Land-use impacts on freshwater and marine environments in New Zealand

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1.1 Pressure-state-impact relationships in New Zealand freshwater and marine environments

The Ministry for the Environment requires a review of peer-review and published or publicly available literature concerning land use impacts on New Zealand's freshwater and coastal environments. The review presented here is structured around pressure-state-impact (PSI) relationships that link land use **pressures** (e.g., elevated contaminant loads, water abstraction) to changes in environmental **state** (e.g., chemical-physical water quality, flow regime), and in turn, to **impacts** on ecological, social, cultural and economic values in New Zealand freshwater and marine environments. The Ministry and Statistics New Zealand have adopted the PSI framework for environmental reporting as it provides more information about human activities and environmental effects than descriptions of environmental state and trends alone. In the current report, the state and impact variables correspond to the environmental reporting indicators for the freshwater and marine domains. All three major classes of pressure variables in PSI relationships will be included in our review: land cover (e.g., high-producing grassland, planted forest), land use classes (e.g., arable cropping, dairying, urban residential), and land management practices (e.g., effluent disposal, stormwater treatment, fertiliser application).

1.1.1 Rivers

Land-use pressures that affect river state and impact river values fall into five broad classes: 1) contaminant loss from land and conveyance to rivers; 2) the presence of livestock in river environments (primarily cattle and deer); and 3) alterations to natural riparian vegetation on river boundaries; 4) abstraction of water from river channels and from aquifers connected to rivers; 5) alterations in natural hydrological processes in addition to direct abstraction (e.g., flow reductions due to evapotranspiration by planted forests); and 6) channel modifications used to prevent erosion and over-bank flooding, and improve flood conveyance (e.g., stopbank construction, channelisation). State changes resulting from these pressures include increased contaminant concentrations, habitat degradation, changes in light and thermal conditions, and flow alteration. The consequences of alterations in river state are wide-ranging, and include loss or reduction in ecological, social, cultural and economic values such as biodiversity, ecosystem functions (e.g., nitrogen removal), populations of threatened species, opportunities for customary harvests, sports fisheries and tourism.

The most common associations between land use pressures and state and impact variables in rivers use land-cover as a pressure. In most of the New Zealand studies that employ land-cover, river monitoring sites are designated as agricultural (i.e., pastoral), planted forest, urban or natural (e.g., native forest, scrub, tussock) based on the single dominant land cover class in the upstream catchment. In other studies, land-cover is used as a continuous variable, and the associations describe changes in the response variable as a function of increasing land cover of a given class (e.g., river nitrogen concentrations as a function of the proportion of urban land cover in the upstream catchment). In several national and regional-scale studies, concentrations of nitrogen (N), phosphorus (P) and the faecal indicator bacterium *E. coli* increased, and visual clarity and invertebrate and fish-based ecological health indicators decreased as the proportion of intensive agricultural and urban cover increased. In one of the national studies, N and P also increased and visual clarity decreased as the proportion of planted forest increased.

The national- regional- and catchment-scale studies that use land cover as a categorical variable also had comparable results. In general, adverse impacts on river water quality and ecological indicators varied between land cover categories in the following order: urban > pastoral > planted forest > natural.

In comparison with land cover, far less research in New Zealand has focused on identifying associations between land use classes (e.g., plantation forestry, sheep and beef grazing) and river water quality and ecological conditions. In the last 20 years, most of the work on the potential effects of animal-based (pastoral) and plant-based (horticultural and arable cropping) agriculture has focused on contaminant losses and their reduction, rather than directly on in-stream effects. The published research on contaminant losses is primarily concerned with pastoral agriculture, and relatively little research covers contaminant losses from horticultural and arable cropping. We summarised a large body of recent research to identify patterns in contaminant losses for different land use classes and for four major classes of contaminants, N, P, sediment and faecal bacteria (usually represented by *E. coli*). The different classes differ in specific sources (e.g., fertiliser, erosion, animal waste) and the pathways by which they are transported to rivers (e.g., leaching, overland flow). For N, reported loss rates ranged from 0.8 to 157 kg $ha^{-1}yr^{-1}$, and for P, loss rates ranged from 0.08 to 7 kg $ha^{-1}yr^{-1}$. For both N and P, losses were greatest for grazed forage crops, and least for deer and sheep grazing. For sediment, reported loss rates ranged from 22 to 8,025 kg $ha^{-1}yr^{-1}$. Sediment losses were greatest for grazed forage crops, and least for dairy farming. There are relatively few studies of faecal microbe losses from pastoral land uses, and too few to distinguish differences among specific land use classes.

Very little research effort has been focused on the direct effects of arable and horticultural agriculture on New Zealand rivers. The level of concern has been greater for pastoral agriculture due to the larger land area used by that sector. In addition, reports of contaminant losses from arable and horticultural crop land are scarcer than for pastoral agriculture, and most of those focus on N leaching and sediment erosion. Large-scale P loss from both arable and horticultural crop land has been assumed to be negligible due to the small areas occupied by these land uses. In contrast, annual N leaching rates in excess of 100 kg ha^{-1} yr^{-1} have been reported from arable crops, which are comparable to those reported for intensive pastoral agriculture.

Compared with land cover and land use, even less research has focused on the effects of specific land management practices such as fertiliser use and agricultural effluent disposal. There are some exceptions, however. The removal, retention and replanting of riparian vegetation in catchments used for pastoral agriculture and forestry is well-studied, and the effects of stock access to and exclusion from rivers are the topics of recent and ongoing research. Removal of riparian vegetation was standard practice in many agricultural areas of New Zealand, and the loss of shade and the loss of contaminant retention by riparian processes has contributed to nutrient enrichment, elevated levels of suspended and deposited sediment, and nuisance growths of macrophytes and algae in many pastoral streams. Long-term studies of the planted riparian buffers have indicated that habitat conditions and biotic communities in adjacent rivers approach reference conditions as the riparian vegetation matures.

Stock access to streams and riparian margins causes bank erosion and reduces soil infiltration, which reduces the capacity for contaminant retention in riparian margins. Stock access also degrades fish and invertebrate habitat, and substantially increases faecal bacteria input. Very recent studies of the effectiveness of stock exclusion indicate reported reductions in suspended sediment concentrations and increased visual clarity in intensively-farmed pastoral streams, and incremental improvements in macroinvertebrate-based ecological health indicators, but the time courses for these improvements appear to be longer than for suspended sediment and visual clarity.

The effects of harvesting planted forests are also intensively studied, as harvesting poses risks of substantial increases in sediment input, loss of shade, increased heating and other pressures. Forestry management practices comprise multiple phases (e.g., land preparation, planting, fertilisation, pest and weed control, harvesting). Harvesting is the only phase for which effects on rivers in planted forest catchment are well characterised, as harvesting has the greatest potential for adverse impacts on freshwater and marine environments, particularly clear-felling to stream margins. Harvesting reduces rainfall interception and evapotranspiration, resulting in increased run-off and soil erosion during the period between harvesting and canopy closure by the following rotation. The increased stream power in post-harvest flow regimes can exacerbate channel bank and bed scouring. In the absence of riparian buffers, tree harvesting can also result in elevated water temperatures, loss of shade and subsequent periphyton proliferations, and accumulations of organic matter in channels.

In addition to the effects of land-derived contaminants and alterations in instream habitat, land use can affect river state and impact river values through changes in flow regimes. These changes reflect a complex series of steps that can include water abstraction from rivers and aquifers, interbasin water transfer, impoundment for electricity generation or future irrigation, irrigation return flows, and alterations in groundwater levels and river flows associated with altered evapotranspiration by catchment vegetation. A recent national analysis of consents to take and use water indicated that irrigation accounts for most out-of-river water use in New Zealand, with remainder used for industry, drinking water and other uses. Among the various classes of water use, irrigation also has the greatest potential to alter flow regimes, based on both total volume abstracted and number of rivers affected. As a broad generalisation, the effects of irrigation takes on river flow regimes are to reduce summer low-flows, reduce low-flow variability, and increase the severity of seasonal flow variation. A reduction in mid-range flushing flows (e.g., flows that mobilise fine sediment or remove growths of nuisance algae) also occurs when abstraction levels are high or where flood harvesting is used to fill off-channel water stores.

1.1.2 Urban streams

Urban development exerts the following pressures on urban waterways: 1) increased volumes of rainfall runoff (stormwater), accelerated transport to receiving environments and reduced recharge, due to piped drainage networks and the replacement of vegetated land cover with impervious cover; 2) increased contaminant input, including sediments from earthworks, copper and zinc derived from traffic sources and building materials; and nutrients and pathogens discharged in wastewater overflows; and 3) modification of stream and river channels and their margins, including straightening, lining, and the removal of riparian vegetation. As for rivers, the primary state changes changes resulting from these pressures are increased contaminant concentrations, habitat degradation, changes in light and thermal conditions, and flow alteration.

In multiple national-scale studies that included categorical comparisons between land-cover classes, state-of-environment (SoE) monitoring sites in the urban class consistently had higher N, P and *E. coli* concentrations, lower visual clarity and lower macroinvertebrate-based health indicator scores than all other land cover classes. Although water quality at urban monitoring sites tends to be poorer than at sites in other land-cover classes, urbanised river reaches comprise less than one percent of New

Zealand river reaches by length, based on the River Environment Classification (REC). Urban land cover comprises less than one percent of all land cover in New Zealand.

Results from a recent analysis of water quality data from Auckland, Christchurch and Wellington and a national-scale study of SoE monitoring sites indicated that all contaminants included in the analyses increased in concentration with increasing urban land cover in the upstream catchment. Urban land cover in these studies was based on a general land-cover category used in the Land Cover Database (LCDB), 'built-up area'. Several studies in New Zealand have used impervious cover as a more fine-grained land-use pressure variable than built-up area. In the latter studies, macroinvertebrate-based ecological health metrics decreased, and concentrations of copper, lead, zinc and polycyclic aromatic hydrocarbons increased with increasing proportions of impervious cover.

Within urban centres, there are some indications of differences in water quality between specific land-use classes. For example, copper, lead and zinc concentrations in urban stream water and sediment and stormwater are generally higher at sites in commercial and industrial areas than in residential areas. Studies of road runoff in Auckland indicate that vehicle use is the principal source of copper in adjacent streams, and that rates of copper and zinc emission are highest at sites with high frequency vehicle braking and acceleration.

The effects of two land management practices on urban stream conditions are well-studied in New Zealand, earthworks during urban construction and stormwater treatment. Earthworks lacking effective erosion control systems are a major source of sediment in urban streams. Rainfall runoff from these sites transports high sediment loads to streams, resulting in reduced water clarity and increased sediment deposition. Turbidity and suspended sediment concentration in urban streams near construction sites can be several orders of magnitude higher than in outlying streams.

Stormwater management in New Zealand has traditionally focused on drainage and flood control, with urban runoff piped and discharged to the nearest waterway with little consideration for ecological impacts. However, since the late 20th century improving stormwater quality has been a research priority. A number of New Zealand studies have assessed the effectiveness of treatment devices in removing suspended solids, organic matter, nutrients and metals from stormwater. The results of these studies indicate that roadside swales and floating wetlands can improve stormwater treatment compared to stormwater ponds alone.

In contrast to the recent focus on stormwater treatment performance, outcomes of stormwater treatment in freshwater and coastal receiving environments have received far less attention. A multidimensional approach that integrates urban planning and water management, water-sensitive urban design (WSUD), has been assessed in recent studies using urban streams in Auckland. The results of those studies indicate that macroinvertebrate-based ecological health scores were higher and copper and zinc concentrations in stream sediments were lower in WSUD catchments compared with catchments that had undergone conventional urban development.

1.1.3 Lakes

The major classes of land-use pressures that affect lake state and impact lake values are contaminant loss from land and input to lakes through groundwater and tributary inflows, and hydrological alterations such as impoundments and abstractions on tributaries. Contaminant loading to lakes affects multiple state variables and values, including generally adverse effects on biodiversity, populations of threatened species, opportunities for customary harvests, sports fisheries and tourism. In addition, faecal contaminants and algal toxins in lakes pose public health risks.

Hydrological alterations affect lake water levels, mixing, thermal regimes and other internal processes, as well as contaminant loading from land.

We reviewed 27 New Zealand studies linking land-use pressures to lake state and impacts. Two types of pressure variables were used in these studies: land cover (as proportions of catchment area in different land-cover classes); and total P and total N loads to lakes, which were estimated from tributary monitoring data or predicted by catchment modelling. The most intensively studied effect of land use pressure on lakes is phytoplankton production in response to nutrient loading. Macro- and micronutrient enrichment due to elevated loading drives phytoplankton proliferations in many New Zealand lakes. In turn, phytoplankton proliferations reduce dissolved oxygen concentrations, degrade habitat quality, alter energy flow through food webs and, in cases of planktonic cyanobacteria, produce algal toxins.

In extreme cases, phytoplankton blooms driven by elevated nutrient loading cause lakes to undergo regime shifts from native macrophyte-dominated, clear-water states to devegetated, turbid states. Numerous shallow lakes in New Zealand have undergone such shifts, and some appear to be in a persistent, degraded state. These shifts increased in occurrence as the proportion of catchment land cover in pasture increased and the proportion in native forest land cover decreased. The presence of pest fish and the invasive macrophyte *Egeria densa* were also positively associated with regime shifts, which suggests that interactions between eutrophication and the presence of invasive species may accelerate regime shifts in shallow lakes.

Our review indicated that both catchment land cover and loads related to land cover classes had negative effects on lake state variables (e.g., trophic state, nutrient concentrations, phytoplankton biomass). Several studies have reported that pastoral land cover is positive correlated with lake trophic state and nutrient concentrations, and negatively correlated with visual clarity and ecological health indicators. In the studies that have used measured and predicted N and P loading to predict lake responses, phytoplankton productivity, lake trophic level and hypolimnetic oxygen depletion increased, and ecological integrity indicators (e.g., zooplankton and macrophyte diversity) decreased in response to increasing nutrient loading. For some response variables, the pressure-response relationship varied widely between lake morphometric classes. This observation suggests that PSI relationships identified in one lake should not be considered widely transferable without testing across a range of lakes.

1.1.4 Aquifers

The major classes of land-use pressures that affect groundwater state and impact values in aquifers are contaminant input from the land surface, injection wells, river seepage and other sources, and hydrologic alterations caused by groundwater abstraction and reduced groundwater recharge. These land use pressures result from land use intensification (particularly increased nutrient input), increased rates of groundwater and surface water abstraction, and changes in vegetation, tillage, irrigation, river flows and other properties that influence rainfall and river recharge.

Successive analyses of data generated by the National Groundwater Monitoring Programme (NGMP) have been used to test for relationships between groundwater chemistry and overlying land cover and land-use patterns. Few detectable associations have come from these analyses, which is due in part to uncertainty about the capture zones of the NGMP wells. In another national-scale analysis that used dissolved reactive phosphorus (DRP) data from > 500 wells, there was a clearer pattern, with elevated groundwater DRP concentrations in the vicinity of dairy farms, and lower, similar DRP concentrations associated with six other land use classes.

Results from smaller scale studies of groundwater chemistry indicate that areas with long histories of intensive farming and high fertiliser use have a high incidence of nitrate contamination. Market garden areas in the Franklin District are an example, where groundwater nitrate concentrations as high as 36 mg N L^{-1} have been reported.

National surveys of pesticides in groundwater have been carried out at 4-year intervals since 1990. In those surveys, more pesticides have been detected in wells surrounded by urban and horticultural land uses compared with other agricultural land uses. Concentrations of some pesticides have decreased in recent groundwater surveys, which is consistent with reported reductions in pesticide use on horticultural crops and orchards following the introduction of integrated pest management schemes. However, residues of some pesticides persist long after their active use ceases, as indicated by detections of the insecticide Dieldrin in groundwater near former sheep-dip sites.

1.1.5 Estuaries and coastal zones

The major classes of land-use pressures that affect state variables and values in estuaries and coastal zones are: 1) physical alteration of habitats through land drainage, sedimentation, and construction; 2) contaminant input from stormwater, sewage effluent outfalls, rivers, road runoff, submarine groundwater discharge and longshore transport; and 3) changes in freshwater inflows due to catchment land use change. The potential effects of these pressures include changes in water quality and habitat quality and availability, reduced native biodiversity and fish stocks, and increased incidences of toxic and non-toxic algal blooms, contaminated fish and shellfish, and exceedance of microbiological standards for contact recreation.

State of the environment monitoring in estuaries and coastal zones in New Zealand is at an earlier stage of development than for freshwater monitoring, and the data needed to link land use pressures to environmental state and impacts is lacking for most locations. Median state and temporal trends in water quality and trophic state have been recently reported for those monitoring sites with sufficient data, but these data have not been associated with land cover or land use in the upstream catchments. However, a recent modelling study used national land cover and land use information to predict land-derived nutrient loads to 415 New Zealand estuaries. These loads were converted to concentrations by applying a dilution model for multiple estuary classes. The predicted concentrations are intended for use in evaluating risks of algal blooms and other responses to elevated nutrient loading.

Despite the scarcity of quantitative PSI relationships for New Zealand estuaries and coastal zones, the qualitative effects on land-derived contaminants on coastal fisheries and coastal biodiversity have been identified in numerous studies, including two case studies in this report (Appendices D and H). Among the most widely observed effects are degradation of habitat due to sedimentation and eutrophication, and reductions and losses of mussel, cockle and scallop beds, seagrass meadows, bryozoan and tubeworm mounds, sponge gardens, kelps beds, and biogenic habitats that these organisms provide.

1.2 Gap analysis

We used two approaches to assess the state of knowledge about land use effects, and the state of data quality and availability concerning those effects. The first approach was a simple matrix of receiving environments, land use pressure categories and response variables that the section authors of this report used for a subjective assessment. The second approach was an email survey of 59 New Zealand experts in land and water management and science. The experts were asked to evaluate the state of knowledge of mechanistic understanding of land use effects and the adequacy of land-use, land-cover and land management practice data, and to provide high-priority steps for filling knowledge and data gaps.

The two clearest patterns identified using the matrix approach were the low levels of information about the effects of land management practices and low levels of information about land-use effects on Māori values in all receiving environment classes. High levels of information availability appear to be limited to water quality in rivers, urban streams and lakes, and ecological values in rivers, as a function of land-cover patterns.

The expert survey participation rate was 58%. The respondents represented regional councils, crown research institutes (CRIs), universities, consultants and independent research organisations, primary sector organisations and central government.

There was a general consensus among respondents on several points concerning the state of understanding about the effects of land use:

- Our understanding of ecological responses far exceeds that for cultural and social responses.
- Our knowledge about N losses from land and effects on N enrichment on freshwater state and values exceeds that for other contaminants (e.g., P, sediment, faecal microbes, pesticides, metals, pharmaceuticals).
- Estuaries and aquifers are understudied relative to rivers and lakes.
- There is a severe lack of knowledge about the environmental effects of specific land management practices.
- Multiple stressor effects (both interactive effects and cumulative effects) in freshwater and coastal ecosystem are major knowledge gaps.

There was also a general consensus among respondents on several points concerning the adequacy and availability of the data needed to assess land use effect:

- Land-cover data, which are generally sourced from the LCDB, are adequate for most purposes, and are widely available. Limitations in land cover data included poor temporal resolution due to the long intervals between versions and coarse classification detail.
- Most respondents rated the current state of land-use data quality as moderate to poor. In particular, data related to horticulture and arable cropping were considered inadequate. Some types of data that are critically needed to link agricultural activities to freshwater are not widely available. Aggregating land use data across regions is difficult due to the lack of a nationally consistent classification.
- The consensus view concerning data that describe land-management practices (LMPs) is that the current data are inadequate and that improving the situation will require major investments of time and funding. The problems most frequently identified were

unreliability due to voluntary reporting, lack of systematic approaches for collecting and classifying LMP data, and severely limited availability of these data.

Regarding the improvements needed in land-cover, land-use and LMP data quality and availability, most respondents referred to the need for an updated version of LCDB, the need for standardisation in collecting and classifying land-use data, increased availability to users, and regular updating. Additional needs included agreeing on the specific variables that should be measured (i.e., identifying LMPs that affect freshwater and coastal environments), and ensuring that a central government department or a research institution is responsible for managing national land use and LMP data.

1.3 Recommendations

The authors of this report collectively provided 28 recommendations for improving knowledge of PSI relationships and filling related knowledge gaps. Some of the recommendations apply to all freshwater and coastal environments, and some are specific to rivers, urban streams, lakes, aquifers and estuaries and coastal zones. In addition to the recommendations that concern receiving environments, the participants of the expert survey summarised above provided multiple recommendations concerning the collection, processing and provision of land cover, land use and land management practice data.

For brevity, the recommendations that apply to all freshwater and coastal environments are summarised here.

- Expand the scope of the PSI model as a framework for organising information and prioritising management. The PSI model is appealing because it is simple, linear and logical, but it does not effectively account for the complex causal networks that characterise aquatic ecosystems, or for the need to rely on proxy variables for pressures and responses.
- Evaluate the effectiveness of industry best management practices and regional and national policies and standards for reducing adverse effects of land use on freshwater and marine environments.
- Improve the state of knowledge about mitigation systems and interventions designed to reduce the impacts of land use pressures on freshwater and coastal ecosystems.
- Advance the use of Māori indicators of freshwater and coastal conditions to develop associations between land use pressure and Māori values. Degradation of customary resources associated with aquatic environments, degradation of mauri (life force), and loss of cultural opportunities are issues of great concern for Māori.

1.4 Case studies

In addition to the main body of the report, we prepared eight case studies to illustrate a wide range of situations in New Zealand where current or historical land use has strongly affected ecological, social and economic values in freshwater and coastal environments. The case studies include streams in the ‘Best Practice Dairy Catchments’, a river in a semi-arid catchment where surface water is over-allocated, a lake and a coastal lagoon that have been degraded by external and internal nutrient loading, but may be on the cusp of recovery, a 12 year-long study of the effects of forestry in an erosion-prone catchment, and the Firth of Thames, where sediment and nutrient loading dating back to 19th Century land clearing has resulted in pervasive changes in ecosystem processes. Collectively, the case studies serve to support the recommendations set out in this report.

1 Introduction

Large-scale land use activities such as agriculture, forestry, mining and urban development generate pressures that can have deleterious effects on freshwater and coastal environments. Land-use pressures form three general classes: 1) discharge of land-derived contaminants (e.g., nutrients, sediment, pathogenic microbes, toxicants) to freshwater and coastal environments; 2) hydrological alterations due to water abstraction, impoundment and transfers; and 3) physical habitat alteration such as wetland draining. This review covers the effects of the first two classes of land-use pressures, contaminant discharge and hydrological alterations. These classes correspond to the limits on resource use required by the National Policy Statement for Freshwater Management (NPS-FM) 2014; limits on both contaminant discharges and water takes are required to ensure that freshwater objectives are met and to prevent over-allocation (New Zealand Government 2014).

All water bodies are receiving environments for land-derived contaminants, even under pristine conditions. Natural erosion, weathering, and biogeochemical processes generate and mobilise contaminants, which comprise natural loads (Snelder et al. 2017). Land use activities increase contaminant loads by increasing input to land (e.g., fertiliser and effluent application, animal stocking, urban populations and traffic volumes), increasing contaminant mobilisation (e.g., through tillage, clear-felling, irrigation) and reducing natural attenuation (e.g., through increasing impervious surfaces, disconnecting rivers from floodplains, draining wetlands).

Prior to the expansion and intensification of agriculture and plantation forestry, point sources such as sewage and industrial outfalls were major sources of land-derived contaminants to New Zealand's freshwater and coastal environments (e.g., Lewis et al. 1986, Rutherford et al. 1989, Vant 2001). Improved water treatment practices and a shift to discharging effluent on land and offshore has progressively reduced point-source inputs (e.g., Burns et al. 2009, Roygard et al. 2012). Contaminant loads are currently dominated by diffuse sources, i.e., multiple, dispersed source areas, from which contaminants are transported to receiving environments through soils, aquifers, rivers and tile drains (Elliott et al. 2005, Howard-Williams et al. 2010). Diffuse contaminant loads are generally more difficult to regulate than point-source loads. The difficulties are due in part to the complexities of contaminant mobilisation, transport, attenuation, and source apportionment in heterogeneous catchments.

Increasing contaminant loading to receiving environments lead to changes in state variables such as contaminant concentrations in water, sediment and biota, deposited sediment cover. The magnitude of state changes depend on contaminant loading rates and physical characteristics of the receiving environment (e.g., residence times, mixing regimes). Contaminant mass balances and dilution models have been used to predict state changes in response to contaminant loading in New Zealand lakes, rivers, wetlands, estuaries and coastal zones (Gibbs et al. 2002, Bayer et al. 2008, Miller and Kuehl 2010, Wilcock et al. 2012, Green 2013, Plew et al. 2018).

Changes in contaminant state cause subsequent changes in ecological, cultural, ecological and social values in freshwater and coastal environments. In the pressure-state-impact (PSI) framework (discussed in detail below), changes in ecological, cultural, economic and social values are referred to as 'impacts'. Elevated contaminant levels have been linked to a wide range of negative impacts in New Zealand and overseas, including loss of lake macrophyte beds, algal blooms, fish kills, habitat degradation due to sedimentation, and human health risks for contact recreation and food consumption (Schallenberg and Sorrell 2009, Cornelisen et al. 2011, Stewart et al. 2011, MacLeod et al. 2012). In extreme cases, contaminant loading results in severe and persistent degradation, as exemplified by anoxic ‘dead zones’ caused by phytoplankton blooms (Rabelais et al. 2014). Persistent dead zones have not been reported from New Zealand, but episodic anoxia has been observed in lakes and coastal areas where land-derived nutrient loading triggered phytoplankton blooms (Mitchell and Burns 1979, Jones and Rhodes 1994).

As is the case for land-use effects on contaminants, hydrological alterations due to land use are issues of global concern. Hydrological alterations refer to changes in hydrological state variables that include the frequency, magnitude and timing of fluctuations in river flows and lake and groundwater levels (Richter et al. 1996, Booker et al. 2014a). Direct hydrological effects of land use are caused by water abstraction, impoundment, artificial drainage and inter- and intra-basin transfers (Duncan and Woods 2013). Indirect land use effects include changes in vegetation and soil properties, which in turn affect evapotranspiration, infiltration and runoff (e.g., Cao et al. 2009, Fahey and Payne 2017). Hydrological alterations caused by land-use pressures have many potential impacts, including changes in aquatic and riparian populations, recreational values, cultural values, and water supplies for out-of-stream use (Poff et al. 2010, Tipa and Nelson 2012, Woodward et al. 2016).

Hydrological effects of land use are scale dependent and cumulative, which can make attributing state changes and impacts to specific land-use activities difficult (Blöschl et al. 2007). Assessing the effects of water abstraction on river flow fluctuations immediately downstream of the abstraction point is generally straight-forward (Duncan and Woods 2013, Lessard et al. 2013), whereas assessing the individual and cumulative effects of multiple hydrological alterations at multiple points in a river network is a major modelling challenge (Ryo et al. 2015, Hoyle et al. 2016). Similarly, catchment-scale effects of single, widespread changes in land cover (e.g., clear-felling, afforestation) can be determined accurately and often measured directly (Fahey and Payne 2017), whereas changes in land cover mosaics in large, heterogeneous catchments have more complex effects, which cannot be measured directly, and for which model uncertainty is typically high (Cao et al. 2009).

There is a widespread consensus that land-use pressures generally have negative impacts in freshwater and coastal environments (Howard-Williams et al. 2010, Poff and Zimmerman 2010). However, targeted management to reduce or prevent negative effects of land use requires more specific information about the effects of different types of pressures on different values in different types of water bodies. In particular, quantitative relationships linking pressure, state and impact variables are needed. These ‘PSI relationships’ are fundamental tools for land and water management in New Zealand and overseas; they are used to identify land-management practices that cause environmental degradation or improvement, to forecast the effects of future changes in land use, to develop limits on resource use, and to evaluate the effectiveness of plans, policies and management actions (Lamon and Qian 2008, Poff et al. 2010, Davies et al. 2014).

Scope and methods

The primary purpose of this report was to compile and summarise PSI relationships that link land use to impacts in New Zealand’s freshwater and coastal environments. Separate summaries were prepared for the major water-body classes in New Zealand that are strongly affected by land-use pressures: rivers, inland lakes, coastal lakes and lagoons, urban streams, aquifers, estuaries and coastal zones. Continental shelf and deep ocean areas were excluded due to lack of information. To facilitate independent evaluations, the summaries were based on publicly available, peer-reviewed publications and technical reports. A large number of publications and reports were compiled and screened for use in the review. To compile this literature, we used Google and Google Scholar searches and literature collections held by the authors and their institutions. A wide range of search terms were used to maximise the number of publication and reports for assessment. We retained publications and reports that contained quantitative or qualitative PSI relationships, ancillary relationships (e.g., contaminant loss rates from land in contrasting land use classes), and the background information reported in Section 2. We focused on primary reports and excluded reviews, except where reviews were used to introduce or explain topic areas. For most PSI relationships, we used publications and reports from New Zealand studies. However, there are severe information gaps about land-use effects in some New Zealand water-body classes (e.g., estuaries and coastal zones), and publications from overseas studies were used in these cases.

For many combinations of land use, water body type and state or impact variable, quantitative PSI relationships are lacking in the peer-reviewed literature. In these cases, we used published qualitative relationships as available. We identified information gaps with a matrix of major land-use and water body classes. We used a survey of New Zealand environmental scientists with knowledge of land-use effects to identify gaps in data and in knowledge of land-use effects.

We investigated the use of the 'Eco-Evidence system' for making inferences about land-use effects on aquatic environments (Norris et al. 2011). The only water body class with sufficient published papers from New Zealand studies to carry out an Eco-Evidence analysis was rivers, using land cover as a land-use pressure. However, the reports of those papers were extremely consistent in terms of direction of land cover-response relationships (i.e., contaminant levels increased, water clarity decreased and invertebrate-base ecological health scores decreased with increasing agricultural and urban land cover). For other explanatory variables related to land use (e.g., stocking rate, impervious cover, stock exclusion) there were too few papers for analyses. Based on these observations, we concluded that Eco-Evidence analyses of papers about land cover was redundant and would not add value to the narrative synthesis.

To add detail to the review of PSI relationships, we include eight case studies that summarise detailed evaluations of land use effects in a wide range of receiving environments.

2 Background

2.1 State and trends in freshwater and coastal environments

National and regional reporting on New Zealand's freshwater and coastal environments consist of two general types of assessments: current state and recent trends in water quality and ecological conditions (e.g., trophic state), and inventories of anthropogenic pressures that may affect those states and trends, including water takes (Office of the Auditor-General 2011, Ministry for the Environment 2014). The raw data for these reports come from regional council state-of-environment (SoE) monitoring programmes, consent databases, national monitoring programmes such as NIWA's National River Water Quality Network (NRWQN)[1] and the GNS Science National Groundwater Monitoring Network (NGMN)[2] , the Agribase spatial farms database[3] and agricultural production surveys and census tables held by Statistics New Zealand[4].

Estimates of water quality and ecological state are generally based on mean or median values for each of a suite of measurement variables. Recent national-scale assessment of the state of estuaries, rivers, lakes, urban streams and aquifers were reported in 2015 and 2016 (Dudley et al. 2017 (coastal waters and estuaries), Larned et al. 2015, 2016 (rivers and lakes), Gadd 2016 (urban streams), Moreau and Daughney 2015). In these reports, current state for each variable at each monitoring site was estimated from 3-10 years of monthly, quarterly or annual measurements, with ending dates between 2013 and 2015. To provide information about variation in water quality and ecological state across New Zealand's heterogeneous environment, site medians are generally grouped into environmental classes, and the class medians are reported. For estuaries, site medians were grouped into hydrogeomorphic classes based on the Estuary Trophic Index, hereafter "ETI classes" (Dudley et al. 2017, Robertson et al. 2016a, b). For rivers, site medians were grouped into River Environment Classification (REC) classes (Snelder and Biggs 2002, Larned et al. 2015, 2016). For lakes, site medians were grouped into elevation and depth classes (Larned et al. 2015). For urban streams, site medians were grouped into classes defined by proportions of urban land cover (Gadd 2016). For aquifers, sites medians were grouped into lithology classes (Moreau and Daughney 2015). As an alternative to grouping monitoring sites into environmental classes, they may be grouped into jurisdictional classes such as regions and cities (e.g., Ballantine et al. 2010). Assessments of mean or median water quality state within regions and cities are needed for some reporting purposes, but this approach does not account for environmental heterogeneity within jurisdictional boundaries.

The assessments of water quality and ecological state summarised above are based on long-term monitoring sites, which comprise a very small proportion of the water bodies in New Zealand. To expand the scope of national reporting to unmonitored sites, statistical spatial models have been developed for predicting water quality and ecological state in river reaches and large lakes across New Zealand, and river hydrological state (Unwin et al. 2010, Clapcott et al. 2012, Booker 2013, Booker and Woods 2014, Snelder et al. 2014, Larned et al. 2016). Statistical models have also been used to estimate reference water quality and ecological state in New Zealand rivers, where reference conditions represent the predicted state in the absence of land-use pressure (McDowell et al. 2013, Clapcott et al. 2017, Snelder et al. 2017, Schallenberg in press,).

[1] www.niwa.co.nz/freshwater/water-quality-monitoring-and-advice/national-river-water-quality-network-nrwqn
[2] www.gns.cri.nz/Home/Our-Science/Environment-and-Materials/Groundwater/Research-Programmes/National-Groundwater-Monitoring-Programme-NGMP
[3] www.asurequality.com/our-solutions/agribase
[4] http://datainfoplus.stats.govt.nz/Item/nz.govt.stats/6362a469-f374-412e-ac25-d76fd0962003

Trend analyses are used in environmental reporting to provide information on the direction and rate of systematic temporal changes in water quality and ecological conditions. Recent national-scale analyses of trends in water quality and ecological conditions were reported in 2015 and 2016, for coastal waters and estuaries (Dudley et al. 2017), rivers and lakes (Larned et al. 2015, 2016), and urban streams (Gadd 2016).

Statistical models that predict trends at unmonitored sites are at an early stage of development. In one of the first such analyses, the directions of temporal trends in the faecal indicator bacterium *E. coli* were predicted in 49,000 river reaches in the Manuwatu-Whanganui Region (Snelder 2018). Similar spatial models of trends in other water quality and ecological variables may be standard components of state and trends analyses in the future.

2.2 Pressure-state-impact relationships

This review uses the pressure-state-impact (PSI) framework to organise information about land use effects on freshwater and coastal environments. The PSI framework, and updated versions such as the driver-pressure-state-impact-response framework serve to organise and convey information about environmental impacts and their putative causes, causal chains, and management responses (European Environmental Agency 1999, Mueller et al. 2015, Scarsbrook and Melland 2015). The Ministry for the Environment and Statistics New Zealand have adopted the PSI framework for environmental reporting as it provides more information about human activities and environmental effects than descriptions of environmental state and trends alone.[5]

In this report, we follow the terminology set out in Oesterwind et al. (2016). Pressures are the proximal causal agents or activities that result in changes in environmental state. Pressures typically refer to human activities such as fishing, fertiliser use, tillage and water abstraction, but may also refer to natural processes such as natural soil erosion. As noted in the next section, pressures in PSI relationships may also be represented by proxy spatial variables such as land cover (discussed in detail in Section 2.3). Pressures are generated by drivers, which are socio-economic phenomena such as water and nutrient demand for producing food and fibre. Therefore, drivers are the ultimate causal agents of changes in state. Pressures are often manageable at local scales, while drivers are generally not. State refers to the prevailing environmental conditions within a given area over a given time period. Impacts are the consequences of state changes in terms of environmental, cultural, social and economic values, e.g., a state change from very low concentrations of a toxicant to high levels can have negative impacts on potable water supplies, recreation, customary harvest, and populations of sensitive species exposed to the toxicant. The PSI model implies a one-directional sequence, where pressures affect state variables, which in turn affect impact variables. In practice, state and impact variables are often used interchangeably; both are used as response variables in assessments of land use effects, and cascades of impacts are common (e.g., nutrient enrichment causes phytoplankton blooms, which decompose, leading to hypoxia or anoxia, which causes fish mortality; Mallin et al. 2006).

In the New Zealand literature about land-use effects on freshwater and coastal environments, most quantitative relationships corresponding to the PSI framework are bivariate pressure-state and pressure-impact relationships. In addition, some linked pressure-state-impact relationships have been reported. In these reports, the causal chain consisting of pressures, states and ecological impacts were elucidated using structural equation methods. For example, structural equation

[5] www.mfe.govt.nz/more/environmental-reporting/reporting-framework

modelling indicated that riparian conditions (pressures) along agricultural streams in Canterbury affected water temperature and benthic organic matter (states), which in turn affected the composition of stream invertebrate communities (impacts) (Greenwood et al. 2012).

2.3 Land cover, land use and land management practices

This review concerns the effects of land use on freshwater and coastal environments. However, the term land use is rarely defined explicitly, it is used in reference to a wide range of activities and spatial properties, and is often used interchangeably with the terms land cover and land management practice (or land management activity). In this review, we consider land use (LU), land cover (LC) and land management practices (LMPs) to be three distinct groups of variables. The definitions used here follow those of Verburg et al. (2011) and ACLUMP (2010): LU describes the purpose for which a parcel of land is used (e.g., deer farming, arable cropping, urban residential); LC describes the observable features on the land surface (e.g., broadleaf forest, exotic grassland, buildings, bare land); and LMPs describe the activities and inputs and outputs that are employed to achieve a given land use (e.g., fertiliser application, forest clear-felling, soil tillage, urban wastewater treatment).

LU and LC are closely related variables; both concern the coverage and configuration of land areas, and analyses of both are based on classification systems that assign cover or use classes to gridcells or polygons on maps. The LU and LC classification systems widely used in New Zealand are not mutually exclusive, and some LC and LU classes are synonymous (e.g., vineyards, orchards). In other cases, broad LC classes encompass multiple LU classes. For example, the high-producing grassland LC class encompasses land used for dairy and deer farming, dairy support, sheep and beef grazing, silage, grass seed and hay (McDowell and Wilcock 2008, Graham et al. 2016). In this review, we differentiate between LC and LU where possible, as they differ in information content and in some data sources as discussed below. In contrast to LU and LC, there is no widely recognised classification system for LMPs in New Zealand. However, LMP classifications have been developed overseas (e.g., ABARES 2016), and could be developed for New Zealand.

The differences between LC, LU and LMPs set out above lead to several differences in their utility as pressure variables in PSI relationships. One of the primary aims of these relationships is to identify activities on land that are likely to lead to degradation (or improvement) in receiving environments. In virtually all cases, PSI relationships are correlative. Therefore, mechanistic relationships between pressure variables and state or impact variables cannot be inferred with certainty. However, LC, LU and LMPs represent a gradient of increasingly direct relationships between human activities and environmental outcomes. (Julian et al. 2017). While LC classes generally have low explanatory power and are clearly proxies for land-use pressures, LC data are publicly available and there is complete coverage of New Zealand. In contrast, LMPs may have higher explanatory power, but LMP data are not widely available.

2.3.1 Land cover data and classification

The primary source of national-scale LC data in New Zealand is the Land Cover Database (LCDB)[6]. Land cover Data for the LCDB are derived from satellite images, converted to spectral reflectance mosaics, and polygons in the mosaics are assigned to LC classes using spectral rules (Dymond et al. 2012). Imagery from the summers of 1996/97, 2001/02, 2008/09, and 2012/13 have been used to generate LCDB Versions 1, 2, 3 and 4, respectively. The number of LC classes in these versions ranged

[6] https://lris.scinfo.org.nz

from 16 to 43; the most current version, Version 4.1 (released February 2017) has 33 mainland classes[7].

Sequential LCDB versions have been used to analyse national-scale LC change (Weeks et al. 2013, Cieraad et al. 2015, Dymond et al. 2017). These analyses have focused on land in natural LC classes, such as indigenous forest, tussock and scrub. National-scale analyses of changes in agricultural, exotic forest and urban LC have not been published, but tables of proportional changes between classes are available on the LRIS portal[8].

2.3.2 Land use data and classification

The primary data sources for national-scale LU classification are the LCDB, Agribase, the Land Environments of New Zealand database, the Agricultural Production Survey, and the Ministry for Primary Industries FarmsOnline database, plus cadastral data from Land Information New Zealand, consents data from regional councils, and ancillary datasets used for validation (Rutledge et al. 2009, Ledgard 2013, Anastasiadis et al. 2014, Semadeni and Elliott 2016). The most up-to-date, national LU maps are for the year 2012. One 2012 map was developed by Ministry for the Environment as part of the Land Use and Carbon Analysis System (LUCAS), for reporting on carbon fluxes and greenhouse gas emissions (Ministry for the Environment 2012, Wakelin et al. 2016). Another 2012 map was developed for running the CLUES catchment model to assess contaminant loading in streams (Daigneault and Elliott 2017). The classification systems underpinning the CLUES LU map are tiered, with broad LU classes (e.g., pastoral, arable, horticultural) subdivided into progressively more detailed subclasses, including geographic areas (e.g., sheep and beef-intensive-Northland, pipfruit-Hawke's Bay).

2.3.3 Land management practise data and classification

As noted above, LMPs comprise a very wide range of practices that include activities, inputs, and outputs, and there are no standardised categories or classification for New Zealand. However, classifications have been developed overseas for agricultural and urban LMPs (e.g., ABARES 2016). There are some impediments to compiling and mapping LMP data in New Zealand, including a lack of standard procedures for characterising LMPs and commercial protection. Despite the impediments, several national and regional-scale LMP inventories have been developed for analyses of land use intensity in New Zealand and overseas (MacLeod and Moller 2006, Erb et al. 2017). The LMPs used in these inventories were animal stocking rates, grazing pressure (using standardised stock units), inputs (synthetic fertilisers, feed supplements, irrigation), and outputs (areal production and production per animal). The data were national and regional totals, and were not geo-referenced.

Despite the shortage of data, the impacts of LMPs in freshwater environments have been assessed in several studies in New Zealand (e.g., Hamill and McBride 2003, Monaghan et al. 2007, Wilcock et al. 2013a, b, Julian et al. 2017). Most of these studies concerned single, small catchments, and the LMP data were sourced from censuses and surveys. In regional and national-scale studies, LMP data are generally modelled (e.g., Julian et al. 2017).

[7] www.lris.scinfo.org.nz/layer/423-lcdb-v41-land-cover-database-version-41-mainland-new-zealand
[8] www.lris.scinfo.org.nz/document/9472-lcdb-v41a-cover-class-change-summary-2008-2012

3 Pressure-state-impact relationships in New Zealand

3.1 Rivers

3.1.1 Water quality and ecological conditions

A. Introduction

This section summarises the state of knowledge about land-use pressures that lead to changes in river state variables, and in turn, have impacts on the ecological, Māori and social values of rivers. We first review associations between agricultural and forestry land cover classes and river state and ecological impacts, then associations between specific agricultural and forestry practices and state and impacts. We note that much of the literature on agricultural land use focuses on contaminant losses without linking those losses directly to changes in river state or impacts; this imbalance in the literature is reflected in our review of agricultural pressures. We finish the section with a discussion of land use effects on river flow regimes. Land use effects on urban streams are reviewed in a separate section (Section 3.2). The separation of urban and non-urban land use effects on streams and rivers reflects the recent approach to national reporting, in which national-scale river state and trends, and urban stream state and trends in selected catchments are reported separately (Larned et al. 2015, Gadd 2016).

We use the pressure-state-impact (PSI) framework described in Section 2.2 to organise information about land use effects on rivers. In the New Zealand scientific literature, land-use pressures that affect river state and impact river values fall into five broad classes: 1) contaminant loss from land and conveyance to rivers; 2) the presence of livestock in river environments (primarily cattle and deer); 3) alterations to natural riparian vegetation on river boundaries; 4) abstraction of water from river channels and from aquifers connected to rivers; and 5) alterations in natural hydrological processes in addition direct abstraction (Figure 3-1). As discussed in Section 2.3, land cover is frequently used as a proxy for land-use pressure in studies of land use effects. In those cases, the specific pressure(s) that causes state changes or impacts cannot be identified.

Although the PSI framework is useful for organising information, and its focus on bivariate relationships reflects the approach used in many studies of land use effects on New Zealand rivers, it does have some limitations. One limitation concerns multiple stressor situations, in which multiple land-use pressures affect a given state variable, or multiple state changes impact a given value (as indicated by the dashed lines in Figure 3-1). There is a general consensus among freshwater scientists that river values are frequently affected by multiple stressors (Townsend et al. 2008), but the research focused on multiple stressor situations lags behind the research focused on bivariate pressure-response relationships. A second limitation concerns the large body of research in New Zealand focused on understanding land-use pressures without explicitly linking those pressures to effects in receiving environments. As noted above, a large proportion of New Zealand research on contaminant losses from land fits into this category. Contaminant leaching and runoff from agricultural source areas is relatively well-studied, but the effects of these losses on distant receiving environments are poorly studied. Part of the problem lies with the complex processes that occur between contaminant sources and receiving environments (e.g., transport, biogeochemical transformation, attenuation). As a consequence, some of the agricultural land-use pressures reviewed in this section are not linked to effects in rivers in the form of quantitative PSI relationships.

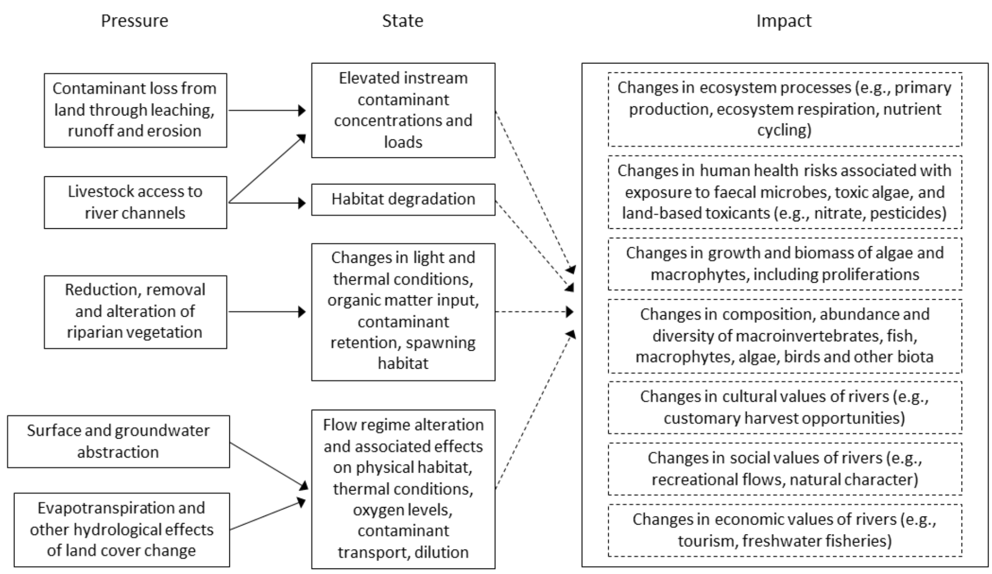

Figure 3-1: General classes of land-use pressures, river state variables and impacts on river environments. Solid arrows indicate the most frequently assessed pressure-state relationships. Dashed arrows indicate that most impacts result from the combined effects of multiple state changes.

B. Associations between land cover classes and river state variables and values

Prior to human settlement, over 80% of the land surface of New Zealand was densely forested. The proportion of native forest cover currently stands at approximately 20%; most of this reduction occurred after 1870, through the conversion of native forest and scrub to farmland, planted (exotic) forests and urban centres (Walker et al. 2006). Changes in land cover have been associated with four general types of state in river ecosystems: instream habitat change, changes in riparian functions, increased input of land-derived contaminants, and changes in flow regimes (Figure 3-1).

Associations between land cover and conditions in rivers are scale-dependent, and the New Zealand research literature reflects this dependence. In most of the papers and reports that link land cover to river conditions, effects of land cover are either analysed at whole-catchment scales or at riparian-zone scales, and the explanatory power of the two scales have been compared in several New Zealand studies. Catchment-scale land cover generally comprises proportions of catchment area upstream of monitoring sites in each of several land-cover classes. Riparian zone land cover data generally comprises the same cover classes summarised above, plus vegetated width and in cases of restored riparian vegetation, plant composition and age (e.g., Parkyn 2004).

B. 1. Catchment land cover

In most New Zealand studies that use catchment-scale land cover, river sites are designated as agricultural (i.e., pastoral), planted forest, urban or natural based on the single dominant land cover class in the upstream catchment. The natural land-cover class is often subdivided into native forest, scrub, tussock and other subclasses dominate by a different indigenous plant taxa. In studies carried out after 2000, the land cover classes correspond to LCDB or and REC land cover classes (e.g., McLay et al. 2001, Snelder and Biggs 2002). Dominance may refer to proportional area alone, or to the predicted influence of various land uses on river conditions. As an example of the latter, the assignment of river sites to REC land-cover classes are based on the following rules: a site is classified as planted forest or natural if those categories account for the largest proportion of the upstream catchment area, unless pastoral land exceeds 25% of the catchment, in which case the segment is classified as pastoral, or urban land exceeds 15% of the catchment, in which case the segment is classified as urban (Snelder and Biggs 2002). The weighted proportions of pastoral and urban land cover are based on the presumption that the areal effects of pastoral and urban land use on rivers are greater than for natural and planted forest land use. Associations between catchment land cover and river water quality and ecological condition have been evaluated using sites distributed across New Zealand, sites distributed across individual regions, and sites located in multiple subcatchments within larger catchments.

National-scale associations between proportions of catchment land cover and river water quality state have been reported in multiple studies, using sampling data extending from 1987 to 2014. In one of the earliest national-scale studies, the "100 Rivers Project", Close and Davies-Colley (1990) reported that proportions of natural, planted forest, urban and pastoral land cover explained variation in concentrations of nitrogen (N), phosphorus (P) and major ions across 101 sites sampled in 1987. For most of the solutes, increasing concentrations were positively correlated with planted forest, urban and pastoral land cover, and negatively correlated with natural land cover. In the same study, Quinn and Hickey (1990) assessed macroinvertebrate communities in 51 of the 101 river sites; the 51 sites represented a gradient of pastoral land cover from < 1 to > 30%. Median macroinvertebrate taxon richness, Shannon diversity, predator biomass and the density of pollution-sensitive Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies) (hereafter referred to as EPT) were all higher at sites with < 30% pastoral landcover that at sites with > 30% pastoral land cover.

Two of the recent national-scale studies were based on 77 sites in NIWA's National River Water Quality Monitoring Network (NRWQN) (Ballantine and Davies-Colley 2014, Julian et al. 2017), and the third used several hundred regional council SoE monitoring sites (Larned et al. 2016). Response variables in the study by Larned et al. (2016) also included the faecal indicator bacterium *Escherishia coli* (*E. coli*), and the macroinvertebrate community index (MCI) as a measure of ecological state (Stark and Maxted 2007). All three studies had comparable results: median concentrations of dissolved and total N and P increased and visual clarity decreased with increasing proportions of pastoral and urban land cover. Larned et al. (2016) reported that median *E. coli* concentrations increased and MCI scores decreased with increasing proportions of pastoral and urban land cover.

The more recent of the two NRWQN studies used 26 year-long water quality time-series (1989-2014) to identify associations between planted forest area chemical and physical water quality (Julian et al. 2017). Planted forest area was positively correlated with total N, nitrate-N, total P and dissolved reactive P (DRP) concentrations and negatively correlated with visual clarity. The authors suggested that historic fertiliser inputs and legacy nutrients from previous agricultural land use may have contributed to the correlations with planted forest area. They also reported negative correlation between the percentage land that had been disturbed by recent forest harvesting and visual clarity.

In addition to continuous variation in land cover, categorical differences in water quality state among REC land cover classes have been reported in several national-scale assessments. In the most recent report (Larned et al. 2016), median concentrations of nitrate-N, ammoniacal-N, total N, total P and *E. coli* increased and MCI scores decreased across land-cover classes in the following order: natural, planted forest, pastoral, urban. Visual clarity was higher in the natural and planted forest classes relative to pastoral and urban classes. These patterns are consistent with land-cover class comparisons using subsets of the same variables in earlier, national-scale reports (Larned et al. 2003, Ballantine et al. 2010).

Regional-scale associations between catchment land cover and water quality and ecological conditions have been identified in several regions, most commonly for assessing ecological health indicators such as macroinvertebrate community metrics. For brevity, we summarise land cover-macroinvertebrate associations in the Waikato Region, where Collier and colleagues have carried out investigations using a network of monitoring sites that span a wide range of instream and catchment conditions (Collier and Olsen 2013). Across Waikato hard-bottom streams, multiple health metrics based on macroinvertebrate community structure (taxon richness, EPT taxon richness, the proportion of EPT individuals (%EPT), MCI scores) were positively correlated with the proportion of natural land cover, and negatively correlated with the proportion of pastoral land cover (Collier 2008, Death and Collier 2010). Distinct thresholds were evident in plots of macroinvertebrate community metrics and natural (native forest and scrub) land cover (Collier and Hamer 2010). These thresholds indicated rapid declines in % EPT and MCI scores as the proportion of natural land cover decreased from 100 to 80%. Another rapid decline occurred between 60 and 0% natural landcover. These non-linear relationships between land cover and ecological health indicators suggest that there is intrinsic resistance in macroinvertebrate communities to minor losses native land cover loss. However, below a threshold (approximately 60% in the Waikato study), ecological health may decline substantially with small additional losses of native land cover.

Catchment-scale associations between land cover and water quality and ecological conditions and contaminant yields have been intensively studied in small catchments (catchment area 0.4 – 2.0 km^2) in and near the Whatawhata Research Centre, Hakarimata Ranges on the west side of the Waipa River, Waikato (e.g., Quinn et al. 1997, Parkyn et al. 2002, Quinn and Stroud 2004). Across the streams draining these catchments, median nitrate-N, ammoniacal-N, and total N concentrations were higher and DRP concentrations and visual clarity lower in pasture- and planted forest-dominated subcatchments compared with native forest-dominated catchments. Similarly, annual yields (in kg ha^{-1} yr^{-1}) of nitrate-N, ammoniacal-N, total N, total P and suspended sediment were higher from pasture- and planted forest-dominated catchments compared with a native forest-dominated catchment (Quinn and Stroud 2004).

Detailed surveys of stream habitat, water quality and macroinvertebrates were carried out in 1992-1993 at 16 Whatawhata sites in pasture-, planted forest- and native forest-dominated catchments (Quinn et al. 1997). Results of the surveys indicated that native forest streams were wider, more heavily shaded and had slower water velocities and less deposited fine sediment than pasture and planted forest streams. Dissolved inorganic N, suspended sediment and turbidity were higher in pasture and planted forest streams than native forest streams, and periphyton biomass was higher in pasture streams that either planted or native forest streams. Three macroinvertebrate metrics distinguished the land cover classes: total macroinvertebrate density was higher and EPT density and QMCI scores (quantitative variant of MCI scores) were lower in pasture streams that either planted or native forest streams.

A subset of the sites used by Quinn et al. (1997) were subsequently used to assess associations between land cover and the abundance, biomass, growth and annual production of the endemic crayfish *Paranephrops planifrons* (Parkyn et al. 2002). In this study, streams in pasture- and native forest-dominated catchments were compared. Crayfish densities were higher in native forest streams, but juvenile growth rates and peak annual biomass were higher in pasture streams. The difference in annual crayfish production between land cover classes was not significantly different, partly due to the trade-off between density and growth rate.

The Motueka River catchment has also been used to identify associations between land cover and water quality (Young et al. 2005), ecological conditions (Shearer and Young 2011) and cultural health (Harmsworth et al. 2011). The water quality study focused on 10 tributaries of the Motueka River with catchment landcover dominated by pasture, horticulture, planted forest or native forest (Young et al. 2005). The results indicated that nitrate-N, total N and *Campylobacte*r concentrations were higher at the pasture and horticulture sites than the native forest and planted forest sites, and nitrate-N, total N and *Campylobacte*r, *E. coli* and suspended sediment concentrations were higher at the pasture and horticulture sites than the native forest sites. No differences among land cover classes were detected for DRP, total P and turbidity. In the ecological study (Shearer and Young 2011), macroinvertebrate communities in 18 tributaries of the Motueka River were used to compare ecological conditions in streams representing combinations of dominant land-cover (pasture, planted forest or native forest) and geology (Separation Point granite, Moutere gravel). Results of the study indicated that MCI scores and the proportion of macroinvertebrates that shred and consume leaf litter were higher at native forest sites than at pasture or planted forest sites.

A study of the Motueka and Riwaka rivers by Harmsworth et al. (2011) is noteworthy as it is one of the very few to link land use with cultural values in rivers. The principal aims of this study were to review the growing body of work on cultural techniques for monitoring Maori values in rivers, and to compare the results of cultural monitoring results with results from scientific monitoring by Tasman District Council. For the purposes of the current review of land use effects, an important result of the Harmsworth et al. study was the strong correlation between cultural stream health measure (CSHM) scores for monitoring sites and native forest land cover in the catchments upstream from the sites. The CSHM is a metric for evaluating river health based on multiple cultural indicators (e.g., clarity, suitability for swimming and cultural harvests). The CSHM is one of three components of the Cultural Health Index, which combines the CSHM with the significance of the site to Māori and its mahinga kai values to generate a holistic assessment (Tipa and Teirney 2006).

The national- and regional-scale assessments summarised above did not include associations between land cover and fish abundance or composition because fish are not routinely monitored in regional council programmes or the NRWQN. However, national and regional assessments of fish occurrence have been carried out using records in the New Zealand Freshwater Fish Database (NZFFD). Joy (2009) summarised the fish data for 22,546 river sites in NZFFD site using an index of biotic integrity (IBI) developed for New Zealand native freshwater fish. The fish IBI is a community-level metric that incorporates the numbers of fish species at a site that are native, riffle-dwelling, benthic, and intolerant to water quality degradation and migration barriers (Joy and Death 2004). The sites were grouped into seven REC land cover classes (native forest, scrub, pastoral, urban, planted forest, bare, tussock) for comparison. Mean fish IBI scores were highest in the native forest and scrub classes, intermediate in the pastoral, urban, bare and planted forests classes, and lowest in the tussock class.

The links between land cover, habitat characteristics and fish composition was also assessed in the Hakarimata Range, Waikato River catchment (Hanchet 1990). In this study, fish were surveyed in 55 stream sites in catchments dominated by pastoral, planted forest, or native forest land cover, or by a mix of pastoral and native land cover. Fish diversity was highest at native forest sites and mixed pastoral-native forest sites and lowest at pastoral and planted forest sites. The habitat factors associated with these patterns included instream woody debris and substrate grain size (highest at native forest sites) and water temperatures (lowest at native forest sites). A second, smaller-scaled study (11 sites) in the same area provided additional information about associations between land cover and fish (Hicks and McCaughan 1997). In this study, fish densities and biomass were an order of magnitude higher at pastoral sites than native or planted forest sties, primarily due to high shortfin eel abundance and sizes at pastoral sites. Estimated eel productivity was about 7 times greater at pastoral sites compared with forested sites, presumably due to greater algal productivity at unshaded pastoral sites, and consequently, greater production of invertebrate prey for eels.

In a study of brown trout and upland bullies at 36 sites in the Manuherikia catchment, Lange et al. (2014) reported a unimodal relationship between pastoral land cover and trout presence: the probability of trout presence increased as pastoral land cover increased from 0 – 20% cover, then decreased to absence at 40% cover. There were no detectable relationships between pastoral land cover and the presence or abundance of bullies.

Several catchment-scale studies in New Zealand have focused on planted forest cover and its associations with river water quality and ecological state. Aristi et al. (2017) reported a positive correlation between the coverage of planted forests in the Maitai River catchment in Nelson and the coverage of *Phormidium*-dominated mats. *Phormidium* is a benthic cyanobacterium (blue-green alga), that can produce toxins. Nutrient concentrations were low in this catchment, and fine sediment inputs from planted forests, were hypothesised as a potential contributor to these proliferations. Parkyn et al. (2006) used 10 subcatchments of the Waipaoa and Mangaoporo Rivers in the Gisborne District to make categorical comparisons of stream water quality and ecological health associated with mature planted forest, grazed pasture and native forest. The study area is in highly erosion-prone soft-rock hill country and pine afforestation was used for erosion control. Median nitrogen and *E. coli* concentrations in the planted forest streams were comparable to those in the native forest streams and lower than in the pasture streams. Macroinvertebrate-based ecological health metrics in the planted forest streams were comparable to those in the native forest streams and higher than in the pasture streams. Across the 10 streams, suspended sediment, visual clarity and turbidity varied to a greater degree with geology and historical land use than with current land use; forest clearing in the 1920s has created a legacy of gully erosion and turbid streams in soft-rock terrain that afforestation has not eliminated.

B. 2. Riparian land cover

The effects of riparian land cover on river conditions may differ from the effects of catchment-wide land cover for two general reasons. First, some catchment-scale processes increase in intensity with increasing proximity to river channels (e.g., denitrification in shallow groundwater and other contaminant attenuation processes, particulate organic matter production) (Howard-Williams and Downes 1984). Second, some terrestrial processes that affect river conditions are limited to riparian zones (e.g., bank stabilisation, shading, lateral hyporheic exchange). The comparative effects of catchment-wide and riparian land cover on rivers are also related to differences in types of land cover in both zones. In some New Zealand catchments, grazing, cultivation or silviculture extends to river margins, and there is no distinction between riparian land cover and land cover in the rest of the catchment (e.g., Renouf and Harding 2015). In other catchments, riparian zones and adjacent land have contrasting land cover and are used for different purposes. Although riparian zones comprise a small proportion of most catchments, the land cover within riparian zones can be more diverse than the surrounding catchment, particularly when the remainder of the catchment is dominated by a single land use (Arbuckle et al. 1999, Harding et al. 2006).

The modern-day land cover in the riparian zones of agricultural and planted forest land result from several alternative types of land management: remnant native vegetation (characterised by mature, native vegetation that may be protected by covenant), retired (untilled, livestock excluded, often dominated by grasses and self-seeded shrubs and trees), planted (characterised by immature or mature native and ornamental trees and shrubs), and utilized (grazed, cultivated or afforested to the margin) (Parkyn et al. 2003, Harding et al. 2006, Renouf and Harding 2015, Maseyk et al. 2018).

In several New Zealand studies, a nested design was used to compare the strength of associations between land cover and water quality and ecological health at whole catchment versus riparian zone scales. In these studies, a riparian corridor was defined extending upstream of the sampling site, and proportions of different land cover classes were determined in both the upstream catchment and the upstream riparian corridor. Buck et al. (2004) compared correlations between water quality variable levels and proportions of pastoral, tussock and scrub landcover at catchment scales and within 120-m wide riparian corridors. Catchment scale landcover was strongly correlated with levels of more water quality variables than riparian-scale land cover, suggesting that extraneous local effects such as livestock access and bank erosion mask effects of riparian land cover. Death and Collier (2010) compared correlations between 22 different macroinvertebrate community metrics and proportions of natural landcover at catchment scales and within riparian zones of unspecified width. The correlations between each macroinvertebrate metric and landcover was stronger at the catchment scale compared to the riparian scale.

In both Buck et al. (2004) and Death and Collier (2010), the effects of environmental factors that are controlled by catchment-scale processes (e.g., flow regimes, water chemistry and temperature) appear to outweigh the effects of factors that are controlled by localised processes (e.g., light levels, channel habitat units). Burrell et al. (2014) used 21 streams in the Canterbury region to represent gradients of intensive agricultural and forested (native and planted) land cover, and compared the independent effects of catchment-scale landcover and riparian land-cover (20-m wide corridors) on stream metabolism. Their results indicated that the stream metabolism was more strongly associated with agricultural land cover at catchment scales and more strongly associated with forest land-cover at riparian-zone scales. In lieu of a nested design, Harding et al. (2006) used longitudinal changes in riparian vegetation to assess the effects of three riparian land-cover classes: pastoral, native fragments bounded by pastoral land cover, and continuous (unfragmented) native forest. Results of this study indicated that stream reaches in continuous native riparian zones had lower light levels, lower water temperatures and higher macroinvertebrate taxon richness, EPT taxon richness and Margalef's biodiversity index than in the pastoral and native fragment riparian zones.

B.3. Summary of associations between land cover classes and river state variables and values

The results of multiple studies that use land cover as a proxy for land-use pressure were highly consistent. Concentrations of the most frequently monitored contaminants (N, P, fine sediment, *E. coli*) increased with increasing pastoral and urban land cover in upstream catchments. Visual clarity and numerous ecological health indicators (e.g., MCI scores, fish IBI scores) declined along the same gradients. These patterns are apparent in riparian, catchment-, regional- and national-scale studies. A small subset of these studies distinguished linear and non-linear association between land cover and water quality or ecological conditions. These distinctions are valuable, as non-linear associations may indicate threshold levels of land cover, beyond which water quality or ecological state may degrade rapidly (e.g., Death and Collier 2010). In contrast to the water quality and ecological health studies, annual production of eels and crayfish were higher in streams dominated by pastoral land cover compared with natural land cover. In these cases, fish and invertebrate productivity appears to be severely limited in shaded forested streams with low primary production.

C. Associations between land use classes, land management practices and river state variables and impacts

Animal and plant based agriculture and plantations forestry account for most of the land use in New Zealand by area. As of 2016, 52% of New Zealand's land surface (263,310 km^2) was used for agriculture and forestry[9] In this section we consider 5 broad agricultural land use classes: sheep and beef (8.5 million ha), dairy (2.6 million ha), deer (325,000 ha with other livestock), arable cropping (449,000 ha), horticulture (191,000 ha for vegetables, berries and orchards), and forestry (1.7 million ha). Within each class, we review the state of knowledge about effects of land management practices.

C.1. Pastoral agriculture

Nutrient loss from land is one of the principle risks that pastoral agriculture poses for downstream receiving environments. Multiple studies of nutrient recycling in pastoral systems have concluded that nutrients ingested by livestock are inefficiently utilised for growth, or milk, meat and wool production (Haynes and Williams 1993). The return of unutilised nutrients via animal excreta is another design inefficiency that results in highly non-uniform distributions of nutrients (and faecal material) on the land surface. An inevitable consequence of these inefficiencies is the increased potential for losses of N and P from land to water due to the limited ability of pastoral plants to rapidly take up the N and P deposited in urine and dung patches, respectively. The other key component of the pastoral grazing system is the soil; many of the effects of pastoral agriculture that directly modify soil properties lead to environmental impacts on water and air quality.

The soil, plant and animal elements of pastoral agriculture introduced above make up the "structure" of pastoral farms, where factors such as stock type, geology, soil type, slope and rainfall strongly influence the transfers of N, P sediment and faecal microorganisms (FMOs) from land to water. Land management practices and intensity affect nutrient flows between soils, plants and animals, and transfers of N, P sediment and faecal microorganisms (FMOs) from land to water.

Most modern pastoral systems continually seek to improve business returns through increasing outputs of saleable product or through decreasing input costs, or both. An inevitable outcome of the former strategy is increased land use intensity and increased inputs of feed, fertiliser and energy. Inefficient nutrient cycling and increased losses of nutrients to the environment are usually an unfortunate consequence of this strategy. Intensive agriculture is known to lose significant amounts of the contaminants listed above: N, P sediment and FMOs (Gillingham and Thorrold 2000, Watson and Foy 2001, Monaghan 2007, Oliver et al. 2005). While these loses are typically not large by agronomic standards, they can significantly degrade aquatic receiving environments. These losses have been shown to increase as farm inputs increase and systems intensify (Ledgard et al. 1999, Monaghan 2005, Scholefield et al. 1993, Watson et al. 2000). In response, a range of management practices has evolved that can reduce some contaminant losses. Management factors such as crop integration, supplementary feeding strategy, wintering and irrigation methods, farm dairy effluent (FDE) handling and riparian protection have all been shown to have large effects on contaminant losses.

[9] http://archive.stats.govt.nz/browse_for_stats/environment/environmental-reporting-series/environmental-indicators/Home/Land/land-use.aspx

Here we review the current state of knowledge about the impacts of New Zealand's pastoral farming activities on water quality, with a focus on transfers of N, P sediment and FMOs from land to water. These are reviewed and discussed within the context of a 'structure-management-intensity' framework that we believe is helpful for explaining some of the contrasts between, and variability within, different categories of pastoral farms. For the purpose of this section, pastoral farming systems are predominantly grass-clover swards grazed by livestock, mainly ruminant cattle, sheep and deer. For completeness, contaminant losses to water from grazed forage crops are also considered because they represent a significant part of livestock diets in some areas of New Zealand.

Our review of land-use pressure associated with pastoral agriculture is based in part on previous reviews (Ledgard 2001, 2009, McDowell 2008, Quinn 2009). We have updated these reviews with recent work, expanded the scope to include contaminant losses and other pressures from a wider range of agricultural landscapes and farm systems, and extracted information about factors that influence contaminant losses (e.g., land slope, rainfall, fertiliser input). The following criteria were used to identify studies to include in the review:

- The study was carried out in New Zealand (with rare exceptions.
- The study was of sufficient size to incorporate the major flow pathways (i.e., overland and/or sub-surface flow) under grazing conditions representative of a particular pastoral land-use class. This criterion eliminated many plot-scale studies that focused on small scale processes and did not contribute to benchmarking contaminant loss rates from land-use classes.
- The sampling frequency in the study was sufficient for making robust estimates of contaminant loss.
- The study was published (with the exception of studies of winter forage grazing and *E. coli* loss rates; in both cases there is little published information).

Most of the studies we reviewed were farm-and paddock-scale studies. We also included catchment-scale studies that covered areas up to 10,500 ha. The catchment scale studies contributed to benchmarking loss rate, and some included multiple contaminants. We recognised that multi-scaled attenuation processes can confound comparisons of losses at different scales, and we checked that by comparing area-specific loss rates across scales.

A summary of the papers and reports we reviewed is in Table 3-1. This table is the basis of the discussions in the remainder of this report. Results are also presented graphically to illustrate the range of reported contaminant losses and variability within land use classes. Sixty two studies were reviewed, representing 77 location-treatment combinations. The majority of studies (26 location-treatment combinations) were from cattle-grazed systems, reflecting the research effort focused on this land use class over the past 20 years. Fewer studies reported contaminant losses from deer, sheep and grazed forage crop classes (8, 10 and 10 location-treatment combinations, respectively). Eight studies documenting losses from non-pastoral land uses were included in the review for comparison.

Table 3-1: Paddock and catchment scale contaminant losses from pastoral agricultural land-use classes. All loss and fertiliser input rates are in kg $ha^{-1}yr^{-1}$, except for *E. coli* loss, which is in cfu $ha^{-1}yr^{-1}$.

Stock	Major LU	Slope	Rain (mm)	Size (ha)	P fert input	N fert input	P loss	N loss	Sed loss	*E coli* loss	Region	Source
Sheep	Rangeland	Easy	690	4,800			0.1				Otago	Caruso (2000)
Sheep	Pasture	Easy	690	300	25		0.1	2	97	8.60E+09	Otago	McDowell and Paton (2004)
Sheep	Pasture	Easy	1,200	16	25		1.3	7	700		Waikato	Cooke (1988); Cooke and Cooper (1988)
Sheep	Pasture	Easy	1,200	1.5	65		0.7	9	1,220		Manawatu	Lambert et al. (1985)
Sheep	Pasture	Easy	1,448	<1			0.1	0.8			Wellington	McColl and Gibson (1979)
Sheep	Pasture	Rolling	1,453	1.4	56		1.1				Northland	McColl et al. (1975)
Sheep	Pasture	Easy	1,295	4	30		0.3	1.3			Wellington	McColl et al. (1977)
Sheep	Pasture	Easy	1,401	<1	45		0.8	4	374		Waikato	Smith (1987)
Sheep	Pasture	Flat-Rolling	1,510	3		0		72			Waikato	Hoogendoorn et al. (2011)
Sheep	Pasture	Rolling	761	2	36	52	0.4	3	595	2E+11	Otago	Monaghan et al. (2017)
Mixed[b]	Pasture	Rolling-Steep	1,600	296			2.4	17.3	2148		Waikato	Dodd et al. (2008b)
Mixed	Pasture	Flat	1,006	10,500	35	35	0.3	18	183		Southland	Thorrold et al. (1997)
Mixed	Mixed	Rolling	1,500	7,500	–		0.8	8			Bay of Plenty	Hoare (1984)
Mixed	Mixed	Rolling	1,923	7,330			0.6	7	128		Bay of Plenty	Williamson et al. (1996)
Mixed	Pasture	Flat	1,330	5,230	60	62	0.8	8	46	7.70E+10	Canterbury	Monaghan et al. (2009)
Mixed	Pasture	Flat	850	2,480	48	95	0.4	8	58	1.30E+11	Southland	Monaghan et al. (2007)
Mixed	Pasture	Steep	1,600	259	21		1.5	10	988		Waikato	Quinn and Stroud (2002)
Mixed	Mixed	Steep	1,600	266	21		1.3	7	2,632		Waikato	Quinn and Stroud (2002)
Mixed	Pasture	Rolling-Steep	1,600	266			2.8	12.8	8025		Waikato	Dodd et al. (2008b)
Mixed	Pasture	Steep	1,000	180	27		1.6	5	1,400		Manawatu	Bargh (1978)
Mixed	Pasture	Easy	1,500	11	25		1.7	12	22		BOP	Cooper and Thomsen (1988)
Mixed	Pasture	Easy	1,200	1.2	65		1.5	12	2,740		Manawatu	Lambert et al. (1985)
None	Native forest	Rolling-Steep	1,600	300			0.83	3.6	440		Waikato	Dodd et al. (2008b)
None	Native forest	Steep	1,600	300			0.6	2	320		Waikato	Quinn and Stroud (2002)
None	Native forest	Easy	1,500	28			0.01	4	27		Bay of Plenty	Cooper and Thomsen (1988)
None	Exotic forest	Easy	1,500	34			0.01	1			Bay of Plenty	Cooper and Thomsen (1988)

Stock	Major LU	Slope	Rain (mm)	Size (ha)	P fert input	N fert input	P loss	N loss	Sed loss	*E coli* loss	Region	Source
None	Native forest	Easy	1,295	11			0.2	0.01			Wellington	McColl et al. (1977)
None	Native forest	Easy	1,295	5			0.1	1.4			Wellington	McColl et al. (1977)
None	Exotic forest	Easy	1,295	4			0.07	0.04			Wellington	McColl et al. (1977)
None	Pasture	Flat-Rolling	1447	1	10	60		3			Waikato	Betteridge et al. (2007)
Deer	Pasture	Easy	687	4	25		0.9	6	4,480	3.41E+10	Otago	McDowell (2007)
Deer	Pasture	Rolling	944	36	35		3	3	3,950	9.79E+09	Otago	McDowell (2007)
Deer	Pasture	Easy	687	4	25		1.4	5	3,356	4.70E+11	Otago	McDowell (2008)
Deer	Pasture	Flat	1,100	32	31		0.6	7	158	1.07E+10	Otago	McDowell et al. (2006)
Deer	Pasture	Easy	1,300	120	30		1.8		850		Otago	McDowell et al. (2008a) + unpub
Deer	Pasture	Rolling	800	25	25		1.4	19	2,068	1.31E+11	Southland	McDowell et al. (2008a) + unpub
Deer	Pasture	Easy	800	24	25		0.6	14	398	5.52E+11	Southland	McDowell et al. (2008a) + unpub
Deer	Pasture	Flat-Rolling	1,510	3		0		67			Waikato	Hoogendoorn et al. (2011)
Dairy	Pasture	Flat		<1		92		46			Waikato	Roche et al. (2016)
Dairy	Pasture	Flat	1,000	<1	44	104	1.0	30			Manawatu	Houlbrooke et al. (2003, 2008)
Dairy	Pasture	Flat	1,000	<1	35	113	1.8	34			Manawatu	Houlbrooke et al. (2003, 2008)
Dairy	Mixed	Rolling	780	4	58	78	1.5		1,250	2.50E+10	Otago	McDowell (2006)
Dairy	Pasture	Flat	1,000	<1	50		0.4	29			Southland	Monaghan et al. (2005)
Dairy	Pasture	Flat	1,000	<1	50	100	0.1	34			Southland	Monaghan et al. (2005)
Dairy	Pasture	Flat	1,000	<1	50	200	0.2	46			Southland	Monaghan et al. (2005)
Dairy	Pasture	Flat	1,000	<1	50	400	0.4	54			Southland	Monaghan et al. (2005)
Dairy	Pasture	Flat	1,100	<1	25	85	0.9				Southland	McDowell and Monaghan (2015)
Dairy	Pasture	Flat	1,100	<1	23	126	4.5				Southland	McDowell and Monaghan (2015)
Dairy	Pasture	Flat	1,132	1,512	78	65	1.2	35	142		Waikato	Wilcock et al. (1999)
Dairy	Pasture	Flat	1,132	1,512	60	80	0.7	13	67		Waikato	Wilcock et al. (2006)
Dairy	Pasture	Flat	1,250	2,090	65	99	0.7	26	149		Taranaki	Wilcock et al. (2007)
Dairy	Pasture	Flat	4,800	600	48	142	7	45	960		Westland	Wilcock et al. (2013a)
Dairy	Pasture	Flat	1,200	13		139		54			Waikato	Beukes et al. (2017)
Dairy	Pasture	Flat	1048	0.1	37	184	0.9	25.5	120	1.83E+11	Southland	Monaghan et al. (2016)

Stock	Major LU	Slope	Rain (mm)	Size (ha)	P fert input	N fert input	P loss	N loss	Sed loss	*E coli* loss	Region	Source
Dairy	Pasture	Flat	700	0.1	15	20	0.08	20	3	5.67E+09	Otago	Monaghan and Smith (2004)
Dairy	Pasture	Flat	700	0.1	15	20	0.17	26	4	3.76E+10	Otago	Monaghan and Smith (2004)
Dairy	Pasture	Flat	1384	6		0		40			Waikato	Ledgard et al. (1999)
Dairy	Pasture	Flat	1384	6		215		79			Waikato	Ledgard et al. (1999)
Dairy	Pasture	Flat	1384	6		413		150			Waikato	Ledgard et al. (1999)
Dairy	Pasture	Flat	1384	6		411		133			Waikato	Ledgard et al. (1999)
Dairy	Pasture	Flat	980	<1		205		19			Manawatu	Christensen et al. (2012)
Dairy	Pasture	Flat	980	<1	14	205		11			Manawatu	Christensen et al. (2012)
Cattle	Pasture	Flat-Rolling	1447	1		82		13			Waikato	Betteridge et al. (2007)
Cattle	Pasture	Flat-Rolling	1447	1		82		5			Waikato	Betteridge et al. (2007)
Cattle	Pasture	Flat-Rolling	1,510	3		0		84			Waikato	Hoogendoorn et al. (2011)
Grazed forage crops												
Deer	Winter crop	Rolling	800	<1	60	45	2		1,012	5.10E+10	Southland	McDowell and Stevens (2008): + unpub
Dairy	Winter crop	Rolling	654	2	44	90	3.8	30	3691	4.4E+10	Otago	Monaghan et al. (2017)
Dairy	Winter crop	Rolling	654	2	44	90	1.3	10	825	1.9E+10	Otago	Monaghan et al. (2017)
Dairy	Winter crop	Rolling	1386	<1		200		153			Waikato	Shepherd et al. (2012)
Dairy	Winter crop	Flat	757	<1	67	67		78			Southland	Smith et al. (2012)
Cattle	Winter crop	Flat	2296	<1	101	147		157			Otago	Smith et al. (2016)
Dairy	Winter crop	Flat	1150	<1				52			Southland	Monaghan et al. (2013)
Dairy	Winter crop	Flat	1,100	<1	48	78	44				Southland	McDowell and Monaghan (2015)
Dairy	Winter crop	Flat	650	<1				83			Canterbury	Shepherd et al. (2017) (Kale)
Dairy	Winter crop	Flat	650	<1				42			Canterbury	Shepherd et al. (2017) (Fodder beet)
Dairy	Summer crop	Flat	970	<1				46			Manawatu	Hanly et al. (2017)

Pastoral agriculture - Nitrogen

Table 3-1 and Figure 3-1 show a wide range of reported annual N loss rates. For pastoral land, these ranged from 0.8 to 157 N $ha^{-1}yr^{-1}$. Figure 3-2 indicates that N losses are greatest for grazed forage crops, followed by cattle-grazed pastoral farms and then the remaining livestock classes, which had broadly similar median losses of 9, 7 and 4 kg N $ha^{-1}yr^{-1}$ for mixed, deer and sheep, respectively (Figure 3-2). As would be expected, N losses to water from forested and un-grazed land were low, ranging from <1 to 4 kg N $ha^{-1}yr^{-1}$. Where ammoniacal-N losses were reported (data not presented here), they usually comprised < 10% of total inorganic N leaching losses. An exception was reported by Shepherd et al. (2017) for a grazed winter forage crop. Dissolved organic forms of N represented a significant proportion of N loss at some locations, including both poorly-drained and naturally well-drained soils (Hoogendoorn et al. 2011, Monaghan 2016).

