\, U / H^{1.33}$
Negulescu-Rojanski	$10.9\, (U / H)^{0.85}$
Owens-Edwards-Gibbs	$5.33\, U^{0.67} / H^{1.85}$
Bennett-Rathbun	$5.59\, U^{0.607} / H^{1.689}$

The most accurate method of estimating the reaeration coefficient is to directly measure the diffusion of an inert tracer gas (e.g. propane, methyl chloride, SF_6) through the surface of the study reach (Wilcock 1984; Wanninkhof et al. 1990; Marzolf et al. 1994;

Young & Huryn 1999). This method is necessary in small turbulent streams where the other techniques work poorly. However, it requires a significant amount of effort and equipment and therefore is probably outside the scope of most regular stream health monitoring programmes.

4.1.4   *Response parameters*

The key parameters derived from measurements of ecosystem metabolism are the rate of gross primary production (GPP) and rate of ecosystem respiration (ER). The ratio of these two parameters (GPP:ER or just P:R) is also very informative since it gives an indication of the food base of the river, or in other words the reliance of the river on algal material produced within the study reach, versus other material either from upstream or the surrounding catchment. If the ratio is >1 then the river may be entirely fuelled by organic carbon produced within the study reach, whereas if the ratio is <1 then at least some organic carbon from other sources is being transported into the reach before being respired. If the P:R ratio is <0.5 then the river ecosystem is being primarily supported by organic carbon from terrestrial sources in the surrounding catchment (Meyer 1989). Net ecosystem metabolism (NEM) is calculated as the difference between gross primary production and ecosystem respiration (i.e. GPP – ER) and has also been widely used as an indicator of the food base of the study reach.

4.2   Data analysis options

The reaeration coefficient, gross primary productivity and ecosystem respiration can be calculated using a relatively simple Excel spreadsheet that is available from Roger Young and will be converted into a more user friendly system during the second year of this project. Oxygen (mg/L and % Saturation) and temperature data for a 24 hour period are copied into the spreadsheet, along with light data (if available). The spreadsheet automatically calculates the rate of change of oxygen concentration and the oxygen deficit at regular intervals ($\leq$15 min) throughout the day. As mentioned above (Section 4.1.3), the reaeration coefficient and respiration rate is calculated from a regression of the oxygen deficit against the rate of change of oxygen concentration during the night time (Figure 1). Gross primary production at intervals throughout the daytime is then calculated from the following equation:

$$GPP = \frac{dO}{dt} + R - kD \qquad (5)$$

where *dO/dt* is the rate of change of oxygen concentration, *R* is the respiration rate, *k* is the reaeration coefficient and *D* is the oxygen deficit. Daily GPP is calculated by summing up the GPP within each interval throughout the day. Corrections for the changes in *R* and *k* with temperature throughout the day are also incorporated in the spreadsheet.

If changes in light intensity at the water surface are measured along with oxygen concentrations then a plot of instantaneous light intensity versus GPP may be very useful for determining if light, or some other factor, is limiting production rates. If GPP increases linearly with light intensity then light is limiting production rates. On the other hand, if GPP initially increases with light intensity before reaching a plateau then some other factor, such as nutrient availability, algal biomass or temperature, is probably limiting primary production rates (Figure 2).
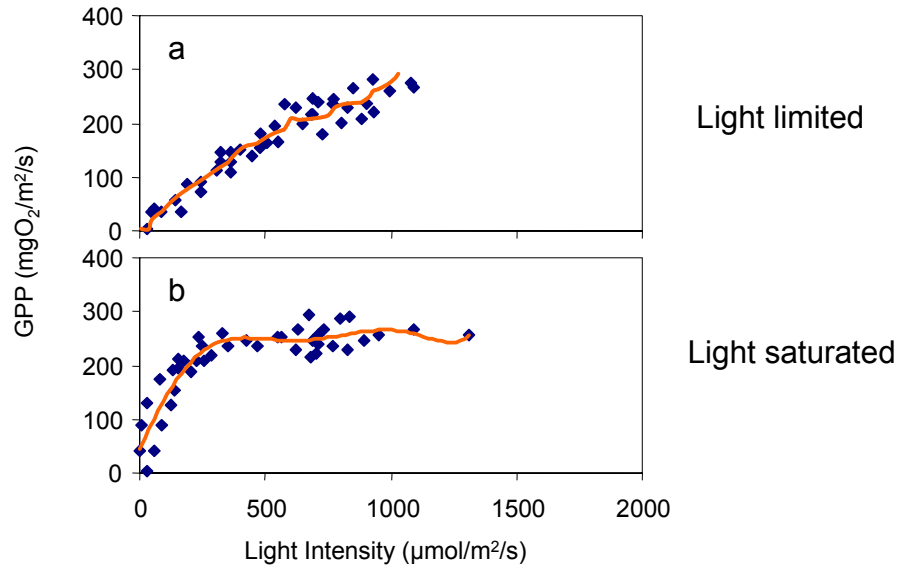
Figure 2. Response of gross primary productivity (GPP) to light intensity in situations where a) light is limiting production rates and b) where light is saturated and some other factor is limiting rates of photosynthesis.

4.3 Advantages and disadvantages of using ecosystem metabolism

Ecosystem metabolism has several specific advantages in addition to the general advantages associated with being a functional indicator (Section 2.2). These include:

- The natural movement and mixing of water in a river means that the measurements are representative of the entire reach and cover the range of habitat types present even though the oxygen concentrations used to calculate metabolism are only measured at one or two specific positions at a site.

- Metabolism measurements are made using dissolved oxygen concentrations and are closely associated with oxygen dynamics within a river. Therefore, even the raw oxygen measurements are of interest in determining river ecosystem health. Associated with this is the fact that the need to measure oxygen concentrations in rivers is easily explained to members of the general public - something that is not so easy in the case of leaf litter processing.

- Metabolism directly assesses the balance between supply and demand of energy in river ecosystems and thus gives an indication of what 'fuels' the ecosystem.

