RIVER WATER QUALITY IN NEW ZEALAND: AN INTRODUCTION AND OVERVIEW

Robert J. Davies-Colley

National Institute of Water and Atmospheric Research (NIWA), PO Box 11-115, Hillcrest, Hamilton, New Zealand.

ABSTRACT: River water quality is important to uses and values within rivers and also in water bodies downstream. Water quality here encompasses: the physico-chemical attributes important to aquatic life and that vary diurnally (e.g. dissolved oxygen, temperature); optical attributes related to transmission of light through water (e.g. visual clarity); the major nutrient elements, nitrogen and phosphorus, that promote plant growth including nuisance algae (nitrate-N and ammonia-N are also toxic to aquatic life); and faecal microbial contaminants that can cause illness to people consuming water or recreating in rivers or downstream waters. River water quality is affected by both point-source pollution (wastewater discharge) and diffuse pollution from land use. In New Zealand, improved wastewater treatment over several decades has resulted in water quality being dominated by diffuse sources. The three major categories of diffuse pollution are (1) fine sediment, causing reduced water clarity and sedimentation of river beds and downstream water bodies, (2) the major nutrients (N, P), promoting aquatic plant (particularly nuisance algae) growth, and (3) faecal microbes, representing a hazard to human users of water or consumers of contaminated shellfish. Toxic metals may contaminate river waters draining relatively small areas, nationally, of urban and mining-affected land.

River water quality in New Zealand, compared with Europe, North America and Asia, may be described as 'fairly good', and is very good (i.e. supports most values including habitat for aquatic life) in rivers draining the conservation estate – lands reserved for ecological and recreation purposes. However, this must be qualified by recognition of widespread diffuse pollution from developed land, particularly pastoral agriculture, with fine sediment causing reduced visual clarity, faecal microbial contamination, and nutrient enrichment. Furthermore, river water quality depends strongly on flow state, and even rivers such as the Motueka (Tasman District, northern South Island), which is generally of 'good' water quality, are typically turbid and faecally polluted in stormflows.

Grazing of livestock (on 40% of New Zealand's land area) mobilises all three major categories of diffuse pollutant, with the result that rivers draining pastoral catchments are moderately degraded. Intensification of pastoral land use generally increases the severity of diffuse pollution. While the worst water quality, nationally, is to be found in (a relatively few) urban- or mine-affected rivers in which other contaminants such as toxic metals are elevated, other soil- and vegetation-disturbing land uses also adversely affect river water quality. For example, cropping (on 1.5% of the land area) mobilises nutrients and sediment. Water quality of rivers in plantation forests (7% of the land area) is generally appreciably better than that of rivers in pasture, and approaches the quality of rivers in native vegetation cover except for periodic 'disturbance' associated with harvest of the tree crop with associated removal of canopy cover. Fine sediment mobilisation remains an issue for plantation forestry.

Unfortunately the water quality of rivers in New Zealand has been declining for the last 25 years, despite the very large expenditure on improved treatment (or diversion from rivers) of city and factory wastewaters – a clean-up tracked by long-term monitoring showing the reduction in certain pollutants such as ammoniacal-N and biochemical oxygen demand. The gains from point pollution control have been negated by steadily increasing diffuse pollution, particularly nitrogen and phosphorus enrichment from intensification of pastoral agriculture. Fortunately, there have been recent encouraging signs that the decline in river health can be arrested, or even reversed, with stabilisation or improvement in a few (polluted) rivers in certain catchments and regions where there has been major effort on improved land management (e.g. riparian fencing and planting) and nutrient controls.

Diffuse pollution is much more difficult to manage than point pollution. It remains to be seen whether the recent freshwater reforms, particularly the national 'bottom lines' protecting secondary contact recreation and aquatic habitat, can provide the desired outcome in terms of improved water quality. Enduring challenges include improving river water quality despite continuing pressures towards pastoral agricultural intensification and urban expansion, and with the additional pressure of global warming driving increases in river water temperatures and declining river flows in some areas.

Some work has been done internationally on valuing rivers, but less on valuing their water quality, and little research has been done in New Zealand. Valuing river water quality seems likely to remain a formidable challenge because of the multiple values and uses of rivers convolved with the multiple dimensions and attributes of water quality that affect (or are affected by) those values.

Key words: diffuse pollution, faecal microbes, fine sediment, nutrients, point pollution, river values

INTRODUCTION

This chapter focuses on river water quality in the sense of water composition as it affects values and uses of rivers and downstream waters. It provides an update of Chapter 11 (on water quality and chemistry of running waters) in *Freshwaters of New Zealand* (Davies-Colley and Wilcock 2004). Since 2004, water quality science has advanced appreciably on several fronts. For example, water quality monitoring and reporting is being improved nationally with the NEMaR (National Environmental Monitoring and Reporting) project (McBride et al. 2013), water quality trends are now better documented and understood, modelling of water quality at the catchment scale has advanced, understanding of diffuse pollution has been improved, and a 'toolbox' of beneficial management practices has been compiled for mitigating pollution from pastoral agriculture (Quinn et al. 2009). There has also been considerable policy action on addressing water quality problems, notably the New Start for Fresh Water, the National Policy Statement for Freshwater Management 2011, the deliberations of the Land and Water Forum (culminating in three major reports), and the Freshwater reforms (MfE 2013). This chapter reviews water quality science and management emphasising the

Davies-Colley RJ 2013. River water quality in New Zealand: an introduction and overview. *In* Dymond JR ed. Ecosystem services in New Zealand – conditions and trends. Manaaki Whenua Press, Lincoln, New Zealand.

last decade, and looks briefly at prospects for valuing ecosystem services associated with water quality in rivers.

RIVERS – HYDROLOGICAL FLUXES

Rivers represent only 0.0002% of water on Earth and only 0.46% of surface fresh water (http://ga.water.usgs.gov/edu/earthwherewater.html 15/4/13). However, rivers are disproportionately important because they are *fluxes* of water when most other components of the hydrological cycle (including snow and ice, lakes, wetlands, and ground waters) behave more like *storages* of water. Rivers may be thought of as the main flux closing the hydrological cycle – returning to the sea a large fraction of rain falling on the land.

Along with water, rivers convey large fluxes of sediment and dissolved material to downstream water bodies (lakes, estuaries) and the sea. These dissolved and suspended constituents are the products of physical and chemical weathering, which over geological time wears down land to sea level. The *constituents* of river waters also relate to their *water quality* (Davies-Colley and Wilcock 2004), which can strongly influence or constrain certain values and uses of river waters. We are concerned with constituents in river waters, not merely as regards the condition of the rivers themselves, but as regards loads of materials and polluting impacts on downstream waters such as lakes, estuaries, and coastal waters.

Rivers are a major part of the cultural identity of New Zealanders. Indigenous New Zealanders (Māori) identify themselves by naming a particular mountain and river. More generally, recreation in and around rivers, including swimming, may be regarded as part of the cultural heritage of all citizens of Aotearoa. Such recreation, and the habitats of aquatic organisms that contribute to aquatic recreation, are strongly dependent on water composition and thus water quality.

DEFINITION OF WATER QUALITY

Water quality is a complex concept related to physical, chemical and biological characteristics of natural water (and therefore to its composition) rather than its level, volume or flow – which are collectively referred to as water *quantity*. However, water quality is not an absolute concept related to water composition alone, but must be defined by reference to how composition affects suitability-for-use of water, or, more widely, its ability to support water *values*. Based on careful consideration of several environmental terms in common use (but not commonly defined!), Johnson et al. (1997) propose the following definition of water quality:

A measure of the condition of water relative to the requirements of one or more species and/or to any human need or purpose.

That is, water quality relates to 'condition' of water as it affects biological habitat and human use. A working definition of water quality for the purposes of this chapter, that is broadly consistent with that of Johnson et al. (1997), but explicitly mentions values of water, is as follows:

The suitability of water composition for supporting a range of water values, including habitat for aquatic life and human uses including recreation.

Most recognised values of water are sensitive to water quality and some depend critically on water quality (as is discussed below). Typically, habitat for aquatic life and recreational use of waters are values most strongly demanding as regards water quality – which, of course, is why these values are explicitly mentioned in our working definition of water quality as given above. Recently, the Freshwater Reform report (MfE 2013) has proposed that suitability as habitat for aquatic life and secondary contact recreation are 'bottom lines' for *all* fresh waters in New Zealand. Some water values are insensitive to water quality, notably use for waste disposal – which actually degrades water quality.

The term water quality is sometimes extended to include biodiversity of waters and biological indicators of river condition or ecological 'health'. However, this chapter uses the term water quality in the more restricted sense related to water composition. Thus, while 'pelagic' microbiota, such as faecal indicator bacteria, are considered part of water quality in the strict sense, biodiversity and biological indicators are beyond the scope of this chapter. The reader is referred to the chapter on freshwater biodiversity in this book as regards river biodiversity.

A note about the relationship of biomonitoring and water quality in assessment of water condition seems called for. Biomonitoring using one or more bio-indicators, such as macroinvertebrate animals (e.g. Davies-Colley et al. 2011), is extremely valuable in assessment of water condition. A particular virtue of biomonitoring is that biological indicators are time-integrating, whereas water quality sampling, except where continuously-monitoring instruments are deployed, provides only a time series of 'snapshots'. Indeed it is sometimes suggested that biomonitoring might replace water quality assessments of water condition because of this time-integrating feature. However, the weakness of biomonitoring in isolation is quickly revealed by a little reflection. First, we usually need to know what is driving poor or declining water condition, on which bioindicators alone may be silent, whereas water composition will usually give at least some hint. Second, and more compelling, we are concerned with condition not merely of the river itself, but also of *downstream* waters such as lakes and estuaries that receive river loads of contaminants. Third, bioindicators reflect, to a large extent, conditions at the reach scale, whereas water quality is spatially integrating – over the whole river catchment. (Indeed, river water quality may be regarded as a powerful indicator of sustainability, or otherwise, of catchment land use.) Therefore, biomonitoring and water quality monitoring should be viewed as complementary and related sources of information on water condition – which are stronger combined than when either is implemented in isolation (Davies-Colley et al. 2011).

River water composition

River water constituents may be categorised into particulate (suspended) materials and dissolved materials. Dissolved materials may be further divided into dissolved gases (e.g. oxygen) and non-volatile solutes (e.g. dissolved salts). Davies-Colley and Wilcock (2004, table 11.1) list constituents of river water with a commentary on the significance of each, and their table is given here (slightly modified) as Table 1. To the categories of constituents present suspended or dissolved in the water column, it is sometimes helpful to add man-made trash, which is unsightly in waters or stranded on shorelines, and may represent a hazard to aquatic life and human recreationalists.

Gaseous solutes

Of the dissolved gases, oxygen is usually of greatest concern because of its oxidising role and crucial support of the respiration of aquatic life (Table 1) (Davies-Colley and Wilcock 2004). The very limited solubility of oxygen in river water (about 10 g m^{-3}),

TABLE 1 Constituents of water important to water quality (modified slightly after Davies-Colley and Wilcock 2004)

Category	Constituent	Importance
Dissolved gases	Oxygen (O_2)	Respiration of aquatic life
	Carbon dioxide (CO_2)	Source of inorganic carbon for plants
Non-volatile solutes	Major inorganic constituents (e.g. Na^+)	Little concern in New Zealand
	Dissolved inorganic phosphorus	Plant nutrient
	Dissolved inorganic nitrogen	Plant nutrient
	Dissolved organic matter (DOM)	Oxygen demand, contains nutrients (C,N, P), absorbs light (shifts water colour, photochemical catalyst)
Particulates	Mineral solids	Major component of sediment load, scatter light strongly, transport sorbed pollutants
	Organic solids	Component of the sediment load, pool of nutrients (C,N, P), oxygen demand, absorb and scatter light
	Microbiological contaminants	Pathogenic microorganisms render water unsafe for consumption or shellfish consumption or water contact.
Trash	Man-made trash, including metal and glass items and (floatable) plastic trash	Trash in or floating on waters can be a hazard to humans and aquatic life. Unsightly aggregations of floatables (affecting amenity value) can occur in dead-zones of rivers, and be stranded in vegetation or along the banks.

depending on temperature) means that this gas may easily become exhausted if oxygen demand exceeds replenishment from the atmosphere by reaeration. Conversely, photosynthesis of aquatic plants releases oxygen, which may become super-saturated in river waters that have relatively poor (slow) atmospheric exchange. Wilcock et al. (1998) discuss the use of diurnal records of dissolved oxygen (DO) to quantify the competing rates of photosynthesis, respiration, and reaeration based on models of oxygen dynamics in rivers.