Some of the structural factors contributing to variability in reported N losses within livestock classes were rainfall and soil type, although correlations between individual structural factors and N loss were not be determined. Particularly large N losses were reported from sites with annual rainfalls greater than 1300 mm (Hoogendoorn et al. 2011, Ledgard et al. 1999). The large N losses reported for grazed forage crops were due to large amounts of mineral N remaining in the soil in late autumn following pasture cultivation and forage crop establishment, and to the deposition of excretal N onto the grazed forage crop during winter.

Nitrogen-fixing clover has traditionally played a critical role in pasture production by adding fixed N to soils for later use for pasture plants. More recently the reliance on clover has been reduced due to the availability of inexpensive N fertiliser, which has allowed total N inputs to pastures to increase. Whilst this has delivered greater pasture yields, Ledgard et al. (2009) document how N leaching from grazed pasture increases exponentially with increased N inputs Figure 3-3). Results from multiple studies indicate that urinary-N accounts for 70-90% of the N leaching. Farm system and life-cycle assessment studies indicate that, at similar levels of N inputs, N losses to waterways from clover/grass pasture systems appear to be similar to those from N-fertilised grass systems (Ledgard et al. 2009).

Some management practices that influence N losses from pastoral agriculture are apparent in the studies in Table 3-1. Taking animals off-paddock at critical times of the year, integrating feed types that reduce urinary N excretion and modifying stocking rates and N fertiliser inputs all reduce N losses to drainage (Ledgard et al. 1999, 2006, Christensen 2012, Monaghan 2016, Beukes et al. 2017). Each of these management practices reduces N losses and can be made without major changes in farm intensity, albeit some measures do incur significant capital or operating cost.

The modifying effects of soils, rainfall, slope, input rates and specific management practices on N losses set out above complicate predictions of loss rates from simple bivariate relationships between pressures and losses. For these reasons, farm-scale tools such as the Overseer Nutrient Budgeting model have been developed to accommodate the components that make up the structure-management-intensity framework we outlined above (Burkitt et al. 2016, Wheeler et al. 2006).

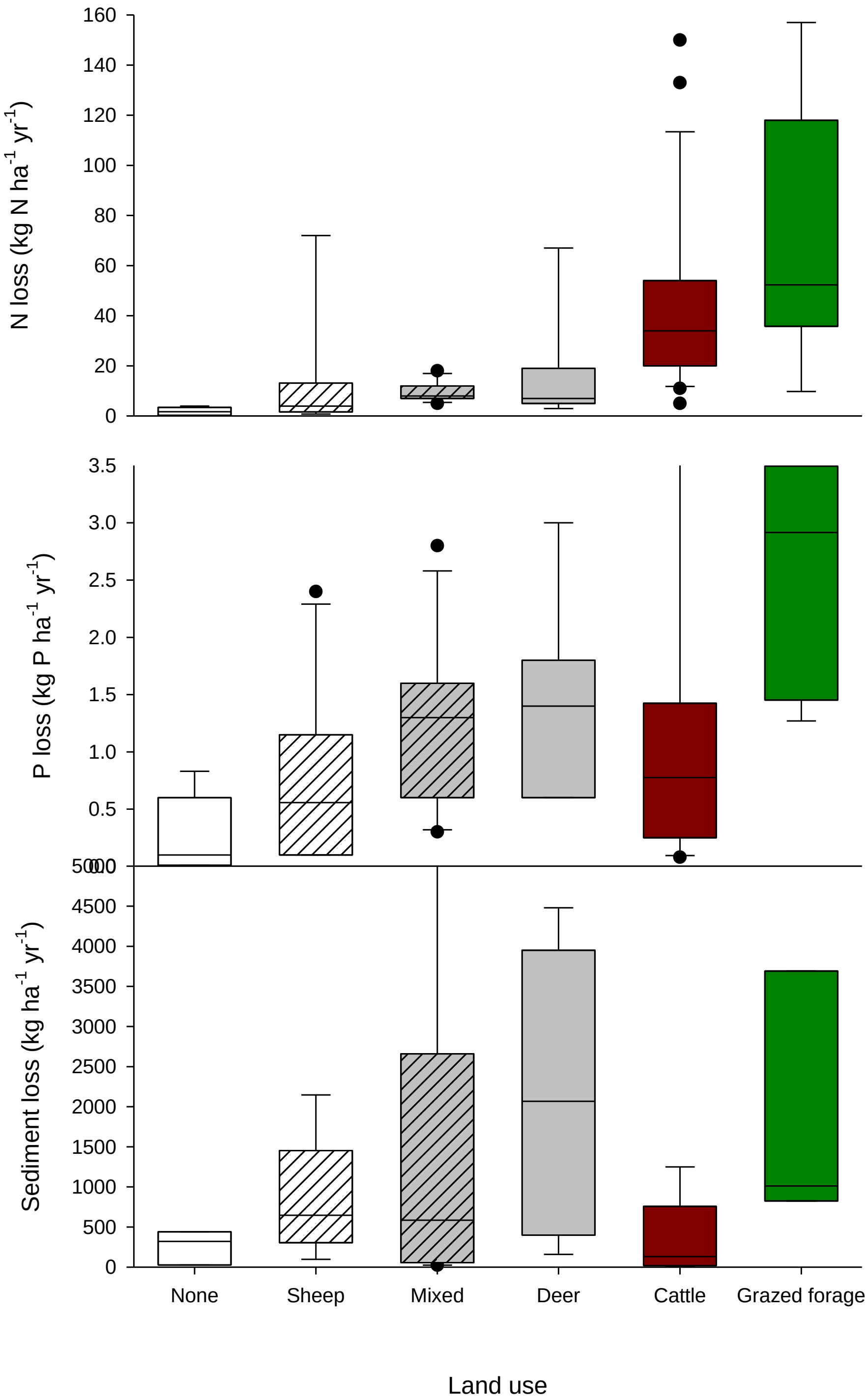

Figure 3-2: Box plots showing median concentration (central horizontal line). 25th and 75th percentiles (box), 10th and 90th percentiles (whiskers), and outliers (dots) for (a) N, (b) P, and (c) sediment annual loads for each pastoral stock class. 'None' refers to non-agricultural rural land uses, such as exotic plantation and native forest. 'Mixed' refers to catchments with more than one stock class. 'Cattle' refers to both dairy and beef stock classes. Data for the box plots are from the references in Table 3-1.

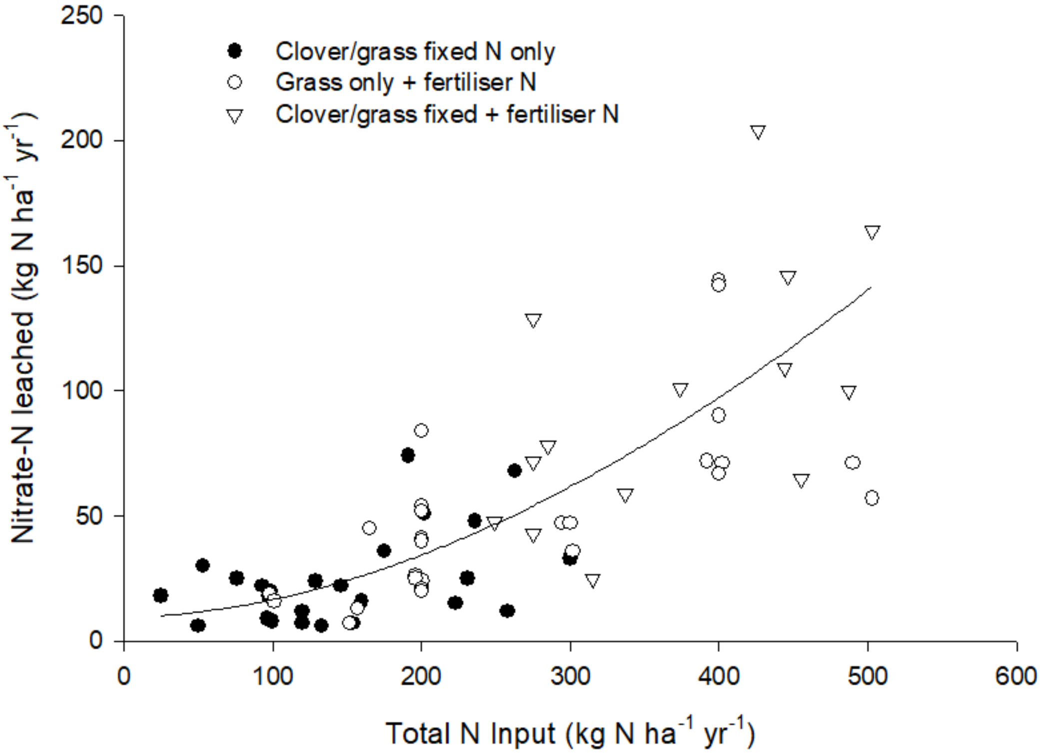

Figure 3-3: Nitrate-N leaching rates from grazed pasture systems as affected by total N input from fertilisers and/or atmospheric nitrogen fixation by clover. Data are a summary of studies from New Zealand, France, United Kingdom and Denmark. The line of best fit is an exponential function obtained by fitting the data on the log scale. Updated from Ledgard (2001).

Pastoral agriculture - Phosphorus

As observed for N, P concentrations in rivers draining pastoral land are frequently higher than in rivers draining land dominated by native vegetation and planted forests (Larned et al. 2016). Elevated P concentrations in pastoral rivers have be attributed to a range of land use and land management practices (discussed below), many of which differ from practices that control N losses. Collectively, the studies undertaken in intensively farmed catchments indicate that P concentrations do not exceed guideline values to the degree observed for N. A relevant example is the dairy catchments study reported by Wilcock et al. (2013b), where median P concentrations exceeded guideline values by 1 to 4 times, whereas median nitrate-N concentrations exceeded guideline values by 1.5 to 13 times.

Losses of P from pastoral land uses ranged from 0.08 to 7 P kg ha^{-1}yr^{-1} (Table 3-1). As noted for N, losses were greatest for grazed forage crops, although this is based on a limited dataset from 3 studies. Notable features concerning P loss in Figure 3-2 are (i) the lower median value of P loss from the dairy land-use class compared to those from deer and mixed land-use classes, and (ii) the exceptionally large losses reported for two studies under dairy land use (Wilcock et al. 2013a, McDowell and Monaghan 2015). P losses to water from forested and un-grazed land were generally low, ranging between 0.01 to 0.83 kg P ha^{-1}yr^{-1}, with a median value of 0.1 kg P ha^{-1}yr^{-1}. Greater variability in P loss rates is evident in all land use classes compared to N loss rates. Some of the key structural factors contributing to this variability are rainfall, slope and soil type. For the mixed land-use class this is illustrated by the relatively large (> 1.5 kg $ha^{-1}yr^{-1}$) P losses at most locations where slope classes were easy or steep and rainfall was 1200 mm per annum or greater. A very large P loss was reported for dairy land use in the study of Wilcock et al. (2013a), reflecting the high potential for P transport in surface flow pathways on the South Island west coast, where rainfall exceeded 4,000 mm per annum. The exceptionally large P losses reported by (McDowell and Monaghan 2015) were attributed to the low P retention capacity of the organic soils evaluated in their study. DRP and/or particulate P made significant contributions to total P losses, as observed in other studies of grazed grasslands (e.g., (Haygarth et al. 1998, Douglas et al. 2007, Bilotta et al. 2008, van der Salm et al. 2012).

Phosphorus input is one of the key management factors influencing the P losses in Figure 3-2 and Table 3-1. Clover has a higher P requirement than grass, and where extra P fertiliser is used for clover/grass pastures, the risk of P loss to waterways is greater than for grass-only pastures. The greater productive potential and profitability of land used for dairy farming has resulted in greater inputs of fertiliser P to dairy catchments, and consequently greater levels of stored P and greater risks of P loss in runoff. McDowell et al. (2003b) describe some of the relationships between soil P fertility and P runoff risk for a range of soil types (Figure 3-4). Animal treading effects on soil structure, particularly during wet conditions, is another intensity factor that governs the pathways of P loss from grazed pastoral land (McDowell et al. 2003a, Monaghan et al. 2016). Treading interacts with soil and animal type such that large P losses can be expected for poorly drained soils and pastures or forage crops that are grazed by heavier classes of livestock (McDowell and Houlbrooke 2009, Monaghan et al. 2016, 2017). Hamill and McBride (2003) suggest that increased dairy farming has been associated with increasing DRP concentrations in rivers in Southland, which reflects the actions of all of the intensity factors listed above.

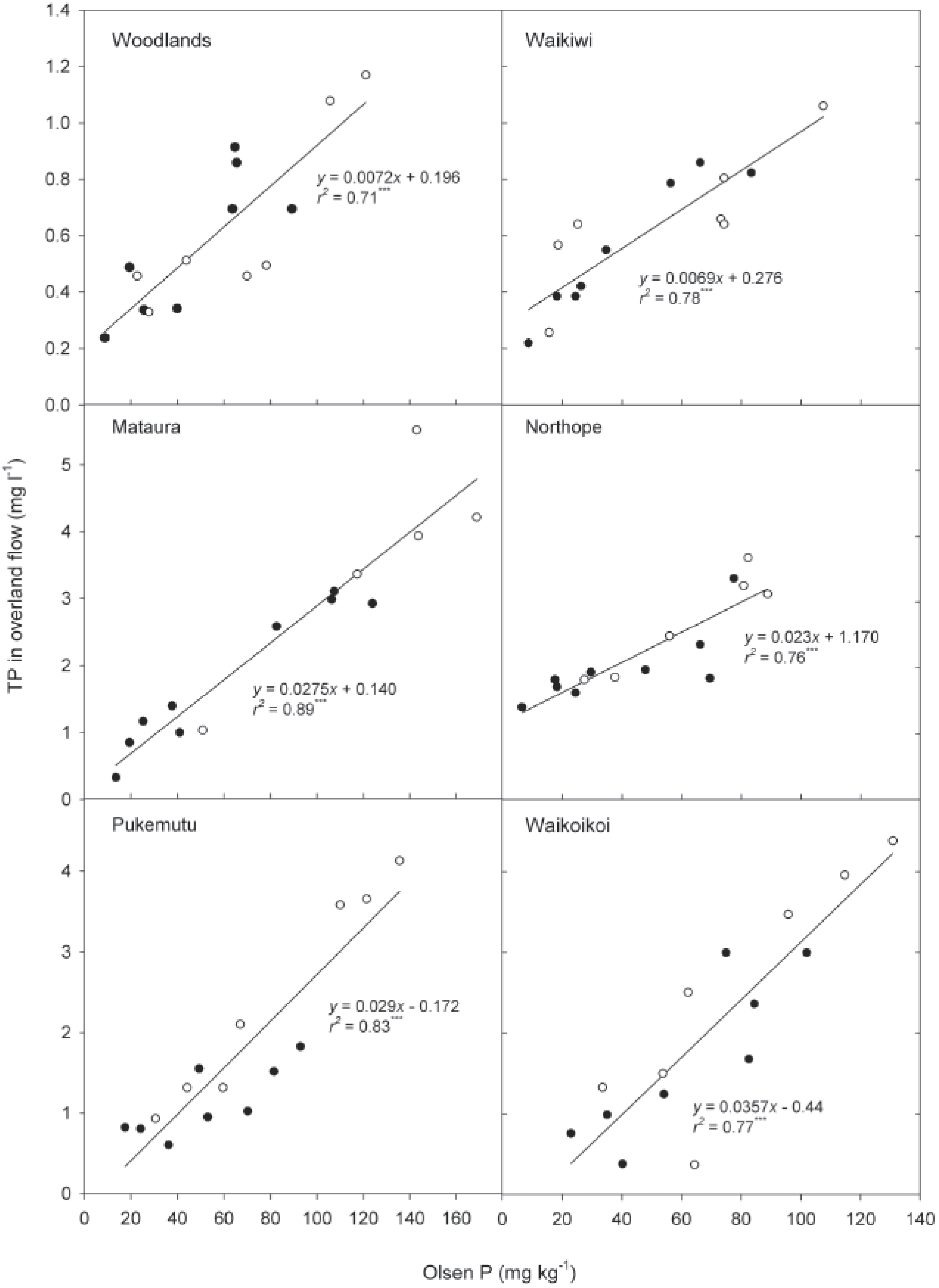

Figure 3-4: Relationship between total phosphorus (TP) in overland flow and Olsen P for six Southland soils. Open and closed symbols represent soils with superphosphate and manure applied to them, respectively (re-produced from McDowell et al. 2003b).

In addition to the Good Management Practices (GMPs) that have been developed and recommended for the management of P fertilisers and soil P fertility, research and extension efforts have been targeted towards ensuring farm dairy effluent (FDE) is managed appropriately and that direct inputs of faecal material to riparian zones is avoided. These measures recognise the P-enriched nature of faecal material and the potential for detachment of P-enriched sediments along riparian margins as a result of animal treading on river banks and beds. Studies by Houlbrooke et al. (2004a, 2008) and Monaghan and Smith (2004) demonstrate how the preferential flow of FDE through mole-pipe drained soils can convey faecal-P directly to rivers. Similarly, the two-pond FDE treatment systems that were common on dairy farms until the 1990s were recognised as significant contributors of P to waterbodies and have been phased out. They are being replaced by land-based application methods, coupled with FDE pond storage facilities where soil risk factors determine they are required (Houlbrooke et al. 2004b, Monaghan et al. 2010). Improvements to flood irrigation systems and their management have also helped reduce the potentially large transfers of P from land to water under this type of water irrigation practise (Monaghan et al. 2009).

The Whatawhata integrated catchment management project was an important case study that explored land use and management change options to improve the economic and environmental performance of hill country farms. Research observations, decision support models and expert stakeholder knowledge were used to develop and implement a plan to afforest 160 of the 296 ha catchment farm with pine and native trees, manage the entire 20-km riparian network via fencing and/or forestry, restore 5 ha of existing native forest, and intensify the remaining pastoral land area (Dodd et al. 2008a). Marked improvements were observed in key environmental performance indicators, with declines in sediment (76%) and phosphorus (62%) loads and faecal coliform (43%) levels (Dodd et al. 2008b). Whilst the authors acknowledged the challenge of better matching land use to land capability, their study demonstrated that significant progress can be made if key land use and land management practices are spatially targeted in hill country landscapes.

Pastoral agriculture - Sediment

Documented losses of sediment to water from pastoral land uses ranged from 22 to 8,025 kg ha^{-1}yr^{-1} (Table 3-1). Notable features of Table 3-1 and Figure 3-2 are (i) relatively large losses from deer and grazed forage crop land, (ii) relatively small losses reported for dairy cattle, and (iii) highly variable loss rates for the mixed, deer and grazed forage land use classes. In the three studies that document sediment losses from native forest land, loss rates were 27, 320 and 440 kg ha^{-1}yr^{-1}. Slope appears to be an important structural factor influencing reported losses: across all of the pastoral and grazed forage land use classes (34 loss measurements), median values of annual sediment loss were 120, 1250, 1035 and 1400 kg ha^{-1}yr^{-1} for flat, rolling, easy and steep slope categories, respectively. Corresponding maximum values of sediment loss were 960, 3950, 8025 and 2632 kg ha^{-1}yr^{-1}, respectively. No clear relationships were evident between rainfall and sediment loss for the 31 results reported for grazed pastoral land uses listed in Table 3-1. The low sediment loss rates from dairy cattle grazed land probably reflect the fact that most of the studies of dairy farms in Table 3-1 were located on flat land with low potential for loss via erosion and/or surface runoff. The higher sediment loss rates reported for deer land use indicates that animal type is another structural attribute that has an important effect on sediment loss. The inclination of farmed deer to wallow in streams and wet areas has been shown to exacerbate soil erosion and sediment losses (McDowell 2007, 2009). Another feature in Table 3-1 and Figure 3-2 is the large sediment loss reported for grazed forage crops. These high loss rates are unsurprising, given the propensity for surface runoff from intensively grazed cropland on rolling topography. Mole-pipe drainage systems have also been identified as a structural factor that can increase sediment loss (Monaghan et al. 2016); given the prevalence of mole-pipe drainage in some regions, they can account for a significant proportion of catchment-scale sediment loss (Walling et al. 2002, McDowell and Wilcock 2004).

River bank collapse and soil erosion caused by cattle grazing in riparian zones can be a major source of sediment in rivers (Wilcock 2008). Stock exclusion from rivers and riparian margins is therefore an important management factor for controlling sediment losses. Wilcock et al. (2013b) reported reductions in sediment loading and increased visual clarity in five dairy catchment streams in response to increased stock exclusion (see Appendix E, Dairy Catchment case study for details). Other management factors for reducing sediment losses from pastoral land use include excluding stock from wetlands and boggy areas, grazing strategies that protect soil structure (particularly in critical source areas (CSAs)), locating forage crops away from CSAs, shaping or bunding farm lanes to prevent surface runoff directly into rivers, preventing deer pacing and wallowing near streams, and tree planting to stabilize soils in erosion-prone areas.

Pastoral agriculture - Faecal microorganisms

The microbiological water quality of rivers in pastoral landscapes is often poor and faecal indicator organism numbers frequently exceed water quality standards (McDowell et al. 2008b, Larned et al. 2016). Muirhead (2015) noted that our current knowledge of the sources and transport of faecal microbes to water is significantly less than our understanding of nutrient and sediment loss processes. Further complexity is added by the significant fluxes of FMOs lost from farms in overland flow that occurs during storm events, a proportion of which may replenish instream (sediment) reservoirs of FMOs (Muirhead et al. 2004), which are subsequently released to the water column under base-flow conditions. In contrast to storm flows, water inputs to streams during base-flow conditions are dominated by groundwater inputs, which are considered to be a very minor source of FMOs to waterways. Some known contaminant sources that can affect a FMO input to rivers independently of rainfall-driven runoff or groundwater flows are deposition of faeces from animals directly into river water (Davies-Colley et al. 2004, McDowell et al. 2008b), discharges of farm dairy effluent (FDE) to rivers via two-pond treatment systems, preferential or overland flow of land-applied FDE (Houlbrooke et al. 2004a, 2008, Monaghan and Smith 2004), and overland flow losses from excess irrigation water (Monaghan et al. 2009, Wilcock et al. 2011) and farm lane-ways (Monaghan and Smith 2012). Modelling analyses indicate that FMO inputs from birds is significant in some catchments where environmental mitigations have been widely implemented (Muirhead et al. 2011).

There are relatively few studies of FMO losses from pastoral land uses. This shortage is due in part to lack of FMO measurements in monitored flows, and to researchers not measuring or reporting the volumes of flow in surface runoff or subsurface drainage when measuring FMO concentrations. The few annual yields that have been documented in Table 3-1 range from 5.7 x 10^9 (grazed dairy pasture; Monaghan and Smith 2004) to 5.5 x 10^{11} MPN ha^{-1} (deer pasture; McDowell et al. 2008a and unpublished results). Given the sparse dataset, no clear effects of land use are evident.

Long-term monitoring of five catchment streams that drain dairy-dominated catchments indicates that *E. coli* concentrations frequently exceeded guidelines for contact recreation (Figure 3-5), despite a number of changes to farm management practices and improving (i.e., decreasing) trends in *E. coli* concentrations in two of the monitored streams (see Appendix F, Dairy Catchment case study for details). Field scale monitoring of *E. coli* losses in the Bog Burn catchment studied by Wilcock et al.

(2013b), indicated that surface runoff from the poorly drained Pukemeutu soil in wet spring conditions accounted for 68% of annual transfers in combined surface runoff and subsurface drainage flows (Monaghan et al. 2016). Protection of soil physical conditions during wet spring conditions through on-off grazing strategies was recommended to reduce these FMO losses.

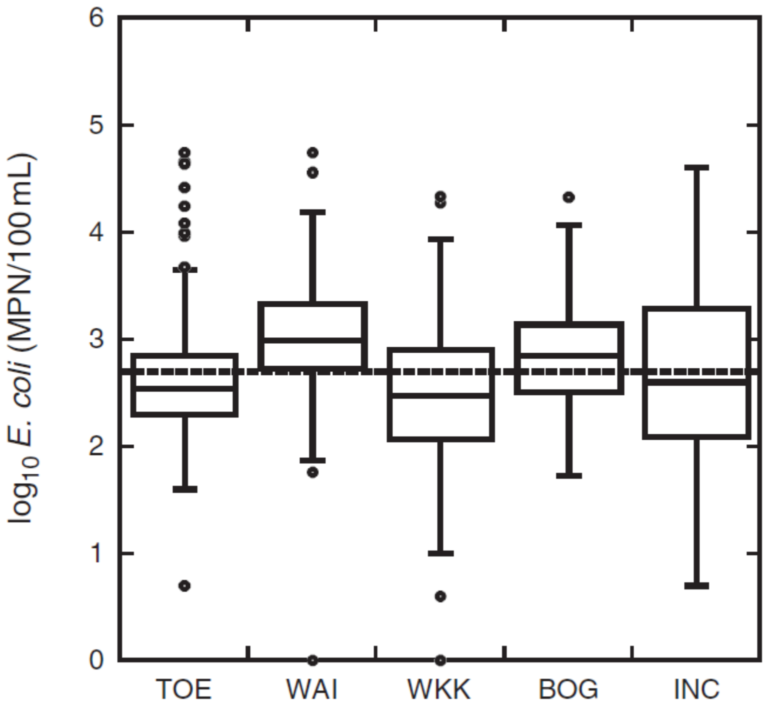

Figure 3-5: Box plots showing log10 E. coli concentration data from five catchment streams that drain predominantly dairy land. Each box contains the interquartile range (IQR) and median (solid line). The whiskers extending from the boxes mark values that are 1.5 times the IQR above and below each box. Values outside this (o) are designated as outliers. A guideline value for contact recreation (ANZECC 2000, Ministry for the Environment 2003) is shown as a horizontal dashed line. TOE = Toenepi; WAI = Waiokura; WKK = Waikakahi; BOG = Bog Burn; INC = Inchbonnie (from Wilcock et al. 2013b).

Modelling approaches for describing the sources, transport and fate of FMOs are less advanced than for nutrients. The information needed to build such models requires metrics for *E. coli* (the FMO indicator) losses from different land uses and land management practices, while factoring in rainfall, soil type and other structural factors. It is evident from Table 3-1 that the data to guide and such model developments are scarce. The effects of land use intensification on pastoral farms is also a key consideration that needs to be accounted for given the likely increase in faecal reservoirs on land in response to increased stocking densities and/or stock productivity. Finally, more quantitative data are required to account for the effectiveness of mitigations and the effect of instream FMO reservoirs. Estimates of the risks of *E. coli* loss based on excretion rates from different types of stock indicate that the *E. coli* reservoir on sheep-grazed pasture is much greater than for dairy cow-grazed pasture (McDowell 2006). This is unexpected as it is widely assumed that *E. coli* losses from dairy land are the highest of the pastoral land use classes. Unfortunately, the data in Table 3-1 are insufficient to compare *E. coli* loss rates across pastoral classes. While this review identifies several structural and management factors that influence the effects of pastoral land use on FMOs in rivers, the data and knowledge needed model and predict these effects accurately are currently lacking in New Zealand and overseas (Cho et al. 2016, Oliver et al. 2016).

Pastoral agriculture - Habitat

The impacts of pastoral agriculture on river habitat in New Zealand have been recently reviewed, and for the purposes of this report we refer readers to those reviews (McKergow et al. 2016, Wilcock et al. 2008, 2013b, Wright-Stow and Wilcock 2017). Some of the key messages from the reviews are briefly described here. The removal of mature riparian vegetation was a common practice for most of the history of pastoral agriculture in New Zealand. The loss of shade and the loss of contaminant retention by riparian processes has contributed to nutrient enrichment, elevated levels suspended and deposited sediment, and nuisance growths of macrophytes and algae in many pastoral streams. Wilcock (2008) describes how rapid plant growth and photosynthetic production alter natural dissolved oxygen (DO) and pH fluctuations in these stream, sometimes elevating pH levels high enough to induce ammonia toxicity. McKergow et al. (2016) documented many of the functions and benefits of well-managed riparian areas and document how some of these can be compromised by pastoral agriculture. Stock access to streams and riparian margins causes bank erosion and reduces soil infiltration, which reduces the capacity for contaminant retention in riparian margins. Stock access also degrades fish and invertebrate habitat. (Wilcock et al. 2013b) and (Wright-Stow and Wilcock 2017) reported reductions in suspended sediment concentrations and increased visual clarity in intensively-farmed pastoral streams, which they attributed to improved riparian protection and improved management of forage crop grazing. Their results also suggested that stock exclusion was improving habitat quality and macroinvertebrate-based ecological health indicators, but the time courses for these improvements appear to be longer than for suspended sediment and visual clarity.

C.2. Arable cropping and horticulture

As is the case for pastoral agriculture, arable cropping and horticulture in New Zealand are under scrutiny with regard to adverse effects on aquatic ecosystems (PCE 2004). The level of concern has been greater for pastoral agriculture due to the larger land area used by that sector, but environmental research to address adverse effects has increased in all three sectors. The traditional research focus for arable cropping and horticulture on optimising plant nutrient use for production (e.g., Goh and Haynes 1983) has expanded to include the quantification and prediction of losses, and techniques to reduce those losses (Cichota et al. 2010, Zyskowski et al. 2016). There are numerous differences in the types of contaminants lost from pastoral and cropping systems, and the processes that control those losses. For example, most of the N leached from dairy farms originates as urine (Vogeler et al. 2013), whereas most of the N leached from cropping and horticultural land originates as fertiliser or plant residues. Contamination of water bodies by faecal microbial pathogens like *Campylobacter* is a major concern for pastoral agriculture, but not for cropping and horticulture. In contrast, contamination by insecticide, fungicide and herbicide residues from cropping and horticultural land is a growing concern due to high application rates per unit land area (Mankeltow et al. 2005).

Published reports of contaminant losses from arable and horticultural crop land are scarcer than for pastoral agriculture. N leaching losses and overland sediment and pesticide losses from arable and horticultural crop land have been measured and modelled (Francis and Knight 1993, Basher et al. 1997, Webb et al. 2001, Lilburne et al. 2010). However, there appear to be no published measurements of P loss from arable crop land (Grey et al. 2016), and P loss from horticultural land has been estimated by difference after accounting for other sources (Alexander et al. 2002). P loss from both arable and horticultural crop land has been assumed to be negligible due to the small areas occupied by these land uses (Parfitt et al. 2013). In contrast, annual nitrate-N leaching rates in excess of 100 kg ha^{-1} yr^{-1} have been reported from arable crops, which are comparable to those reported for intensive pastoral agriculture.

Organic farming and no-tillage farming are expanding subsectors of arable and horticultural farming in New Zealand (Cooper et al. 2010, Ward and Siddique 2015). The adverse environmental effects of organic farming are often presumed to be lower than for conventional farming, including reductions in nutrient input requirements, but there is little information from New Zealand farms to test that presumption. Meta-analyses of numerous overseas studies suggest that N and P losses from organic and conventional farms are roughly comparable, when normalised by N and P input or by crop yield (Mondelaers et al. 2009, Gomerio et al. 2011, Tuomisto et al. 2012). The same studies report that organic farms generally have higher soil organic matter and lower soil erosion rates, which may reduce sediment runoff. In New Zealand, the comparative effects of organic, no-till and conventional land management practices have been assessed using terrestrial ecological health indicators such as soil properties, insect and bird diversity, and broader ecosystem service provisions (Sandhu et al. 2008, MacLeod et al. 2012, Todd et al. 2016). However, comparable assessments of the effects of organic and no-till agriculture on river health have not been published.

As is the case for pastoral agriculture, knowledge about contaminant losses from arable and horticultural land very rarely extends to contaminant input to rivers, and even more rarely to changes in river state or ecological impacts. This is a substantial research gap, particularly in comparison with the large body of evidence from overseas concerning effects of arable cropping and horticulture on rivers (e.g., Stone et al. 2005).

C.3. Forestry

Planted forests cover approximately 6.4% of New Zealand's land area. These forests are dominated by *Pinus radiata* (90%), most are in private ownership (92%), and most are certified by the Forest Stewardship Council, a global forest certification system (73%). Approximately 20% of the planted forest estate is in steep hill country (>20° slope) and the remainder is in on lower gradient land (Dunningham et al. 2012). In recent years, the total planted forest area has declined, particularly in regions where the land has been converted to more intensive agricultural practices such as dairying. At the time of this report, the government's programme to plant one billion trees over 10 years (between 2018 and 2027) is in an early planning stage, but this programme has the potential to substantially expand planted forests for both timber and non-timber values.

Forestry pressures

Planted forests often comprise large contiguous areas of land, and they have the potential to exert both positive and negative pressures on the state of receiving freshwater environments (Figure 3-6). Those pressures include rainfall interception and evapotranspiration, run-off and erosion, input of vegetative material to rivers and changes in solar radiation reaching river surfaces.

Due to their canopy roughness and large leaf surface areas, planted forests have higher rainfall interception and higher evapotranspiration rates than agricultural and urban areas. In turn, high interception and evapotranspiration reduces water availability for run-off and erosion (Figure 3-6) (Fahey et al. 2004, Davie and Fahey 2005). As they mature, planted forests increase soil stability and reduce shallow erosion through root reinforcement, particularly in steep, erosion-prone terrain where afforestation is used for erosion control as well as production (Maclaren 1996, Marden 2012, Marden et al. 2014, Phillips et al. 2017). However, soil stabilisation by afforestation may have an upper limit that is associated with natural erosion rates and historical forest clearance. Planted forests also reduce light levels at river surfaces (i.e., ʽstreamlighting'), and there is a strong positive correlation between channel width and light penetration; in mature pine forests, stream channels with widths < 3 m have median lighting approximately 1% of full sunlight (Davies-Colley and Quinn 1998). Planted forests in New Zealand are characterised by high rates of organic matter input to rivers in the forms of logs, branches, trigs, leaves needles, and insects. In shaded forest streams, this land-derived or ʽallochthonous' input far exceeds the instream or ʽautochthonous' production of organic matter in the form of algae and macrophytes. Allochthonous inputs vary widely through the forest rotation cycle, with low to moderate rates of input during maturation, higher pulses during thinning, and highest inputs during harvesting, mainly in the form of logging slash (Quinn et al. 1997, Collier and Baillie 1999, Thompson and Townsend 2004).

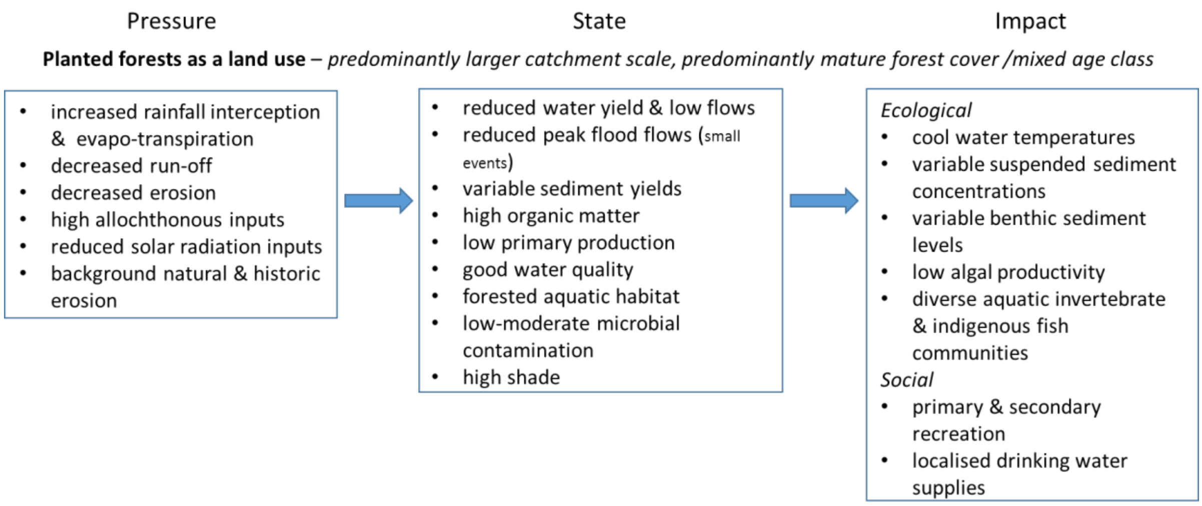

Figure 3-6: Key pressure-state-impact relationships for planted forests as a land use.

Effects of afforestation

Afforestation refers to the conversion of land from grassland or other land-cover classes to planted forest, and subsequent forest growth in terms of height, areal cover and biomass. Afforestation in New Zealand occurs most frequently on grassland and scrubland. *Pinus radiata* forests usually reach canopy closure 6-9 years after planting and are harvested after 25-35 years (Cooper et al. 1987, Eyles and Fahey 2006, Rolando et al. 2013, 2016, Smaill and Clinton 2016, Fahey and Payne 2017).

Catchment water yields generally decrease progressively during the afforestation cycle as canopy interception and evapotranspiration increases (Figure 3-6). For example, conversion of tussock grassland to planted forest in the Glendhu experimental catchment in east Otago reduced annual water yield by 130 mm (compared with the unplanted control catchment) at the time of canopy closure, and by 273 mm at maturity. Low and peak flows in the streams draining the Glendhu catchment followed a similar trend, with a 26% reduction in low flow in the afforested catchment and a 78% reduction in peak flows during small floods (Fahey and Jackson 1997, Fahey and Payne

2017). Smaller reductions in water yields have been reported when planted forests replace indigenous scrub or gorse, in comparison with tussock grassland (Davie and Fahey 2005).

A national-scale survey of annual water and sediment yields measured in small catchments before and after afforestation indicated that afforestation caused water yields to decrease by 25 to 81% and sediment yields to decrease by 27 to 95% (Blaschke et al. 2008). The wide ranges in yields reflect the wide ranges of pre-existing vegetation types and planted forest ages in the survey.

Duncan and Collins (2013) summarised the results of several New Zealand paired-catchment studies of afforestation. Reductions in runoff following afforestation were reported for all studies. These results were used to develop a rainfall-runoff relationship for planted fully forested catchments. The relationship indicated that afforestation reduced runoff by 58% at a rainfall of 1000 mm.

Annual sediment yields from catchments dominated by planted forest tend to be low compared to other land uses, although sediment yields can vary by orders of magnitude in response to rainfall, geology, topography, historic land management practices and stage of forest rotations (Figure 3-6). The highest sediment yields reported were associated with large storms in soft-rock geological terrain in the eastern North Island (Hicks 1990, Fahey et al. 2004, Basher et al. 2011, Marden et al. 2014, Quinn and Phillips 2016). In an 11-year study of the Pakuratahi catchment in Hawke's Bay, annual sediment yields from planted forest (713 t km^{-2} yr^{-1}), which included sediment pulses from harvesting activities, were lower than from the adjacent pasture-dominated catchment (1168 t km^{-2} yr^{-1}) (Fahey and Marden 2006; see Appendix G Pakuratahi case study for details).

Allochthonous inputs provide an important energy source in planted forest streams, particularly in small shaded streams with low primary productivity. Annual litter inputs to three mature planted forest streams in the Waikato Region averaged 485 g dry mass m^{-2} yr^{-1} (Scarsbrook et al. 2001). Similar input rates have been reported from mature planted forest streams in the Otago Region (Townsend et al. 1997a). In a nation-wide assessment, the standing stock of large woody debris (logs and branches) in the channels of planted forest stream averaged 112 m^3 ha^{-1} (Baillie et al. 1999). This estimate suggests that planted forest streams tend to have lower standing stocks of large woody debris than native-forest streams, for which an average of 206 m^3 ha^{-1} was reported from a survey of 18 streams (Meleason et al. 2005). In-stream decay measurements indicate that large woody debris composed of *Pinus radiata* can persist in hydrologically stable streams for up to 20 years (Collier and Halliday 2000). Large woody debris contributes to habitat diversity and sediment and organic matter retention in planted forest streams, which can in turn benefit aquatic macroinvertebrate and native fish communities through habitat and food provisioning (Quinn et al. 1997, Collier and Halliday 2000, Collier and Smith 2003, Baillie and Davies 2002, Baillie et al. 2013).

Results from several catchment-scale studies indicated that instream nutrient and *E. coli* concentrations decreased following afforestation. Instream nitrate-N, total P and DRP concentrations declined within 4-5 years of tree planting in a small (34 ha) catchment in the central North Island (Cooper et al. 1987, Davis 2014). In three small pasture catchments in and adjacent to the Whatawhata Research Centre, *E. coli* concentrations declined by a factor of 10 (to a median concentration of 41 *E. coli* 100 mL^{-1}) two years after conversion of pasture land cover to planted forest (Donnison et al. 2004). Similar reductions were observed in the same study seven years after planting; these reductions were attributed to livestock exclusion with streamside fences and eradication of feral animals at the time of planting (Donnison et al. 2004).

As part of long term study of integrated catchment management at the Whatawhata Research Centre, a 95-ha subcatchment of Mangaotama Stream was converted from 100% grazed pasture to 100% *Pinus radiata* (Quinn et al. 2009). Livestock were excluded from the catchment and a 10-m wide unplanted riparian buffer was established along the tributary stream margin. Six years after tree planting, the following changes were observed in the tributary stream: the channel had narrowed by approximately 30%, periphyton biomass had decreased by 80%, and the values of four macroinvertebrate-based ecological health indicators increased significantly: EPT taxon richness, % EPT density, quantitative MCI score, and macroinvertebrate IBI. Six years after planting, the macroinvertebrate community composition was close to that in an adjacent reference stream with native forest land cover.

Effects of forestry practices

A typical planted forest rotation comprises land preparation, tree-planting, thinning and pruning, fertilisation, pest and weed control, and harvesting. The effects of fertiliser applications, pest and weed control and harvesting on river environments have been assessed in New Zealand planted forests and are summarised in this section. There are large differences in the amount of published information that exists about the effects of these practices, which is reflected in the level of detail in the following summaries. We know of no published information on the effect so thinning and pruning on river environments.

The two published reports about the effects of fertiliser applications in planted forests on stream water quality in New Zealand are over 40 years old (Leonard 1977, Neary and Leonard 1978). These reports were based on five case studies on the North and South Islands. In four of the case studies, trees were fertilised 5-6 years after planting. In the fifth case, trees were fertilised approximately 20 years after planting. Application rates were 200 or 500 kg ha^{-1} for urea and from 400-1250 kg ha^{-1} for superphosphate. Peak dissolved and total N and P concentrations in adjacent streams occurred on the day of fertiliser application or shortly after, depending on soil and groundwater transport. Instream N and P had returned to baseline levels within two months of fertiliser application. In view of changes in forestry practices and fertiliser formulations and application rates in the decades since the two reports, updated information on effects of fertiliser use on freshwater environments is needed.

Herbicides are used in plantation forestry for pre- and post-planting weed control. The main herbicides used are glyphosate, terbuthylazine and hexazinone (Rolando et al. 2013). Instream herbicide residues were recently monitored in two central North Island studies before and after the aerial application of a herbicide mix of terbuthylazine and hexazinone (Baillie et al. 2015, Baillie 2016). Peak herbicide concentrations were detected in the streams on the day of herbicide application (Figure 3-7) or during a rainfall event within one month of application. Herbicide concentrations rapidly decreased downstream and were at or below detection limits at the base of the catchments. Terbuthylazine concentrations exceeded WHO and New Zealand drinking water concentrations for ≤ 24 hours, otherwise herbicide concentrations remained below drinking water standards and lethal concentrations for aquatic organisms.

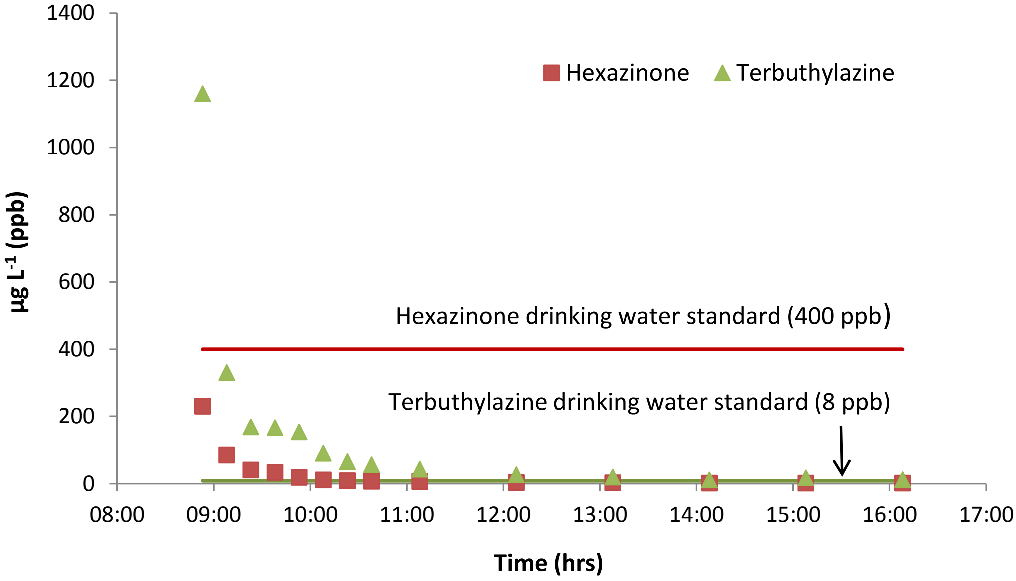

Figure 3-7: Concentrations of terbuthylazine and hexazinone measured in a small stream on the day of aerial herbicide application (from Baillie et al. 2015).

The most common fungicide used in planted forests is cuprous oxide, for the treatment of *Dothistroma* needle blight. *Pinus radiata* is highly susceptible to *Dothistroma* infection from four to fifteen years of age (Bulman et al. 2004). To assess the impact of fungicide treatments on water quality, cuprous oxide was aerially applied to three *Pinus radiata* forests of varying age classes in the central North Island. Peak copper concentrations in stream water across the three sites ranged from 28 to 60 μg L^{-1}, but declined rapidly within hours of application (Baillie et al. 2017). In this study, instream copper concentrations remained below New Zealand drinking water standards and posed a low risk to aquatic organisms, based on the exposure times to the concentrations measured in this study.

The harvesting phase of the forestry cycle has the greatest potential for adverse impacts on freshwater and marine environments, particularly clear-felling to stream margins (Baillie and Neary 2015, Quinn and Phillips 2016). Tree removal reduces rainfall interception and evapotranspiration, resulting in increased baseflows, flood peaks and water yields (Wood and Fahey 2006). Planted forests on steep slopes with highly erodible soils are vulnerable to increased run-off during rainfall events throughout their rotation, and surface erosion processes such as landslides and run-off from logging roads and landings increase sediment loss during and after harvesting. Sediment yields can increase up to five times from pre-harvest yield, and generally decline to pre-harvest levels within two to six years of harvesting. The increased stream power in post-harvest flow regimes can exacerbate channel bank and bed scouring, further increasing sediment yields and suspended sediment concentrations (Fransen et al. 2001, Phillips et al. 2005, Marden et al. 2006, Basher et al. 2011).

There is a 'window of vulnerability' for sediment loss and debris flows following forest harvest due to the loss of soil reinforcement from the harvested trees and the restoration of reinforcement by the next rotation of planted trees (Phillips et al. 2017). Debris flows may be generated by storms during these periods, and can transport large quantities of sediment and logging slash to waterways, and then to depositional zones such as floodplains and estuaries. The negative impacts of large debris flows include water quality and aquatic habitat degradation, damage to infrastructure (i.e., roads, bridges, culverts) and neighbouring properties, safety risks, clean-up costs, and adverse publicity, which affects the forest industry's social 'licence to operate' (Gray and Spencer 2011, Cave and Davies 2017).

The quantities of organic matter (predominantly logging slash) deposited in rivers during forest harvesting vary with harvest methods. The highest inputs are associated with cable harvesting where timber is extracted across the stream channel. The lowest inputs are associated with ground-based harvesting and 'stream-cleaning' (removal of logging slash from the stream channel) and the presence of vegetated riparian buffers (Thompson et al. 2009).

In warm climate areas, post-harvest maximum water temperatures may reach stress thresholds for sensitive invertebrate and fish species (Quinn et al. 1994, Richardson et al. 1994, Olsen et al. 2012). In turn, elevated water temperatures and instream respiration can cause decreases in average and daily minimum dissolved oxygen concentrations (Collier and Bowman 2003, Baillie et al. 2005).

Periphyton productivity generally increases after forest harvesting (Boothroyd et al. 2004, Death and Death 2006, Thompson et al. 2009, Reid et al. 2010), in response to the combination of increased solar radiation (Davies-Colley and Quinn 1998, Boothroyd et al. 2004, Quinn and Wright-Stow 2008), post-harvest nutrient pulses (Collier and Bowman 2003, Davis 2014), and increased water temperatures (Boothroyd et al. 2004, Baillie et al. 2005, Quinn and Wright-Stow 2008).

The complex interactions between the range of physical, chemical and biological disturbances associated with clear-cut harvesting up to the stream edge, can have significant impacts on aquatic invertebrate communities (Figure 3-8).

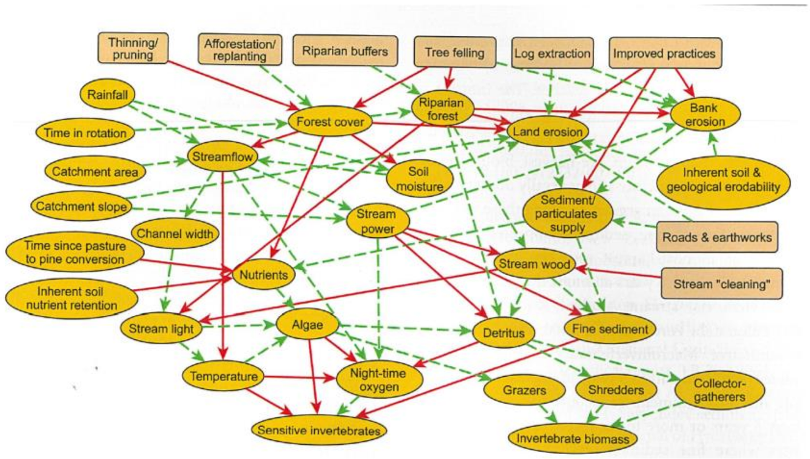

Figure 3-8: Causal pathways linking forestry practices and state variables to aquatic invertebrate biomass and community structure as indicators of ecological impact. Red arrows indicate negative relationships (an increase in the driver leads to a decrease in the response variable). Green arrows indicate positive relationships (an increase in the driver leads to an increase in the response variable). Forest management practices are in rectangles (from Quinn and Phillips 2016).

Harvesting generally causes aquatic macroinvertebrate community to shift from a predominance of taxa associated with mature forested streams, to communities similar to a predominance of taxa associated with pasture streams, with high proportions of pollution tolerant taxa such as Chironomidae (midges), Diptera (flies), Mollusca (snails) and Oligochaetes (worms) (Graynoth 1979, Death et al. 2003, Quinn et al. 2004, Baillie et al. 2005, Thompson et al. 2009, Reid et al. 2010). In most of these studies, the changes in community composition were accompanied by reductions in macroinvertebrate-based ecosystem health metrics.

The results of several New Zealand studies indicate that the adverse effects of forest harvesting can be reduced by retaining vegetated riparian buffers (Graynoth 1979, Boothroyd et al. 2004, Quinn et al. 2004, Thompson et al. 2009). Riparian buffers of sufficient width and height maintain shade at pre-harvest levels, minimise post-harvest increases in stream temperature and periphyton growth, and reduce sediment input to channels. Results from an intensive study of four catchments draining the Whangapoua Forest on the Coromandel Peninsula illustrate the effects of clear-cut harvesting to the stream edge versus retaining riparian buffers (Rowe et al. 2002, Boothroyd et al. 2004, Quinn et al. 2004). In this study, sites lacking riparian buffers were characterised by higher stream lighting wider channels, higher bank erosion, higher periphyton biomass, and lower macroinvertebrate taxon richness, lower EPT taxon richness, lower % EPT density, and lower macroinvertebrate IBI scores compared with sites with intact riparian buffers. Fish surveys at the harvested sites indicated that banded kokopu (*Galaxius fasciatus*) abundance was lower at sites lacking buffers than at sites with intact buffers, whereas redfin bully (*Gobiomorphus huttoni*) abundance was higher at sites lacking buffers.

In a second study of fish responses to forest harvesting, Graynoth (1979) reported a decline in the abundance of dwarf galaxias (*Galaxias divergens*) and longfin eel (*Anguilla dieffenbachi*) following clear-cut harvesting to the stream edge in the Golden Downs State Forest, Tasman District. The same harvesting activities were associated with localised mortality of brown trout (*Salmo trutta*) in the Motueka River, which was attributed to increased sediment and logging slash in the stream channel and elevated water temperatures and observed algal proliferations. There were minimal impacts of harvesting on fish fauna at an adjacent site where a riparian buffer was retained.

Stream size and portion of the catchment harvested are key factors influencing the effects of forest harvesting, and variation in these factors contributes to the variation reported in recovery trajectories for state variables and ecological values (Collier and Smith 2005, Quinn and Wright-Stow 2008). Recovery times following forest harvest and replanting vary from months to several years for most chemical, physical and biological variables and are typically near pre-harvest condition by the time of the next canopy closure (Baillie and Neary 2015, Quinn and Phillips 2016). However, some recovery times occasionally extend into the mid-rotation phase of the forestry cycle. For example, Quinn and Wright-Stow (2008) predicted that thermal regimes in streams 6-12 m wide would take around 12 to 16 years to recover to a pre-harvest state.

Apart from one case study that assessed the impact of harvesting best management practices on stream environments (Eyles and Fahey 2006; Appendix G Pakuratahi case study), there is limited information to date on the effect of forest industry standards (New Zealand Forest Owners Association 2007). Similarly, there is insufficient information to assess the effectiveness of recent National Environmental Standards for Plantation Forestry (New Zealand Government 2017) on reducing adverse effects of planted forests on freshwater and marine environments.

3.1.2 Flow regimes and water resources

A. Introduction

Hydrological regimes of rivers (i.e., the natural range and variation in flow) and other aspects of aquatic habitat such as substrate, water quality, and temperature regimes create a template for ecological processes (Resh et al. 1988; Poff et al. 1997) and maintenance of native biodiversity (Bunn and Arthington 2002). Developing a better understanding of the relationships between flows, river state variables and impacts on river values is essential for effective flow management (Jowett and Biggs 2008; Poff et al. 2010). Numerous studies of New Zealand river have contributed to our current understanding of these relationships (e.g., Mosley 1982, Snelder et al. 2011, Booker et al. 2014a, Greenwood and Booker 2016). Far fewer studies have identified the links between those relationships and land use.

B. The National Policy Statement for Freshwater Management and water resource use limits

The National Policy Statement for Freshwater Management (NPS-FM) is important when considering the influence of land use on flow regimes because it guides how regional councils (and others) define and consider management of flow regimes and water resources. This section first describes the need for setting water resource use limits under the NPS-FM. Definitions of the term "flow regimes" and hydrological indices used to describe flow regimes and their potential influence on in-stream values are then outlined. Potential influence of land use on flow regimes are then discussed. Land use effects are divided into those with direct and indirect influence on river flows. Direct influences on river flows are caused by water abstraction, impoundments or transfers. Indirect influences on river flows are caused by alteration of evaporation or runoff rates.

A need to set bottom lines for water use has arisen in New Zealand due to widely distributed abstractions, and to dams and diversions on larger rivers (Ministry for the Environment 2015). Rivers in New Zealand are generally short and steep, and rainfall-runoff is generally high (Tait et al. 2006). These conditions create flashy flow regimes with relatively little capacity to store or manipulate flows (Booker 2013). There is high demand for water use (Li et al. 2011) coupled with a lack of feasible engineering solutions to strategically design environmental flows, such as inter-basin transfers or dams with large storage capacities. Additional drivers for setting bottom lines on water use are the recognition of indigenous relationships and rights, and the need to use community-wide collaborative approaches to freshwater management (Tipa et al. 2016).

To address these issues, the NPS-FM includes a statement that recognises the national significance of fresh water and Te Mana o te Wai. In this case, Te Mana o te Wai represents the innate relationship between te hauora o te wai (the health and mauri of water) and te hauora o te taiao (the health and mauri of the environment), and their ability to support each other while sustaining te hauora o te tāngata (the health and mauri of the people) (Ministry for the Environment 2015, p27). The statement emphasises the importance of identifying, through the planning process, community and tāngata whenua (indigenous Māori people of New Zealand) values that will collectively recognise the national significance of fresh water and Te Mana o te Wai. The NPS-FM requires regional water management plans to establish freshwater objectives and enforceable water resource use limits in the form of water quantity limits for all bodies of fresh water. Limits on the maximum use of water resources must therefore be set to avoid over-allocation. The intention is that these limits provide clarity regarding water availability for public, industrial, and agricultural uses, whilst also ensuring protection of social, Māori and environmental values (Ministry for the Environment 2015). Several key concepts are outlined in the NPS-FM relating to water quantity resource use limits to be set. A freshwater objective is a statement describing a desired environmental outcome required to be met in a freshwater management unit. These objectives may be expressed at different levels of detail or precision. A freshwater management unit is the water body determined as the appropriate scale for setting freshwater objectives and limits for freshwater accounting and management. Over-allocation is defined as the situation where the water resource either has been allocated to users beyond a limit or is being used to a point where a freshwater objective is no longer being met. It should be recognised that it may be impossible or very expensive to definitively prove that freshwater objectives are no longer being met as a consequence of flow regime changes due to various considerations (Table 3-2).

Table 3-2: Considerations for quantifying the extent to which water quantity limits are protective of freshwater objectives. Derived from Booker (2018).

Consideration	Issues
Uncertainties in comparing how much water is actually being used versus consented to be taken	Inaccuracies in measuring takes Non-recording of takes for permitted activities Permits allowing water to be taken at a rate of less than 5 litres/second are not required to supply records of takes under the Resource Management (Measurement and Reporting of Water Takes) Regulations 2010 Inconsistencies in temporal resolution (15 minute, hourly, daily, monthly, annually) Lack of records for flow returning to a river Inaccuracies in estimating evaporation or runoff for indirect effects of land use change on flow regimes
Uncertainties in relationships between time-series of takes and time-series of river flows	Estimating quantity and timing of streamflow depletion from groundwater takes is uncertain Some abstracted water may augment river flows via unrecorded return flows (flows back to a river) or seepage (through inefficient irrigation practices)
Uncertainties in relationships between river flows and ecological attributes (e.g., periphyton, macrophytes, invertebrates, fish, birds) due to additional influences	Nutrients concentrations Sediment transport, suspended sediment concentrations Physical habitat and geomorphological template Dissolved oxygen Temperature Other contaminants Presence of invasive species (e.g., *Didymosphenia geminata*) Various biotic interactions, characteristics and processes such as trophic interactions (feeding and the food chain), resistance (ability not to change under stress) and resilience (ability to return to pre-stressed state)
Freshwater objectives may not be being met because ecological attributes can be stressed by many factors other than flow alteration	Any factor stated in Points 3 a-h above Naturally occurring low flows
Difficult in isolating which aspects of the flow regime are influencing freshwater attributes	The frequency and duration of both low and high flow events can influence ecological attributes Flow regimes can be described by a multitude of hydrological variables. Seasonal flow patterns will be important for some species, but not others
There is natural spatial variability in flows and the states of ecological attributes	The relative hydrological influence of a single take will diminish with distance downstream as tributaries add more flow to the river There may be critical locations such as spawning habitat or river mouth openings that more strongly influence a freshwater attribute than other locations Some locations have naturally occurring low flows, and therefore naturally stressed ecological states
There is spatial variation in freshwater values	Some attributes will be highly valued in some locations, but be less highly valued (or not relevant) in other locations. For example, several threatened native fish species are restricted to specific regions or catchments Alternatively, some species considered culturally important for food gathering may be important in some locations, but not in others

Under the NPS-FM, water quantity limits must comprise at least a minimum flow (Qmin) and a maximum allocation rate (ΔQmax) (Ministry for the Environment 2015). Qmin specifies the flow below which no further water is to be taken. ΔQmax specifies the maximum rate of abstraction. ΔQmax represents a limit to total allocation defined by the maximum rate of abstraction summed across all upstream abstractors. When these water quantity limits are enforced they have two consequences. First, the rate of abstraction at any point in time must never exceed ΔQ. Second, flow must not fall below Qmin unless this occurs in the absence of abstractions. Enforcement of these limits requires either full or partial restriction of abstractions at lower flows (Figure 3-9). A change in either Qmin or ΔQmax involves a three-way trade-off between: a) minimising hydrological impact on in-stream values; b) ensuring reliability of water supply; and c) allowing larger volumes of water to be abstracted. A change to either of Qmin or ΔQmax necessitates a change in these points. Minimising alteration of natural river flows always comes at the expense of reduced reliability or allowing smaller volumes to be abstracted.

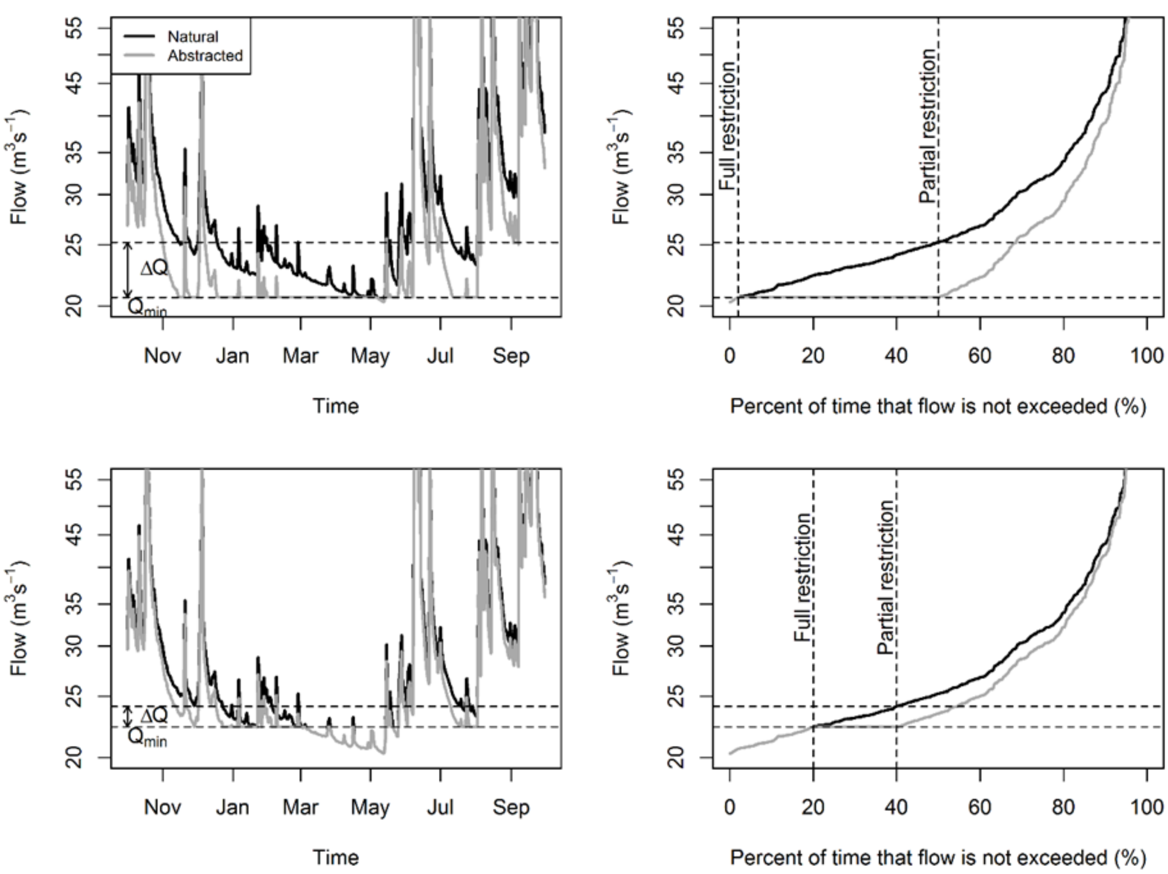

Figure 3-9: Example hydrographs (left) and flow duration curves (right) under natural and hypothetical resource enabling (top) and environmentally conservative (bottom) altered flow regimes, as defined by water resource use limits specifying a minimum flow (Qmin) and a total allocation (ΔQmax). From Booker (2018).

Setting limits for the use of a water resource ensures that a predetermined level of alteration to flow regimes is not exceeded, and they can provide information about the state of water availability for current and potential users through comparison between the limits and present allocation. The same comparisons can identify water bodies that are under-, fully-, and over-allocated. This information in important for water resources planning. However, no prescriptive guidelines have currently been published describing how over-allocation should be calculated. Neither the mathematical form of this calculation, nor the temporal and spatial scale at which it should be applied have been clearly described. The lack of prescriptive instructions allows for flexibility, for example, “the geographical and temporal definition of over-allocation will relate to the detail of the freshwater objective for a particular freshwater body” (Ministry for the Environment 2011, p24). While the NPS-FM clearly calls for over-allocation to be avoided, the process applied to achieve this goal has not yet been defined.

C. Definition of flow regimes

“Flow regime” is a phrase often used to describe the collective properties of river flow time-series. River flow is widely viewed as a “master variable” that controls or influences channel geomorphology, sediment, habitat size, physical habitat templates, disturbance regime, food resources and water quality including nutrients, dissolved oxygen and water temperature (Poff 1997). Ecological and evolutionary processes in river ecosystems are therefore highly influenced by their flow regimes.

River flows vary in time and space in response to variation in weather and climatic, topographic, geological and vegetation conditions, and to human activities such as water abstraction and impoundment. These fluctuations can be characterised using a suite of hydrological indices, each of which quantifies a different aspect of the flow regime. No specific suite of indices is applicable to all rivers, because there are major differences between rivers in total flow ranges, in temporal patterns of flow fluctuations, and in the ecological effects of flow.

D. Hydrological indices

Various hydrological indices can be calculated from flow time-series. Each index is designed to quantify a particular characteristic of the time-series. For example, application of the Indicators of Hydrologic Alteration (IHAs: Richter et al. 1996) has been an approach used to characterising natural flow regimes and quantifying human modification of flows. The IHAs originally comprised 32 hydrological indices (and 32 associated measures of variation) (Richter et al. 1996). Richter et al. (1996) suggested that together these 32 indices provided information on the ecologically relevant features of surface and ground water hydrology, but the number of indices was later expanded to 51 (and 51 associated measures of variation) (Table 3-3; Richter et al. 1997). In subsequent studies, further hydrological indices have been suggested as influencing or being correlated with a variety of ecological states and processes. For example, in New Zealand the Ministry for the Environment periphyton guidelines (Ministry for the Environment 2000; Biggs 2000) link periphyton biomass with nutrient concentrations and a hydrological index defined as the frequency of high flow events exceeding three times the median flow (FRE3). Other studies have used different multiples of the median flow to identify the frequency of events exceeding different ecologically relevant thresholds. For example, Townsend et al. (1997b) calculated FRE5 when comparing various metrics of disturbance to macroinvertebrate species traits and species richness. Indices characterising low flow have also been linked to invertebrate communities (Booker et al. 2014a), fish passage (Mosley 1982), and availability of suitable physical habitat for fish (Snelder et al. 2011). Low flow can be characterised by mean annual low flow (MALF) or mean flow in the naturally driest calendar month (Qlow).

Table 3-3: Example hydrological indices (after Richter et al. 1997).

Group	Parameter description	Abbreviation
1) Magnitude of monthly flows	Mean value for each calendar month Median value for each calendar month	e.g., MeanSep e.g., MedianSep
2) Magnitude and duration of annual extremes	Annual minima 1-day means Annual minima 3-day means Annual minima 7-day means Annual minima 30-day means Annual minima 90-day means Annual maxima 1-day means Annual maxima 3-day means Annual maxima 7-day means Annual maxima 30-day means Annual maxima 90-day means Number of zero flow days Base flow index	Mean1DayFlowMins Mean3DayFlowMins Mean7DayFlowMins Mean30DayFlowMins Mean90DayFlowMins Mean1DayFlowMaxs Mean3DayFlowMaxs Mean7DayFlowMaxs Mean30DayFlowMaxs Mean90DayFlowMaxs ZeroFlowDays BFI
3) Timing of annual extremes	Julian day of annual maximum Julian day of annual minimum	JulianMin JulianMax
4) Frequency and duration of high and low pulses	Number of low pulses within each water year Mean duration of low pulses Median duration of low pulses Number of high pulses within each water year Mean duration of high pulses Median duration of high pulses	nPulsesLow MeanPulseLengthLow MedianPulseLengthLow nPulsesHigh MeanPulseLengthHigh MedianPulseLengthHigh
5) Rate and frequency of flow changes	Mean of all positive differences between daily values Median of all positive differences between daily values Number of all positive differences between days Mean of all negative differences between daily values Median of all negative differences between daily values Number of all negative differences between days Number of hydrologic reversals	meanPos medianPos nPos meanNeg medianNeg nNeg Reversals

Hydrological indices are often calculated from mean daily flow time-series, but can also be calculated from flow time-series with more frequent observations such as 15-minute flow data. Observed flow time-series are only available where a gauging station with a relationship between stage height and water discharge is known. Alternatively, hydrological models have been used to calculate estimated flow time-series or particular hydrological indices at ungauged sites across New Zealand (Booker and Woods 2014; McMillan et al. 2016). For most hydrological indices it is possible to compute a value for each year of record. Both the central tendency and inter-annual variation of the index can be subsequently calculated. For example, a value such as the low (minimum) flow can be computed for each year of an entire multi-year time-series. Both the standard deviation and the mean of the annual low flows (known as MALF) can therefore be reported because both the long-term mean and inter-annual variation in low flow are potentially important characteristics of flow regimes. Inter-annual variation may be of interest because, in a natural situation, it can be used to characterise the range of natural variation experienced by organisms living in the river. Long-term means may be of more interest when comparing flow regimes from sites across the landscape or when comparing scenarios of flow alteration.

The length of flow time-series and the degree of hydrological alteration in a catchment must be carefully considered when calculating hydrological indices. Record length is important because there may be large between-year differences in calculated annual values, and because the inter-annual distribution of particular hydrological indices may have skewed rather than normal distributions. For example, if calculated from a relatively short record, MALF can be strongly affected by the inclusion of one particularly low flow year. Furthermore, hydrological indices cannot be considered to be stationary; temporal patterns in climate and/or landcover can cause hydrological indices to be trended (steady or abrupt change through time) or auto-correlated (cyclical patterns in time). Evidence has been given for inter-decadal patterns in some, but not all, indices for particular regions of New Zealand but not others (e.g., McKerchar and Henderson 2003; Booker 2013). Strong inter-annual and intra-annual variability in river flows has also been demonstrated across New Zealand, with inter-annual patterns exhibiting temporal auto-correlation (Booker 2013; Caruso et al. 2016). This means that each year is not independent of the previous year. Caruso et al. (2016) demonstrated that inter-annual patterns in floods and droughts were not common across the regions of New Zealand; low flow years in Hawke's Bay did not necessarily coincide with low flow years in Canterbury. This is important when considering whether space-for-time-substitution can be used to assess land use change impacts on flow regimes.

Given the large number of indices, hydro-ecological researchers and river managers are therefore confronted with the task of having to choose among a large number of competing hydrologic indices to reduce computational effort and index redundancy prior to characterising natural flow regime, assessing land use change impacts on flow regimes, or setting environmental flow standards. This task is further complicated because many indices calculated from natural (and altered) flow time-series are highly correlated (Figure 3-10). To help the process, hydrological indices may be organised into groups designed to represent similar aspects of flow regimes. For example, Olden and Poff (2003) grouped 171 published hydrological indices into five categories: magnitude (94 indices), frequency (14 indices), duration (44 indices), timing (10 indices) and rate of change (9 indices).

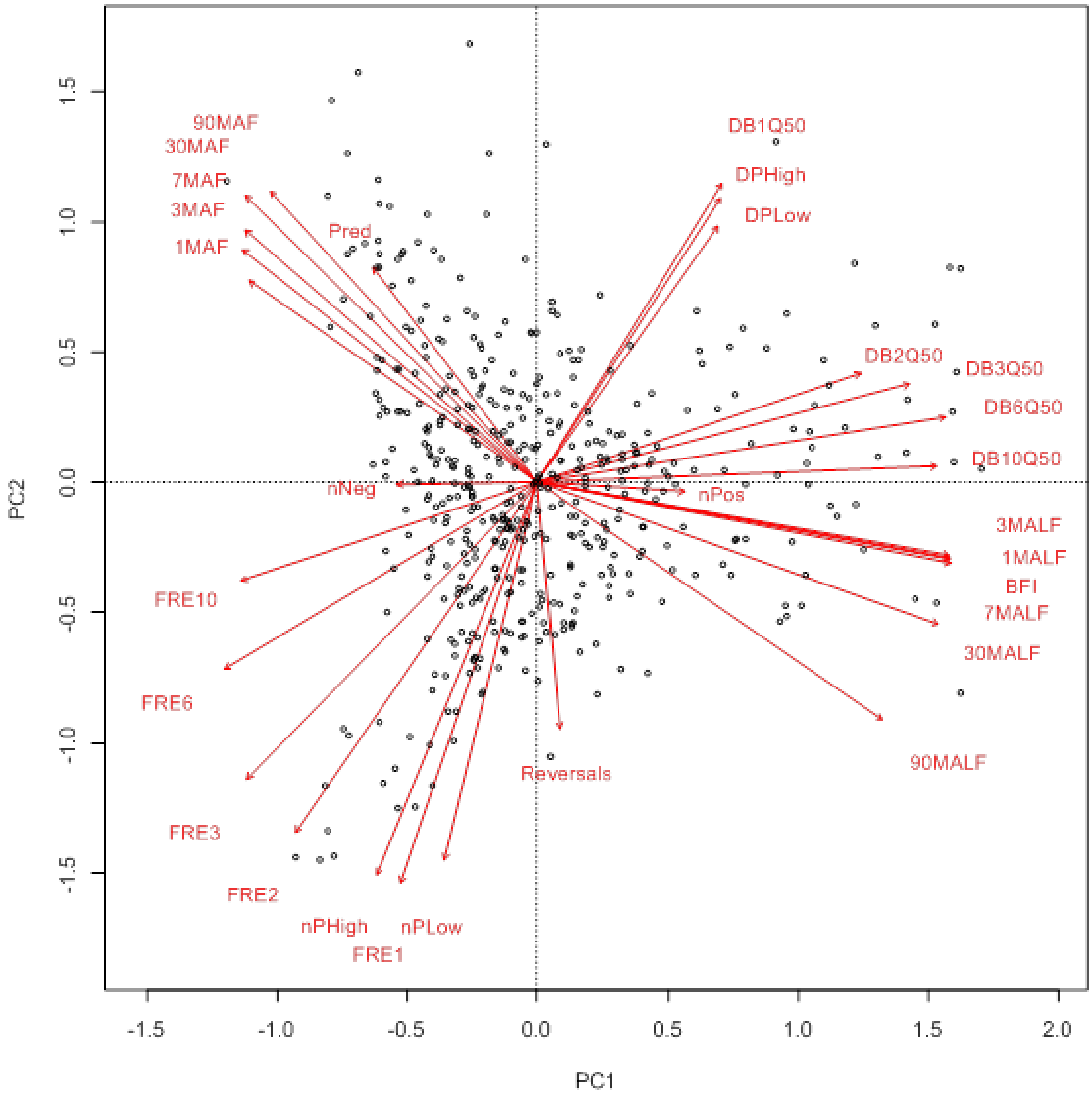

Figure 3-10: PCA ordination of 25 hydrological indices, using daily flow records from 438 river gauging stations distributed across New Zealand. Circles: scores of gauging stations on two principal components. Arrows: strengths of correlations between indices and the first two principal components.