- A study of metabolism at a particular site could be planned, conducted and completed within just a couple of days, assuming the necessary equipment was available.

Unfortunately there are also some disadvantages, which include:

- At least one data-logging oxygen meter is ideally required to make metabolism measurements and this equipment is reasonably expensive. A basic oxygen meter with no data-logging facility could potentially be used, but the need for regular

measurements throughout the night makes this option tiring and unpleasant. Many regional councils already have their own oxygen logging equipment, so this should not be too much of a problem for regional council staff. However, the availability of suitable equipment for other groups involved with environmental monitoring is a concern.

- Measurements of oxygen concentration are required over at least a 24 hour period; therefore equipment is normally left unattended at the sampling site for an extended period of time. The likelihood of theft, vandalism or sudden changes in flow needs to be considered before leaving expensive equipment unattended.

- Metabolism measurements can be made in most rivers and streams relatively easily using just one oxygen logger. However, in small, turbulent systems with low productivity, considerably more effort and equipment is required due to uncertainties with measuring oxygen exchange through the water surface (see Young & Huryn 1999). These requirements limit the feasibility of making metabolism measurements for routine environmental monitoring in these types of streams.

- The fact that oxygen measurements integrate ecosystem processes over a relatively wide area and range of habitats is generally an advantage, but could be a disadvantage if the aim of a study was to address the health of specific habitats. If only one oxygen meter is used, the extent of the area of river bed upstream that influences the oxygen concentrations, and metabolism measurements, is not clear. If this is a concern, then two oxygen loggers can be used to calculate metabolism that applies only to the reach of river between the oxygen loggers.

4.4 Summary of controlling factors and likely response to stressors

Ecosystem metabolism can be influenced by a wide variety of factors, some that vary naturally and others that are strongly influenced by human disturbance of ecosystems. Some factors affect both GPP and ER, while others only influence one of these parameters Table 3 summarises the likely responses to various environmental stressors found in New Zealand. A detailed review of the literature used to construct this table is presented in Section 4.5 for those who require more information on particular controlling factors.

Table 3. Likely responses of primary productivity (GPP) and ecosystem respiration (ER) to a range of stressors commonly occurring in New Zealand.

Stressor	Change	Response	Comments
Temperature	Warmer water	Increase ER	Only weak evidence for this
Sediment	Higher turbidity	Decrease GPP	If river depth is sufficient to limit light
Sediment	More moveable substrate	Decrease GPP	
Sediment	Block connection with hyporheic zone	Decrease ER	
pH	Acid conditions	Decrease GPP & ER?	
Nutrients	Nutrient enrichment	Increase GPP and ER	
Organic pollution	Input of organic waste	Increase ER	Possible increase in GPP too, if nutrients

			released
Toxic chemicals	Toxic inputs	Decrease GPP and ER	May be offset by nutrients in toxic discharge
Riparian vegetation	Loss of stream-side vegetation increasing light availability	Increase GPP	ER may also increase if system is dominated by algal respiration
Riparian vegetation	Increase organic matter inputs	Increase ER	
River regulation	Decrease flows and flow fluctuations	Increase GPP and ER	
Channelisation	Loss of habitat heterogeneity	Decrease GPP and ER	
Flow fluctuations	Floods	Decrease GPP strongly, decrease ER a little, decrease P/R	
Flow fluctuations	River drying	Increase GPP, P/R	
Aquatic plant management	Plant removal	Decrease GPP and ER	Only if macrophytes are major contributors to metabolism

4.5 Literature review of factors controlling ecosystem metabolism

As mentioned above, the components of ecosystem metabolism are influenced by a range of factors. This section reviews the large body of existing knowledge on the factors controlling metabolism. We initially focus on factors that primarily vary naturally and then move to factors influenced more directly by human disturbance. This section is reasonably detailed and readers are encouraged to focus on Table 3 for an initial overview of the factors controlling ecosystem metabolism. However, if more information is required then this section is the place to look.

4.5.1 *Position in the river hierarchy*

One of the key predictions of the river continuum concept (Vannote et al. 1980) is that the ratio of GPP to ER should change in a predictable manner from the headwaters to the lower reaches of natural river systems. In forested headwaters, dense shading from surrounding trees is predicted to restrict primary productivity and the P/R ratio will be much less than 1. Further downstream as the channel widens, more light will reach the river surface promoting primary productivity on the river bed, leading to P/R ratios of around 1. In the lower reaches of very large rivers, primary production is predicted to be limited by the depth and turbidity of the water, leading to a reduction in the P/R ratio.

These predictions have been tested in many locations and generally have been supported by data (Naiman 1983; Bott et al. 1985; Chessman 1985; Naiman et al. 1987; Minshall et al. 1992; McTammany et al. 2003), although there have been some fundamental differences in grassland/prairie systems (Wiley et al. 1990; Young & Huryn 1996) and in rivers with strong floodplain connections (Junk et al. 1989; Meyer & Edwards 1990). In grassland/prairie river systems, the headwaters are not shaded and have high rates of primary production and thus high P/R ratios. P/R ratios decline downstream in these systems in response to increased river depth and turbidity (Wiley et al. 1990; Young et al. 1996). The lower reaches of rivers with strong floodplain connections are heavily influenced by organic matter from the surrounding floodplain and may have very high respiration rates and thus extremely low P/R ratios (Meyer & Edwards 1990).

4.5.2 *Bed morphology and substrate*

The most important effect of bed morphology on ecosystem metabolism relates to the size of the hyporheic zone beneath the stream bed. This zone is where exchange occurs between surface waters and groundwaters. A large proportion (50-85%) of total ecosystem respiration occurs in the hyporheic zone (Grimm & Fisher 1984; Naegeli & Uehlinger 1997; Fellows et al. 2001) so streams with a large hyporheic zone have markedly higher respiration rates (and lower P/R ratios) than similar streams with smaller hyporheic zones (Mulholland et al. 1997; Fellows et al. 2001; Mulholland et al. 2001).