Dissolved inorganic carbon (Table 1) is important as the main source of carbon for aquatic plant assimilation. Furthermore, dissolved inorganic carbon is important for regulating pH. 'Buffering' of pH by the carbonate system (Stumm and Morgan 1981) explains why most New Zealand river waters ('bicarbonate waters', Close and Davies-Colley 1990) are in the (near neutral, but tending to slightly alkaline) pH range between about 6.5 and 8.5 (Davies-Colley and Wilcock 2004).

Non-volatile solutes

Despite their high concentrations, major inorganic constituents (the four major cations: Na^+, K^+, Ca^{2+}, Mg^{2+}; three major anions: Cl^-, SO_4^{2-}, HCO_3^-, and silica, SiO_2) have rather minor significance to water quality. That is, these major constituents do not usually constrain water uses, at least not in New Zealand, in contrast to semi-arid or arid areas like parts of Australia with salinised soils. However, these major inorganic constituents can sometimes be useful as water tracers, notably to indicate groundwater-surface water exchange in New Zealand. For routine monitoring purposes, the major ions are usually substituted collectively by electrical conductivity – which correlates well with total dissolved solids (Close and Davies-Colley 1990).

Dissolved nitrogen and phosphorus species are perhaps of greatest concern among solutes in rivers, mainly because these elements are readily available forms of commonly growth-limiting nutrients for aquatic plants including algae (Davies-Colley and Wilcock 2004). Furthermore, nitrogen in ammonia and nitrate forms (CCME 2012) is toxic to aquatic animals. Dissolved organic matter (DOM) is important for a range of reasons, including its content of nutrient (N, P) and its oxygen demand. The light-absorbing (usually refractory) fraction of DOM, coloured dissolved organic matter (CDOM; otherwise known as aquatic humus), is significant for attenuating light penetration into waters (e.g., Julian et al. 2008) and for its catalysis of photochemical reactions of importance including sunlight inactivation of contaminant microbes (e.g., Sinton et al. 2002).

River waters potentially contain a very wide range of other non-volatile solutes (not listed in Table 1) that may be of concern in particular situations, including trace metals and specific organic compounds including biocides. Trace metals, including toxic elements such as lead and copper, are usually of more interest in rivers with urban- or mine-affected catchments than in rural (pastoral) catchments.

Particulates

The particulate categories of most concern in rivers are mineral solids, organic solids, and microbial contaminants (Table 1) (Davies-Colley and Wilcock 2004). Collectively, particulate constituents scatter light strongly, and this optical feature is routinely exploited in nephelometry (measurement of side- or back-scattering of light, Davies-Colley and Smith 2001) which provides an index of particulate concentration known as turbidity. Despite its usefulness in general water quality work, turbidity is only a relative, arbitrary index of the concentration of light-scattering particles that is not generally suitable for enumerating water quality guidelines and standards (Davies-Colley and Smith 2001). Visual clarity, which is inversely related to turbidity and is 'exactly' related to fundamental optical properties of water, is more suitable for such standards.

Mineral solids are comparatively dense (e.g. quartz has a density 2.65 times that of water) and scatter light strongly. Most mineral solids do not absorb light strongly, with the notable exception of ferric iron compounds (Bowers and Binding 2006). Clay minerals (layer silicates occurring as plate-shaped particles) are particularly strongly light scattering because of their small size and unusually large specific surface area. The density of mineral solids means they have relatively high fall velocities and tend to settle on the bed of rivers and downstream water bodies causing a range of adverse effects. Organic solids, in contrast to mineral solids, are much less dense and often strongly light-absorbing as well as light-scattering. But organic solids are perhaps most significant in waters as a pool of nutrient elements (C, N, P) that may be mineralised by bacterial action – with the consumption of dissolved oxygen. Microbiological contaminants that are a threat to human users of water may be thought of as a special category of fine organic particles. They fall into four main sub-categories, in order of increasing size and decreasing 'mobility' in waters: viruses, bacteria, protozoan cysts, and worm parasites (Dufour et al. 2012).

Although it is convenient to consider particulate constituents as falling tidily into the three categories of mineral solids, organic solids and contaminant microbiota, reality is not so simple. River particles are frequently aggregations of primary particles ('flocs') bound together by surface forces and organic polymers of bacterial origin (Droppo 2001). Furthermore, mineral solids can adsorb organic matter, including CDOM, organic particles, and microbiota, onto their surfaces.

Trash

Man-made trash in rivers is a potential hazard to humans and wildlife including native birds, and floatable trash is unsightly on the surface of rivers or downstream waters, or stranded on shorelines (Table 1). Trash is not normally considered a river water constituent (and might, by some definitions be considered beyond the scope of water quality), but its importance as a pollutant is recognised overseas (Moore et al. 2011) and it has recently been identified as an increasing issue for rivers and beaches downstream of urban areas in New Zealand (Young and Adams 2010).

RIVER WATER QUALITY AND WATER COMPOSITION

Water quality, as we have seen, relates to water composition, but knowing water composition alone is not helpful unless a means of interpreting constituent concentrations in terms of suitability-for-use is applied. Many people find it confusing that some minor constituents of water are so much more important than major constituents in terms of supporting water values. Furthermore, while some water quality variables relate directly to water composition (e.g. dissolved oxygen, nutrients), others relate only indirectly (e.g. conductivity, visual clarity) and one (temperature) is *unaffected* by water composition. Finally, in rivers, water composition and water quality vary strongly with time, mainly with flow condition, so it is not particularly helpful to refer to a certain river as of 'good' or 'bad' water quality without reference to the time distribution of variables constraining water quality. For example, the Motueka River (Tasman District, northern South Island) could be described as of good quality for most of the time (Young et al. 2005), but under stormflow conditions this otherwise fairly unpolluted river is much more turbid than usual and heavily laden with faecal bacteria from pastoral agriculture in its lower reaches (McKergow and Davies-Colley 2010; Wilkinson et al. 2011).

Recently, the National Environmental Monitoring and Reporting (NEMaR) project, run by NIWA for the Ministry for the Environment, recommended the variables and indicators of water condition to be measured (monthly) by all 16 regional authorities across New Zealand. The NEMaR project, which used expert panels comprising regional council as well as research institute and university scientists, recognised those variables needed for national reporting as 'core' water quality variables (e.g. total nitrogen) *versus* 'supporting' variables that need to be measured (in some cases less frequently than monthly) to help interpret the 'core' variables (e.g. conductivity). 'Special interest' variables need not be measured routinely at all sites, but may be measured at particular sites as regional concern dictates. For example, the toxic metal copper is a special-interest variable that might be measured at urban- or mine-affected river sites, and organic biocides might be measured more widely in special temporary campaigns.

The NEMaR water quality variables recommended for rivers (McBride et al. 2013) are almost identical to those that have been measured routinely (monthly) for more than 24 years in the New Zealand's National Rivers Water Quality Network (NRWQN) (Davies-Colley et al. 2011). Indeed the NRWQN provides a model for regional monitoring as recommended in NEMaR (McBride et al. 2013). Table 2 shows how water composition relates to the NRWQN (and NEMaR) water quality variables.

Water quality variables

Table 2 shows that some constituents of water are directly measured in routine state-of-environment monitoring – notably dissolved oxygen (DO), forms of the major limiting nutrients nitrogen and phosphorus, and *Escherichia coli* (*E. coli*) bacteria as an indicator of faecal pollution of water.

The remaining variables measured in the NRWQN (and recommended for routine measurement by NEMaR) – visual clarity, nephelometric turbidity, CDOM, pH, electrical conductivity, and temperature – do not measure water composition directly. The two optical variables, visual clarity and turbidity, are inverse correlates, both of which relate to light scattering by particles. Visual clarity is more useful than turbidity (Davies-Colley and Smith 2001), and there is some redundancy in measuring both turbidity and visual clarity. However, turbidity provides a useful quality assurance (QA) check on (field) visual clarity measurement – which is not repeatable (McBride et al. 2013). A third optical variable is the light absorption coefficient in the blue to near-UV range (typically at 440 nm – blue light, or 340 nm – near-UV) of dissolved humic matter or coloured dissolved organic matter (CDOM) – which is conveniently isolated by membrane filtration. Electrical conductivity, as we have seen, reflects the total ionic content of water. pH is a measure of the concentration (more strictly the activity) of the hydrated hydrogen ion, conventionally expressed on a negative log base-10 scale.

Temperature, in contrast to all the other variables measured in the NRWQN, is a purely physical variable essentially *unaffected* by water composition. Temperature is, nevertheless, usually considered part of water quality because it so strongly affects chemical and biochemical equilibria and reaction rates in water – affecting, for example, DO solubility in water and rates of DO consumption by respiration.

Some readers may be surprised to find that total suspended solids (TSS), a direct measure of the organic solids plus mineral particulate content of river water, is not measured in the NRWQN (Davies-Colley et al. 2011) and is only included in NEMaR as an (optional) supporting variable. This reflects, particularly, the relatively high cost of TSS analysis compared with cheap optical correlates (visual clarity, turbidity) making the former an expensive option in routine, indefinite monitoring (McBride et al. 2013). If TSS is needed, indirect estimation by correlation with turbidity or visual clarity is far cheaper – once the latter variables are locally 'calibrated' to TSS. (Recently, TSS and other sediment assays were temporarily added to the NRWQN to provide such estimates.) Another reason for not including TSS in routine, indefinite state-of-environment monitoring is that TSS is probably of most concern as regards sedimentation, and sediment loads are dominated by stormflows, which are only occasionally intercepted by routine (usually monthly) monitoring. That is, monthly

TABLE 2 Water quality variables in the NRWQN (New Zealand's National Rivers Water Quality Network) and recommended in NEMaR (National Environmental Monitoring and Reporting project) in relation to water composition (modified after Davies-Colley et al. 2011)

Category of water quality variable	Determinand (units)	Method	Rationale in NRWQN	Relationship to river water composition
General physico-chemical	**Dissolved oxygen (DO)** ($g\ m^{-3}$; % saturation)	Field oxygen probe	Essential for respiration of aquatic life, maintains oxidising conditions including for photooxidation (numerical standard in RMA[1])	**Direct measurement of a water constituent – dissolved oxygen is an important gaseous solute**
	pH (- log base-10 activity of H^+ ion)	Glass electrode standardised with buffers (laboratory or field)	Aquatic life protection; pollution indicator; acidification (numerical standard in RMA)	Determined by composition of water (acid-base pairs), particularly the inorganic carbon composition. (pH and dissolved inorganic carbon relate to the CO_2 content)
	Temperature (°C)	Field thermistor probe or thermometer	Thermal conditions (climate change), dissolved oxygen interpretation; aquatic life protection (numerical standard in RMA)	NOT related to water composition (a purely physical variable, but one which strongly influences composition via effects on equilibria and reaction rates in water)
	Conductivity ($mS\ m^{-1}$ at 25°C)	Conductivity probe (laboratory or field)	Simple surrogate for total dissolved solids or salinity	Determined by total ionic composition of water
Nutrients (nitrogen and phosphorus)	**Dissolved forms (3 variables): nitrate- plus nitrite-N; ammoniacal-N, dissolved reactive phosphorus (DRP)** (all $g\ m^{-3}$)	Various colorimetric methods, usually measured by flow injection analysis (FIA)	Ammoniacal-N and nitrate-N can be toxic to aquatic life and degrade potable supply; dissolved forms of N and P promote nuisance plant growths	**Direct measure of water constituents - dissolved inorganic forms of the major nutrient elements, N and P**
	Total-N and total-P (all $g\ m^{-3}$)	Digestion to desorb or mineralise N and P, then as for dissolved N and P	Nutrient status/eutrophication	**Direct measure of water constituents – total of the major nutrient elements, N and P** (other than occluded in mineral particles)
Optical variables	Turbidity (nephelometric turbidity units, NTU)	Nephelometry (calibrated to standards, e.g. of Formazine)	Index of light scattering by particles (and of total particulate content). Supports on-site visual clarity measurement	Related to total particulate matter (PM) content of water, and used as an index of this
	Light absorption coefficient of a membrane filtrate (g_{340}, g_{440}; both as m^{-1})	Spectrophotometry at 440 and 340 nm (with a near-infrared measurement to correct for residual light scattering)	Relates to water colour, light climate for aquatic plants, and organic character of water	g_{340} and g_{440} provide an index of the coloured dissolved organic matter (CDOM) fraction of DOM in river water
	Visual clarity (m)	Field observation of the extinction distance of a black disc (m)	Protection of the visual range in waters for aquatic animals (e.g. fish) and human recreational users (descriptive standard in RMA)	Related to total light attenuation (by both scattering and absorption) in water, and therefore to total particulate matter (PM) and to light-absorbing organics (CDOM, POM[2])
Microbiological indicator	***Escherichia coli*** (culture-forming units, cfu per 100 mL)	Membrane filter or MPN[3] methods specified precisely in standard methods (APHA 2005)	Indicator of faecal microbial pollution and thus risk of exposure to faecal pathogens	**Direct measure of a microbial constituent of water – *E. coli* indicates possible presence of faecal pathogens**

[1] RMA = Resource Management Act 1991
[2] POM = Particulate organic matter
[3] MPN = Most probable number

measurement of TSS to estimate sediment load, although feasible in theory, is inefficient in practice, because most river samplings are at baseflow conditions during which sediment flux is very low. Sediment loads are usually better estimated from special storm sampling programmes that use event-triggered autosamplers to obtain samples at high flow conditions (e.g., Basher et al. 2011). Davies-Colley and Smith (2001) have argued that visual clarity is a preferred variable for environmental standards and management compared with either TSS or turbidity for reasons of cost and environmental relevance.