An updated method for identifying an ecologically meaningful sub-set of hydrological indices was suggested as part of the Ecological Limits of Hydrological Alteration (ELOHA) approach (Poff et al. 2010). In this approach hypothesised flow-ecology relationships developed through a theoretical basis are used to identify ecologically relevant aspects of flow regimes at a regional scale. Empirical relationships developed in rivers with similar hydrological (and possibly geomorphological) conditions are then used to quantify these relationships. Hydrological alteration and ecological status are then observed to update and inform these hypotheses and flow-ecology relationships. Some aspects of this approach match reasonably well with the effects-based approach applied in New Zealand and outlined within the Resource Management Act (RMA), Ministry for the Environment flow setting guidelines (Ministry for the Environment 1998, 2008), and NPS-FM. Poff et al. (2010) suggested that three criteria should be used to select hydrological indices for river flows (e.g., when building a river classification). First, the indices should collectively describe the full range of natural hydrologic variability, including the magnitude, frequency, duration, timing and rate of change of flow events. Second, the indices should represent hydrological phenomena that affect ecological conditions, will therefore be important in assessing ecological responses to hydrologic alteration. In the New Zealand context, this criterion can be interpreted to mean that there must be a measurable link between the hydrological index and an objective that has been set for an in-stream value (e.g., Table 3-2). Third, the indices should be amenable to management actions, so that environmental flow standards can be developed using the indices.

E. Relating hydrological indices to water resource use limits

Booker (2018) demonstrated how water resource use limits can be related to ecologically-relevant hydrological indices. Mean flow in the naturally driest calendar month (Qlow) was calculated to represent conditions in a low flow period because it has been linked to invertebrate communities (Booker et al. 2014a). The frequency of events exceeding three times the natural median flow (FRE3) was calculated because it has been linked to periphyton removal in gravel bed rivers (Biggs 2000). Qlow and FRE3 were calculated under natural conditions and for permutations of Qmin and ΔQmax. The results showed how Qmin and ΔQmax interact to influence Qlow (Figure 3-11). FRE3 was solely influenced by ΔQmax. These relationships varied between two example sites due to different hydrological regimes. This demonstrates the difficulties of comparing limits and potential influence of land use change between sites; and indicates that similar changes in land use change may not result in similar hydrological alterations at different sites.

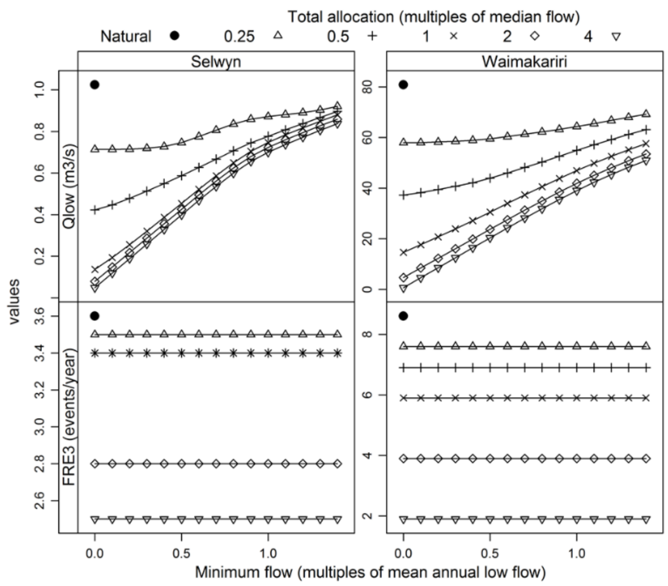

Figure 3-11: Changes in two ecologically relevant hydrological variables with changes in water resource use limits at two sites. FRE3 is the frequency of events exceeding three times the median after having applied a 5-day window. Qlow is mean flow in the naturally driest calendar month: February for Selwyn River at Coes Ford; April for Waimakariri River at Old Highway Bridge. Calculation period was 1985-1994 inclusive. From Booker (2018).

F. Direct effects of land use: water abstraction, impoundments and transfers

There is high demand for water use across New Zealand with respect to pasture production (Li et al. 2011). An excess of evaporation over rainfall for much agriculturally suitable land, coupled with economic drivers, has led to large areas of irrigated land. This area has been estimated to be 800,000 ha across New Zealand (Dark and Kashima 2017).

Water abstraction for some activities such as domestic use, fire-fighting and farm animal stock drinking is permitted without consent under New Zealand's Resource Management Act. Consents are also often not required for certain purposes if rate of take is very low. However, most abstractions associated with land uses require consents from the regional authority. This means that many land use changes that have prompted water abstractions will be associated with consents. National analysis of these consents indicates that irrigation causes the largest demand for consumptive water whether measured by either consented volume or number of consents, although industrial, hydropower, drinking and other uses are also present (Figure 3-12; Booker 2018). The results showed that both maximum instantaneous rate of take and maximum annual take volumes were useful for describing the influence of consents (and therefore land use associated with irrigation) on flow regimes around the country. When mapped onto the national river network, it could be seen that the effects of both groundwater and surface water abstractions are likely to be widespread across New Zealand. Results showed how surface water consents are distributed around the country whereas groundwater consents are concentrated in particular zones where aquifers are present and groundwater demand is high. Canterbury has by far the greatest number of consents. Irrigation has by far the greatest potential to cause widespread alteration of river flows. Impoundments for hydro-power schemes (or hydro-power in conjunction with storage of water to be used for irrigation) have potential to cause large changes to flow regimes for a few rivers.

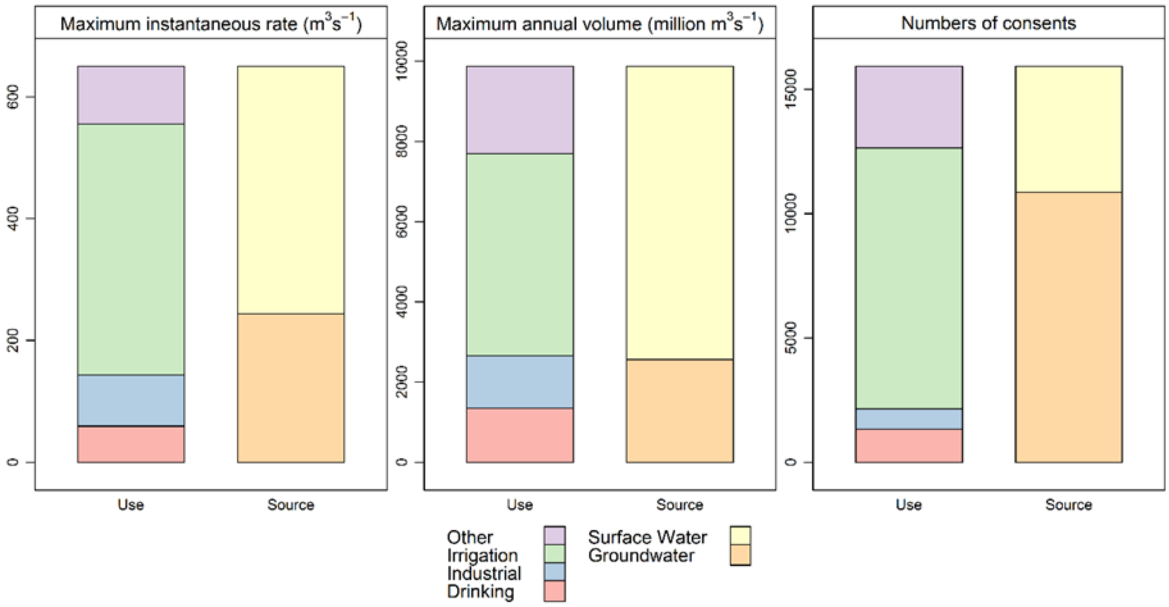

Figure 3-12: Consented abstractions summarised across New Zealand.

Because responsibility for water management is devolved to regional councils there is no nationally consistent format that must be followed when specifying consent conditions across New Zealand. This has led to a proliferation of ways in which consents act to control abstractors. This freedom has allowed flexibility within the consenting process. It also causes difficulties when assessing the impacts of land use on flow regimes (Booker 2018). This is because some consents have conditions that specify relatively high minimum flows, whereas other consents will allow continuous abstraction. Where minimum river flows (or minimum groundwater levels) are specified the abstractions are restricted, or must cease, during low periods. These consent conditions are specifically therefore designed to reduce the hydrological impact of these abstractions at lower flows or to prioritise some water users over others. There is also considerable uncertainty associated with calculating the extent and magnitude of streamflow depletion from groundwater abstraction (Barlow and Leake 2012), especially in small low-lying catchments with alluvial geologies (Nyholm et al. 2003) or in complex geological settings with multiple aquifers and aquitards (Zlotnik and Tartakovsky 2008). The method applied by Booker (2018) to estimate potential stream depletion from consented takes was deliberately conservative, and assumed a worst-case-scenario for streamflows; i.e., that all groundwater abstractions result in streamflow depletion. That method was designed to be transparent and easily applied given data availability at the national scale. The method therefore lacked detailed understanding of groundwater-surface water interactions gained through field experiments (e.g., Hunt et al. 2001) or detailed modelling (e.g., Hunt 2012).

Despite the complications mentioned above, the most likely impacts of irrigation takes on flow regimes are to reduce summer low flows, reduce low flow variability, and increase the severity of flow regime seasonality. A reduction in mid-range flushing flows (e.g., flows that mobilise fine sediment or remove growths of nuisance algae) can occur in situations with extremely high abstraction or where flood harvesting is used to fill off-channel water stores (Figure 3-11). The magnitude of effect will depend on several factors, including:

- consent conditions limiting the annual volume, instantaneous rate of take, or other restrictions;
- size of river (in terms of flows) being influenced;
- position of abstractions within the catchment; and
- the potential for cumulative effects of many abstractions.

Booker et al. (2014b) explored how, once rules to define a minimum flow and an allocation rate have been established, differences in implementation of these limits can lead to vastly different outcomes for both water users and the downstream flow regime. The effect of changing the number, spatial configuration and strategy for implementing water takes was explored by modelling sets of hypothetical water takes across an example catchment. Results indicated that, due to cumulative effects and the connected nature of river networks, some implementation strategies can ensure that specified limits are met at a downstream monitoring point, but may not guarantee that these limits are complied with elsewhere in the catchment. This work therefore demonstrated how impacts of abstraction (for irrigation) can vary greatly across a river network, even if the rate of take is constant in terms of litres per hectare; a medium rate of abstraction can have a very large influence on flows in a small river, compared to the same rate of abstraction from a larger river.

Cumulative impacts arise because the connected and hierarchical nature of river networks control how abstractions will combine to have cumulative effects; i.e. the sum of all abstractions accumulates in the downstream direction, but flow also naturally increases with distance downstream (this is not always the case as some rivers lose flow naturally; Larned et al. 2011). As a result, the value of supplying a cubic metre of water per second may be the same for abstractors in different locations within the same catchment, but the influence of a cubic metre of water per second for maintaining ecological values is highly influenced by the size and natural flow rate of the river reaches that are being depleted.

G. Indirect influences of land use: evaporation and runoff rates

Indirect influences of land use change on river flows are caused by alteration of evaporation rates or runoff rates. See Section 3.2 on urban streams for the effects of increased impermeable surfaces on flow regimes. Evidence from field studies (e.g., Fahey and Jackson 1997) and numerical modelling (e.g., Dymond et al. 2012) have both indicated that afforestation generally reduces river flows, and deforestation generally river flows. The results of field studies of afforestation and forest harvest effects are summarised in Section 3.1.1, above.

Dymond et al. (2012) applied the WATYIELD water-balance model developed by Fahey et al. (2010) to estimate runoff from various land cover classes over a range of rainfall levels and New Zealand soil types (Figure 3-13). Their results indicated that at 600 mm of annual rainfall, the water yield of grass is approximately twice that of forest, and for tussock it is approximately ten times that of forest. At 1000 mm of annual rainfall, the water yield of pasture is 70% more than forest, and for tussock it is approximately three times that of forest. At 2000 mm, the water yield of grass is 50% more than forest, and for tussock it is approximately twice that of forest. Dymond et al. (2012) suggested that these land-cover effects that need to be considered when developing policy concerning land-use change, especially in catchments that experience water shortages. Afforestation and destocking that results in the re-establishment of shrubland can reduce water yield.

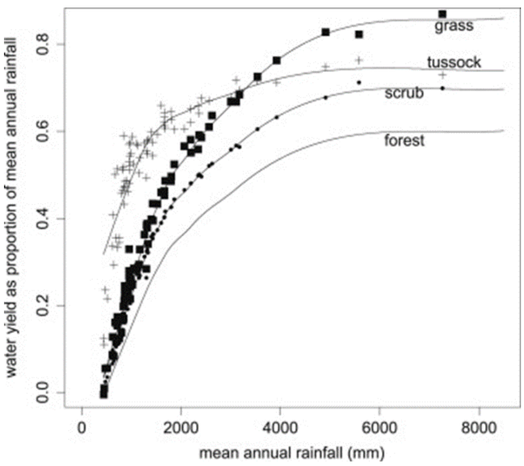

Figure 3-13: Water yield for basic land covers as a proportion of mean annual rainfall. Each point corresponds to a WATYIELD run for the most common soil type in a level II land environment (Leathwick et al. 2003), with the plotting position relative to the forest result for that soil. From Dymond et al. (2012).

The effects of wetland drainage on flow regimes in New Zealand has been poorly studied compared with the influence of afforestation and forest harvesting. Overseas studies indicate that flow regimes can change substantially following wetland drainage. For example, Holden et al. (2006) compared runoff production in intact and drained blanket peat catchments. They found that overland flow was significantly higher in catchments with intact wetlands, and throughflow higher in drained catchments.

3.2 Urban streams

Introduction

This section summarises published research findings on pressure-state-impact (PSI) relationships and trends relating to New Zealand's cities and towns. Urban land-use pressures can exert a change in a wide range of state variables relating to urban streams and rivers. Taken in combination, these changes in state have been found to result in marked impacts on the ecological functioning and composition of biological communities in urban streams, a condition that has been termed the 'urban stream syndrome' (Walsh et al. 2005).

The pressure-state-impact relationships associated with urban development can be conceptualised as follows (Suren and Elliott, 2004; Harding et al. 2016), summarized in Figure 3-14.

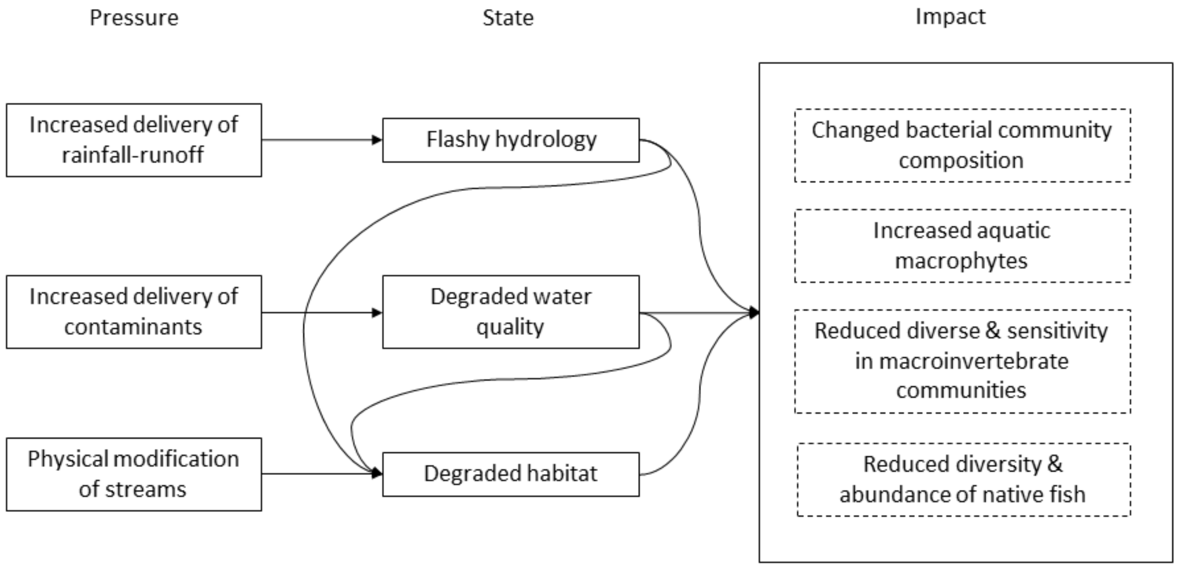

Figure 3-14

Urban development exerts the following pressures on urban waterways:

- Increased volumes of rainfall runoff (stormwater), delivered more quickly to streams, and reduced rates of infiltration and groundwater recharge. These changes result, firstly, from the replacement of pervious, vegetated land covers with impervious covers such as concrete, asphalt and roofing materials and, secondly, from the construction of piped drainage networks.

- Increased generation and delivery of a range of contaminants. These include: sediment, especially during the earthworks phase of urban development; ecotoxicants such as copper and zinc derived from traffic sources and building materials; and nutrients and pathogens discharged in wastewater overflows.

- Modification of stream and river channels and their margins. This includes the straightening and lining, for instance with concrete, of channels to improve conveyance of flows and the removal of riparian vegetation.

The resulting changes in the state of urban streams and rivers fall into three broad areas:

- Changed hydrology: in particular, urban development results in a 'flashy' hydrology characterised by an increase in the volume of storm flows and the frequency and magnitude of flood events, with a contrasting reduction in base flows.

- Changed water quality: urban streams are characterized by elevated concentrations of suspended solids, metals, nutrients and faecal indicator bacteria associated with stormwater discharges and wastewater overflows. Water temperature can also be elevated, partly in response to stormwater discharges generated from relatively warm impervious surfaces, but also influenced by a lack of riparian vegetation to cast shade. Dissolved oxygen concentrations can be low in urban streams, especially where elevated nutrient concentrations and lack of shade allow algae or aquatic macrophytes to flourish.

- Changed habitat: as a result of channel modifications, the geomorphology of urban streams can feature reduced diversity, both in terms of channel form and in substrate composition. Culverts, weirs and dams are also often constructed in urban streams. The removal or changes to the composition of riparian vegetation not only affects shading, but also the replenishment of woody debris in streams and the nature of the terrestrial-aquatic vegetated interface.

The impacts of these changes in state include:

- Changes to the composition of bacterial communities;

- Increased abundance of aquatic macrophytes (as noted above), including invasions of exotic species;

- Reduced diversity and/or abundance of benthic macroinvertebrates, with communities often limited to the more pollution-tolerant taxa; and

- Reduced diversity and/or abundance of native fish species, including those with a migratory component to their lifecycle, and the presence of exotic pest fish species.

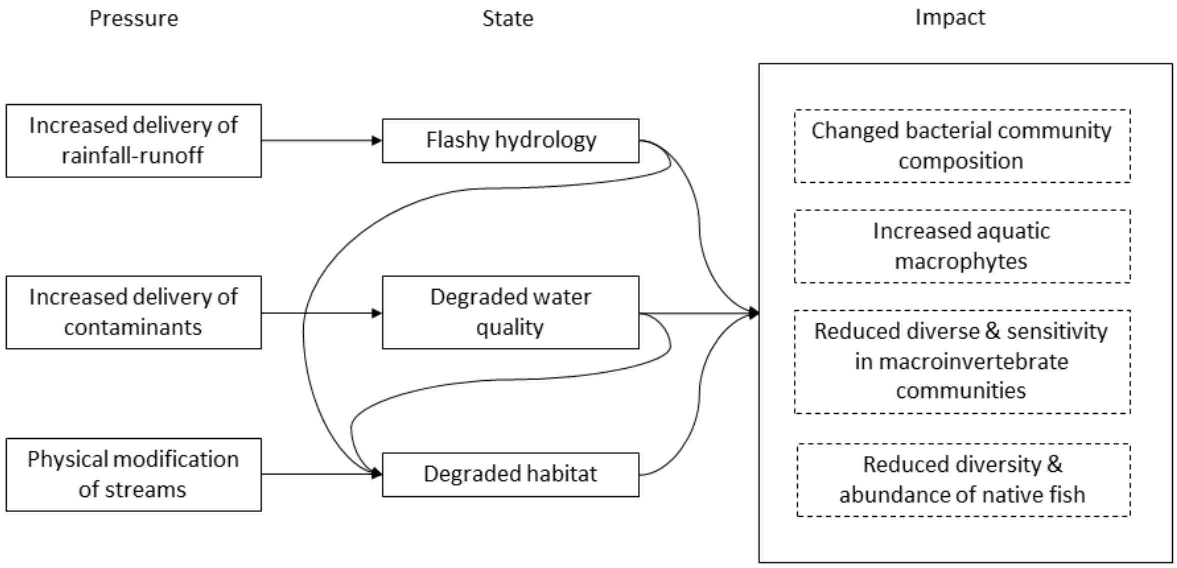

Figure 3-14: Summary of pressure-state-impact relationships affecting urban streams and rivers.

The following sections summarise studies that have investigated the influence of land-based pressures on urban rivers and streams from three complementary perspectives: those of land use, land cover and land management practices. The review focuses on the response of physical-chemical water quality variables and benthic macroinvertebrate community metrics, key indicators of state and impact, respectively. Where information exists, the review also describes studies of temporal trends in these state and impact variables.

Many of New Zealand's towns and cities are located on or near coastal and estuarine waterbodies. In some locations, coastal receiving environments receive urban-derived contaminants delivered via rivers and streams while in others, discharges occur directly from stormwater and wastewater networks. PSI relationships relating to urban land use effects on coastal water bodies are considered in Section 3.2.

Land use

This section describes the findings of studies that have assessed, firstly, the influence of urban land use as a single, broad land use category and, secondly, the influence of different types of urban land use on streams and rivers. It is relevant to note that several of the studies summarised below describe urban land use as a 'land cover' class based on, for instance, areas defined as 'built-up' in successive versions of the New Zealand Land Cover Database (LCDB). The built-up class in the LDCB provides a broad indication of the extent of an urban area, including impervious surfaces such as buildings, roads and paved areas and pervious vegetated areas (excluding parks, which is a separate land cover class). For this review, we have adopted the approach that estimates based on the LDCB are indicative of the extent of urban land use (being the aggregate of many cover types), rather preserving the 'land cover' terminology of the original studies. Our consideration of land cover relationships then focuses on the influence of aggregated impervious covers and specific cover types.

Urban land use as a single category

National scale studies

Ballantine and Davies-Colley (2014) reviewed the influence of land use on water quality monitored over the period 1989-2009 at 77 sites that make up New Zealand's National Rivers Water Quality Network (NRWQN). The majority of water quality variables were found to be strongly related to the proportion of urban land use in a catchment (taken from the LCDB2). Median values of nutrient concentrations (total P, DRP, total N and nitrate-N), conductivity, temperature, dissolved organic matter and turbidity were positively correlated with the percentage of urban land use, while water clarity was negatively correlated. Correlations were significant at the 99% level.

Larned et al. (2016) investigated the current state (2009-2013) and temporal trends (2004-2013) in New Zealand rivers from physical-chemical water quality and benthic invertebrate data collected at over 900 monitoring sites across New Zealand, the majority of which are operated by regional councils but also including the NRWQN sites. The proportion of urban land use in a catchment was assessed as being the sum of four urban cover types in the LCDB3 (built-up areas, urban parks, mines and dumps and transport infrastructure). Monitoring sites were classified as urban where urban land cover exceeded 15% of the catchment.

Median concentrations of nutrients (total P, total N, ammoniacal-N and nitrate-N) were highest at urban monitoring sites and exceeded ANZECC trigger values. Similarly, median *E. coli* concentrations were also highest at urban sites and exceeded the NPS-FM minimum acceptable state for primary contact at all urban sites. Median MCI scores were lowest at urban sites and corresponded with a 'poor' water quality grade. Consistent with these findings, the results of multiple regression analyses indicated that nutrient and *E. coli* concentrations were positively correlated with the proportion of urban land use in a catchment. MCI scores and visual clarity were negatively correlated with urban land use.

However, among other land uses, urban sites exhibited improving trends (>1.5% improvement p.a.) in total P, DRP ammoniacal-N and visual clarity. There was a small degrading trend (<1% p.a.) in MCI scores, with a stronger degrading trend (<1% p.a.) at urban sites on lowland streams and rivers.

In a focused analysis of the water quality state and trends in urban rivers and streams, Gadd (2016) reviewed data from monitoring sites in Auckland, Christchurch and the Wellington region. As well as nutrients and *E. coli*, this review also included data on dissolved copper and dissolved zinc, two ecotoxicants that are frequently found at elevated levels in urban stormwater. The earlier reviews of national datasets described above did not consider these variables because they are not routinely measured in non-urban water quality monitoring programmes. Urban land use areas in this study were derived from the LCDB4 and, in another contrast with the previous studies, were based on only the built-up and transport infrastructure cover types.

Analysis of water quality state was based on data collected at 50 or more sites (depending on the water quality variable) over the three years 2013 to 2015. Concentrations of dissolved copper and zinc, nitrate-N and, to a lesser extent, ammoniacal-N and *E. coli* were higher at sites with higher proportions of urban land use (Figure 3-15). Water quality guidelines and standards were found to have been exceeded at many sites: the NPS-FM A/B threshold for *E. coli* was exceeded at the majority of sites, and the default ANZECC dissolved zinc guideline for protection of 95% of species was exceeded at around half of all sites.

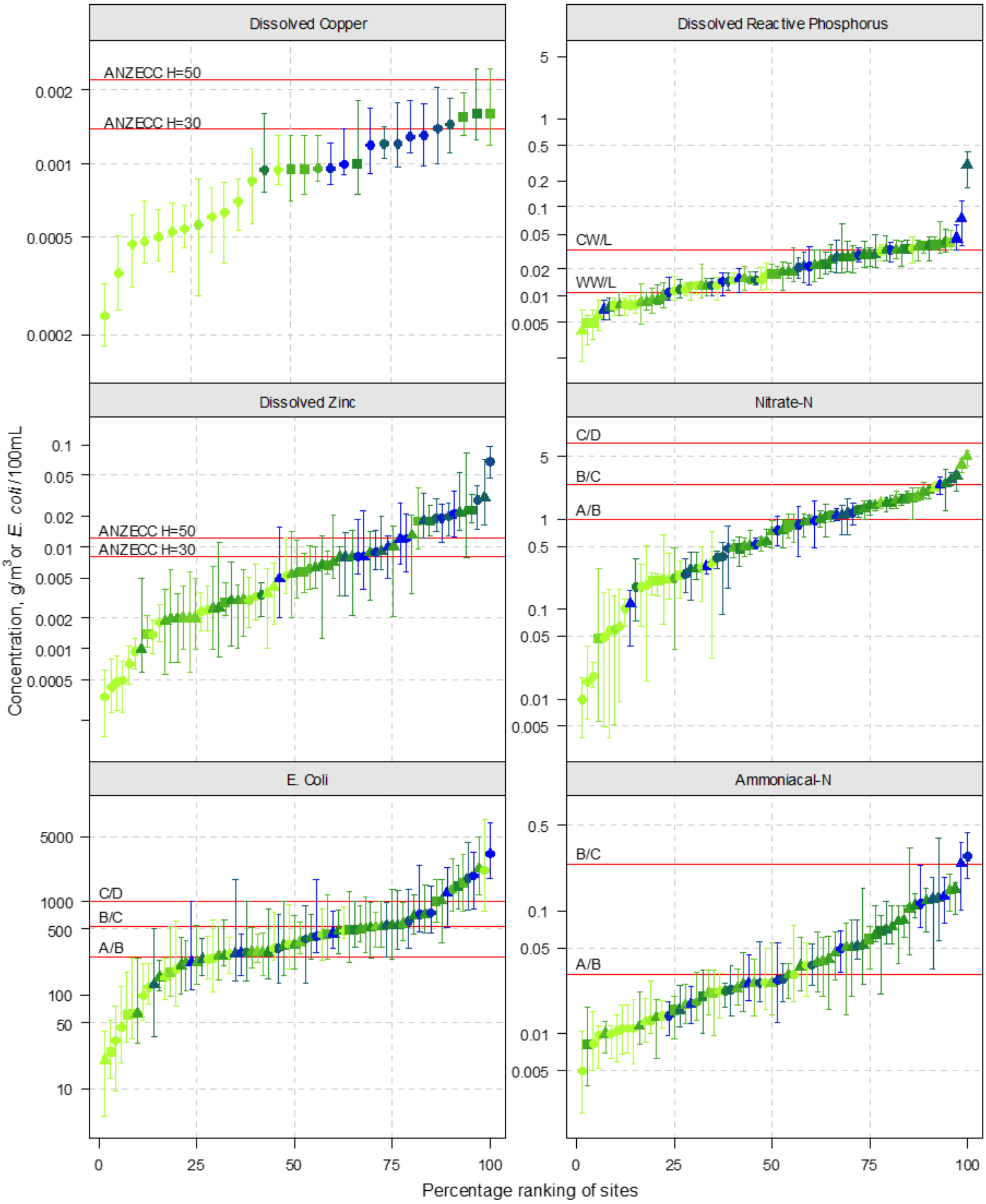

Figure 3-15: Urban stream water quality at monitoring sites in Auckland, Christchurch and Wellington ordered by median values and coloured by percent urban land co cover in upstream catchment. Colours indicate % urban land cover from light green (0%) to dark blue (100%). Marker indicates median value for each site and bars represent interquartile range (25-75%iles). Marker shapes indicate city as follows: Auckland: circle; Wellington: square; Christchurch: triangle. Red lines for nitrate-N, ammoniacal-N and *E. coli* represent NPS-FM NOF attribute thresholds between bands. Red lines for dissolved copper and zinc represent ANZECC

(2000) trigger values at hardness (H) of 30 or 50 g/m3 as $CaCO_3$. Red lines for DRP represent guidelines for Warm Wet, Lowland streams (WW/L) and Cold Wet, Lowland streams (CW/L). Note log-scale on y-axes. Reproduced from Gadd (2016).

Analysis of water quality trends was based on data collected over the eight years 2008 to 2015, but at fewer sites than in the analysis of state (reflecting a lack of long-term monitoring sites in some urban locations). For most variables, more sites were found to have indeterminate trends than either decreasing or increasing trends. However, DRP was found to have decreased at a majority of sites and, of sites that were not indeterminate, dissolved Zn, nitrate-N and ammoniacal-N decreased at more sites than they increased. The converse was true for *E. coli*, although the study noted that the increasing for *E. coli* trend tended to be more prominent at sites with relatively low proportions of urban land use. Analysis of long-term (20 year) trends at five Auckland sites generally confirmed the results of the 8-year analysis, although two sites exhibited an increasing 8-year trend (one each in dissolved zinc and nitrate-N) that contrasted with decreasing 20-year trends.

Regional Council State of the Environment Reporting

Several regional councils have reported on relationships between urban land use extent and the results of SoE monitoring programmes. The following summaries are taken from council reports.

Stevenson et al. (2010) assessed the state and trends of river water quality in the Canterbury Region. Data collected at 167 sites over the period 2002-2008 were used to assess the state of physical-chemical water quality. Sites classified as "Spring-fed Plains – urban" (17) had: the lowest median concentrations of DO; among the highest median concentrations of DIN (along with non-urban spring-fed streams of the Canterbury Plains, these were at 'enriched' or 'excessive' levels); and the highest concentrations of nitrate-N, DRP and *E. coli*. The study also analysed trends in water quality at 82 sites over the 12-year period commencing in 1997 and found increasing trends in ammoniacal-N at a number of urban monitoring sites. In contrast, NO_x-N decreased at several of the urban sites while DRP increased at some and decreased at others.

Perrie et al. (2012) assessed the state and trends of river water quality and ecology in the Wellington Region. Data collected at 55 sites (10 for metals) over the period 2008-2011 was used to assess the current state of physical-chemical water quality. Best on the land use classification applied in the River Environment Classification (REC), urban sites had the highest concentrations of nutrients (NO_x-N and DRP) and *E. coli*. Results for metals were not grouped by land use, reflecting the limited number of sites at which they were measured. Applying Greater Wellington RC's Water Quality Index (WQI), which combines median values of six key water quality variables (Perrie 2007), the study rated five of seven urban monitoring sites as having 'poor' water quality, with the two remaining rated as 'fair'. Applying the Canadian Council of Ministers for the Environment (CCME) WQI gave similar results. MCI scores, assessed from surveys at the same 55 sites over the period 2009-2011, fell in the "fair" quality range for the majority of urban sites, with one in the "poor" range. Urban sites had the lowest mean MCI score of any land use type.

The Wellington Region study analysed trends in physical-chemical water quality over the period 2006-2011 and found decreasing concentrations of nutrients (nitrate-N, ammoniacal-N, total N, DRP and total P) at some urban sites and no trends at others, but there were no urban sites with increasing trends of these variables. Despite decreases, however, concentrations at many of these sites still exceeded ANZECC guideline values. In contrast to the improvement in water quality, trends in macroinvertebrate metrics across the region were almost all negative, and these included sites dominated by urban land use.

Holland et al. (2016) reported on the state of river water quality in the Auckland region. Monitoring data from 36 monitoring sites was used to calculate the CCME WQI. On average, the WQI score for urban sites over the period 2012-2015 was the lowest of any land use type, giving an overall class of "fair". Buckthought and Neale (2016) conducted 10-year trend analysis (2005-2014) on the data from 23 of the Auckland river water quality monitoring sites and 6-year trend analysis on the data from a further six sites. The study reported significantly decreasing trends in total and dissolved copper and lead concentrations in data from urban sites as a whole. The same was not true for zinc concentrations, although there were decreasing trends at individual urban monitoring sites.

Young et al. (2013) reviewed continuous stream temperature monitoring data collected over the period 2010-2012 at seven Auckland SoE monitoring sites. Daily mean and maximum temperatures were higher at four urban streams than at streams in a pastoral and two forested catchments. Two of the urban streams had markedly high maximum temperatures and daily temperature fluctuations than either the rural or other urban streams. This was attributed to the relatively low level of shade cover (around 20% compared with 40% or greater at the other sites) and greater extent of channel modification (such as concrete-lining) at these two sites.

Neale et al. (2017) assessed the state of river ecology in the Auckland Region from the results of macroinvertebrate sampling at 71 sites over the period 2011-2013. Sites with at least 10 years of data (51 sites) were used for trend analyses. Sites in urban catchments had the lowest taxon richness, MCI scores, and %EPT metrics. MCI scores at 83% of the urban sites were in the "poor" quality class, compared with 30% for the region as a whole. However, MCI scores at low scoring urban sites were found to have improved. The authors noted that an improvement at a highly impacted site could be explained by the chance addition of taxa starting from a very low baseline MCI score.

City and catchment scale studies

Christchurch City Council operates an extensive river water quality monitoring programme, and produces annual reports on state and trends (e.g., Margetts and Marshall, 2016). While the analysis of the monitoring data does not explicitly consider relationships with the proportion of urban land use upstream of each monitoring site, overall results indicate poorer water quality in the urban Heathcote, Styx and Avon catchments than in rural areas such as the Otukaikino catchment.

Hall et al. (2001) compared water quality variables and macroinvertebrate community composition at urban, agricultural and native bush sites in the Water of Leith stream catchment near Dunedin. Urban sites had the highest nitrate and faecal coliform concentrations, the lowest mean abundance levels of EPT taxa and the highest mean abundance of pollution tolerant Oligochaeta (segmented worms). Reflecting these findings, the urban sites had the lowest MCI and QMCI scores.

Collier et al. (2009) monitored macroinvertebrate communities at 28 urban and 19 peri-urban (rural) streams sites in and around Hamilton. Both groups of sites were dominated by pollution-tolerant taxa and there was no statistical difference in MCI scores between them.

Moffett et al. (2015) used population density as a measure of urban intensity in a study nutrient processing in 24 stream sites in Auckland and Christchurch (spring and summer 2012-13). The authors found statistically significant positive relationships between urban intensity and concentrations of DIN and ammoniacal-N. Nutrient concentrations in highly urbanized catchments were up to 10,000 times higher than in native forest catchments.

Stansfield (2016) reported on water quality and ecology at 19 stream sites in west Auckland's Project Twin Streams (PTS) catchments. Sites were assigned to urban, pastoral and forest groups according to their classification in the REC. Relative to the other land uses, the water quality of urban stream sites tended to feature: high turbidity, temperature and concentrations of suspended solids, *E. coli*, nutrients, dissolved metals and sediment metals; and low concentrations of dissolved oxygen. Urban streams tended to have the lowest number of taxa, MCI scores and EPT metrics. MCI scores at the urban sites were indicative of moderate to severe pollution.

Specific urban land uses

In an early New Zealand study of urban water quality, Williamson (1985) conducted water sampling in three urban catchments, one being in a residential area of Hamilton and the others on Auckland's North Shore which, while also being predominantly residential, contained areas of commercial, industrial and developing land. Concentrations of suspended solids, DRP, total P and three metals (chromium, nickel and zinc) were significantly lower at the sampling site in the Hamilton residential catchment than at those in the mixed land use catchments. Other variables (COD, organic N, lead, and VSS) were not significantly different between the sites and the author concluded that, overall, geographical differences were more important than urban infrastructure in determining variations in stormwater quality in these catchments.

Subsequent to the enactment of the Resource Management Act in 1991, the city-wide effects of stormwater discharges have come under closer scrutiny and, accordingly, there has been a growing need for locally-based information on stormwater quality and its impacts on receiving environments. In Auckland, this need prompted the most extensive data collection exercise to characterise urban stream and stormwater quality yet undertaken in New Zealand. As part of its Integrated Catchment Study (ICS, Sharman et al. 2006), Auckland City Council funded a programme of storm event-based automatic sample collection from the city's stormwater network and urban streams. Sites were selected so as to be able to characterise stormwater quality discharged from areas within three broad land use categories: residential, commercial and industrial (Timperley et al. 2005). The results indicated that copper concentrations were similar across all three types of land use (Figure 3-16). In contrast, zinc concentrations were elevated in stormwater from industrial catchments, reflecting the widespread use of galvanised steel roofing on industrial buildings (Figure 3-16). The effects of steel roofing are discussed in detail in the section on specific urban land covers below.

In more recent studies conducted in Christchurch, samples of stream sediments were analysed for a range of contaminants, including metals commonly present in stormwater (Gadd and Sykes 2014, Gadd 2015). Concentrations of copper, lead and zinc in samples collected at sites in the Avon River catchment were found to be generally higher at locations associated with predominantly commercial and industrial land use than in residential areas (Figure 3-17). In contrast, sediment metal concentrations in samples collected at sites in the Heathcote River catchment were found to be unrelated to land use. The authors noted that this may have been partly because all urban sites included at least some residential land use (i.e., there were no purely commercial/industrial sites) but also attributed the results to the limited number of sampling sites and significant sediment quality variation within each land use group.

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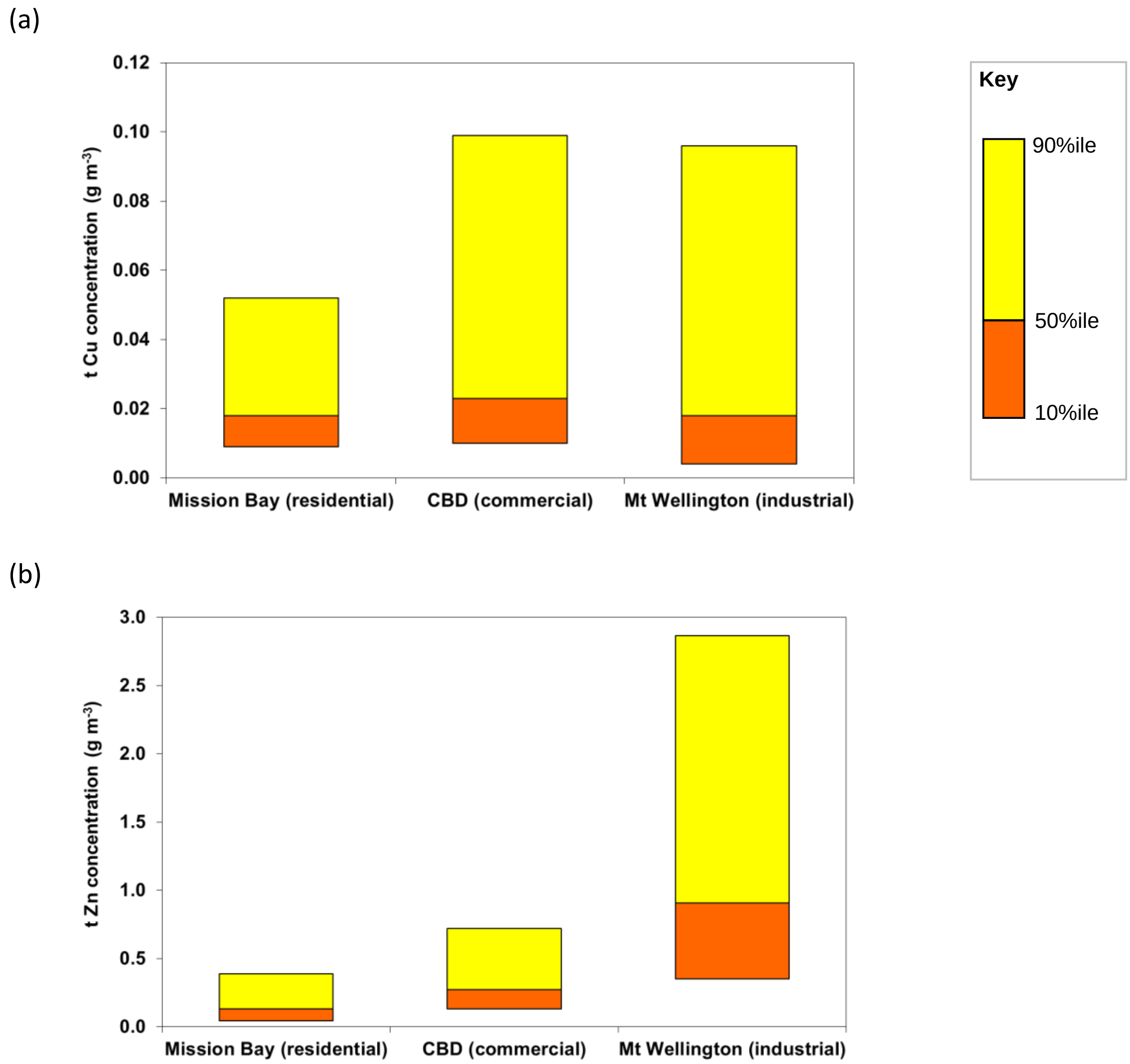

Figure 3-16: Concentrations of (a) total copper and (b) total zinc measured in samples of stormwater collected at residential, commercial and industrial sampling sites in Auckland as part of the Integrated Catchment Study. From Sharman et al. (2006).

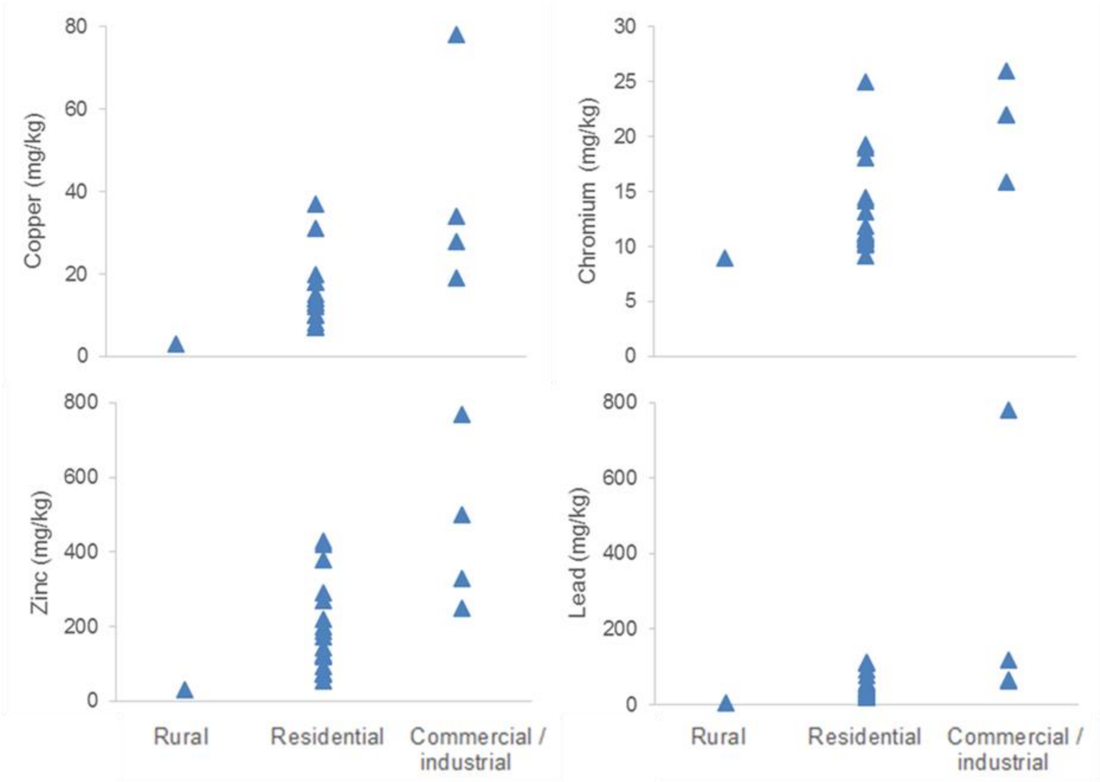

Figure 3-17: Metal concentrations in stream sediment samples collected from rural, residential and commercial/industrial sites in the Avon River catchment. Modified from Gadd (2014).

In another Christchurch study, Murphy et al. (2014) investigated atmospheric sources of stormwater contaminants by sampling runoff from concrete boards deployed in areas of three different land use (residential, industrial and airport "airside" land). Pollutant loadings for all contaminants (TSS, arsenic, chromium, manganese, nickel, lead, strontium and zinc) were significantly higher in the industrial area then in the residential and airside areas, a finding that was attribute to topography rather than land use. Similarities in the temporal distribution of contaminant loadings and in the ratios between different metals indicated the likelihood of a common, but uncertain, source of atmospheric pollutants that was unrelated to land use.

In addition to comparisons across broad urban land use types, other studies have investigated the quality of stormwater from specific, and significant, sources within the urban landscape. These included other land uses (roads[10]) as well as specific land cover types (roofing materials, see Section Specific urban land covers). An investigation of road runoff quality was conducted over two periods of sampling at Richardson Road on the Auckland Isthmus (Timperley et al. 2005, Moores et al. 2009), providing information on the contribution of roads to city-wide loads of suspended sediments, copper and zinc. Roads were found to be the single most important source of copper, although a large proportion of catchment copper loads were unaccounted for (Figure 3-18). While a subsequent assessment was able to account for a higher proportion of the copper mass budgets, a relatively

[10] Roads are a land use, rather than a land cover, as they can be constructed of different materials (asphalt, concrete, gravel) and can include vegetated margins and median strips.

large component remained unaccounted for, the most likely reason considered to be uncertainty around estimates of copper deriving from vehicle brake pad wear (Kennedy and Sutherland 2008).

[1] Roads are a land use, rather than a land cover, as they can be constructed of different materials (asphalt, concrete, gravel) and can include vegetated margins and median strips.

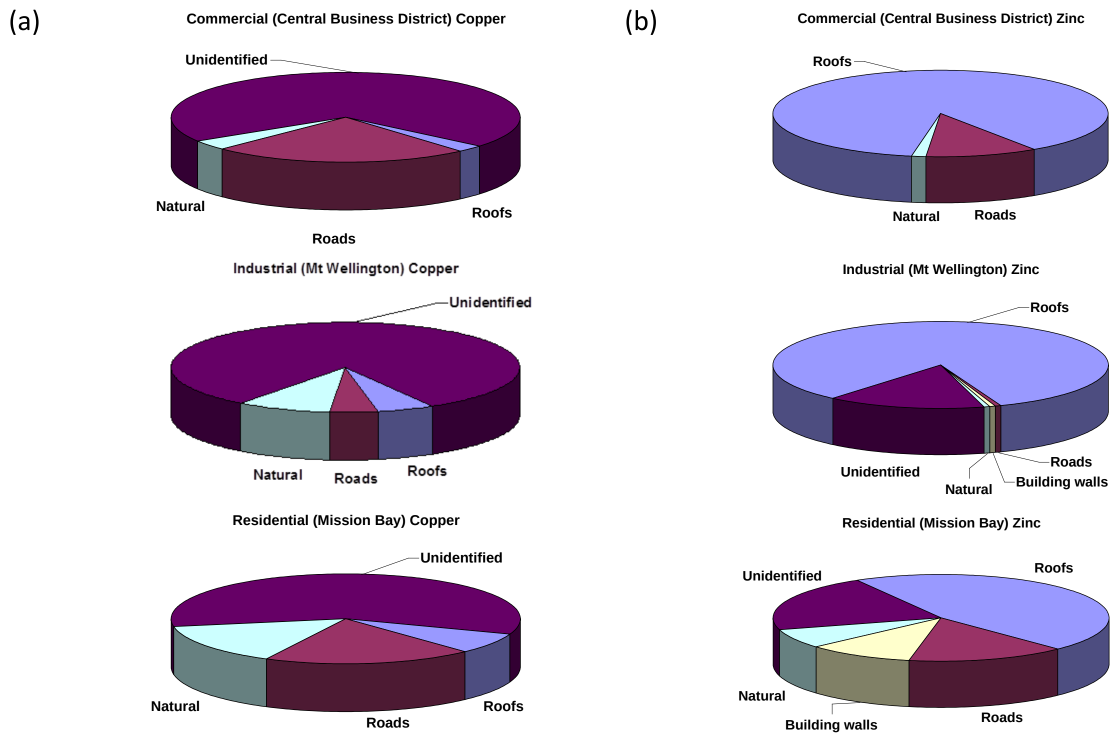

Figure 3-18: Mass budgets for (a) copper and (b) zinc estimated for commercial, industrial and residential stormwater catchments. Reproduced from Timperley et al.

In addition to the Richardson Road studies, the Ministry of Transport (MoT) and New Zealand Transport Agency (NZTA) have also invested in research to assess the importance of roads as sources of stormwater contaminants. An Ministry of Transport study from the early 2000s estimated vehicle emission factors of a range of contaminants based on international literature review, sampling of road dust in Waitakere City and Wellington and analysis of tyre and brake pad samples, key sources of zinc and copper, respectively (summarised in Kennedy 2003). More recently, Moores et al. (2010) conducted a road runoff sampling programme on four state highways in the Auckland region to investigate the influence of traffic characteristics on variations in the vehicle emissions of copper and zinc. The results were compared with those of previous published and unpublished New Zealand studies, including some of those described above, finding that higher rates of copper and zinc emission were associated with sites of more frequent braking and acceleration.

All the studies above have investigated the influence of specific land uses on water body state (water quality and sediment quality). In the only New Zealand example that we know of that has attempted to systematically investigate the impact of a specific urban land use on stream ecology, Shaver and Suren (2011) compared macroinvertebrate communities upstream and downstream of six state highways. The selection of sampling locations reflected roads having a relatively high contaminant-generating potential (more than 10,000 vehicles per day (vpd)), a lack of other urban or rural contaminant sources, streams with relatively limited dilution potential, and sufficiently similar stream characteristics above and below the point of road stormwater discharge. The results of the studied indicated that there were few detectable differences in macroinvertebrate-based ecological health metrics upstream and downstream of the road runoff discharge points.

Recognising the considerable growth in locally-collected urban water quality in recent years, NIWA, Auckland Council and the University of Auckland partnered in the development of an urban runoff quality database. The database holds data collected by councils, government agencies and research institutions throughout New Zealand and includes measurements of physical-chemical properties and contaminant concentrations in samples of both stormwater and urban streams. The database can be freely queried through a web-based tool, the Urban Runoff Quality Information System (URQIS[11]; Gadd et al. 2014) which aims to provide a reliable and consistent source of urban runoff quality information for use in resource management processes and research activities. Inclusion of a range of metadata in the database allow summary results to be compared according to land use.

Figure 3-19 provides examples of outputs from URQIS; boxplots of the concentrations of total zinc and copper in samples of untreated stormwater associated with a more refined breakdown of urban land uses than the broad categories described earlier in this section. Consistent with findings summarised above, the URQIS dataset indicates greater land use variation in zinc concentrations than in copper: median zinc concentrations vary by an order of magnitude, with a range of 0.07 – 0.67 mg/L, while median copper concentrations fall within the relatively narrow range 0.013 – 0.02 mg/L. Concentrations of total zinc are highest in stormwater samples collected from light industrial and city centre commercial (CBD) locations, followed by medium density residential areas. Relatively low concentrations of total zinc have been measured in samples from roads and low-density residential areas. Stormwater samples from CBD and heavily trafficked roads (>20,000 vpd) have the highest total copper concentrations. The relative ranking of other land uses is somewhat counter to expectations, with roads carrying 5-20,000 vpd having lower total copper concentrations than low density residential areas.

[11] https://urqis.niwa.co.nz/#/report

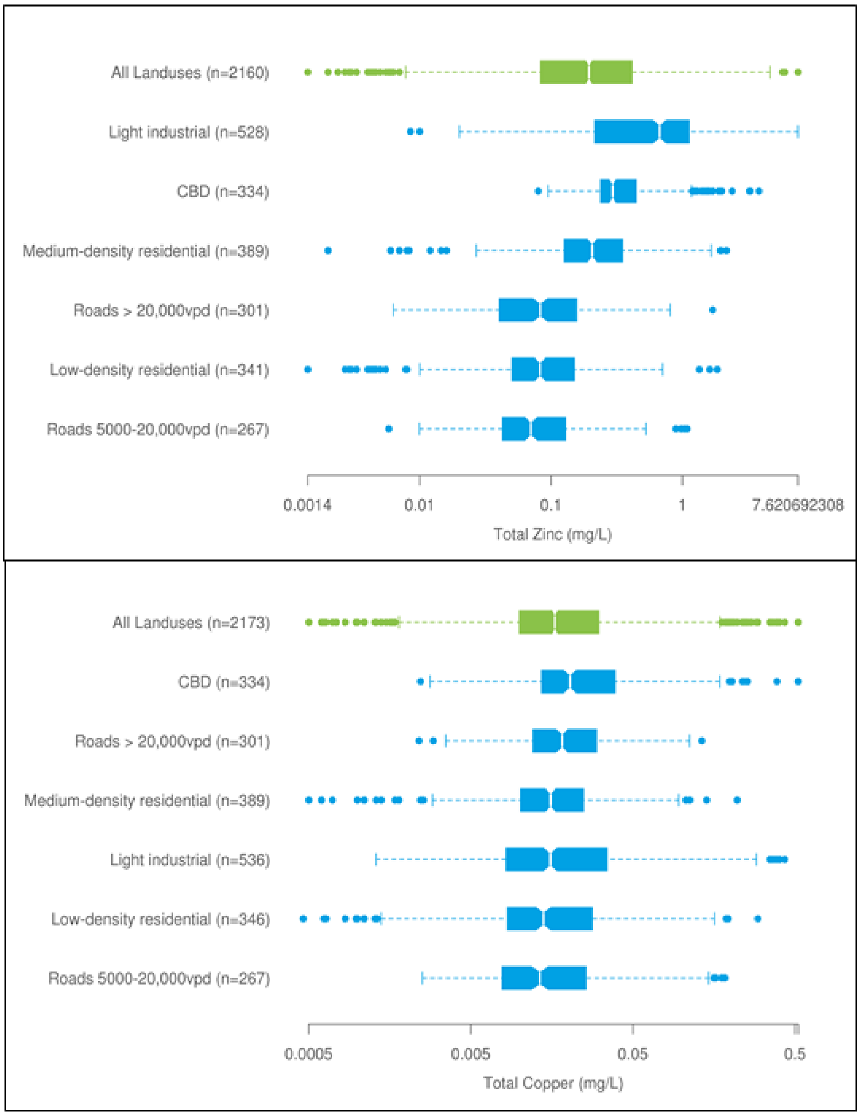

Figure 3-19: Examples of outputs from URQIS: box plots of total zinc concentrations (top) and total copper concentrations (bottom) in untreated stormwater, by land use. From Gadd et al. (2014).

Land cover

This section first summarises studies that have investigated PSI relationships associated with impervious surfaces (for example, roofing materials, concrete and asphalt) as a single, aggregate land cover class. It then summarises studies that have focused on specific types of impervious surfaces.

Impervious land covers

National scale study

Clapcott et al. (2012) investigated relationships between land-use pressures, including impervious cover, and a range of water quality and ecological variables based on collated measurements made as part of nationwide and regional monitoring programmes. The study analysed water quality data collected at between 360 and 578 sites (depending on variable) over the period 1996-2002 and macroinvertebrate survey data from over 2600 sites, collected over varying time periods, mostly since 1990. The proportion of impervious land cover in the catchment of each monitoring site was derived from a GIS layer identifying impervious features such as roads and buildings.

Using boosted regression trees, the land-use pressures were found to explain 20-57% of the deviance in three water quality variables. However, other than for median DRP concentrations, impervious cover was a less important factor in explaining deviance in the water quality variables than the other land-use pressures, percentage removal of indigenous vegetation and stream N loading. The three land-use pressures together explained 6-36% of deviance in macroinvertebrate metrics, with impervious cover again making a lesser contribution to explanatory power than the other land-use pressures. Despite its lower contribution to the overall results, the authors found that almost all metrics "indicated reduced ecological integrity at very low levels of impervious cover, indicating that any levels of impervious cover >0 have a visible effect on stream integrity." The study found a threshold at 10% impervious cover, describing this as an "upper limit for maintaining stream integrity."

City scale studies

Allibone et al. (2001) investigated the influence of impervious cover on macroinvertebrate communities surveyed at 35 urban stream sites in Auckland. Statistically-significant negative relationships were found between the percentage of impervious cover and three macroinvertebrate community metrics. EPT taxa richness was found to decline rapidly as percentage imperviousness increased, from a count of 11 at 10% impervious cover to 4 or fewer at 20% or greater (Figure 3-19).

Collier and Clements (2010) investigated the influence of impervious cover on stream macroinvertebrate communities at 25 urban stream sites in Hamilton. Macroinvertebrate metrics were found to decline with increasing impervious cover across a range of scales. The authors found that local-scale impervious cover (within the immediate stream corridor adjacent to survey sites) was more strongly related to macroinvertebrate metrics than the proportion of impervious cover in the wider catchment.

As part of their survey of sediment quality at sites in the Avon River catchment (see Section Specific urban land uses), Gadd and Sykes (2014) investigated relationships between percentage impervious cover and sediment concentrations of copper, lead, zinc and polycyclic aromatic hydrocarbons (PAHs). While concentrations of all four contaminants increased with the proportion of impervious cover, the relationships were relatively weak (Figure 3-20). Other factors, including broad land use (Figure 3-16) and specific industrial activities and land fills were considered likely sources of the variability in the relationships between percentage impervious cover and sediment quality.

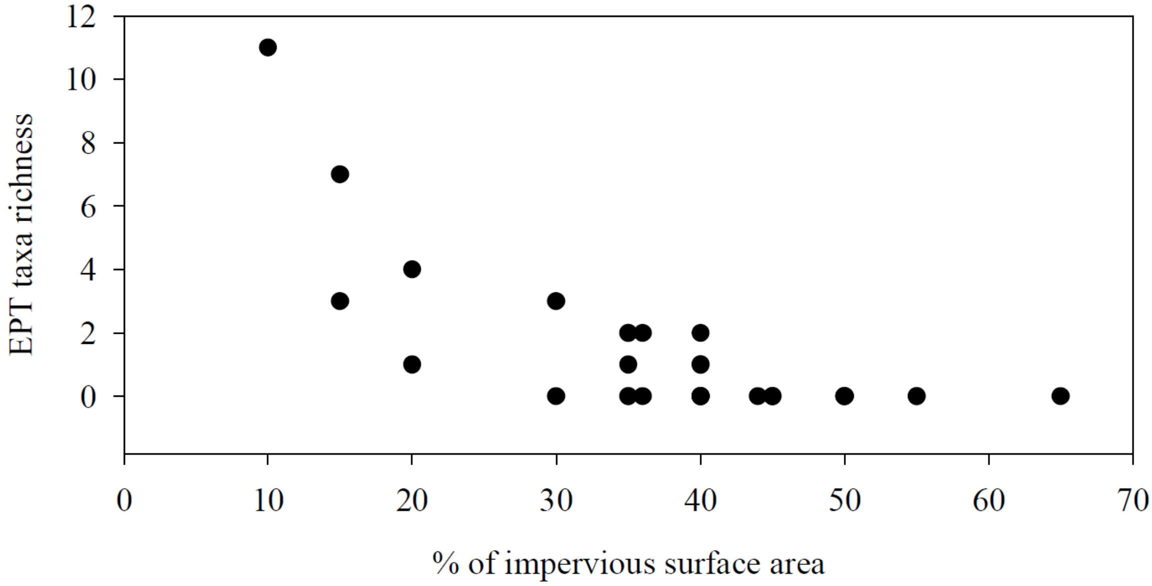

Figure 3-20: Relationship between impervious cover (%) and EPT taxa richness in 35 Auckland urban streams. From Allibone et al. (2001).

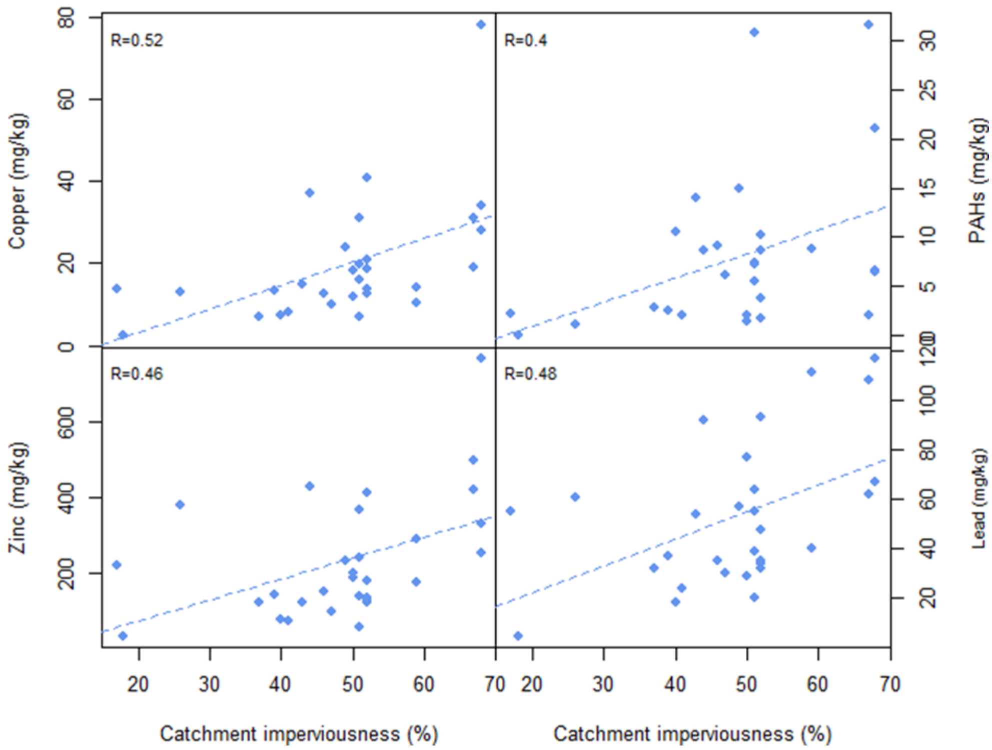

Figure 3-21: Relationships between metal concentrations in stream sediment samples and impervious cover (%) at sites in the Avon River catchment. From Gadd (2014).

Specific urban land covers

Roofing materials

A study of roof runoff quality in Auckland compared contaminant concentrations discharged from different roofing materials and in different conditions (Kingett Mitchell Ltd and Diffuse Sources Ltd, 2003). Notably, dissolved zinc concentrations discharged from galvanised iron roofs were found to be markedly higher than from other materials, identifying this roof type as a key source of zinc in urban stormwater, especially in areas of industrial and commercial land use (see Figure 3-17). Subsequent research in Christchurch has found roof age and rainfall pH to be an important factor in the leaching of zinc from galvanised roofs (Wicke et al. 2014).

As well as zinc, roofs can also be an important source of copper: Pennington and Webster-Brown (2008) and Charters et al. (2016) found markedly elevated concentrations of copper in stormwater runoff from copper roofing at locations in Auckland and Christchurch, respectively.

Road and paving materials

Wicke et al. (2012) and Murphy et al. (2015) sampled stormwater runoff from experimental concrete- and asphalt- surfaced boards at a car park and residential area, respectively, in Christchurch. Runoff from concrete boards was found to have higher TSS concentrations than that from the asphalt boards, attributed to the smoother surface characteristics of the concrete. In contrast, copper and zinc were adsorbed by the concrete, resulting in the discharge of lower loads than from the asphalt boards. A permeable asphalt board was found to trap suspended solids but provide little or no retention of copper or zinc, which were discharged in the dissolved form (Murphy et al. 2015).

Moores et al. (2013) found that the characteristics of stormwater discharged from porous asphalt vary with age of the road surface. Runoff samples collected from a recently-sealed section of Auckland's northern motorway contained markedly lower suspended solid and particulate metal concentrations than those from a six-year old section of the road.

Bare earth

The construction phase of the urban development process involves the removal of vegetative land covers and the exposure and compaction of bare earth. Unless well managed (see Section Erosion and sediment control), rainfall runoff from earthworks sites can contained elevated suspended solids concentrations, resulting in reduced water clarity, impacts on fish feeding and behaviour and the smothering of stream beds (Suren and Elliott 2004).

Williamson (1993) reported mean values of turbidity, suspended solid concentrations and visual range for low flow conditions in New Zealand streams. Streams in areas of construction earthworks had 20 times the turbidity, over 10 times the SSC and $1/10^{th}$ of the visual range of streams in residential areas.

The influence of earthworks runoff on stream water quality is especially marked during storm events. Winter (1998) reported that runoff from an area of earthworks at Albany on Auckland's North Shore had a maximum SSC of 20,000 g m^{-3}, around 200 times the background concentration in the receiving stream. Monitoring at Long Bay, also on Auckland's North Shore, have measured median and 95^{th} percentile TSS concentrations in earthworks runoff that were more than 100 and 3500 times higher, respectively than the equivalent background stream concentrations (Basher et al. 2016).

Land management practices

A range of practices exist that aim to reduce the impact of urban development on receiving water bodies. Some of these involve the direct management of different land covers. For instance, erosion control practices on earthworks sites involve replacing one cover type (bare earth) with another (e.g., a mulch or geotextiles). Source control of zinc involves replacing high-yielding roof materials (unpainted galvanized steel) with low-yielding alternatives (painted or coated materials). Riparian restoration projects typically involve planting of shrubs and trees, replacing grassed stream banks with a narrow forested strip.

Other practices do not directly involve land management but, instead, aim to manage urban water quality impacts by intercepting contaminants prior to their discharge receiving water bodies. For example, sediment retention ponds and stormwater treatment ponds are deployed widely for this purpose during the construction and post-construction phases of urban development, respectively.

The following sections describe studies that have investigated the performance of both direct and indirect land management practices in urban areas. Notably, the review excludes any coverage of practices to manage roof-generated loads of stormwater metals because, as far as we are aware, no specific studies of the performance of such practices exist

Stormwater treatment

Stormwater management has traditionally focused on drainage and flood control, with urban runoff piped and discharged to the nearest waterway with little consideration given to its potential impacts on the character and functioning of receiving water bodies. However, since the late 20th century increasing attention has been given to addressing stormwater quality issues.

Stormwater ponds have been the principal form of treatment, functioning by sedimentation of stormwater solids and associated particulate forms of contaminants. Internationally, a significant combined monitoring effort has revealed that ponds are limited in their effectiveness for the removal of finer solid matter and dissolved contaminants (for instance, see the US-based International Stormwater Best Management Practices (BMP) Database[12], Wright Water Engineers Inc. and Geosyntec Consultants 2017), prompting the development and adoption of alternative forms of stormwater treatment. Stormwater treatment in New Zealand is increasingly employing devices such as wetlands, vegetated swales and bioretention systems (or raingardens). These systems employ biofiltration and/or infiltration, where stormwater contaminant concentrations are reduced by soil and plant matter interception and uptake, and can be employed as part of a wider water sensitive approach to urban development (see Section Water sensitive urban design).

A number of New Zealand studies have assessed the performance of individual stormwater treatment devices. Moores et al. (2010, 2013) sampled a pond and swale adjacent to an Auckland motorway and found the latter to be more effective for metal removal because of its better performance on dissolved metals. As a potential innovation to improve the performance of stormwater ponds, Auckland Council initiated a programme of research to investigate the potential to retrofit existing ponds with floating treatment wetlands (FTW). Initial small-scale mesocosm and pilot trials demonstrated the potential of these systems for the removal of suspended solids, organic matter, nutrients and metals (Headley and Tanner 2012). A subsequent full-scale trial near Auckland's northern motorway involved reconfiguring an existing stormwater pond, with half

[12] http://www.bmpdatabase.org/

retrofitted with a full-scale floating wetland and half left unvegetated as a control. Intensive monitoring of physical-chemical parameters and analysis of water and sediment samples found that the presence of plant roots in the water column increased the trapping of particulate contaminants, with increased sequestration of metals and nutrients in the pond sediments and reductions in effluent concentrations of suspended solids, metals and nutrients compared to the control effluent (Borne et al. 2013, 2014).

As well as the growing use of bioretention systems, a wide range of proprietary devices, such as catchpit inserts, media filters and hydrodynamic separators have also become commercially available in New Zealand in recent years. Suppliers' performance claims leading to the acceptance of proprietary devices by regulators in New Zealand have been largely based on results reported from the United States, where a number of states require new stormwater treatment technologies to be assessed according to one of a number of rigorous testing protocols. However, in a field assessment of the performance of three commercially-available media filters, Moores et al. (2012) found that device performance was strongly influenced by the characteristics of influent runoff and the bypassing of treatment during high flows, leading to guidance on the design and maintenance of media filters.

While there has been a considerable research effort, both internationally and within New Zealand, to evaluate the water quality performance of individual stormwater treatment devices, the assessment of receiving water body outcomes of stormwater treatment has received far less attention. The few studies that we are aware of have compared outcomes under conventional and 'water sensitive' approaches to urban development and stormwater management. Accordingly, these are summarized below in Section Water sensitive urban design.

Water Sensitive Urban Design

Water Sensitive Urban Design (WSUD) is an alternative to conventional forms of urban development that aims to integrate urban planning and water management in order to better manage, among other things, water quality in natural waterbodies. While different jurisdictions place emphasis on different aspects of WSUD (Fletcher et al. 2014), the following concepts are particularly evident in a New Zealand 'understanding' of what WSUD comprises. (see, for example, Lewis et al. 2015).

Firstly, WSUD aims to limit stormwater runoff and contaminant generation at source by minimising the construction of impervious surfaces, such as roads and roofs. This can be achieved, for instance, by building clusters of multi-storey dwellings in order to retain relatively large areas of undeveloped green space. Secondly, WSUD aims to maintain the functioning of natural drainage systems, rather than replacing stream networks with piped systems. In combination, these practices aim to maintain characteristics of catchment hydrology, including infiltration, groundwater recharge and stream flow characteristics, similar to those that existed pre-development. Thirdly, WSUD uses green technologies to better manage stormwater in a way that complements its approach to land use planning. The use of permeable paving, for instance, helps to promote infiltration and reduce stormwater runoff. Bioretention systems, or raingardens, also provide for runoff control while providing treatment to improve stormwater quality via the removal of contaminants as stormwater infiltrates through an engineered soil media. Wetlands also provide stormwater treatment and runoff control, as well as providing habitat and amenity services.

There is a significant body of international research on the performance of WSUD infrastructure, for instance involving monitoring the effectiveness of individual green technologies for managing stormwater. As noted above, the international BMP database is perhaps the best know repository of device-scale performance data. In New Zealand, there have been performance assessments of a range of WSUD technologies, including bio-retention devices (Trowsdale and Simcock 2011), permeable paving (Fassman and Blackbourn 2011) and green roofs (Voyde et al. 2010).

In contrast, only a small number of studies have assessed the cumulative effects of the implementation of WSUD at the catchment or receiving environment scale. Dietz (2007) wrote that "many large-scale housing projects utilizing LID[13] techniques have been installed around the country [the USA]. However, to date only one such project has had monitoring of stormwater quantity and quality to investigate the cumulative impact of LID systems."

However, since then the results of several additional US studies comparing water quality and hydrology in catchments with different forms of land use and development configurations have been published. Bedan and Clausen (2009) assessed water quality and hydrology in a study of paired catchments in Connecticut, one adopting WSUD practices including grass swales, cluster housing, shared driveways, rain gardens, and a narrower pervious concrete-paver road, and the other a traditional subdivision. In another paired catchment study, Winston et al. (2013) assessed the water quality and hydrology resulting from the retrofitting of a North Carolina residential area with bioretention devices and permeable paving. In both studies, the WSUD catchment generally out-performed the traditional development or control catchment. Peak stream flows were maintained or reduced in the WSUD catchments, while markedly increasing in response to traditional development. WSUD also resulted in a reduction in catchment loads of a range of water quality variables, including nutrients and metals.

In a New Zealand study, van Roon and Rigold (2016) reported on differences in stream ecology associated with alternative development approaches in the Flat Bush area of south Auckland. Macroinvertebrate sampling was conducted over the period 2005-2013 in 10 sub-catchments, varying in terms of dwelling density (medium or countryside living), riparian vegetation and the extent to which urban design and stormwater management featured WSUD. The authors reported markedly higher macroinvertebrate metric scores in streams in WSUD catchments than in those developed according to conventional practices. Scores for the WSUD catchments were also generally better than those for urban streams previously surveyed across the Auckland region.

In another area of Auckland currently undergoing WSUD development, Auckland Council is monitoring stream hydrology, water and sediment quality and ecological indicators at Long Bay on the North Shore. Mills (2016) reported on the results of sediment quality monitoring conducted before and during the development of the area. Stream sediment copper and zinc concentrations remained below effects thresholds, in contrast to those in a stream draining a neighbouring area of earlier urban development. While there was some evidence of increasing sediment metal concentrations, the authors were cautious about the reliability of trend analysis results. To our knowledge, results of the other monitoring conducted at Long Bay to date have not been published.

Erosion and sediment control

A wide range of erosion and sediment control practices are used on urban earthworks sites to reduce sediment-laden runoff reaching receiving water bodies. These include:

[13] 'LID' stands for Low Impact Design, an alternative name for WSUD

- erosion control practices for managing sediment generation at source, for instance, the use of mulches and geotextiles to protect areas of bare earth and stabilization by hydroseeding; and
- sediment control practices for managing sediment discharges from a site, for instance, the use of sediment retention ponds and temporary measures such as silt fences.

Basher et al. (2016) reviewed literature on studies investigating the performance of both groups of practices. Readers are referred to that report for a summary of international research, most of which has been conducted in the United States.

Since the late 1990s there has been a growing interest in establishing the performance of erosion and sediment control practices in New Zealand, especially in the Auckland Region. While we provide a summary of some of the key local studies below, readers are again referred to Basher et al. (2016) for a comprehensive review of New Zealand research in this area.

An experimental study at Albany, North Shore found that mulching could reduce sediment loads discharged from bare earth by 85% or greater (Auckland Regional Council 2000). In another Albany-based study Winter (1998) found that a sediment retention pond reduced sediment loads by 70-99% by monitoring influent and effluent quality over 11 storm events. However, despite the capture of the majority of the sediments entering the pond, the mean suspended solids concentration in pond effluent samples was still ten times higher than in background stream samples.

In order to address this issue, in recent years enhanced erosion and sediment control practices that utilise chemical treatments have become increasingly common. These include the addition of coagulants to sediment retention ponds in order to flocculate suspended fine sediments and enhance their capture and settlement. In a study during the construction of Auckland's northern motorway extension, Moores and Pattinson (2008) found that chemical treatment of a sediment retention pond resulted in an overall marked improvement on that of an untreated pond, but that performance varied with storm event characteristics and the management of the chemical dosing system. More recent monitoring at the Long Bay development found that sediment concentrations in effluent discharged from chemically-treated ponds was generally consistent with background concentrations in receiving streams (Basher et al. 2016).