Large substrate particles are less likely to be moved during floods than smaller particles and therefore provide a more stable surface for algal biomass to accumulate, potentially allowing faster rates of production (Biggs et al. 2001). Studies comparing rates of production on rock and fine sand enclosed within experimental chambers have found higher production rates on rock (Rosenfeld & Roff 1991; Rier & King 1996). It is not clear if these results can be extrapolated to a whole-stream scale, but presumably if all other controlling factors were equal, higher rates of primary production would be expected in streams with large, stable substrate particles.

Several studies have suggested a positive relationship between respiration rates and the amount of fine sediments on the stream bed (Hedin 1990; Hill et al. 1998). The likely mechanism for this response is unclear, but may be related to increased organic matter in fine sediments, or perhaps an increase in surface area available to be colonised by microbes.

Recent research has suggested that substrate heterogeneity can also influence rates of GPP and ER (Cardinale et al. 2002). Experimental riffles with artificially high substrate heterogeneity had higher rates of GPP and ER than natural riffles, or riffles with artificially low heterogeneity. The authors attributed their results to alterations in flow velocities and turbulence intensity near the stream bed resulting from different levels of physical habitat heterogeneity (Cardinale et al. 2002).

4.5.3 *Flow fluctuations*

Increased shear stresses near the river bed associated with increases in flow can be sufficient to slough algae from the bed (Peterson 1996). High flows often also carry high concentrations of abrasive sediment that is capable of stripping algae from substrate (Peterson 1996). Both primary production and ecosystem respiration are substantially reduced by high flows and then recover during the subsequent inter-flood period. However, primary production appears to be more sensitive to flow fluctuations than ecosystem respiration, and so bed-moving high flows tend to result in reductions in the P/R ratio (Young & Huryn 1996; Uehlinger & Naegeli 1998; Uehlinger 2000; Uehlinger et al. 2003).

River drying will also alter rates of metabolism. Molla et al. (1996) showed that a Mediterranean stream switched from a heterotrophic system (P/R <1) during periods of continuous surface flow to an autotrophic system (P/R >1) when flows declined, leaving a series of superficially isolated pools with higher productivity. In another study, Hill & Gardner (1987) compared metabolism throughout a year in two Texan streams, one with perennial flow and one with intermittent flow. No significant difference in productivity was found between the two streams, suggesting that primary producers are able to recover quickly after dessication – productivity in the intermittent stream, following four

months without flow, returned to levels comparable with the perennial stream within one month. In contrast, ecosystem respiration was significantly lower in the intermittent stream and recovered more slowly after stream drying (Hill & Gardner 1987).

4.5.4 *Temperature*

Increases in temperature, up to a certain tolerance limit, are expected to enhance rates of primary production and ecosystem respiration (Phinney & McIntire 1965). However, the effects of temperature at an ecosystem scale appear to be weak (DeNicola 1996; Mulholland et al. 2001). We are not aware of any studies that have conclusively shown a link between primary production rates and temperature in natural streams, although several studies have suggested modest effects of temperature on ecosystem respiration (Bott et al. 1985; Hill & Gardner 1987; Hedin 1990; Howarth et al. 1992; Sinsabaugh 1997; Hill et al. 1998, 2000, 2002) and there are abundant data showing significantly higher respiration in summer compared to winter (see review by Webster et al. 1995).

4.5.5 *Nutrients*

Nutrients can stimulate both primary productivity and ecosystem respiration (Odum 1956; Bott et al. 1985; Bowden et al. 1992; Guasch et al. 1995; Hill et al. 2000; Mulholland et al 2001). Fertilisation of an arctic tundra stream with phosphate alone, or with ammonium and phosphate together, increased photosynthesis and respiration rates (Bowden et al. 1992). Similarly, experimental channels in a Spanish stream that were enriched with nitrogen, phosphorus and potassium had higher primary production and respiration rates than control channels (Guasch et al. 1995). In a comparison of respiration in 371 streams across different parts of the U.S.A., Hill et al. (2000) found that respiration rates were strongly correlated with nutrient concentrations. In a comparison of 8 streams across North America, Mulholland et al. (2001) found that dissolved phosphorus concentrations explained a significant amount of the variation in both GPP and ER among the streams. However, nutrient concentrations were of secondary importance, behind light and size of the hyporheic zone, in explaining the respective variations in GPP and ER (Mulholland et al. 2001). In New Zealand, the effects of different nutrient levels also appear to be of secondary importance, with nutrient concentrations rarely included in models explaining variation in GPP or ER among streams (Young 1998). Nevertheless, sites with high nutrient inputs are often characterised by very high rates of GPP and ER (Wilcock et al. 1995; 1998).

Recent research has indicated that metabolism rates are an indicator of biotic demand for nutrients (Hall & Tank 2003). Streams with high metabolism rates, particularly GPP, have high nitrogen uptake rates and hence can control concentrations of nutrients downstream (Hall & Tank 2003). So, although high nutrient concentrations can stimulate GPP and ER, high rates of metabolism can reduce concentrations of dissolved nutrients.

4.5.6 *pH*

The effects of pH on metabolism rates have generally received little study. Niyogi et al. (2002) examined the effects of acid mine drainage on primary production rates in Rocky Mountain streams in the U.S.A. and found a negative relationship between primary production rates and an index of acid mine stress. However, this pattern appeared to be driven by deposition of metal oxides in many of the acidic streams, which reduced primary production rates, rather than a direct effect of low pH. The few sites with low

pH and low metal oxide deposition had primary production rates similar to nearby pristine streams. Niyogi et al. (2002) observed high algal biomass in some of these acidic streams not affected by metal oxide deposition and suggested that this may be a response to a lack of grazing pressure, since many of the grazing invertebrates are very sensitive to low pH and are not present in stressed sites.