Flow measurement in river water quality

Discharge of rivers is an extremely important supporting variable in river water quality. There are at least three major reasons for needing flow measurements in river water quality monitoring. First, flow strongly affects river water composition and thus quality, so flow is often needed to interpret water quality on particular occasions. Second, for time-trend analysis, it is usually important to flow-adjust water quality data so that water quality trends driven by trends in flow can be distinguished from trends driven by other influences such as land-use change. Third, for some important contaminants of river water, notably nitrogen, phosphorus, and sediment, loads on downstream waters need to be calculated; this is done by first calculating flux (g s^{-1}) as concentration (g m^{-3}) multiplied by flow (m^3 s^{-1}) and then integrating over time to give load (g). Note that all river samples in the NRWQN are 'flow-stamped' (Davies-Colley et al. 2011) and this is also recommended in NEMaR (McBride et al. 2013).

Ideally, flow is measured continuously at or near the river monitoring site to provide interpretive data in terms of antecedent flow and for estimating loads of contaminants. Alternatively, flow at the monitoring site at times of sampling can be estimated from correlation between the flow at the monitoring site measured on a few occasions versus flow at a continuous monitoring station located more distantly, but preferably on the same main stem river (McBride et al. 2013).

RIVER WATER QUALITY AND FLOW

Flow of rivers, as we have seen above, is an extremely important 'supporting' variable in river water quality because many variables vary strongly with flow. Some constituents, particularly when sourced mainly from groundwater or wastewater discharges, correlate inversely with flow because of the dilution effect of high water flows following rainstorms. Figure 1 shows conductivity correlating inversely with flow in the Manawatu River at Palmerston North (NRWQN site WA8) due to progressive dilution of salts at elevated flow. Other constituents may be sourced mainly from land use in the catchment (i.e. diffuse pollution sources) and tend to be washed in or otherwise amplified during rainstorms – so giving a positive correlation of concentration and flow. A positive correlation of concentration and flow is one of the characteristics distinguishing diffuse pollution from point-source pollution (Davies-Colley 2009).

Most of the water quality variables in Table 2 correlate positively with flow in most New Zealand rivers – the exception being a few rivers that are strongly affected by large wastewater discharges. For example, the Tarawera River exhibits a negative correlation with flow of certain constituents discharged from large pulp and paper mills near Kawerau. Conductivity is a notable

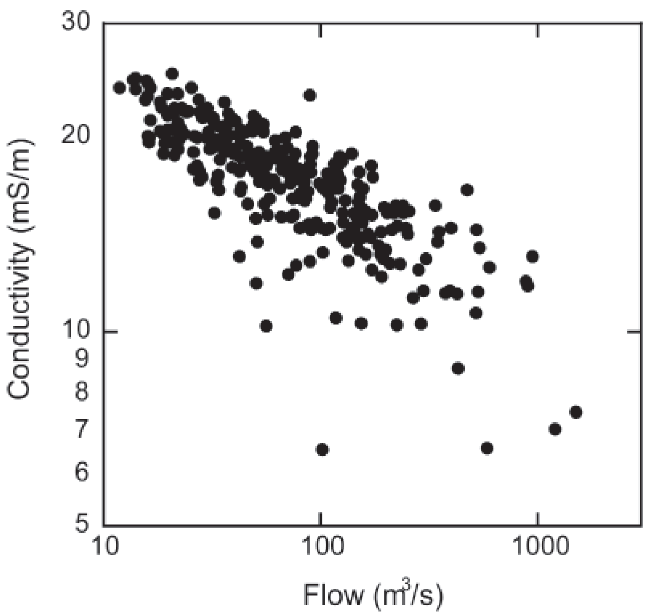

FIGURE 1 Typical inverse relationship of conductivity and flow in an (uncontrolled) New Zealand river. Data are from the Manawatu River at Teachers College, Palmerston North (NRWQN site WA8), for the years 2003–2012.

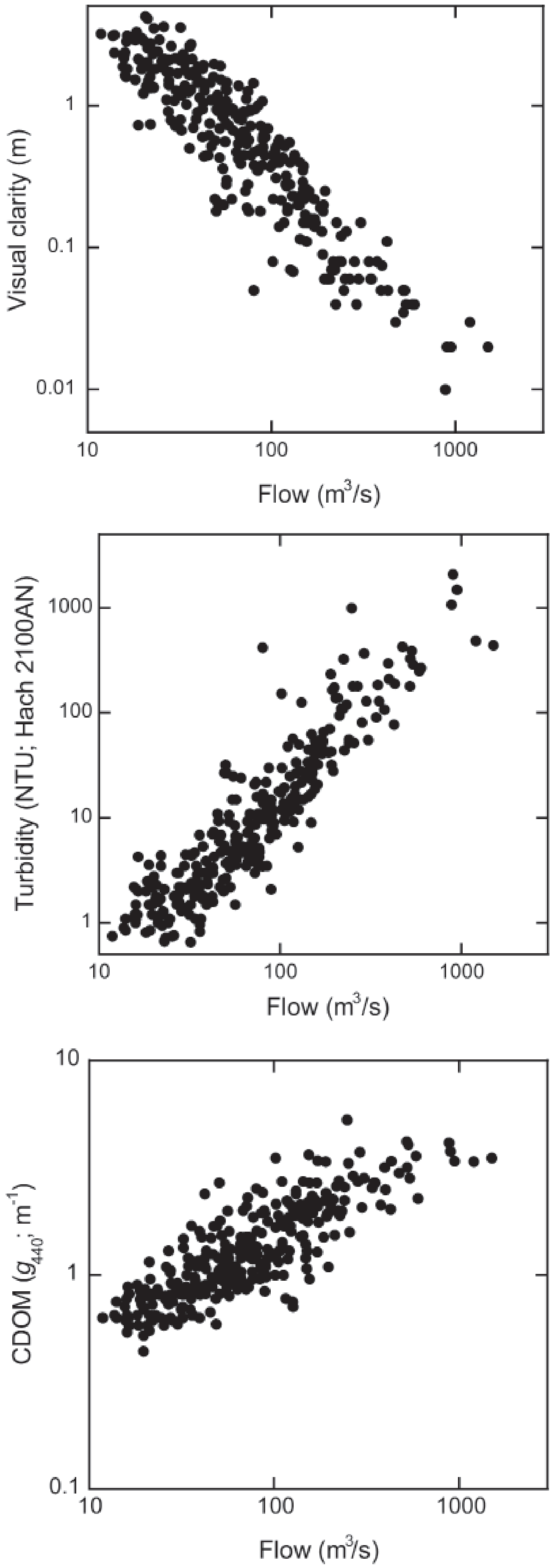

FIGURE 2 Typical power law relationships of visual clarity, turbidity and CDOM *versus* flow in an (uncontrolled) New Zealand river. Data are from the Manawatu River at Teachers College, Palmerston North (NRWQN site WA8), for the years 2003–2012.

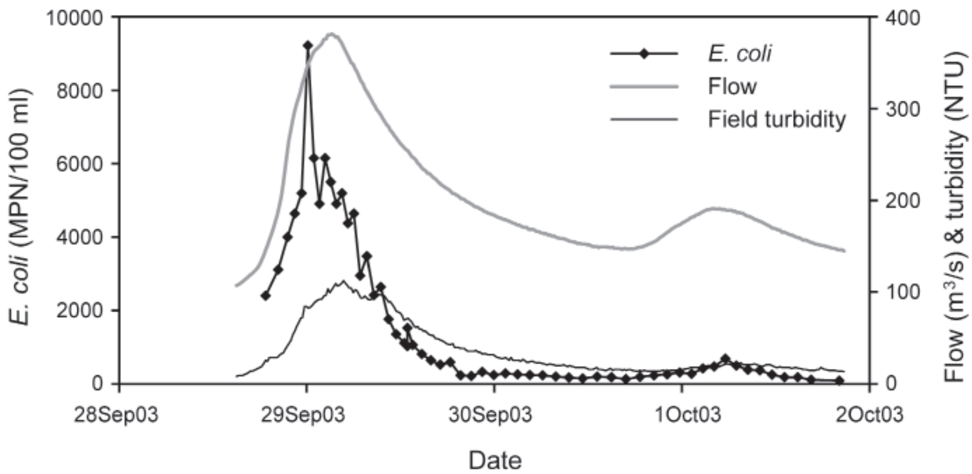

FIGURE 3 *Escherichia coli* and turbidity plotted together with the hydrograph over a flood event in the Motueka River at Woodman's Bend (data from McKergow and Davies-Colley 2010). MPN = most probable number; NTU = nephelometric turbidity units.

exception; this typically correlates inversely with flow because of the dilution of groundwater seepages that are relatively high in dissolved salts with fresh surface runoff or quick-flow following rainstorms (Figure 1).

Particulate constituents and related water quality variables, including particulate forms of nitrogen and phosphorus, turbidity and *E. coli*, all typically correlate positively with flow in rivers. Because of the inverse relationship of visual clarity and turbidity, the former measure declines with flow (Davies-Colley and Smith 2001). The typical positive correlation of TSS (a 'lumping' index of particulate matter) with flow in rivers is well known (e.g., Hicks et al. 2000).

Smith et al. (1997) used data from the NRWQN to examine flow relationships of optical water quality variables in New Zealand rivers. Turbidity was typically a power law function of flow, similar to TSS, but usually with a lower exponent – which was attributed to the increasing coarseness of suspended loads (and decreasing specific turbidity) as flow increases (Davies-Colley and Smith 2001). Consistent with this interpretation, Basher et al. (2011) presented SSC vs turbidity plots for seven sites in the Motueka River that were mostly concave up, showing that turbidity does not increase as rapidly as sediment concentration. Smith et al. (1997) found that typically visual clarity in rivers is, correspondingly, a declining power law function of flow (Figure 2). Recently Elliott et al. (2013) used data from the NRWQN to confirm that light attenuation, calculated from visual clarity observations, is strongly correlated with flow; however, the relationship is sometimes non-linear in log-log space, in contrast to Figure 2 (i.e. a power law is not a universal model). Elliott et al. (2013) found that light attenuation followed similar patterns to suspended sediment, and had similar relationships to catchment attributes, but there were differences attributable to variation in the optical properties of suspended particles between different rivers.

Smith et al. (1997) found that CDOM, as indexed by g_{340}, was also a power law function of flow in most rivers (as illustrated in Figure 2), but the power law exponent, while still positive, was much lower (i.e. CDOM increases much less sharply with flow than turbidity or light attenuation). For example, the median power law exponent for CDOM versus flow in the NRWQN was +0.50, compared with −1.08 for visual clarity (inverse relationship) and +1.34 for turbidity versus flow. It is not immediately obvious why CDOM, a solute, should generally increase with increasing flow in rivers when other solutes, notably salts as indexed by conductivity, generally dilute at elevated flow. The reason may be that CDOM is much higher in zones of organic concentrations within catchments, such as wetlands and in the soil solution, but is comparatively low in groundwater. Thus, while groundwater solutes such as salts are diluted during high flows, quickflow through soils and surface runoff from wetlands yields stormflows with elevated CDOM.