Urban stream restoration

Urban stream restoration typically involves riparian planting but can also include restoring 'natural' channel forms and the daylighting of piped stream sections. A number of studies have investigated the extent to which stream restoration in New Zealand cities has resulted in an improvement in stream ecological health. Two Christchurch studies found little or no improvement in macroinvertebrate metrics following stream restoration (Suren and McMurtrie 2005; Winterbourn et al. 2007 (cited in Harding et al. 2016)), attributed to a lack of improvement in water quality (Harding et al. 2016). In contrast, at one Christchurch location fish species diversity improved markedly and this was attributed to the improvement in physical habitat quality resulting from restoration (Suren and McMurtrie 2005).

In west Auckland, Project Twin Streams (PTS) has involved riparian planting of 56 km of stream banks since 2003. Despite this significant restoration effort, Stansfield (2016) found that MCI scores for restored urban streams in the PTS catchments was not significantly different to those for other urban streams covered by Auckland Council's SoE monitoring programme. As noted earlier (Section Urban land use as a single category), scores for macroinvertebrate metrics at the PTS urban sites were lower than at forested and pastoral stream sites.

In another Auckland restoration project, two 90 m reaches of piped urban streams were daylighted in 2013. Neale and Moffett (2016) found that daylighting resulted in changes to the structure of macroinvertebrate communities, although with only one of 12 metrics (EPT richness at one of the two sites) showing a statistically significant improvement. The authors linked differences in the responses of the two sites to differences in the strength of upstream source populations from which recolonization could occur. They also suggested that the relatively limited extent of the improvement observed could reflect the relatively short-time frame of the study or the characteristics of the restored stream substrate.

3.3 Lakes

Introduction

Lakes are sinks for land-derived contaminants and other land-use pressures generated in their catchments. As such, lakes are subject to the PSI framework set out in Section 2.2. In addition, lakes are "reactors" where physical, chemical and biological processes interact over time to mediate the lake's state and impacts on lake values. These mediating processes are not the focus of this section, however. The focus issue is how catchment land-use pressures affect the states of New Zealand's lakes and their values.

Pressure-state-impact framework as applied to lakes

We reviewed 27 New Zealand studies linking land-use pressures to lake state and impacts in order to assess the state of knowledge regarding the PSI framework in relation to lakes and to weigh the evidence about land-use effects on New Zealand lake values.

Land use and land management practices exert a wide range of pressures on lakes. The primary pressures can be grouped into two general classes, contaminant loads and hydrological alterations (Figure 3-22). For both classes, lake tributaries and groundwater inflows provide the conduits by which land-use pressures impinge on lakes. For example, most of the sediment, faecal microbes, phosphorous and particulate N that originates on land and enters lakes is conveyed by tributaries. Nitrate, pesticides and other mobile contaminants may be conveyed by tributaries or groundwater flowpaths that discharge to lake beds. The processes by which these contaminants are lost from land is covered in Section 3.1.1.

Water abstraction and other hydrological alterations associated with land use affect water budgets, water levels, mixing, thermal regimes and other internal processes, as well as contaminant loading from land.

These pressures comprising contaminant loading and hydrological alteration can affect three main characteristics of the state of lakes: water quality, hydrology and vulnerability to invasive species.

Macro- and micro-nutrient loading to lakes can fuel phytoplankton proliferations (Schallenberg 2004), which can in turn reduce dissolved oxygen concentrations, habitat availability and quality, alter energy flow through food webs and produce algal toxins (Wood et al. 2016). Changes to lake hydrology can negatively impact lake water levels, which impacts on littoral, benthic habitats. Furthermore, hydrological alteration affects water residence time, flushing rates and the nutrient retention capacity of lakes.

The consequences of alterations in lake state are wide-ranging, and include loss or reduction in ecological, social, Māori and economic values such as biodiversity, populations of threatened species, opportunities for customary harvests, sports fisheries and tourism. In addition, faecal contaminants, algal toxins, pesticides and other toxicants and agricultural pharmaceuticals pose public health risks.

The ecological impacts of land-use pressures can destabilise lake ecosystems, and increase their vulnerability to invasion by non-indigenous species. If populations of invasive species become established in compromised lakes ecosystems, their impacts can cause further destabilisation, leading to shifts in ecosystem structure and functioning (Schallenberg and Sorrell 2009). Ultimately, biodiversity and a range of ecosystem services may be negatively affected by such perturbations (Figure 3-22).

In addition to contaminant loading and hydrological alteration, land use on and adjacent to lake shorelines is associated with wetland drainage, clearance of native terrestrial vegetation and replacement with invasive non-native vegetation such as willows (*Salix cinerea*, *S. fragilis*) and rushes (*Juncus articulatus*) (Tanner 1992). Studies of the effects of altered lakeshore vegetation have focused on aquatic habitat modification (e.g., Chisnall 1996, Jellyman and Sykes 2009). Other potential effects include alterations in contaminant input from shallow groundwater, alterations in terrestrial carbon input from litter, alterations in lake-shore erosion, and competitive exclusion of native lakeshore species.

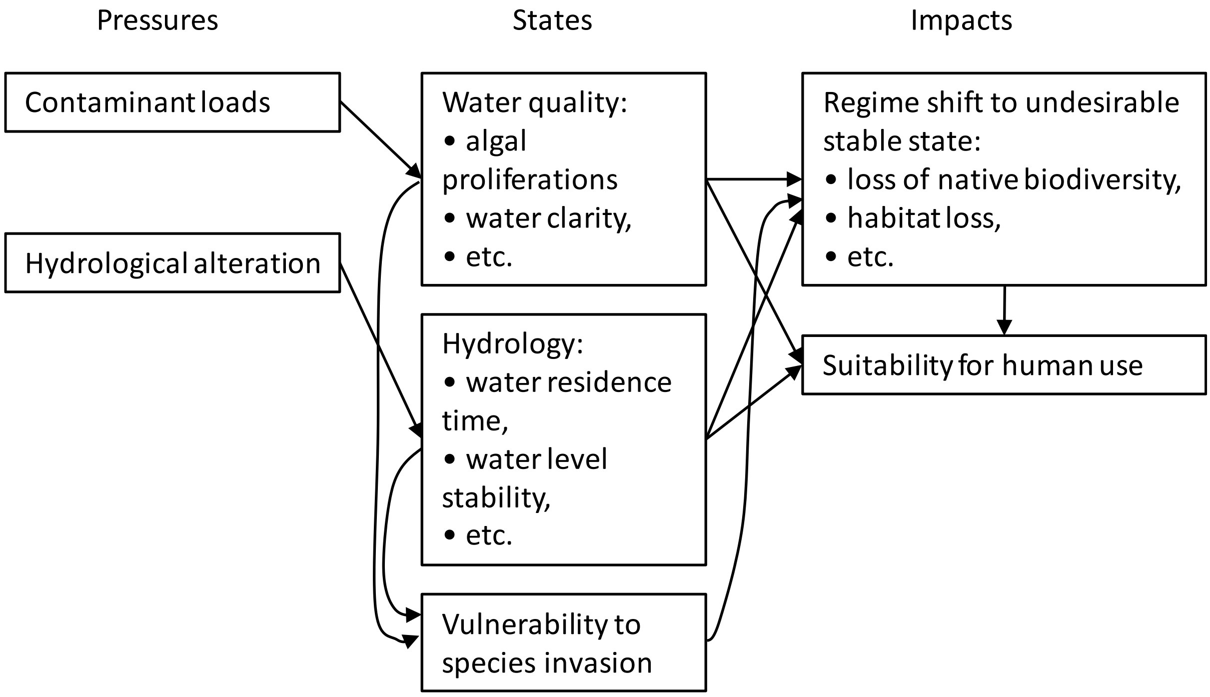

Figure 3-22: Pressure-state-impact framework as applied to land-use effects on lake ecosystems.

Land use effects on lake water quality and ecological conditions

We compiled reports of studies that specifically assessed the relationship between land management and lake ecosystems. The studies are summarised in Table 3-3 and have been separated into pressure-state relationships and pressure-impact relationships.

Pressures examined in these studies were either expressed: (1) as percentages of catchment area in different classes of land cover (obtained from various versions of the New Zealand Land Cover Database), (2) as total P and total N loads to lakes obtained using the CLUES model (partially derived from land cover data: http://www.mpi.govt.nz/environment-natural-resources/water/clues), or (3) the total N and total P loads were measured directly in tributary streams and rivers and variously apportioned to agricultural sources.

No studies were found that directly linked pressures from specific land management practices to lake responses. However, Vant and Huser (2000) showed that dairy cow stocking rates in nine catchments in the Waikato region correlated strongly with specific N yields from the catchments (Figure 3-22), which suggests that stock density is also related to N input to lakes. Nevertheless, our analysis would have benefitted from the existence of PSI relationships related to specific land use activities such as stocking rates, fertiliser application rates, and cropping information.

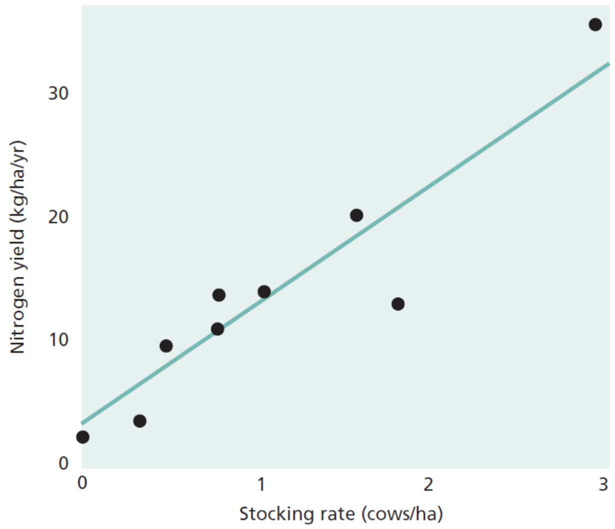

Figure 3-23: Correlation between dairy cow stocking rates and specific N yield from nine catchments in the Waikato Region. R^2 = 0.93, P < 0.0001. Reproduced from PCE (2004) from data in Vant and Huser (2000).

A wide range of lake state indicators were used in these studies including water quality variables (e.g., trophic state, nutrient concentrations, phytoplankton biomass), macrophyte cover, and bioindicators such as the diversity of macrophytes, rotifers, benthic macroinvertebrates, phytoplankton and zooplankton (Table 3-4). The water quality, macrophyte and macroinvertebrate state variables are similar to river state variables that have often been linked to land-use pressures (see Section 3.1), however the zooplankton, phytoplankton, and rotifer diversity bioindicators are specific to lakes.

Fewer studies focused on relationships between catchment pressures and impacts on lake values (Table 3-4). However, we found five studies that focused on impacts, including the probability of an ecological regime shift, the probability of lake macrophyte loss, and effects on the multi-component metric of lake health called ecological integrity (Schallenberg et al. 2011). Ecological regime shifts are common in shallow lakes that are exposed to high nutrient and/or sediment loads and the shift from clear-water to turbid-water phases can occur rapidly when a pressure threshold, or tipping point, is exceeded (Scheffer 2004). Ecological resistance and resilience to pressures plays an important role in regulating the state and impacts of lakes under pressure from high nutrient and/or sediment loads. Resistance and resilience in the undesirable, turbid state can result in inertia to recovery even when nutrient and sediment loads are substantially reduced (Dodds et al. 2010). For these reasons, the impacts of regime shifts on lakes can be severe and, therefore, regime shifts should be avoided if possible.

Pressure-state relationships

The 22 studies on lake pressure-state relationships generally indicated that both catchment land cover and loads related to land cover classes had negative effects on lake state variables (Table 3-4). Relationships have been based on statistical correlations among lakes along a pressure gradient (e.g., Figure 3-24; Figure 3-25), nutrient budgets for specific lakes (e.g., Table 3-5), or modelled or estimated catchment nutrient losses coupled with a process-based model of lake responses (e.g., Table 3-6).

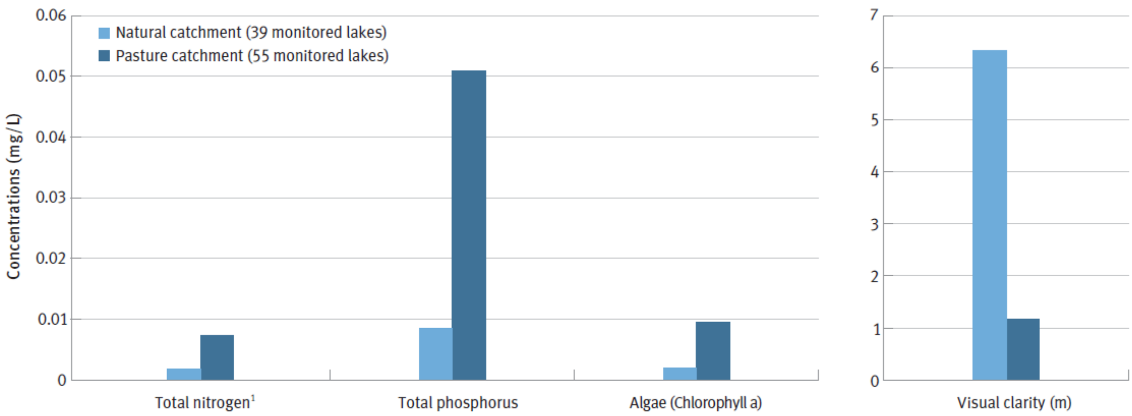

Figure 3-24: Comparisons of water quality variables between lakes in catchments with are predominantly in pasture and those predominantly in natural vegetation. Total N data have been scaled by 1/10. Reproduced from Ministry for the Environment (2007).

Comparisons of water quality indicators from 94 lakes across New Zealand clearly showed that the average water quality in lakes categorised has having predominantly natural catchments was far better than water quality in lakes categorised as having predominantly pasture catchments (Figure 3-24; Ministry for the Environment (2007)). Mean N, P and phytoplankton concentrations were 2 to 6 times higher in lakes in predominantly pasture catchments, and mean visual clarity was 5 times greater in lakes in predominantly natural catchments.

Table 3-4: New Zealand studies showing direct effects of land management on lake ecosystems. Pressure-state relationships are in the top 22 rows and pressure-impact relationships are in the bottom five rows.

Study	Lake(s)	Lake Response(s)[1]	Pressure(s)[2]	Demonstrated association	Strength/Confidence[3]
Pressure-state relationships					
Abell et al. (2011)	101 NZ lakes	total N, total P	% High producing grassland catchment cover % Urban catchment cover % Exotic forestry catchment cover	Statistical correlation	R^2 between 0.04 and 0.52
Drake et al. (2011)	43 Shallow NZ lakes	Chl *a*, macrophytes	CLUES N and P loads	Statistical correlation	R^2 between 0.22 and 0.31
Galbraith and Burns (2007)	45 Otago lakes, ponds, wetlands, estuaries, reservoirs	total P, total N, DRP	% Pasture catchment cover	Statistical correlation	R^2 between 0.48 and 0.58
Kelly et al. (2013)	18 South Island shallow lakes	total P, total N, Chl *a*, TLI, LakeSPI, water clarity, macrophyte cover, macroinvertebrate and macrophyte communities	CLUES N and P loads	Modelling and statistical correlation	R^2 between 0.36 and 0.86
Kelly et al. (2014)	27 Canterbury high country lakes	total P, total N, Chl *a*, TLI, LakeSPI, water clarity, macrophyte community	CLUES N and P loads	Modelling and statistical correlation	R^2 between 0.02 to 0.73
Kelly et al. (2016)	27 Northland dune lakes	total P, total N	CLUES N and P loads	Modelling and statistical correlation	R^2 between 0.01 to 0.49
Özkundakci et al. (2014)	24 NZ deep lakes	Rotifer species richness, macroinvertebrate species richness, zooplankton species richness	CLUES N and P loads	Statistical correlations	R^2 between 0.38 and 0.55
Weaver et al. (2017)	Lake Wanaka (Otago)	Phytoplankton	Nitrate and DOC loads	Statistical correlation	Moderate
MfE (2007)	NZ lakes	total N, total P, Chl *a*, Water clarity	% Pasture, % Natural	Statistical correlations	High

			catchment cover		
Verburg et al. (2010)	NZ lakes	TLI, (LakeSPI)	% Pasture, % Native catchment cover	Statistical correlations	R^2 = 0.27, R^2=0.28 (R^2=0.26, R^2=0.10)
Schallenberg (in press)	NZ lakes (shallow, deep and brackish)	total P, total N, Chl *a*, trophic level index, water clarity, native fish, macrophyte, macroinvertebrate, rotifer and phytoplankton communities	% Native catchment cover	Statistical correlations	High
Rutherford et al. (1996)	Lake Rotorua (Bay of Plenty)	total P	total P load	Modelling	Moderate
Spigel et al. (2015)	Lake Benmore (Canterbury)	Trophic state	CLUES N and P loads	Modelling	High
Norton et al. (2014)	Lake Ellesmere (Canterbury)	Trophic state	CLUES N and P loads	Modelling	High
Fish (1969)	Lake Rotorua and others (Bay of Plenty)	Eutrophication (excessive plant growth)	N and P loads from pasture	Budget	Moderate
Lovegrove (1985)	Lake Alexandrina (Canterbury)	Eutrophication	Nutrients	Budget	Moderate
Vant and Huser (2000)	Lake Taupo (Waikato)	Phytoplankton	total N loads	Budget	High
Verburg et al. (2013)	Lake Brunner (West Coast)	Phytoplankton	total P and total N loads	Budget	Moderate
Williamson et al. (1996)	Lake Rotorua (Bay of Plenty)	Trophic state	total P load	Budget	Moderate
PCE (2006)	Lake Rotorua (Bay of Plenty)	Lake Health	total P and total N load	Budget	High
Burns et al. (1997)	12 Bay of Plenty lakes	Trophic state	% Pasture catchment cover	Various	Moderate to High
Pressure-impact relationships					
Drake et al. (2011)	43 Shallow NZ lakes	Ecological integrity	CLUES N and P loads	Statistical correlations	R^2 between 0.22 and 0.31

Özkundakci et al. (2014)	24 NZ deep lakes	Ecological integrity	CLUES N and P loads	Statistical correlations	R^2 between 0.38 and 0.55
Schallenberg and Sorrell (2009)	37 NZ shallow lakes	Severe loss of macrophytes and regime shift from clear to turbid water	% Pasture catchment cover, % Forest catchment cover	Statistical correlations	R^2 = 0.92
Schallenberg et al. (2017)	Waituna Lagoon (Southland)	Severe loss of macrophytes	N load, P loads	Statistical correlations and modelling	Moderate
Norton et al. (2014)	Te Waihora (Canterbury)	Cyanobacteria risk	CLUES N and P loads	Modelling	Moderate

[1] Chl *a* – chlorophyll *a*; DRP – dissolved reactive P; TLI – trophic level index; LakeSPI – lake submerged plant index

[2] DOC – dissolved organic carbon

[3] Either the reported strength of the correlation reported in the study, or our inferred degree of confidence in the association

Catchment land cover and lake state have also been statistically related by correlating catchment land cover to water quality indicators. For example, Verburg et al. (2010) reported a positive correlation between the percentage of pasture land cover in 112 lake catchments and lake trophic state, and a negative correlation between the percentage of pasture land cover Lake SPI index scores (Figure 3-25).

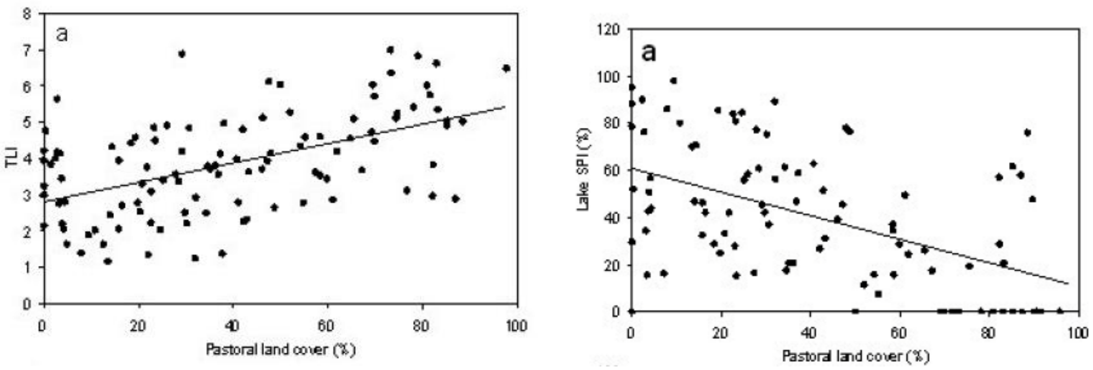

Figure 3-25: Correlations between lake trophic level index (TLI; left) and submerged plant index (Lake SPI; right) vs. percentage of the catchment in pasture in 11 lakes. Reproduced from Verburg et al. (2010).

The contributions of land use activities to lake trophic state have also been inferred using nutrient budgets, which attribute fractions of the total lake nutrient load to different land parcels in the catchments, which comprise multiple land cover classes. For example, the annual N and P budgets for Lake Rotorua (Bay of Plenty) in 2004 showed that pasture in the catchment contributed 52% of the N and 28% of the P to the lake (PCE 2006; Table 3-5). The same budgets showed that exotic forestry contributed 2.5% of the N and 1.5% of the P to the lake. In this case, the lake has a substantial internal load of both N and P from the sediments to the water column and the estimates of the contributions of land uses were underestimates because they did not account for the proportion of the internal load attributable to historical nutrient inputs from land-based activities.

Using simple input-output models incorporating lake volume and water residence time (e.g., Kelly et al. 2013), the loads calculated in the Lake Rotorua budgets can be used to estimate contributions of the different land use classes in the catchment to lake nutrient concentrations. In this way, the link between land cover and lake state (as contaminant concentrations) can be assessed.

Table 3-5: Annual N and P loads to Lake Rotorua for 2004, showing the importance of pasture as a source of N and P to the lake. Reproduced from PCE (2006).

Nutrient source	Catchment area (ha)	% cover	N load (tonnes yr^{-1})	% total N load	P load (tonnes yr^{-1})	% total P load
			From land			
Native forest & scrub	10,588	23.9	42	5.5	1.3	3.3
Exotic forest	9,463	21.4	28	3.7	1.0	2.4
Cropping & horticulture	282	0.6	17	2.2	0.6	1.4
Pastoral	20,112	45.4	573	75	18.1	45.7
Lifestyle	556	1.3	11	1.5	0.5	1.3
Urban	3,267	7.4	50	6.6	3.8	9.6
Springs	–	–	–	–	13.0	32.8
Geothermal	–	–	42			
Total	**44,268**	**100**	**763**	**100**	**39.7**	**100**
			Sources other than land			
Rain	–		32		1.2	
Internal load	–		308		24.0	
Lake	8,079					
Total	**52,347**		**1,103**		**64.9**	
Wildfowl			1.43		1.37	

The third way that links between land use and lake state have been characterised is by coupling process-based lake models to either catchment models or measured catchment nutrient loads. Whether modelled by a catchment model such as CLUES, or measured as part of a nutrient budget, the nutrient loads can be apportioned to different land cover classes and in this way the effects of land use on lake state attributes can be estimated. For example, the effects of agricultural intensification and increasing irrigation on the trophic state of Lake Benmore were modelled by linking CLUES-derived nutrient loads from different land use scenarios to the lake process-based models DYRESM (hydrodynamic) and CAEDYM (biological) (Spigel et al. 2015). The exercise produced 4-dimensional relationships linking land use, catchment N and P loads, and lake trophic state (Figure 3-26).

Pressure-impact relationships

The five studies we reviewed that examined lake pressure-impact relationships generally indicated that catchment land cover and contaminant loads related to land cover classes both had negative impacts on lake values (Table 3-4). Relationships were demonstrated either by: (1) statistical correlations among lakes along a pressure gradient (e.g., Figure 3-26) or by (2) the modelling of lake responses to different catchment nutrient load scenarios.

Where statistical correlations were provided, the strengths of the correlations are shown (Table 3-4). Where modelling was used to demonstrate relationships and no statistics were provided, then we subjectively attributed a confidence class to the information published, indicating how strong the evidence was in confirming a substantial effect of land-use pressures on lake state.

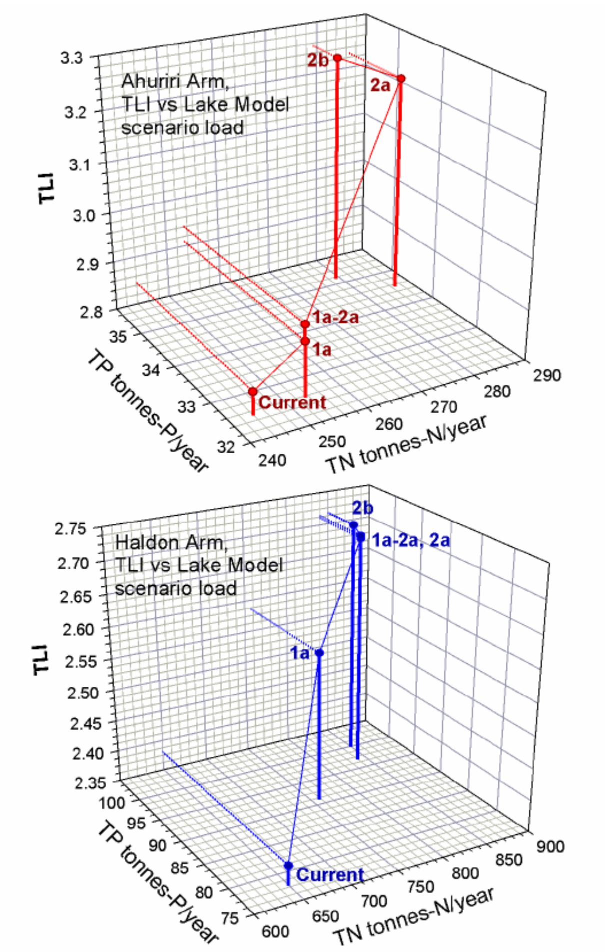

Figure 3-26: The modelled response of the trophic state (TLI) of Lake Benmore in relation to different scenarios of total P (TP) and total N (TN) loading. Nutrient loads were generated by CLUES catchment modelling based on different land use scenarios. The scenarios are labelled Current, 1a, 1a-2a, 2a and 2b in order of increasing N and P loading. Reproduced from Spigel et al. (2015).

Shallow lakes can undergo rapid degradation in response to increasing nutrient loading. This has often been linked to the sudden loss of submerged macrophytes, which are foundations species in shallow lakes. The loss of macrophytes from a lake bed facilitates a shift in lake ecosystem structure and function, often resulting in phytoplankton dominance and decreased water clarity (Scheffer 2004). Schallenberg and Sorrell (2009) identified 37 lakes in New Zealand that had apparently undergone such regime shifts from a clear-water, macrophyte dominated state to a turbid, phytoplankton-dominated state. They also identified a number of similar shallow lakes which had not undergone such regime shifts, using the combined dataset to assess the relationship between regime shifting in shallow lakes and land cover in their catchments. Their results indicated that regime shifts in lakes increased in occurrence as the proportion of catchment land cover in pasture increased and the proportion in native forest land cover decreased (Figure 3-27). They also found that the presence of *Egeria densa* (an invasive macrophyte) and the presence of certain pest fish species were positively associated with regime shifting in shallow lakes, suggesting that interactions between eutrophication and the presence of invasive species may accelerate regime shifts in shallow lakes.

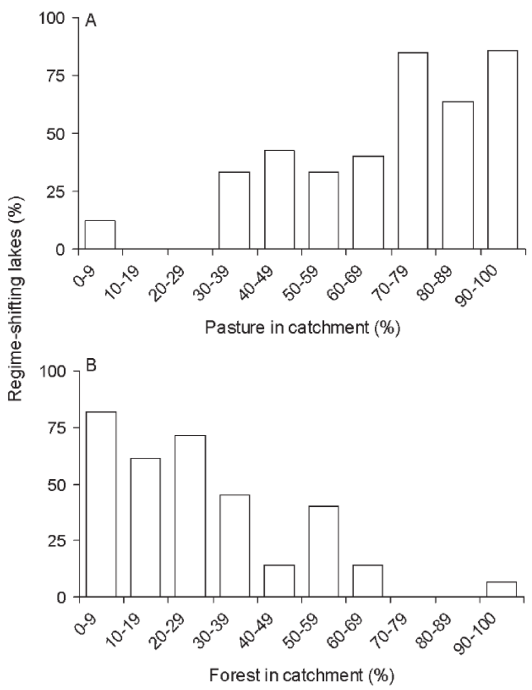

Figure 3-27: Percentages of lakes with catchments in different pasture (A) and forest (B) classes that have undergone regime shifts. (i.e., catastrophic loss of macrophytes and increases in turbidity). Analysis is based on 37 shallow lakes that have undergone regime shifts and 54 shallow lakes that haven't. Reproduced from Schallenberg and Sorrell (2009).

Modelling has also been carried out to assess impacts of land use on shallow lakes. For example, process based models were used to estimate the risk of cyanobacterial bloom occurrence in Te Waihora/Lake Ellesmere under different catchment nutrient load scenarios, where the nutrient loads were generated by the CLUES catchment model (Norton et al. 2014; Table 3-6). The model predicted that the risk of blooms would increase only under a land use scenario which would result in a 50% reduction in the N load, compared to the load observed during the model calibration period.

While the coupling of catchment and lake process-based models provides a mechanistic rationale linking land use to lake state and impacts, such complex renderings of relationships are also subject to numerous assumptions and limitations (Spigel et al. 2015). Validation of model outputs against measured data provides a rough indication of uncertainty and bias in the model (Fedra et al. 1981), but commonly used regression-based validation techniques do not assess a number of sources of modelling error that complex, deterministic models are subject to (Loehle 1997).

On the other hand, the empirical, among-lake correlations and associations in Table 3-4 and discussed above are only suggestive of a causal relationship because correlations only quantify associations - they do not confirm cause and effect linkages (Havens 1999).

Table 3-6: Comparison of trophic level index (TLI) and cyanobacterial risk for different catchment nutrient loading scenarios (estimated with the CLUES catchment model) for Te Waihora/Lake Ellesmere. Predictions are based on DYRESM/CAEDYM modelling. The Cal scenario represents the modelled state of the lake during the calibration period. TLI-3 excludes water clarity in the index. Table is reproduced from Norton et al. (2014).

Scenario	Cyanobacteria risk	TLI-3	TLI-4	Notes
SM	0.65	6.5	6.6	50% reduction in TP, 50% increase in TN
S2+	0.66	6.5	6.6	50% reduction TP, 29.7% increase in TN
S3 50% reduction	0.67	6.3	6.5	50% reduction in TP
S3	0.69	5.7	6.0	50% reduction in TN and TP
S2a	0.72	7.2	7.2	75.9% increase in TN
S2	0.73	7.3	7.3	10% increase in TP load, 75.9% increase in TN
S1	0.74	7.2	7.2	35.9% increase in TN
S2c	0.74	7.4	7.3	20% increase in TP load, 75.9% increase in TN
Cal	0.74	6.9	7.0	Calibration
S3 50% reduction	0.77	6.2	6.4	50% reduction in N

In general, the pressure-state and pressure-impact relationships that we collated ranged from strong to weak or non-significant, and while each approach used to demonstrate these relationships has weaknesses, the multiple lines of evidence presented in Table 3.1, collectively indicate that pasture-based land uses are often linked to the degradation of lakes. Specifically, the proportion of the catchment in pasture is often positively related to eutrophication and its symptoms such as regime shifts and cyanobacterial blooms.

Lake trends

The Ministry for the Environment and the OECD publish trends in New Zealand lake water quality and ecological variables. National trends in lake water quality have been assessed based on median trend magnitudes for each variable within lake classes (e.g., low-elevation, deep lakes, high elevation shallow lakes), and by tallying the number of lakes within each class with improving and degrading trends in each variable (e.g., Verberg 2010, Larned et al. 2015). Based on the tallying method, Larned et al. (2015) reported that lakes with decreasing chlorophyll *a*, trophic lake index, ammoniacal-N, total N and total P levels outnumbered lakes with increasing levels of the same variables over the period 2004 to 2013. In contrast, lakes with increasing nitrate-N concentrations outnumbered lakes with decreasing nitrate concentrations.

Land use change at the national scale over the period 1996 to 2012 was recently summarised in Ministry for the Environment and Statistics New Zealand (2018). This report indicated that rates of change in areal land uses declined from the period 1996 to 2002 (when over 200,000 ha of land changed land-cover class) to the period 2006 to 2012, when just over 50,000 ha of land changed land-cover class. The trends over the latter period showed net minor losses in grasslands and indigenous forests and shrubland (-1.3 and -0.7%, respectively), while land in exotic forest and urban land uses increased (+11.2 and +10%, respectively).

We found no analyses relating recent changes in land cover or land use with changes in lake state or impacts. However, a report by the Parliamentary Commissioner for the Environment (2013) shows a strong positive correlation between the change in the predicted amount of N pollution to freshwaters and the predicted change in land used for dairy farming amongst the regions of New Zealand (Figure 3-28). This is an indication that trends in land conversion to dairy farming may relate positively to N pollution to lakes.

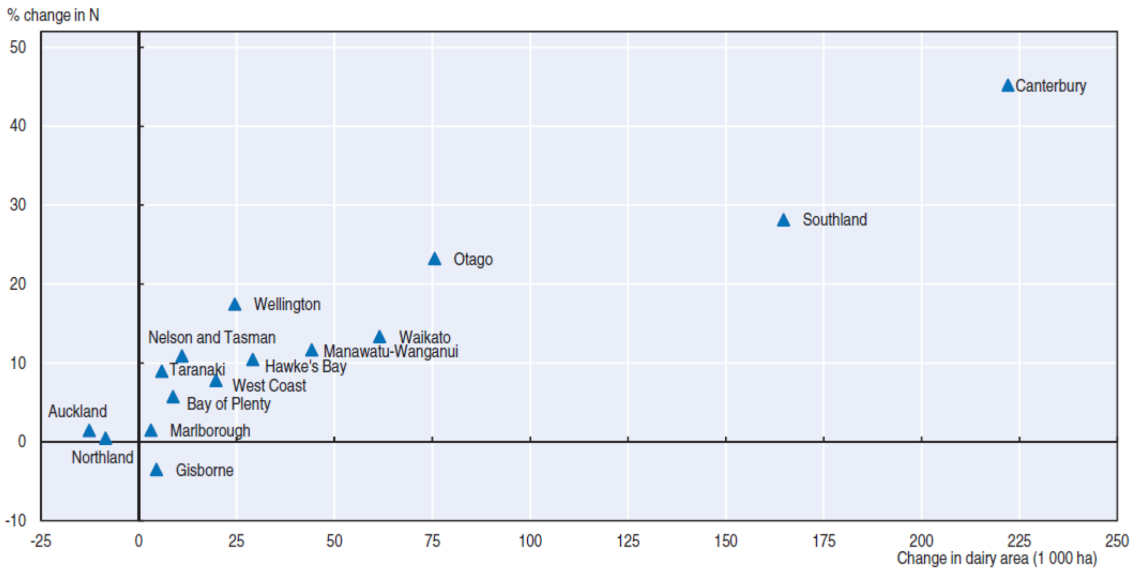

Figure 3-28: Projected change in N pollution per region and projected change in land area per region converted into dairy farming. Reproduced from PCE (2013).

3.4 Aquifers

Introduction

This section summarises published information on PSI relationships and trends in New Zealand's aquifers. Agricultural and horticultural and urban land use exert the following pressures on aquifers:

- Increased contaminant input from the land surface, injection wells, river seepage and other sources.
- Groundwater depletion and reduced groundwater recharge. These land use pressures result from increased rates of groundwater and surface water abstraction, and changes in vegetation, tillage, river flows and other properties that influence rainfall and river recharge.

Changes in the environmental state of aquifers as a response to land use pressures fall into two broad areas:

- Altered groundwater quality. Aquifers under intensive agricultural, horticultural and urban land use are characterised by high concentrations of nutrients (Close and Flintoft 2004, McDowell et al. 2015), and elevated levels of mobile pesticides and faecal bacteria (Close et al. 2008, Close and Humphries 2016).
- Altered groundwater quantity, including alterations in long-term water table elevations and short-term fluctuations (at both aquifer scales and near wells), in river recharge and spring flows, and in lake-bed and submarine discharge (Fenemor and Robb 2001).

The impacts of these changes in state may include:

- Changes in the health of groundwater-dependent ecosystems, including wetlands, rivers, lakes, riparian vegetation, and the invertebrate, protozoan and microbial communities in aquifers
- Changes in the risk of seawater intrusion
- Changes in ecosystem services (e.g., water purification).
- Changes in water supplies and supply reliability for irrigation, drinking water and other uses.

The following sections summarise studies concerning the effects of land use, land cover and land management practices on aquifers. Response variables included physical-chemical water quality variables (e.g., nutrients, pesticides, microbial contamination), and other indicators of state and impact. We also summarise studies of temporal trends in these state and impact variables.

Associations between land use classes and groundwater state

National-scale studies

Nutrients

Morgenstern and Daughney (2012) identified natural baseline groundwater quality and effects of land use intensification by relating groundwater chemistry to water age. Plots of hydrochemistry and field parameters versus groundwater age were used to distinguish water quality parameters that increase in concentration with age and are therefore from geological sources, from those affected by recent land use. The nitrate data in oxic groundwaters of the National Groundwater Monitoring Programme (NGMP) data showed a record of land-use change in New Zealand, with nitrate levels tracking land use intensification. Old groundwater with a mean residence time (MRT) > 130 years reflected the natural baseline conditions prior to anthropogenic land-use impact. In young groundwater (< 130 years) recharged after agricultural intensification commenced, nitrate-N (up to 2.5 mg L^{-1}) and pesticides were the most representative indicators for the effects of land-use intensification. A transition to slightly elevated nitrate-N concentrations (> 2.5 mg L^{-1}) due to low-intensity land-use was observed in groundwater recharged prior to approximately 1880. Five NGMP sites had younger groundwater with nitrate-N concentrations up to 34 mg L^{-1}. A significant increase in nitrate and other agrochemicals due to high-intensity agriculture was observed in groundwater recharged since 1955. Pesticide concentrations were also elevated in young groundwater (< 60 years), further indicating land-use effects.

Groundwater chemistry data from the NGMP sites for the period 2004-2013 were used for a recent state and trend analysis (Moreau and Daughney 2015). In approximately 40% of the NGMP monitoring sites, the groundwater was strongly affected by human activities. These 'impacted sites' had elevated nitrate-N concentrations, without the elevated iron and manganese concentrations associated with rock dissolution. The most common trends in the NGMP data analysis were increasing concentrations (i.e., degrading conditions) in groundwater nitrate-N, ammoniacal-N, DRP, iron, and manganese. No associations were detected between solute concentrations (or trends) and the land use or land cover surrounding the wells. Previous publications based on NGMP data have also reported no detectable associations between land use and groundwater chemistry. These results are likely due in part to uncertainty about the capture zones of the NGMP wells; delineation of capture zones will aid in identifying the land use corresponding to groundwater sampled by each well (C Daughney, GNS Science, personal communication).

McDowell et al. (2015) carried out a meta-analysis of New Zealand datasets to identify potential associations between P concentrations in soil, surface water and groundwater and land use, with an emphasis on dairying. They use groundwater data from 540 monitoring wells. The groundwater component of the meta-analysis was based on DRP alone, as other fractions such as total P and organic P are not routinely measured in groundwater. Results of the study indicated that there were significant differences between land-use classes in mean DRP concentration in both surface and groundwater, and that groundwater associated with dairy farms was the most DRP-enriched. Groundwater DRP concentrations associated with six other land-use classes were lower and did not differ significantly.

Pesticides

National groundwater pesticide surveys have been carried out in New Zealand at four-yearly intervals since 1990. Some regions also undertake their own more intensive groundwater pesticide monitoring programmes. Here we review the results of two recent national surveys. The fifth national survey was undertaken in 2006 (Gaw et al. 2008). Of the 163 wells sampled, 31 (19%) tested positive for pesticides (19 different pesticides and their metabolites), and two or more pesticides were detected in 13 (8%) wells. Pesticides were detected in one or more wells in 11 out of the 14 regions sampled (no detections in Northland, Hawke's Bay, and Taranaki). Herbicides were the pesticide group most commonly detected (12 different herbicides detected), followed by insecticides (5), and fungicides (2). There was a total of 50 pesticide detections and of these, 37 were herbicides with 25 detections of triazine herbicides. Simazine (11 wells) and terbuthylazine (8 wells) were the two most frequently detected pesticides. Concentrations of two of the 50 pesticide detections were > 1 μg L^{-1} and only 1 of the positive pesticide detections (alachlor, 34 μg L^{-1}) exceeded the New Zealand Drinking Water Standard (NZDWS). The effect of land-use type on pesticide detection was tested for shallow wells for which there were sufficient data. Results of that analysis indicated that more pesticides were detected in wells surrounded by urban and horticultural land uses compared with other agricultural land uses. Pesticide concentrations in groundwater had decreased since the previous survey, which was consistent with reported reductions in pesticide use on horticultural crops following the introduction of integrated pest management schemes and changes to weed management techniques in orchards (Manktelow et al. 2005).

The seventh national survey was carried out in 2014, using 165 wells (Close and Humphries 2016). Pesticides were detected in 28 wells (17%), of which 10 had two or more pesticides. Pesticides were detected in at least one well in each of six of the 13 participating regions. Pesticides were not detected in sampled wells from Hawkes Bay, Taranaki, Horizons, Wellington, Marlborough, Canterbury, and Otago. Herbicides were the most frequently detected pesticide group: four insecticides and two fungicides were detected. There were 31 separate detections of triazine herbicides and terbuthylazine the most frequently detected pesticide, with 16 detections. There were four pesticide detections with concentrations > 0.001 mg L^{-1}, but only one of the sampled wells exceeded the NZDWS. That exceedance was for the insecticide Dieldrin, which was widely used in New Zealand in the 1960s for controlling parasites on sheep. Dieldrin has not been in active use for decades, but residues are still present in soil and groundwater where sheep dip wastewater was discharged. The rate of pesticide detections in 2014 was similar to that in previous surveys, with little change in types of pesticides.

Groundwater levels

Groundwater levels have fallen in recent history in some flood plains, and this may be related to land drainage and groundwater abstraction for irrigation. Lowered aquifer levels have led to sea water intrusion in a number of places, including aquifers on the Heretaunga and Waimea Plains, and small aquifers situated beneath rural seaside communities such as those on the Coromandel Peninsula (Brown et al. 1999). In 1990 seawater intruded 600 m inland in a shallow gravel aquifer at Lower Moutere near Nelson after irrigation caused groundwater levels to drop (Taylor and Smith 1997). Changes in management have generally reversed seawater intrusion and many councils now use networks of sentinel wells to monitor for seawater intrusion.

Regional council state of the environment reporting

In this section we review several regional council reports that are based on groundwater SoE monitoring programmes. The reports reviewed here are limited to those that linked groundwater state variables to land use or land cover.

White and Close (2016) reviewed SoE groundwater quality monitoring data for the Waikato, Greater Wellington, and Southland regions. For the Waikato region, nitrate-N concentrations in 14% of the 110 SoE monitoring wells were greater than the NZDWS of 11.3 mg L^{-1}, and nitrate-N concentrations were < 1 mg L^{-1}in 24% of the wells. Lower nitrate-N concentrations were observed in a subset of the Waikato wells used for school water supplies (e.g., 2% of wells were > 11.3 mg L^{-1}, 55% of the wells were < 1 mg L^{-1}). This pattern was attributed to the abandonment and replacement of school-supply wells when groundwater quality exceeded the NZDWS, and to a higher proportion of anoxic groundwater in school-supply wells. Trends were analysed for 30 of the Waikato wells that were monitored quarterly for 15-20 years. Nitrate-N concentration were increasing in 37% of the wells and decreasing in 23% of the wells.

In the White and Close (2016) review of groundwater in the Wellington region, median nitrate-N concentrations ranged from <0.002 to 11 mg L^{-1}. *E. coli* was detected on at least one sampling occasion in 26 of 44 wells (59%) tested, which may reflect poor well-head protection rather than underlying groundwater quality. However, one well at Te Horo Beach in Kapiti consistently had *E. coli* concentrations in excess of the drinking-water standard (< 1 cfu 100 ml^{-1}) and water quality in this bore was clearly impacted by local land use (on-site wastewater treatment).

In the White and Close (2016) review of groundwater in the Southland region, the influence of land-use activities on groundwater quality was evident in the elevated nitrate-N concentrations (7% of bores sampled exceeded the NZDWS) in intensively-farmed areas. There was also a high incidence of microbial contamination. However, microbial contamination is generally a localised issue that can be reduced by adequate well-head protection and by ensuring bores are located appropriately with respect to potential contaminant sources.

Several studies that have referenced long-term SoE groundwater quality monitoring data. Monitoring data from the Franklin Local Board area, south of Auckland, showed elevated and increasing concentrations of nitrate-N in shallow groundwater (Meijer et al. 2016). The Patumahoe and Pukekohe soils in the Franklin area have been used for intensive horticulture for over 100 years. These soils are used primarily for short rotational potato and onion production and are classified as having a high leaching risk (Houlbrooke 2008). In addition to horticulture, the Franklin area is also used for dairy and drystock farming. Application rates of fertilisers in this area vary between crops (Cathcart 1996, Francis et al. 2003, Scoble 2000), but Meijer et al. (2016) reported that it is not uncommon to apply 1 tonne of fertiliser ha^{-1} of potato and onion crops. Although high nitrate-N concentrations in groundwater (up to 36.1 mg N L^{-1}, Figure 3-29) in the Franklin area appear to be linked to soil classes and agricultural land use, there were insufficient data to substantiate these links. Research is currently underway to develop a dual stable isotope abundance technique to determine the source of groundwater nitrate in rural and urban areas of Franklin.

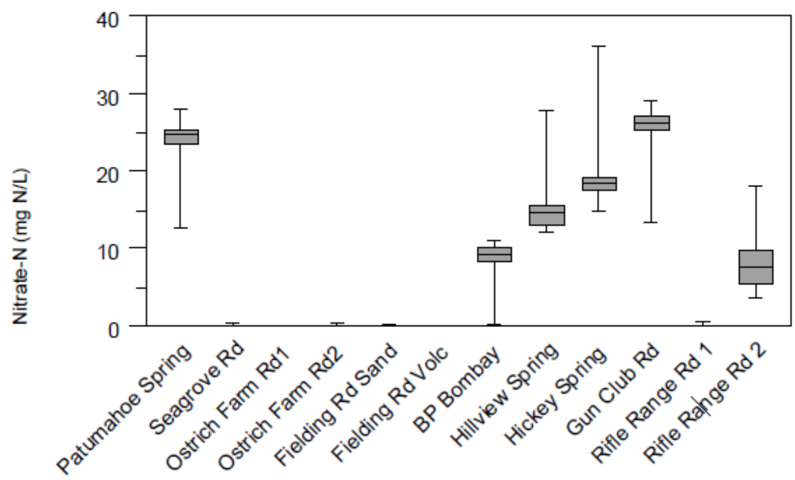

Figure 3-29: Box-and-whisker plots showing the distributions of nitrate-N concentrations at groundwater and spring sites in the Franklin area. Boxes represent interquartile range, mid-lines are medians, and the whiskers are maximum and minimum values (modified from Meijer et al. 2006).

Trend analyses were carried out with the nitrate-N data from the Wellington Regional Council (GWRC) Groundwater Quality SoE (GQSoE) monitoring programme by Baker (2017). The GQSoE programme incorporated quarterly monitoring of water quality in wells in four of five GWRC water management areas. The trend period was October 2003 to 30 June 2016. Of the 46 sites analysed, three had increasing trends in nitrate-N and 11 had decreasing trends. The three wells with increasing trends had low median concentrations (0.005-1.07 mg L^{-1}), which were not considered environmentally significant as they were below the 2.4 mg L^{-1} ANZECC threshold. The increasing trends were attributed to increased pastoral farming in the capture zones of the three wells. The decreasing trends in other sites were attributed to conversion of pastoral farms to lifestyle and/or viticulture.

In the Hawke's Bay Region, groundwater 144 shallow wells (< 40 m depth) were sampled and analysed for nitrate, ammonia and faecal coliforms (Baalousha and Larking 2008). Nitrate-N concentrations in eight wells exceeded the maximum acceptable value of 11.3 mg L^{-1} and 17 wells had detectable levels of faecal coliforms (>1 cfu 100 ml^{-1}). All the wells with exceedances were shallow and the land use in their vicinity was pastoral farming and horticulture. More investigations of the polluted wells were deemed necessary to identify the nitrate sources.

Hanson et al. (2006) reviewed the occurrence of bacterial detections in Environment Canterbury's water quality database over the period 1986 to 2004. Fecal bacteria (fecal coliforms *and E. coli*) were detected in 1732 (19%) of the samples in the database, collected from 502 wells (40% of the 1260 wells sampled). Total coliforms were detected in 3426 (42%) of the samples analysed. The highest bacteria counts (*E.coli* >2400 cfu 100 ml^{-1}) came from samples that were either visibly contaminated or from wells that were located near clear sources of contamination (e.g., septic tank boulder pits, wastewater disposal areas, or refuse pits) under heavily grazed or irrigated agricultural land.

Catchment and farm-scale studies

Groundwater nitrate-N concentrations exceeding the NZDWS (11.3 mg N l^{-1}) have been a problem in the Waimea Plains, Nelson for decades (Stanton and Martin 1975). Stevens (2005) analysed groundwater from 90 wells in the Waimea Plains aquifer and confirmed that nitrate contamination has persisted and is widespread; nitrate in 36 wells exceeded the NZDWS. This contamination has been associated with a combination of domestic wastewater discharge, historic discharge of piggery wastes, discharge of dairy and/or other animal effluent, increased animal stocking rates, fertiliser use associated with intensive land uses (horticulture, market gardening, dairying), inappropriate discharge of waste hydroponics water from glasshouse horticulture, and the discharge of groundwater with elevated nitrate concentrations from the underlying unconfined aquifer. Stewart et al. (2011) used N isotopes (^{15}N) to identify input sources of the nitrate in Waimea Plains groundwater, estimated groundwater residence times using tritium concentrations and combined those results to reconstruct the history of nitrate input on the plains and infer future changes. Results of the study were used to identify two types of nitrate contamination in Waimea Plains groundwater: 1) diffuse contamination in the eastern plains area, which was attributed to the combined effects of inorganic fertilisers and manures for market gardening; and 2) point source contamination attributed to a large piggery. Tritium measurements in wells were used to give MRT's, for groundwater in different parts of the Waimea plains. Mean ages were youngest in the area where nitrate concentrations were highest. The timing of the derived nitrate input history indicated that both the diffuse sources and the point source were present from the 1940s, a period of agricultural intensification on the plains. Major sources of nitrate at that time were located on the main groundwater recharge zone of the plains, leading to contamination of the Upper and Lower Confined

Aquifers. The contamination plume then flowed north, affecting wells for decades. Input of nitrate to the groundwater has decreased since 1988 due to closure of the piggery. A subsequent decrease in nitrate concentrations has been observed in wells gradually travelling northward in the plains.

Morgenstern et al. (2004) undertook hydrochemical and age dating measurements in the western Rotorua and northern Okareka Lake catchments to assess historical and current groundwater quality. The study involved age dating at five springs and seven groundwater wells to estimate travel times for nutrient-enriched groundwater moving from farms to springs, streams, and the lakes. Nitrate-N concentrations were significantly higher for water recharged after land-use intensification. The natural background level of nitrate-N (before land-use intensification) was estimated to be 0.15 mg L^{-1}, and the current recharge level was 2.7 mg L^{-1} (9% of the nitrate-N in the young groundwater is derived from the land-use in the catchment). Total P was very low in young water (<0.04 mg L^{-1}), but increased with decreasing young water fraction, and was considerably higher in the oldest water (1.3 mg/L). These results suggested that the P in the groundwater does not originate from land-use practices, but is leached from the volcanic lithology.

Close et al. (2008) investigated microbial groundwater quality at a border-strip irrigated dairy farm in the Waikakihi catchment in Canterbury. *E. coli* were detected in all monitored wells on the farm, and in 75% of 135 individual groundwater samples. *Campylobacter* were detected in 12% of the samples. *Campylobacter* were also detected in 90% of cow faecal samples tested at the farm. *E.coli* detections were higher during the irrigation season than in winter; variation in *Campylobactor* detections between seasons was minimal. *E.coli* detections peaked following ‘high risk’ events, when irrigation occurred within a few days of stock grazing near the wells.

3.5 Estuaries and coastal zones

Introduction

In this section we review published information on PSI relationships and trends in New Zealand's coastal aquatic environments. Coastal PSI relationships associated with land-use effects can be summarized as follows:

Land development exerts the following pressures on coastal aquatic environments:

- Physical alteration of coastal habitats such as dunes, estuarine wetlands, seagrass beds and mangroves through coastal urban, agricultural and industrial development (GESAMP 2001). These alterations include land drainage, sedimentation, and construction of landfills, recreational, harbor and aquaculture facilities.
- Increases in the loads of contaminants to estuaries and coastal zones. These contaminants include sediment (Thrush et al. 2004), toxicants such as copper and zinc (Abrahim and Parker 2008), and nutrients and pathogens (Nixon et al. 1996, Collins et al. 2007, Barr et al. 2013). Sources of coastal contaminant loads include stormwater, industrial and sewage effluent outfalls, rivers, road runoff, submarine groundwater discharge, and longshore transport from adjacent estuaries and coastal zones (Carpenter et al. 1998; Boehler et al. 2017, Stewart et al. 2017).
- Changes in freshwater flows to coastal waters through urban and agricultural development, such as replacement of highly vegetated land covers with grassland or impervious urban land cover, and the construction of piped drainage networks.

The resulting changes in the state of coastal aquatic environments fall into two broad areas:

- Changes in habitat quality. Sedimentation and other coastal modifications can greatly alter the geomorphology of estuaries and coasts, and habitat quality for wildlife. This is particularly true for the texture and chemistry of benthic substrate in estuaries. The abundance and composition of submerged, emergent and estuarine riparian vegetation, shellfish beds, fish spawning areas and wading bird habitat are frequently affected by habitat alteration (GESAMP 2001, Morrison et al. 2009).
- Changes in water quality. Estuarine systems have inputs of both freshwater from land and marine water. Because most of the water in New Zealand's terrestrial water bodies discharges to the ocean, many of the land use impacts on freshwater state also affect coastal water quality state. For example, estuaries in New Zealand with lower salinity (indicating proportionally greater contributions of land-derived fresh water) tend to have higher concentrations of nutrients and faecal indicator bacteria (Dudley et al. 2017). Because N is the nutrient that generally limits peak primary production in New Zealand coastal waters, increased terrestrial N loading can result in coastal algae blooms and other symptoms of eutrophication, e.g. reduced clarity and alterations in dissolved oxygen fluctuations (Howarth and Marino 2006). Increased sediment loads increase coastal suspended sediment concentrations and reduce clarity in coastal water (Davies-Colley and Smith 2001).

The impacts of these changes in state include:

- Degradation or reduction in fish spawning and nursery grounds (Morrison et al. 2014);
- Reduced diversity and/or abundance of native and harvested finfish species, including those with a migratory component to their lifecycle (Morrison et al. 2009);
- Increased eutrophication and toxic and non-toxic algal blooms (Howarth and Marino 2006);
- Increased microbial pathogen concentrations pose a health risk for contact recreation, sport fisheries, and customary harvests (Malham et al. 2014);
- Increased toxicant concentrations in shellfish, fish and birds pose health risks for customary harvests and sport and commercial harvests (El-Din Bekhit et al. 2011, MacKenzie 2014, Glover et al. 2016).
- Reduced diversity and/or abundance of benthic macroinvertebrates, including shellfish, with communities often limited to the more pollution- or sediment-tolerant taxa (Pratt et al. 2014).

As with the preceding reviews of freshwater environments, we distinguish between PSI relationships in which pressures are represented by land use, land cover and land management practices. We focus on the responses of physical-chemical water quality variables and benthic macroinvertebrate community metrics, as key indicators of state and impact, respectively. Because coastal areas receive much of their nutrient and sediment loads from rivers, many of the PSI relationships in New Zealand's freshwater environments are certain to apply to coastal zones as well. However, studies that have specifically focused on PSI relationships in coastal areas are scarce. Assessments of the state of New Zealand's coastal area based on water quality monitoring are at an earlier stage of development than for freshwater (Larned et al. 2015, Dudley et al. 2017). Furthermore, coastal systems differ from freshwater environments in that changes to state and impacts driven by contaminant loads from land are dependent on the interactive effects of freshwater discharge, tidal mixing of fresh and salt water, and coastal basin morphology (Paerl 2009). These interactive effects have motivated recent research to quantify the roles of coastal hydrosystems (Hume et al. 2016) in modifying pressure state and pressure impact relationships in New Zealand (Robertson et al. 2016a, b, Plew et al. 2018, Zeldis et al. 2018 a, b, c).

Land use

Plew et al. (2018) provided one of the few examples of a national scale pressure-state study for New Zealand's coastal zones. This study presented a simple dilution modelling method for predicting potential (i.e., assuming no biological uptake or losses) nutrient concentrations in estuaries and applied it to New Zealand estuaries. The study used both land cover and land use information; input data used to predict freshwater inflows and land-derived nutrient loads were from NIWA's CLUES model, which uses a suite of land cover and land-use models (Elliott et al. 2016). CLUES model outputs for terminal river reaches are used as inputs for estuarine dilution calculations. Validation data for Plew et al. (2018) were sourced from a recent data assimilation and review of council coastal water quality monitoring programmes (Dudley et al. 2017). A comparison between modelled potential nutrient concentrations and total-N concentrations measured by councils showed a significant positive correlation. Furthermore, because changes in N availability are often responsible for trophic changes in estuaries and coastal zones (Barr et al. 2012, Barr et al. 2013), the dilution modelling approach or its successors can be used to predict changes in estuarine trophic state resulting from changes in land use (Dudley and Plew 2017).

The approach of Plew et al. (2018) has been incorporated into the New Zealand Estuarine Trophic Index (ETI) (Zeldis et al. 2017c). This tool uses the potential nutrient concentrations to calculate a eutrophication susceptibility score. The calculation is based on relationships between N availability and excessive growth of macroalgae and phytoplankton in New Zealand coastal waters (Figure 3-31). The calculation of eutrophication susceptibility scores incorporates N load to the estuary and hydrosystem type as predictors, as well as continuous hydrogeographical variables, such as the proportion of intertidal area in an estuary (Robertson et al. 2016b). If the user identifies an estuary as intermittently open and closed, eutrophication susceptibility scores are provided for both its open and closed state.

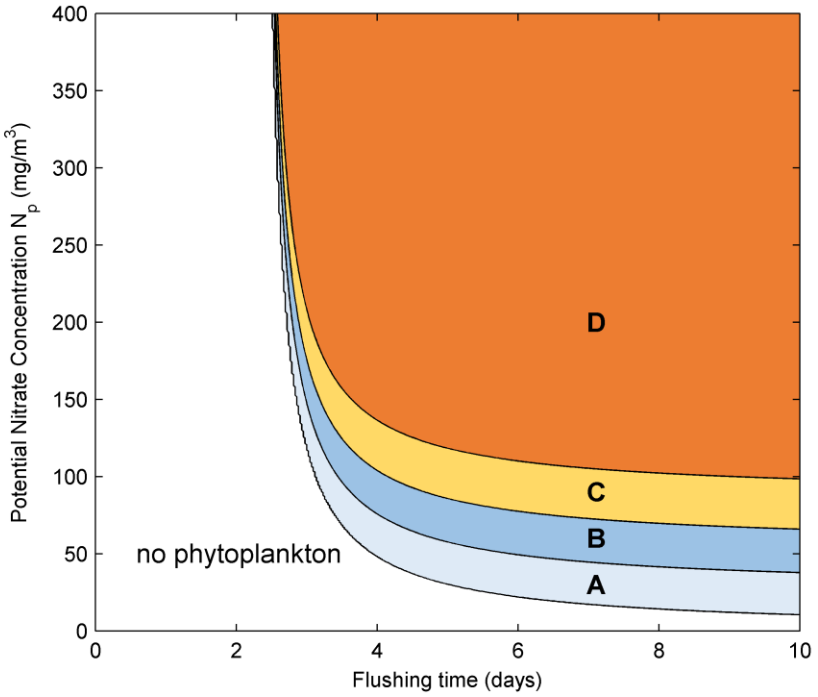

Figure 3-31: ETI susceptibility bandings for phytoplankton based on flushing time and potential N concentration. This graph shows model output based on assumed N half-saturation coefficient of 35 mg/m^3 and a net specific growth rate of 0.43 day^{-1}. Bandings A-D represent increasing risk of eutrophication via excessive growth of phytoplankton.

Green (2013) developed a framework for setting limits on estuarine sediment loads, using data derived from CLUES model outputs. In the example provided, Green calculated limits for sediment loading to the Pāuatatanui estuary, Wellington. This study also explains the place of his proposed framework in New Zealand estuarine management.

Morrison et al. (2009) reviewed effects of land-derived contaminants on coastal fisheries and coastal biodiversity in New Zealand. Of particular relevance to this report, the review of Morrison et al. identified physical alterations in coastal habitat caused by sedimentation and eutrophication. The impacts of sedimentation and eutrophication include modification or loss of nursery habitats, especially those composed of habitat-forming (biogenic) species. These biogenic species include green-lipped and horse mussel beds, seagrass meadows, bryozoan and tubeworm mounds, sponge gardens, kelps/seaweeds, and a range of other 'structurally complex' species (Vooren 1975, Francis et al. 2005, Kappel 2005, Petes et al. 2007). The authors presented evidence and case studies illustrating how these modifications and losses can degrade commercial fisheries. For finfish, this evidence is based primarily on estuarine habitat alteration. The report includes sections on the physical connections of land-based activities to the marine environment, mechanisms of impact, a review of species likely to be impacted, case studies and a gap analysis for further research.

Gibbs (2008) provides a valuable method for tracing changes in estuarine habitat (sedimentation rates), back to changes in land use. The method uses compound-specific isotope data and an isotope mixing model to establish proportional contributions of a 'library' of catchment soil types to estuarine sediments. Because the ages of estuarine sediment layers can be estimated using radioactive tracers, the method of Gibbs (2008) can be applied to sediment cores to estimate changes in sediment sources through time. When compared with retrospective changes in catchment land use this technique can provide information on how changes in land-use pressure have influenced sedimentation rates through time (Handley et al. 2017). This method has been widely applied and refined since its initial publication (Vale et al. 2016, Upadhayay et al. 2017, Mabit et al. 2018).

Land cover

This section summarises the two studies we located that reported associations between land cover and state and impact variables in Zealand estuaries and coastal zones.

Bierschenk et al. (2012) examined the influence of catchment land cover on benthic algal accrual rates and cellulose breakdown. These response variables are indicators of the degree to which nutrient availability limits benthic primary production and microbial activity. Study sites were selected to represent a salinity gradient from freshwater to near-marine. Bierschenk et al. (2017) extended the work Bierschenk et al. (2012) to included biological traits of benthic invertebrates and community metrics of benthic invertebrates along the freshwater-marine continuum. In both studies, 'land use intensity' was represented by the percent cover of natural vegetation in the catchment upstream of each site. Results of the studies indicated that three invertebrate trait categories were directly related to catchment land cover and 11 trait categories were related to deposited fine sediment levels which were, in turn, related to land cover. Invertebrate community metrics (total invertebrate abundance and taxonomic richness) decreased along the salinity gradient from freshwater to near-marine, but did not vary strongly with land cover.

Land management practices

We were unable to locate any studies that reported associations between land management practices and state or impact variables in New Zealand estuaries and coastal zones. However, in studies carried out overseas, contaminant concentrations and loads in estuaries have been positively correlated with the density of wastewater treatment plants and livestock feedlots, human population density, and fertilizer application rates in the catchment, and to volumes of sewage effluent and stormwater discharged into estuaries (e.g., Billen et al. 2001, Fisher et al. 2006, Rothenberger et al. 2009). Presumably, similar relationships apply to New Zealand estuaries in catchments with agricultural and urban land use; the lack of comparable quantitative information is primarily due to the shortages of both estuary monitoring data and land-management practices data.

Temporal trends

Dudley et al. (2017) investigated the current state (site medians and quantiles from data collected from 2011-2015) and temporal trends (1998-2015 and 2008-2015) in coastal water quality. The data were collected at 347 monitoring sites across New Zealand operated by regional councils and unitary authorities. Because variation in water quality across New Zealand's coastal hydrosystems are dependent on dilution and export of freshwater as well as contaminant loads from land, site statistics were grouped into hydrogeomorphic classes that correspond to these dilution and export features. Classes were assigned using ETI methodology as described above.

No comparisons of water quality statistics with catchment land use, land cover or land management practices were made in the study of Dudley et al. (2017). Comparisons between ETI class provide some information about the general effects of land use because classes with greater freshwater influence are likely to have greater concentrations of land-derived pollutants (Figure 3-33). However, these comparisons do not provide information about the effects of specific land uses or land management practices. Dudley et al. (2017) provided a summary of state and trend results at each for which data was provided by councils. These data can be used in the future to assess relationships between water quality at the coastal monitoring sites and land use in the corresponding catchments.

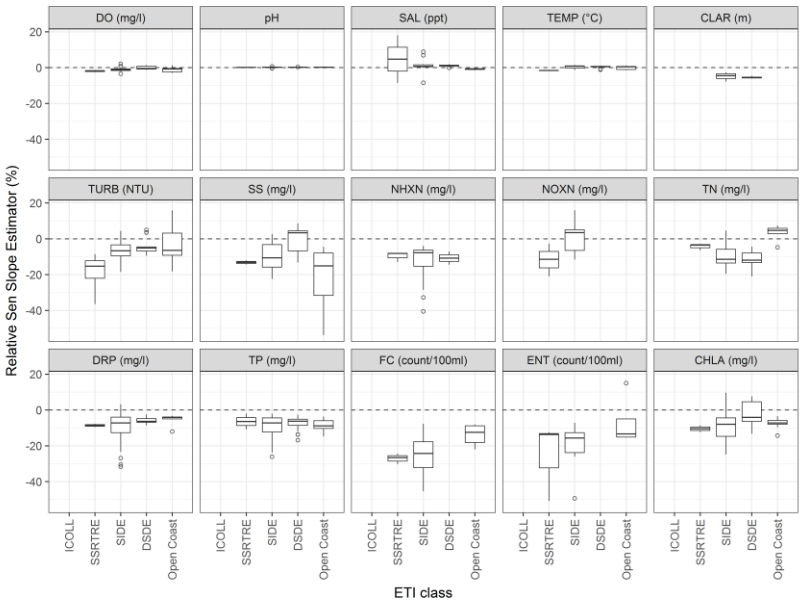

Figure 3-33: Summary of 8-year trends in New Zealand coastal water quality. Box-and-whisker plots show the distributions of site trends within ETI classes. The line within in each box indicates the median of site trends, the box indicates the inter-quartile range and the whiskers extend from the box to the largest value within 1.5 x the inter-quartile range. Outliers (any data beyond the whiskers) are indicated by open circles. Plots, left to right and top to bottom show trends in water column dissolved Oxygen (DO), ph (pH), salinity (SAL), temperature (TEMP), visual clarity (CLAR), turbidity (TURB), suspended solids (SS), ammonium and ammonia (NHXN), nitrate and nitrite (NOXN), total N (TN), dissolved reactive P (DRP), total P (TP), faecal coliforms (FC), enterococci (ENT), and chlorophyll a (CHLA). From Dudley et al. (2017).

4 Gap analysis

4.1 Introduction

Land-use pressure-state-impact relationships such as those discussed in Section 3 are basic tools for environmental management. A diverse set of these relationships are needed for management tasks such as setting limits on resource use, evaluating the effectiveness of management actions and restoration projects, and predicting effects of changes in land use pressures in unmonitored locations and in the future. There is a general shortage of PSI relationships to meet these needs, both in New Zealand and internationally (e.g., Doody et al. 2016).

It is clear from our review that there are many combinations of land use pressure and environmental response for where no reliable PSI relationships have been developed. For many other combinations, the existing relationships are limited in spatial scale or resolution, limited to very specific response variables, have low precision or a combination. We used a simple matrix to summarise these gaps and shortages.

The aim of relating land use pressures to state variables and impacts is straightforward when point-source contaminant discharges (e.g., wastewater outfalls) are the primary pressure. However, large-scale, spatially complex land use is of primary concern today, and relating land use to environmental effects is now more challenging. This challenge is partly due to limited knowledge of catchment processes that link land use pressures to environmental responses, and partly due to limited availability of data that describe land use pressures and environmental responses. We used an email survey of New Zealand environmental scientists and stakeholders to clarify these limitations and compile recommendations to resolve them.

4.2 Methods

4.2.1 Matrix of information availability

Our initial intent was a to use a detailed matrix to organize and rate assessments of PSI relationships corresponding to all possible combinations of land use pressures and state and impact variables, in each of the receiving environments we considered, at multiple scales. This level of detail proved unworkable. There is too much variation across publications in spatial scale, detail, statistical power, choice of pressure and response variables and other parameters to subdivide published studies into multiple, clearly delineated groups and rate the level of information in each group. In lieu of that detailed, quantitative approach, we used coarse subdivisions and the subjective assessments of the section authors of this report. We used five categories of response-variables, and split the approaches used for land use pressure into three categories. The land use categories corresponded to those most frequently used in New Zealand studies of PSI relationships: land cover/land use classes at regional-to-national or local-to-catchment scales, and land management practices. The five response-variable categories were water quality (e.g., nutrient concentrations, visual clarity), hydrological conditions (e.g., flow regime, groundwater level, lake and coastal inflows), ecological values (e.g., biotic community metrics, algal blooms, ecosystem processes); Māori values (e.g., cultural health index, cultural harvest opportunities), social values (e.g., risk exposure to infectious microbes, toxic algae, fisheries).The receiving environment classes were the same ones used in Section 3 of this report: rivers, urban streams, lakes, estuaries and coastal zones, and aquifers.

Ordinal levels of information availability (high, moderate, low and gap) were assigned for each class of receiving environment by the coauthor who led the corresponding subsection in Section 3.

Although assigning levels of information was subjective, the coauthors had a high level of knowledge about the research carried out in New Zealand in their specialist areas.

4.3 Results

4.3.1 Survey of experts

The survey consisted of 14 questions grouped into six topics (Table 4-1). The questions were intentionally open-ended as our aim was to elicit more information than simply categorical responses. Due to the open-ended questions, we did not use a vote-counting approach to tally the responses. Instead, we summarized closely related responses to each of the questions.

A cover letter was sent with the survey form to explain the literature review and provide background needed to clarify the survey questions. In particular, the cover letter defined the three major classes of pressure variables used in our review: land cover (e.g., high-producing grassland, planted forest), land use classes (e.g., arable cropping, dairying, urban residential), and land management practices (e.g., effluent disposal, stormwater treatment, fertiliser application). We noted that there are overlaps between the three classes as they are used in New Zealand, and interdependencies among them.

The survey was sent by email on 14-15 March 2018 to 59 potential respondents, who represented regional councils, central government, universities, agricultural sector groups, Crown Research Institutes, independent research organisations and consultancies. Several respondents passed their survey on to a more qualified person in the same organization, and others filled in the survey jointly with colleagues.

Table 4-1: Email survey used to compile information from New Zealand experts in land and water management about data quality and availability and the state of knowledge about land-use effects on freshwater and coastal environments.

Issues	Questions
Adequacy of mechanistic understanding	1. How do you rate the current state of mechanistic or process understanding of the effects of land use on ecological, cultural and social responses in freshwater and coastal environments? 2. What improvements are needed?
Adequacy of land-use data	3. How do you rate the current state of data quality and quantity (spatial extent, spatial resolution, temporal resolution, classification detail) concerning land use (e.g., arable, horticulture, dairy classes)? 4. How do you rate the current state of land-use data availability? 5. What improvements are needed?
Adequacy of land-cover data	6. How do you rate the current state of data quality and quantity (spatial extent, spatial resolution, temporal resolution, classification detail) concerning land cover (e.g., high-producing exotic grassland, low-producing exotic grassland, exotic forest)? 7. How do you rate the current state of availability of land-cover data? 8. What improvements are needed?
Adequacy of land management activity data	9. How do you rate the current state of data quality and quantity (spatial extent, spatial resolution, temporal resolution, classification detail) concerning land management activities, including mitigation actions (e.g., stocking rates, fertiliser application, stock exclusion, forage cropping, tillage method)? 10. How do you rate the current state of availability of land management activity data? 11. What improvements are needed?
Gaps in mechanistic understanding	12. What are the top three gaps that impede our understanding of land-use effects on freshwater and coastal environments?
Data gaps	13. What are the top three priorities for improving the availability of data needed to understand land use effects? 14. What are the biggest obstacles to acquiring data?

4.3.2 Matrix of information availability

Our broad-scaled assessment of gaps in published information about the associations between land use pressures and water quality, hydrology and ecological, cultural and social values is summarised in Table 4-2. The two clearest patterns are the low levels of information about the effects of land management activities (far right column in Table 4-2) and low levels of information about land-use associations with cultural values in all receiving environment classes. High levels of information availability appear to be limited to water quality in rivers, urban streams and lakes, and ecological values in rivers, as a function of land-cover patterns. For the remaining combinations, information availability is moderate at best.

We suggest that there are four general reasons for scarcity of information about effects of land-use pressures indicated in Table 4-2. First, independent information about land use and about state and impact variables is insufficiently 'joined up'. For example, regional and city council SoE monitoring programmes are producing data about water quantity, water quality and ecological conditions at many sites in each class of receiving environment. However, these data are not routinely linked to land use pressure data. In fact, most analyses of SoE monitoring data focus on identifying temporal trends and central tendencies (i.e., median or mean) in state and impact variables, with little reference to drivers. In a subset of these analyses, monitoring sites are grouped by environmental class, which provides some indication of potential pressures but no quantitative association. Second, land-use data in the form of land-management practices are scarce in New Zealand as noted in Section 2.3. This issue was prominent in the survey of experts summarised below. Third, data to describe cultural values in freshwater and coastal environments are moderately scarce, although work is underway to expand the Cultural Health Index assessments and link them to land use. Fourth, estuary and coastal monitoring programmes in New Zealand are at an early developmental stage, and associations with catchment-scale land use have only been initiated in the last two years, as noted in Section 3.5.

Table 4-2: Ordinal levels of information about associations between land use pressures and response variables. The receiving environments correspond to the subsections in Section 3. Four colour-coded levels of information availability were assigned to each combination of land-use pressure and response category, based on the review of literature concerning each receiving environment. The term 'gap' indicates that no published information was located for a pressure and response combination, or that the existing information anecdotal or extremely limited in scope.

		Land use pressure categories		
		Land cover/land use classes		Land management activities
Receiving environment	Response variable category	National / regional scale	Catchment / local scale	
Rivers	Water quality	High	High	Low
	Hydrology	Moderate	Moderate	Low
	Ecological values	High	High	Low
	Cultural values	Low	Low	Gap
	Social values	Moderate	Moderate	Low
Urban streams	Water quality	High	High	Moderate
	Hydrology	Moderate	Moderate	Moderate
	Ecological values	Moderate	Moderate	Low
	Cultural values	Low	Low	Gap
	Social values	Low	Low	Gap
Lakes	Water quality	High	High	Gap
	Hydrology	Gap	Low	Gap
	Ecological values	Moderate	Moderate	Gap
	Cultural values	Gap	Gap	Gap
	Social values	Low	Low	Gap
Aquifers	Water quality	Moderate	Moderate	Low
	Hydrology	Low	Moderate	Low
	Ecological values	Low	Low	Gap
	Cultural values	Gap	Gap	Gap
	Social values	Low	Low	Low
Estuaries and coastal zones	Water quality	Low	Moderate	Gap
	Hydrology	Low	Low	Gap
	Ecological values	Moderate	Moderate	Gap
	Cultural values	Gap	Gap	Gap
	Social values	Low	Moderate	Gap

4.3.3 Survey of experts

The survey participation rate was 58% (34 surveys completed out of 59 requests). The respondents represented regional councils (10 respondents), CRIs (8), universities (7), consultants and independent research organisations (6), primary sector organisations (3) and central government (1). Two respondents declined to participate due to time constraints and one had an inoperable email address. Summaries of the most common responses to each survey question are listed below, with some additional, insightful comments by individual respondents.

Issue 1. Adequacy of mechanistic understanding

Q1. How do you rate the current state of mechanistic or process understanding of the effects of land use on ecological, cultural and social responses in freshwater and coastal environments?

There was a general consensus on several points concerning the state of mechanistic understanding:

- Our understanding of ecological responses to land-use pressures far exceeds that for cultural and social responses.
- Our knowledge about N losses from land and effects on N enrichment on freshwater state and values exceeds that for other contaminants.
- Estuaries and aquifers are understudied relative to rivers and lakes.
- There is a severe lack of knowledge about the environmental effects of specific land management practices.

Several respondents from regional councils considered that mechanistic understanding of land-use effects is generally adequate. This may reflect the need for council staff to make management decisions with incomplete information and high uncertainty. As one council scientist put it, the level of understanding is "good enough to act on". A related view was that conceptual understanding of PSI relationships is adequate, but the land-use data needed to shift from conceptual knowledge to quantitative models and forecasting was inadequate.

Q2. What improvements are needed?

Most of the responses to this question referred to the need to undertake research and monitoring in order to alleviate the problems identified in the first question, e.g., knowledge of PSI relationships in estuaries, lakes and aquifers need to reach the level that exists for rivers.