4.5.7   *Turbidity/Suspended sediment*

Turbidity potentially influences metabolism rates through two mechanisms – firstly reducing the amount of light passing through the water column and reaching primary producers on the river bed, and secondly smothering primary producers on the bed. Suspended sediment may also have abrasive properties and strip algae from the bed during high flows (Peterson 1996). Davies-Colley et al. (1992) showed a reduction in periphyton production rates downstream of a clay discharge and attributed this to a reduction in the amount of light reaching the river bed. Similarly, Young & Huryn (1996) showed that primary production in the lower reaches of the Taieri River was limited by light availability during a wet year when the lower reaches were turbid, whereas primary production rates were much higher during a dry year when the lower reaches were clear. Respiration rates were not influenced by turbidity to the same extent as productivity, so the P/R ratio was <1 under turbid conditions, but ≈1 when the water was relatively clear.

As mentioned above, the extent of the hyporheic zone is important for controlling rates of ER. Therefore, an increase in sedimentation has the potential to block the connection between surface waters and the hyporheic zone (Boulton et al. 1997) and thus decrease ER. However, we are not aware of any studies that have examined this possibility.

4.5.8   *Light*

The amount of light reaching primary producers on the stream bed appears to be the main factor influencing rates of GPP in rivers. Many factors control the amount of light reaching a particular reach of a stream or river and include the amount and type of riparian vegetation, orientation of the valley and slope of the banks. The amount of light passing through the water column will depend on water clarity. Light input will also vary seasonally with changes in day length and sun angle. Shading from riparian vegetation is particularly important and changes associated with leaf fall or riparian vegetation removal can have dramatic effects on stream metabolism. For example, Hill et al. (2001) measured metabolism throughout a year in two deciduous forest streams in eastern Tennesee. Rates of GPP declined by 75% after leaves emerged and shaded the stream. Comparisons of streams with different amounts of canopy cover have also emphasised the strong relationship between light availability and primary production at a broader scale (Naiman 1983; Bott et al. 1985; Webster et al. 1995; Young 1998; Young & Huryn 1999; Mulholland et al. 2001). Light intensity is unlikely to affect respiration rates directly, but there may be correlations with respiration rates in streams where respiration is predominantly associated with algal biomass, which may be abundant in well-lit streams (Bunn et al. 1999).

4.5.9   *Riparian vegetation*

The amount and type of riparian vegetation has an important role in controlling the light climate of streams. Reductions in canopy cover can have profound effects on rates of GPP and stream health (Bunn et al. 1999). Streams with intact riparian cover generally

have P/R ratios considerably less than 1, but after vegetation clearance P/R ratios may exceed 1, indicating a change from an ecosystem reliant on organic matter from outside the system to one where the majority of organic matter is produced on site. In a Queensland river, Bunn et al. (1999) showed that a canopy cover of >75% ensured that metabolism rates were similar to those in undisturbed forest streams, while a reduction in canopy cover below 40-50% resulted in increased GPP and ER and severe reductions in stream health associated with a change from palatable microalgae to inedible filamentous algae and macrophytes.

The type of riparian vegetation may also affect rates of respiration via alterations in the supply and type of organic matter delivered to the stream (Hedin 1990). Deciduous trees will drop large amounts of organic matter into streams in a short period. This large input of organic matter provides an important food resource in streams where the inhabitants are able to process the material. However, leaf-eating invertebrates are uncommon in many New Zealand streams and large inputs of willow leaves, for example, can increase respiration rates and lead to low dissolved oxygen concentrations if they are not flushed out of the system.

4.5.10 *Keystone species*

Metabolism does not appear to be influenced to the same extent as leaf litter processing by the presence or absence of particular species. Nevertheless, trophic cascades, where changes at one trophic level influence other trophic levels, are well-known in the scientific literature. Biggs et al. (2000) compared periphyton biomass and production among six streams with different top-level predators (brown trout and *Galaxias*) to determine the strength and implications of trophic cascading observed in earlier experiments in artificial stream channels (Flecker & Townsend 1994). Periphyton biomass was significantly higher in the trout streams than in the *Galaxias* streams, as had been found previously, but this did not translate into differences in production rates. However, in a more intensive study at two of the same sites, Huryn (1998) observed a 6-fold difference in annual net primary production between a trout stream and a *Galaxias* stream. It is possible that other factors may also have been responsible for this difference in primary production rates between streams, but the results show that the presence or absence of a particular species may be important.

4.5.11 *Organic pollution*

Discharge of organic pollutants into rivers can lead to prolific growths of sewage fungus and filamentous green algae and increased respiration rates (Rutherford et al. 1987). Quinn & McFarlane (1989) showed that respiration rates downstream of several waste water discharges on the Manawatu River ranged from $12.5 - 37.6$ $gO_2/m^2/day$, compared with a maximum respiration rate of only 18 $gO_2/m^2/day$ upstream of the discharges. The effects of organic pollution on GPP are not so clear with some studies suggesting inhibition of primary production in reaches just downstream of waste water discharges (Odum 1956; Rama Rao et al. 1979). Urban streams are often characterised by higher rates of GPP and ER than neighbouring forested streams, but many still have low P/R ratios that appear to be the result of inputs of organic pollution such as sewage effluent (Paul & Meyer 2001).

4.5.12   Toxic chemicals

High concentrations of toxic chemicals, such as metals, generally appear to slow rates of GPP and ER. For example, Hill et al. (1997) showed that GPP and ER decreased significantly (GPP, 10.9 to 0.83 $gO_2/m^2/day$; ER 0.65 to 0.02 $gO_2/m^2/day$) between reference and impacted sites in a Rocky Mountain river affected by elevated metal concentrations. As mentioned earlier, Niyogi et al. (2002) also studied primary productivity in Rocky Mountain streams impacted by acid mine drainage. High metal concentrations impaired rates of primary production in their study too, although this appeared to be due to physical stress associated with deposition of metal oxides rather than a direct toxic effect of the dissolved metals. The opposite pattern was observed by Crossey & LaPointe (1988) who observed an increase in GPP and ER, and a decrease in the P/R ratio, at sites downstream of a discharge of heavy metals. However, nutrient concentrations were also elevated slightly by the discharge and may have been responsible for this different response. Other toxicants, such as an increase in level of salinisation, have also been shown to depress rates of GPP and ER (Davies 2003)

There is also evidence that some toxicants affect GPP in a different way to ER. For example, Maki & Johnson (1976) found that a lampricide (TFM) suppressed GPP by 25-50%, but increased ER by 3-50%.