The positive correlation of particulate constituents and flow was traditionally attributed to wash-in of particulate contaminants by overland flow. However, although wash-in may be the main ultimate source of these contaminants, entrainment from the stream channel or banks is often the dominant (proximal) source during flood events. Evidence for this includes the time relationship of contaminants and flow over flood hydrographs. For example, Figure 3 shows the typical relationship of *E. coli* with flow in rivers – with the indicator bacteria concentration peaking well ahead of the flow peak. McBride (2011) applied kinematic wave theory in an analytical model to explain why *E. coli* (derived from bed sediment entrainment) arrives ahead of flood peaks, in contrast to a pathogen (*Campylobacter*) that does not appear to survive well in sediments and arrives later on the hydrograph, presumably with wash-in of faecal matter. Clockwise loop rating curves for sediment are also common in streams and small rivers with bed or bank erosion, but in larger rivers or where sediment comes from remote areas of the catchment the sediment peak may lag behind the flow peak implying an anti-clockwise loop rating (e.g., Hughes et al. 2012).

VALUES AND USES OF RIVER WATERS – DEPENDENCE ON WATER QUALITY

RiVAS, the River Values Assessment System, was developed (Hughey and Booth 2012) to identify primary attributes (a subset of a full list of attributes of rivers) that describe the value of a particular river for a particular water use. The RiVAS method involves an expert panel using a quantitative, standardised approach to rate attributes (6–10 'indicators') affecting particular river values for particular rivers. These primary attributes are indicators for monitoring. So far, the RiVAS method has been applied to 11 different river water values (Hughey and Booth 2012). They present a matrix of water values (uses including passive uses) *versus* river attributes – some of which are identified as indicators that constrain values.

Table 3 lists the 11 values of water recognised by Hughey and Booth (2012) and shows that most, but not all, of these values are sensitive to water quality. Indeed, it is clear that, overall, water quality variables are *the* most generally important attributes constraining water uses. Water quality variables are therefore, perhaps not surprisingly, the most important indicators for evaluating suitability-for-use of rivers.

However, the RiVAS work shows that a number of other attributes of rivers besides water quality attributes may also affect values, including water *quantity* attributes such as flow or level, and built structures and access. For example, although swimming is very strongly sensitive to water quality (notably microbiological quality and visual water clarity; also presence of trash), swimming also depends strongly on state of flow in rivers and direct access or built structures like jetties to provide convenient access.

Hughey and Booth (2012) proposed that RiVAS provides a basis for selecting indicators of water values for monitoring purposes, including, but not limited to, water quality variables. They point out that simple count data are useful for almost all water values and uses, but such data are seldom collected. These would include, for example, the count of swimmers using a particular access point ('beach') on a river over a year, and the average count of native birds using a particular river reach.

Recently, the Freshwater Reforms report (MfE 2013) outlined values (N = 14) for rivers to be used in a National Objectives Framework. The values listed in MfE (2013) are similar to those analysed via RiVAS, but a few other values were included: stock watering, secondary contact recreation (essentially all active recreation besides swimming), and aquaculture. For each value a list of attributes to be managed is given by MfE (2013) – most, but not all, of which are water quality attributes. Water quality in the sense of suitability-for-use can be defined with regard to different uses or values of rivers. The National Objectives Framework envisages defining a set of bands (A, B, C, D) for each attribute and each value/water use. An 'A band' rating would indicate excellent water quality and 'C' a pass, while 'D' would indicate unacceptable conditions (a 'fail'). For some combinations of value and attribute, scientific information is available to underpin the framework, but there are probably a significant number of combinations where new scientific work will be required so that the framework can be populated over time (MfE 2013).

Waste disposal is an important value not recognised by either MfE (2013) or Hughey and Booth (2012) and one that is noteworthy for being completely insensitive to water quality but which itself degrades water quality. However, certain other uses are only slightly sensitive to water composition, including hydropower and irrigation (Table 3). At the other extreme, aquatic habitat and human recreation are extremely sensitive to water quality (Table 3). MfE (2013) envisages the National Objectives Framework requiring that aquatic ecosystem health and human health for secondary contact are protected in *all* water bodies – so as to provide national 'bottom lines' in terms of aquatic habitat and recreational condition of fresh waters.

POINT-SOURCE AND DIFFUSE POLLUTION OF RIVERS

Point versus diffuse pollution

The term 'pollution' applied to a river usually evokes the image of a pipe discharging wastewater from a factory or municipal sewage works. Of course these so-called 'point' sources of pollution do strongly affect water quality. However, in New Zealand, point sources, with a few notable exceptions, have been largely cleaned up over the past three decades or more, leaving diffuse sources predominant (Davies-Colley 2009; Howard-Williams et al. 2011). Diffuse pollution is defined slightly more widely than non-point pollution by the International Water Association (IWA) Specialist Group on Diffuse Pollution, with multiple distributed small point sources such as tile drains in agricultural land or sewer leaks in urban land regarded as diffuse sources. Accordingly, Campbell et al. (2004) define diffuse pollution as

Pollution arising from land-use activities (urban and rural) that are dispersed across a catchment, or subcatchment, and do not arise as a process effluent, municipal sewage effluent, or farm effluent discharge.

TABLE 3 Water quality (WQ) variables in relation to values of water (as discussed by Hughey and Booth 2013)

Category of value	Value/use	Water quality dependence (variables of concern)
Recreation	Salmonid angling	Temp, DO, visual clarity, N and P
	Swimming	Microbiology, visual clarity, nutrients
	Whitewater kayaking	Microbiology, visual clarity
	Whitebaiting	Visual clarity (mainly)
Character	Natural character	All major areas of WQ
Ecological	Native birds	All major areas of WQ (except microbiology)
	Native fish	All major areas of WQ (except microbiology)
Cultural	Tangata whenua	All major areas of WQ?
Development	Irrigation	Insensitive to WQ
	Hydropower	Insensitive to WQ
	Potable water	Sensitive to several areas of WQ, notably microbiology
Other values	Waste disposal	Unaffected by WQ, but *affects* WQ
	Livestock watering	Microbiology (mainly)

Water quality and diffuse pollution from land use

Water quality is strongly related to land use, as implied by the above definition of diffuse pollution. In fact, (good) water quality may be viewed as an ecosystem service provided, mainly, by (minimally disturbed) land. In-stream attenuation also contributes to good water quality, notably by so-called nutrient 'spiralling' (transformation and uptake of inorganic nitrogen and phosphorus) and by clarifying water (reducing particulate contaminants) by hyporheic exchange (exchange of water and constituents between bed sediment interstices and overlying water) with underlying sediment plus 'filtration' by plant biomass in rivers. Undisturbed land with indigenous vegetation cover generally yields very good water quality; disturbed land yields degraded water quality because of mobilisation of, particularly, fine sediment, the nutrients nitrogen and phosphorus, and faecal microbes. These 'big three' categories of diffuse pollutants are considered in turn below.

Certain types of land use mobilise other categories of diffuse pollutant (Campbell et al. 2004). For example, mining and urban land uses (occupying comparatively small proportions of New Zealand's land area) can mobilise heavy metals that are very toxic to aquatic life. Pesticides and herbicides can be mobilised from both urban and rural land areas (Parkyn and Wilcock 2004). The following discussion is confined to the 'big three' pollutants characteristic of the dominant land use in New Zealand, pastoral agriculture.

Effects of major categories of diffuse pollutant

Fine sediment — This is sometimes referred to as the universal water pollutant, both because it is mobilised by all soil- and vegetation-disturbing land uses and because of its wide range of adverse effects on aquatic ecosystems. Sediment also acts as a major carrier of other pollutants sorbed on sediment particles, including phosphorus and toxic metals. Effects of fine sediment when deposited are often severe and include shoaling or infilling of lakes, reservoirs and estuaries, and smothering of benthos, notably in estuaries (Thrush et al. 2003), by major deposition events. In rivers and streams, clogging of bed sediments, with consequent inhibition of hyporheic exchange, is probably the most severe effect of fine sediments. In New Zealand a major effort to improve assessment of fines sedimentation in rivers has recently been completed (Clapcott et al. 2011).

Fine sediment can also have a wide range of impacts when still suspended, including, at fairly high concentrations, damage to gills and respiratory structures of aquatic animals. However, the optical effects of fine sediment are probably of most concern while suspended within river waters (Davies-Colley and Smith 2001). Fine sediment scatters light intensely and may also absorb light, and both optical processes lead to light beam attenuation and reduced visual clarity (Davies-Colley 1988; Zanevald and Pegau 2003). Reduction in visual clarity is among the most severe impacts of fine sediment because it alters the 'visual habitat' (and thus behaviour) of fish and birds, and reduces the amenity value for recreationalists (Davies-Colley et al. 2003). Furthermore, fine sediment reduces light penetration to aquatic plants in waters, thus reducing their productivity. Fine sediment is typically more strongly light scattering than light absorbing, but Kirk (1985) has explained that light scattering also contributes to reduced light penetration – because multiple scattering forces light to take a tortuous path down through the water column so increasing the probability of its being absorbed. Reduced light penetration may have a severe impact on standing waters downstream of rivers (Davies-Colley and Smith 2001) such as lakes and estuaries into which rivers discharge fine sediment.

Nitrogen and phosphorus — These nutrients are usually the growth-limiting factors for aquatic plants as well as terrestrial plants. Unfortunately, while fertile land is usually considered a good thing, fertile water causes a range of 'nuisances' associated with excessive aquatic plant growth, notably the growth of benthic algae (periphyton) in unshaded rivers with a range of mostly adverse ecological shifts (Biggs 2000). Rivers also act as a conduit for nutrient loads to water bodies downstream that respond adversely to nutrient loading. In lakes, nutrient enrichment promotes the growth of phytoplankton, rendering lake waters turbid and reducing light penetration to (desirable, sediment-stabilising) benthic plants. Eutrophication also makes lakes prone, in fine, calm weather, to blooms of cyanophytes (photosynthetic blue-green bacteria) that form unsightly surface scums and other nuisances, including being toxic to humans and animals (Conley et al. 2009). Eutrophication of estuaries is typically combined with increased sediment loading and promotes phytoplankton and nuisance macroalgae such as sea lettuce. Phytoplankton and fine sediment combine to reduce light penetration through the estuarine water column, which may eliminate desirable benthic plants such as seagrasses with major adverse shifts in estuarine ecology.

As well as being, typically, one of the two main growth-limiting nutrient elements in waters, nitrogen is toxic to aquatic life in its ammonia and nitrate forms. The toxicity of ammoniacal-N in the (unionised) ammonia form is well known, and guidelines have been promulgated for many years (e.g, ANZECC 2000). The toxicity of nitrate has only fairly recently been recognised (e.g., CCME 2012), but has major implications for ecological damages in some of our nitrate-enriched pastoral streams and rivers. Ecotoxicological experiments on a range of New Zealand freshwater organisms have recently been interpreted to propose a guideline value of 1.5 g m^{-3} nitrate-N for a 'high' level of protection of aquatic life in rivers (Hickey 2013).

Faecal microbes — These may, for some purposes, be considered a special kind of fine, low density organic sediment or bio-colloid. Faecal microbes of human health concern come from warm-blooded animals including humans, livestock, and feral birds and animals (Dufour et al. 2012). Intuitively, human faecal matter is perceived as most hazardous (and is most aesthetically repellent – probably for related reasons!), but various zoonotic diseases of humans may be present in the faeces of a wide range of warm-blooded animals and birds, notably (in New Zealand) the bacteria *Campylobacter* and the protozoans *Giardia* and *Cryptosporidiium*. Viruses are generally very host-specific, so animal viruses are not (usually) a threat to humans, and animal viruses are not (usually) considered a pollutant of river waters (Dufour et al. 2012).

A wide range of disease-causing infectious agents (pathogens) from faecal contamination may cause (sporadic) episodes of water pollution, so routine, indefinite monitoring of the many pathogens that could be present in rivers is prohibitively expensive and is almost never done. Instead *indicators* of faecal pollution are used as sentinels in river monitoring to indicate the presence of (recent) faecal contamination and, therefore, the potential presence of human pathogens. In fresh waters, including rivers, the favoured faecal indicator is the bacterium *E. coli*, which is almost always present in stools of warm-blooded animals. Some studies have shown a useful degree of correlation of certain faecal pathogens with *E. coli*, including *Campylobacter,* which is prevalent in New Zealand (Till et al. 2008), so the monitoring of *E. coli* is likely to remain the main tool for routine water monitoring, including rivers (McBride et al. 2013).