In addition, respondents identified the need for agricultural scientists who focus on contaminant losses from source areas to interact more closely with ecologists who focus on effects in receiving environments, and with the catchment modellers who can link sources to receiving environments.

Issue 2. Adequacy of land-use data

Q3. How do you rate the current state of data quality and quantity concerning land use?

Most respondents rated the current state of land-use data quality as moderate to poor. Several respondents noted that there are multiple sources of land-use data (e.g., council, CRI and MPI datasets) that could supplement primary sources such as AgriBase, but those additional data need to be compiled and processed into usable forms, and IP and privacy issues need to be resolved.

Several respondents noted that some types of data that are critically needed to link agricultural activities to freshwater are not included in Agribase. Furthermore, gaps in AgriBase are not well-covered in other existing databases. In particular, data related to horticulture and arable cropping were considered inadequate. Horticulture and arable data are particularly difficult to capture due to variable rotations within and between years. In addition, most land-use databases represent a complicated mixture of data from different sources, different methods (e.g., remote imagery, modelling, voluntary surveys), and different levels of reliability. These mixtures present challenges for assessing the effects of different land use pressures (e.g., empirical land-cover data versus modelled stock densities). Although most regional councils compile land-use data, several respondents noted that these data have limited detail and aggregating across regions is difficult due to the lack of a nationally consistent classification.

Several respondents noted that land use changes more rapidly than coarse-grained land cover, and that data users need more up-to-date land-use data.

Q4. How do you rate the current state of land-use data availability?

The consensus among respondents was that the availability of land-use data is low compared with the need, and that the primary cause is that most land-use data are privately held. Commercial protection of some land-use data reduces its availability, although to a lesser degree than for land management practices data (see Question 10). Several respondents noted that the costs to access some land-use databases exceeded the budgets of their organisations, and that the perceived lack of benefit for data owners in making land-use data available further reduces access.

Urban land-use data may be an exception to the availability issues set out above. Land-use data are held by councils with large urban centres and widely available to data users.

Q5. What improvements are needed?

Most respondents referred to the need for standardisation in collecting and classifying land-use data, increased availability to users, and regular updating. A stable long-term funding source would be needed to support these changes. Several respondents noted that land-cover data from LCDB are frequently used in lieu of land-use data because land cover is a workable proxy, and the data are easily available and free of charge.

For urban land use, land use zoning data held by individual councils would be valuable if aggregated into a national urban land use zone layer.

One respondent suggested that land-use data should be maintained and provided to users by a single, non-commercial organisation. The maintenance of LCDB data by Manaaki Whenua Landcare Research and its provision through the LRIS portal could be a model for this approach.

Issue 3. Adequacy of land-cover data

Q6. How do you rate the current state of data quality and quantity concerning land cover?

Almost all respondents rely on LCDB, and most consider it sufficient for their purposes. Limitations that were noted about LCDB included poor temporal resolution due to the long intervals between versions, coarse classification detail and the scarcity of ground-truth data.

As with land-use data, land-cover data for analyses of urban land-use pressures appears to be an exception. A water quality scientist commented "Urban land-cover data quality and quantity is variable. The LCDB class 'built up' is poor in terms of suitability for investigating relationships with specific urban covers or % impervious (since 'built up' includes all urban land covers other than large parks etc). However, some councils have GIS layers defining the extent of impervious covers, such as building footprints and roads. From this type of data, combined with land use zoning data, it is possible to delineate and classify urban areas into the types of land covers required for predictive modelling of contaminant loads and/or for use in correlative studies of land cover versus receiving environment state".

Q7. How do you rate the current state of land-cover data availability?

The near unanimous view is that the public availability of LCDB greatly benefits stakeholders in land and water management.

Q8. What improvements are needed?

There was a consensus among respondents that LCDB5 is urgently needed as a first step, because the 2011/2012 imagery used for LCDB4 is no longer accurate for some cover classes. Following the update to Version 5, several respondents recommended annual updates, and possibly seasonal updates as image processing increases in efficiency.

Another consensus view was that LCDB needs more detailed land cover classes and greater spatial resolution. Specific suggestions included more indigenous vegetation classes, distinction of wetland from pasture, irrigated from non-irrigated pasture, and delineation of riparian vegetation.

With regard to urban land cover, the water quality scientist quoted above suggested that LCDB should replace the current 'built up' cover class with urban impervious and urban pervious classes.

A university respondent noted "I do not think we are bringing through a cohort with the capability to exploit this technology as well as what is required for land use assessment and planning purposes".

Issue 4. Adequacy of land management practices data

9. How do you rate the current state of data quality and quantity concerning land management practices (LMPs)?

The consensus view was that LMP data are inadequate and that improving the situation will require major investments of time and funding. The problems most frequently identified were unreliability due to voluntary reporting, lack of systematic approaches for collecting and classifying LMP data, and severely limited availability.

With regard to voluntary reporting, one respondent noted that there is a risk of bias in LMP datasets, as land owners who are managing their land well and are well informed at a policy and practice level are probably most willing to share data.

It was suggested that regional councils should be the primary sources of LMP data, and most councils already compile LMP data, at least informally. However, there is wide variation between councils in data sourcing, structuring, internal use and external provision.

One respondent noted that the application of detailed LMP data for developing PSI relationships has been largely restricted to case studies such as catchment research projects (e.g. Best Practice Dairy Catchments), and regional council projects (e.g. SLUI).

Several respondents noted that the problem of rapid temporal change noted for land use classes information is even more severe for land management practices, which can change daily.

Q10. How do you rate the current state of availability of land management practice data?

The consensus view was that the availability of spatial datasets of rural and urban LMPs at catchment, regional and national scales is minimal. Three causes for this problem were identified: lack of standard procedures for data collection and classification, rapid change in many variables, and the fact that a large proportion of LMP data are privately owned and/or commercially sensitive.

Numerous respondents commented on the privacy issue, particularly in terms of agricultural practices. They pointed out that private landowners see little benefit in providing data (i.e., they see no value proposition), they are concerned that their own data may be used against them by regulators, and they perceive a lack of consistency in the type of data they are expected to provide.

In contrast to the view that LMP data are largely unavailable, a smaller group of respondents noted that LMP data collection is growing rapidly, driven in part by expanding requirements for farm environmental plans. In addition, spatial models of some LMP variables (e.g., stock unit density) have been developed recently. However, growth of data does not necessarily equate with increased availability as few LMP datasets are in the public domain.

Q11. What improvements are needed?

Most respondents identified the need for standard procedures for collecting and classifying or coding LMP data. Additional needs included agreeing on the specific LMP variables that are should be measured (i.e., identifying LMPs that affect freshwater and coastal environments), and ensuring that central government or a research institution is responsible for managing national LMP data.

A more general, but insightful comment was that evidence is needed that LMP data can be linked to environmental outcomes. This evidence would help build a rationale for overcoming the multiple challenges associated with LMP data. A related step is addressing the reluctance of land owners to provide data; land owners need clarity about the benefits of providing data, and they need simple and reliable reporting systems. One of the benefits identified by respondents was that land owners can demonstrate that they have implemented best management practices and mitigation strategies.

Issue 5. Gaps in mechanistic understanding

Q12. What are the top three gaps that impede our understanding of land-use effects on freshwater and coastal environments?

Several researchers noted that New Zealand land and water management and biophysical science is primarily focused on four contaminants (N, P, fine sediment, and faecal bacteria), and needs to increase the level of research on other contaminants (e.g., pesticides, metals, pharmaceuticals) and non-contaminant land-use pressures (e.g., flow regime alteration).

Several researchers identified multiple stressor effects (both interactive effects and cumulative effects) in freshwater and coastal ecosystem as major knowledge gaps.

Individual respondents identified a wide range of knowledge gaps, including:

- Nitrogen leaching.

- Surface water-groundwater exchange and its effects on N and P cycling.
- Effectiveness of mitigation systems on land (e.g., erosion control) and interventions in water bodies (e.g., flushing flows, managed lake openings, managed aquifer recharge).
- Temporal change in land use and land management practices.
- Measuring land use as a continuous variable.
- Indicators for measuring complex land use and land use intensification.
- Lag times and legacy contaminant dynamics.
- Relationships between land use pressures and cultural values for iwi/hapū.
- Spatial representation of cultural values and their current and future state.
- The dependency of cultural rights and interests of land use and ecosystem processes.

Issue 6. Data gaps

Q13. What are the top three priorities for improving the availability of data needed to understand land use effects?

Most of the responses to this question fell into two groups: improvements in monitoring and data generation, and improvements in data provision for users.

Recommendations related to data generation included the following:

- Synoptic measurements of pressure, state and impact variables.
- Improved monitoring of land management practices.
- Increased monitoring in estuaries and coastal zones.
- Mapping and monitoring critical source areas for contaminants.
- Agreement on indicators for reporting land use pressures and environmental responses.
- A prioritisation of the many potential measurement variables, to keep monitoring programmes affordable.

The most frequent recommendations related to the provision of data included the following:

- User-friendly and robust data services.
- Removing proprietary and privacy barriers to data provision.
- Integration with data systems used by Manawhenua.
- Data-sharing agreements among data users (e.g., between CRIs).
- Integration of datasets from multiple sources.
- Rigorous checks on the accuracy of modelled land use and LMP data.

Q14. What are the biggest obstacles to acquiring data?

There was a general consensus that privacy issues, data quality (i.e., accuracy) and costs are the greatest impediments for data users. Some respondents noted that limits on data provision are partly due to limited environmental monitoring. This could be addressed through a combination of expanded SoE monitoring networks, greater investments in autonomous sensors, greater use of remote sensing for monitoring freshwater and coastal environments, and syntheses of modelled and observed data. For land owners, the biggest obstacle to providing data was considered to be uncertainty about how the data will be used.

5 Recommendations

5.1 Introduction

In this section we provide recommendations for improving knowledge of PSI relationships and filling related knowledge gaps. Some of the recommendations apply to all freshwater and coastal environments, and some are specific to rivers, urban streams, lakes, aquifers and estuaries and coastal zones. In addition to the recommendations set out here, which are based on receiving environments, multiple recommendations concerning the collection, processing and provision of land cover, land use and land management practice data were provided by the respondents to our email survey, and are set out in Section 4.

5.1.1 All freshwater and coastal environments

Expand the scope of the PSI model as a framework for organising information and prioritising management. This recommendation applies to all receiving environments. The PSI model is appealing because it is simple, linear and logical, but it does not effectively account for many of the complexities that characterise aquatic ecosystems (e.g., time lags between pressures and responses, multiple stressor situations, resistance to recovery when pressures are reduced).

Evaluate the effectiveness of industry best management practices and regional and national policies and standards for reducing adverse effects of land use on freshwater and marine environments.

Improve the state of knowledge about mitigation systems and interventions designed to reduce the impacts of land use pressures on freshwater and coastal ecosystems. There is a wide range of knowledge gaps across these systems, including scaling up from pilot- to operational scales (e.g., denitrifying bioreactors), evaluating long-term performance (e.g., riparian planting and constructed treatment wetlands) and testing effectiveness of modelled mitigations (e.g., testing the effectiveness of environmental flows that are based on hydraulic-habitat modelling).

Advance the use of Māori indicators of freshwater and coastal conditions to develop associations between land use pressure and Māori values. Degradation of customary resources associated with aquatic environments, degradation of mauri (life force), and loss of cultural opportunities are issues of great concern for Māori. Scientific evidence to support these concerns is growing slowly (e.g., Harmsworth et al. 2011), but more evidence will strengthen the case for land-use management to protect cultural values.

5.1.2 Rivers

Increase the degree to which land-use pressures such as contaminant loss from land and hydrological alterations are linked to changes in river state and impacts, both statistically and in process models. Contaminant loss processes (e.g., leaching, runoff), hydrological processes (e.g., evapotranspiration, water yield) and ecological, social and cultural values in rivers are all major research areas for environmental scientists in New Zealand. However, the research carried out in these area is highly siloed and insufficiently joined up. This situation is an impediment to improving our ability to predict the responses of river values to contaminant losses and hydrological alterations upstream. The two key requirements are greater investment in catchment modelling and interoperable modelling, and greater emphasis on multidisciplinary research.

Increase the number of river monitoring sites in planted forest to improve the robustness of statistical analysis and national reporting. In national- and regional-scale analyses of river water quality and ecological conditions, monitoring sites are often grouped into land use or land cover classes. This approach treats monitoring sites as replicates within each class in order to make robust estimates of state and trends. Relatively few river monitoring sites are located in catchments dominated by planted forests so large-scale estimates for this class are generally uncertain.

Improve the state of knowledge about the effects of fertiliser use and tree thinning and pruning in planted forests on freshwater and marine environments.

Develop clear procedures for preventing over-allocation of river flows, and procedures for resolving over-allocation where it occurs, as stipulated by the NPS-FM.

Initiate a periodic monitoring programme to determine the presence and levels of contaminants in rivers, in addition to those included in regional council SoE monitoring programmes. In agricultural areas, target contaminants may include pesticides and agricultural antibiotics and other veterinary pharmaceuticals. In urban and industrial areas, target contaminants may include pesticides, petroleum products, solvents, pharmaceuticals, dissolved metals (in addition to copper and zinc) and personal-care products.

5.1.3 Urban streams

Compile spatial data layers with impervious and pervious cover, stormwater infrastructure, and peri-urban and suburban residential land. As a first step, a partial national dataset can be constructed using city and regional council data, but this will require substantial work to standardize data.

Improve and standardize definitions of urban land cover and land use classes (e.g., industrial, commercial, residential, built-up). At present, data users use a range of non-standard definitions.

Initiate long-term studies in catchments that are undergoing (or are planned to undergo) urban development. These studies are needed to track changes in environmental state and ecological, cultural and social values through the development phases. The same studies will be invaluable for evaluating the effects of future changes in urban environments. Those changes can include climate change, expanded use of water-sensitive urban design (WSUD), shifts from suburban sprawl to high density housing, changes in use of petrol and electric cars, improvements to public transport, and expanded use of low yields brakes, tyres and building materials.

Similarly, initiate multi-catchment studies in areas undergoing alternative forms of urban development (e.g., WSUD versus convention urban water management). We note that the lack of evidence that WSUD contributes to achieving environmental objectives has been identified as a barrier to WSUD uptake.

Improve the state of knowledge about contaminant yields for different land cover and land use categories, and for specific urban activities, source control measures (e.g., effects of roof replacement of stormwater metal concentrations), and mitigation systems.

5.1.4 Lakes

Focus lake research on impacts of land use pressure on ecological, cultural and social values in lakes. Most of the New Zealand lake studies to date have reported changes in lake state in response to land use pressures, rather than impacts on values.

Substantially expand lake state-of-environment monitoring. There are more than 3,800 large lakes in New Zealand. Less than 100 of these lakes are monitored regularly by regional councils to evaluate the state of water quality and ecological values and to track trends. The scarcity of regular, long-term monitoring data makes it difficult to develop robust pressure-response relationships for most lakes, which in turn impedes effective lake management.

Improve the degree to which regime shifts in New Zealand lakes (e.g., from clear and macrophyte-dominated to turbid and phytoplankton-dominated) can be predicted in advance. These predictions will facilitate the use of management actions aimed at preventing degradation, rather than relying on retrospective management to reverse degradation.

5.1.5 Aquifers

Improve the knowledge required to link groundwater at different points in a catchment to land use and land cover upgradient from each point. Several national scale data analyses have reported no detectable associations between chemical variables in groundwater and the overlying land use and land cover. These results are likely due in part to uncertainty about the capture zones of wells. In other words, it is not clear what area of land surface (and its corresponding land use) influenced the chemistry sampled in a given well. Delineating capture zones is a substantial modelling task, but is likely to produce valuable information.

As part of the same need to improve knowledge of land use effects on groundwater, long-term catchment studies in areas undergoing agricultural intensification or de-intensification are needed to develop quantitative PSI relationship. Alternatively, space-for-time substitution approaches can be used.

Expand the current research focus on groundwater N to include more work on other anthropogenic contaminants (e.g., metals, agricultural and medical pharmaceuticals, personal-care products, viruses), ecological response variables (e.g., stygofauna) and ecosystem services (e.g., attenuation of pathogens and other contaminants). This research will improve understanding about land use effects, but it is also critical to monitor a wider range of contaminants that pose ecological and human health risks.

5.1.6 Estuaries and coastal zones

The following recommendations are summarised and updated from a recent Ministry for the Environment report on state and trends in estuarine and coastal water quality and ecological conditions (Dudley et al. 2017).

The current national-scale estuarine and coastal monitoring network (composed of aggregated regional council sites) has few sites, and they are unevenly distributed across the classes of estuarine and coastal hydrosystems that exist in New Zealand. The number of sites within each class needs to be increased to facilitate comparisons between classes. As noted above for rivers, monitoring sites within classes function as replicates in these comparisons, and the statistical power of comparisons depends in part on site replication. The current network has very low statistical power.

The current estuarine and coastal monitoring network does not accurately represent the relative abundance of different hydrosystems in New Zealand. The lack of a representative network makes it difficult to make unbiased assessments of water quality and ecological conditions across regions or nationally. As with increasing statistical power, increasing representativeness will require more monitoring sites.

Contaminant loading to coastal hydrosystems should be assessed by monitoring water quality in terminal river reaches, within estuaries and on the adjacent open coast. Data from terminal river reaches and open coasts are needed to determine the relative importance of land and open ocean as contaminant sources to estuaries, and this information is needed to assess effects of land use on estuarine conditions.

There is a lack of national consistency in the measurement variables and methods used in regional council estuarine and coastal monitoring programmes. These inconsistencies reduce the data and sites that can be used in national analyses and can create regional biases. All estuarine and coastal monitoring programmes should use the National Environmental Monitoring Standards for coastal water quality.[14]

Ecological health indicators should be included in estuarine and coastal monitoring programmes to facilitate the development of water quality criteria (e.g., bands for NOF attributes). Candidate indicators include methods in New Zealand estuarine trophic index (Zeldis 2017a, b, c) and the New Zealand traits-based macroinvertebrate Index (Hewitt et al. 2012, Rodil et al. 2013).

Limit setting to reduce land-use effects on estuarine and coastal hydrosystems should account for differences among hydrosystem classes in sensitivity to land use pressure.

The first recommendation listed above for rivers also applies to estuaries and coastal zones: there is a need to increase the degree to which land-use pressures hydrological alterations are linked to responses in estuaries and coastal zones, both statistically and in process models. As with rivers, understanding these linkages can be impeded when research focused on separate components (e.g., contaminant loss from land, ecological effects in estuaries) is insufficiently joined up.

[14] National Environmental Monitoring Standards (NEMS). 2017. Water Quality Part 4: Sampling, Measuring, Processing and Archiving of Discrete Coastal Water Quality Data (http://www.nems.org.nz/documents/water-quality-part-4-coastal-waters).

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Case studies

Appendix A Manuherikia River

Summary

- Seasonal water scarcity in the Manuherikia catchment of Central Otago poses a risk of conflicts between agricultural land use and some ecological and social values associated with rivers.
- Ecological values include several species of endemic, endangered fish, bird and invertebrates, a diverse invertebrate fauna that is indicative of good ecosystem health, and generally good water quality.
- Social values include a nationally important trout fishery.
- Intensive water management via storage, abstraction and conveyance through races and river channels has made it possible to irrigate 25,000 ha of the catchment, but the catchment is overallocated and new regulations to resolve this are in development.
- Water management has strongly modified flow regimes in the Manuherikia River and its tributaries, included increased summer flows in the mainstem below the Falls Dam and depleted flow in tributaries due to abstraction and impoundment.
- Estimated large-scale surface water abstraction and proportions of catchments in high intensity land cover have been linked to changes in fish abundance and invertebrate community structure.

Setting

The Manuherikia River drains a 3035 km^2 catchment in Central Otago, bounded by the Hawkdun, Saint Bathans and Dunstan Ranges. Catchment elevation ranges from 2300 m above sea level to 150 m at the confluence with the Clutha River at Alexandra. The catchment has two major valleys separated by the Raggedy Range, the Manuherikia Valley, drained by the Manuherikia mainstem and tributaries, and the Ida Valley, drained by the Ida Burn, Pool Burn, Manor Burn and tributaries.

The Manuherikia catchment is in a continental climate zone with an annual mean air temperature of 10° C (range -20 to 35° C), very low annual rainfall at low elevations (< 500 mm/yr), and snow-dominated precipitation at high elevations (> 1000 mm/yr). The annual average rainfall at Alexandra is 340 mm, the lowest recorded in New Zealand, and the annual average deficit is 300 mm. The Manuherikia catchment is widely regarded as water scarce during the growing season, with agricultural intensification dependent on water storage and irrigation.

The Manuherika River has a variable planform that in turn creates a wide range of aquatic and riparian habitats. Upstream of Falls Dam, the river has a braided planform with multiple active channels on a braid plain that is up to 1 km across. This river section is the least modified and is used by numerous endemic birds as discussed below. Downstream of the dam, the river plan form ranges from semi-braided to strongly hillslope constrained, including the gorge section between Ophir and Springvale. In river sections that pass through intensively farmed areas, bank stabilisation by willows and other non-native riparian trees has reduced channel migration. The largest tributary, the Ida Burn, has a planform that ranges from semi-braided to hillslope confined, with willow-lined banks in some farmed sections.

Agriculture in the Manuherikia catchment

Land cover in the Manuherikia catchment is currently dominated by high producing exotic grassland on the valley floors and lower hillslopes, and low-producing exotic grassland and native tussock on higher and steeper slopes (Olsen et al. 2017). Land use in the Manuherikia catchment is currently dominated by sheep, beef and deer farming, viticulture, fruit growing and arable cropping, with a small area (approximately 10%) in dairy farming or dairy support. Sheep and beef stocking rates are relatively low in the non-irrigated upper reaches and higher on low-elevation, irrigated paddocks.

Agricultural land use in the Manuherikia catchment is supported by six irrigation schemes and their infrastructure, including multiple dammed reservoirs (Falls, Ida Burn, Pool Burn, Upper and Lower Manor Burn) and interconnecting races. The largest reservoir is behind Falls Dam (total storage 10.4 million m^3) on the Manuherikia mainstem, 59 km upstream from the Clutha River confluence. The irrigation schemes and private users abstract surface water through off-takes in the middle and lower Manuherikia River and tributaries. Approximately 25,000 ha of land in the lower catchment are currently irrigated.

With regard to freshwater ecosystems, there are two broad classes of agricultural land-pressures of concern in the Manuherikia catchment, input of land-based contaminants (primarily nitrogen, phosphorus, fine sediment and pathogenic faecal bacteria) and river flow alteration due to agricultural water management. Water management and flow alteration discussed in detail in the following section.

The risk of input of land-based contaminants to freshwater systems can be partially assessed by considering the risk of loss from the land surface. Nitrogen and phosphorus losses under current land use have been estimated using OVERSEER models for five farm systems (Watkins et al. 2015). Estimated annual nitrogen loss rates ranged from 3 to 63 kg N ha^{-1} yr^{-1}, and annual phosphorus loss rates ranged from 0.1 to 3.2 kg P ha^{-1} yr^{-1} (Table A-1).

Table A-1: Estimated annual nitrogen and phosphorus loss rates from case-study farms in the Manuherikia catchment. The predominant mechanisms for nutrient loss from each farm system are shown in descending order of contribution to the total loss rate. No information about mechanisms of nutrient loss was provided for dryland sheep farms. Data from Watkins et al. 2015.

Farm system	Total N loss (kg N ha^{-1} yr^{-1})	Total P loss (kg P ha^{-1} yr^{-1})
Irrigated sheep	14 (urine leaching = irrigation outwash > fertiliser/effluent leaching > overland flow)	2.8 (overland flow > irrigation outwash)
Irrigated dairy support	37 (urine leaching > fertiliser/effluent leaching > irrigation outwash)	3.2 (overland flow > irrigation outwash)
Irrigated mixed arable	63 (fertiliser/effluent leaching > irrigation outwash = urine leaching)	3.2 (overland flow > irrigation outwash > fertiliser/effluent leaching)
Partially irrigated sheep	9 (fertiliser/effluent leaching > urine leaching > irrigation outwash)	0.8 (overland flow > irrigation outwash)
Dryland sheep	3	0.1

Contaminant loss from land, as exemplified in, is the first in a sequence of steps by which land-based contaminants are transported to surface water bodies. Those steps include infiltration, groundwater and overland flow, and attenuation by biogeochemical process. Information about contaminant input to surface water in the Manuherikia catchment is rare, but a study of Thompson's Creek (a tributary of the Manuherikia River near Ophir) by Wilcock et al. (2011) is an exception. In this study dissolved inorganic nitrogen (DIN), dissolved reactive phosphorus (DRP) and *E. coli* concentrations at a site upstream of border-dyke irrigation were compared with a downstream site that received irrigation outwash. The median DIN, DRP and *E. coli* concentrations downstream were 4, 9 and 39 times higher than the upstream concentrations. Note that border-dyke irrigation is being replaced with spray irrigation in the Manuherikia catchment, and this may reduce contaminant discharge to surface water. However, Thompson's Creek remains the most severely nitrogen, phosphorus and *E. coli*-enriched site among the regional council monitoring sites in the catchment, as discussed below.

Water resources management

Most of the current water allocations in the Manuherikia catchment are authorised by historic mining rights, or 'deemed permits'. As of February 2017, there were 213 permitted surface water takes in the catchment, for a total allocation of approximately 32 $m^3 s^{-1}$. Actual takes in 2012 were estimated to total 16 $m^3 s^{-1}$ (Olsen et al. 2017). The deemed permits expire in 2021 and resource consents from Otago Regional Council (ORC) will be required for most abstraction. In the same time period, ORC is required to set al.location limits under the National Policy Statement for Freshwater Management (NPS-FM). The NPS-FM specifically requires cases of over-allocation to be resolved.

Schedule 2A of the current regional Water Plan identifies minimum flows and allocation rates for specified sites in the region (as at October 2013). The Manuherikia River upstream of Ophir currently has a minimum flow of 0.82 $m^3 s^{-1}$, but minimum flows have not been set downstream of Ophir or on any tributaries. The Manuherikia River has a primary allocation limit of 3.2 $m^3 s^{-1}$ from the headwaters to the Clutha River confluence, this primary allocation limit is currently exceeded.

To address the over-allocation problem, ORC is developing a plan change, "Plan Change 5C (Manuherikia Catchment: Integrated Water Management)"[15]. Plan Change 5C would enable setting minimum flows at one, two or three sites. One of those sites is to be Manuherikia at Ophir, where the minimum flow would increase from the current 0.82 $m^3 s^{-1}$ to between 1.5 and 2.5 $m^3 s^{-1}$. This increase would likely improve conditions for instream ecological and recreation values, and potentially reduce supply reliability for water users. The Falls Dam reservoir is periodically entirely drawn down under current conditions and it could do so more frequently if minimum flows are increased. The proposed plan change retains the primary allocation limit of 3.2 $m^3 s^{-1}$ from the headwaters to the Clutha River confluence. The proposed plan change is currently in the community consultation phase.

Several proposals have been advanced recently to enable increased water storage in the Manuherikia catchment and an expansion of the irrigated area by up to 21,000 ha. The Manuherikia Catchment Water Strategy Group was set up to develop the proposals and oversee feasibility studies. The proposals with the largest potential effects on water resources are for raising the height of the Falls Dam and constructing new dams on Hopes Creek and the upper Ida Burn. In addition, proposed

[15] https://www.orc.govt.nz/plans-policies-reports/regional-plans/water/manuherikia-catchment-integrated-water-management

improvements in irrigation scheme efficiency and piped supplies may reduce leakage to groundwater, and subsequent groundwater discharge to the Manuherikia River.

Flow regime

Flow regimes in the Manuherikia River and its major tributaries are strongly modified by surface and groundwater abstraction, reservoir storage, transfer through irrigation races, and flow augmentation in natural channels. The flow regime in the Manuherikia mainstem downstream of Falls Dam is largely controlled by the dam operation. The Falls Dam is located near the top of the catchment, but intercepts most of the surface water flow in the catchment; the median inflow is 5.3 $m^3 s^{-1}$. Water is released from the dam to convey irrigation water to water users downstream, these releases elevate the river flow downstream of the dam during the irrigation season, and this flow is progressively reduced with distance downstream by off-takes. The Falls Dam has minimum residual flow of 0.5 $m^3 s^{-1}$.

In contrast to the augmented flow below Falls Dam during the irrigation season, abstraction on tributaries that lack minimum flows severely depletes or dries these tributaries during the irrigations season (Otago Regional Council 2012). Some of these tributaries may have naturally intermittent flow regimes, but abstraction will increase the duration and/or frequency of intermittence.

Flow regime alteration by water users can be evaluated using differences between observed and naturalised flow; naturalised flows are estimates of flow in the absence of abstractions or other modifications. Naturalisation can be applied to either flow time series or to the many flow metrics used to summarise flow regimes. ORC used naturalised 7-day mean annual low flow (7dMALF) as an evaluation metric (Olsen et al. 2017). The 7dMALF is the average of the lowest arithmetic mean of seven consecutive daily average flows. Comparisons of observed and naturalised 7dMALFs are useful indicators of the effects of flow depletion on aquatic habitat availability (Jowett et al. 2008). In the Manuherikia case, observed 7dMALFs at sites downstream of Falls Dam were 23 to 69% of the naturalised 7dMALFs, indicating a strong reduction in natural low flow levels. The observed 7dMALF immediately below Falls Dam was approximately 10% higher than the naturalised 7dMALF, due to summer release of impounded water.

Recent daily flow data from a recorder on the Manuherikia River at Ophir are shown in Figure A-1. The hydrograph indicates that large seasonal fluctuations occur at this site, with winter flows characterised by floods, freshes and recessions, and summer flows characterised by stable low flows and occasional floods caused by convective rainstorms (e.g., the summers of 2014 and 2017 in Figure 2). Strong seasonal variation in flows are typical for New Zealand rivers with snowmelt or winter rain-dominated high flows and relatively low groundwater-fed base flows. However, the summer flow regime at the Manuherikia River at Ophir is also strongly influenced by the Falls Dam, 35 km upstream, and by abstraction and return flows between the dam and the recorder. The 7dMALF at this site is 2.2 m3 s-1. During particularly dry summers, flow may fall below the 7dMALF for extended periods. For example, daily flows at the Manuherikia at Ophir were less than the 7dMALF for 63 consecutive days during the drought of 1998-1999 (Caruso 2001).

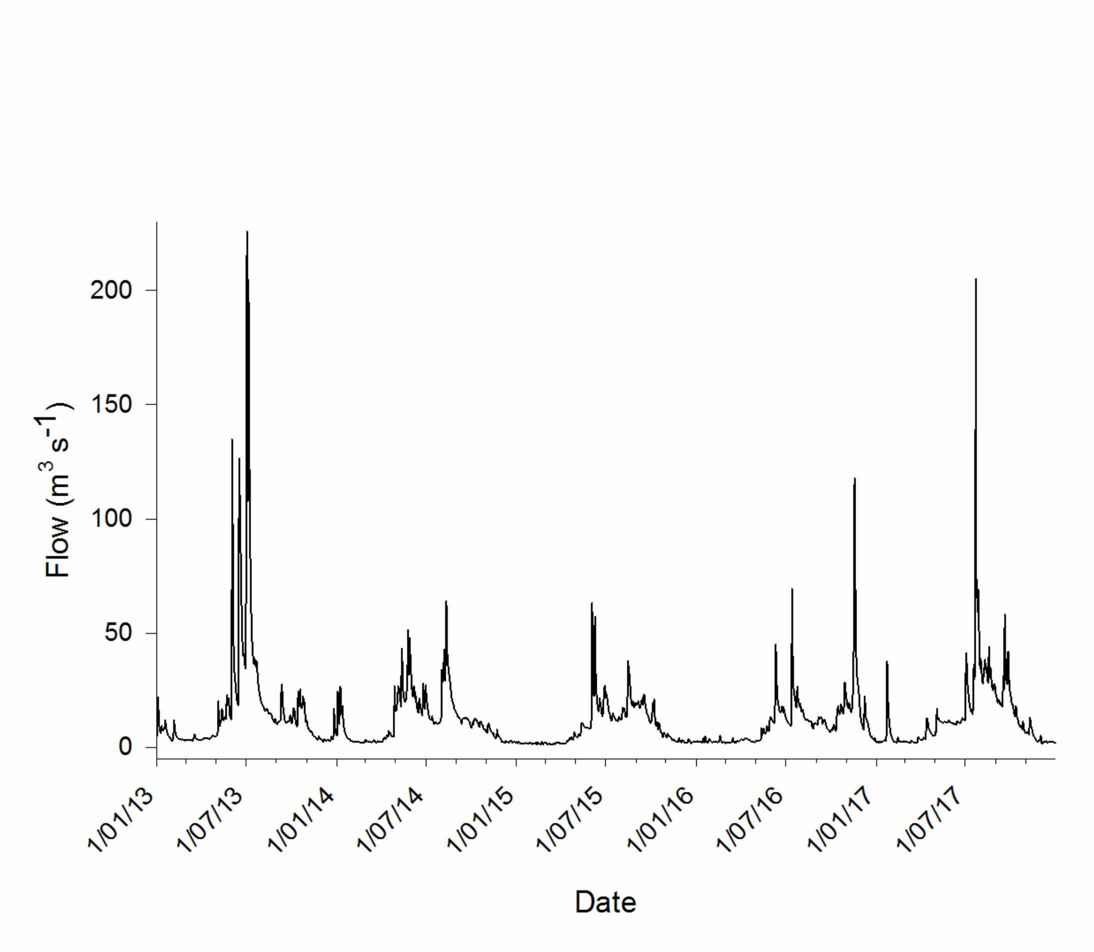

Figure A-1: Hydrograph for Manuherikia River at Ophir, 2013-2017.

Instream values

Water quality

River water quality in the Manuherika River is monitored at five sites by Otago Region Council as part of the council's state of environment monitoring programme (Table 2). Physical and chemical water quality variables at these sites are monitored monthly. Water quality state at the sites are posted on the ORC water quality reporting website[16]. Several other sites are monitored only for microbiological water quality during the bathing season. A summary of current water quality state at the five monitoring sites is in Table 2. ORC has set numeric limits for five water quality variables for sites in each of five Receiving Water Groups in the region (i.e., freshwater management units); the sites in the Manuherikia catchment are in Receiving Water Group 2 and the limits are shown in the Table 1 heading. ORC assigns water quality grades (from poor to excellent) based on the number of variables for which the current state complies with the limit. The Manuherikia at Blackstone and Dunstan Creek at Beatties Road are graded excellent, the Manuherikia at Galloway is graded good, Manuherikia at Ophir is graded fair, and Thompson's Creek is graded poor.

[16] https://www.orc.govt.nz/plans-policies-reports/reports-and-publications/water-quality/state-of-the-environment-water-quality-reports

Table A-2: Water quality state at ORC state of environment monitoring sites in the Manuherikia catchment. Values are medians from monthly samples over the period July 2012 to June 2016. Values in parentheses are the numerical limits set out in Schedule 15 of the ORC Water Plan. Median values in red exceed the Water Plan limit. NO3-N: nitrate-nitrogen, NH4-N: ammoniacal nitrogen, DRP: Dissolved reactive phosphorus, *E. coli*: faecal indicator bacterium *Escherichia coli*, TN: total nitrogen, TP: total phosphorus. NA: not applicable, as there are no numerical limits for TN and TP.

Monitoring site	Grade	NO3-N (mg/L) (0.075)	NH4-N (mg/L) (0.10)	DRP (mg/L) (0.01)	*E. coli* (cfu/100 ml) (260)	Turbidity (NTU) (5)	TN (mg/L) NA	TP (mg/L) NA
Manuherikia at Ophir	Fair	0.067	0.019	0.037	320	3.7	0.35	0.06
Thompson's Creek	Poor	0.178	0.024	0.077	1100	5.6	0.76	0.15
Manuherikia at Blackstone	Excellent	0.004	0.008	0.005	170	5.0	0.13	0.02
Dunstan Creek	Excellent	0.052	0.008	0.005	82	0.9	0.13	0.01
Manuherikia at Galloway	Good	0.025	0.010	0.018	170	2.8	0.25	0.03

The nutrient concentrations in Table E-1 can also be compared to the water quality 'trigger values' for upland sites set out in the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC 2000, Table 3.3.10). The nitrate-N trigger value was exceeded at Thompson's Creek, the ammoniacal-N and total N trigger values were exceeded at Thompson's Creek and the Manuherikia at Ophir, and the DRP and TP trigger values were exceeded at Thompson's Creek, the Manuherikia at Ophir and the Manuherikia at Galloway.

Biological values

Fish

Nine native fish species occur in the Manuherikia River including bullies (*Gobiomorphus* spp.), koaro (*Galaxias brevipinnis*), longfin eels (*Anguilla dieffenbachia*) and three endangered, non-diadromous galaxiids (*Galaxias anomalus*, *Galaxias paucispondylus,* and *Galaxias* sp. D) (Goodman et al. 2014). The river also supports a significant brown trout (*Salmo trutta*) fishery, discussed below. While brown trout represent important sports fisheries in many New Zealand rivers, including the Manuherikia, trout predation also poses a threat to native fish populations (McIntosh et al. 2010). Brown trout occur both above and below each of the major irrigation dams in the Manuherikia catchment, so these dams do not provide a refuge for vulnerable native fish. Despite this, Leprieur et al. (2006) reported that *G. anomalous* populations have minimal overlap with brown trout in the Manuherikia catchment. Instead, *G. anomalous* and brown trout appear to be separated by hydraulic habitat conditions; *G. anomalous* are most likely to occur in reaches where water abstraction causes severe flow reductions, including the formation of isolated pools with high water temperatures and low dissolved oxygen; brown trout are excluded by these habitat conditions (Leprieur et al. 2006).

Aquatic macroinvertebrates

Lange (2014a) collected a total of 108 aquatic macroinvertebrate taxa in the Manuherikia River and tributaries, of which 85 were aquatic insects. None of the taxa collected by Lange are listed as threatened or at risk by the Department of Conservation (Grainger et al. 2013). However koura

(Paranephrops zealandicus) occurs in the Manuherikia catchment and is classified as at risk – declining.

Aquatic invertebrates are monitored annually at three ORC state of environment sites in the Manuherikia catchment, Dunstan Creek at Beatties Road, Manuherikia at Blackstone and Manuherikia at Ophir. Several metrics are used to provide an indication of ecosystem health based on the presence and abundance of macroinvertebrate taxa and their tolerance to nutrient enrichment and organic pollution. The metrics used by ORC are number of taxa, the number of taxa consisting of pollution sensitive aquatic insects in the orders Ephemeroptera, Plecoptera and Trichoptera (termed 'EPT richness'), the New Zealand Macroinvertebrate Community Index (MCI) and the Semiquantitative Macroinvertebrate Community Index (SQMCI) (Stark and Maxted 2007). A summary of the most recent macroinvertebrate metric scores at the three monitoring sites in the Manurherikia catchment is given in Table A-2. The observed range of MCI and SQMCI scores in New Zealand have been subdivided into classes ranging from poor to excellent. The MCI and SQMCI scores were good and excellent, respectively, at Dunstan Creek and Manuherikia at Ophir, and fair and excellent, respectively, at Manuherikia at Blackstone. Approximately half of the taxa collected at each site were EPT taxa. These results suggest that ecosystem health at the monitoring sites was relatively good at the time of sampling.

Table A-3: Macroinvertebrate community metric scores at monitoring sites in the Manurherikia catchment, Summer 2016-2017.

Monitoring site	Number of taxa	EPT richness	MCI	SQMCI
Manuherikia at Ophir	25	13	111	6.1
Manuherikia at Blackstone	26	13	98	6.6
Dunstan Creek	23	13	120	7.2

Birds

Several species of endemic birds use islands in the Manuherikia River for nesting and foraging habitat, including black-fronted terns (Chlidonias albostriatus), South Island pied oystercatchers (Haematopus ostralegus), banded dotterels (Chandarius bicintus), and black-billed gull (Larus bulleri). These species are classed as threatened or at risk by the New Zealand Department of Conservation: black-fronted terns are nationally endangered, South Island pied oystercatchers as declining, and banded dotterels as nationally vulnerable. In addition, several non-threatened native bird species use Manuherikia River islands, including large numbers of black-backed gulls (Larus dominicanus). The braided section of the river is internationally important for black-fronted terns, but repeated surveys indicate that breeding populations of this species have declined recently (O'Donnell et al. 2011).

Fisheries

Four sports fish occur in the Manuherikia River and its tributaries and reservoirs, brown and rainbow trout, brook char and perch. The Manuherikia mainstem supports economically important brown and rainbow trout fisheries, including a backcountry fishery upstream of Falls Dam (Otago Fish and Game 2015). The tributaries that drain the Ida Burn Valley do not support trout fisheries due to low fish numbers and small sizes (ORC 2012).

Effects of land use on instream values

Quantitative relationships between land use and instream values in the Manuherikia catchment have been identified in two studies, one focused on on aquatic macroinvertebrates (Lange et al. 2014a) and the other on brown trout and native upland bullies (*Gobiomorphus breviceps*) (Lange et al. 2014b). In both studies, two catchment-scale explanatory variables were used to quantify effects of land-use pressur: the proportion of land in intensive agriculture in the catchment upstream of each sampling site, and the proportion of stream flow abstracted in the upstream catchment. The proportion of catchment area classed as LCDB-2 'high-producing exotic grassland' was used to estimate intensive agriculture. The proportion of abstracted stream flow was estimated as the difference between modelled natural stream flow and current flow during irrigation seasons. The two modelled flow scenarios were based on the ACRU model set up for the Manuherikia (Kienzle and Schmidt 2008). The sampling sites represented a gradient in percent cover in intensive land use from 0 to 95% and a gradient in abstracted stream flow from 0-92%. In addition to the two land-use variables, three instream predictor variables were used in both studies: nitrogen and phosphorus concentrations, deposited fine sediment, and instantaneous stream flow at the sampling sites. Generalised linear models were used to quantify relationships between the land use and instream explanatory variables and fish and invertebrate abundance and composition.

The strongest relationships linking fish presence and density and invertebrate community structure to land use were:

- A unimodal relationship between % intensive agriculture and trout presence (increasing probability of presence as intensive agriculture increased from 0 – 20% cover, then decreasing to absence at 40% cover.
- A negative relationship between % flow abstracted and trout presence.
- A negative relationship between % flow abstracted and trout density.
- Negative relationships between % intensive agriculture and invertebrate taxon richness, functional diversity, EPT richness, % EPT and MCI score.
- Negative relationships between % flow abstracted and EPT richness and MCI score.

The negative relationships between trout presence and density and % flow abstracted is consistent with observations that trout are excluded from stream reaches where flow is severely reduced by natural processes or abstraction or natural recession (Leprieur et al. 2006, ORC 2012). No relationships were detected between upland bully presence or density and either land use variable. This observation suggests that upland bullies tolerate a wider range of agricultural land-use pressure than brown trout, but the two explanatory variables, % flow abstracted and % intensive agriculture, are very general and more specific land-use pressures may influence upland bully distribution.

Flow management and water use in the Manuherikia catchment appear to have a combination of positive and negative effects on aquatic biota. For example, flow supplementation through the release of water from storage during the irrigation season, in combination with increased minimum flow requirements, are likely to reduce flow intermittence, reduce water temperatures, increase hydraulic habitat (i.e., increased depth, wetted width and velocity), and possibly improve chemical water quality through dilution. However, extreme low flows resulting from abstraction and naturally low runoff also provide some native species with a refuge from predatory brown trout (Leprieur et al. 2006, ORC 2012). One such tradeoff was set out in an Otago Regional Council investigation of flow requirements in the lower Ida Burn: "Optimal habitat for longfin eel includes pools for daytime refuge, and riffles and runs for feeding at night. However, if these habitats are maintained throughout the irrigation season it is predicted that brown trout will persist in this reach and the local population of Central Otago roundhead galaxias will become extinct" (ORC 2012). In view of these tradeoffs, it may be necessary to prioritise instream values and regulate flow regimes to optimise the high-priority values.

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Appendix B Wairewa/Lake Forsyth

Summary

- Te Wairewa/Lake Forsyth is a coastal lake/lagoon in Canterbury which is in a supertrophic state partially due to current and historical land use practices in its catchment.
- High nutrient loads result in the lake being a "flipping" lake, subject to switching between a plankton-dominated, turbid state (usually exhibiting toxic cyanobacteria blooms) and a macrophyte-dominated, clear water state.
- The presence of cyanobacteria blooms is attributed to current catchment phosphorus loads and substantial internal phosphorus loading, which is partially a legacy due to poor historical land use practices in the catchment.
- A new regional Land and Water Plan targets catchment phosphorus loads in an attempt to restore the lake to an improved, eutrophic state by 2030.

Introduction

Te Wairewa/Lake Forsyth (henceforth called Wairewa) is a supertrophic, intermittently closed and open lake/lagoon (ICOLL) located in a small, steep catchment on the Banks Peninsula. It has a surface area of 6.3 km^2 and a maximum depth of around 3 m, although its water level varies with the opening and closing of its outlet to the sea. Its 110 km^2 catchment, which drains part of the volcanic Banks Peninsula, is mostly steeply sloped and rocky, with shallow soils, except in the lowest altitudes (Figure B-1).

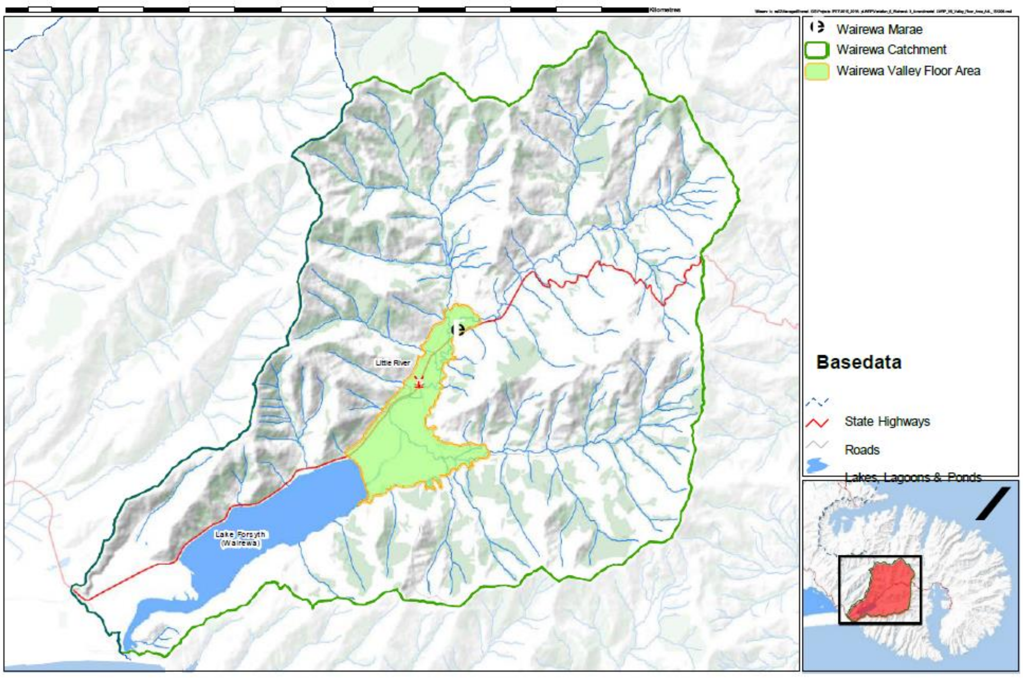

Figure B-1: Wairewa/Lake Forsyth and its catchment. Reproduced from ECAN (2017).

The lake is a tribal taonga (treasure) for Kāti Irakehu and Kāti Makō mainly because of the abundance of tuna/eels that has sustained local Maori for generations. In the 1840s, at the time of European settlement of the region, the barrier bar at the Wairewa outlet was very narrow when the lake was closed, suggesting that the lake was undergoing a change from being an embayment open to the ocean to being an ICOLL (Soons 1998). Since the mid-19th century artificial openings of the lake have been carried out to regulate water levels and allow for fish migrations.

Lake trophic state and dynamics

Much has already been written about historical land use in the catchment and its effects on the lake (e.g., Pyle 1992, Main 2002, Woodward and Shulmeister 2005, Flint 2007, Schallenberg and Schallenberg 2013). Briefly, the lake's poor condition is due to a combination of factors including the relatively phosphorus-rich soils found in the catchment (Lynn 2005) (Figure B-2), the historical and current land uses which have enhanced the loads of sediment and phosphorus to the lake (Hamilton and Mitchell 1997; Davie undated), the recycling of historical phosphorus loads from the lake bed into the water column (Waters 2016), the geochemical effect of saline intrusions to the lake (Schallenberg 2014; Waters 2016), and the intermittent nature of the flushing of the lake water to the sea afforded by the managed opening of the lake outlet (Schallenberg 2014, Waters 2016).

Since the early 1900s, the lake has been in a nutrient-enriched state (Figure B-3) and the tuna/eel fishery is considered to have suffered as a result (Davie 2005). The pressures listed above (e.g., erosion-prone phosphorus-rich catchment, land use, salinity, restricted flushing) have contributed to a lake condition that can be quite variable from year-to-year, but is generally characterised by a supertrophic state (Figure B-3), resulting in frequent toxic cyanobacterial (e.g., *Nodularia spumigena*) blooms.

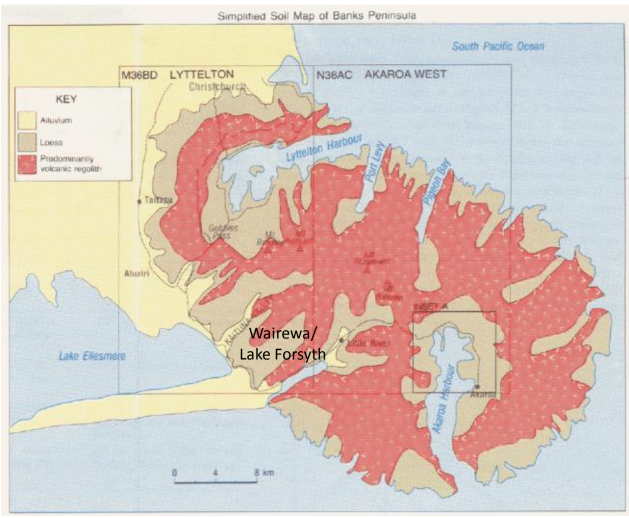

Figure B-2: A simplified soil map of the Banks Peninsula showing the dominant volcanic-derived soils (red shading) in much of the Wairewa catchment. Reproduced from Lynn (2005).

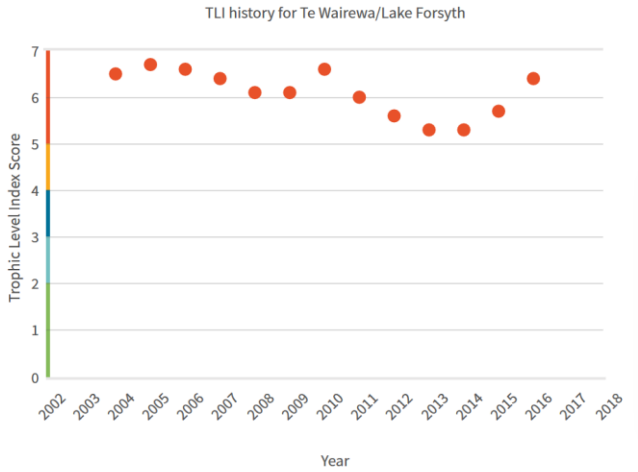

Figure B-3: Trophic state (TLI) of Wairewa/Lake Forsyth from 2003 to 2016. Scores > 5 indicate supertrophic conditions (very poor water quality). Data and graph from the LAWA website (https://www.lawa.org.nz/explore-data/canterbury-region/lakes/te-wairewalake-forsyth/).

In the period 2003 to 2016, Wairewa's annual average trophic state varied by almost two TLI units (varyng from highly supertrophic to almost eutrophic). This illustrates the phenomena of regime shifts and alternate stable states, which are common in shallow lakes (Scheffer 2004, Schallenberg and Sorrell 2009). Many shallow lakes impacted by high nutrient loads undergo regime shifts between macrophyte-dominated clear water states and turbid, phytoplankton dominated states (Schallenberg and Sorrell 2009). Wairewa was identified by Schallenberg and Sorrell (2009) as a lake which undergoes such regime shifts and the period from 2012 to 2015 in Figure B-3 highlights the effect of macrophyte recolonisation of the lake after a period of phytoplankton dominance and very low macrophyte biomass.

Such "flipping" lakes can be understood to have a pressure-state (P-S) relationship that accommodates two alternate stable states within a specific range of pressure. For example, Wairewa may exhibit a P-S relationship like that shown in Figure B-4, where within a specified range of nutrient loading, the lake state is unstable and may alternate back-and-forth between two relatively stable states, lasting for a number of years (Scheffer 2004). Schallenberg and Sorrell (2009) analysed 37 "flipping" lakes and 51 non-flipping lakes and showed that the prevalence of lake flipping in New Zealand shallow lakes was positively related to the % of the lake's catchment that is in pasture and was negatively related to the % of the lake's catchment that is in forest. At the time of Schallenberg and Sorrell's (2009) analysis, the catchment of Wairewa constituted 69% pasture and 28% forest, which supported the overall finding based on analysis of 88 lakes. Schallenberg (2014) explains in more detail how the theory of alternate stable states applies to the specific issues and attributes of Wairewa.

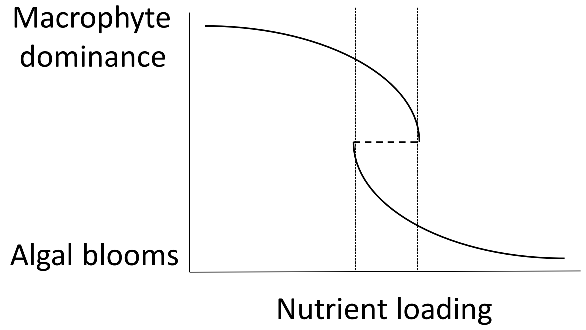

Figure B-4: A hypothesised pressure-state relationship for Wairewa, showing a range of pressure (nutrient loading) within which two states (macrophyte dominance occur. Based on the theory of alternate stable states (Scheffer 2004).

Catchment and in-lake nutrient drivers of phytoplankton blooms

Waters (2016) calculated a phosphorus budget for Wairewa over a 15-month period (from December 2012 to March 2014). During this time, over 7000 kg P yr^{-1} was transported to the lake from the catchment. Eighty percent of this load was due to particulate P transported during large flood events, particularly from the Okana River sub-catchment. Seventy percent of the external P

load was retained in the lake's sediments and the P budget indicated that the total flux of P from the lake bed sediment to the water column was greater than the external load that occurred during the summer months. The budget al.so indicated that increases in total P concentrations that fuelled algal blooms in the lake were consistent with this magnitude of internal P loading.

Waters' (2016) detailed study, which also included experiments and empirical measurements, suggested that the geochemical release of phosphorus from the lake's sediment due to the effects of dissolved oxygen depletion, elevated pH, and changes in salinity, substantially increased phosphorus availability in the lake, especially during summer - the time when phytoplankton blooms are more likely to occur. The substantial internal load that Waters (2016) calculated relates to the legacy of phosphorus now stored in the lake's sediments that was historically delivered to the lake from the catchment.

These results show that the P-S-I relationship in this lake, and similar lakes, can be complex and that the recycling of historical P loads within the lake may persist for a long time. The implications of this for lake restoration are that a beneficial effect of reduction of catchment phosphorus loads to the lake will take time to affect a reduction in phosphorus availability, and potentially phytoplankton blooms.

Schallenberg (2014) reviewed evidence that phytoplankton blooms in Wairewa are specifically regulated by phosphorus availability. Previous empirical and experimental studies suggested that phosphorus availability was the key driver of phytoplankton blooms. However, by examining antecedent nutrient conditions in the lake prior to significant algal blooms between 1996 and 2013, Schallenberg (2014) inferred that pulses of both phosphorus and nitrogen recorded in the lake had often preceded phytoplankton blooms. Thus, his conclusion was that both P and N availability (at different times or in simultaneously) contribute to phytoplankton blooms in the lake.

While a number of studies have focused on drivers of phytoplankton blooms in the lake, the "flipping" behaviour of the lake and the resulting occasional presence of macrophytes continues to influence the water quality of the lake in certain years. For example, in the years 2010 to 2015, the lake bed at the northern (inflow) end of the lake developed a substantial cover of macrophytes, which improved water clarity in the northern end of the lake, substantially improving the trophic state of the lake in general (Figure B-3) (Ngai Tahu 2014). However, by 2016, toxic cyanobacterial blooms had returned to the lake as the macrophyte beds reduced in abundance and cover (Stuff 2016), highlighting the alternate stable states that the lake is subject to under current nutrient conditions.

A new land and water plan targets suspended sediment and P loading

After consultation with Iwi and other locals and stakeholders, the Canterbury Regional Council brought into effect its Land and Water Regional Plan in 2017 with a specific set of goals and rules for Wairewa, known as Plan Change 6 (ECAN 2017). The Plan introduced some specific rules about land use in the catchment designed to facilitate recovery of the lake to the stated TLI goal of 5 (with an aspirational TLI target of 4). The main focus of the plan change was to reduce P inputs from the lower, flatter part of the catchment, where most of the sediment and P load to the lake originates (ECAN 2017). Among the initiatives supported are:

- Addressing the key source of phosphorus in the catchment by encouraging works to stabilise the banks of rivers in the Valley Floor Area.
- Preventing stock accessing the river banks in the Valley Floor Area so as to reduce bank erosion and collapse, and prevent animal effluent entering waterways.
- Intercepting phosphorus-rich sediment before it enters the lake through the construction of a sediment retention basin or wetland at the head of Te Roto ō Wairewa/ Lake Forsyth.
- Providing for lake investigations to address the phosphorus that is already present in the lake.
- Requiring community wastewater treatment systems to remove phosphorus from discharges where practicable, and minimise the volume of wastewater.
- Introducing a minimum flow and allocation limit for the Ōkana, Ōkuti and Takiritawai rivers and their tributaries that protects the ecosystems and cultural health of these water bodies.
- Setting a nitrate-nitrogen concentration for rivers that protects the existing high water quality.
- Providing for a lake opening and closing regime that maintains lake levels for flood control and land drainage, while recognising the cultural values of Ngāi Tahu. [ECAN 2017].

The Plan also explicitly defines outcomes to be achieved by 2030 for the tributary rivers (Table B-1) and the lake (Table B-2).

Table B-1: Freshwater outcomes to be achieved for Wairewa tributaries by 2030. Reproduced from ECAN (2017).

Freshwater Management Unit	River type	River	Ecological Health Attributes			Periphyton Attributes[3]		Siltation Attribute[3]	Human Health for Recreation Attributes				Cultural Attribute
			QMCI [1,3]	Dissolved oxygen [min % saturation] [min]	Temp [max] [°C]	Chlorophyll a maximum biomass [mg chl-a/m²] [no more than 17% of samples]	Filamentous algae >20mm [max cover of bed]	Fine sediment <2mm diameter [max cover of bed] (%)	Cyano-bacteria [% mat cover]	SFRG[2]	*E.coli* [*E.coli /100mL*] Annual median	95th percentile	
Wairewa catchment	Banks Peninsula	Ōkuti River Ōkana River Takiritawai River	5	90%	20	120	20	20	30	Good	260	260	Freshwater mahinga kai species sufficiently abundant for customary gathering, water quality is suitable for their safe harvesting, and they are safe to eat.

1. QMCI = Quantitative macro invertebrate community index.
2. SFRG = Suitability for Recreation Grade from Microbiological Water Quality Guidelines for Marine and Freshwater Recreational Areas 2003.
3. These attributes only apply to wadeable areas of wetted riverbed. For the purposes of this table, wadeable areas are defined as reaches of the river up to 600mm in depth

Table B-2: Freshwater outcomes to be achieved for Wairewa by 2030. Reproduced from ECAN (2017).

Lake Type	Ecological health Attributes				Trophic Attribute	Visual Quality Attribute		Human Health for Recreation Attributes				Cultural Attribute
	Dissolved Oxygen (min) Hypo-limnion saturation [%]	Epilimnion saturation [%]	Temp (max) [°.C]	Lake SPI [1] [min grade]	TLI2 (max) annual average]	Colour	Macrophytes	Cyanobacteria [either mm3/L or cells/mL] [80th percentile]	SFRG	E.coli [E.coli/100ml] Annual median	95th percentile	
Coastal lake	70	90	19	Moderate	**5**	The natural colour of the lake is not degraded by more than 5 Munsell units	The spatial extent of native macrophyte beds is showing an increasing trend	1.8	Good	260	260	Freshwater mahinga kai species sufficiently abundant for customary gathering, water quality is suitable for their safe harvesting, and they are safe to eat.

1. Lake SPI = Lake Submerged Plant Indicators from Clayton J, Edwards T (2002) Lake SPI: a method for monitoring ecological condition in New Zealand lakes (Technical report version 1 by NIWA).
2. TLI = Trophic Level Index from: Protocol for Monitoring Trophic Levels of New Zealand lakes and reservoirs (Report by Lakes Consulting, March 2000). The scale is from less than 1 (very low nutrients) to more than 7 (very high nutrients). The TLI is calculated at TLI3 (using TP, TN and Chl. a.).
3. SFRG = Suitability for Recreation Grade from: Microbiological Water Quality Guidelines for Marine and Freshwater Recreational Areas, Ministry for the Environment, June 2003.

Current and historical land use has undoubtedly contributed to the degraded state of Wairewa, but a number of other factors have strongly mediated the pressure-response relationship. The large reservoir of P in the lake sediments reflects historically high P loading from the catchment and this sediment P reservoir will delay beneficial outcomes resulting from the new rules which are intended to reduce catchment sediment and P loading and thereby restore lake water quality. In-lake remediation actions have been considered for this lake (Schallenberg (2014) and a new opening to the ocean has recently been cut to allow better flushing and hydrological management (Ngai Tahu 2014). Nevertheless, the target date of 2030 for the new Plan outcomes to be realised attests to the difficulties of restoring such degraded lakes.

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Appendix C Lake Hayes

Summary

- Lake Hayes is an iconic high-country lake that has become eutrophic due to elevated nutrient inputs from its catchment.
- The Lake Hayes catchment has been used for agricultural since late 1800s, but severe degradation was not observed until the late 1960s, when the bottom waters of the lake became anoxic and nutrients previously bound to the lake sediment began to dissolve into the water column.
- The acute degradation phase of the late 1960s followed the drainage and conversion of wetlands in the catchment farm land. This land-use change resulted in noticeable discolouration of the main inflow stream for many years.
- Catchment nutrient loads have stabilised in the past 20 years, but Lake Hayes remains eutrophic due to its internal phosphorus load.
- Recent evidence suggests that Lake Hayes could be entering a recovery phase, but severe algal blooms occur during most summers.
- New residential development plans in the catchment threaten the recovery of Lake Hayes and recent studies highlight management and restoration actions that will be needed to ensure that the lake does not shift back to a degraded state.

Introduction

Lake Hayes is a 280 ha lake (31 m maximum depth), located in the Arrow Basin between Queenstown and Arrowtown. It is situated in one of the most scenically attractive landscapes in New Zealand and it is important for recreation and tourism (Cromarty and Scott 1995). A vegetated margin surrounds the lake, including wetlands that support rare and threatened fauna such as koaro, longfin eel, New Zealand shovellers, marsh grebes, Australian coots and great crested grebes (Cromarty and Scott 1995). Some of the lake margin is recreational and wildlife reserve, belonging to a larger 354 ha wildlife refuge area, which includes the lakebed. A shared-use track circumnavigates the lake and the lake and its surrounds are used for recreation and tourism (Figure C-1).

Figure C-1: Lake Hayes. (Photo: M. Schallenberg).

Lake Hayes was an important food gathering area for Māori, leading to its recognition as a treasured resource (Waahitaoka) (ORC 2009). The Otago Regional Council Lake Hayes Management Strategy states that "the conservation of the Lake Hayes resource is of regional and national importance both economically, recreationally and for its intrinsic and scenic values." (ORC 1995).

Since the late 1960's, Lake Hayes has had severe algal blooms as a result of elevated nutrient loading from the main tributary, Mill Creek, and from nitrate-enriched groundwater springs at the northern end of the lake (Bayer and Schallenberg 2009). From the 1960's to 2010, the lake was in a degraded state, with blooms of cyanobacteria (blue-green algae), green algae and dinoflagellates occurring intermittently throughout that > 40-year period. After 2010, blooms of the dinoflagellate alga *Ceratium hirundinella* occurred during most summers. However, *Ceratium hirundinella* blooms did not occur in two recent summers and the lake maintained a historically high water clarity (Schallenberg and Schallenberg 2017).

Importance of the Mill Creek catchment

Land cover in the Lake Hayes catchment (Figure C-2) was likely dominated by kahikatea forest prior to 1740. A large wetland occupied the mid-western part of Mill Creek catchment, several small swamps were located to the west and north of the lake, and extensive riverine marshes lined the banks of Mill Creek and smaller tributary streams (Robertson 1988).

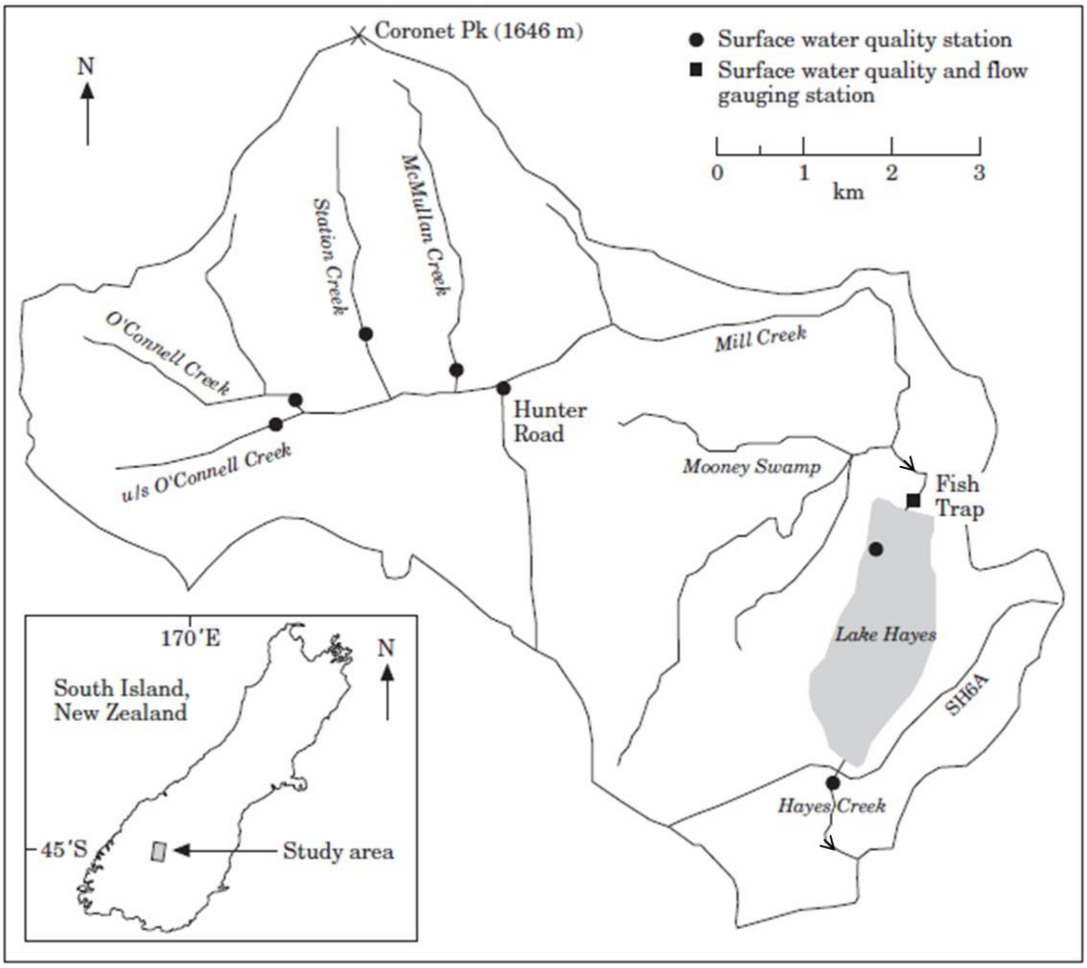

Figure C-2: The Lake Hayes catchment showing the tributary water sampling network used by Caruso (2001). The figure is adapted from Caruso (2001).

The kahikatea forest was largely destroyed by fire around 1740 and further deforestation followed to the late 1800's, as miners and settlers required wood for shelter and fuel (Robertson 1988). Following deforestation of the catchment, native tussock grassland occupied most of the catchment's high country and wetlands dominated the low-lying areas (Robertson 1988). The downstream transport of fine sediment and nutrients toward Lake Hayes would have been impeded by the wetlands despite the land-cover change. Thus, until the early 1990s, relatively low concentrations of contaminants from the catchment would have been transported to the lake by Mill Creek.

From the early-to-mid 1900's, much of the land in the Lake Hayes catchment was converted to sheep, beef and dairy farming. Superphosphate fertilizer was introduced in the 1950s and farming intensified as aerial topdressing became common (Robertson 1988). From 1912 to 1955, a cheese factory north of the lake released whey effluent into Mill Creek, estimated to be equivalent to a phosphorus (P) load of around 1000 kg/yr (Robertson 1988). Whey was also fed to pigs, likely contributing more contaminants to the Mill Creek catchment.

Major drainage and channeling works occurred in 1961-1962, when wetlands in the mid-catchment were drained and artificial channelisation was introduced into what was soon to be high producing exotic grasslands. The drainage works affected 80 to 120 ha of wetland, resulting in very high sediment loads in Mill Creek and into Lake Hayes (Robertson 1988). Brown water flowing through Mill Creek was commonly observed in 1961 and continued sporadically while drainage works in the wetland areas continued over the next few decades. Locals who remember this era consider the wetland drainage in the early 1960s to have been a major turning point for the health of Lake Hayes.

In addition to contributing land-derived nutrients to the lake, the conversion of wetlands into pastoral grasslands reduced the attenuation of contaminants from the middle and upper part of the catchment. Robertson (1988) estimated that 80% of the P load in Mill Creek had been attributable to tributaries above the large wetland area and suggested that the wetland would have trapped substantial amounts of P and as well as denitrified substantial amounts of nitrogen (N) flowing downstream via Mill Creek. Thus, prior to the loss of the wetlands in the mid-catchment, nutrient loads to the lake via Mill Creek would have been substantially lower than post-wetland removal.

A number of studies have recommended wetland restoration/re-establishment and on-farm BMP's for the Lake Hayes catchment (Robertson 1988; ORC 1995; Bayer and Schallenberg 2009; Ozanne 2014). Wetland re-establishment was discussed by Robertson (1988) as a means to recover lost ecosystem services such as nutrient retention, flood retention and denitrification.

Robertson (1988) suggested potential mitigation measures and best management practices that could reduce inputs of P into the lake, most of which involved improving catchment land use practices such as reducing fertiliser use and runoff, controlling channelling and drainage operations, establishing riparian buffer zones, and carefully managing future land development minimise P losses to the lake. The improvement of land use practices in the catchment was highlighted by Robertson (1988) and ORC (1995) to be crucial for the restoration of Lake Hayes.

In 1995, the Otago Regional Council developed the Lake Hayes Management Strategy with the stated goal of improving 'the water quality of Lake Hayes, to achieve a standard suitable for contact recreation year-round and to prevent further algal blooms' (ORC 1995). The strategy identified issues in the catchment affecting water quality and outlined ambitious actions to be taken to reduce the P load of the catchment, including negotiating with landowners around Mill Creek to establish riparian zones, encouraging the protection and re-establishment of wetlands, and encouraging more sustainable land use in the catchment. In response to this report, septic tanks in proximity to Lake Hayes were decommissioned.

ORC (1995) concluded that the Mill Creek catchment is the main source of nutrients to Lake Hayes. Around 80% of the catchment P load to the lake is attached to sediment particles. High historical catchment P loads settled to the lake bottom to become an internal P load, which greatly contributed to the decline and persistence of poor lake health seen over recent decades (Schallenberg and Schallenberg 2017, Gibbs 2018).

Caruso (2000) examined spatial and temporal variation in P concentrations and fluxes through the Mill Creek catchment. His analysis revealed a strong correlation between flow events and total P concentrations, with the chemograph of total P following the hydrograph by around 4 to 6 hours during a flood in August 1997. He also found that total P concentrations didn't show clear seasonal patterns at a number of sites in Mill Creek during his study in 1997. At a site just upstream from Lake Hayes, baseflow total P loading accounted for 47% of the annual total P load, whereas loading during the period of snowmelt accounted for 30% of the annual load. In comparison, just one flood event, occurring over a 26 h period, contributed 23% of the total annual total P load to the lake from Mill Creek.

An assessment of the spatial distribution of total P concentrations and fluxes through the Mill Creek catchment indicated that a substantial portion of the total P in Mill Creek originated from a relatively small sub-catchment in the upper catchment called O'Connell Creek and from the lower catchment (Figure C-2) (Caruso 2001). This analysis of critical source areas contributed to a risk-based P targeting analysis for P load reduction at the catchment, sub-catchment and farm scales. Caruso (2001) concluded that:

> *"The worst individual properties were targeted based on the combination of intensity of cattle and sheep grazing, fertilizer usage, bank erosion and location in the worst subareas.... In addition to concentrations, average and extreme loadings are important. Data on catchment characteristics, particularly land use, are needed for targeting, but are not always readily available at small scales."*

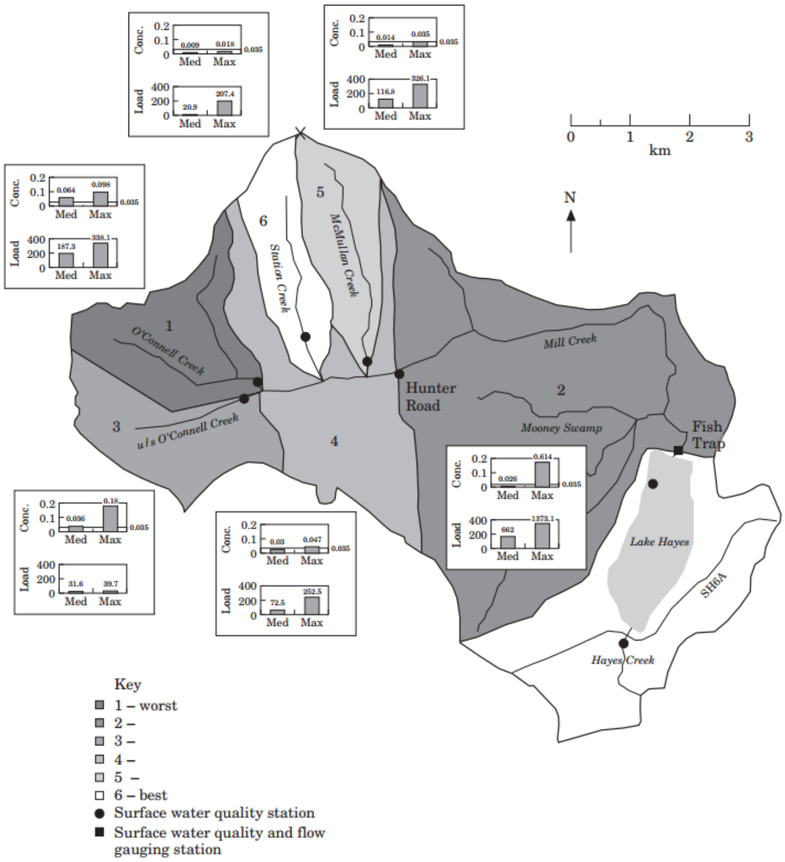

Figure C-3: Mill Creek catchment phosphorus targeting from Caruso (2001), showing catchments ranked from highest to lowest P contribution. Reproduced from Caruso (2001).

Intensification of agriculture in the catchment has been fostered by the use of irrigation water from the Arrow River irrigation scheme which was reported by Robertson (1988) to use 1.75 m^3/s of water from the Arrow River to irrigate 1100 ha of farmland in the Lake Hayes catchment.

Effects on the lake

The recreational brown trout fishery of Lake Hayes has been a regionally important fishery (H. Trotter, Otago Fish and Game, pers. comm.). But its popularity with anglers decreased rapidly in the mid 2000's (Figure C-3), due to negative effects of blooms of the dinoflagellate, *Ceratium hirundinella*, which began in the mid-2000s (Bayer et al. 2009; Schallenberg and Schallenberg 2017).

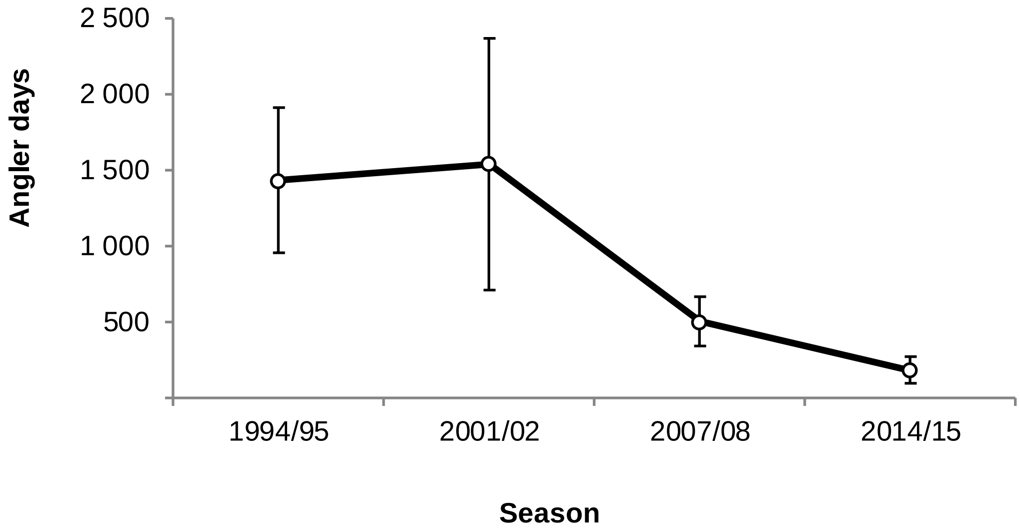

Figure C-4: Annual angler days for Lake Hayes estimated from the National Angling Survey (Otago Fish and Game, unpublished data). Reproduced from Schallenberg and Schallenberg (2017).