4.5.13   River regulation and confinement

Uehlinger et al. (2003) measured ecosystem metabolism in a Swiss river downstream of a large dam before and after an experimental flood. The large dam had substantially changed the flow regime of the river from the natural state, leaving only a small residual flow that lacked the power to transport coarse sediments. The river ecosystem downstream of the dam was dominated by dense algal mats and moss beds due to the lack of flushing flows. The experimental floods were an attempt to improve the habitat quality of the river and reduced GPP and ER by 64 and 36 %, respectively.

In a comparison of a regulated and unregulated reach of an American river, Munn & Brusven (2004) found higher rates of GPP and ER downstream of a large dam. They attributed the difference to extensive growth of aquatic moss in the regulated reach. The lack of gravel recruitment past the dam resulted in a stable armoured stream-bed, providing good conditions for moss growth, despite regular and large fluctuations in flow

Channelisation of rivers is also expected to result in changes to ecosystem metabolism. Gelroth & Marzolf (1978) showed a major difference in GPP in natural versus channelised reaches of a Kansas stream, with a four-fold increase in GPP in the channelised reach. Ecosystem respiration was not affected to the same extent, so the P/R ratio was also much higher in the channelised reach. This response may have been due to changes in habitat heterogeneity associated with channelisation, but was more likely due to the loss of riparian vegetation and associated shade along the channelised reach (Gelroth & Marzolf 1978).

4.5.14   Aquatic plant management

Prolific growth of aquatic macrophytes can lead to problems with hydraulic efficiency of waterways and also lead to large diel changes in dissolved oxygen concentrations. Therefore, removal of aquatic plants is commonly undertaken to address these issues. Removal of large amounts of plant material would be expected to have a profound effect

on rates of metabolism. In a study of two Swiss streams before and after macrophyte removal, Kaenel et al. (2000) measured a 70% decline in GPP and ER in one stream that was associated with macrophyte removal. However, neither GPP nor ER was significantly affected in the other stream. Wilcock et al. (1999) also found no substantial difference in GPP or ER in a New Zealand stream after macrophyte removal, while Simonsen & Harremoes (1978) reported that GPP and ER continued to increase despite manual plant removal and was only reduced once mechanical dredging was used to clear their study streams. These variable responses to macrophyte removal suggest that in some streams macrophytes contribute only a small proportion of the ecosystem metabolism. Other less conspicuous primary producers such as benthic algae appear to make a larger contribution to rates of GPP and ER despite their relatively low biomass.

4.5.15 *Multiple stressors*

Many of the stressors listed above will often occur together and may either complement or counteract each other. For example, agricultural development is often associated with removal of riparian vegetation, and increased nutrient and sediment delivery to streams. Riparian vegetation removal will increase light available for primary production and this effect will be enhanced further by an increased supply of nutrients. High rates of GPP and ER have been observed in agricultural streams and rivers in response to abundant light and nutrient levels (Wiley et al. 1990; Wilcock et al. 1998; Young & Huryn 1999). However, increased concentrations of suspended sediment and turbidity will tend to counteract these effects and lead to declines in GPP at sites where the combination of water depth and turbidity restricts the amount of light available at the riverbed (Wiley et al. 1990; Young & Huryn 1996). Similar contradictory effects may occur with waste discharges where organic waste will tend to increase ecosystem respiration rates, but toxic industrial waste may reduce rates of GPP and ER (Rama Rao et al. 1979).

4.6 Case Study B – the health of Otago's large lake-fed rivers

Consider the following example, which uses measurements of ecosystem metabolism to compare the health of the large rivers flowing out of the Southern Lakes. The size of these rivers makes it difficult, and in some cases dangerous, to sample macroinvertebrates or periphyton across the full variety of habitat types present. Invertebrate samples that have been collected in these rivers in the past have consistently given low MCI scores which would normally indicate moderate to severe pollution (ORC 2000). However, these results were presumably due to the lake-fed nature of these rivers reducing macroinvertebrate diversity, rather than poor ecosystem health. In fact these rivers have very high water quality, among the highest fish densities in the country (Teirney & Jowett 1990) and would be considered very healthy by most people.

> *Measurements of ecosystem metabolism in 4 lake-fed rivers were carried out in summer 1997. A single oxygen logger was safely deployed in a wadeable area of each river for just over 24 hours and was programmed to measure oxygen concentration, conductivity and water temperature every 15 minutes. Reaeration coefficients were calculated using the oxygen record during the night time. Gross primary production ranged from 1.2 to 11.5 $gO_2/m^2/day$, while ecosystem respiration ranged from 0.9 to 4.6 $gO_2/m^2/day$. The P/R ratios were high and >1 in three of the four rivers that were monitored, reflecting the stable substrate and flows that are normally characteristic of these rivers. However, in the fourth river the P/R ratio was only 0.7 suggesting that the ecosystem in this river was functioning quite differently to that in the other rivers. This 'different' river was the only one with an artificially controlled flow regime and had more regular and*

larger changes in flow than the other rivers. Although these were only one-off measurements, this difference in the food-base of the river ecosystem indicates a reduction in ecosystem health and life supporting capacity in the modified river. Incidentally, anglers have reported that fish populations in this particular river are very low and rated it poorly compared to other lake-fed rivers in the region.