RIVER WATER QUALITY – LAND USE PATTERNS

Effects of pastoral land use

Pastoral agriculture is the single largest category of land use in New Zealand and occupies about 40% of the nation's total land area (MfE 2007). Indigenous forest cover has been reduced from an estimated 85% to 23% of New Zealand's land area (MfE 2007). Profound changes in environmental conditions of streams accompany forest clearance, with impacts being exacerbated by subsequent drainage of wetlands, channelisation of streams, fertilisation, tillage, and grazing of livestock (Parkyn and Wilcock 2004; PCE 2004; Quinn et al. 2009).

A large number of studies in New Zealand have examined effects of pastoral land use on stream water quality. Parkyn and Wilcock (2004) (see also PCE 2004) reviewed studies up to that time, building on an earlier review of pastoral agriculture effects on water quality by Smith et al. (1993). More recent reviews include that of Quinn et al. (2009). These reviews have reported the consistent finding that water quality in pastoral streams and rivers is degraded compared with that in comparable streams in native vegetation cover. Streams in grazed pasture have increased runoff due to lower interception and evapotranspiration of grasses compared with forest (Fahey and Rowe 1992) plus reduced infiltration owing to soil compaction by livestock trampling (Nguyen et al. 1998). The increased runoff (and lowered infiltration) is a major driver of reduced stream water quality.

Furthermore, livestock are themselves sources of diffuse contaminants, both directly with mobilisation of nitrogen, phosphorus, and faecal microbes in their urine and dung, and indirectly with fine sediment eroded from livestock-damaged soils (Collins et al. 2007). Cattle are attracted to water, and cause damage to riparian areas and stream banks and channels to which they have access (Trimble and Mendel 1995). Moreover, some fraction of cattle dung, with its associated microbial pollutants and phosphorus, is deposited directly into streams and drains by cattle (Collins et al. 2007). Estimates of cattle dung deposited directly in channels vary from about 0.5% to 8% of total, and may depend on the type of cattle (e.g. beef versus dairy) and factors such as land slope and climate. The high-end of that range was reported for beef cattle in semi-arid rangeland of the western USA (Belsky et al. 1999) and rather lower direct deposition rates by cattle (albeit still representing very significant pollution) appear to apply in New Zealand (Collins et al. 2007).

Findings from numerous New Zealand studies of widespread diffuse pollution of pastoral streams and rivers by fine sediment, nutrients, and faecal microbes have been confirmed in a recent overview of diffuse pollution in New Zealand by Howard-Williams et al. (2011). These findings are also broadly consistent with studies in the international literature on livestock impacts as reviewed by Campbell et al. (2004).

Sediment — The mobilisation of fine sediment in streams draining livestock pasture (reducing visual clarity) (Quinn et al. 1997) can be attributed to increased runoff, hillslope erosion and bank disturbance along channels accessed by livestock (Parkyn and Wilcock 2004). Cattle, with their high hoof pressures, are very damaging of soils, particularly in wetlands and riparian soils, and on steep slopes, on which the shearing action of their hooves is readily apparent. Furthermore, their attraction to water results in severe damage to riparian areas and stream banks (Trimble and Mendel 1995). Sediment yields are typically higher in sheep-beef and deer farming than from dairy farms, apparently because of the gentler slopes typical of dairying (McDowell and Wilcock 2008). All these 'disturbances' by livestock tend to increase erosion and the mobilisation of fine sediment into streams and rivers draining livestock pasture. Sedimentation studies (coring from sedimentary basins) suggest about a 10-fold increase in sediment yield under pasture compared with forest, although sediment load studies in rivers usually yield more modest amplification factors in the range 2- to 5-fold (Hicks et al. 2004). For example, for hill country near Whatawhata, Quinn and Stroud (2002) reported a 3-fold greater sediment yield in a pasture catchment than in a nearby paired native forest catchment (that was somewhat disturbed by feral mammals). An even lower amplification factor was reported for the same two catchments by Hughes et al. (2012) after integrated catchment management (ICM) action in 2000–2001 that was designed to improve water quality and stream ecological health in the pasture catchment (Quinn et al. 2007).

Nitrogen and phosphorus — These nutrients are typically higher in pasture than in forested streams (Quinn et al. 1997, 2009; McDowell et al. 2009). The mobilisation of nitrogen and phosphorus from livestock pasture may be attributed to soil erosion (carrying particulate N and P) and to deposition of livestock urine and dung – including directly into the channel by cattle (Parkyn and Wilcock 2004). A monitoring study of five intensely farmed dairy catchments (Wilcock et al. 2007) revealed poor water quality and particularly high nitrogen yields, but with considerable potential for management intervention to improve water quality. Fertilisation of land provides a top-up of catchment stores of nutrients that would otherwise be progressively depleted by export from the catchment by runoff in streams and rivers as well as livestock products. Nitrogen in urine patches deposited on soil is rapidly transformed into ammoniacal-N and thence oxidised to nitrate, which is highly mobile and easily leached beyond the root zone into groundwater. This groundwater eventually re-emerges in surface water, but in some cases only after major lag times, sometimes decades. For example, nitrogen entering Lake Taupo in spring-fed streams is a legacy of farming in the lake catchment ranging from 20 to 75 years earlier (Morgenstern et al. 2011).

Pathways for phosphorus differ appreciably from those for nitrogen. Phosphorus is more concentrated in livestock dung, which can reach stream channels by land runoff or by direct deposition by cattle. Phosphorus is more strongly associated with sediment (and yields broadly correlate with those of sediment) because of erosion of P-enriched soils. McDowell et al. (2009) have emphasised the importance of nitrogen to phosphorus ratios as regards limitation of periphyton in rivers, by reference to the ratios of these elements in typical algal biomass. Thus nuisance periphyton (benthic algae) may be limited by nitrogen or phosphorus or co-limited.

Faecal pollution — Faecal microbial pollution from livestock pasture, as indicated by *E. coli*, (Donnison and Ross 1999; Collins et al. 2007) may be attributed (ultimately) to livestock wastes washing in from contributing areas, particularly areas of soil compacted by livestock trampling, and from cattle directly entering channels. As a result, typical (median) *E. coli* concentrations in pastoral streams average about 20-fold higher than in forested land (Smith et al. 1993). The zoonotic pathogens *Campylobacter*, *Cryptosporidiium* and *Giardia* are also present at much higher concentrations in pastoral streams (Donnison and Ross 1999), although not necessarily with a high correlation versus *E. coli* owing to contrasting environmental behaviour (e.g. Stott et al. 2011) as well as sporadic presence of pathogens. (*E. coli* is shed by all animals, but pathogens only by infected individuals).

A clear finding from recent studies is that faecal pollution is much higher in rivers during stormflows than at baseflows (Davies-Colley et al. 2008; McKergow and Davies-Colley 2010). This stormflow pollution used to be attributed to wash-in of faecal matter, but the timing of the faecal pollution (peaking well ahead of flood peaks) and artificial flood experiments (producing similar peaks of faecal pollution in absence of rainfall and wash-in) suggest that most of the faecal microbes are stored in stream bed sediments from which they may be mobilised by flood wave fronts (Stott et al. 2011). The new understanding of river faecal dynamics has been the basis of modelling efforts in New Zealand (McBride 2011; Wilkinson et al. 2011).

Intensification of pastoral land use

Intensification of pastoral agriculture in New Zealand has been an important feature of recent years (MacLeod and Moller 2006). Intuitively, it would be expected that the *intensity* of land use for pasture would affect river water quality. Again New Zealand studies generally confirm that expectation (Quinn et al. 2009). Most studies of water quality have found monotonic degradation of water quality with increasing pastoral intensity. For example, Hamill and McBride (2003) compared trends (1995–2001) in water quality variables in the Southland Region, and found that increasing dairying was associated with decreased water quality, notably worsening dissolved oxygen conditions and increased dissolved reactive phosphorus (DRP). McDowell et al. (2009) modelled DIN and DIP yields from a typical sheep-beef farm and a typical Southland dairy farm in 1958, 1988, 1998, and 2008 – a 50-year period of increasing intensification. There were major increases in DIN over that timespan, but *decreases* in DIP (attributable to improved effluent management), from the dairy farm (increasing the N:P ratio and driving towards P-limitation in receiving waters), with modest increases of both DIN and DIP from the sheep-beef farm (and minor increase in N:P ratio).

McDowell and Wilcock (2008) have reported modest P yields and relatively low sediment yields from dairying, despite much greater N-yields than other farming systems. Therefore conversion to dairying may be expected to increase N yields but not necessarily other diffuse pollutants. However, Quinn et al. (2009) have recognised that improving farm practices (implementation of BMPs) may at least 'hold the line' on water quality despite intensification.

Specific yields of sediment, nutrients, and microbes increase near-linearly with increasing stocking density (Parkyn and Wilcock 2004). For example, Vant (2001) reported that total nitrogen (TN) correlated strongly with stocking density (cows per hectare) averaged over catchments in the Waikato Region. As the trend to pastoral land intensification continues, particularly the expansion of dairying, we may expect water quality to continue to decline, unless beneficial management practices (BMPs) are sufficiently efficacious and widely adopted to compensate. Wilcock et al. (2007) reported that BMPs implemented in intensively managed dairy land are yielding distinct benefits in stream water quality, albeit from a degraded condition.

Effects of other land uses in New Zealand

We have examined the effect of pastoral land use on water quality. The question arises: what is the effect of other land uses on water quality? Cropping and horticulture together occupy only about 1.5% of New Zealand's land area, but are strongly associated with increased nutrients and fine sediment (but not microbial pollution) – consistent with findings in other countries (Campbell et al. 2004).

Urban land use is also small in areal terms, with about 1% of New Zealand under impervious surface including urban and transport. However, urban effects on water quality are often severe. Indeed, the streams and rivers in 'worst' ecological condition in New Zealand are probably those draining urban catchments (Suren and Elliott 2004). This is attributable mainly to accentuated flood peaks due to rapid runoff from extensive impervious area (and a corresponding dearth of baseflow). However, urban water quality is typically poor with elevated fine sediment concentrations (and correspondingly low visual clarity), elevated nutrients, and faecal microbial pollution. Toxic metals and floatable trash add to the water quality burdens on urban streams (Williamson 1991).

Plantation forestry is a fairly important land use in New Zealand in areal terms (about 7%). Intuitively, plantation forestry might be expected to be a fairly 'benign' land use as regards water quality because of the dampening effects of the tree canopy on hydrology and because of mostly low levels of fertiliser application and soil disturbance (apart from roading) and low densities of livestock or feral animals. New Zealand studies over a range of climate and geological regions bear out this expectation in the main (Fahey et al. 2004). For example, Quinn et al. (1997) reported strong water quality patterns with land use in streams draining hill country in the western Waikato Basin. Water quality was fairly good in streams draining first-rotation pine plantations converted from pasture except for high turbidity and low visual clarity. More recently, Parkyn et al. (2006) reported trends across streams in contrasting land-use in soft-rock catchments of the Gisborne District, where pasture streams had severely degraded water quality (and low stream ecological 'health' as indicated by macroinvertebrate communities). Streams draining mature pine plantations converted from pasture had fairly good water quality, as regards nutrients and faecal microbes. However, high sediment yields and turbidity (low visual clarity) persisted under pines where deep-seated gullies had been initiated under pasture prior to land use conversion. Parkyn et al. (2006) speculated that if pine trees are harvested up to the stream edge, destabilisation of stream banks with consequent impacts on water quality and stream health can be expected.

A confounding phenomenon with conversion of pasture to plantation forestry is that shading under the developing tree canopy is expected to initiate a phase of streambank erosion as channels widen following elimination of (bank-armouring) pasture grasses (Davies-Colley 1997). This channel-widening phenomenon likely caused the high turbidity and low visual clarity in plantation forest streams studied by Quinn et al. (1997).

Although plantation forestry greatly improves water quality compared with pre-existing pasture, there remain concerns around the harvesting phase. Harvesting of the tree crop results in removal of the tree canopy and generally increased disturbance of the catchment soils with consequent sediment generation and nutrient mobilisation (Fahey et al. 2004).

National-scale patterns of water quality with land use

The patterns of water quality with land use discussed above in regard to streams with near-homogeneous catchments are also seen nationally, and in large rivers – despite differences in geology, soils, topography, and climate. Davies-Colley (2009) reported a correlation matrix (reproduced here as Table 4) showing that median values of key water quality variables at NRWQN sites correlated strongly with the percentage of catchments in pasture and more weakly with percentage of catchments in indigenous forest and plantation forestry. Increasing pastoral development degrades water quality and the best water quality is seen in indigenous forest. Even cropping exhibited a clear water-quality-degrading signal (Table 4).