Since the 1950s, when the lake's bottom waters were oxygenated in summer (Jolly 1959), water clarity has been variable, but has often been quite poor (e.g., eutrophic or water clarity below 3.6 m) due to algal blooms. From 1970 onward, nitrogen-fixing cyanobacteria (e.g., *Dolichospermum* sp.) have often been part of the phytoplankton community, sometimes occurring as the dominant species (Burns and Mitchell 1974; ORC 1995). Nitrogen-fixing cyanobacteria may outcompete other phytoplankton when excess phosphorus is available because the cyanobacteria are able to fix nitrogen from the air.

Since at least 1970, the bottom waters of Lake Hayes have become anaerobic (with a complete loss of dissolved oxygen) (Burns and Mitchell 1974) and summer deoxygenation of the bottom waters has been recorded whenever it has been measured, since that time (Robertson 1988; ORC 1995; Bayer et al. 2008; Bayer and Schallenberg 2009; M. Schallenberg, unpublished data; ORC, unpublished data). The loss of dissolved oxygen from the bottom waters not only excludes fish, zooplankton and many invertebrates from the cooler bottom waters of the lake, but it also causes biogeochemical changes in the lake sediments, releasing sediment-bound phosphorus into the water column (Bayer et al. 2008). Since at least 1970, the summer bottom waters have been releasing significant amounts of phosphorus into the lake water (Mitchell and Burns 1981; Robertson 1988; Bayer et al. 2008; Bayer and Schallenberg 2009; Schallenberg and Schallenberg 2017; Gibbs 2018), recycling historically accumulated and immobilised P back into the lake ecosystem and further fuelling algal and cyanobacterial blooms. Severe blooms eventually settle to the lake bed delivering more P to the sediments as dead phytoplankton cells, where they decompose and consume oxygen, contributing to the next year's summer deoxygenation. This internal anoxia-phosphorus-algae feedback cycle has contributed to maintaining Lake Hayes in a eutrophic state for much of the time since at least 1970 (Figure C-4), despite the fact that external nitrogen and phosphorus loading from Mill Creek and the springs at the northern end of the lake had decreased into the 1990s and early 2000s (Caruso 2000; Bayer et al. 2008; Bayer and Schallenberg 2009).

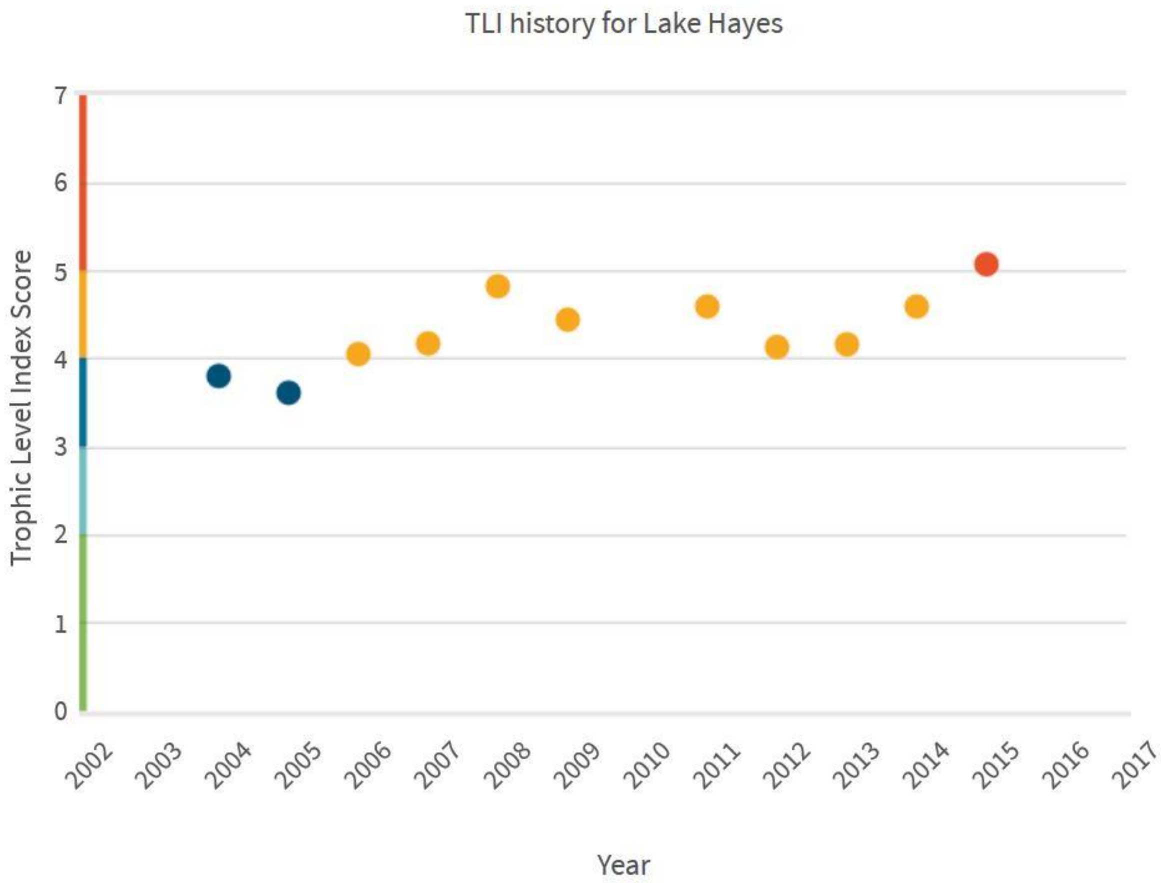

Figure C-5: The trophic level index (TLI) for Lake Hayes from 2004 to 2015. TLI is an index that tranforms and averages the lake water quality indicators: total P, total N, chlorophyll a (an indicator of phytoplankton biomass) and Secchi disk depth (a measure of water clarity). A TLI between 3 and 4 is mesotrophic (good water quality), between 4 and 5 is eutrophic (poor water quality), and between 5 and 6 is supertrophic (very poor water quality). Data and graph are from the LAWA website and are reproduced from Schallenberg and Schallenberg (2017).

The average water clarity of the lake had been moderately poor but rather stable from 1970 to 2006 (Figure C-5), until *C. hirundinella* began to bloom in the lake (Bayer et al. 2008). This dinoflagellate is a large, spikey, motile (swimming) alga that is mixotrophic, meaning that it can gain energy both via photosynthesis (as plants do) and also by feeding on bacteria (as some protozoans and zooplankers do). Mixotrophic dinoflagellates including *C. hirundinella* have been reported to become abundant and dominant in the phytoplankton communities of lakes during periods of lake recovery from eutrophication (Jeppesen et al. 2003; Gerdeaux and Perga 2006, Mehner et al. 2008), partly due to their ability to supplement their nutrient requirements by feeding on bacteria (Gerdeaux and Perga 2006).

Experiments on Lake Hayes phytoplankton communities in 2006 (Bayer et al. 2008) indicated that the growth of the phytoplankton community, dominated by *Ceratium* at the time, was stimulated by additions of N and the trace elements boron and zinc. Phosphorus additions did not stimulate phytoplankton production. This result supported an analysis of N:P ratios in the lake, which also suggested that P was often in surplus in the lake water relative to N (in relation to the nutrient demands of phytoplankton) (Bayer et al. 2008).

While the lake has suffered severe *C. hirundinella* blooms in most summers since 2006, in the summers of 2009/10 and 2016/17, the lake experienced unprecedented water clarity and very low *C. herundinella* biomass. Schallenberg and Schallenberg (2017) suggested that food web interactions related to summer persistence of the grazing zooplankter, *Daphnia pulex*, in the lake may have contributed to the sudden shifts of Lake Hayes from a eutrophic condition with severe summer *C. hirundinella* blooms to occasional summers with very low algal biomass.

Lake Hayes may be entering a phase of recovery whereby nutrient availability is approaching a recovery tipping point and food web interactions in some years may tip the lake into a temporary recovery from eutrophication (Schallenberg and Schallenberg 2017).

Restoration options

Restoration options for Lake Hayes have been proposed since at least the 1970s. While Mitchell and Burns (1972) discussed options including diversion of inflows away from the lake, flushing the lake with irrigation water and oxygenation of the bottom waters, Robertson (1988), Schallenberg and Schallenberg (2017), and Gibbs (2018) emphasised the importance of continued reduction of catchment contaminant loads.

Although nutrients in Mill Creek and the springs seem to have stabilised (Bayer and Schallenberg 2009), Mill Creek remains higher than average for upland streams in *E. coli* counts, turbidity and total nitrogen, total oxidised nitrogen, ammonium and total P concentrations, while the concentrations of *E. coli* have been increasing in the past decade (https://www.lawa.org.nz/explore-data/otago-region/river-quality/clutha-river/mill-creek-(fish-trap)/). In addition, substantial areas of land in the catchment have been designated for further development in the proposed Queenstown Lakes District Plan. Together, this highlights the importance of implementing best land use management practices (BMPs) in the catchment. Mitigations should focus on *E. coli*, fine sediment, phosphorus and nitrogen and should also account for fluxes of these during flood flows. BMPs may include carefully managing farming practices to minimise soil erosion and nutrient leaching, managing new land developments to minimise losses of sediment and nutrient to waterways, and excluding stock from waterways. The Otago Regional Council is planning to conduct a sub-catchment survey of tributary contaminant concentrations and fluxes, based on Caruso (2000). Furthermore, Caruso's (2001) strategy for targeting BMPs in the sub-catchments, and on the farms, that are contributing high contaminant loads could be a useful approach for further minimising contaminant loads from the catchment to the lake. BMPs could also be developed for lifestyle blocks, golf courses and any common activities in the catchment that disturb ground or add nutrients to the soil or waters:

- To facilitate these actions, Schallenberg and Schallenberg (2017) suggested that the following steps could be undertaken.

- Identifying iwi, stakeholders, industries, scientists and other interested parties.
- Reviewing the ORC (1995) Lake Hayes Management Strategy and its implementation.
- Undertaking a catchment-wide N, P, sediment and *E. coli* survey based on the design of Caruso (2000) to identify current hotspots where contaminants enter Mill Creek and the lake.
- Determining the feasibility of setting nutrient caps on the catchment.
- Collaboratively develop a lake/catchment management plan with community participation at all stages.

Summary and conclusions

The information presented in this case study is summarised in a timeline describing the history of catchment events that had the potential to impact the condition of Lake Hayes (Figure C-5). The number and diversity of potentially significant historical events show the complexity of the Lake Hayes system and that a combination of historical factors (e.g., fish introductions, fertiliser overuse, dairying, wetland drainageand current factors (e.g., *C. hirundinella* blooms, *D. pulex* dynamics, decreasing nutrient loads) can, together, impact the current condition of a lake.

This case study also highlights how both in-lake physico-chemical conditions (e.g., dissolved oxygen concentrations) and species-specific ecological factors (*C. hirundinella*, *D. pulex*) can mediate the effects of catchment contaminant loads (i.e., pressures) on the state of a lake and impacts on it. The internal P load issue and the importance of groundwater to the N budget of Lake Hayes both indicate that time lags and legacy loads can confound what might theoretically seem to be direct links between catchment pressures and lake responses.

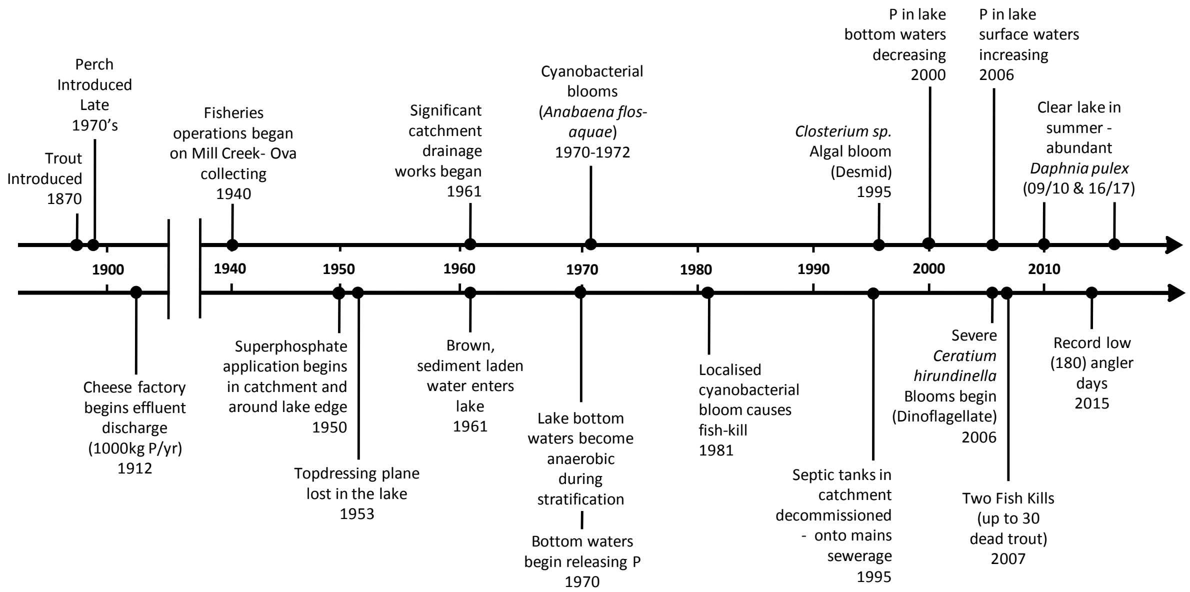

Figure C-6: Historical timeline of some key events impacting Lake Hayes.

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Appendix D Queen Charlotte Sounds

Ecosystem Service Review of Marlborough Sounds Pāua

Summary

Aotearoa Fisheries Ltd, known as Moana New Zealand is the largest Māori seafood company and the only one owned by all Iwi. Pāua, the Māori name for abalone, is a taonga and important to their business. Being Māori owned, Moana’s tikanga (ways of working) strongly underpins their operations including being kaitiaki, or guardians, of the domain of Tangaroa (God of the Sea) for this, and future generations. Held in perpetuity their fishery quota assets must be carefully stewarded so Moana has undertaken a sustainability journey including to align with community values.

As part of this, in 2015, with support from sustainability advisers Terra Moana Limited (TML), Aotearoa Fisheries conducted a Qualitative Ecosystem Service Review (ESR) of New Zealand pāua. The Pāua Industry Council (PIC), the National Institute of Water and Atmospheric Research (NIWA) and Otago University School of Marine Science were core technical partners. One of five such ESRs it was led by Landcare Research for the New Zealand Sustainable Business Council Business and Biodiversity project supported by the Department of Conservation.

Moana’s pāua harvest managers provided technical advice about the daily business realities and issues pāua divers face in harvesting a wild resource, including marine environmental observations. The ESR enabled Moana to evaluate its dependence and impact on ecosystem services and to identify the resulting business risks and opportunities. This ESR was a first for the New Zealand seafood industry and globally for a commercial seafood representing a step change in understanding commercial fisheries and their environmental and fishery management needs.

The AFL ESR illustrated three key types of change to be aware of in Moana’s pāua operations:

1. ***Environmental*** e.g., sediment, localised temperature or pH changes.
2. ***Fishing*** whether commercial, recreational, customary or illegal - for species that interact as an ecological community in kelp habitats (blue cod, kina, pāua, rock lobster).
3. ***Dynamic*** e.g., global scale forces such as climate change (multiple time and spatial scales).

Furthermore, an associated preliminary valuation of the quota value loss incurred over this period, at least partially, related to lost kelp habitat was estimated at ~$20 million. Furthermore, despite 4 years of voluntary cuts by industry, for the 2016/2017 fishing year, the Pāua 7 Fishery Total Allowable Commercial Catch was cut 50% to and remains 93.62 tonnes.

Whilst establishing causality is problematic, since the ESR, awareness and public discourse have grown about the sediment impacts on coastal seafood habitats and production – especially large pulsed forestry derived sediment that this ESR identified as a key risk to pāua fisheries (and coastal habitats and therefore other valuable seafood such as rock lobster) and which is thus this case study’s focus.

Moana will begin an ESR of their Whangaroa Harbour oyster farming operations in 2018.

Background

In 2014, supported by the Department of Conservation to introduce the concept of natural capital to New Zealand business, the Sustainable Business Council and Landcare Research facilitated five ecosystem service reviews (ESR).

As kaitiaki of significant Māori fisheries assets, Aotearoa Fisheries Limited (AFL) adopted a sustainability journey to safeguard the future of wild and farmed seafood. Being Iwi owned AFL hold an infinite approach to business recognising their natural resources are permanent assets, and therefore needing deeper understanding to future proof their businesses. AFL joined the New Zealand Sustainable Business Council (SBC), and with Terra Moana Ltd (TML), their sustainability advisers, piloted the corporate Ecosystem Service Review (ESR) in their pāua business. This was a world first[17] for a commercial fishery and a unique opportunity to explore the alignment between Te Ao Māori (the Māori worldview) and the ecosystem service framework.

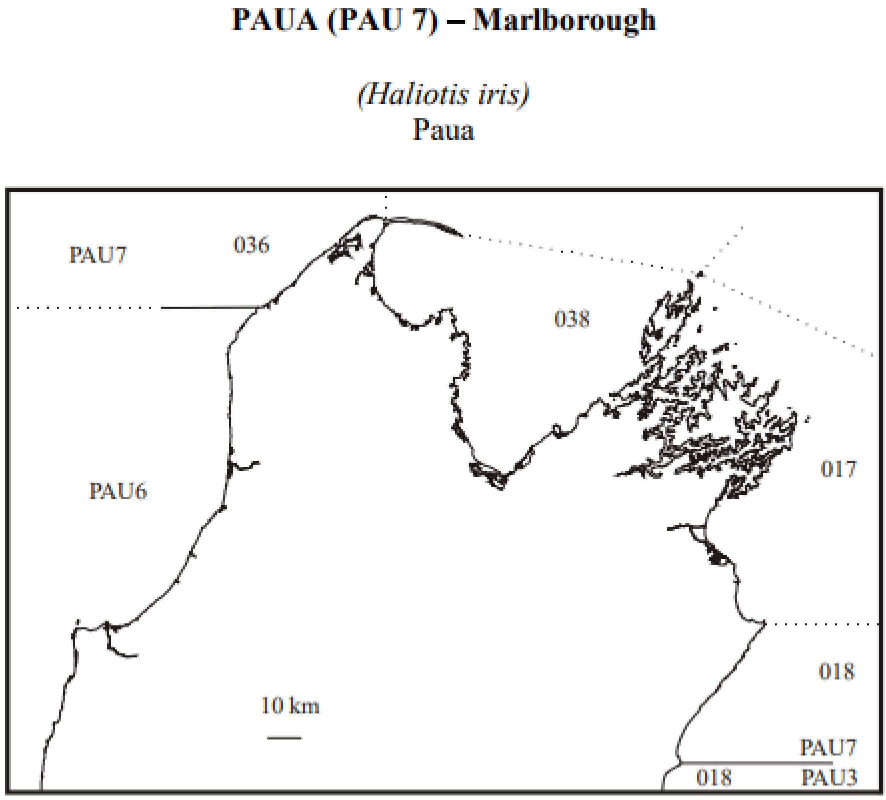

Figure D-1: Paua (Pau 7) - Marlborough.

AFL understands that natural capital and ecosystem services provide the goods and services i.e. benefits that humans derive from nature, biodiversity and the environment, directly and indirectly. Through Te Ao Māori, AFL deeply understand this interconnectedness, that humans are part of healthy ecosystems and the necessity of living that worldview through practicing kaitiakitanga.

The ESR scope was defined as the Pāua Fishery in Queen Charlotte Sound of the Marlborough Sounds because i) pāua is a significant quota asset to AFL, ii) the fishery was in trouble, iii) there were stakeholders willing to participate and, iv) pāua is a taonga to Māori. This pāua fishery is a sub-set of

[17] Short, K. *Scaling Up Seafood Sustainability, An Illustrated Journey*. Solutions Journal. March-April 2015

the Pāua 7 Management Action Committee area. The review explored if and how the coastal ecosystems where pāua occur are changing and the potential impacts this might have on the provisioning of ecosystem services for the fishery and related fisheries such as blue cod and rock lobster. The review identified issues to consider more deeply and potentially address, including the declining catches of pāua and kelp habitat loss in the Marlborough Sounds.

Aotearoa Fisheries Ltd and Te Ohu Kai Moana

Created in 2004, Aotearoa Fisheries Ltd was formed to manage the fisheries assets returned to Iwi under the Treaty of Waitangi Fisheries Claims Settlement, known as the Sealord Deal. Iwi have asset holding companies and some also own seafood businesses including Ngati Porou and Ngai Tahu. Overall, AFL owns the majority of wild pāua quota, the majority of pacific oyster aquaculture, the only farmed pāua operation nationally and is a major quota owner in significant North Island inshore fisheries such as tarakihi and snapper. All seafood is supplied to AFL by independent contract aquaculture operators, fishers and divers. AFL is an integrated seafood company with state of the art infrastructure. It has significant partnerships with Iwi such as: i) in the Chatham Islands, ii) the Iwi Collective Partnership and iii) the second largest rock lobster company, Port Nicholson Fisheries.

As part of a strategic brand positioning for Māori seafood, Aotearoa Fisheries Ltd changed its public name to Moana New Zealand in 2016, legally remaining Aotearoa Fisheries Ltd. Moana New Zealand is the largest processor of pāua in New Zealand, both wild and farmed selling a prized seafood to customers in New Zealand and especially in Asia. Traditionally, Moana New Zealand used the prime pāua meat for canning, sent the trimmings to the nutraceutical sector, and the shells to the ornament trade. This picture is changing, however, with increasing interest in live export. Key pāua markets are China, Malaysia, Singapore and Hong Kong.

Te Ohu Kai Moana (TOKM) was created to support Iwi/Māori through the fisheries settlement and to manage the process of allocating the returned fisheries assets. It maintained a 20% share of AFL quota ownership until, in a review in 2014, Iwi/Māori decided that would be fully distributed to AFL. TOKM's updated strategy reflects this and which now, related to this case study, includes advocating for better understanding of Habitats of Particular Significance for Fisheries Management (Fisheries Act Sn 9(c)).

ESR Benefits to AFL

The Ecosystem Service Review was beneficial to Moana New Zealand in:

- Providing their own picture of what's happening to marine resources.
- Improving their understanding of the terrestrial impacts affecting the economic and ecological viability of the fisheries which Moana New Zealand's business depends upon, and which risks the sustainability of Māori fisheries assets overall.
- Giving new access to government.
- Improving access to science, scientists, communities and expertise to develop solutions.
- Creating new dialogue within AFL across the Divisions.

Moana New Zealand is fully committed to ensuring it provides customers and shareholders with the confidence that their seafood products come from responsible, environmentally safe and sustainably managed seafood production systems. Moana New Zealand was alarmed by the learnings of the ESR especially how it highlighted the potential impact of land-based activity (sedimentation, run-off) on the kelp flora. Like a slowly boiled frog, the impacts are too often overlooked in the short term and over the longer term the consequences of leaving the underlying causes unchecked appear to be significant. This applies to both the commercial sector and to recreational fishers, tourism operators and the broader Marlborough community and economy – and potentially elsewhere in New Zealand. The consequences flow on to diminish all natural capital, the essence of taonga, and quality of life on earth.

Update / the Current Situation

Since the ESR was released by the Sustainable Business Council in 2015, key developments include:

- The Ministry for the Environment and Sustainable Business Council have continued to apply natural capital concepts, including through the Natural Resources Sector (NRS).
- A preliminary valuation of the quota value loss incurred from the decline in kelp habitat was estimated at $20 million (elaborated below).[18]
- Despite 4 years of voluntary cuts by industry, for the 2016/2017 fishing year, the Pāua 7 fishery[19] Total Allowable Commercial Catch was cut 50% to and remains 93.62 tonnes.
- Awareness has grown in the forestry sector of the impacts of poor harvesting operations on coastal marine ecosystems, habitats and thus risks to seafood production.
- Moana New Zealand and SCION, the forestry research institute have lodged a MBIE bid to develop a quantitative ecosystem services modelling approach to inform decision making across connected biomes (land and sea) and to improve forestry practices.
- The National Environmental Standard for Plantation Forestry (NES PF) has been released with strengthened provisions for protecting waterways from sediment during harvest operations.
- Whilst causality is always difficult to establish, the issue of sedimentation impacting coastal fisheries, including in the fishing industry, has continued to feature in the media[20] and key reports[21].
- Marlborough District Council and SCION analysed forestry setback options for the Marlborough Sounds (Nov 2017).
- Furthermore, at the recent Māori Fisheries Conference (March 2018) Te Ohu Kai Moana explored how Section 9(c) of the Fisheries Act and could be better employed to protect coastal seafood from terrestrial inputs in its workshop "Taking Responsibility

[18] Natural Resources Sector Business Leaders Forum. "*Capturing the Value of Nature in Decision Making"* Case Study. Aug 2015.
[19] See map pg 2.
[20] https://www.stuff.co.nz/southland-times/opinion/103341285/letters-lets-revisit-a-bad-decision-and-make-it-good
https://www.stuff.co.nz/national/99132882/not-everyone-is-so-sure-its-a-good-idea-to-introduce-pua-to-starfish
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http://www.eds.org.nz/our-work/events/past-eds-events/voices-from-the-sea-book-launch/

for Managing the Effects of Fishing on the environment" (27.3.18). See below: Key Next Steps.

- This MfE cross biome report reflects these complex issues at the margins/across boundaries.

ESR Method

The ESR was conducted using internationally accepted best practice via the World Resources Institute Corporate Ecosystem Service Review methodology. Landcare Research technically supported the application of the ESR in New Zealand. The steps followed were:

1. Determine the scope of the assessment. The ESR was focused on the operational issues facing the production of wild pāua in the marine environment and its processing facility, Prepared Foods had recently moved into a new high environmental specification facility with efficiency programmes.

2. Identify the priority ecosystem services. The ESR enabled the company to prioritise key ecosystem services by evaluating the degree of dependence and impact on a range of ecosystem services.

3. Communicate. A team was identified to document the status of each priority ecosystem service as follows:

- Erosion and Wildfoods: Pāua Industry Council.
- Regional Climate Change: Otago University.
- The three social/cultural priority ecosystem services: Terra Moana and Moana New Zealand.

A summary business case study report[22] was compiled for a broad audience and the findings were presented in a range of fora including the New Orleans SeaWeb Seafood Summit[23], the SCION Forest Ecosystem Services Forum[24], the Solutions Journal and online[25] internationally, and alongside a preliminary valuation (presented below) for the Natural Resource Sector Business Leaders Forum.

Findings

Priority Ecosystem Services

Provisioning: (1) Wildfoods were selected as Aotearoa Fisheries pāua operations depend upon the provision of pāua as a wildfood.

Regulating: (2) Regional climate change and (3) erosion were selected as Aotearoa Fisheries pāua operations are dependent upon global and regional climate, freshwater timing and flows, erosion control, water purification and waste treatment, and natural hazard mitigation.

Cultural: Aotearoa Fisheries pāua operations are affected by (4) recreation and ecotourism, (5) education and inspiration and (6) ethical and spiritual values.

[22] http://moana.co.nz/wp-content/uploads/2017/06/AFL-1501-ESR-Paua-Book-V7-final.pdf
[23] http://media.wix.com/ugd/387f67_3458faabb992471b8d8a9949d1bbe348.pdf
[24] http://media.wix.com/ugd/387f67_f6354feb3b0046129c6391580ec64d53.pdf
[25] https://impactalpha.com/tending-the-ocean-not-just-the-seafood-valuing-ecosystem-services-in-new-zealand/

Types of Change

Furthermore, the AFL ESR illustrates the interactions of **three types of change**:

1. Environmental e.g., sediment, localised temperature or pH changes.

2. Fishing whether commercial, recreational, customary or illegal - for species that interact as an ecological community in kelp habitats (blue cod, kina, pāua, rock lobster).

3. Dynamic e.g., global scale forces such as climate change (multiple time and spatial scales).

Erosion - Summary

Given the sedimentation topic of this review, below is an excerpt from the Erosion Priority Ecosystem Service prepared by Tom McCowan, Scientist, Pāua Industry Council.

This Priority Ecosystem Service review examines the trends and drivers of sedimentation into the marine environment and its potential effects on Pāua fisheries. Sedimentation because of human land-based activities (erosion) is recognized as a threat of global concern to marine environments. Sedimentation in coastal environments has been identified as a major issue in New Zealand, where large-scale deforestation for agriculture, forestry and urbanization has occurred over erosion prone landscapes. The habitats most severely affected by sedimentation are those which are critical for pāua and key components of pāua ecosystems (e.g., food sources such as *Macrocystis pyrifera*). There have been concerns over the potential effects of sedimentation on pāua fisheries for several years, particularly in areas of high forestry and agricultural activities (e.g., PAU7), where changes in levels of sedimentation and observed effects are reported.

Evidence for direct effects of sedimentation on pāua was reviewed as well as effects on M. pyrifera (an important part of pāua ecosystems in some locations). Research suggests that sedimentation events can restrict settlement of pāua larvae, cause larval mortality, and change behavior in adult pāua making them more susceptible to predation and dislodgment. Sedimentation can also restrict the recruitment of kelp spores, cause mortality of settled juveniles by smothering, and restrict growth in juveniles by increasing water turbidity. Settlement of fine sediments on the laminae of mature plants can also restrict their photosynthetic ability and nutrient exchange. These findings all suggest that sedimentation ultimately adversely affects pāua fisheries by limiting recruitment success (into the fishery) and reduces overall productivity in areas where M. pyrifera is a critical food source. These findings align with the SCION (Nov 2017) forestry setbacks report commissioned by Marlborough District Council.

While there is no published scientific evidence for direct links between increasing sedimentation, a reduction in kelp abundance and reduced pāua fishery productivity, there are anecdotal reports which strongly suggest that this pattern exists in some areas. For example, in the Marlborough Sounds (PAU7), it is believed that increased sedimentation from forestry has resulted in a dramatic decline in M. pyrifera abundance in some areas, which is also linked with a decline in the productivity of the pāua fishery in these areas. This pattern begins to become more apparent when comparisons are made with reports from other Quota Management Areas (QMAs), where declines in fishery productivity have also been linked with specific areas where sedimentation has also been identified as a threat (or conversely, reports of no noticeable declines where sedimentation is not an issue, e.g., PAU5B, Stewart Island). This is an important area for further research including investigating different datasets for more robust correlations.

An assumption in fisheries stock assessment models is that fishing pressure is the only stressor on fisheries – which is clearly not the case. This review suggests that environmental impacts (in this case sedimentation) may also act as a significant stressor on pāua fisheries in some areas. This highlights the need to consider the impacts that environmental changes have on fisheries when making management decisions and implementing management tools.

Consequently

As mentioned, an assumption in fisheries stock assessment models is that fishing pressure is the only stressor in the fishery, and that environmental stressors remain constant, despite dramatic ongoing changes in coastal ecosystems over the last 100 years. Management initiatives are currently based around altering fishing pressure (changing size limits or the TACC); however, the ongoing cumulative effects of environmental stressors on the ecosystem are poorly understood (Morrison et al. 2009; MacDiarmid et al. 2012). This review highlighted the potential of sedimentation to significantly adversely affect pāua fisheries, and the need to give greater management consideration of these effects. For example, in areas where reseeding is routinely undertaken, care should be taken to reseed in areas where there is intact native vegetation, or where forestry is present and logging will not occur in the near future, or to avoid reseeding during seasons where high rainfall is expected. There could also be a move towards using new indicators to monitor fishery health (in conjunction with standard fisheries stock assessment indicators). For example, in particularly susceptible areas, metrics such as measures of land use change, forestry harvest timeframes, frequency of storm events or levels of turbidity may be able to be used to predict the status of the fishery. This is an area for further research.

The potential magnitude of environmental effects on susceptible pāua fisheries can be analysed by comparing the relative status of fisheries such as PAU5B and PAU7. These have undergone similar histories of over-exploitation followed by exploitation-based management controls, yet PAU7 has not had the estimated biomass recovery. While there is no hard evidence to suggest this is due to changing environmental effects, the literature and anecdotal observations presented in this review suggest they may be significant. There is evidence to suggest that certain drivers of sedimentation (storm events, sea-surface temperatures, land use change) will only increase in the future, meaning this is something that must be given serious consideration in monitoring fishery health in the future and in future management decisions.

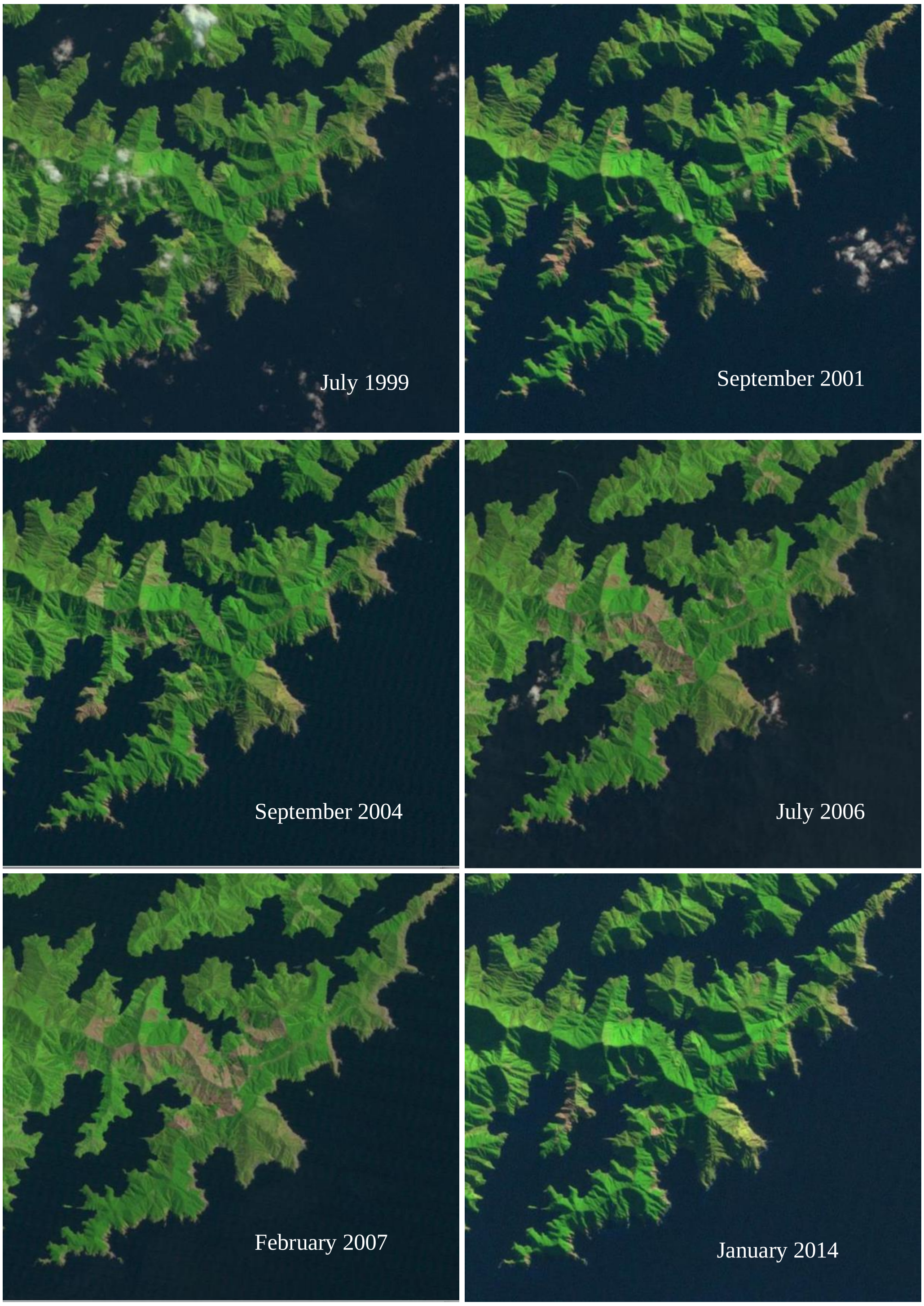

Figure D-2: Selected aerial photographs of forestry activity in the southern Marlborough Sounds showing changes in forestry activity over the last 15 years. (images from the United States Geological Society).

Figure D-3: An estimated 2 tonnes (pers. comm. A. Col, 2014) of Pāua washed ashore after extreme sedimentation and storm events at Kaka Point, Catlins, in late 2013. (photos from Ayson Col).

Figure D-4: Approximate depletion of Macrocystis pyrifera beds in Queen Charlotte Sound based on historical accounts (shown as % loss since 1965). (pers. comm. D. Baker, 2014).

Figure D-5: Massive Macrocystis beds washed ashore in approximately 1980 after a southerly storm at Anakakata Bay, outer QSC, a phenomenon that has not occurred for over two decades. (photo from Dave Baker).

Valuation

Summary

A conservative preliminary economic valuation estimated the potential ecosystem service loss from decline in pāua kelp habitat since 2001 as:

- NZ -$5.94m combined revenue loss (2001-2014) to quota owners and harvesters, and therefore local economies, and some of this is this in then reflected in.
- NZ$ -19.13 million of capital value balance sheet losses to quota owners.
- Total estimated value change ~ -$20m.

Detail

Following the ESR, MBIE, TML and AFL collaborated to explore what valuation approaches could be used to better understand the impact of the apparent ecosystem change, i.e. kelp loss. This took account of the May 2015 NZIER produced a report: "Capturing Natural Capital in Decision Making". The Key Points highlighted below notes some of the challenges faced in economic analysis of such values and which many acknowledge will be further exacerbated in the marine environment.

Failing to value natural capital leads to sub-optimal decision-making
The basic premise of natural capital approaches is to recognise some value for natural resources that are commonly regarded as value-less. Without a value, natural resources are prone to under-weighting in decision processes and excessive use causing externality effects on current and future generations…

Numerous options for assessing natural capital exist

There is a range of methods for natural capital assessments, including:

- ***Wealth accounting*** *is the complement to income accounting of the System of National Accounts – total wealth comprising physical, financial, natural, human, institutional and social capitals*
- ***Extended income accounting****, which removes natural resource depreciation from gross income measures to obtain net economic welfare or green GDP*
- ***Ecosystem services*** *approaches measure service flows from specific natural resource/asset classes that make them valuable to human well-being*

These methods have high data requirements and are time consuming. Due to data and time shortages, alternative approaches were considered that could use fishing quota and Annual Catch Entitlement (ACE) values as substitutes to assess potential ecosystem services contribution or loss.

The value of fisheries is traditionally measured through quota prices which arise from:

- The NPV (net present value) of the future stream of earnings related to the right to a kilogram (kg) of ACE (Annual Catch Entitlement) and which is the annual right to catch (usually someone else's) quota.
- That "right to a kg of ACE" is directly related to the health of the environment that supports stock productivity and the supporting management framework effectiveness.
- The price of a kg of ACE (as represented by annual cash flow) that is paid to the quota owner is therefore directly related to the health of the fishery.

Two approaches were used to consider economic value loss:

1. Using a simplified price/cost table.
2. Mapping of TACCs versus catch and discount / risk relationships.

Table D-1: Simplified Price/Cost Table (NZD and all numbers are for example purposes only).

Current international (CIF) price for pāua (Pp)	78
	40
Payment to Quota Holder (Q)	
Of Which Payment to Diver (D)	20
Payment to Factory operations (Processing /Packaging) (F)	38

Here Economic Rent (or supra-normal profit, above the return that is required to keep capital and labour employed in the fishery) is:

Received by Quota holder (Q (1 - Cc) - D); where Cc is the percentage cost of capital

Received by Diver rd x D; 0<rd≤1

Received by Factory rf x F; 0<rf≤1

The fractions rd and rf are the shares of income which are economic rent. Total rent is therefore:

Q (1 - Cc) - (1 - rd)D + rf F

Opportunity costs (unavoidable resource costs of production per kg) are:

Pp - (Q (1 - Cc) - (1 - rd)D + rf F)

Pp is unaffected by the output of the fishery. The level and distribution of rent - the values of Q, D, F, rd and rf - will depend on this price, and the underlying resource costs of harvesting and processing.

For a given installed processing capacity, processing cost per unit will depend strongly on throughput. Harvesting costs will depend on the location and density of the pāua stock, this being a function of the state of the supporting ecosystem as well as the intensity of past harvesting. Changes in the level of recreational take may be an important factor here.

The value of Quota per kg will be the NPV (net present value) of the future stream of earnings related to the right to a kg of ACE (Annual Catch Entitlement). The market price will be higher than the NPV of a quota holder's annual stream of rents as defined above, as it will include the NPV of financing costs at Cc. Unfortunately, given the commercial sensitivity of key market price data it is not incorporated here.

Mapping of TACCs versus catch and discount / risk relationships.

Figure D-6 below identifies the TACC vs Catch Trends and highlights the industry voluntary shelving to date attempting to stem the stock downward trend. For many it would be easy to assume fishing pressure was the most obvious of a range of potential causes and history would suggest that the original TACCs, like many at the introduction of the Quota Management System, were set too high. However, if one considers that fishing pressure was not the sole cause of the decline then Figure D-6 graphically illustrates the potential ecosystem value loss, at least to the commercial sector. Sadly, catch is all that management and industry track.

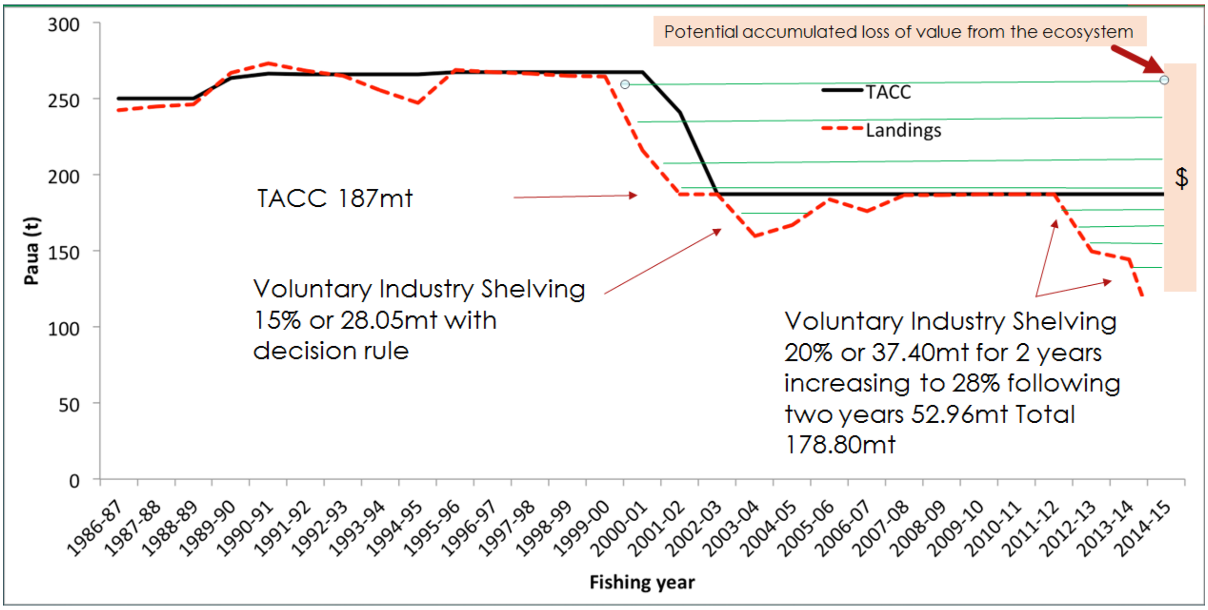

Figure D-6: Historical pāua 7 TACC and Catch Trends.

Sitting behind Figure D-6 is a simple economic story which requires separating quota ownership from quota harvesting. For the quota owner the story identifies the relationship between risk and associated discount rates to annual cash flows (ACE) and therefore NPV. Quota values are therefore a factor of (AC x P/AH-C) = (cash flow) whereby:

- AC = ACE kg's caught.
- P = price per kg.
- AH = ACE held (revenue per kg owned).
- C = cost of government levies.

The quota value of each kg or price that one is prepared to pay is then a factor of risk and the application of associated discount rates. High risk rates i.e. 25% deliver a NPV multiplying factor of 4 whilst conversely low risk at for e.g., 6% delivers an ACE multiplying factor of 16.66 (1/6).

Establishment of the risk or discount rate is critical and pāua fisheries are renowned for their difficulty to manage. Without comprehensive ecosystem monitoring (and associated adaptive management) i.e. more than catch vs TAC or catch per unit rates (CPUE) currently used, we are mostly fishing in the dark. Their sedentary nature means they are more susceptible to serial depletion and environmental changes than many other species.

In a perfect world the cuts noted in Figure D-6 could be regarded as an "investment" and or astute management action by the quota owner to reduce the risk of uncertainty and therefore increase the multiplying factor of cash flows to maintain overall balance sheet and quota value. In addition, the quota owner would expect to achieve a return on the investment (ROI) as the benefits of the "investment" transpired into quota increases over time.

Unfortunately, that is where the perfect world in this scenario ends. Under the New Zealand Quota Management System the quota owner has no ability to constrain additional harvest pressure nor extract payments from externalities i.e. increased recreational harvest, land-based activity or global impacts (warming). Hence the reason why industry generally oppose TACC decreases. Furthermore, the costs of implementing marine monitoring regimes that provide better understanding of subtle, yet cumulative impacts are usually considered prohibitively expensive rather than being sensibly designed and collaboratively funded to better inform both management and the industry (the investors) to make the most appropriate management decisions.

In fact, one would say because of the absence of such data risk the discount rates should still be high because of the lack of certainty going forward. This is not a recipe for encouraging investment or improving community wellbeing.

The real loser in this model is the harvester (Diver) who experiences lost revenue under the following scenario (A x C x P) = (Annual Revenue) whereby:

- A = ACE Leased.
- C = Catchability.
- P = Price = annual revenue.

Each year where ACE leased remains uncaught reduces diver profitability that cannot be recaptured.

Therefore even under a conservative estimate model and little or no chance of quota owners getting a return on their "investment", the potential ecosystem service loss since 2001 could be reflected as:

- NZ -$5.94m combined revenue loss (2001-2014) to quota owners and harvesters, and therefore local economies, and some of this is then reflected in.
- NZ$ -19.13 million of capital value balance sheet losses to quota owners[26].
- Total estimated value change ~ -$20m.

Key Issues and Opportunities

The ESR identified key actions with the priorities being:

1. To understand the ***multiple, cumulative stressors*** affecting pāua and kelp (Macrocystis pyrifera) in the Marlborough Sounds (sedimentation, fishing and regional climate change (pH, temperature and acidification). Kelp is also habitat to kina, rock lobster, blue cod and other valued species that interact with pāua both ecologically and from fishing.

[26] **Quota loss** is attributed to the volume of quota MT'S in the TACC reduction 2001 (53.49) multiplied by the average 2014 equivalent value price of quota for that period $357,718.40 per mt = $19,13m
Annual revenue loss = Total of 2014 equivalent value ACE prices applied per annum X loss of catch opportunity each year (Quota Cut plus shelving where applicable) = $5,94 over the period. In simplified terms: If you owned a farm that was 100 Acres and every acre was worth $359,000 and the government took 10 acres off you this would equate to a balance sheet loss of $3.59m unless you a) received compensation or B) the remaining 90 Acres was then worth the same as the 100. The value of the 100 is of course relative to the productivity of the total so taking the 10 away is unlikely to be supplemented by the 90. This is particularly so with fishing quota where diversification is not possible. Similarly your total annual income would have been related to that 10 acres that was taken off you. In this case like fishing quota the earning value is always the same so what you lose is 10 acres worth of income each year the 10 acres is taken off you and or there is no chance of getting it back …the loss equivalent to ACE Annual Catch Entitlement lease revenue the quota owner would have normally received.

2. ***Manage all forms of pāua removals*** - whether recreational, customary, commercial or illegal. Other than illegal, it is pointless laying blame and whilst we must learn from the past, there is clear common interest to secure a healthy and well-managed resource. Critically, responsibility (and therefore investment) must be more carefully shared between government, customary, recreational and commercial sectors. To begin with we recommend:

- ***Recreational.*** Trial voluntary recreational fishing reporting. It is critical to understand the real time level of removals to inform more responsive and timely management decisions.
- ***Commercial***. Enable sustainable management initiatives by the Pāua 7 Management Committee to be included in decision making specifically:
- ***Pāua MAX****:* Bring previously productive areas back into production by enhancement and proactive management.
- ***Dataloggers****: C*ooperate with Fisheries New Zealand to collect fishery-dependent (i.e. the harvesters collect it as they fish) high resolution data for better assessment and management procedures.
- ***Citizen Science****:* Build on the goodwill of divers whether in the pāua industry or community who are regularly in the water and train them (e.g., Reef Check) to build a current and ongoing picture of life under the water, what is there and what's happening.
- ***Regional Authorities:*** Engage with regional councils responsible for coastal waters to improve environmental information and reduce the impacts of land-based activities on the marine environment.

3. ***Collaborate.*** Collaboration is key to pursue resolution of complex resource management challenges. AFL, TML and SBC have built a solid initial interest amongst Iwi, industry, science, local and central government and community stakeholders (including Marlborough Marine Futures) and are committed to deepen relationships. AFL is keen to understand how best to support engagement whilst also enabling shorter term action and outcomes to emerge.

4. ***Economically value the ecosystem service loss.*** The section above explains the valuation approach that can be employed. There is much more research and development needed to develop effective techniques to apply economic valuation to marine ecosystem services as per Figure D-7 below.

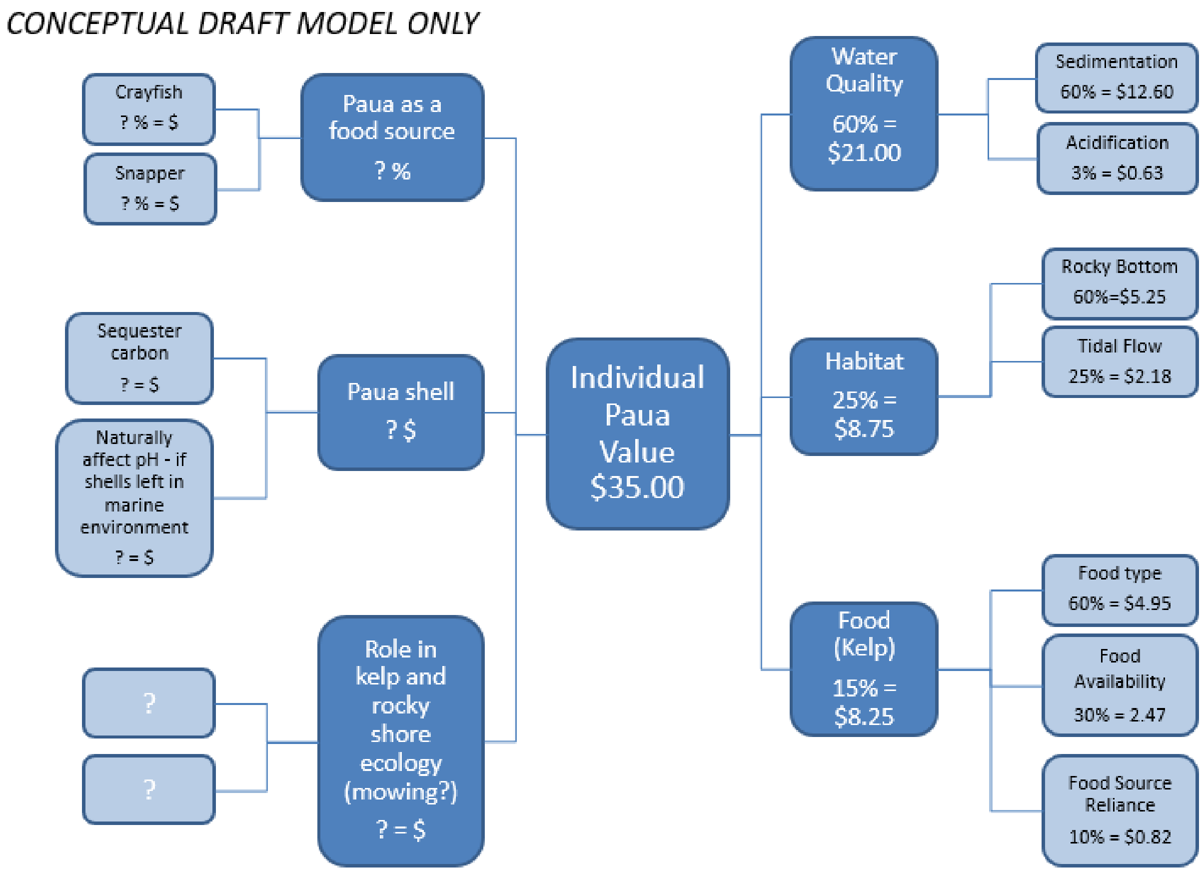

Figure D-7: Conceptual Model of how the ecological inputs required to produce a pāua could be valued and which could guide investment in their protection, management, or restoration. N. Wragg/T. Craig pers. comm. 2015.

ESR Insights

The following insights can be gleaned from the ESR assessment exercise[27]:

- The ESR provided an excellent tool for companies to engage both internally and with external stakeholders. The systematic and comprehensive nature of the process meant that external stakeholders were able to identify with one or more ecosystem service. As a result, participants felt their interests were being considered in the process. Internally, it provided a mechanism for staff to explore issues and management areas in a different light with new opportunities being mutually recognised and which made company commitments easier to secure.

- New insights were revealed. For example, current ingredient procurement processes did not capture all ecosystem service-related risks and one company realised the degree to which they were dependent on the activities of other resource users. This latter example highlighted the importance of engaging in other regional and national

[27] Landcare Research, 2015

dialogues they were previously unaware of and identified potential additional marketing opportunities.

- Robust discussions about ecosystem service risks and opportunities were still possible even where extensive and detailed information on conditions and trends was not available.
- While not a prerequisite, independent facilitation adds value to the ESR process by providing a neutral view point and enhanced credibility of the outcomes.

For Moana New Zealand

Moana New Zealand and the Pāua Industry Council both agreed that the ESR was a useful process. The ESR project provided insights into Moana New Zealand's interaction with the ecosystem services that are important to the health of pāua. It deepened the company's understanding of the broader marine environmental issues affecting its pāua business and the opportunities to address them.

The ESR motivated further discussions and investigations in Moana New Zealand's sustainability, marketing, and procurement. Following the ESR, the team presented the results across key Moana New Zealand management as well as to the PIC Executive and Pāua2 Management Action Committee. It identified important questions for further assessment including identifying priority risk sites for the wild fishery, the financial business implications from cumulative environmental risks, and identifying possible solutions and opportunities to address these collaboratively.

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Appendix E Best Practice Dairy Catchments

Background

The Dairy Catchments study began in mid-2001. The project initially comprised four catchments located in key geo-climatic dairying regions with specific challenges for farmers, with the aim of minimising adverse effects on natural waters. Key objectives of the study were to (i) establish baseline water quality under present farming methods and land use intensity, (ii) define linkages between land use and water quality in order to derive BMPs, and (iii) monitor changes in water quality as a range of improved land management practices were adopted by farmers. Funding for the study was mainly sourced from Fonterra Cooperative Group, the Sustainable Farming Fund (MPI) and the Pastoral21 research consortium (MBIE- and Industry-funded). Local Regional Council resources were also used to support parts of the initiative.

The four study catchments initially established in 2001 were Toenepi (Waikato); Waiokura (south Taranaki); Waikakahi (south Canterbury) and Bog Burn (central Southland). A fifth catchment, Inchbonnie, in the Lake Brunner catchment (West Coast) was included in the study in 2004 Figure E-1). Some of the criteria for catchment selection were:

- Dairy farming was the dominant land use, and farming practices, climate, soils, farm management and topography were generally representative of the region.
- A usefully diverse range of catchments, topographies and climates were chosen that would enable extension of the results more generally.
- Streams were large enough to be regarded as significant by farmers (i.e., not a drain or ephemeral creek) but small enough to have a single major land use. In effect this meant that streams had mean flows of 200-600 L s^{-1} and catchment areas of 6-63 km^2. This had the advantages that streams were hydrologically manageable, especially with regard to flood flows, and lag effects were minimised as much as possible.

The study approach combined surface water quality and quantity monitoring data with approximately 3-yearly surveys of farming management practices, and biennial surveys of soil fertility and quality, to understand land–water linkages. Supplemented by some key experimentation and aggregated farm-scale modelling, appropriate management practices were identified and advocated by the catchment teams to encourage changed practice and improve water quality outcomes. An important aspect of the study was its goal of identifying and adopting improved land management practices that targeted some specific recreational, ecological or community values that were identified and agreed upon by stakeholders within each region. Some of these key values were:

- Fishery – Bog Burn, Waikakahi, Waiokura, Inchbonnie.
- Recreation – Bog Burn, Waiokura, Toenepi, Inchbonnie.
- Domestic water supply – Bog Burn.
- Land drainage – Bog Burn, Waikakahi.

Monitoring data were also augmented by biological surveys of benthic invertebrates and stream habitat in each of the streams and compared with relatively un-impacted sites nearby. A key feature of the study was that it was a collaboration involving the dairy industry as a whole, farmers in each of the catchments (and later, others from further afield), regional council scientists, fertiliser companies, MAF and the scientific and technical advisory agencies (viz. AgResearch, Dexcel/DairyNZ and NIWA).

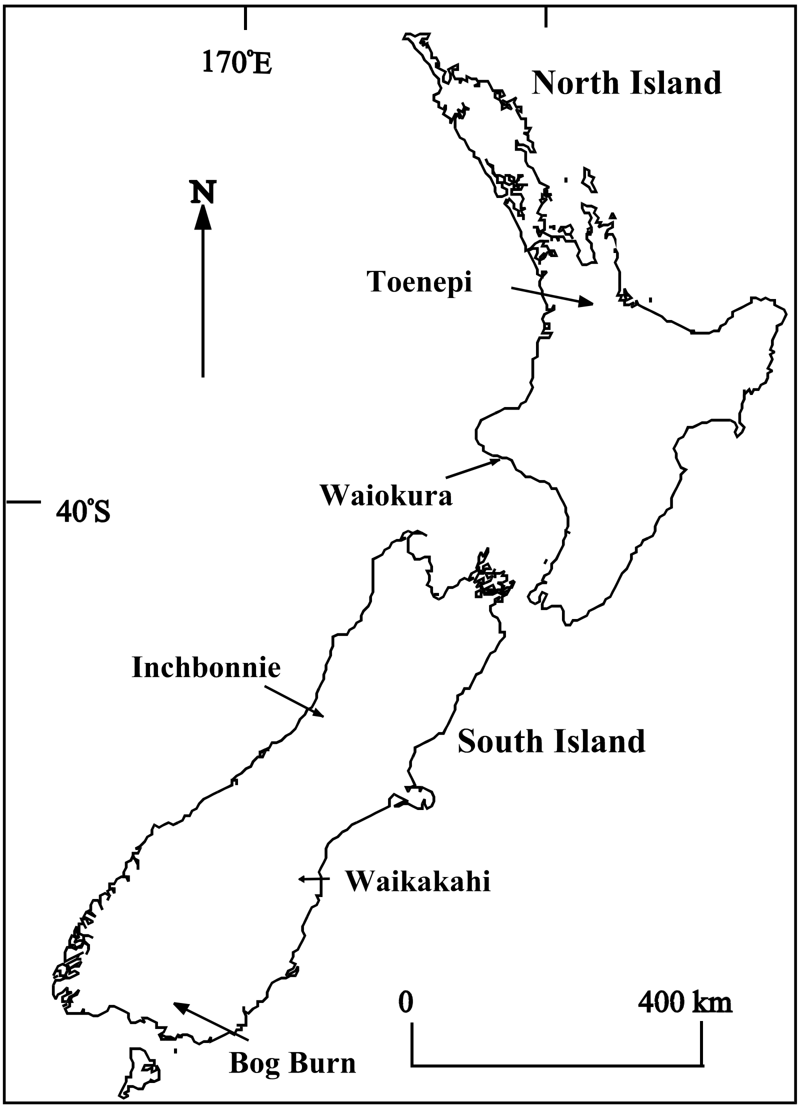

Figure E-1: Approximate locations of the five dairy study catchments.

Findings

Catchment monitoring showed that water quality was initially degraded with respect to N, P, suspended solids (SS), Escherichia coli (E. coli) concentrations and macroinvertebrate composition (Wilcock et al. 2013b; Wright-Stow and Wilcock, 2017). These findings were not altogether unsurprising and are typical of catchments with intensive pastoral land use. Trend analysis has however shown some significant improvements in many analytes and in many of the catchments (Table E-1). This was most evident for sediment and water clarity which are thought to have improved due to increased riparian protection (all catchments; Plate 1) and the more judicious grazing of winter forage crops (Waikakahi catchment). Regular surveys of managers and landholders within all catchments documented increases in farm productivity over the life of the project, and particularly in the Waikakahi and Bog Burn catchments where land conversion to dairy farming was still underway. This contributed to increased loads of N to water in the latter 2 catchments, and probably to the increase observed for E. coli concentrations in Bog Burn stream (Table E-1).

Table E-1: Summary of water quality trends in the five catchment streams (from Wright-Stow and Wilcock 2017).

Catchment	Period	total N	NOx-N	total P	*E. coli*	Sediment	Clarity
Toenepi	1995-2013	NST	NST	NST	NST	↓	↑
Waiokura	2001-2013	↓	↓	NST	↓	↓	↑
Waikakahi	1994-2013	↑	↑	NST	↓	↓	↑
Bog Burn	2001-2013	↑	↑	NST	↑	↓	NST
Inchbonnie	2004-2011	↓	↓	↓	NST	↓	↑

Notes: Arrows indicate significant (P < .05) upward or downward trends over the entire sampling duration. NST is either no significant trend or a zero-trend slope.

The implementation of improved effluent and water management practices were identified as likely contributors to observed reductions in E. coli concentrations in the Waiokura and Waikakahi catchments. In the case of Waiokura, this was helped by the on-going shift away from 2-pond farm dairy effluent (FDE) treatment systems to land application, in many cases using deferred effluent irrigation practices. The gradual removal or laser-levelled re-bordering of flood irrigation bay (Monaghan et al. 2009; Plate 2) and improved FDE irrigation practices are believed to have contributed to the reduced concentrations of E. coli observed in Waikakahi stream.

Figure E-2: Plate 1: Riparian protection and planting works in the Waikakahi and Waiokura catchments.

Although water quality in the five dairy streams improved somewhat in response to the implementation of the management practices described above, further measures will be needed if streams are to comply with guidelines for slightly disturbed lowland river ecosystems or contact recreation (Wilcock et al. 2013b). This raises the question of whether "slightly disturbed" is an appropriate status threshold for catchments that generate considerable socio-economic activity. Current scientific endeavour, such as that undertaken within the Our Land and Water National

Science Challenge initiative, is seeking to develop tools and processes that can guide catchment deliberations to help produce mutual benefits whereby aspirations for economic and environmental outcomes are fully considered. These integrated assessments will be particularly important for analytes such as nitrogen (N), which is relatively mobile in the landscape and, partly in consequence, relatively expensive to mitigate.


Figure E-3: Plate 2: Irrigation improvements in Waikakahi catchment: re-bordering of irrigation bays or shift to spray irrigation.

Key messages

- Each of the Dairy Catchments provided an outdoor laboratory where farmers, scientists, regulatory agencies and industry and community groups deliberated on the ecological and agricultural status of soils and streams.
- The scale and structure of the study provided meaningful data and insights that helped to accelerate the implementation of selected measures that were identified to protect and enhance some key catchment values. Although water quality was initially assessed as relatively poor, the trajectories of improved water quality metrics (16) outnumbered those where water quality was declining (5); no significant trends were discernible for the remaining 9 (water quality x catchment) metrics (Table E-1).
- The 3–7-year periodicity of major climate cycles that affect rainfall and catchment runoff, the volatility of market forces influencing farm income and costs, and a slow rate of farmer adoption of some BMPs mean that catchment monitoring programs in New Zealand need to be much longer than 10 years if they are to detect changes in water quality caused by farmer actions.

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Appendix F Pakuratahi Stream

Summary

- Harvesting planted forests of radiata pine on soft-rock hill country in the coastal Hawke's Bay region has the potential to increase erosion and sedimentation, posing a risk to downstream receiving aquatic environments. The Pakuratahi land use study was designed to assess that risk.
- Harvesting increased annual water yields by an average of 22% compared with a nearby pasture catchment.
- Peak storm flows increased in the planted forest catchment in during the harvest and immediate post-harvest period, exceeding peak flows in the pasture catchment, but were below pasture peak flows for the remainder of the post-harvest period.
- Prior to harvest, sediment yields were 3-4 times higher from the pasture catchment compared with the planted forest catchment. During the logging component of the harvesting phase (roading and logging), sediment yields were 2-3 times higher than the pasture catchment. Forest management practices of roading and infrastructure management, oversowing and rapid replanting to re-establish vegetation cover reduced sediment yields to pre-harvest levels within 2-3years.
- Over the 11-year period of sediment monitoring sediment yields were substantially lower in the planted forest compared with the pasture catchment (713 t/km^2 and 1168 t/km^2 respectively) due to the lower erosion rates in the planted forest catchment.
- No statistical changes in water quality were detected in the planted forest catchment following harvesting.
- The forested streams contained a high diversity of native fish and aquatic invertebrate fauna prior to harvesting. Harvesting had minimal impacts on native fish communities. Invertebrate communities showed a marked change from a mayfly (Ephemeroptera) dominated composition to more disturbance tolerant taxa (i.e., Chironomids, Elmidae, *Potamopyrgus antipodarum*), similar to invertebrate communities in the pasture streams.
- Invertebrate communities recovered within 2-3 years of harvesting at the base of the catchment, where flows were higher and sediment inputs minimal. In headwater streams communities had yet to recover 5-6 years after harvest, most likely due to the increase in fine sediment and limited hydrological capacity to flush the sediment from the system.

Background

In 1993, a 12-year paired catchment study (Pakuratahi land use study) was established in the steep coastal soft-rock erosion-prone hill country in the Hawke's Bay region, in response to concerns that harvesting of planted forests would increase erosion and sedimentation with subsequent impacts on water quality, flow and aquatic biota. The aim of the study was to monitor a range of in-stream parameters (discussed in more detail below), to assess the impacts of forestry from the pre-harvest (1993-1997), harvest (December 1997-October 1999), and post-harvest phase of the forestry cycle that included replanting in 1998-2000, through to canopy closure in 2005 when the replanted trees were 5-7 years old. An adjacent pasture catchment provided comparison as a control (Eyles and Fahey, 2006a).

The planted forest Pakuratahi catchment (345 ha) and pasture Tamingimingi catchment (795 ha) are located in the Hawke's Bay region, approximately 18 km north of Napier and 3 km inland from the coast (Fig. G1). The Hawke's Bay region has a temperate climate with warm summers and moderate winters, but is prone to the extremes of drought and major storm events, and highly variable rainfall patterns. Mean annual air temperature at Tangoio, 5 km north of the catchment is 13.3°C and mean annual rainfall (1951-1980) is 1501 mm although annual variation is high. The climate at the study site for the trial duration was typical for coastal Hawke's Bay, experiencing both droughts and variable rainfall. This included several high rainfall events all of which were below the estimated 5-year return period for a 24-h rainfall event. Rainfall at the two rain gauges at the study site (Figure F-1) averaged 1246 and 1044 mm/yr (Fahey and Marden, 2000, Wood and Fahey, 2006).

Approximately 50% of the pasture catchment and 60% of the planted forest catchment are in slopes ≥ 20° with more deeply incised valleys in the planted forest catchment (Fransen and Brownlie, 1996). The catchments are underlain by coastal sedimentary geology comprising a mix of greywacke conglomerate, silica-rich and shelley sands or sandstone, shelly limestone and mudstones deposited in terrestrial and marine environment, dominated by the Kaiwaka Formation. Boundaries between more permeable layers of sands, gravels and loess and impermeable layers of mudstone and ash, predispose these sites to slope failure. Soils have developed from the weathering, erosion and deposition of bedrock materials, along with rhyolitic airfall deposits from the Taupo Volcanic Zones. Tephric (Recent) soils are located predominantly along the ridge lines, Orthic soils and Rendzic Melanic soils cover the majority of the upper and lower slopes, with fluvial Recent soils occurring in the valley bottoms and floodplains (Fransen and Brownlie, 1996).

The original indigenous vegetation cover that was present prior to human settlement, was removed by a combination of fire, vegetation clearance and conversion to pasture. The loss of forested cover accelerated erosion processes, exacerbated by major storm events along with a major earthquake in 1931. While both catchments reverted back into scrub and weeds to varying extent, historically, the Tamingimingi catchment has been more intensively grazed than the Pakuratahi where much of the catchment was in scrub prior to planting in *Pinus radiata* in 1971/72. Concerns that forest harvesting may renew historical erosional processes with potential impacts on receiving aquatic environments was a key factor influencing the establishment of this study (Fransen and Brownlie, 1995, Eyles and Fahey, 2006b).

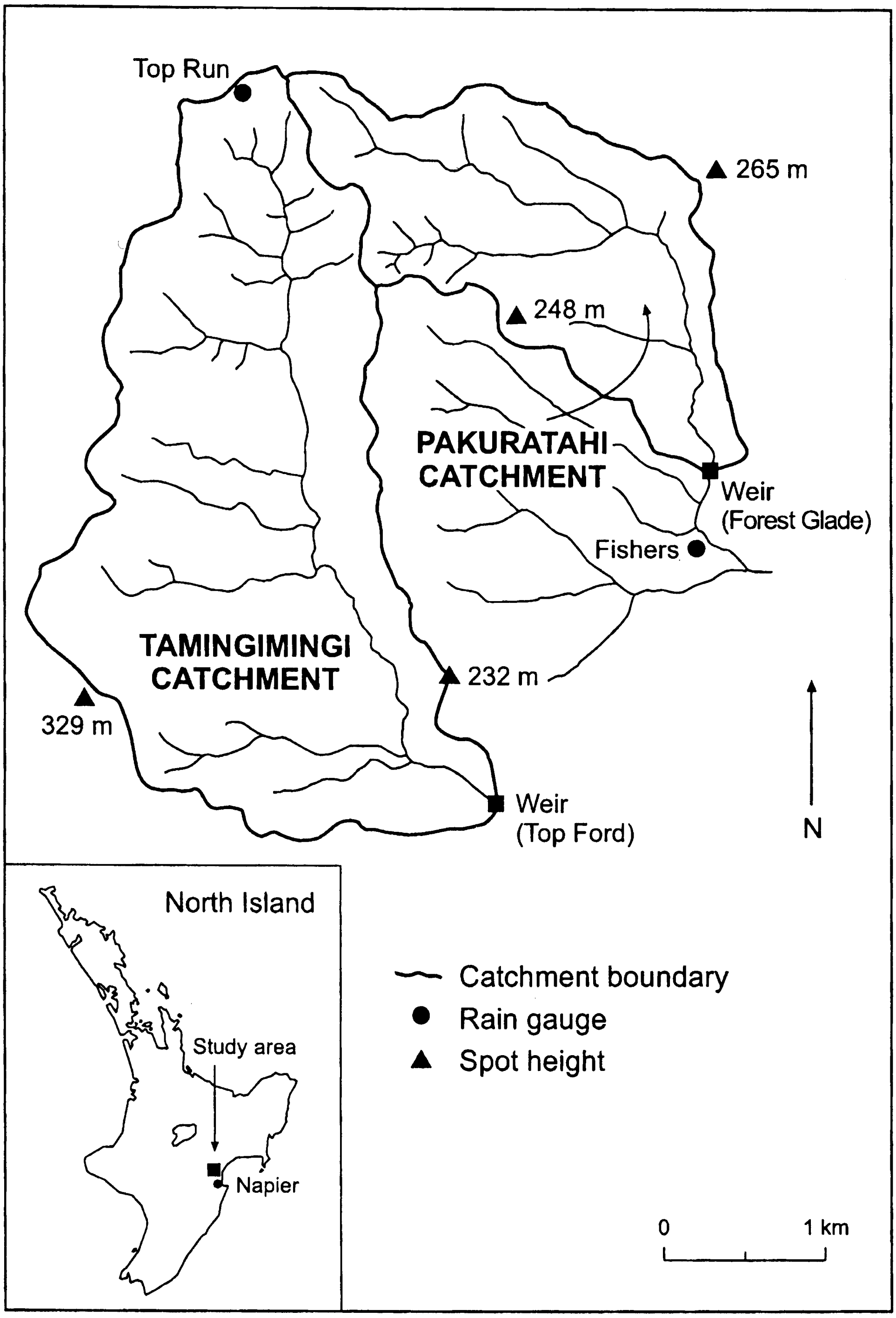

Figure F-1: Location of the Pakuratahi land use study (Fahey and Marden, 2000).

Current land use/management in the catchments

Pasture (Tamingimingi)

Pasture is the predominant land cover in the Tamingimingi catchment (≈ 93%) with small scattered patches of indigenous vegetation. There are two main farm owners in the catchment. Land use in the catchment is in a mix of sheep and cattle farming with low average stocking rates of 9.8-12.4 su/ha. Fertiliser application rates average 250 kg/ha of superphosphate, applied biannually. The farms are susceptible to droughts and floods and slip erosion is common with some eroding gully heads in the catchment. Both properties were managed consistently throughout the study period, providing a comparative 'control' for the forestry catchment (Eyles, 2006).

Planted forest (Pakuratahi)

The trial covered three main phases in a forestry cycle; pre-harvest, harvest (includes roading and harvesting), and post-harvest through to canopy closure. Land cover and land use in the Pakuratahi catchment at the commencement of the trial was predominantly planted forest (≈85%) with another ≈7% in indigenous vegetation located mainly in riparian stream margins and gully headwaters. Tree socking was typically 225 stems/ha with grazed (feral goats) grass undergrowth.

Roading and harvesting activities commenced in January 1998 and were completed in October 1999 (Gilmore, 2006). The programme involved considerable infrastructure upgrade and additional roads and landings. Most of the area (85%) was harvested using cable logging with ground-based operations confined to ridge crests. Best management practices used to minimise environmental impacts included; two-staging of logs (trees were extracted to a landing and then transported to a centralised landing outside of the catchment for processing) to minimise landing size, confining roading and landing infrastructure to ridges wherever possible, use of tall yarders to maximise lift and minimise soil disturbance during log extraction, extracting logs away from the stream channel wherever possible to protect riparian vegetation and minimise logging slash inputs, felling patterns to minimise windthrow and post-harvest maintenance of roading and landing infrastructure to maintain water and logging slash controls.

Post-harvest land preparation included aerial desiccation of weeds and regenerating radiata pine, followed by oversowing of a mix of grasses to re-establish ground vegetation cover, reduce sedimentation and assist with weed control (Gilmore, 2006). Pest control programmes were in place for goats and possums. The Pakuratahi catchment was replanted with Pinus radiata between 1998 and 2000 at 830-850 stems/ha with additional planting set-backs from stream edges. The catchment was selectively thinned to maintain desired stocking rates and pruning (up to 6 m) was confined to the easier terrain in the catchment. By the end of the project (December 2005) the replanted forest had achieved canopy closure, with a dense undergrowth of weeds, grasses and regenerating natives. The main pressures on the aquatic environment in the forestry catchment were from harvesting activities that removed the forest cover and the impact of this process on the stream hydrology and sediment regime in the catchment.

Forestry effects on hydrology

As the pasture catchment is more than twice the size of the forested catchment, hydrological variables expressed as 'per unit area' have been used for comparative purposes (e.g., Figure F-2) (Wood and Fahey, 2006). In the pre-harvest phase water yields were higher from the pasture catchment than the planted forest where the canopy cover would have intercepted a higher percentage of the rainfall (Figure F-2). This pattern was reversed during the harvest phase when the forest canopy was removed and continued through the post-harvest vegetation recovery phase through to canopy closure. A similar pattern was observed for annual water yields which were 6% lower in the forested catchment compared with the pasture catchment in the pre-harvest phase, but exceeded annual water yields from the pasture catchment by an average of 22% during the harvest phase. This difference declined to 16% in the post-harvest phase as the forest cover re-established. By canopy closure at the end of the project annual water yield in the forested catchment, exceeded the pasture catchment by 5% but had yet to return to pre-harvest levels.