5. HOW DO THESE MEASUREMENTS RELATE TO ECOSYSTEM HEALTH?

As with any indicator, guidance is needed about what the measurements mean in terms of ecosystem health, ideally with specific values that indicate a likely transition from 'good' to 'poor' ecosystem health.

5.1 Leaf litter decomposition

Gessner & Chauvet (2002) have tentatively proposed a framework for assessing functional stream integrity using leaf litter processing rates (Table 4). This framework includes two approaches, firstly a reference site approach where the results at test sites are compared with those at appropriate reference sites. For example, a leaf litter decay rate at a test site that is within 30% of the decay rate at reference sites would indicate good ecosystem health, whereas a decay rate of <50% or >200% of that at the reference sites would indicate severely impaired health. Values between these extremes would indicate more mild effects on ecosystem health. Scores could be assigned to each of these criteria as a simple way of indicating the health of different sites (Table 4).

The alternative approach is to set absolute values, which can be compared with test site results (Table 4). For example, Gessner & Chauvet (2002) have suggested that leaf litter processing rates between 0.01-0.03 generally indicate good ecosystem health, while values outside this range indicate either mild or severe effects on ecosystem health (Table 4). However, it is impossible to set sensible absolute values without information on the type of leaves used, or the characteristics of the sites tested. Nevertheless, after classifying rivers into different types and gaining sufficient information on breakdown rates in reference streams within each river type, it would be possible to devise a series of appropriate absolute values that could be used for comparison with test sites of the same type. The New Zealand river environment classification system (Snelder et al. 2004) would be ideal for this purpose.

Table 4. Framework for assessing functional stream integrity using leaf litter processing rates. Modified from Gessner & Chauvet (2002).

Method	Assessment parameter	Criterion	Score
Comparison with reference	Ratio of breakdown rates at test (k_t) and reference (k_r) sites	$k_t:k_r = 0.75\text{-}1.33$	2
		$k_t:k_r = 0.5\text{-}0.75$ or $1.33\text{-}2.0$	1
		$k_t:k_r < 0.5$ or >2.0	0
Absolute value	Breakdown rates at test site (day^{-1})	$kt = 0.01\text{-}0.03$	2
		$k_t = 0.005\text{-}0.01$ or $0.03\text{-}0.05$	1
		$k_t < 0.005$ or >0.05	0
Absolute values of ratio	Ratio of breakdown rates in coarse (k_c) and fine (k_f) mesh bags	$k_c:k_f = 1.2\text{-}1.5$	2
		$k_c:k_f = 1.5\text{-}2.0$ or <1.2	1
		$k_c:k_f >2.0$	0

To improve the sensitivity and robustness of leaf litter assays, Gessner & Chauvet (2002) also proposed using the ratio of leaf breakdown rates in coarse-mesh and fine-mesh bags (Table 4) since changes in this ratio would indicate shifts in the balance between the contribution of microorganisms and leaf-eating invertebrates to decay processes.

However, it is not clear if this would be an appropriate indicator in New Zealand given the low diversity and abundance of leaf-eating invertebrates in most New Zealand streams and rivers (Winterbourn et al. 1981). Gessner & Chauvet (2002) also suggested that the ratio of breakdown rates of fast decomposing and slow decomposing leaves might be a useful indicator.

5.2 Ecosystem metabolism

The framework proposed by Gessner & Chauvet (2002) for interpreting leaf decomposition rates seems like a sensible approach and could be adapted to cater for ecosystem metabolism measurements. To do this, metabolism data from a range of sources (Wiley et al. 1990; Young & Huryn 1996; Webster & Meyer 1997; Wilcock et al. 1998; Young & Huryn 1999; Mulholland et al. 2001; Hall & Tank 2003; McTammany et al. 2003) were examined and split into two groups; 'reference' sites that drain relatively natural catchments and 'impact' sites that drain intensive agricultural land. We used the distribution of data from the reference sites to develop some interim criteria for interpreting metabolism data (Figure 3 and 4, Table 5). Values between the lower and upper quartiles (25th – 75th percentiles) were arbitrarily considered to represent good health, as has been done previously for other biological indices (Gerritsen 1995; Barbour et al. 1996; Maxted et al. 2000). Values within this range are given a score of 2. For GPP, values less than the 25th percentile or between the 75th and 95th percentiles were considered to represent satisfactory health and given a score of 1, while values beyond the 95th percentile were considered to represent poor ecosystem health and have a score of 0 (Figure 3). For ER, a similar system was used, except that values below the 5th percentile, as well as above the 95th percentile, were considered to represent poor ecosystem health and given a score of 0 (Figure 4). This difference between the GPP and ER criteria is due to the fact that very low GPP values are not necessarily indicative of poor health – small, pristine forested streams, for example, may have very low rates of GPP. However, extremely low rates of ER are more likely to indicate problems with ecosystem health.
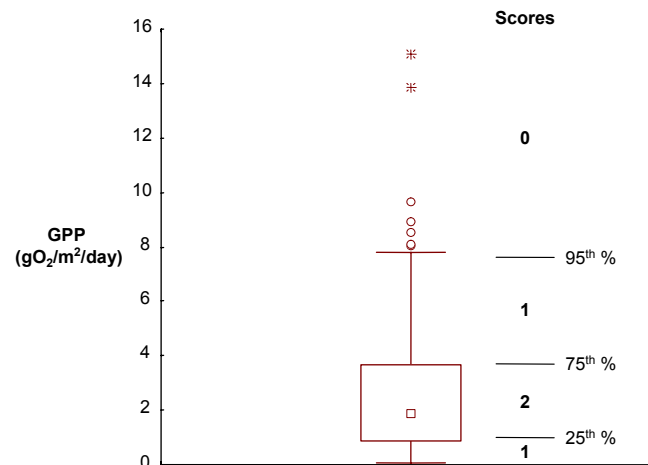
Figure 3. Distribution of rates of GPP from 'reference' sites and proposed criteria that could be used to determine the health of streams. Scores could be given to values fitting within each class as shown.
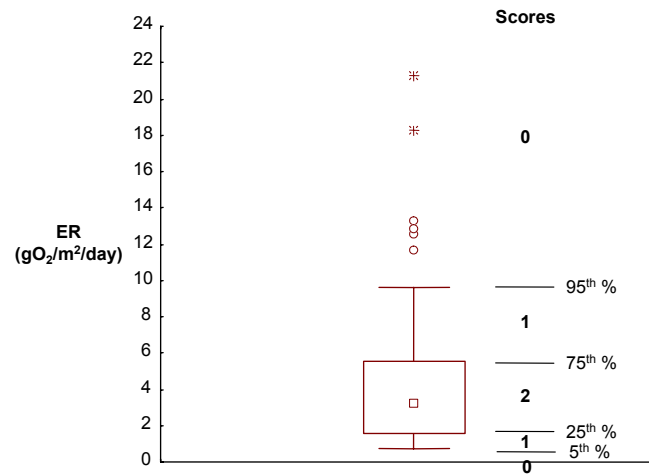
Figure 4. Distribution of rates of ER from 'reference' sites and proposed criteria that could be used to determine the health of streams. Scores could be given to values fitting within each class as shown.