Figure 4 shows total nitrogen and *E. coli* medians at the 77 NRWQN sites plotted against percent pasture. Both variables have strong correlations, implying a causative relationship (which, of course, is confirmed from other studies including those – reviewed above – in homogeneous small catchments).

TABLE 4 Correlation matrix of water quality relationships to land use (from Davies-Colley 2009). Correlation coefficients are non-parametric Spearman rank coefficients – which indicate strength of monotonic, but not necessarily linear, relationships, and NRWQN data are for the years 2005–2008. Land use classifications were taken from MfE (2007) based on the LCDB2. All the correlations are significant (95% confidence interval for 75 d.f., rs > 0.225) with the exception of that for *E. coli* versus cropping and horticulture

Variable	% Pastoral	% Cropping and horticulture	% Native forest	
Total nitrogen	0.85	0.45	−0.39	
Total phosphorus	0.70	0.24	−0.32	
Visual clarity	−0.45	−0.24	0.30	
Escherichia coli	0.80	(0.17)	−0.34	
				Total
NZ land area (km^2)	107672	4174	65672	271900*
% of land area*	39.6	1.5	24.1	100%

(*) Remaining areas (amounting to 34.8% of total), include 9.7% Tussock, 9.3% Scrub, 7.7% Water (incl. snow and ice), 7.2% Plantation forest, and about 1% impervious surface.

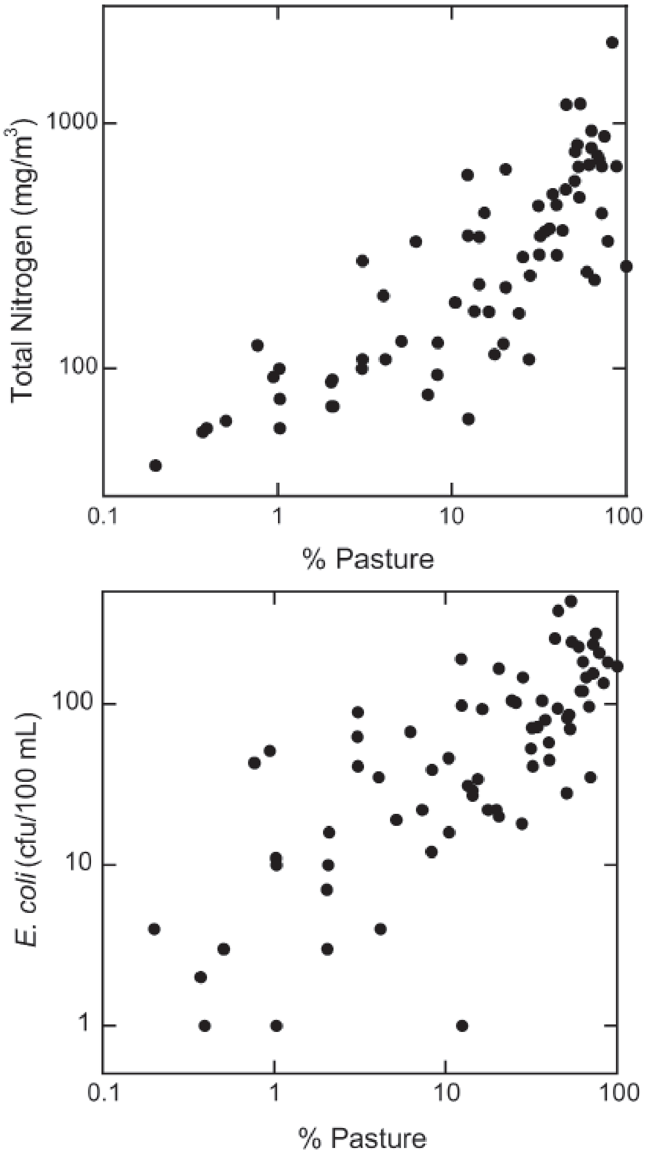

FIGURE 4 Total nitrogen and *E. coli* medians at 77 NRWQN sites plotted versus percent pasture in the river catchments (data as collated by Davies-Colley (2009) for the years 2005–2008)).

The strong relationships of total nitrogen and *E. coli* with percent pasture are expected, given that nitrogen mobilisation and faecal pollution are both strongly linked to the presence of livestock. The positive correlation of total phosphorus, and inverse correlation of visual clarity, with percent pastoral (Table 4) are also as expected, although the absolute values of the correlations for these variables are weaker. This is probably because they both relate to fine sediment and erosion, which is affected by factors other than livestock grazing, such as geological and soil factors (Davies-Colley 2009).

TIME TRENDS IN RIVER WATER QUALITY

State-of-environment monitoring and reporting is concerned not merely with current state of the environment, but also with its trend over time. As regards river water quality, there is intense interest in whether a particular river, and rivers regionally and nationally, are getting worse or improving over time – most often in relation to land management changes, but also, potentially, in response to climate change. The public perception is that water quality in New Zealand is getting worse and that this decline is driven mainly by pastoral intensification (Hughey et al. 2011), a perception that is broadly borne out by formal trend analyses of river monitoring data at the national scale, as is discussed below.

The first major trend analysis for water quality in New Zealand was that of Smith et al. (1996) using the first 5 years of data (1989–1993 inclusive) for all 77 sites in the NRWQN. This analysis presented several innovations in trend detection, particularly the use of a (non-parametric) de-seasonalised trend statistic, setting the standard for subsequent trend analyses of river water quality in New Zealand. A software package (Time Trend) has since been developed, and made available as freeware (www.niwa.co.nz/our-science/freshwater/tools/time-trend). Smith et al. (1996) also reported on the patterns of correlation of NRWQN variables with flow that permit 'flow adjustment' of water quality data for time-trend analysis.

Smith et al. (1996) found generally improving river water quality in the first five years of the NRWQN (1989–2003 inclusive), particularly for the South Island. Nitrate-N, total phosphorus, and biochemical oxygen demand trended downwards and there were upward trends for conductivity, DO, visual clarity, and CDOM. Downwards trends in water temperature were linked to (temporary) global cooling caused by the Mt Pinatubo volcanic eruption (1991).

With the expectation that climate change will affect water quality, Scarsbrook et al. (2003) studied associations between river water quality and the El Niňo Southern Oscillation (ENSO) climate cycle. ENSO has long been known to affect flow (mainly via rainfall) and air temperature. Scarsbrook et al. (2003) found that 13 water quality variables measured routinely in the NRWQN all correlated significantly, although weakly, with the Southern Oscillation Index over 1989–2001. The strongest relationships with ENSO were for water temperature and nitrate-N. The authors suggested that such influences of climate cycles could potentially mask water quality trends caused by land-use change.

A comprehensive analysis of trends in river water quality at NRWQN sites was reported by Scarsbrook (2006). Strong decreasing trends in biochemical oxygen demand and ammoniacal-N were attributed to reducing loads from point pollution sources. Scarsbrook (2006) found an (improving) trend toward increasing visual clarity, which may reflect improving soil conservation practices as well as reduced point-source loading of light-attenuating materials. However, other trends indicated deterioration in water quality, including increasing nitrate-N, total nitrogen, DRP, and total phosphorus. These nutrient enrichment trends were attributed to increasing diffuse pollution from pastoral agriculture that was overwhelming gains from improved wastewater treatment and consequently reduced point-source pollution.

Ballantine and Davies-Colley (2009) reported trend analyses on the NRWQN data for 1989–2007. Their analysis broadly confirmed the findings of Scarsbrook (2006) that gains from clean-up of point-source pollution have been increasingly negated by the impacts on water quality of diffuse sources, mainly intensification of pastoral agriculture driving nutrient enrichment. However, in some degraded pastoral rivers there was a suggestion of a slowing in the decline in water quality, with reduction in certain key pollutants, notably nitrogen. Davies-Colley (2009) speculated that this might be the signature of improved farm practices such as riparian fencing and nutrient budgeting.

More recent analysis of national trends in the NRWQN (Ballantine and Davies-Colley 2010), and also in regional council data (Ballantine et al. 2010), confirmed generally declining water quality in New Zealand rivers attributable (mainly) to intensification of pastoral agriculture. However, several relatively polluted

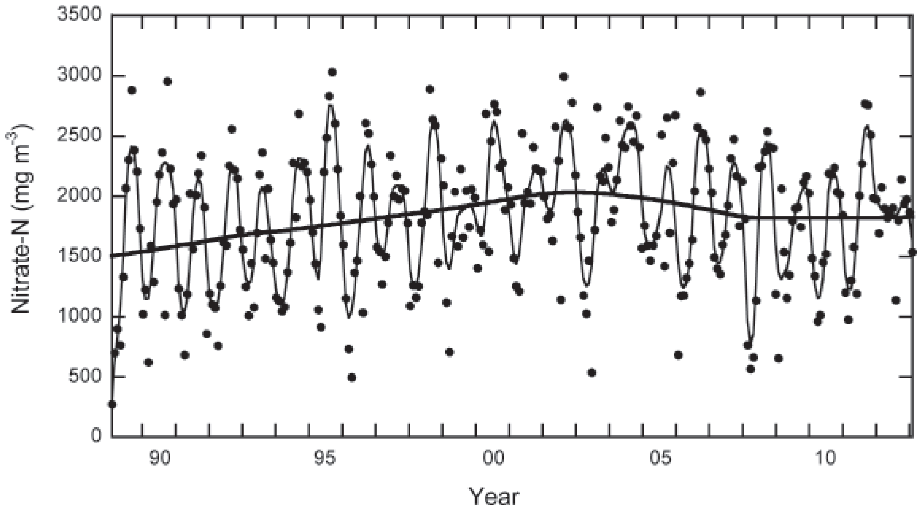

FIGURE 5 Trends in nitrate-N in the Waingongoro River at SH45, Taranaki Region (NRWQN site WA3). A running median smoother picks out the seasonal pattern of nitrate with recurrent winter-spring peaks. A LOWESS smoother (span of 20% of the data) emphasises overall patterns. A long-term decline in water quality (increasing nitrate) seems to have recently been replaced by an improving trend (reducing nitrate).

pastoral rivers in areas of 'mature' pastoral land management, such as the Waikato, Manawatu, and Taranaki, display recently slowed water quality declines. And some rivers appear to have actually *improved* in water quality. For example, in the Taranaki Region, water quality at some monitoring sites appears to have improved (Figure 5), plausibly as an outcome of widespread riparian management efforts in that region (Gary Bedford, Taranaki Regional Council, pers. comm.).

Arrest of the widespread decline in river water quality, with modest improvement in some key variables, is encouraging for showing that continuing degradation in our rivers and downstream waters is not inevitable and can even be reversed. For example, eutrophic Lake Rotorua, Bay of Plenty, has shown clear trends of improving water quality since about 2001, most likely in response to basin-wide reductions in nutrient loading (Abell et al. 2012).

ENDURING CHALLENGES IN RIVER WATER QUALITY

Although water quality is routinely perceived as the largest environmental problem by New Zealanders (Hughey et al. 2011), it continues to be degraded by land uses, particularly pastoral land use, and other strongly soil-disturbing land uses such as (opencast) mining and urban expansion. These sources of diffuse water pollution continue to negate the gains made from improved wastewater treatment and the resulting clean-up of point-source pollution in New Zealand.

Surveys by Hughey et al. (2011) have shown that New Zealanders are increasingly aware that pastoral farming is the main source of degradation of water quality in this country. This rising awareness of the source of the problem should drive increasing action by the pastoral industries towards improving their environmental performance.

That said, it needs to be recognised that controlling land-use impacts on water quality (diffuse pollution) is difficult – much more so than controlling point-source pollution. Changing to more benign (rural) land uses than grazed pasture is an option that may well have to be pursued in some areas, particularly in the catchments of some sensitive lakes such as Taupo (Morgenstern et al. 2011). Otherwise, beneficial management practices (BMPs) seem to hold out hope that at least a modest improvement in water quality can be achieved by catchment-wide implementation of good practice. BMPs may be put into two main categories: those that reduce contaminants at source and those that reduce entry of such contaminants into streams once mobilised. Both will be needed. In areas of urban expansion, increasing adoption of 'water sensitive design' (control of stormwater at source) will be needed, as well as management of stormwater, once generated, by detention ponds and wetlands (Campbell et al. 2004).

Quinn et al. (2009) discuss various mitigation strategies (BMPs) for pastoral agriculture with reference to a 'toolbox' (McKergow et al. 2007; Monaghan et al. 2008) for predicting efficiency and estimating costs of controlling mobilisation of nutrients, sediment and faecal microbes. For example, deferred irrigation of dairy-shed effluent (or adoption of Advanced Pond Systems, APS) is expected to greatly reduce runoff of effluent pollutants from wet soils in wet weather. Similarly, the use of herd homes or stand-off pads for dairy cattle should greatly reduce wet weather runoff of diffuse pollution from wet grazed pasture (Collins et al. 2007).