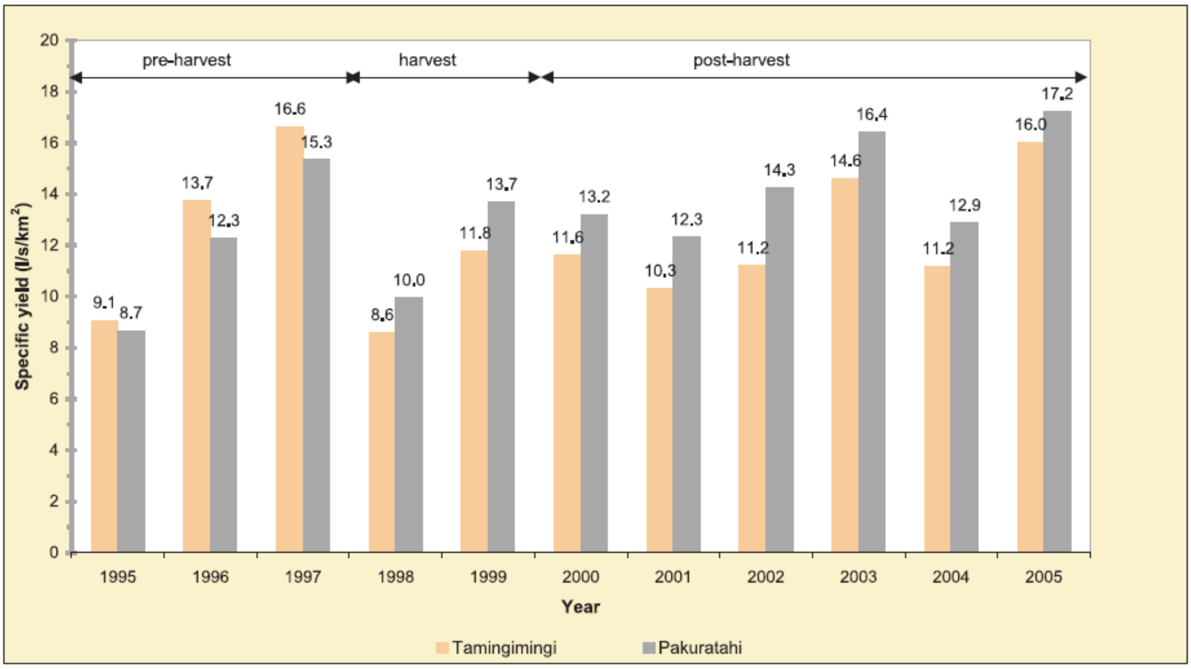

Figure F-2: A comparison of specific yields from the Tamingimingi (pasture) and Pakuratahi (planted forest) catchments (Wood and Fahey, 2006).

During storm flows, peak storm run-off was greater from pasture compared with planted forest in the pre-harvest period. This trend was reversed during the harvest phase although there was a lag effect after harvesting commenced, before this reversal occurred (Figure F-3) (Wood and Fahey, 2006). Within six months of harvest completion, pasture peak flows were once again higher than the planted forest and this trend persisted for the remainder of the post-harvest phase. However, minimum annual 7-day low flows were higher from the planted forest compared with the pasture catchment in the pre-harvest phase and this was attributed to the higher contribution of groundwater seepage from limestone outcrops in the forested catchment. This difference increased to an average of 0.26 mm/yr during the harvest phase and persisted into the immediate post-harvest phase, declining in the final years of the project where low flows approached pre-harvest levels (Wood and Fahey, 2006).

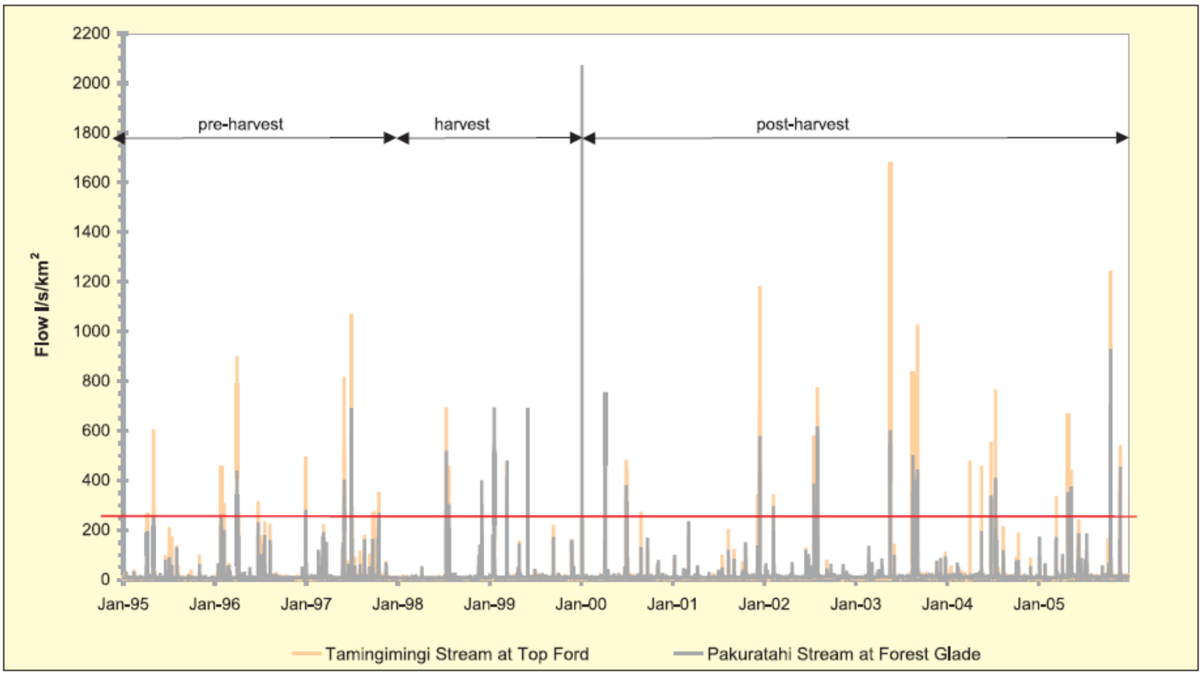

Figure F-3: Area standardised hydrographs for the Tamingimingi (pasture) and Pakuratahi (planted forest) catchments (Wood and Fahey, 2006).

Forestry effects on sedimentation

The main contaminant discharged to the stream system from the forestry catchment was sediment. Prior to harvesting, sediment yields from the pasture catchment were 3.7 times higher than from the planted forest catchment (Figure F-4) (Fahey and Marden, 2006, Fahey and Marden, 2000, Fahey et al. 2003). During the harvest phase, which included both the roading preparation and harvesting of the trees, sediment yields increased from the planted forest catchment and were 1.4 times higher in comparison to the pasture catchment. For the harvesting component only, the sediment yield was 2.6 times higher (Figure F-4) (Fahey and Marden, 2006). The main sediment sources were cutbank and sidecast failures, shallow landslides and channel bed and bank scouring. In the first year after harvest sediment yield from the planted forest catchment still exceeded pasture but for the remainder of the post-harvest period, the pasture was yielding four times more sediment than the planted forest catchment. Post-harvest management of roading and landing infrastructure, oversowing to re-establish ground cover and reduce run-off and rapid replanting contributed to the rapid decline in sediment yields within two years of the completion of harvesting in the catchment. Entrapment and storage of sediment in headwater stream channels by vegetation and logging slash most likely contributed to the decline as well (Baillie, 2006). Over the 11-year period of sediment monitoring sediment yields were substantially lower in the planted forest compared with the pasture catchment; 713 t/km2 and 1168 t/km2 respectively (Fahey and Marden, 2006). This was attributed to the lower erosion rates in planted forest catchment compared with pasture, resulting in overall lower sediment yields in spite of the sediment pulse during harvesting.

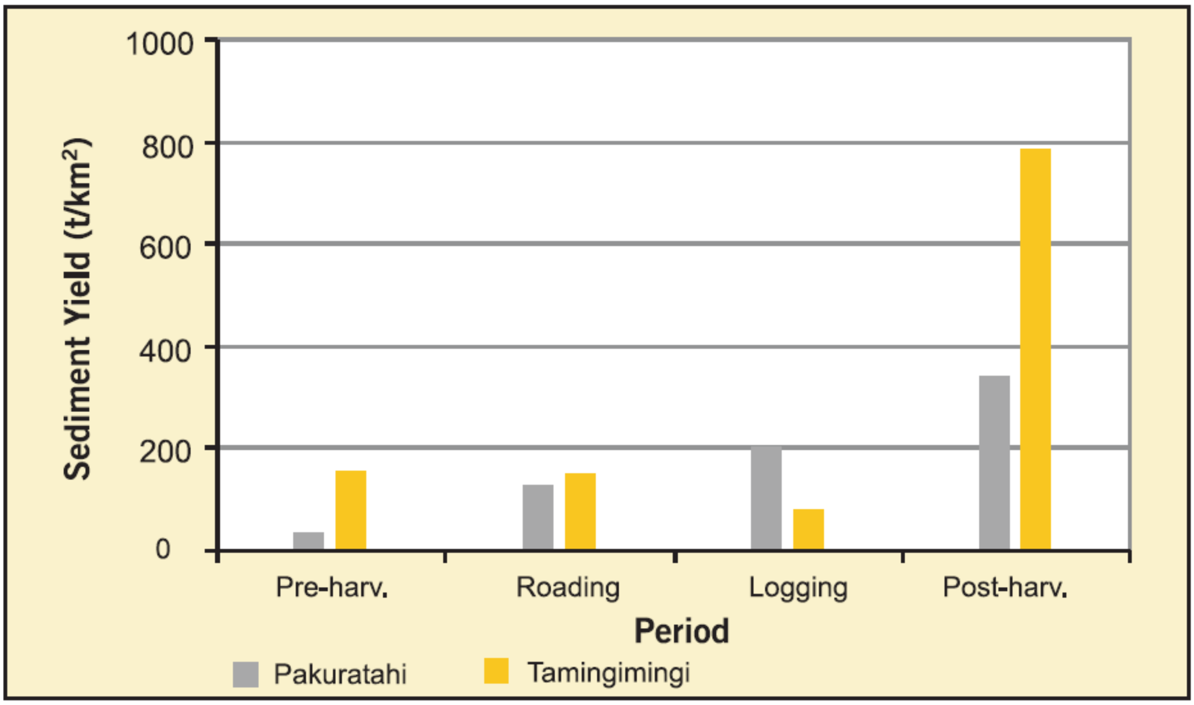

Figure F-4: Estimated sediment yields from the Tamingimingi (pasture) and Pakuratahi (planted forest) catchment (Fahey and Marden, 2006).

Forestry effects on water quality

A wide range of water quality variables were monitored in both catchments at varying sampling intervals. Key variables are presented in Table F-1 but refer to Fahey and Stansfield (2006) for full details. While some variables showed slight increases and occasionally decreases after harvest, no significant differences were detected in black disc visibility, turbidity, electrical conductivity, nitrate-N, ammoniacal nitrogen, TKN, total P and total dissolved P. Electrical conductivity and a range of cations and anions showed slight, but non-significant increases after harvest. Indicators of microbial contamination also showed no significant response to harvesting. Water temperature increased in both catchments after harvest indicating a greater influence of climate conditions than harvesting. Stream water pH was in the alkaline range for both catchments, a reflection of the carbonate lithologies present, and the post-harvest increase in pH was only significant in the pasture catchment, indicating that harvesting was not the causal factor.

Table F-1: Water quality in the forestry and pasture catchment before and after harvest. Median values and sample sizes ().

Monitoring site	nitrate-N (mg/L)	NH4-N (mg/L)	TSP (mg/L)	FC (cfu/100 ml)	SS (mg/L)	Turbidity (NTU)	TN (mg/L)	TP (mg/L)
Forestry								
Before harvest	0.66(64)	0.022(32)	0.002(40)	1(1)	2(65)	1(64)	0.33(5)	0.006(65)
After harvest	0.477(27)	0.01(60)	0.003(49)	25(25)	1.5(59)	0.925(60)	0.789(6)	0.005(60)
Pasture								
Before harvest	0.5(61)	0.026(30)	0.003(38)	31(1)	2.5(63)	1(62)	0.52(3)	0.011(62)
After harvest	0.49(27)	0.015(60)	0.007(50)	120(25)	2(59)	0.65 (60)	0.745(6)	0.01(60)

nitrate-N: nitrate-nitrogen, NH4-N: ammoniacal nitrogen, TSP: total soluble phosphorus, FC: faecal coliforms, SS: suspended solids, TN: total nitrogen, TP: total phosphorus.

Impact on the aquatic biota

Native fish

Monitoring of fish populations at the base of the two catchments recorded ten native fish species, seven of which were present in both catchments; longfin eel (*Anguilla dieffenbachia*), shortfin eel (*Anguilla australis*), bluegill bully (*Gobiomorphus hubbsi*), common bully (*Gobiomorphus cotidianus*), redfin bully (Gobiomorphus huttoni); inanga (*Galaxias maculatus*) and torrent fish (*Cheimarrichthys fosteri*). Koaro (*Galaxias brevipinnis*), banded kokopu (*Galaxias fasciatus*) and common smelt (*Retropinna retropinna*) were rarely caught and found in the forested catchment only (Black, 2006). These species are currently classified as either 'Not Threatened' or 'Declining' (Goodman et al. 2014).

Prior to harvesting feral goat grazing along the riparian margins of the monitored reach in the forested catchment, maintained a similar bedform (pool/riffle/run morphology) to that in the pasture catchment. After harvest, the increase in solar radiation, along with pest control, has resulted in abundant regrowth of blackberry, grasses, and sedges, reducing the channel width, and altering both the habitat and hydrological regime. These changes have not affected overall fish biodiversity, but have influenced the proportional abundance of individual species, although the author considered that changes insufficient to indicate a negative impact from harvesting (Black, 2006). However, fish sampling was confined to the base of the catchment, the point in the stream system that was the least impacted by harvesting activities, therefore impacts on native fish populations in the more impacted headwater streams are unknown. The spring-fed base flows in both catchment have maintained higher base flows than would normally occur in non-spring-fed streams, moderating temperature increases that typically occur in small headwater streams following harvesting and buffering the impacts of land-use (Quinn and Wright-Stow, 2008, Baillie et al. 2005). These factors have likely contributed to the diverse fish populations present throughout the study period.

Aquatic macroinvertebrates

Aquatic invertebrates were monitored at three sites in both the planted forest (Figure F-5, P1-P3) and pasture catchments (Figure F-5, T1-T3) and sampled at varying intervals through the study period. Benthic sediment and periphyton were monitored at the same sites (Death and Death, 2006, Death et al. 2003). Aquatic invertebrate communities in the planted forest catchment were dominated by a diversity of mayfly species prior to harvest but changed markedly in the post-harvest period to communities dominated by Chironomidae, *Aoteapsyche* sp., Elmidae, Ostracoda and *Potamopyrgus antipodarum*, similar to invertebrate communities in the pasture catchment (Figure F-5). Taxa numbers declined significantly in the immediate post-harvest period in the forested catchment (Death et al. 2003). The Macroinvertebrate Community Index (MCI) and Macroinvertebrate Community Index (QMCI), biological indicators of water quality, were primarily in the 'clean water' category in the headwaters prior to harvesting and ranged from 'clean' to 'possible mild to probable moderate pollution' at the base of the catchment. Both these indicators declined after harvest to 'probable severe pollution' at all three sites, reflecting the shift to more disturbance tolerant taxa in the post-harvest period. These changes were attributed to increased light levels following canopy removal, a short-term significant increase periphyton biomass in the immediate post-harvest period and more particularly a significant increase in the proportion of sand and silt in the stream channel, which persisted in the two headwater sites (Death et al. 2003, Death and Death, 2006).

In the smaller headwater streams some community characteristics were just returning to pre-harvest conditions or had yet to recover 5-6 years after harvest (Death and Death, 2006). In some headwater reaches, sediment, regenerating vegetation (grasses and blackberry) and logging slash had completely infilled the stream channel (Baillie, 2006). Full recovery of flow, habitat and the aquatic invertebrate communities in these streams will be influenced by their capacity to flush the sediment from their systems (Death and Death, 2006). In contrast, at the base of the forested catchment (Fig. G5, P3) where flows and flushing capacity were higher and sediment accumulation was minimal, aquatic invertebrate communities had largely recovered within three years of harvesting (Death and Death, 2006).

Invertebrate community recovery from a harvesting disturbance is taking longer than from a storm event in July 1997 when the catchment was in mature forest and where communities recovered within 5 months (Death et al. 2003, Death and Death, 2006) (Figure F-5).

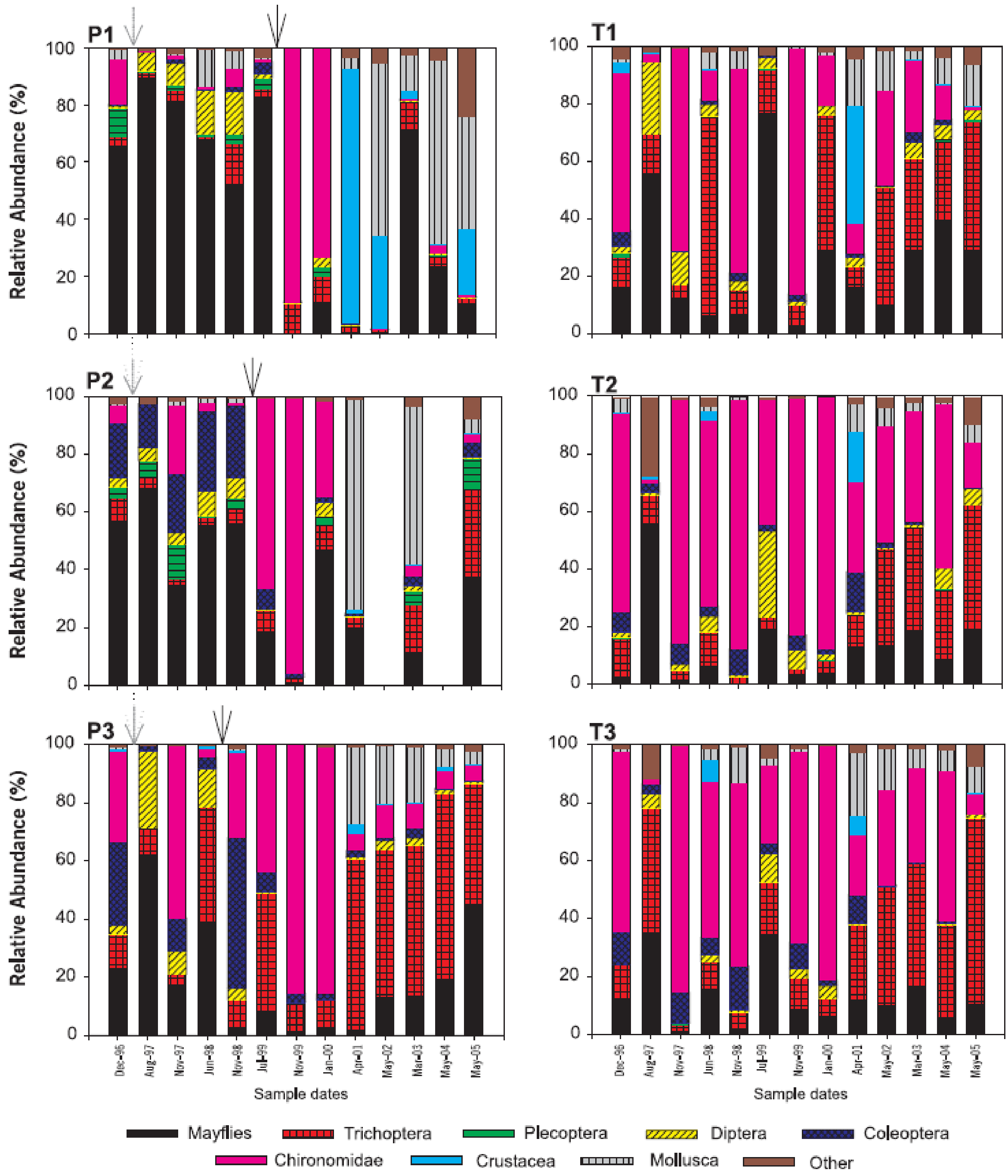

Figure F-5: Relative abundance of invertebrates collected in 4 Surber samples between December 1996 and May 2005 in 3 planted forest streams and 3 pasture streams. A dashed arrow indicates the July 1997 storm and a solid arrow the harvest date (Death and Death, 2006).

Conclusions

The pressure of catchment-scale forest harvesting on receiving aquatic environments was assessed by comparing a range of variables before and after harvesting and against a 'steady state' land-use of agriculture in an adjacent catchment. The state of in-stream parameters of water yield, peak storm flow, sediment yield, a range of water quality variables, benthic sediment and chlorophyll *a* (periphyton) were measured throughout the pre-harvest, harvest and post-harvest phase. These parameters have been used, where applicable, to explain the relationships between forestry land-use and land management practices and the impact on receiving environments, using native fish and aquatic invertebrates used as the indicators.

- There was a strong relationship between the three forestry phases and water yield, and peak storm flows. Both variables increased after harvest followed by a subsequent decline toward pre-harvest levels, although the recovery trajectory was more rapid for peak storm flows, than for water yield.
- Directly linking the pressure of harvesting on the state of water yield and peak storm flows to the impact on ecological conditions was more tenuous and descriptive. While no direct analyses were undertaken between water yield or flows and aquatic invertebrate community composition, the higher post-harvest water yields would have contributed to the higher flows at the base of catchment, assisting in flushing fine sediment from the system, which contributed to the reduced impact of harvesting and a more rapid recovery of aquatic invertebrate communities, compared with the headwater sites. It is likely that these processes also contributed to the lack of definitive impacts on native fish communities.
- No relationships were identified between the three forestry phases and a range of water quality variables. For stream temperature, the high spring-fed base flows in the catchments most likely provided a buffer against post-harvest stream temperature increases, often associated with forest canopy removal (Baillie and Neary 2015 and references therein).
- While there was a short-term significant increase in periphyton abundance immediately after harvest, the increase was not considered sufficient to impact on aquatic invertebrate community composition.
- There was a significant increase in the percentage of sands and silt prior to and immediately after harvest. However, this impact on benthic sediment was minimal at the base of the catchment but persisted in the small headwater tributaries, highlighting the spatial and temporal variability in responses.
- The strongest causal link identified in this site was: harvesting (pressure) → increased sediment yields (state) → increased benthic fine sediment (state) → shift to disturbance tolerant aquatic invertebrate communities (impact) → replanting and oversowing (pressure) → decrease in sediment yield (state) → varying spatial recovery trajectories of aquatic invertebrate communities toward preharvest composition.

Comparisons with the pasture catchment was useful in separating harvesting impacts from other seasonal and inter-annual variations.

No information was found on the impact of harvest on the cultural, economic and social values associated with the freshwater environments in these catchments. A storm event after harvest generated a debris flow in one of the headwater streams that travelled downstream and into the main stem of the Pakuratahi River (Baillie, 2006). However, no information was found on whether there were any impacts to downstream freshwater and coastal receiving environments or downstream infrastructure (i.e., fences, culverts, bridges).

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Appendix G Canterbury Plains Aquifers

Summary

- Canterbury, as the largest region in New Zealand, has the largest water resources and also the highest water use.
- In recent years there has been an increase in groundwater abstraction volumes (for irrigation and drinking water supply) and the development of deeper groundwater resources, with annual groundwater use exceeding annual surface water uptakes.
- Intensification of agriculture has led to increasing loading of nutrients (e.g., nitrate), pesticides, bacteria and sediment, on the land surface, which poses a significant risk of contamination of groundwater sources within the region.
- Current research is examining the abundance of micro-, meio- and macrofauna and their relationship with the surrounding environment and groundwater quality.

Setting

The Canterbury Plains encompasses the land between the Orari and Ashley rivers in the south and north, the river-gorges in the west and the seashore and inland boundary of Banks Peninsula in the east and southeast. The Plains have very little topographical variation with a gradual increase in elevation from the coast up to the foothills as well as slight depressions along the path of the rivers. Glacial outwash deposits of the major rivers have built up the Canterbury Plains from the Southern Alps, and fluvial episodes have then re-distributed the sediments. The plains extend over a 50 km wide by 150 km long area from Timaru to the Waipara River. The Canterbury Plains are a complex of coalescing fans overlying a basement of Permian to Jurassic Torlesse Super Group rocks. The fans were deposited during the late Tertiary and Quaternary periods by eastward-flowing rivers that emerged from the foothills of the Southern Alps. The inland plains are comprised predominantly of gravels (> 500 m thick). At the coastal margins of the plains, fine-grained marine sediments with layers of coarser gravels and sands of the coastal fan complex result in a layered sequence of gravel aquifers separated by aquitards of relatively less permeable marine sediment (Cameron, 1992).

Aquifer geometry and hydraulic properties

Quaternary gravel aquifers (e.g., Springston/Christchurch Formation, Riccarton Gravel Formation, Linwood Gravel Formation, Burwood Gravel Formation, Wainoni Gravel Formation) are widespread and their thickness range from 250 m up to 600 m (Brown and Weeber, 1992). The groundwater aquifers are unconfined or semi-confined under much of the inner plains, becoming confined near the coast around Kaiapoi, Christchurch and Lake Ellesmere. The aquitards (e.g., Bromley Formation, Heathcote Formation, which in the western parts of the Christchurch area are alluvial sediments) were deposited in interglacial periods, although there are water bearing gravel lenses within these generally low permeability sediments (Davey, 2006). Highly permeable buried river channels with open framework gravels, which form a complex network at all depths in the aggradational deposits of the lower plains, are the preferred flow paths for groundwater in the Canterbury Plains (Wilson, 1973; Taylor et al. 1989; Bal, 1996; Davey, 2006; Dann et al. 2007). These gravel lenses have vertical thicknesses from a few centimetres to a metre, and the horizontal widths tend to be from sub-metre to a few metres wide (Scott et al. 2011).

The shallow and deep aquifer zones are separated by a semi-confining layer of lower permeability clay-bound gravels, which is thought to behave like a leaky aquitard. Aquicludes are known only in the vicinity of Christchurch although they may also occur over small areas at other locations near the seashore. Calculated aquifer transmissivity values for the Central Plains area range from 200 to 15,770 m^2/d with a geometric mean value of 2,050 m^2/d (Williamson et al. 2010). In general, transmissivity and storativity values are higher in the shallow aquifer zone than the deeper zone.

Flow regime

The Canterbury Plains are relatively dry and susceptible to prolonged periods of drought.

Typical summers have low rainfall that coincide with increased temperature, while winter conditions lead to increased rainfall and decreased temperatures (Harrison and Gomez, 2013)). Changes in climate (i.e. increased temperatures will impact Canterbury water supply through decreased rainfall, increased evapotranspiration and decreased water storage (Srinivasan et al. 2011). Geological, hydrological, isotope (tritium and ^{18}O) and chemical (mainly nitrate and chloride concentrations) evidence has been interpreted to give a picture of the recharge sources and flow patterns of the important groundwater resource in the deep glacial and interglacial deposits of the sector of the Canterbury Plains between the Selwyn and Ashley Rivers. Most of the groundwater in the Canterbury plains aquifers are river-recharged, but some areas of significant local precipitation recharge contribution have been clearly identified by ^{18}O and chemical concentrations (mainly nitrate and chloride) (Taylor et al. 1989). Pressure distribution, tritium and chemical data reveal that the artesian ground-water underlying Christchurch ascends from deeper aquifers into the shallowest aquifer via gaps in the confining layers of which most of this flow is induced by withdrawal. The Christchurch aquifers are recharged by infiltration from Waimakariri River in its central Plains reaches, and the resulting flow regime is east- and southeast-directed. Recharge in the Ashburton-Hinds plain comes from rainfall, irrigation, seepage from both the Hinds River and the Ashburton River/Hakatere, and water losses from the Valetta irrigation scheme (Cameron, 1992; Hanson and Abraham, 2010). The general direction of groundwater flow in the Ashburton-Hinds plain is from northwest to southeast, parallel to the flow of the Hinds River and Ashburton River/Hakatere (Davey and Attema, 2004). Prior to the commencement of major irrigation schemes in the Canterbury region, rainfall and seepage from rivers and streams was the main source of groundwater recharge. After the commissioning of irrigation schemes there has been significant groundwater recharge by irrigation water, which maintains an artificially high groundwater table during parts of the year (e.g., summer). Groundwater discharges from the groundwater system in the Canterbury region through seepage into rivers and also via several springs (Hanson and Abraham, 2010; Scott et al. 2011). In addition, some discharge may also occur along the coast or through the sea floor off the coast (Taylor, 1996; Brown 2001); however, this discharge is not well investigated or documented.

Groundwater levels

Groundwater levels vary seasonally and are often lowest towards the end of the summer irrigation season. Seasonal fluctuations in groundwater levels are about 0.7 m for each of the confined aquifers and from 5 to 10 m for the inland unconfined aquifers (Cameron, 1992). A 30 m bore located at the Canterbury Museum that taps the first confined aquifer shows that the average groundwater level declined by about 0.5 - 1.0 m over the period 1895- 1905, and then remained steady (but fluctuated seasonally) through to 1992 (Cameron, 1992). The natural fluctuating patterns in groundwater levels is accentuated by the increasing abstractive demand. Compared to December records from previous years, 52% of the wells monitored by Environment Canterbury in December 2016 had 'low' groundwater levels (0 to 1 standard deviation below the mean) and 9% had ‘very low’ levels (more than 1 standard deviation below the mean). This trend was particularly notable in the coastal Selwyn-Waihora area (the spring-fed tributaries of Te Waihora/Lake Ellesmere), the Ashburton plains and the West Melton area (Environment Canterbury, 2016).

Water resources management

Groundwater is the major source of drinking water in the Canterbury region. Almost three quarters of the community water supply, including all of Christchurch, is sourced from groundwater. Most private, domestic water supplies are also derived from groundwater (Hanson et al. 2006). As of 2017 there were 29,983 permitted groundwater takes in the Canterbury region, for a total allocation of approximately 1.45 billion m^3 (Canterbury Maps, 2017a). However, for the majority of consents, no explicit account has been made of actual use, the consent conditions theoretically allowing the abstraction to occur at the maximum volume for 365 days. Some takes will occur year-long (e.g., public supply) and many others will only occur over a shorter time frame (e.g., irrigation consents) (Aitchison-Earl et al. 2004). In recent years there has been an increase in groundwater abstraction volumes and the development of deeper groundwater resources, with annual groundwater use exceeding annual surface water uptakes. Allocation has reached a stage where Environment Canterbury considers some groundwater zones to be over allocated and where new run-of-river takes are less able to provide reliable supply for irrigation (Figure G-1). In 2010, over 100% of the groundwater in the Rangitata-Orton groundwater allocation zone and 98% in the Orari-Opihi groundwater allocation zone had been allocated (Davey, 2004).

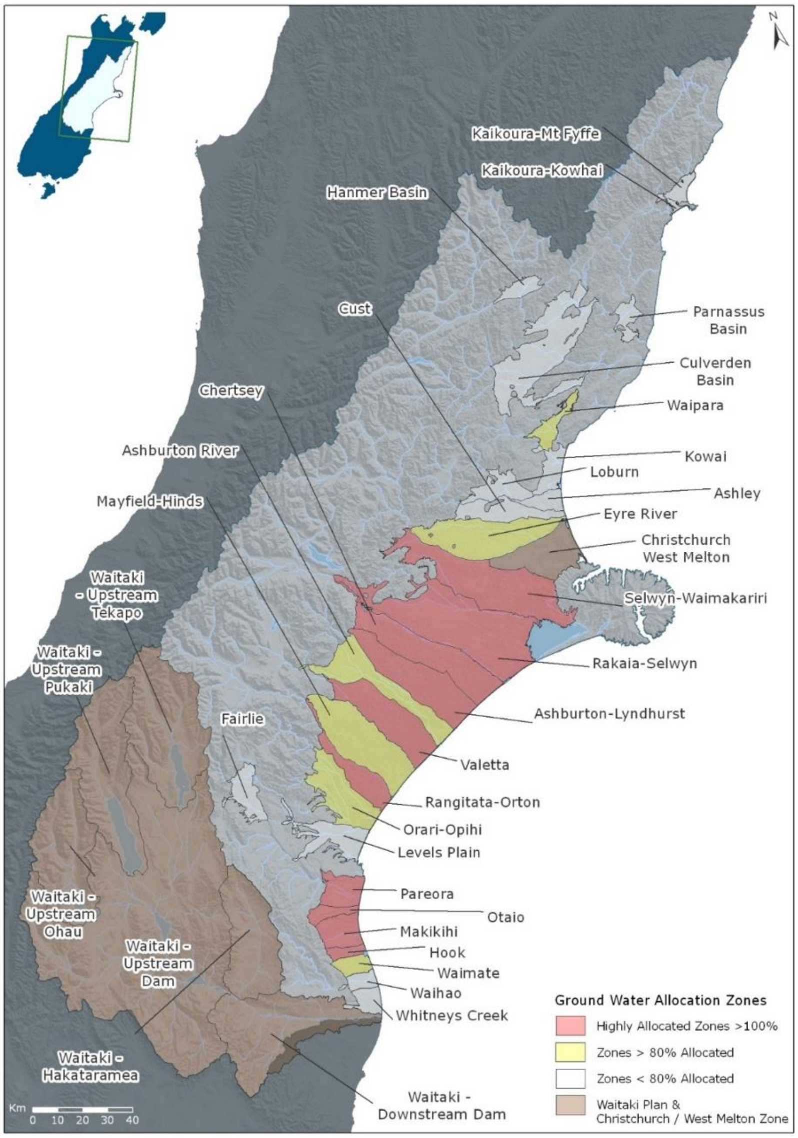


Figure G-1: Groundwater allocation zones and allocation amount in the Canterbury Region (Environment Canterbury (2018)).

The allocation of water for irrigation purposes in Canterbury is primarily the responsibility of Environment Canterbury, however is largely influenced by the Resource Management Act (RMA).

Environment Canterbury's "Canterbury Water Management Strategy", states that "Environment Canterbury is responsible for managing the region's water resource, including the flows and levels in any water body; control of taking, use, damming and diversion of water; the allocation of water and the control of discharges" (Environment Canterbury, 2009). However, as mentioned, a large portion of the Canterbury Plains is already over allocated for groundwater take consents. Therefore, appropriate management of the groundwater resources in Canterbury is very important in order to meet the requirements of the RMA. The Canterbury Water Management Strategy (Canterbury Water, 2009) was developed for the sustainable management and development of the region's water resources. Concerning groundwater allocation, the strategy aims to: address water brokerage, transfer/allocation of consents and charging regimes to encourage reconfiguration of existing consents, and to drive efficiency of water use; reconfiguration of water consents in conjunction with additional water from storage; implement a regional programme that will address rules to ensure water allocation is managed in the public interest, including levies to fund environmental restoration.

Agriculture in the Canterbury aquifer catchment

The aquifer system of the Canterbury Plains is essential to maintaining the agricultural productivity of the land. In 2012, the spatial extension of agriculture in Canterbury covered an area of 2,519,110 ha (Statistics New Zealand, 2012). Over the last two decades, agricultural production in the Canterbury region has grown as a result of the increasing use of inputs, such as fertilisers, supplementary feeds and irrigation water, accompanied by the conversion of plantation forests and areas of extensive sheep and beef grazing into dairy farms (Scott et al. 2011; Lilburne et al. 2013). Dairy farming requires much higher levels of water input when compared to other farming types. Some of these activities lead to increasing loading of nutrients, particularly nitrate, on the land surface, which poses a significant risk of contamination of both surface and groundwater sources within the region. In the 1980s, Bowden et al. (1983) and Burden (1982, 1984) reported that new irrigation schemes and the resultant intensification of land use in the central Canterbury plains could lead to higher nitrate concentrations in groundwater, which in turn could threaten its use as a source of drinking water. To counter this agricultural intensification, Environment Canterbury encourages the use of modern farming management practices to limit the leaching of nutrients to the groundwater environment.

Canterbury’s land cover is characterised by extensive areas of grassland (45%) of which three quarters is exotic grassland. Native tussock grasslands make up 15%. Other herbaceous vegetation (freshwater and saline vegetation, flaxland) make up a very small proportion of land cover (< 0.5%). Urban/bare/lightly vegetated surfaces, which include urban areas, roads and railways, as well as natural bare surfaces, together make up 13% of Canterbury's land cover. Urban areas are a small proportion at 1%. Forest, scrub/shrubland, including exotic and indigenous vegetation combined, make up 19% of Canterbury's land cover. Cropland (arable and horticulture) 6%, and water bodies (lakes or ponds, rivers, and estuaries) comprise the remaining 2%. By 2012 (compared to 1996), the area of grassland, and scrub and shrubland, had decreased whereas the area of urban bare/lightly vegetated surfaces, cropping, forest, and water bodies had increased (LAWA, 2018).

About 70% of all irrigated land in New Zealand is in Canterbury. Agricultural land use in the Canterbury region is supported by a number of irrigation schemes and their infrastructure (Table 1), with a total area of 507,418 ha irrigated (Dark et al. 2017). Canterbury accounted for nearly 60% of the national increase in irrigable land from the year 2007 – 2012 and this has been largely attributed to the increase in dairy farming (Bascand, 2012). Table G-1 outlines the irrigation schemes in the Canterbury region and the consent and total ha serviced by the scheme along with the method of irrigation.

With regard to groundwater aquifers, there are two broad classes of agricultural land-pressures of concern in the Canterbury region, input of land-based contaminants (primarily nitrogen, phosphorus, fine sediment and pathogenic faecal bacteria) and groundwater flow alteration due to agricultural water management.

Table G-1: Irrigation schemes in the Canterbury region (Canterbury Maps, 2017b).

Scheme Name	Consent ha	Border ha	Spray ha	Total ha	Method
Ashburton-Lyndhurst	25000	24500	4500	24592	Border-dyke and Spray
Aviemore	400	0	0	0	Border-dyke and Spray
Balmoral	5500	3657	1643	5300	Border-dyke and Spray
Barhill-Chertsey	40000	0	0	0	Spray
Benmore	8000	0	0	0	Border-dyke and Spray
Cascade	884	0	0	0	Spray
Clayton Road	383	0	0	0	Spray
David Rutherford	650	0	0	0	Border-dyke and Spray
Eiffelton	2290	0	0	0	Spray
Fereday	1700	0	0	0	Border-dyke and Spray
Greenstreet	2347	0	0	0	Border-dyke and Spray
Hassall-Zino	200	0	0	0	Border-dyke
Kakahu	3200	0	0	0	Spray
Levels Plains	5600	739	2386	3125	Border-dyke and Spray
Lower Waitaki	0	10484	5750	16234	Border-dyke and Spray
Lynnford	541	0	0	0	Spray
Maryburn	270	0	0	0	Border-dyke and Spray
Mayfield-Hinds	34000	0	0	32000	Border-dyke and Spray
Mitchells	152	0	0	0	Spray
Morris Road	492	0	0	0	Spray
Moy Flat	147	0	0	0	Spray
Northbank	1319	0	0	0	Border-dyke
Pangborn and Thomas	405	0	0	0	Border-dyke
South Rakaia	1055	0	0	0	Border-dyke
Spencer Bower and Prattley	2201	0	0	0	Border-dyke and Spray
Totara Valley	1400	0	0	0	Spray
Upper Waitaki	1920	0	0	0	Border-dyke
Valetta	7000	3805	496	4301	Border-dyke and Spray
Waiau	14380	9922	4458	14380	Border-dyke and Spray
Waimakariri	18000	500	14500	15000	Border-dyke and Spray
Waireka Downs	419	0	0	419	Border-dyke
Morven Glenavy	-	-	-	-	-
Rangitata South	-	-	-	-	Spray

Groundwater quality

Due to the intensification of agriculture in Canterbury, there is increasing evidence that Canterbury's groundwater is becoming degraded as a result of increasing inputs of nutrients, bacteria and sediment from these changing land uses (Environment Canterbury, 2008). Concern over nitrate concentrations dates back to at least the 1970s, when high nitrate concentrations were reported in the Lincoln area (Adams et al. 1979, Saffigna, 1977). Shallow groundwater, and water recharged to depth by rivers other than the Waimakariri River, irrigation and local precipitation on the unconfined western areas of the Plains, are more susceptible to agricultural and other pollutants (Taylor et al. 1989).

Nutrients

Hanson (2002) reviewed nitrate-N concentrations in Canterbury groundwater using existing data held in Environment Canterbury's water quality database. Concentrations in 942 of the 14,015 samples (6.7%) collected from 2350 wells exceeded the NZDWS of 11.3 mg/L, while almost two-thirds of the samples had concentrations less than 5.65 (one half the NZDWS). Concentrations ranged from below detection limits to a maximum of 89 mg/L (Figure G-2).

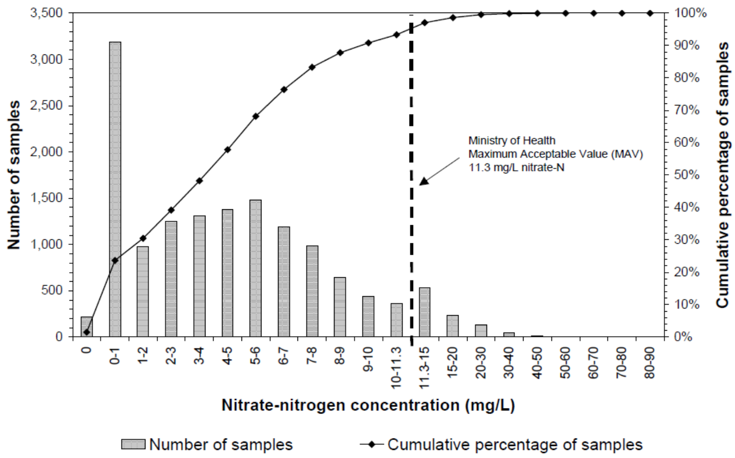
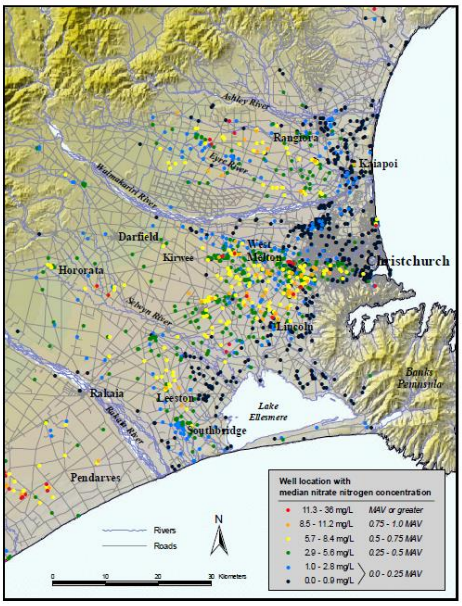
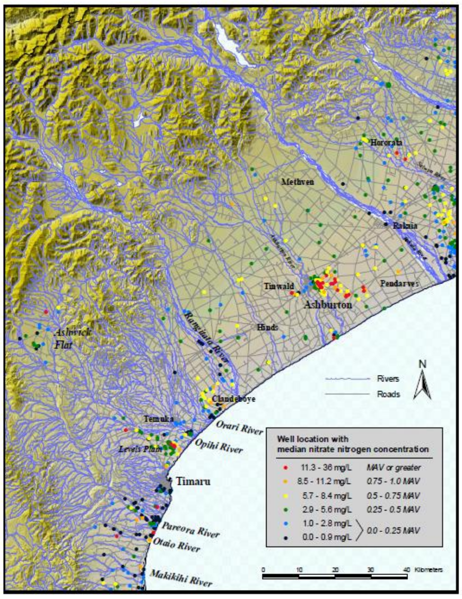


Figure G-2: Nitrate-nitrogen concentrations.

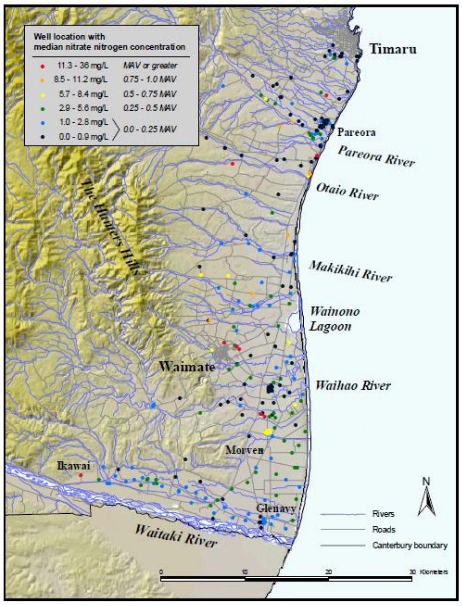
While nitrate-N concentrations varied with time and location, some patterns were able to be identified (Figure G-3). nitrate-N concentrations ranged from below detection limits (0.05-0.1 mg/L) to a maximum of 89 mg/L. nitrate-N concentrations were < 1 mg/L over well-defined areas in the coastal confined aquifers and in areas dominated by recharge from rivers (e.g., north Canterbury Plains, between the Waimakariri and Ashley rivers, near the coast and along Eyre River). Outside of these areas, where groundwater is recharged by soil drainage, concentrations were generally > 3 mg/L, indicating the influence of agriculture and waste disposal activities (e.g., south and west of Christchurch, Lincoln/Burnham/West Melton areas, Spreydon/Sydenham). In shallow, unconfined groundwater not diluted by recharge from surface water, nitrate-N concentrations fluctuated (2-6 mg/L over a single year) on a seasonal cycle, with higher concentrations in winter and spring and lower concentrations in autumn. Long-term increasing trends (median: 0.2 mg/L per year) were identified in 43 wells (e.g., lower half of the plain, and in Waitaki, Ashwick Flat, Culverden and Kaikoura). Of the 43 wells, 21 wells distributed across the plain (e.g., south-western Christchurch, near the Rakaia River and north of the Waimakariri River) showed decreasing trends.



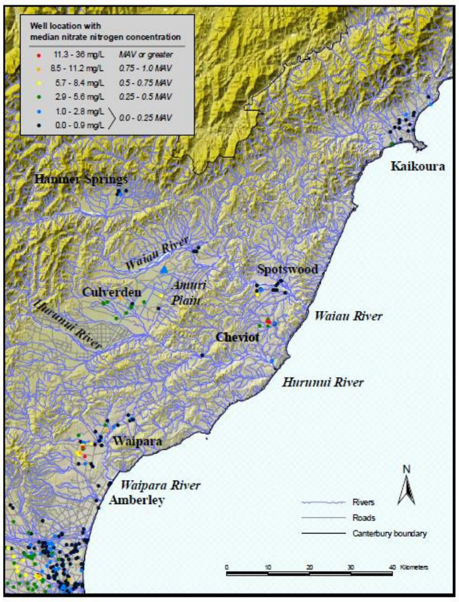
Northern and central Canterbury Plains



Southern Canterbury Plains



Between Timaru and the Waitaki River



Northern Canterbury

Figure G-3: Median nitrate-N concentrations across the Canterbury plains (modified from Hanson, 2002).

In 2004, nitrate-N concentrations were measured in groundwater samples from 121 wells on the Ashburton-Hinds plain, near Tinwald (Hanson and Abraham, 2010). The investigation area encompassed 600 km^2, between the Hinds River and Ashburton River/Hakatere (Figure G-4). As mentioned previously, three irrigation schemes (Valetta, Eifleton and Lynnford) service the area. The plain has a number of groundwater-fed streams and drains, which discharge to coastal waters. There have been some concerns about the concentration of nitrate in spring-fed of coastal streams mid-Canterbury (Meredith et al. 2006).

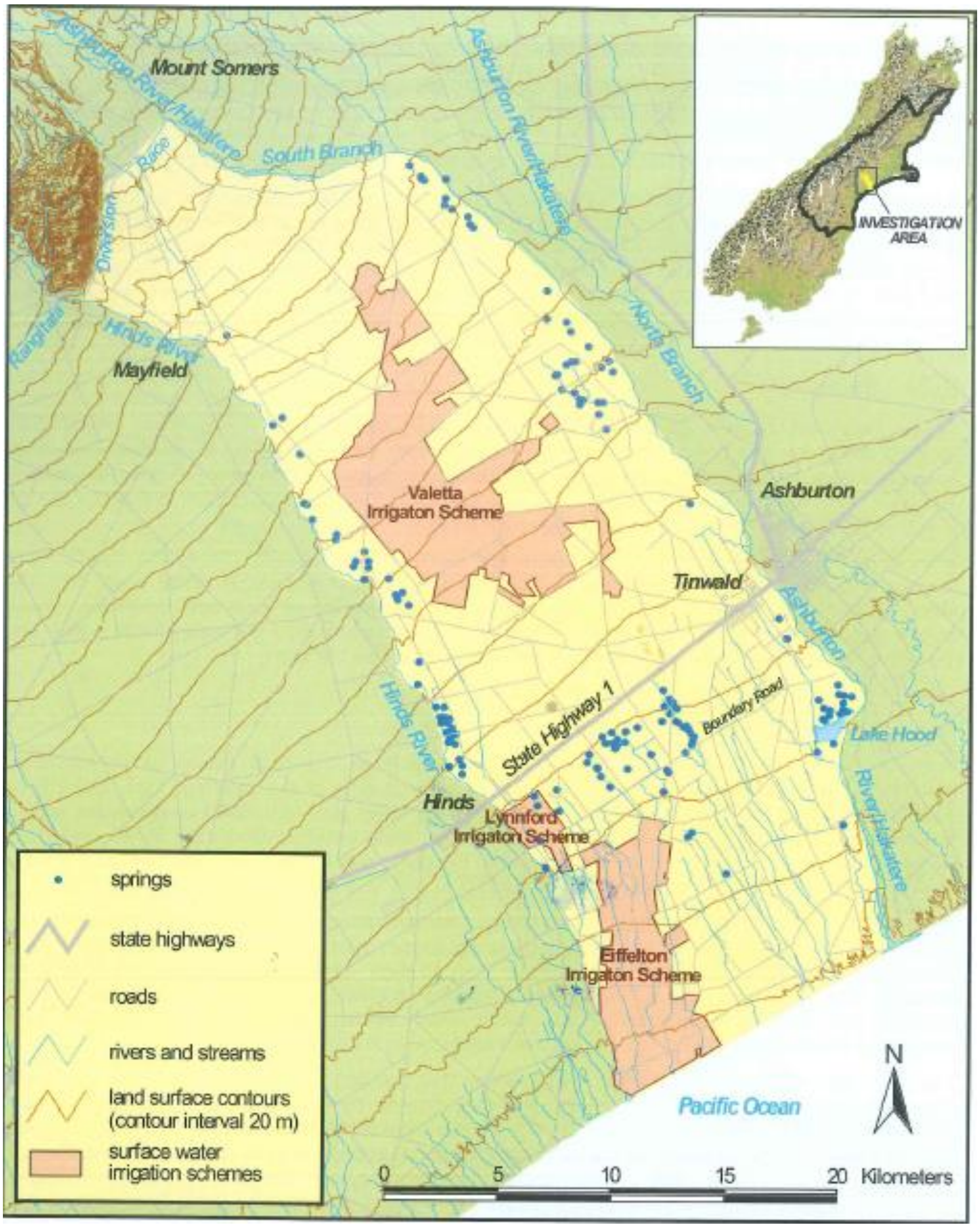


Figure G-4: Ashburton/Hinds plans investigation area, including topography, surface water, drainage, springs and irrigation schemes (Hanson and Abraham, 2010).

Pre-2004, nitrate-N concentrations ranged from 0.6 to 13 mg/L (Abraham and Hanson, 2004), with two wells located in the Tinwald area with observed concentrations above the NZDWS. In 2004, nitrate-N concentrations ranging from 0.1 mg/L (the detection limit) to 22.4 mg/L were observed, with the highest concentrations (> NZDWS of 11.3 mg/L) measured in Tinwald, where there is a transition zone between high-permeability sediments beneath the upper plain and the lower-permeability sediments near the coast (Hanson and Abraham, 2010). Samples from 20 of the 121 wells sampled had concentrations above the NZDWS and samples from wells had concentrations above half the NZDWS. Concentrations above the NZDWS were also measured outside this area (e.g., nitrate-N > 8.4 mg/L in the middle of the plain), with the majority of the wells located southeast of

State Highway 1. In the upper part of the plain, at depths of greater than 60 m below the water table). In addition, there was a distinction between nitrate-N concentrations and whether they were observed within or outside the Valetta irrigation scheme (< 5.6 mg/L and 5.7-8.4 mg/L, respectively). nitrate-N concentrations were lower (2.9-8.4 mg/L). Down-gradient of Tinwald, nitrate-N concentrations reduced to values less 2.9 mg/L, coinciding with an area of reducing conditions within the aquifer (i.e. high iron (> 1mg/L) and manganese (> 0.5 mg/L) concentrations). Since 2004, nitrate-N concentrations in the Ashburton-Hinds plains have generally increased (Hanson and Abraham, 2010, Hanson and Abraham, 2011). In 2009, Environment Canterbury's annual groundwater quality survey included 8 wells, all of which were sampled in 2004. nitrate-N concentrations in 3 wells (K37/0216, K37/0358, K37/0468) showed clear increasing trends from 2004 to 2009 (K37/0216: 3.6-6.3 mg/L to 2.2-12.3 mg/L; K37/0358: 13-14.5 mg/L to 11.5-18.2 mg/L; K37/0468: 7.3 mg/L to 4.2-10 mg/L).

Extensive areas of groundwater with nitrate-N concentrations above the NZDWS have been reported elsewhere in Canterbury, including the area between the Ashburton River/Hakatere and the Rakaia River (Abraham and Hanson, 2004, Hayward and Hanson, 2004) and the area between the Rangitata River and the Orari River (Hanson, 2002, Scott et al. 2011). A survey of groundwater quality in the lower Rangitata-Orari Plain (which extends from the foothills of the Southern Alps in the north-west to the Pacific Ocean in the south-east and is bounded to the north-east and south-west by the Rangitata and Orari Rivers, respectively) was undertaken in summer and autumn of 2008, sampling 32 shallow wells (< 20 m deep) and 10 deep wells (> 20 m deep) between State Highway 1 and the coast in order to determine the extent of groundwater nitrate contamination and to identify the sources of this contamination (Scott et al. 2011). Of the wells that were sampled, 5 of them were part of Environment Canterbury's ongoing monitoring programme, 16 wells were monitored as part of the conditions of Fonterra's discharge consents and some wells were sampled only for this investigation. Up until this time a detailed review of the groundwater in the Rangitata-Orari Plain had not been conducted. As well as pressures of increase abstraction on groundwater volumes within the plain, several land use activities (e.g., discharges of animal effluent and dairy processing wastewaters and leaching of agrichemicals applied to land) has the potential to affect the quality of the groundwater in the Rangitata-Orari Plain. In this study, samples were analysed for nitrate-N and δ15N to investigate possible sources of contamination. Results from the 2008 Rangitata-Orari survey showed: > 5 mg/L nitrate-N in most of the shallow groundwater wells in the lower plain, including areas away from the dairy factory discharges; > 11.3 mg/L nitrate-N in most wells around the dairy factory wastewater discharge areas; < 5 mg/L nitrate-N in most of the deep wells and in some shallow wells where nitrate may be diluted by alpine river recharge; and close to the detection limit (0.1 mg/L nitrate-N) in shallow and deep groundwater from reducing zones near the coast. A higher median concentration of nitrate-N (6.7 mg/L) was observed in shallow groundwater compared to other areas in Canterbury (4.3 mg/L from Environment Canterbury's 2008 annual survey of 302 wells; 4.0 mg/L in wells in the Timaru district (Abraham and Hanson, 2009)) with the exception of some areas near Ashburton and Culverden. Natural tracers for determining possible recharge sources indicated that: recharge from the Rangitata River influences groundwater adjacent to the river; shallow groundwater near the upper and middle reaches of the Orari River and Coopers Creek is influenced by Orari River recharge; and both shallow and deep wells show probable land surface recharge signatures on the Orton Plain (Scott et al. 2011).

A 2006 investigation of groundwater quality in 37 wells in the Culverden Basin (between the townships of Waikari in the south to Waiau in the north), North Canterbury, indicated that ambient groundwater quality in the basin was high with relatively few exceedances of the NSDWS (Abraham and Hanson, 2006). The wells (22 wells < 25 m deep, 15 wells > 25 m deep) were sampled for nitrate-N, ammonia-nitrogen, iron, manganese, boron, major anions and cations, pH and conductivity. nitrate-N concentrations greater than the NZDWS was observed in 2 of the wells sampled (located at Manson Flat, M33/0005: 14.9 mg/L, N33/0206: 21.1 mg/L), with the contamination appearing limited in its areal extent. Samples from an additional 8 wells in the northern part of the basin had nitrate-N concentrations greater than half the NZDWS (5.65 mg/L). nitrate-N concentrations appeared to increase with increasing distance from the Hurunui and Waiau Rivers.

Pesticides

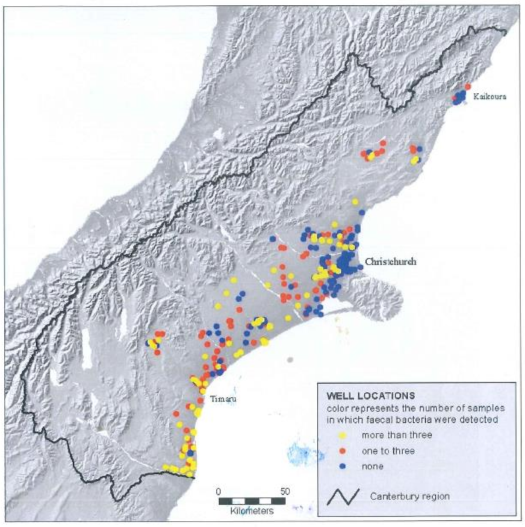
The pesticides atrazine (0.01-0.31 µg/L) and simazine (0.01-0.78 µg/L) have previously been found at low levels in several wells in the South Canterbury area (Levels Plains-Temuka) possibly associated leaching via irrigation combined with low levels of dilution by river water (Smith, 1993). Low levels of simazine (0.06-0.09 µg/L) and terbuthylazine (0.05 µg/L) were observed in the 1994 national pesticide survey (Close, 1996). In the last two national pesticide surveys (Close and Skinner, 2012, Close and Humphries, 2016) there have been no pesticide detections observed in the Canterbury Region.

Microbial contamination

Although much attention has been focused on groundwater contamination by nutrients and organic chemicals, the degradation of groundwater quality in Canterbury by microbial contamination may be more prevalent.

Hanson et al. (2006) reviewed Environment Canterbury's water quality database for bacterial detections (faecal coliforms up until the late 1990s and *E. coli* after that) from 1260 wells over the period 1986 to 2004. The aim was to examine the extent of contamination in the region's groundwater. Most of the wells sampled were privately-owned wells used for domestic, stock-water or irrigation purposes, while some of the wells were community water supply wells owned by city or district councils. The private domestic wells introduced bias toward groundwater known or suspected to be contaminated due to their location near septic tank discharges. Very few of the wells were purpose-built groundwater quality monitoring wells. Faecal bacteria (*faecal coliforms and E. coli*) were detected in 1732 (19%) of the samples in the database, collected from 502 wells (40% of the 1260 wells sampled). Of the 9215 samples that were analysed for faecal bacteria, 8243 were also analysed for total coliforms. Total coliforms were detected in 3426 (42%) of the samples analysed. The samples tested for total coliforms were collected from 1099 wells and 694 (63%) of these wells had at least one detection. Figure G-5 shows the faecal bacteria and total coliforms (387) results, respectively, from wells with recorded depths less than 50 m that were sampled more than three times. 171 out of 415 wells produced no samples with detections for faecal bacteria, with over half of these located within the areas of the Christchurch artesian aquifer system which extends from Lake Ellesmere to the coast north of Christchurch. 103 out of 415 wells produced at least three samples with detections for faecal bacteria; however very few of these wells were located within the Christchurch aquifer area. Concerning total coliforms, only 60 out of 387 wells had no detections and all of these wells but a few were located in the Christchurch aquifer area. The highest bacteria counts (faecal coliforms or *E. coli*) recorded in the database came from samples that were either visibly contaminated or from wells that were located near clear sources of contamination (e.g., septic tank boulder pits, wastewater disposal areas, or refuse pits). These sources are further discussed in the section on the effects of land use on groundwater quality below.

In the 2015 Annual groundwater quality survey (Wong, 2015) *E. coli* was detected in 22 (6.7%) of the wells sampled. This was a higher detection rate than the previous year (12 detection), but still lower than 2011 to 2013 when we detected *E. coli* was detected in more than 10% of the wells sampled. 19 samples with *E. coli* detections came from wells less than 20 m deep, and only 3 samples with detections came from wells deeper than 20 m. E. coli was detected in one well that was 125 m deep; no detections had been observed in this well in the past. The highest *E. coli* count recorded in this survey was 1553 MPN/100 ml, which came from an 8 m deep well located in the Lower Waitaki – South Coastal Canterbury zone, down-gradient of a wastewater treatment plant.

(a)


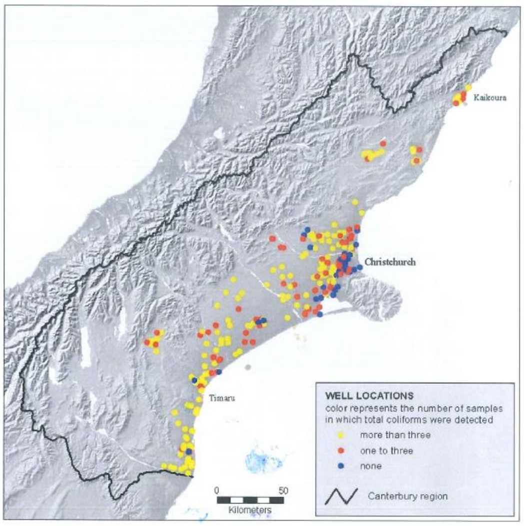
(b)


Figure G-5: Faecal bacteria (a) and total coliform (b) results from 415 and 387 wells (< 50 m depth), respectively, sampled more than three times.

The effects of septic tank effluent discharge on groundwater quality was investigated at Oxford, northern Canterbury by Hughes (1993). Groundwater quality monitoring showed a significant degree of groundwater contamination. Background water samples were free from faecal contamination, and only showed intermittent presence of very low levels of coliform organisms. However, concentrations of total and faecal coliform bacteria in excess of NZDWS (median: <1-172 MPN/100 ml) were detected up to 900 m down-gradient of the nearest septic tank. Elevated levels of chemical indicators (nitrate-N, chloride) were also detected in all monitoring wells, indicating chemical enrichment of the groundwater close to Oxford by septic tank effluent.

Close et al. (2010) presented the results of monthly monitoring (from 2001-2006) for faecal coliforms (540 samples) and Campylobacter (582 samples) in groundwater wells at the Lincoln University Dairy Farm. In the up-gradient wells, 5.4% of samples had detections of faecal coliforms at concentrations of 1 cfu/100 ml or greater. In the down-gradient wells, 2.5% of samples had detections of faecal coliforms at concentrations of 1 cfu/100 ml or greater. Over the five-year period, there were only 0.9% of samples with detection of Campylobacter. For the period between July 2005 and June 2006, Campylobacter was detected in one sample from an up-gradient well and in two samples from down-gradient wells, with the isolates being identified as C. jejuni. These results indicate that there is little microbial contamination coming from the Lincoln University Dairy Farm.

Biological values

Even though 80% of water is in the ground, micro-, meio- and macrofauna represent a relatively unexplored yet increasingly important ecological indicator of health in this system. ESR and NIWA have been examining the abundance and species richness of this fauna in the Canterbury region in order to understand their relationship with the surrounding environment (water chemistry, geology, seasonality). It is hoped with that improved understanding, monitoring and capacity building with others, we can gain a better picture of these organisms' role, including sensitivity to overlying surface water changes which could lag shallower systems, and their importance in preventing subsidence and clogging of aquifers, bioremediation of waste, nutrient cycling and improving underground water quality.

There have been relatively few studies in New Zealand looking at groundwater ecology and ecosystem function, even fewer relating to land use (Boulton et al. 2008, Chilton, 1881,1882, Climo, 1974, Hartland et al. 2011, Scarsbrook et al. 2003, Sinton, 1984, Williamson et al. 2012, Weaver et al. 2015). One study investigated the macroinvertebrate abundance in a sewage polluted aquifer in Canterbury (Sinton, 1984). The study found that there was a significant increase in macroinvertebrate abundance in the contaminant plume derived from sewage irrigation practice. The study concluded the macroinvertebrates were either directly or indirectly (through feeding on biofilms) feeding on organic matter derived from the sewage. It was also noted that there were a high number of dead macroinvertebrates present indicating the sewage irrigation may have a detrimental effect on macroinvertebrate abundance.

Overseas there is also sparse information on the variation of groundwater micro and macrofauna in relation to land use impacts (Danielopol et al. 2003, Korbel et al. 2013, Korbel and Hose, 2015). A study in New South Wales, Australia investigated the impact of land use activities on groundwater fauna (Korbel et al. 2013). The study found that there was a greater macroinvertebrate abundance and diversity at irrigated sites. They also found the diversity of microbial assemblages varied with land use. A further study in the same aquifer further studied the microbial and macroinvertebrate diversity across the alluvial aquifer (Korbel and Hose, 2015). They concluded that macroinvertebrate diversity was affected by groundwater quantity rather than quality whereas microbial diversity did show a difference in response to water quality (amongst other parameters). Danielopol et al. (2003) reviewed the state of groundwater ecosystems and the future prospects with changes in land use in Europe and globally. The review pointed to a number of potential issues related to land use activities and the effect it may have on groundwater ecosystems. One is the effect of land use activities on water quantity and flow. The review suggests that an increase in organic matter loading will lead to increased areas of anoxia and thus would lead to clogging effects as organisms grazing on microbial biofilms would be restricted by the reduction in oxygen. Clogging would lead to changes in water quantity due to changes in flow. The review also points to evidence that increases water takes impact groundwater fauna detrimentally. Stygofauna abundance and richness was greatest at irrigated sites, with the composition of the assemblage suggestive of disturbance. Microbial assemblages and water quality also varied with land use. As mentioned above this area of research is in its infancy still and there is an urgent need for fundamental research both in New Zealand and internationally before we can address any land use impacts.

Effects of land use on groundwater quality

Nutrients

Qualitative relationships between land use and groundwater concentrations of nutrients (with particular emphasis on nitrate) in the Canterbury aquifer have been identified in numerous studies, which are summarised below.

On the Canterbury plains, generally, the highest nitrate-N concentrations in groundwater are found in areas where the general concentrations are below NZDWS but above concentrations that would be considered natural. In shallow wells of the southern Plains area, between Timaru and the Rakaia River, average nitrate-N concentrations are generally in the range of 2 to 8 mg/L. Higher concentrations are found in localised areas, especially in areas where dairy and meat packing industries discharge waste effluent to land (Hanson, 2002). In several of these areas, especially south-west of Christchurch and on either side of the Eyre River, numerous wells have median concentrations above 5.6 mg/L (half the NZDWS). The concentrations in these areas probably reflect the land uses, which include agriculture, lifestyle blocks, and small communities (Hanson et al. 2002). Increasing trends in nitrate-N across the Canterbury plains may be caused either by the accumulation of nitrate in the groundwater from continued land use practices or by changes in land use (e.g., changes to more intensive agricultural activities, increased rates of wastewater effluent application). Seven of the 43 wells reviewed by Hanson (2002) with increasing nitrate trends were resource consent monitoring wells associated with land disposal of effluent from meat processing plants or a dairy factory, and all of these had trends with slopes of 0.5 mg/L or greater. Decreasing trends in nitrate-N may also be caused by changes in land use (e.g., changes to less intensive agriculture, reduction in waste effluent disposal rates). Land use activities in the area surrounding wells with increasing concentrations tend to include intensive agricultural activities like effluent spreading, dairy farming and horticulture (Table G-2). Decreasing trends could also be caused by increased abstraction rates, which would increase the hydraulic gradients around the well and could cause more water to be drawn from areas with lower nitrate concentrations, such as deeper groundwater or nearby rivers (Hanson, 2002). Land uses around wells with decreasing trends include a higher proportion of grazing and residential land (Table G-2).

Table G-2: Land use activities surrounding wells with increasing and decreasing nitrate-N concentrations in groundwater (modified from Hanson, 2002).

Well No.	Well depth	Median nitrate-N (mg/L)	Land use on well property	Land use up-gradient of well
Increasing nitrate-N trend				
J38/0045	24	5.6	Lifestyle block-grazing	Grazing-cattle
K38/0412	4.9	5.3	Farm house/yard-grazing	Grazing-cattle
L36/0200	30.8	4.8	Farm yard-dairy	Dairy
L37/0020	67.66	6.6	Farm yard-dairy	Dairy
L37/0254	70.7	6.6	Farm yard-dairy	Dairy/cropping
L37/0397	45.5	7.8	Meat works/effluent disposal	Meat works/effluent disposal
M36/0456	9	6.3	Farm yard-horses	Dairy
M36/0698	25	1.7	Public supply	Residential/crop
M36/3588	12.2	5.2	Farm yard-dairy	Dairy
M36/3596	9.1	2.8	Farm yard-dairy	Dairy
M37/0065	18.3	2.0	Farm yard-grazing/crop	Cropping
Decreasing nitrate-N trend				
J37/0012	6.7	4.6	Farm yard-sheep	Grazing-sheep
J38/0055	5.5	5.6	Lifestyle block-sheep	Grazing-sheep
K36/0118	11.3	5.2	Farm house-sheep	Grazing-sheep
L37/0422	53.8	23.0	Meat works/effluent disposal	Meat works/effluent disposal
M35/1860	78.5	0.7	Public supply	Industry/residential
M35/1883	28.9	7.8	Yard-meat processing	Industry/residential
M36/5128	12	0.3	Residential	Residential
M36/2528	33.8	7.7	Public supply	Industry/residential

Bidwell et al. (2009) assessed: the effects of existing land-use on the Canterbury Plains on groundwater quality if existing land-uses and land-use practice was to continue in the long-term; the scale and patterns of change in groundwater quality if there was a reduction in nitrate inputs from existing land uses, the scale and patterns of change in groundwater quality from a widespread increase in nitrate (land-use intensification); and the difference in water quality between shallow and deep groundwater, and the role of river recharge in dispersing nitrate concentrations. Modelling of nitrate discharge from land to groundwater is achieved by combining a GIS-based map of land use and a "look-up table" of nitrate discharge rates for each type of land use, soil type and climate zone. The range of land use types corresponded to those available on the land use databases and was supplemented by additional information from the Environment Canterbury consent database and some remote sensing data. For assessment of the deep groundwater quality of Central Canterbury in this study, model predictions of nitrate concentration using AquiferSim were presented as maps of average values for the depth ranges of 0 – 50 m and 100 – 150 m below the groundwater surface. It was found that nitrate discharge from agricultural land use on the Canterbury Plains has the potential to cause levels of nitrate concentration in shallow groundwater (< 20 m below the groundwater surface), at some localities, that exceed the NZDWS (Figure G-6). There are many locations where the map legend indicates that predicted values of nitrate concentration are close to or above 12 mg/L, which means that this recharge water is at or above the drinking water limit. In addition, groundwater quality was observed to improve with depth below the groundwater surface due to dispersive mixing with high quality groundwater from river recharge.

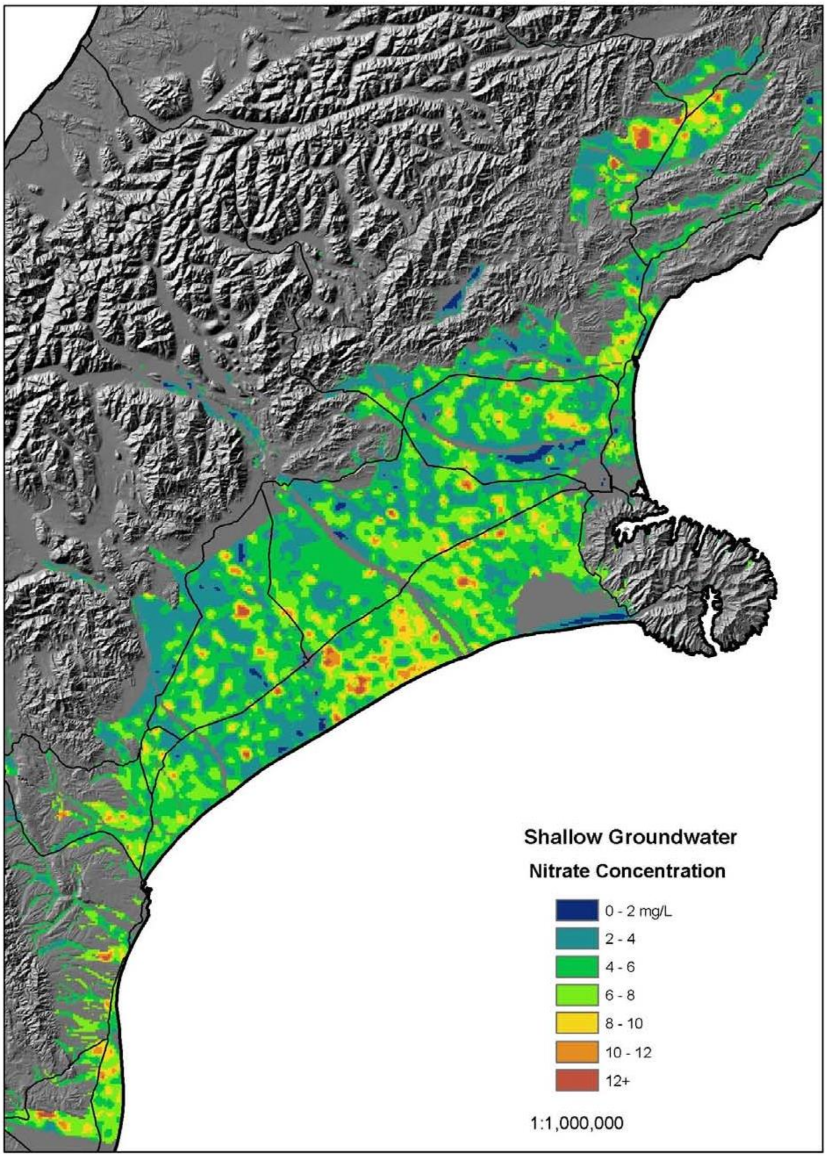


Figure G-6: Model prediction of nitrate concentration in shallow groundwater for Canterbury Plains from agricultural use on the Canterbury Plains (Bidwell et al. 2009).

Bidwell et al. (2009) modelled the likely effect of improved practices for reduction of nitrate discharge from agricultural land (Figure G-7). This map was produced by applying a 20% reduction to the nitrate concentration component of the nitrate discharge data in Figure G-6. This shows a visually detectable change in shallow groundwater quantity, in general, and a decrease in localities where access to drinking water quality might be an issue. However, this approach is approximate because improved practices would involve changes to both concentration and soil-water drainage.

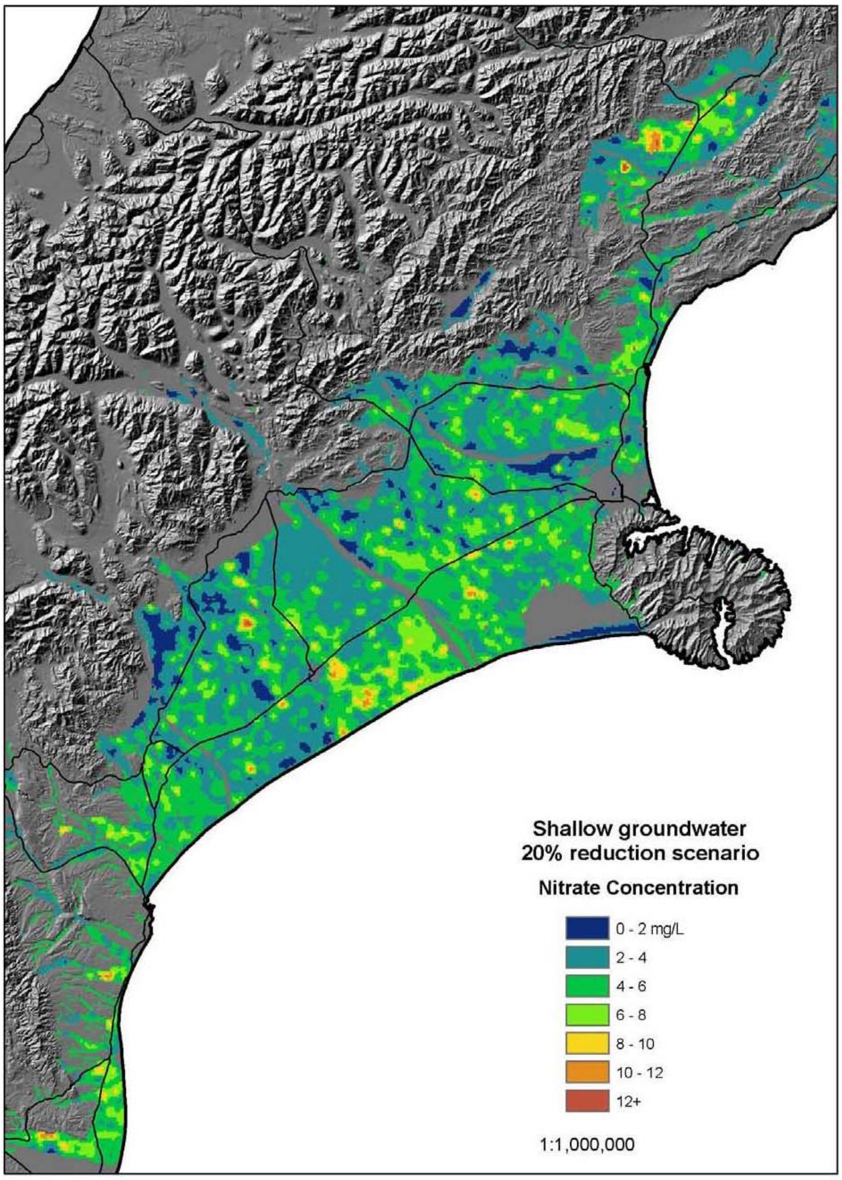


Figure G-7: Model prediction of nitrate concentration in shallow groundwater, for Canterbury Plains. Model prediction of nitrate concentration in shallow groundwater, for Canterbury Plains, by applying a 20% reduction in the nitrate concentration of the nitrate discharges in Figure H-6 (Bidwell et al. 2009).