Table 5. Framework for assessing functional stream integrity using metabolism data.

| Method	Assessment parameter	Criterion	Score
Comparison with reference	Ratio of GPP at test (GPP_t) and reference (GPP_r) sites	$GPP_t:GPP_r = 0.4\text{-}1.5$	2
		$GPP_t:GPP_r = 0.1\text{-}0.4$ or $1.5\text{-}3.0$	1
		$GPP_t:GPP_r <0.1$ or >3.0	0
	Ratio of ER at test (ER_t) and reference (ER_r) sites	$ER_t:ER_r = 0.4\text{-}1.4$	2
		$ER_t:ER_r = 0.2\text{-}0.4$ or $1.4\text{-}2.5$	1
		$ER_t:ER_r <0.2$ or >2.5	0
Absolute value	GPP at test site ($gO_2/m^2/day$)	$GPP_t = 0.8\text{-}4.0$	2
		$GPP_t = <0.8$ or $4.0\text{-}8.0$	1
		$GPP_t >8.0$	0
	ER at test site ($gO_2/m^2/day$)	$ER_t = 1.5\text{-}5.5$	2
		$ER_t = 0.7\text{-}1.5$ or $5.5\text{-}10.0$	1
		$ER_t <0.7$ or >10.0	0

GPP = gross primary production rates, ER = ecosystem respiration rates

As an example of how this framework may work, we compared the metabolism data from the 'reference' sites with data from the 'impact' sites (Figure 5). Almost all of the data from the impact sites indicated either poor or satisfactory health according to the criteria set using the reference site data. Data from only 2 sites (out of 51) indicated good ecosystem health.
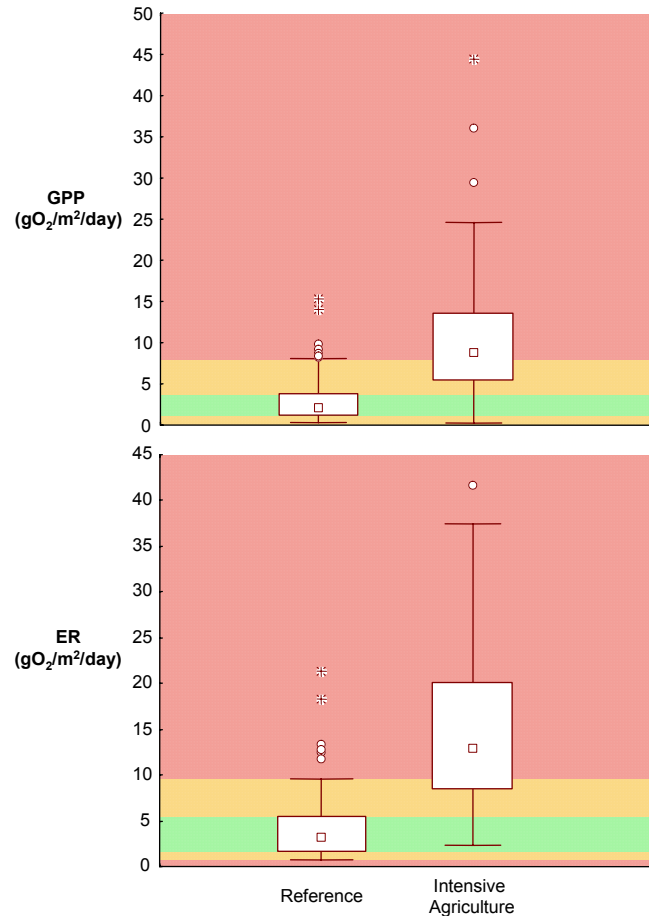
Figure 5. Comparison of rates of GPP and ER between 'reference' sites and 'impact' sites draining intensive agricultural land. The colour bands correspond to criteria from Table 5 – green = good health, orange = satisfactory health, red = poor health.