There are also some well-known options in the 'tool box' (Monaghan et al. 2008) for control of diffuse pollutants from pastoral farming, once generated. Wetlands have been referred to as the 'kidneys' of landscapes (Mitsch et al. 2009) and the widespread destruction of natural wetlands by drainage has greatly reduced the diffuse pollutant 'filtering' capability of agricultural land. Existing wetlands should ideally be protected (fenced) and maintained for their pollutant-transforming functions. Tanner and Sukias (2007) suggest that, where natural wetlands have been destroyed, constructed wetlands amounting to about 1% of the catchment area can usefully attenuate the 'big three' categories of diffuse pollutants (nutrients, sediment and faecal microbes) before they enter streams proper. Similarly, riparian buffers, ideally comprised of forest plantings with upslope grassed filter strips, can reduce diffuse pollutant loading on streams by a wide range of mechanisms (Quinn et al. 2007). These include reduction in pollution at source by exclusion of livestock, sedimentation of particulate contaminants on riparian soils, infiltration of runoff water into riparian soils, and uptake and transformation of nutrients by riparian soils and plants.

Climate change seems likely to drive an increasingly invidious interaction with water quality later this century. The two main climate drivers are likely to be (1) reduced flows (at least in the already drier areas east of the main dividing ranges) – causing less flow for dilution of contaminants, and (2) higher river water temperatures (implying more rapid biochemical reactions), leading to more rapid spiralling of nutrients and more frequent and widespread oxygen stress in waters. Analysis of data from the NRWQN sites (Piet Verburg, NIWA, pers. comm.) suggests widespread increase in river water temperatures in New Zealand, at a rate broadly as expected from reported air temperatures. It remains to be seen how the interacting pressures of increasing water temperatures and reduced river flows in some areas will further pressure water quality of rivers and downstream receiving waters that are already heavily loaded by diffuse pollutants (Howard-Williams et al. 2011).

River water quality as an ecosystem service

River waters represent a very wide range of benefits associated with consumptive use (e.g. water supply, irrigation), direct use (e.g. hydro-power, recreation, waste disposal) and non-use values (water-based scenery, aquatic biodiversity) (Table 3). Where there are markets for such services (e.g. hydropower generation) monetary valuation is relatively straightforward. But more often, monetary valuation is difficult and indirect (Turner et al. 2010).

Water quality is, to a large extent, an ecosystem service of *land* (particularly riparian land and wetlands). More specifically (good) water quality is an ecosystem service provided by (relatively) undisturbed land on which generation of diffuse pollutants is limited, and these pollutants usually just move downslope a short way before being trapped, so that they seldom reach stream channel networks. River channels themselves have impressive self-cleaning capacity, particularly as regards hyporheic exchange, which clarifies water by deposition of fine particles (Packman and MacKay 2003), and nutrient transformation. Therefore (good) river water quality may be viewed as an ecosystem service of (relatively undisturbed) land *and* (healthy, unsedimented) river channels.

A number of studies have been performed over several decades on valuing fresh waters, and North American studies (1971–1997) are reviewed by Wilson and Carpenter (1999). Three main methods for valuing non-market aspects of fresh waters were used in almost all of the reviewed studies: (1) the travel cost method – in which the cost to all visitors of travelling to a water body and conducting passive or active recreation is surveyed to estimate aggregate value, (2) hedonic valuation – in which the effects on market values (typically property prices) of aquatic ecosystem services are estimated by use of statistical models, and (3) contingent valuation methods – based on surveys, using questionnaires, querying people's willingness to pay (WTP) for services associated with water (including *improved* services associated with improved water quality).

An example of the hedonic valuation approach using property prices affected by river water quality is that of Poor et al. (2007) who estimated values of improvements (reductions) in suspended sediment and nitrogen in the St Mary's River basin draining to Chesapeake Bay, eastern USA. Two fairly recent examples of the contingent valuation approach applied to rivers are, first, that of Loomis et al. (2000), who valued the rehabilitation of ecosystem services on the (agriculturally impacted) South Platte River near Denver, Colorado, USA, and, second, that of Bateman et al. (2006), who valued water quality improvements for the urbanised River Tame running through Birmingham, England.

In some such studies water quality is considered separately from water quantity. Where rivers and lakes are valued in their entirety, it is difficult to disentangle water *quality* (which is multi-dimensional and complex – Table 2) from water *quantity* aspects such as flow. Indeed, the multiple dimensions of river water quality combined with the large number of values and uses of rivers (Table 3) present a formidable obstacle to valuation. Where water quality *has* been the special focus, often just one (usually easily understood) aspect of water quality has been emphasised such as visual clarity or dissolved oxygen (Wilson and Carpenter 1999).

We can get a crude insight into the likely scale of (non-market) river water quality value in New Zealand from ecosystem service valuation exercises at national scale. For example, tourism contributes NZ $6.2 billion per year to GDP. Obviously a considerable portion of this (perhaps, conservatively, around 10%) may be associated with our many rivers, which contribute greatly to scenic beauty and are the focus of considerable recreational (including tourist) activity (angling, hiking, swimming, jet boating). This would suggest an aggregate value of our rivers, *for tourism alone*, of perhaps NZ$0.6 billion per year of which perhaps half relates to water quality rather than quantity.

A potential approach for assessing the value of ecosystem services associated with river water quality lies in considering restoration costs (Spangenberg and Settele 2010), but such approaches appear not to have yet been applied to river water quality in New Zealand. This is not surprising, because of the 'multi-dimensionality' of water quality and overall ecosystem complexity. Pricing nature will always be difficult, and it is perhaps most difficult where, as with water quality, there are multiple (sometimes interacting) aspects and dimensions and multiple values and uses.

ACKNOWLEDGEMENTS

This work was supported by the 'Cumulative effects of multiple stressors…' programme (MBIE contract CO1X1005) and a NIWA project on water monitoring and reporting in the 'Values, Monitoring and Outcomes' (VMO) research programme led by Landcare Research (MBIE contract CO9X1003). I am grateful for the many discussions over many years with colleagues from NIWA and other agencies with whom I have worked on aspects of river water quality, including Graham McBride (water quality monitoring design, statistical tools, microbial water quality), Richard Storey (bio-monitoring), Bob Wilcock (effects of pastoral agriculture), John Quinn (river ecosystems), Graham Bryers (water quality monitoring operations), Deborah Ballantine (water quality time trends), Robert Ward (Emeritus Professor at Colorado State University; water quality monitoring design) and Dave Smith (retired; water quality monitoring design and operation). Special thanks to Bob Wilcock and Graham McBride for review of an early draft of the manuscript. Thanks to colleagues in the VMO programme, particularly Suzie Greenhalgh, Andrew Fenemor, and Garth Harmsworth (all of Landcare Research), and Jim Sinner (Cawthron Institute), for collegiality at the water science–policy interface. I also thank staff of the Ministry for the Environment and state-of-environment officers from all of the regional councils in New Zealand with whom I have worked in the NEMaR (National Environmental Monitoring and Reporting) project.

REFERENCES

Abell J, Hamilton D, McBride C, Stephens T, Scarsbrook M 2012. Analysis of Lake Rotorua water quality trends: 2001-2012. Environmental Research Institute Report, 010, University of Waikato. Report to DairyNZ. 22 p.

ANZECC 2000. Australian and New Zealand guidelines for fresh and marine water quality, Vol. 1. The Guidelines. Australia and New Zealand Environment and Conservation Council (ANZECC) and Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ).

APHA 2005. Standard methods for the examination of water and wastewater. 21st Edition (Eaton AD, Clesceri LS, Rice EW, Greenberg AE eds), American Public Health Association, American Waterworks Association, Water Environment Federation. Washington DC (www.standardmethods.org)

Ballantine DJ, Davies-Colley RJ 2009. Water quality trends at NRWQN sites for the period 1989–2007. NIWA Client Report HAM2009-026 for Ministry for the Environment. 36 p.

Ballantine DJ, Davies-Colley RJ 2010. Water quality trends at NRWQN sites for the 21 years: 1989-2009. NIWA client report HAM 2010-142 for Ministry for Environment. 14 p.

Ballantine DJ, Booker D, Unwin M, Snelder TH 2010. Analysis of national river water quality data for the period 1998–2007. NIWA Client Report CHC2010-038 for the Ministry for the Environment. 71 p.

Basher LR, Hicks DM, Clapp B, Hewitt T 2011. Sediment yield response to large storm events and forest harvesting, Motueka River, New Zealand. New Zealand Journal of Marine and Freshwater Research 45: 333–356.

Bateman IJ, Cole M, Georgiou S, Hadley D 2006. Comparing contingent valuation and contingent ranking: A case study considering the benefits of urban river water quality improvements. Journal of Environmental