The framework for interpreting metabolism results that we have proposed here is very broad and could be tightened up considerably using data from appropriate local reference sites, rather than the broad range of relatively natural sites we used here. For example, the 'reference' sites used above included sites with both open and closed canopies. As mentioned earlier in this report, light is an important variable controlling rates of GPP and therefore the degree of shading is likely to have a large impact on GPP measurements. This effect is clearly shown in Figure 6 and demonstrates how reference sites with closed canopies have quite a different pattern of GPP and ER compared to larger or grassland reference sites with open canopies. A test of the impact of riparian vegetation clearance of a small forested stream, for example, clearly should use reference sites with closed canopies for comparison. There is also evidence in Figure 6 that streams and rivers with relatively stable flows (spring fed or lake outlets) have somewhat different rates of metabolism than other systems, and therefore any tests of environmental impacts in these types of systems may require information from unimpacted streams with stable flows.
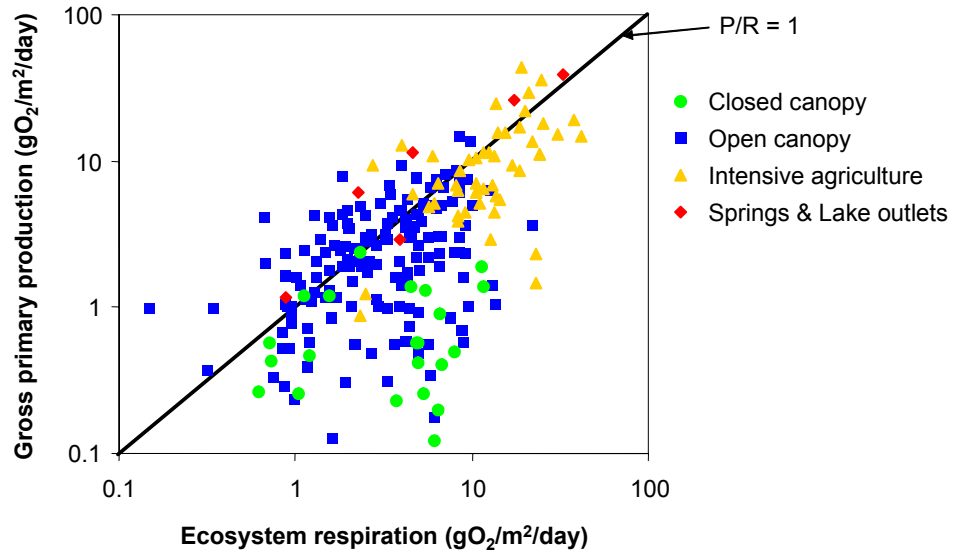
Figure 6. Plot of ER versus GPP for reference sites with closed and open canopies, 'impact' sites draining intensive agricultural land, and stable flow sites in spring-fed streams or lake outlets.

Other metrics, such as the P/R ratio, may also be useful for detecting particular types of environmental stress where the effects do not apply equally to both GPP and ER. For example, the P/R ratio did not show any clear difference between the 'reference' and 'impact' sites mentioned above (Figure 7a), but there was a clear difference in P/R between the closed canopy and open canopy reference sites (ANOVA, $F = 16.1$, $P<0.001$, Figure 7b).
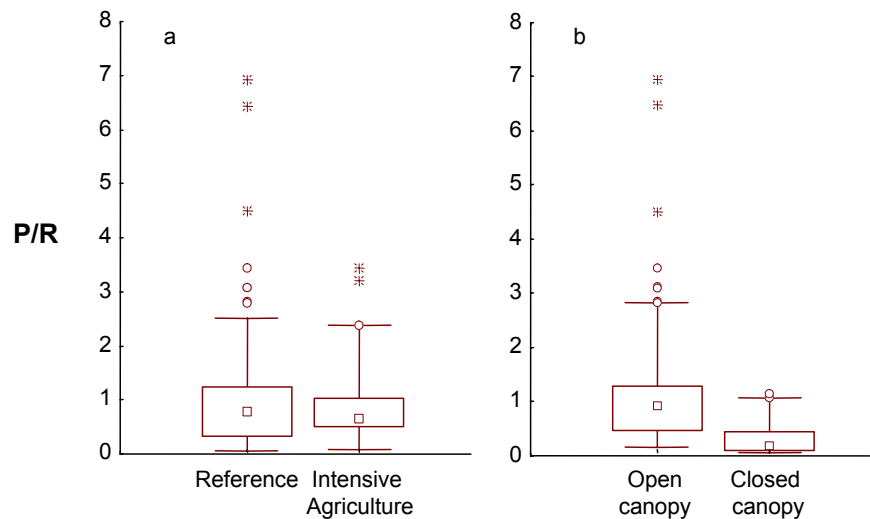
Figure 7. Comparison of the P/R ratio for a) reference versus impact sites and b) open-canopy and closed-canopy reference sites.

6.   SUMMARY

In this report we have introduced the concept of using functional indicators of river ecosystem health and focussed on two types of functional indicators that we believe have the most potential for use as regular monitoring tools – leaf litter decomposition and ecosystem metabolism.   The use of functional indicators may complement more traditional biomonitoring methods and improve monitoring programmes in a variety of ways including:

- a more complete picture of health is measured

- measurements provide an integrated measure of stream health over a moderate time period and across different habitat types

- the health of a range of habitats can be measured

- there is no reliance on the presence or absence of a particular set of species

- costly taxonomic expertise is not required

- functional measurements are often directly linked with the environmental stress so it may be straightforward to trace the cause of a problem

We then reviewed methods for measuring leaf litter decomposition and ecosystem metabolism and provided suggestions for how these methods could be used in regular monitoring programmes. A summary of the scientific literature on factors that control rates of leaf litter processing and ecosystem metabolism is also provided so that predictions can be made of the response of the indicators to a range of environmental stressors. The advantages and disadvantages of each of these indicators is also summarised.

We considered including detailed protocols in the report describing the methods used to measure leaf litter decomposition and ecosystem metabolism. However, we would prefer to receive feedback from the potential users about the suggested approaches before developing more detailed protocols. We imagine that protocols will evolve somewhat during the project as the practicalities of using the techniques for regular monitoring become more evident. Examples where the use of functional indicators have assisted monitoring efforts are included in the report as short case studies and provide further guidance on the practicalities of using these techniques.

Finally, we have suggested some preliminary criteria that could be used to interpret results from these indicators. We see these proposed criteria as only a starting point with substantial need for them to be refined to cater for conditions in different parts of the country, or particular types of rivers. The regional case studies that will be carried out in conjunction with council staff in the 2nd year of this project will assist with this. Lessons learnt during the case studies will be incorporated into a final report produced near the end of the project, to provide the basis for improved monitoring of river health in the future.

7. ACKNOWLEDGEMENTS

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