Management 79: 221–231.
Belsky AJ, Matzke A, Uselman S 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. Journal of Soil and Water Conservation 54: 419–431.
Biggs BJF 2000. Eutrophication of streams and rivers: dissolved nutrient-chlorophyll relationships for benthic algae. Journal of the North American Benthological Society 19: 17–31.
Bowers DG, Binding CE 2006. The optical properties of mineral suspended particles: A review and synthesis. Estuarine Coastal and Shelf Science 67: 219–230.
Campbell N, D'Arcy B, Frost A, Novotny V, Sansom A 2004. Diffuse pollution. An introduction to the problems and solutions. 1st edn. London, IWA. 322 p.
CCME 2012 Canadian water quality guidelines: nitrate ion. Canadian Water Quality Guidelines: Scientific Criteria Document PN 1470.
Clapcott JE, Young RG, Harding JS, Matthaei CD, Quinn JM, Death RG 2011. Sediment assessment methods: protocols and guidelines for assessing the effects of deposited fine sediment on in-stream values. Cawthron Institute, Nelson New Zealand. 108 p.
Close ME, Davies-Colley RJ 1990. Baseflow water chemistry in New Zealand rivers. 1. Characterisation. New Zealand Journal of Marine and Freshwater Research 24: 319–341.
Collins R, McLeod M, Hedley M, Donnison A, Close M, Hanly J, Horne D, Ross C, Davies-Colley R, Bagshaw C et al. 2007. Best management practices to mitigate faecal contamination by livestock of New Zealand waters. New Zealand Journal of Agricultural Research 50: 267–278.
Conley DJ, Paerl HW, Howarth RW, Boesch DF, Seitzinger SP, Havens KE, Lancelot C, Likens GE 2009. Controlling eutrophication: nitrogen and phosphorus. Science 323: 1014–1015.
Davies-Colley RJ 1988. Measuring water clarity with a black disc. Limnology and Oceanography 33: 616–623.
Davies-Colley RJ 1997. Stream channels are narrower in pasture than in forest. New Zealand Journal of Marine and Freshwater Research 31: 599–608.
Davies-Colley RJ 2009. Land use and water quality in New Zealand – an overview. Water Air and Soil Pollution 162: 32–35.
Davies-Colley RJ, Smith DG 2001. Turbidity, suspended sediment, and water clarity: A review. Journal of the American Water Resources Association 37: 1085–1101.
Davies-Colley RJ, Wilcock RJ 2004. Water quality and chemistry in running waters. In: Harding J, Mosley MP, Pearson CP, Sorell B eds Freshwaters of New Zealand. Christchurch, Caxton Press. Pp. 11.1–11.17.
Davies-Colley RJ, Vant WN, Smith DG 2003. Colour and clarity of natural waters. Science and management of optical water quality. Caldwell, New Jersey, reprinted by Blackburn Press. 310 p.
Davies-Colley RJ, Nagels JW, Lydiard E 2008. Faecal bacterial dynamics and yields from an intensively dairy-farmed catchment. Water Science and Technology 57: 1519–1523.
Davies-Colley RJ, Smith DG, Ward R, Bryers GG, McBride GB, Quinn JM, Scarsbrook MR 2011. Twenty years of New Zealand's National Rivers Water Quality Network: benefits of careful design and consistent operation. Journal of the American Water Resources Association 47: 750–771.
Donnison AM, Ross CM 1999. Animal and human faecal pollution in New Zealand rivers. New Zealand Journal of Marine and Freshwater Research 33: 119–128.
Droppo IG 2001. Rethinking what constitutes suspended sediment. Hydrological Processes 15: 1551–1564.
Dufour A, Bartram J, Bos R, Gannon V eds 2012. Animal waste, water quality and human health. Geneva, World Health Organization and International Water Association. 476 p.
Elliott AH, Davies-Colley RJ, Parshatom A, Ballantine DJ 2013. Load based approaches for modelling visual clarity in streams at regional scale. Water Science and Technology 67: 1092–1096.
Fahey B, Duncan MJ, Quinn J 2004. Impacts of forestry. In: Harding J, Mosley MP, Pearson CP, Sorell B eds Freshwaters of New Zealand. Christchurch, Caxton Press. Pp. 33.1–33.16.
Fahey BD, Rowe LK 1992. Land-use impacts. In: Mosley MP ed. Waters of New Zealand. Wellington, New Zealand Hydrological Society Inc. Pp. 265–284.
Hamill KD, McBride GB 2003. River water quality trends and increased dairying in Southland, New Zealand. New Zealand Journal of Marine and Freshwater Research 37: 323–332.
Hickey CW 2013. Updating nitrate toxicity effects on freshwater aquatic species. NIWA Client Report HAM2013-009. 39 p.
Hicks DM, Gomez B, Trustrum NA 2000. Erosion thresholds and suspended sediment yields, Waipaoa River Basin, New Zealand. Water Resources Research 36: 1129–1142.
Hicks DM, Quinn JM, Trustrum NA 2004. Stream sediment load and organic matter. In: Harding J, Mosley MP, Pearson CP, Sorell B eds Freshwaters of New Zealand. Christchurch, Caxton Press. Pp. 12.1–12.16.
Howard-Williams C, Davies-Colley RJ, Rutherford JC, Wilcock RJ 2011. Diffuse pollution and freshwater degradation: New Zealand perspectives. In: van Bochove E, Vanrolleghem PA, Chambers PA, Therialt G, Novotna B, Burkart MR eds Issues and solutions to diffuse pollution. 14th International Water Association (IWA) conference on Diffuse Pollution and Eutrophication. Pp. 126–140.
Hughes AO, Quinn JM, McKergow LA 2012. Land use influences on suspended sediment yields and event sediment dynamics within two headwater catchments, Waikato, New Zealand. New Zealand Journal of Marine and Freshwater Research 46: 315–333.
Hughey KFD, Booth KL 2012. Monitoring the state of New Zealand rivers: How the River Values Assessment System can help. New Zealand Journal of Marine and Freshwater Research 46: 545–556.
Hughey KFD, Cullen R, Kerr GN 2011. A decade of public perceptions of the New Zealand environment: a focus on water and its management. Public perceptions of environment in New Zealand. 25 p. http://nzae.org.nz/wp-content/uploads/2011/08/Hughey_et_al_A_Decade_of_Public_Perceptions.pdf
Johnson DL, Ambrose SH, Bassett TJ, Bowmen ML, et al. 1997. Meanings of environmental terms. Journal of Environmental Quality 26: 581–589.
Julian JP, Doyle MW, Stanley EH 2008. Empirical modeling of light availability in rivers. Journal of Geophysical Research – Biogeosciences 113: G03022.
Kirk JTO 1985. Effects of suspensoids (turbidity) on penetration of solar radiation in aquatic ecosystems. Hydrobiologia 125: 195–208.
Loomis J, Kent P, Strange L, Fausch K, Covich A 2000. Measuring the total economic value of restoring ecosystem services in an impaired river basin: results from a contingent valuation survey. Ecological Economics 33: 103–117.
MacLeod LM,, Moller H 2006. Intensification and diversification of New Zealand agriculture since 1960: an evaluation of current indicators of land use change. Agriculture, Ecosystems and Environment 115: 201-218.
McBride GB 2011. Explaining differential sources of zoonotic pathogens in intensively-farmed catchments using kinematic waves. Water Science and Technology 63: 695–703.
McBride GB, Davies-Colley RJ, Bargh M 2013. Toward a nationally consistent and dependable surface freshwater monitoring programme for New Zealand. Description and costs including quality assurance. NIWA Client Report HAM2013-024. 41 p.
McDowell RW, Wilcock RJ 2008. Water quality and the effects of different pastoral animals. New Zealand Veterinary Journal 56: 289–296.
McDowell RW, Larned ST, Houlbrooke DJ 2009. Nitrogen and phosphorus in New Zealand streams and rivers: control and impact of eutrophication and the influence of land management. New Zealand Journal of Marine and Freshwater Research 43: 985–995.
McKergow LA, Davies-Colley RJ 2010. Stormflow dynamics and loads of Escherichia coli in a large mixed land use catchment. Hydrological Processes 24: 276–289.
McKergow LA, Tanner CC, Monaghan RM 2007. Stocktake of diffuse attenuation tools for New Zealand pastoral farming systems. NIWA Client Report HAM2007-161. 105 p.
MfE 2007. Environment New Zealand 2007. Wellington, Ministry for the Environment. 428 p.
MfE 2013. Freshwater reform 2013 and beyond. Wellington, Ministry for the Environment. 57 p.
Mitsch WJ, Gosselink JG, Zhang L, Anderson CJ 2009. Wetland ecosystems. Hoboken, Wiley.
Monaghan RM, De Klein CAM, Muirhead RW 2008. Prioritisation of farm-scale remediation efforts for reducing losses of nutrient and faecal indicator organisms to waterways: a case study of New Zealand dairy farming. Journal of Envionmental Management 87: 609–622.
Moore CJ, Lattin GL, Zellers AF 2011. Quantity and type of plastic debris flowing from two urban rivers to coastal waters and beaches of Southern California. Journal of Integrated Coastal Zone Management 11: 65–73.
Morgenstern U, Hadfield J, Vant WN 2011. Lake Taupo – threat from delayed impact of land use via old groundwater inflow. International Water Association (IWA) conference of the Diffuse Pollution Specialist Group (DIPCON2011).

Nguyen ML, Sheath GW, Smith CM, Cooper AB 1998. Impact of cattle treading on hill land 2. Soil physical properties and contaminant runoff. New Zealand Journal of Agricultural Research 41: 279–290.
Packman AI, MacKay JS 2003. Interplay of stream-subsurface exchange, clay particle deposition, and streambed evolution – art. no. 1097. Water Resources Research 39: 1097.
Parkyn S, Wilcock RJ 2004. Impacts of agricultural land use. In: Harding J, Mosley MP, Pearson CP, Sorell B eds Freshwaters of New Zealand. Christchurch, Caxton Press. Pp. 34.1–34.16.
Parkyn SM, Davies-Colley RJ, Scarsbrook MR, Halliday NJ, Nagels JW, Marden M, Rowan D 2006. Pine afforestation and stream health: a comparison of land-use in two soft rock catchments, East Cape, New Zealand. New Zealand Natural Sciences 31: 113–135.
PCE 2004. Growing for good? The sustainability of intensive farming in New Zealand. 254 p.
Poor PJ, Pessagno KL, Paul RW 2007. Exploring the hedonic value of ambient water quality: A local watershed-based study. Ecological Economics 60: 797–806.
Quinn JM, Stroud MJ 2002. Water quality and sediment and nutrient export from New Zealand hill-land catchments of contrasting land use. New Zealand Journal of Marine and Freshwater Research 36: 409–429.
Quinn JM, Cooper AB, Davies-Colley RJ, Rutherford JC, Williamson RB 1997. Land-use effects on habitat, periphyton, and benthic invertebrates in Waikato hill country streams. New Zealand Journal of Marine and Freshwater Research 31 (Special issue): 579–597.
Quinn JM, Dodd M, Thorrold BS 2007. The Whatawhata integrated catchment management project: the story so far. Proceedings of the New Zealand Grasslands Association 69: 229–233.
Quinn JM, Wilcock RJ, Monaghan RM, McDowell RW, Journeaux PR 2009. Grassland farming and water quality in New Zealand. Irish Journal of Agri-environmental Research 7: 69–88.
Scarsbrook MR 2006. State and trends in the National River Water Quality Network (1989–2005). Wellington, Ministry for the Environment. 42 p.
Scarsbrook MR, McBride CG, McBride GB, Bryers GG 2003. Effects of climate variability on rivers: Consequences for long term water quality analysis. Journal of the American Water Resources Association 39: 1435–1447.
Sinton LW, Hall CH, Lynch PA, Davies-Colley RJ 2002. Sunlight inactivation of fecal indicator bacteria and bacteriophages from waste stabilization pond effluent in fresh and saline waters. Applied and Environmental Microbiology 68: 1122–1131.
Smith CM, Wilcock RJ, Vant WN, Smith DG, Cooper AB 1993. Towards sustainable agriculture: freshwater quality in New Zealand and the influence of agriculture. MAF Policy Technical Paper 93/10. Wellington, Ministry of Agriculture and Forestry.
Smith DG, McBride GB, Bryers GG, Wisse J, Mink DFJ 1996. Trends in New Zealand's National River Water Quality Network. New Zealand Journal of Marine and Freshwater Research 30: 485–500.
Smith DG, Davies-Colley RJ, Knoeff J, Slot GWJ 1997. Optical characteristics of New Zealand rivers in relation to flow. Journal of the American Water Resources Association 33: 301–312.
Spangenberg JH, Settele J 2010. Precisely incorrect? Monetising the value of ecosystem services. Ecological Complexity 7: 327–337.
Stott R, Davies-Colley RJ, Nagels JW, Donnison AM, Ross C 2011. Differential behaviour of Escherichia coli and Campylobacter spp. in a stream draining dairy pasture. Journal of Water and Health 9: 59–69.
Stumm W, Morgan JJ 1981. Aquatic chemistry. 2nd edn. New York, John Wiley.
Suren A, Elliott AH 2004. Impacts of urbanisation on streams. In: Harding J, Mosley MP, Pearson CP, Sorell B eds Freshwaters of New Zealand. Christchurch, Caxton Press. Pp. 35.1–35.17.
Tanner CC, Sukias J 2007. Multiyear nutrient removal performance of three constructed wetlands intercepting tile drain flows from grazed pastures. Journal of Environmental Quality 40: 620–633.
Thrush SF, Hewitt JE, Norkko A 2003. Catastrophic sedimentation on estuarine sandflats: recovery of macrobenthic communities is influenced by a variety of environmental factors Ecological Applications 13: 1433–1455.
Till DG, McBride GB, Ball A, Taylor K, Pyle E 2008. Large scale freshwater microbiological study: rationale, results and risks. Journal of Water and Health 6: 444–460.
Trimble SW, Mendel AC 1995. The cow as a geomorphic agent—A critical review. Geomorphology 13: 233–253.
Turner RK, Morse-Jones S, Fisher B 2010. Ecosystem valuation: a sequential decision support system and quality assessment issues. Annals of the New York Academy of Sciences 1185: 79–101.
Vant WN 2001. New challenges for the management of plant nutrients and pathogens in the Waikato River, New Zealand. Water Science and Technology 43: 137–144.
Wilcock RJ, Nagels JW, McBride GB, Collier KJ 1998. Characterisation of lowland streams using a single-station diurnal curve analysis model with continuous monitoring data for dissolved oxygen and temperature. New Zealand Journal of Marine and Freshwater Research 32: 67–79.
Wilcock RJ, Monaghan RM, Thorrold RS, Meredith AS, Betteridge K, Duncan MJ 2007. Land-water interactions in five contrasting dairy catchments: issues and solutions. Land Use and Water Resources Research 7: 2.1–2.10.
Wilkinson RC, McKergow LA, Davies-Colley RJ, Ballantine DJ, Young RG 2011. Modelling storm-event E. coli pulses from the Motueka and Sherry Rivers in the South Island, New Zealand. New Zealand Journal of Marine and Freshwater Research 45: 369–393.
Williamson RB 1991. Urban runoff data source book: A manual for preliminary evaluation of stormwater impacts. Hamilton, Water Quality Centre, DSIR. 71 p.
Wilson MA, Carpenter SR 1999. Economic valuation of freshwater ecosystem services in the United States: 1971–1997. Ecological Applications 9: 772–783.
Young M, Adams NJ 2010. Plastic debris and seabird presence in the Hauraki Gulf, New Zealand. New Zealand Journal of Marine and Freshwater Research 44: 167–175.
Young RG, Quarterman AJ, Eyles RF, Smith RA, Bowden WB 2005. Water quality and thermal regime of the Motueka River: influences of land cover, geology and position in the catchment. New Zealand Journal of Marine and Freshwater Research 39: 803–825.
Zanevald JRV, Pegau WS 2003. Robust underwater visibility parameter. Optics Express 11: 2997–3009.