Environment Canterbury collected 1922 records for dissolved reactive phosphorus (DRP) concentrations in groundwater samples from 918 wells across the region between 1995 and 2014. Scott and Wong (2016) analysed the data to characterise the state, distribution and trends in DRP concentrations. They also used well construction details, spatial data on soil properties and land use and other groundwater chemistry data to look for factors that could be controlling the trends in data. While the distribution of DRP concentrations in groundwater was similar to that published in national DRP statistics, the proportion of wells with higher DRP concentrations was slightly lower in Canterbury. The median concentration of DPR in groundwater in Canterbury was 0.007 mg/L (1995-2014) compared to 0.01 mg/L (1995-2008) in all of New Zealand groundwater. The number of wells with a median > 0.009 mg/L in Canterbury was 44% (402 wells) compared to 54% (383 wells) New Zealand wide. The number of wells with a median > 0.03 mg/L in Canterbury was 13% (123 wells) compared to 28% (200 wells) New Zealand wide. Wells with higher DRP concentrations were mostly located in the Hurunui-Waiau, Orari-Opihi-Pareroa and Lower Waitaki-South Coastal Canterbury zones under pastoral farming, lifestyle blocks and horticulture rather than under arable cropping or forested land. While there are some areas of intensive agriculture within these zones which could be the source of DRP, other zones with intensive agriculture (for example, Ashburton) were found to have relatively low DRP concentrations in the groundwater. Land use information from the AgriBase™ dataset was used to look for possible links between land use and groundwater DRP concentrations. It was found that land use may have an effect on median DRP only if the samples were not oxic. Oxic samples, which made up the bulk of the groundwater data, had very similar median DRP concentrations across all the land use classes. The outlier and extreme values occur more frequently in the land use classes where livestock, or septic tanks on lifestyle blocks, may be present. When only anoxic samples were considered, there did appear to be some enrichment of DRP in groundwater under land uses involving grazing livestock (sheep/beef/deer, dairy and horses/goats/pigs). However, with very few anoxic groundwater samples below arable cropping, support grazing and forestry, there was not enough data to draw strong conclusions about the impact of these land uses. In addition, data for forest and natural vegetation in all redox classes was limited to only a few wells where these land classes occur. Geological sources of phosphorus (for example, peat and older marine deposits), and its contribution to DRP in groundwater, were found North of Christchurch and south of Timaru along the coast and below rolling hill country.

The main source of high nitrate-N concentrations in groundwater observed in the Ashburton-Hinds plain area in 2004 was determined to be due to diffuse nitrate leaching from agricultural land (Hanson and Abraham, 2010). Farming is the main land use on the Ashburton-Hinds plain, with a mixture of pastoral farming (sheep, beef, and/or deer), dairying, and cropping (Figure 7). Since the 1990s there has been a trend of intensification in farming with many farms converted to irrigated dairy (Engelbrecht, 2005). Cropping is common in the coastal area and in the areas along the river margins above State Highway 1, particularly along the southwest side of the Ashburton River/Hakatere. Most farms within the Valetta irrigation scheme area involve intensive dairy livestock production. Outside of the Valetta scheme is dominated by sheep and other pastoral farming, with a trend of conversion to more intensive farming practices (Hanson and Abraham, 2010). Agricultural activities are the most obvious source of nitrate-N concentrations in the Ashburton-Hinds plain, but there are also potential sources of localised contamination. It was postulated that cropping and winter fallow activities, on the south bank of the Ashburton River/Hakatere northwest of Tinwald (Figure 8), on soils with low nitrate uptake contributed to the high concentrations in the Tinwald area. Fallow land is estimated to have low nitrate uptake that could result in high nitrate leaching rates (Bidwell et al. 2003). The dilution of nitrate in soil drainage water caused by flood irrigation in the Valetta irrigation scheme area (See Figure G-4) accounted for the lower concentrations outside the Tinwald area. Potential point sources included a fertiliser store and stock sale yards in Tinwald (nitrate-N concentrations up to 18 mg/L observed in a nearby well) as well as a truck wash on Frasers Road, across from Tinwald golf club.

Most of the contamination in the area between the Ashburton River/Hakatere and the Rakaia River and the area between the Rangitata River and the Orari River is probably caused by diffuse leaching from agricultural land. In the Rangitata-Orari Plain, dairy farming and seasonal grazing occupy most of the land south of State Highway 1, along with minor sheep and beef farming and arable cropping. In the upper part of the plain, north of State Highway 1 along the Rangitata and Orari Rivers. The foothills are dominated by deer farming and forestry (Scott et al. 2011). Irrigation and conversion from dryland agriculture to dairying has intensified in this area in recent years.

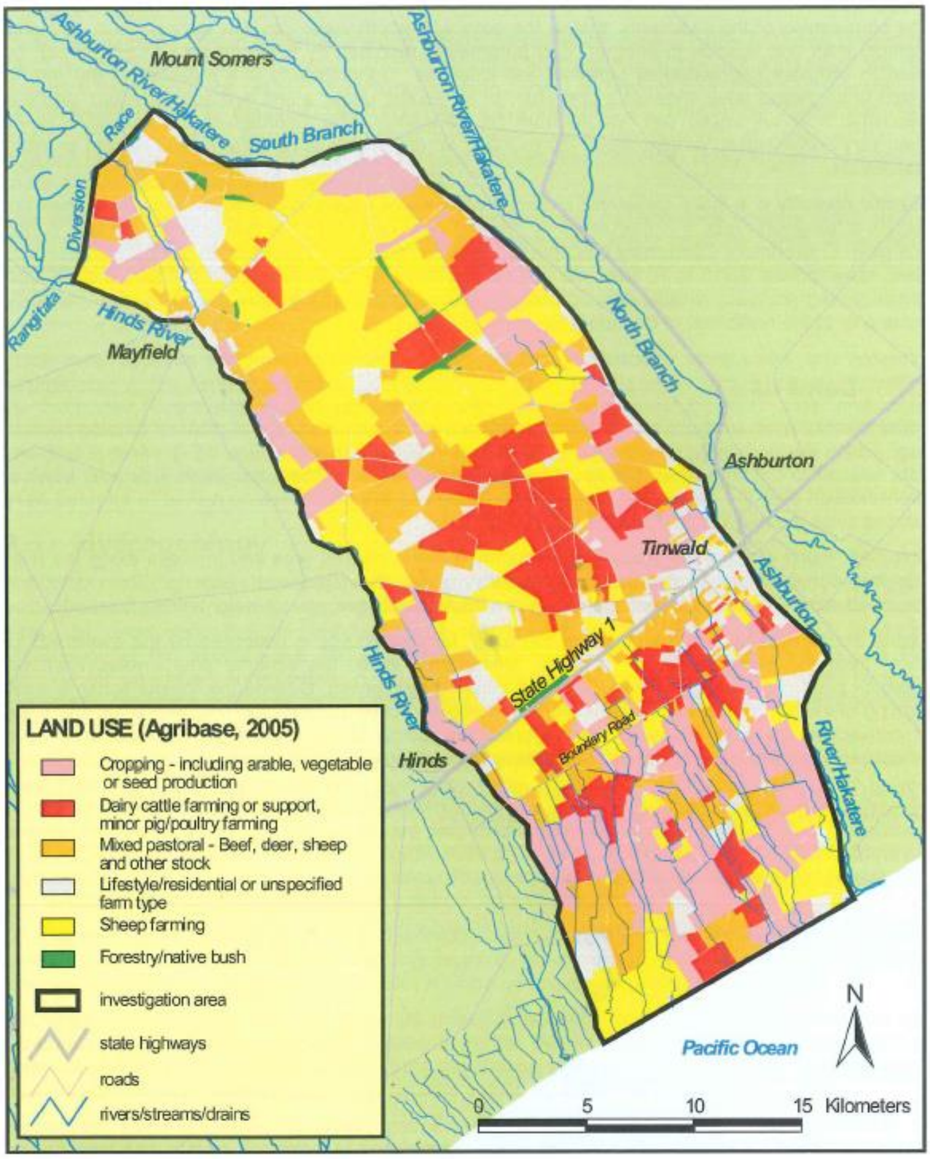


Figure G-8: Land use on the Ashburton-Hinds plain (Agribase, 2005 in Hanson and Abraham, 2010).

However, the highest nitrate-N concentrations in these areas appear to be associated with specific point sources, particularly wastewater disposal from meat processing plants or dairy factories (Hanson, 2002, Abraham and Hanson, 2004, Hayward and Hanson, 2004, Scott et al. 2011). In the lower Rangitata-Orari plain, a large dairy processing factory at Clandeboye is the most prominent industry in the area. The factory holds various discharge consents for the disposal of wastewater (up to 26,750 m^3/d) to land across much of the lower Rangitata-Orari Plain (1400 ha) that has an effect on the quality of the groundwater. Some areas around the dairy factory have been irrigated with wastewater for more than 2 decades. Of the wells surveyed in the 2008 Rangitata-Orari survey (Scott et al. 2011), 3 of the wells with the highest nitrate-N concentrations were located down-gradient from the dairy factory wastewater irrigation area where there is likely to be little dilution of contaminant introduced by wastewater irrigation of leached from soils by land surface recharge in this area. The increasing trends in nitrate-N concentrations in wells near the dairy factory around the year 2000 was postulated to be due to the occurrence or lactose processing at the factory and the subsequent decrease in dissolved organic carbon and biological oxygen demand of the wastewater limiting the extent of denitrification in the groundwater (Justin O'Brien, Environmental Officer, Fonterra, pers. comm. in Scott et al. 2011).

Because of the strong contrast between calcium dominance of natural groundwater and sodium dominance of wastewater and sewage effluent discharges, the ratio of sodium to calcium ions (Na/Ca) in groundwater tested in the 2009 survey was also used to delineate a zone of influence of the dairy factory wastewater and sewage effluent irrigation and showed that wastewater and sewage effluent irrigation was the major source of nitrate-N contamination in the most affected areas (Scott et al. 2011).

From the analysis of $\delta^{15}N$ for nitrate in groundwater in the 2008 Rangitata-Orari survey it was concluded that (Scott et al. 2011):

- four of the samples had probably animal origin (manure or sewage effluent) for the nitrate (3 of these wells are down-gradient from the sewage effluent irrigation area and their nitrate concentrations have decreased by denitrification, the fourth well is near the dairy factory wastewater irrigation area but may also have another localised source of nitrate contamination;
- most samples lie within the overlapping ranges of $\delta^{15}N$ for synthetic fertilisers and nitrate derived from soil organic nitrogen and the origin of the nitrate could not be defined with this tool.

Other consented discharges in the Rangitata-Orari Plain include by-wash water from the Rangitata Water irrigation scheme, which is discharged into boulder holes near the Rangitata River, a solid waste discharge consent for the closed Timaru District Council landfill at Peel Forest and an industrial stormwater discharge consent for a grain storage shed at Orton (Scott et al. 2011). Stormwater, which contains metals, from a saw mill and lumber yard in Arundel is also discharged to land under a resource consent, however groundwater sampling from a nearby well has found concentrations of metals (e.g., copper and boron) below the maximum limit set out in the resource consent (2 mg/L) (Scott et al. 2011).

Widespread cropping up-gradient of the Chertsey-Dorie-Rakaia area, between the Ashburton River/Hakatere and the Rakaia River may be the source of high nitrate-N concentrations in groundwater within this area (Hayward and Hanson, 2004, Abraham and Hanson, 2004). North-west of the town of Rakaia there are several wells with nitrate-N concentrations close to or in exceedance of 8.5 mg/L (Abraham and Hanson, 2004). The source of nitrate contamination in the Ashburton district is attributed to agricultural land use, compounded in some areas by large industrial discharges to land (Abraham and Hanson, 2004, Hayward and Hanson, 2004). The disposal of effluent from a meat processing plant north-east of Ashburton has created plumes of nitrate contamination in groundwater down-gradient of the effluent disposal sites (Hayward and Hanson, 2004).

In the 2006 Culverden Basin groundwater quality survey (Abraham and Hanson, 2006), elevated nitrate-N concentrations in some of the wells were postulated to be due to border strip irrigation. Prior to the 1980s, the Culverden Basin was primarily dominated by dry land farming (80% sheep grazing, 10% cattle grazing, 10% cropping; Tonkin and Taylor, 1985). With the construction of major border dyke irrigation schemes, the availability of irrigation water led to rapid land use intensification and the conversion of dry land farming to dairy farming in areas irrigated by the Balmoral and Waiau irrigation schemes. Outside of these areas farming including grazing of sheep, beef and deer. On the north bank of the Hurunui River is the Balmoral Forest, a 10,000 ha pine plantation. Mixed cropping occurs along the river margins and there are a number of vineyards on the south bank of the Hurunui River. There are about 133 consented discharges to land in the Culverden basin with 85% involving discharge of effluent from dairy, piggery and human activities (Abraham and Hanson, 2006). Pre-irrigation the spatial variability in mean nitrate-N concentrations was high (2.0-10.7 mg/L) in shallow groundwater (Close and Woods, 1986, Close, 1987). Average nitrate-N concentrations increased in the pre-irrigation period and decreased in the post-irrigation period. These trends were attributed to increased leaching due to disturbance of the topsoil during land contouring and a wetter year before commencement of irrigation, followed by excessive drainage after the irrigation scheme began. In a well (N33/0206) located in the west of the basin, nitrate-N concentrations have increased from 4.6 mg/L in 1996 to 24.7 mg/L in 2005 (Abraham and Hanson, 2006). While the source of the high nitrate-N concentrations is not clear for this well, land use in the area is dominated by intensive dairying and border dyke irrigation. In recent years there has been a change from border strip irrigation to spray irrigation in the Culverden basin that will reduce drainage amounts and may lead to higher concentrations of nitrate in the shallow groundwater.

Lilburne et al. (2013) estimated nitrate-N leaching rates under rural land uses in Canterbury and compared the nitrate-N load for non-pastoral land uses and pastoral land uses. A regional GIS land use map developed by Environment Canterbury staff with assistance from Landcare Research and a nitrate-N leaching rate lookup table using nitrate-N discharge rates from Lilburne et al. (2010) were used to model (using OVERSEER®6, SCION SWatbal, LUCI and SPASMO), at a regional scale, the potential changes to water quality of changing agricultural land uses for the Canterbury Water Management Strategy (Bidwell et al. 2009). It is important to note that modelling was undertaken to develop a comprehensive and robust set of values due to inadequate information on nitrate-N leaching across a range of farm systems, climate and soil types. Leaching was modelled under lifestyle blocks, turf grass-golf courses, outdoor pigs, arable farming and pastoral farming and an estimate of leaching under forestry was also provided. The results from the LUCI and SPASMO modelling were used for the non-pastoral land uses (i.e. arable, lifestyle blocks, berry and pip fruit, grapes). The SCION SWatbal results were used for exotic and native forestry. To ensure a consistent set of inputs for the modelling, the Canterbury Region was divided into four coastal rainfall zones (650 mm/y, 750 mm/y, 850 mm/y and 950 mm/y) and two inland rainfall zones (550 mm/y, 900 mm/y). The region’s soils were grouped into seven categories, according to their profile available water storage and drainage characteristics. Figure G-9 compares the nitrate losses between agricultural sectors.

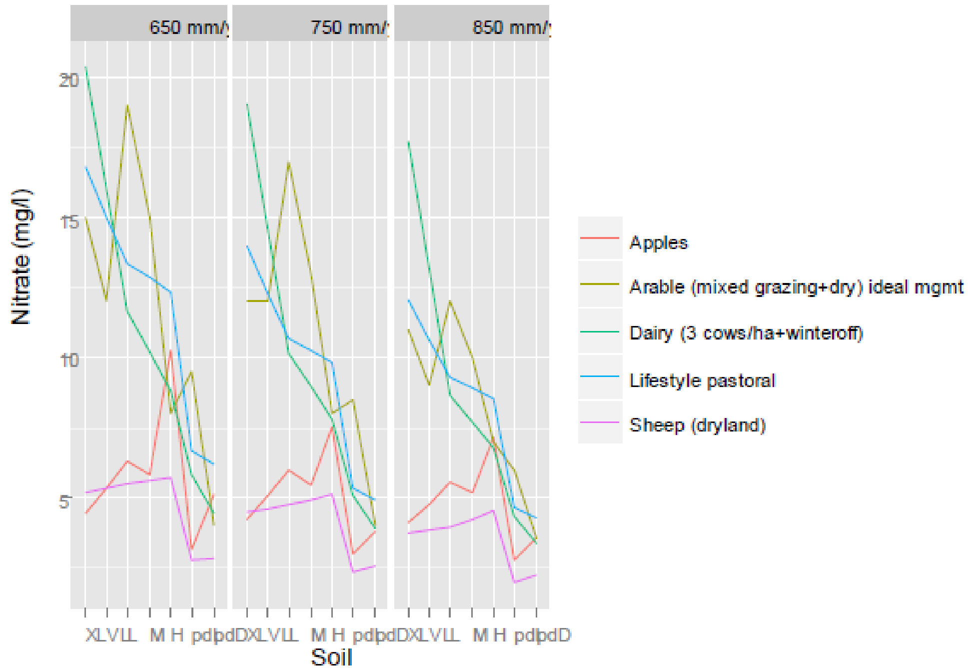


Figure G-9: nitrate-N losses (concentration) between agricultural sectors by soil type and climate zone (650 mm/y, 750 mm/y and 850 mm/y) (Lilburne et al. 2013).

Close and Abraham (2012) examined nitrate data (1990-2010 period) from wells in Canterbury to determine whether such data can be used to provide estimates of inputs from different land uses. Land use information for an area immediately up-gradient of a well was obtained and the nitrate data summarised for each land use category. Criteria for the land use being > 50% and > 75%, together with the average depth of water between the well screen and the water table being < 12 m and < 18 m, were used to assess the variability of nitrate concentrations for each land use. nitrate data was compared to model-based estimates of leaching for different land uses as reported by Lilburne et al. (2010). Mean nitrate concentrations under cropping ranged from 7.1 to 9.7 mg/L, which fell within the range given by Lilburne et al. (2010). Mean nitrate concentrations under dairying ranged from 10.4 to 11.1 mg/L, which was lower than values estimated by Lilburne et al. (2010). Mean nitrate concentrations under sheep ranged from 4.2 to 6.7 mg/L, similar to the mean value of 6.3 mg/L estimated by Lilburne et al. (2010). As the mean for the 20-year period (1990-2010) may not reflect land uses that have changed recently, wells with the most restrictive data were re-assessed using the most recent two years of data from each well. Three of the land uses showed increases in mean nitrate concentrations with dairying showing the largest increase (1.97 mg/L), which was attributed to intensification in dairying over the 15 years prior to the study, and sheep and cropping showing increases of ~ 1.3 mg/L, which was attributed to improvements in irrigation efficiency and a subsequent increase in nitrate leaching to groundwater.

Spring-fed streams draining the lower and coastal parts of the Canterbury Plains are nutrient rich was excessive nitrate concentrations (Meredith and Hayward, 2002). Median nitrate concentrations in coastal streams draining the plains between the Rangitata and Rakaia Rivers are >5.6 mg/L (Meredith et al. 2006). Groundwater with high nitrate concentrations, as a result of irrigation and land use activities further up the plains, is likely to be the principal source of elevated nitrate-N concentrations (Meredith et al. 2006).

Microbial contamination

The study by Hanson et al. (2006) determined the source of contamination for those wells where faecal bacteria was greater than 2000 MPN/100 ml (Table G-3). The highest bacteria count recorded in the Environment Canterbury database was a total coliform count of 24,000,000, which was detected in well M36/4517 in 1994. This well was driven through a septic tank boulder hole, so it may reflect septic tank effluent rather than groundwater. This same sample had a faecal coliform count of 4,600,000, which was also the highest faecal coliform count in the database. However, three earlier samples from this well in 1992-93 had no bacteria detections apart from one sample with a total coliform count of 7. Most of the wells with detections for *E. coli* > 2,400 MPN/100 ml were located near septic tanks, under heavily grazed or irrigated agricultural land or near oxidation ponds. Poor well-head protection of these wells was determined to be the cause of most of the contamination observed, with bacterial detections halted after well-head protection was improved.

Table G-3: Summary of wells where faecal bacteria counts were >2000 MPN/100 ml (modified from Hanson et al. 2006).

Well #	Samples	Detections	Highest count	Source
M36/4517	4	1	4,600,000	Septic tank boulder hole
L37/0602	2	1	> 2,400,000	Storage tank
L37/0391	28	14	1,300,000	Oxidation pond for feedlot effluent
M35/0733	1	1	360,000	Construction of oxidation ponds for municipal wastewater
K39/0020	1	1	> 200,000	Septic tank
L37/0915	45	22	> 200,000	Disposal of meatworks effluent
M36/3747	13	11	29,000	Refuse pit
J40/0042	75	65	> 20,000	Grazing animals on heavily stocked irrigated paddock; nearby land use includes sheep, dairy, cattle, deer and cropping
L37/0393	24	11	12,000	Feedlot effluent
M36/3867	6	6	6,700	Refuse pit
L37/1051	9	1	5,200	Feedlot effluent
J39/0082	1	1	2,700	Horse paddock
J39/0232	58	31	> 2,400	Heavily irrigated golf course, also adjacent sheep farming and lifestyle blocks; dead animals sometimes found in well
J40/0343	79	37	> 2,400	Grazed irrigated paddocks on dairy farm at bottom of moderate to intensively farmed catchment, old landfill up-gradient
J40/0163	101	66	3,400	Border-strip irrigation runoff from sheep farm, birds use nearby grain silo as food source and faecal matter has been observed on top of well

As a result of rapid increases in urban and high density rural development, effects of domestic wastewater disposal by land irrigation on groundwater quality at two sites in Burnham and

Templeton, on the central Canterbury Plains was investigated by Martin and Noonan (1977). Land irrigation schemes have been operating at Templeton and Burnham for many years. At the time, the Templeton sewage treatment plant discharged an average of 550 m^3 of effluent onto a 25 ha border-dyked disposal area. The source of sewage for the Burnham scheme is the Burnham Military Camp, which had an average of 1300 m^3 of treated sewage discharged daily onto a 23 ha border-dyked disposal area. Groundwater chemistry analysis in bored at Templeton indicated higher chemical concentrations down-gradient of the disposal area, suggesting that the irrigation scheme is causing contamination of the aquifer system. Analysis of groundwater at both sites suggested that some chemical and microbiological contamination was occurring as a consequence of land irrigation of domestic sewage. Little microbial contamination was found in the wells down-gradient from the border-dyke area at Burnham (during periods of no or low rainfall) unless irrigation of effluent was being carried out. Large numbers of faecal coliform bacteria were found in the groundwater following heavy rainfalls. In addition, during a period of time when effluent was being irrigated, there was a marked increase in the numbers of faecal coliform bacteria. In contrast to the Burnham results, microorganisms were found in a number of bores at Templeton even when there was no heavy rainfall suggesting that contamination of groundwater was occurring during effluent irrigation. Irrigation of border-dyke strips lying on the groundwater flow line through observation bores at Burnham showed that microorganisms could move a distance >900 m at a rate of ~150 m/d.

Sinton (1982) carried out a groundwater quality survey at Yaldhurst, west of Christchurch. The study area consisted of 120 households each served by an individual septic tank system and a domestic well. A baseline groundwater quality survey was undertaken. Approximately 33% of the wells contained coliform bacteria, faecal coliform bacteria or faecal streptococci. A subset of 25 wells was sampled fortnightly from January to August 1977. Results showed low nitrate-N levels throughout the test period. Of the 25 wells tested, 23 wells exhibited intermittent bacterial contamination. Although no correlation was observed between septic tank proximity and microbial contamination, soakage pits were suspected to be the main source of the microbial contamination.

Sinton (1986) investigated the effects of two methods of septic tank effluent disposal (on pasture land adjacent to a sewage plant treating domestic effluent of a military camp) on the microbial quality of alluvial gravel aquifers at Burnham, in the Canterbury Plains. Septic tank effluent (1000L/d) is discharged into either a 5.5 m deep soakage pit or an 18 m deep injection bore. A tracer bacterium, *E. coli* PB 922, was injected into the effluent sampling riser during the 2nd discharge pulse of the day. Effluent discharge from the soakage pit into the unconfined groundwater and from the injection bore into the confined groundwater was monitored using arrays of shallow and deep bores. The movement of faecal coliform bacteria 9 m from the soakage pit into an unconfined aquifer, and 42 m from the injection bore into a confined aquifer was observed. Faecal coliform bacteria was detected in all the shallow bores, suggesting the radial spread of soakage pit leachate into the unconfined groundwater. Faecal coliform bacteria was also found in all the deep bores. Diurnal fluctuations in faecal coliform concentrations were correlated with periods of effluent discharge.

Pang et al. (2006) modelled the impact of clustered disposal systems on nitrate and faecal coliforms in groundwater in a rural community near Christchurch. The model included nine disposal boulder pits, situated 4 m below the surface in alluvial gravel media, in a domain of 3.3 km by 30 m (including both unsaturated and saturated zones). Water movement between the ground surface and the disposal pits was simulated using HYDRUS-1D. The performance of the two-dimensional model was evaluated using monitoring data obtained from a 1977 study. Both observed and simulated results showed that multiple clustered disposal systems have a significant cumulative impact on nitrate concentrations in groundwater, but the impact of faecal coliforms from individual systems was localised.

In a study by Sinton et al. (2005), three microbial tracers (*Escherichia coli* J6-2, a somatic coliphage (ØESR1) and endospores of *Bacillus subtilis* var. *niger* NCIB 8649 tracer strain JHI) were added to effluent flood irrigated onto border dyke strips at Templeton sewage treatment and flood irrigation scheme, 10 km south of Christchurch. All three tracers, and three effluent indicators (e.g., faecal coliforms, F-RNA phages, and chloride) were observed ~ 10 hr after irrigation in a bore (24 m deep, screened over lower 12 m), approximately 100 m downstream. Reductions in microbial concentrations were ≈100 times greater than for chloride, and occurred rapidly, suggesting that up to 99% of the microorganisms underwent early exclusion from macropore flow and were removed. However, results showed that substantial numbers of bacteria and viruses could potentially reach the groundwater through macropores beneath effluent irrigation schemes located on alluvial gravel formations. This provides useful information on horizontal microbial transport in groundwater down-gradient of effluent irrigation schemes in alluvial gravel outwash areas.

In the study by Close et al. (2006), similar rates of detection of faecal coliforms and *Campylobacter* (slightly lower in the down-gradient wells) between the up- and down-gradient wells on Lincoln University Dairy Farm, with all levels being 3 cfu/100 ml or less after the first year of monitoring. These results indicated little, if any, impact of dairying with spray irrigation at the current irrigation rate on microbial quality of groundwater. It was thought that most faecal coliforms and *Campylobacter* died off during percolation before they were able to reach the water table. All of the detections of faecal coliforms and 2 of the 3 detections of *Campylobacter* were attributed to heavy rainfall events in May and June 2006.

From this case study, it can be seen that the significant increase in groundwater use and changes in land use in the Canterbury Region appears to have detrimental effect on groundwater quality of the aquifers. For example, groundwater allocation has reached a stage some groundwater zones (e.g., Rangitata-Orton groundwater allocation zone) to be over allocated. Agricultural intensification and the conversion of plantation forests and areas of extensive sheep and beef grazing into dairy farms (which requires greater amounts of water) has led to increasing use of inputs, such as fertilisers, supplementary feeds and irrigation water. This has subsequently led to increased leaching of nutrients and bacteria, as evidenced by increasing trends in nitrate concentrations and detections of bacteria in groundwater.

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Appendix H Firth of Thames

Summary

- The Firth of Thames is an estuarine embayment occupying the Hauraki Depression between the Coromandel and Hunua Ranges. Prior to human habitation, land cover in the Hauraki Plains was mainly native forest. The Plains have been cleared of native forest and are currently dominated by agricultural land-use.
- The intertidal flats in the southern Firth have been accreting sediment over the last 90 years. Present-day suspended-sediment input from the Waihou and Piako rivers represents only about 40% of the estimated ~430,000 t y^{-1} of sediment depositing in the southern Firth. The apparent discrepancy between present day sediment delivery by rivers and the annual sedimentation in the Firth could be due to the reworking by waves and currents of legacy sediments deposited in the Firth during large-scale deforestation and mining activities of the 1800’s and early 1900’s.
- The historical deposition of fine muds may have impaired re-establishment of bivalve and suspension-feeder reefs that were once present in the Firth. The loss of hard reefs is likely to have reduced the Firth’s biotic resilience and filtration capability. Since fine sediments in the Firth are largely the legacy of past human activities, there are limited opportunities to mitigate sediment effects.
- Almost 90% of dissolved inorganic N (DIN) loading to the Firth is from rivers draining its catchment, with the remainder originating from the Hauraki Gulf offshore. Nutrient inputs make the Firth more respiratory than productive. The Firth is sensitive to excessive nutrients because its flushing rate is low, which promotes phytoplankton growth and sedimentation. It stratifies seasonally in its deeper parts leading to depleted oxygen. Its high respiration renders it susceptible to reduced oxygen and lowered pH (acidification).
- Water quality monitoring in the Firth over the last 15 years shows large increases in DIN and dissolved organic nitrogen (DON). N loading from the catchment have not increased enough to explain the increased DIN and DON levels. A hypothesis is proposed that N loss (denitrification) in the Firth has instead decreased.
- Judging by its loading from catchments and historical changes in land-use, the Firth has probably changed from an N-poor oligotrophic system to a moderately enriched mesotrophic system.

Introduction

Managing the Firth of Thames region sustainably depends on a well-developed ability to understand and predict the drivers of its ecosystem services. These services are intimately tied to the health of the region’s ecosystem components. The 2017 Hauraki Gulf State of Environment Report (Kelly et al. 2017) presented situation analyses and status of key indicators of ecosystem health, including fisheries, toxic chemicals, sediment and benthic health, mangroves, nutrient loading, microbial contamination, invasive species and biodiversity. The case study presented here focuses on the Firth benthic and pelagic ecosystems, with sections on geomorphology and sedimentation, benthic and pelagic nutrient dynamics, phytoplankton dynamics and ecology, and the status of ecosystem stressors associated with oxygen depletion and catchment-driven ocean acidification. It considers the sensitivity of these benthic and pelagic indicators to land-use and other factors affecting the region. Key primary sources of this information are Swales et al. (2015), Zeldis et al. (2015) and Zeldis and Swaney (2018).

Physical setting

The Firth of Thames, including its northern approaches to the latitude of Waiheke Island, is an 1100 km2 (Figure H-1) mesotidal estuarine embayment occupying the Hauraki Depression, bounded to the east and west by the Coromandel and Hunua Ranges, respectively (Swales et al. 2015). It progressively shoals from a maximum depth of 40 m at its northern inlet near Waiheke Island to the extensive intertidal flats of the southern Firth.

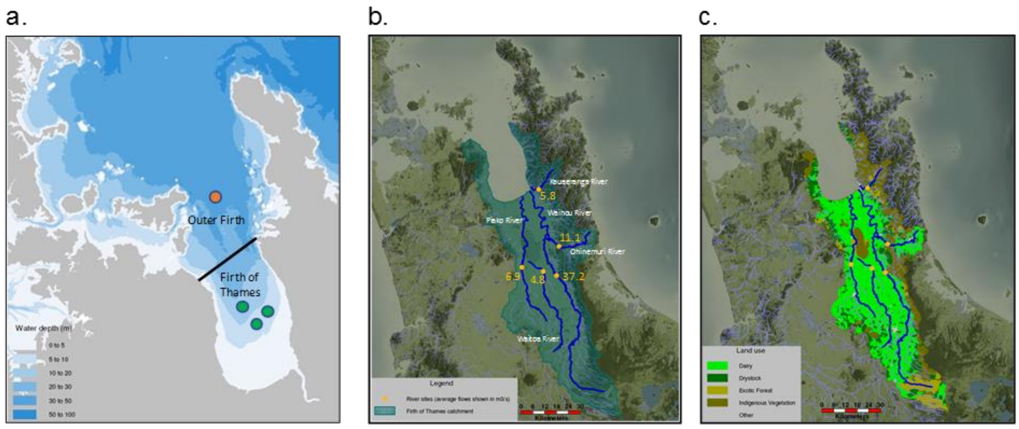


Figure H-1: (a): Location of Firth of Thames, with NIWA Firth of Thames monitoring site (orange dot) and 'inner Firth' sampling sites (green dots) referenced in this report (Zeldis et al. 2015). (b) Firth catchment and major rivers (with mean flows in m3/s). (c) major catchment land uses.

Geomorphology and sedimentation

Prior to human habitation (>1000 y bp) the landcover consisted of podocarp–hardwood forests on the ranges and freshwater marshes and kahikatea occupied the Hauraki Plains (McGlone et al. 1984, Newnham et al. 1995). Large-scale deforestation began shortly after European settlement in the mid 1800's (Brownell 2004). Along with gold-mining, deforestation substantially increased sediment loads to the Firth. Sediment-cores from the southern Firth indicate that formerly sandy intertidal flats prior to the early 1900's today are buried under ~2 m of mud. In the southern Firth, intertidal flats have been accreting for the last 90 years at rates an order of magnitude greater (~25 mm y-1) than most studied North Island estuaries. In addition, mangrove forests have been prograding. Muddy sediments have been accumulating adjacent to the mangroves (up to 1 km from the mangrove fringe) and on intertidal flats at similar high rates (~25 mm yr-1 since 1970s) (Swales et al. 2015).

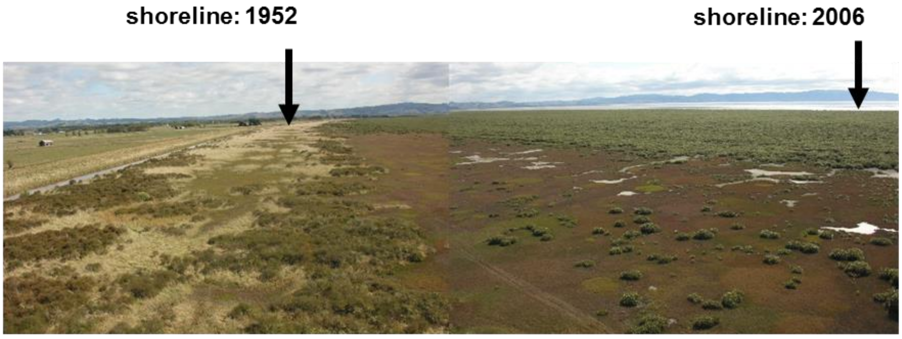


Figure H-2: View of southern Firth of Thames shoreline looking west, showing progradation over time. Source (Swales et al. 2007).

A sediment budget for mangrove forest, intertidal flat, lower intertidal and shallow subtidal sub-environments of the southern Firth has been assembled (Swales et al. 2015, Zeldis et al. 2015). Sediment accumulation rates (SARs) indicate that ~110,000 t y^{-1} of fine sediment is accumulating in the mangrove forest and a similar amount in the upper intertidal flat immediately adjacent. By comparison, an estimated 200,000 t y^{-1} of fine sediment is accumulating on the 210 km^2 of lower intertidal – shallow subtidal zone of the southern Firth, south of Tararu–Kaiaua at rates of 2 mm y^{-1}. This rate is similar to data from the outer Firth. This SAR value is less than half the rate indicated by the sediment-mass deposition near the peak of the catchment deforestation and mining activities in the late 1800s to early 1900s. The present subtidal sedimentation rates (~2 mm/yr) have accelerated substantially over the last 100–150 years, associated with land-use practices (see sediment budget).

The present-day annual suspended-sediment loads of 160,000 t y^{-1} and 30,000 t y^{-1} for the Waihou and Piako Rivers, respectively are estimated (Hicks et al. 2011), representing only about 40% of sediment depositing in the southern Firth. A sediment budget shows that about half of this is occurring in the upper-intertidal flat/mangrove-forest complex, which alone more than accounts for all of annual river load and is accumulating fine sediments at an order of magnitude higher rate than in the southern Firth as a whole. The apparent discrepancy between sediment delivery by rivers and the annual sedimentation in the Firth could be due to the reworking by waves and currents of legacy sediments deposited in the Firth during large-scale deforestation and mining activities during the late 1800s to early 1900s. It is suggested that the intertidal flats were largely built by muds reworked by waves from the shallow subtidal zone and transported onshore and deposited since at least the 1920s. This mechanism also explains the relatively recent development of the mangrove forest since the 1950s (Swales et al. 2015).

Numerical modelling (Figure H-3) and satellite remotely sensed data (not shown) on total suspended sediment (TSS) show greatest concentrations in the inner Firth of Thames, before rapidly dropping to very low levels seaward. Other numerical modelling showed that both maximum suspended sediment concentration (SSC: mg/l) and final sediment deposition footprint (mm deposited) were largest for the Waihou River, followed by the Piako and the Kauaeranga Rivers (Zeldis et al. 2015). All three rivers deposited sediment in the southern Firth, close to their respective mouths. Even large discharges under strong winds (40 km h^{-1}, i.e., storms) resulted in very little sediment escape from the entrance of the Firth to the wider Hauraki Gulf, with SSC greater than 5–10 mg/l not extending beyond the northern entrance to the Firth.

Inner Firth subtidal sediments are very muddy (70-100% mud) and become sandier further seaward (Figure H-3) (Zeldis et al. 2015). However, their carbon contents (2.2%) are only slightly higher than other documented North Island estuaries (~1.8%) (Pratt et al. 2014). The% carbon levels are within the range of eutrophic estuaries reported overseas. Intense microbial remineralisation of terrestrial organic matter is indicated, with the preferential replacement of this material by marine sources (Sikes et al. 2009, Uhle et al. 2007). There appears to have been a reduction in the coarser size fractions (gravel and sand) in the outer Firth over 1999–2003, with a concomitant increase in the proportion of mud. This has been matched by increases in the organic carbon and total organic matter content and a decrease in the molar C:N ratio (i.e., less degraded organic material) in the surficial sediments through to 2012 (Zeldis et al. 2015).

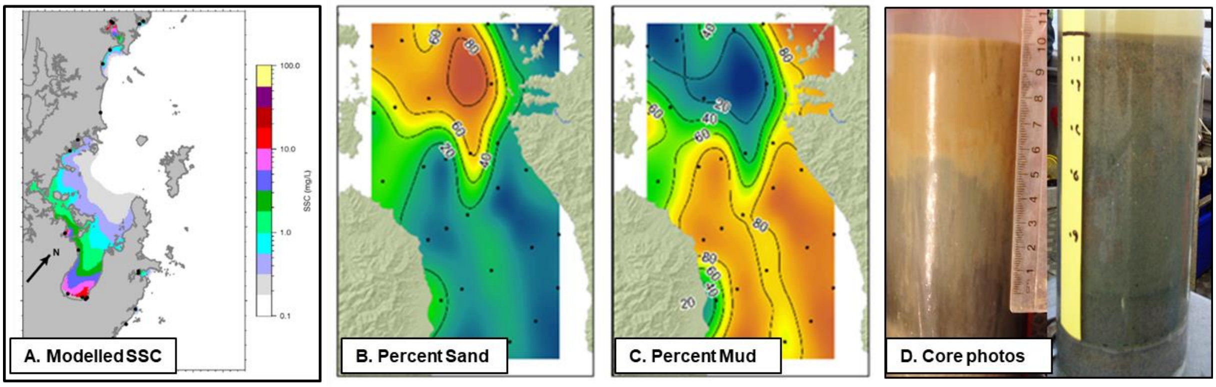


Figure H-3: (a) Modelled spatial distribution of fine sediment in Hauraki region showing dominant inputs from Firth of Thames rivers. Shown is fine suspended sediment concentration (SSC) in the upper water column, averaged over a two-year simulation (M. Hadfield, NIWA, unpubl. data). (b and c) Surface sediment properties sampled during NIWA RV Kaharoa voyage SEA0201 (December 2002) in the Firth of Thames, showing relative proportions of sandy and muddy sediments, respectively. Station locations are shown by black dots. Data from Zeldis et al. (2015). (d) Photographs comparing sediment cores from inner (left) and outer (right) Firth of Thames sites, collected in May 2012 (Clearwater and Depree, NIWA, unpublished data).

The shallower inshore area (<7 m depth) supports benthic micro-algal (BMA) populations that exhibit low chl-a:phaeopigment ratios relative to other North Island estuaries (Zeldis et al. 2015). This may be a consequence of their high mud contents which affect biogeochemical processes and ecological function. There is limited information on subtidal benthic faunal communities in the Firth and the wider Hauraki Gulf. Benthic infauna communities in the southern Firth have similar abundance and biomass to those in the outer Firth, but with peaks in most parameters occurring either on the margins or across the central Firth. Meiofauna biomass peaks occur near the central Firth in association with some of the biogeochemical parameters (e.g., bacteria, % N) and with a transition from the muddy, inner to sandy, outer Firth environments. There may also be localised increases associated with the mussel farms on the western and eastern edges of the central Firth. The inner, subtidal Firth is depauperate in macrofauna and infauna, relative to the central Firth. This is likely to be associated with the progressive muddying of this environment (see below).

Thresholds associated with sediments

Effects of suspended sediment concentration (SSC) and sediment deposition were reviewed using New Zealand and overseas research and used to predict effects of sediment loading from rivers emptying into the Firth. In the present day and over short time-scales, ecological responses to elevated SSC and sediment deposition in the inner Firth are likely to be relatively small, or negligible for the following reasons.

- Because the intertidal and shallow subtidal areas in the southern Firth of Thames are already extremely muddy (60-70% mud), the benthic macrofaunal community is likely to be dominated by a limited number of mud-tolerant species. The large SSC-sensitive suspension-feeding taxa (sponges, ascidians, horse mussels, green lipped mussels, pipis, cockles) are not likely to be common. With the baseline already shifted by historical fluvial inputs, the response of the system to high SSC may now be relatively small.
- The most pronounced impacts of sediment loading are generally associated with sudden, thick deposition events, which can severely affect benthic macrofaunal communities and biophysical processes. However, uncalibrated Deltares modelling (Zeldis et al. 2015) predicted deposits <1 mm thick under the worst-case scenarios. This thickness is sufficient to induce behavioural responses in some species, but it will not have large effects on macrofaunal abundance, richness or community composition unless deposits of this thickness occur with a relatively high frequency (2–3 times per month over the course of 6 or more months).
- Ongoing monitoring of the intertidal southern Firth by the Waikato Regional Council of benthic macrofauna and surficial mud content has shown little evidence of ecologically significant changes over the past 10 years of monitoring. However, these studies would not have resolved impacts occurring prior to the monitoring.

A more pervasive concern from deposition of fine muds in the Firth operating over the long-term could be impaired re-establishment of bivalve and suspension-feeder reefs that were once present in the Firth until over-fishing in the 1960s (Zeldis et al. 2015). The loss of hard reefs, due to historical sedimentation, overfishing and damage from trawling gear, is likely to have reduced the overall biotic resilience and filtration capability of the Firth. Negative feedbacks from this loss of ecosystem service includes heightened sedimentation and a shift into an alternative, less desirable, stable state which sediments are reinforcing.

Reversibilty and remediation of sediments effects

It was concluded (Zeldis et al. 2015) that over the period of human habitation, but especially during the last century, land use in and around the Firth has impacted its ecology and functioning. Surficial sediments in the southern Firth are considerably muddier than prior to European land clearance, resulting in decreased water column clarity and accentuated by the historic loss of shellfish biophysical services from over-harvesting in the 1960s. This could reflect the earlier loss of highly sensitive species (large, structure-forming, suspension-feeding benthic organisms). The state of the Firth may be relatively stable now, and somewhat inured to new inputs of sediments. It is likely that the ability of the benthic ecosystem to return to its former state (i.e., with large, structure-forming, suspension-feeding benthic organisms) is being impeded by negative feedbacks associated with its current state. It is likely that any recovery would not follow a linear path back to original conditions but would display non-linearity associated with biological feedbacks involved in recolonization and reef-building. Reef recovery could result in positive feedbacks with improved sediment health.

Nutrient metabolism and dynamics

Nutrient supply is a vital element of coastal marine ecosystem services. Nutrients supplied by ocean mixing, river inputs and internal recycling fertilise primary production by phytoplankton at the base of the food chain, supporting natural ecosystems, fisheries and aquaculture. Assimilation of nutrients by coastal waters is another critical ecosystem service, by ameliorating effects of excessive nutrient loading from land which can otherwise lead to the environmental hazards of eutrophication (MacDiarmid et al. 2013).

The delivery of nutrients to New Zealand coastal waters has increased dramatically in post-colonial New Zealand (Snelder et al. 2017, Zeldis et al. 2010), exacerbated by erosion, deforestation and land-use intensification (see Sediment section). Primary biomass stimulated by inorganic nutrient loading, and organic matter loaded directly to the system, is eventually respired, depleting oxygen and liberating carbon dioxide into the water, leading to lowered oxygen (hypoxia) and pH (acidification) (Paerl 2006, Sunda and Cai 2012, Sutula 2011). Thus, the cycles of nutrients, oxygen and carbon are intertwined, and understanding their relationships is important for managing potentially serious hazards to coastal ecosystem health. Their relationships can be summarised as:

$$CO_2 + \textit{inorganic nutrients} + \textit{water} \leftrightarrow \textit{organic matter} + O_2.$$

From left to right, the uptake of carbon dioxide and nutrients forms organic matter and evolves oxygen via photosynthesis (autotrophy). From right to left, organic matter is respired, consuming oxygen, reforming dissolved nutrients and producing carbon dioxide (heterotrophy). This balance of production and respiration (p-r) defines the ecosystem's Net Ecosystem Metabolism, or NEM (Caffrey 2004, Gordon et al. 1996, Zeldis and Swaney 2018). The relationship describes fluxes of dissolved inorganic carbon (DIC) and O_2, which control ecosystem stressors of decreased pH and deoxygenation (Cai et al. 2017, Kemp et al. 2005, Sunda and Cai 2012, Waldbusser and Salisbury 2014).

Nitrogen (N) is particularly significant, because it is usually the limiting nutrient for biological production in coastal waters and often sets the rates of these processes (Howarth and Marino 2006 , Vitousek and Howarth 1991). Eutrophication is ameliorated by denitrification, the release of gaseous nitrogen (N_2) out of the system by microbial processes operating in coastal marine sediments (Cornwell et al. 1999, Seitzinger 1988). Deleterious synergistic effects may occur if near-seabed waters become hypoxic or the sediments become excessively enriched with organic matter, removing appropriate sediment conditions supporting denitrification (Kemp et al. 2005, Boynton and Kemp 2008, Eyre and Ferguson 2009, Hale et al. 2016).

In the Firth, seasonally-resolved mass-balance budgets of water, salt and nutrients were used to describe Hauraki Gulf and Firth system nutrient dynamics including NEM and denitrification, using Land-Ocean Interactions in the Coastal Zone (LOICZ) methodology (Zeldis et al. 2015; Zeldis and Swaney 2018). This was done for two years, in budgets separated by 12 years (2000-01 and 2012-13). Key results were:

- Annual mean water residence time (turnover) of the Firth was 21 days in 2000-01 and 24 days in 2012-13.
- Firth catchments contributed 78-85% of total dissolved inorganic N (DIN) inputs to the total Firth N load, with the remainder coming from offshore. The catchment inorganic N loads (river plus atmospheric inputs) were dominated by river inputs (>90%) (Zeldis

and Swaney 2018). This was contrasted with other LOICZ budgeting done for Golden and Tasman Bays, whose catchments contributed only about 20% of their totals (Zeldis and Swaney 2018).

- Firth primary production of organic matter was often slower than its respiration, so Firth NEM was often negative (net-heterotrophic). It was most heterotrophic in autumn and winter. In comparison, the Hauraki Gulf offshore of the Firth was nearly balanced between net-autotrophy and heterotrophy. Approximately balanced results were also obtained for Tasman and Golden Bays (Zeldis and Swaney 2018, Zeldis 2008) (Figure H-4).
- Large percentages of total export of N from the Firth were via denitrification (means of 79% in 2000-01 and 48% in 2012-13), indicating the critical importance of this ecosystem service. Estimated mean denitrification in 2012-13 was, however, only 42% of the rate in 2000-01 (Figure H-4).[28]
- The values of NEM for the Firth placed it among the more heterotrophic values derived from ca. 150 budgets in the global LOICZ budget database, while the net denitrification values were near the mode of values in the database (Figure H-4).

[28] This percentage was calculated using only spring, summer and autumn seasonal surveys, common to both the 2000-01 and 2012-13 surveys (Zeldis and Swaney 2018).

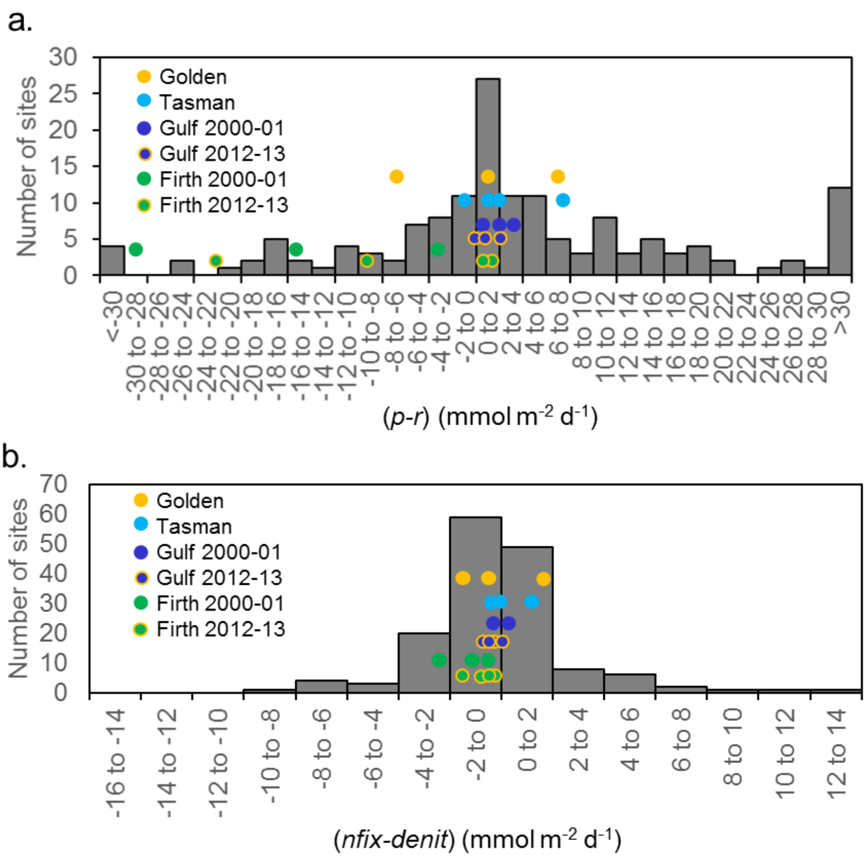


Figure H-4: Histograms. of (a) 150 values of the difference of system production and respiration (p-r) and (b) 155 values of system net denitrification (difference of N fixation and denitrification (nfix-denit)) from a meta-dataset of LOICZ budgets (Smith et al. 2010). The histograms are overlaid with seasonal (p-r) and (nfix-denit) values for Golden and Tasman Bays, Hauraki Gulf and Firth of Thames from budgets of those systems. Golden and Tasman Bays and Hauraki Gulf had relatively neutral (p-r), showing that their Net Ecosystem Metabolism was nearly balanced, whereas (p-r) for Firth of Thames was frequently strongly net-heterotrophic. The net denitrification (nfix-denit) rates for all 4 bays were near the mode of the meta-dataset (Zeldis and Swaney 2018). The LOICZ metadata had upper and lower 5th percentiles removed.

Studies of benthic nutrient, oxygen and carbon fluxes involved in NEM were made in the Firth, using direct shipboard experiments. Strong increasing gradients of benthic Sediment Community Oxygen Consumption (SCOC) were found from the northeast shelf, into the Gulf and thence into the Firth (Giles et al. 2007). This showed that benthic mineralisation of organic matter is most active inshore – where nutrient loading is highest. The increasing SCOC correlated positively with particulate organic C and N, sediment chlorophyll-*a* (chl-*a*), phaeopigments and silt/clay fraction (i.e., increased toward and through the Firth). Within the Firth, SCOC rates measured in 2003 were highest at the shallowest inshore sites (locations in Figure H-1), a result repeated with even higher values in 2012 (Zeldis et al. 2015).

Benthic micro-algae (BMA) were more abundant in the shallow inner Firth than outer Firth, reflecting a higher benthic light environment inshore. Irrespective, all sites had net O_2 consumption by the sediments (Zeldis et al. 2015), consistent with the mass-balance budget findings of net-heterotrophy (see above). Sediment heterotrophy at the inner Firth was twice that at the outer Firth. Ammonium fluxes from the sediments were 2-4 times higher at the inner Firth than the outer Firth and were relatively unaffected by BMA abundance. Ammonium porewater concentrations at inner Firth sites were about 3 times greater than at outer sites. Rates of SCOC at the outer Firth site suggested a 1.5-1.7-fold increase in oxygen consumption rates between 2003 and 2010 (Zeldis et al. 2015). Although this comparison was compromised by differences in sampling seasons (summer vs autumn), it was consistent with increased muddiness of sediments between 2003 and 2012 (Sediments section).

Based on published relationships between denitrification efficiency (amount denitrified / total N flux) and CO_2 efflux rates (assumed equal and opposite to SCOC (Cook et al. 2004, Eyre and Ferguson 2009), data from the 2012 Firth benthic SCOC surveys showed that outer Firth sites were within the range at which denitrification efficiency was optimal (~-800 µmol O_2 $m^{-2}h^{-1}$). At the inner Firth, with higher SCOC (up to -1800 µmol O_2 $m^{-2}h^{-1}$), efficiency was predicted to drop to around 50%. The mechanism by which this can occur was summarised by Sutula (2011) and Boynton and Kemp (2008). Coupled denitrification in estuarine sediments involves a finely balanced relationship between aerobic and anaerobic bacterial communities in close proximity in surficial sediments. Estuarine denitrification can therefore be limited by conditions of organic enrichment of sediments and low bottom water oxygen. Resulting low redox conditions and shallow penetration of oxygen into sediments inhibit nitrification, and consequently denitrification. Under these conditions, more of the organic N deposited to sediments is returned to the water column as ammonium rather than N_2 (Kemp et al. 1990).

The role of the pelagic zone in NEM was assessed using water column oxygen evolution and uptake experiments in shipboard incubations at the inner and outer Firth sites during autumn (March) 2012 (Zeldis et al. 2015). Key results were:

- All incubations drew down O_2, whether in light or dark, indicating strong net-heterotrophy.
- Net O_2 draw-down (heterotrophy) in incubations was greater at the inner sites than the outer site by about 2-fold. Although draw-down was slower at outer Firth site, the deeper water column there sustained particularly low O_2 because it was isolated from the atmosphere by vertical water column physical stratification (see Oxygen and pH section).
- Gross O_2 production in light was ~2 x greater at inner than outer sites, consistent with 2-fold greater mean chl-*a* at inner sites, and indicating a more productive inner Firth phytoplankton community than in the outer Firth.
- Water column-integrated values of daily net O_2 consumption were compared with the benthic daily net O_2 consumptions in co-incident sediment core incubations, at inner and outer sites. Ratios of water column-to-sediment respiration were 50:50 at inner sites and 90:10 at outer sites. Thus, water column respiration equalled or exceeded benthic NEM over most of the Firth, a finding in common with estuarine systems overseas.

Nutrient concentrations have been sampled at the NIWA Firth monitoring site (Figure 1) approximately every 3 months over the 15 years between September 1998 to July 2013 (excepting July 2001 to December 2002), to determine trajectories and drivers of change of Firth water quality (Zeldis et al. 2015, Zeldis and Swaney 2018) (Figure H-5). Time-series analyses showed that dissolved inorganic N (DIN sum of ammonium and nitrate) increased by about 5% y^{-1} (Table H-1) from 1998-2013.

Dissolved organic nitrogen (DON) also increased. In contrast, dissolved inorganic phosphorus (DIP) did not change significantly and dissolved organic phosphorus (DOP) decreased. The ratio of DIN/DIP increased strongly (4% y^{-1}) indicating a significant enrichment of N in the system relative to P and that a long-term (decadal-scale) shift in the nitrogen environment had occurred. This was supported by the similar changes in phytoplankton and bacteria (Figure H-5 and next section).

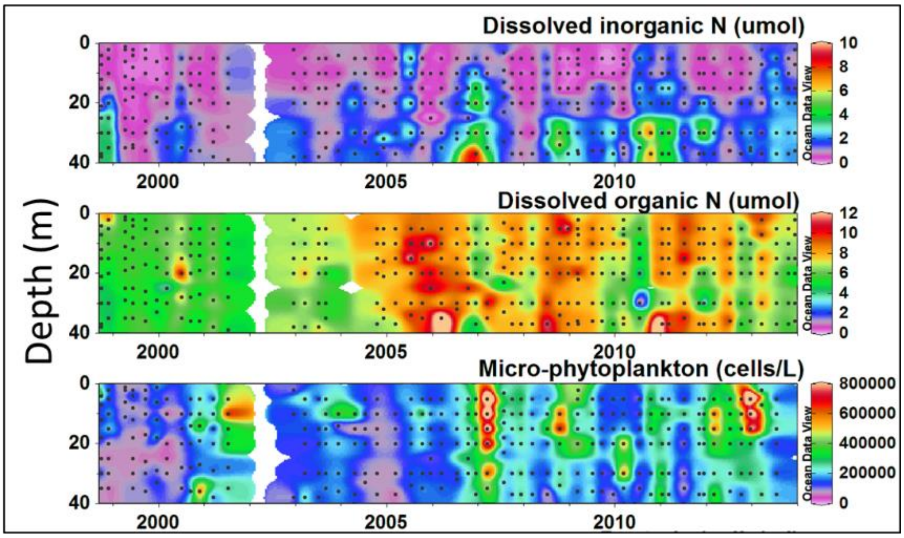


Figure H-5: Concentrations (µmol) of nutrients plotted by depth and time at the Firth of Thames monitoring site 1998¬–2013. Ticks on the x-axis correspond to 1 January of each year. Depths of water samples are shown by black dots.

Table H-1: Non-parametric seasonal Kendall tests for time trends in areal (m-2) nutrient concentrations (µmol) and ratios. Data from October 1998 to July 2013 were grouped by season (seasons used were: Sep – Nov, Dec – Feb, Mar – May, Jun – Aug for spring, summer, autumn and winter, respectively). Significant trends are indicated in by red P values and green limits on 95% CI's (see Appendix A in Larned et al. (2015)).

Nutrient variable	Sample size	Median value	*P*	Median slope (annual)	5% confidence limit for slope	95% confidence limit for slope	Percent annual change
DIN	57	43.3	0.01	2.22	0.78	3.26	5.14
DIP	57	14.6	0.69	-0.1	-0.31	0.17	-0.68
DIN/DIP	57	3.1	0.04	0.13	0.05	0.24	4.15
DRSi	47	408.0	0.25	-8.69	-18.23	3.49	-2.13
DON	53	278.7	0.02	4.92	1.49	8.31	1.76
DOP	51	6.9	0.04	-0.38	-0.7	-0.08	-5.48

To examine if the trends in nutrient levels over the time series arose from changed offshore oceanographic circulation patterns, time trends of salinity, temperature and DIN were examined (Zeldis et al. 2015). There was no evidence of higher salinity and cooler water (with respect to freshened and warmer coastal water) at the monitoring site, which would indicate increased offshore, higher DIN, water. Current meter time series near the seabed and near surface showed no trends in rates of exchange of outer Firth waters with the seaward Hauraki Gulf. Freshwater flows from rivers also showed no significant time trends, indicating that rate of estuarine circulation would not have changed (consistent with the current meter records, above), further discounting the likelihood that increased offshore advection of deep water caused higher nutrient levels.

Southern Oscillation Index (SOI) state is related to wind directions and upwelling frequency in the NE North Island, with westerly winds (upwelling favourable) more common during negative SOI (MacDiarmid et al. 2013). SOI varied between negative and positive states over frequencies considerably higher than the long-term increasing trend observed in the DIN time series, and was most persistently negative (upwelling favourable) when DIN concentrations were low. It therefore did not predict the nutrient trends. Satellite remotely-sensed sea surface temperature (SST) data for the shelf north of the Gulf also did not exhibit a long-term cooling trend, discounting an upwelling source for increased inshore nutrient levels.

River water quality changes also do not appear to have been a strong potential driver of the recent trends in enrichment. Published water quality trends (Vant 2013) showed improvements in total N in the Piako and deteriorations in the Waihou. The Waihou (which contributed approximately 60% of the TN loadings to the Firth in the mass balance) showed 'important deteriorations' (increasing total N slopes of 1.0 and 1.7% y^{-1}) at two sites and a 'minor' increasing slope of 0.5% y^{-1} at a third site. These rates of change were considerably less than the trends in DIN in the marine record. Further discussion of drivers of change in the marine record is made below.

Thresholds associated with nutrients

The Firth is now dominated by catchment-derived N inputs with offshore supply of nutrients playing a lesser role. However, prior to historical land-use intensification, ocean-side loading was likely to have contributed a much larger percentage to a much lower nutrient load, overall (Snelder et al. 2017). The reasoning for this is that for developed catchments of the Hauraki Plains, point and diffuse (agricultural) human sources contribute about 8% and 70%, respectively of total N load to the major rivers, with 'natural' sources the remainder (Vant 2013), a result consistent with other work showing dominant effects of intensification on nutrient export (Quinn and Stroud 2002, Snelder et al. 2017) in New Zealand landscapes. Intensification has therefore greatly increased nutrient loads in the major Hauraki rivers (Waihou and Piako). They now are among the most heavily N-loaded rivers in the North Island. (Snelder et al. 2017). The fact that they terminate in a relatively enclosed water body (the Firth) rather than the open sea further accentuates their potential for trophic impact (Ferreira et al. 2005) (considered further in the Phytoplankton section).

As noted, the important increases in N concentrations in the last 15 years in the Firth do not appear to be explained by decreases in river water quality. However, the dominant role of denitrification in loss of N from the system means that decreases in denitrification efficiency could have important effects on water quality. Studies in Chesapeake Bay (USA) and elsewhere have shown that denitrification efficiency drops with degree of organic enrichment and anoxia in bottom waters (Cook et al. 2004, Eyre and Ferguson 2009, Hale et al. 2016, Kemp et al. 2005, Kemp et al. 1990). Lowered denitrification efficiency would lead to greater N recycling in the system, consistent with observations of Firth enrichment over the last 15 years. The mass-balance analyses showed reduced denitrification rates (by ~ 60%) between 2000-01 and 2012-13, and findings of organic enrichment and increased ammonium efflux in sediments over a similar time frame are consistent with this hypothesis. The accelerated availability of nutrients to primary producers is a positive feedback on the eutrophication process.

While nutrients are considered key drivers of primary (phytoplankton) and secondary (e.g., oxygen, pH) indicators of trophic state (Hughes et al. 2011) nutrient loading levels or concentrations are often not ascribed thresholds or used in a limits-setting context (Sutula 2011). The situation is complicated because the trophic response depends on physical and hydrodynamic factors which differ across different estuary types, such as temperature, stratification, residence time etc., in addition to nutrient loading. Concentrations, as distinct from loads, are particularly problematic in that they often do not reflect loading rates or nutrients available to primary producers, during nutrient-limited phases of the annual cycle (Bricker et al. 2003). Thus, managers have found it challenging to use loads or concentrations of nutrients in coastal systems in a limits-setting context with a high level of confidence (Sutula 2011).

Notwithstanding, below is described some ways Firth nutrients have been evaluated in terms of thresholds. With respect to the (ANZECC 2000) guideline trigger levels the water column-integrated median DIN values at the Firth monitoring site (~20 mg m^{-3}) are below the unimpaired (ANZECC 2000) level (30 mg m^{-3}) and are in the 'good' category of the USA National Estuarine Eutrophication Assessment guidelines (Bricker et al. 2003). Further inshore in the Firth, the annual ranges of DIN were wide, with high values (about 200-900 mg m^{-3}) in winter (Vant 2011), when concentrations could be expected to represent 'conservative' values relatively unaffected by (light-limited) uptake. Such values place the inner Firth within the 'fair' ANZECC category. This example illustrates the effects that the physiography and seasonality of an estuary can have on the assessment of nutrient levels against trophic criteria.

Nutrient levels at which phytoplankton growth can be expected to maximise (from literature: between 14 and 70 mg m^{-3} DIN (Eppley 1969, Morel 1987)) were regularly met by DIN values in the inner Firth (medians 6-45 mg m^{-3} DIN) (Vant 2011). Also, DIN concentrations deep in the water column at the Firth monitoring site (where light-limitation is often strong) often lay within this range (median about 28 mg m^{-3} DIN).

The importance of physiography in determining trophic response was illustrated by the case of Chesapeake Bay, USA. Chesapeake Bay has an annual mean areal loading rate of about 14 g N m^{-2} y^{-1}. While this is not particularly high in comparison with estuaries globally (Boynton and Kemp 2008), Chesapeake Bay is an iconic example of severe environmental degradation driven by anthropogenic nutrient loading, with intense, recurring deep-water hypoxia, compromised biodiversity and impaired fisheries (Kemp et al. 2005, Murphy et al. 2011). Chesapeake Bay has physiographic similarities to the Firth – it is a large coastal embayment that is subject to seasonal stratification and has long residence time and relatively clear waters, which favour retention, growth, and senescence of phytoplankton and bacteria. The mass-balance budget estimate of watershed loading received by the Firth was 4.5 g N m^{-2} y^{-1}, which can be compared with the estimate for Chesapeake Bay of 14 g N m^{-2} y^{-1}, above. If the Firth were to receive more areal loading it could show similar responses as Chesapeake Bay because of its similar physiography. The point at which this might occur is not presently known.

Reversibility and remediation associated with nutrients

It is likely that there has been a historic, progressive shift in trophic baseline of the Firth, increasing its dissolved nutrient stocks as agriculture has developed (Zeldis et al. 2015). Reversing this trend would entail significant changes to land use which has developed over many decades. In recent times (last 15 years), increases in DIN and DON have occurred in the Firth. These recent trends of declining water quality in the Firth are adding to the 'shifted baseline' that has occurred historically in the Firth over decades (as also discussed in previously for sediments). However, the Firth it is not yet displaying the serious eutrophic conditions occurring in highly compromised systems such as Chesapeake Bay or other systems overseas (see next section). This may not be the case if current trends in N concentration continue to increase.

Phytoplankton and bacteria dynamics

Phytoplankton (planktonic micro-algae) form the autotrophic basis of the marine food web in the Hauraki Gulf and Firth, consuming inorganic nutrients during photosynthesis, forming organic matter and feeding the heterotrophic (non-photosynthetic) food web. As such, they are key intermediaries in the NEM of the coastal ecosystem.

The seasonal abundances of phytoplankton are governed most strongly by light levels and nutrient supply (Boynton et al. 1982). Zooplankton graze the phytoplankton, and their abundance is often tightly linked to phytoplankton productivity. Seasonality of phytoplankton and zooplankton in the Hauraki region was described by Chang et al. (2003), Hall et al. (2006) and Zeldis and Willis (2015). Phytoplankton growth is light-limited in winter, allowing inorganic nutrients supplied by ocean exchange, river inputs and recycling to accumulate. Increasing light in spring accelerates growth and nutrient uptake. By early summer, growth maximises and nutrients (especially nitrogen) are reduced at rates faster than their re-supply. In late summer and autumn nutrient limitation becomes most intense. The phytoplankton become re-distributed toward the deeper water column where higher nutrient levels have persisted under lower light conditions. Ultimately, with onset of increasing nutrient and light-limitation, growth stops, whereupon the cells senesce and decompose. Bacteria are intertwined in this functioning, consuming and respiring the organic particulate matter and accompanying dissolved organic matter, and regenerating inorganic nutrients while consuming oxygen. These productive and respiratory metabolic functions are the means by which phytoplankton and bacteria mediate the levels of O_2 and CO_2, and consequent symptoms of anoxia (Conley et al. 2009, Paerl 2006) and acidification (Provoost et al. 2010, Sunda and Cai 2012).

Phytoplankton biomass and growth are demonstrably related to nutrient loading rates (Boynton and Kemp 2008, Boynton et al. 1982) but the relationships between the nutrient drivers of the phytoplankton and the hazards of eutrophication are not consistent across different types of coastal ecosystems (Cloern and Jassby 2008, Monbet 1992). Instead, the relationships are governed by physiographic differences specific to estuary type (Hughes et al. 2011). This is evident across New Zealand's estuaries (Robertson et al. 2016) (Zeldis et al. 2017). Many of our shallow estuaries are well-flushed, with short water residence times that deliver phytoplankton to the sea before they can assume bloom proportions. Others are turbid with poor light that limits growth even under heavy nutrient loading. The long residence times and relatively clear waters of large coastal bays such as the Firth of Thames, mean such systems are sensitive to nutrient enrichment. They are susceptible because, in addition to potentially high nutrient loading from large catchments, they can support phytoplankton blooms which undergo complete life cycles of growth, retention and senescence (Ferreira et al. 2005), capable of engendering eutrophication.

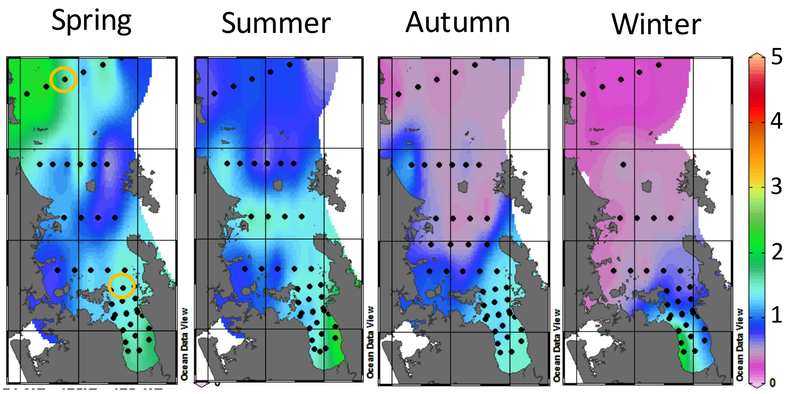
Phytoplankton biomass has been measured using chlorophyll-*a* (chl-*a*) samples collected over the greater Hauraki Gulf and Firth many times, dating from 1996 (Zeldis et al. 2015). Biomass was concentrated in the inner Gulf coastal areas, and was particularly high in the Firth (Figure 6 a). In the Firth, biomass varied seasonally, generally greatest in spring and least in winter. It was widely distributed in spring, but became progressively restricted to inshore areas (especially the Firth) from summer through winter. In spatial surveys conducted in 2012-13 (see Oxygen and pH section) highest biomass was found in the inner Firth in autumn, when it exceeded 5 mg m^{-3} over a large area. In more recent surveying (2015-2017) of the inner Firth, chl-*a* levels up to 22 mg m^{-3} have been sampled (J. Zeldis unpubl. data).

Primary production (carbon fixation) was greater in the Firth than offshore on the shelf (Figure 6 b) (Gall and Zeldis 2011), driven by higher phytoplankton biomass, proximity to nutrient loading from rivers, and greater nutrient recycling from the benthos and the water column. Primary production rates in the Firth were greatest in summer, intermediate in spring and autumn, and reached minima in winter as light became limiting.

Annual average rate of primary production in the Firth was ~ 190 g C m^{-2} y^{-1} (Zeldis et al. 2015) which places the Firth at the mesotrophic level of productivity in the spectrum from oligotrophy to eutrophy in the analysis of Nixon (1995)[29]. Systems with this level of production can be expected to export a high proportion (~45%) of production out of the upper water column to be degraded at depth. Chl-*a* data from moored fluorometers and shipboard sampling at the NIWA Firth of Thames monitoring site showed the deepening of the phytoplankton in autumn indicating the exhaustion of upper water column nutrient, and re-distribution of senescent phytoplankton under nutrient stress. Implications of these dynamics for oxygen depletion and pH are discussed in the Oxygen and pH section.

[29] The (Nixon 1995) trophic classification designates oligotrophic as <100, mesotrophic as 100–300, eutrophic as 301–500 and hypertrophic as >500 g C m^{-2} y^{1}.

a.



b.

C Fixation (mg C m^{-3} d^{-1})	Outer Firth	Shelf
Spring	24	8
Summer	66	7
Autumn	27	12
Winter	19	9

Figure H-6: (a): Surface chlorophyll-a (averaged over upper 15 m) sampled over seasonal NIWA voyages made between 1996 and 2013. Sampling sites shown by black dots (Zeldis et al. 2013), many of which are overlaid. b): net-primary production (carbon fixation) for each season at outer Firth and northeast shelf sites (orange rings in the Spring figure) (Gall and Zeldis 2011, Zeldis and Willis 2015).

To determine long-term variation in phytoplankton, samples were collected at the NIWA Firth monitoring site approximately every 3 months over the 15 years between September 1998 to July 2013 (excepting July 2001 to December 2002). Very small pico-phytoplankton (<2µm cell size) and bacteria were collected at the Firth site from December 2002. Time-series analyses showed significant increasing trends for both chl-*a* and its breakdown product (phaeopigment) in the lower half of the water column (20-40 m depth) (4.2% y^{-1} increase for chl-*a*), but no trends in the upper water column (<20m). Cell counts of total micro-phytoplankton increased significantly (6.9% y^{-1}) over 1998-2013 (Table 2, Figure 5). When analysed by functional groups, diatoms increased by 4.6% y^{-1} and a diverse category composed of taxa (called small flagellates) increased by 6.7% y^{-1}. Dinoflagellates showed no trend. Large centric diatoms increased by 8.9% y^{-1}.

Table H-2: Non-parametric seasonal Kendall tests for time trends in areal cell counts (cells m-2 x 10-3) integrated over the water column for total micro-phytoplankton and its taxonomic components. Data from October 1998 to July 2013 were grouped by season (seasons used were: Sep – Nov, Dec – Feb, Mar – May, Jun – Aug for spring, summer, autumn and winter, respectively). Significant trends are indicated in by red P values and green limits on 95% CI's (see Appendix A in Larned et al. (2015)).

Counts	Sample size	Median value	*P*	Median slope (annual)	5% C.L. for slope	95% C.L. for slope	Percent annual change
All micro phyto	53	6938291	0.00	477211	223557	646149	6.9
Diatoms	53	1129149	0.03	51870	14361	96930	4.6
Dinoflagellates	53	237451	0.39	-5096	-18291	4258	-2.2
Small flagellates	53	5374044	0.00	361234	175643	477743	6.7
Large centric diatoms	53	341824	0.01	30252	8796	55277	8.9

Prokaryotic pico-phytoplankton and bacteria also showed significant increasing trends. Bacteria biomass (7.0% y^{-1} increase) was larger than summed micro-phytoplankton biomass and so was a very important component of the plankton.

Nutrient enrichment affects phytoplankton biomass, group composition and ecology. Increased biomasses accompanied the nutrient enrichment of the Firth over 1998-2013 (Nutrient section), with 'diatom and small flagellates' and bacterial biomasses increased 88% and 101%, respectively, while dinoflagellates showed no trends. Some dinoflagellates have a higher phosphorus (P) requirement than some other species groups (Smayda 1997) and P concentration, unlike N, did not increase. Thus, the enhancement of N, along with adequate silicon, may have increased diatom abundance relative to dinoflagellates. The dissolved organic nitrogen pool also increased significantly. This would support taxa known to consume dissolved organic matter, including mixotrophic taxa common in the 'small flagellates' group and bacteria.

Nutrient enrichment is also known to favour the downward transport of algal carbon in coastal systems, due to higher algal biomasses and shifts toward larger algal species (i.e., large centric diatoms) (Riegman 1995). This was consistent with the observed increase of chl-*a* biomass in the lower water column (63%) over 1998-2013 (Zeldis et al. 2015).

Along with light, the main factor governing phytoplankton production in the Firth is nutrient loading from its catchment – this was evident from the strongly increasing gradients of phytoplankton biomass and productivity from the outer Hauraki Gulf to the inner Gulf coastal zone and especially the Firth. The correlations of nutrient increases with phytoplankton and bacterial increases and taxon shifts seen during 1998-2013 was also evidence of this dependence. The mass-balance budget (Nutrient section) showed a dominant effect of catchment-side loading on NEM, contributing an estimated 87% of its DIN load, and rendering the system frequently net-heterotrophic in its sediments and water column via the seasonal formation of oxidisable organic matter within the Firth. Catchment loading was thus likely to have a pervasive effect on present day Firth phytoplankton and bacterial ecology. This was in agreement with Firth biophysical modelling by Broekhuizen and Zeldis (2007) which predicted that doubled nutrient loads would increase in phytoplankton biomass by 100% in the southern Firth, and 300% increases under 5-fold load increases.

Prior to historical land clearance and intensification, offshore waters were probably a much more important contributor to a much lower overall load to the Firth (Nutrient section). Since then, it is land-side changes that have most influenced changes in Firth primary production and trophic status. The findings that *ca*. 80% of Hauraki Plains loading is anthropogenic in origin, combined with the dominance of catchment side loading shown here, leads to the conclusion that land-use intensification has at least doubled the historic N loads to the Firth. Calculations of Snelder et al. (2017) indicate that this increase is likely to be even larger (order of 4-fold: their Figure 5). These findings lead to the further conclusion that the Firth was probably oligotrophic (in the Nixon (1995) classification) prior to its catchment modification (Zeldis et al. 2015).

Thresholds associated with phytoplankton and bacteria

Resource managers have found it challenging to ascribe ‘hard and fast’ benchmarks for phytoplankton biomass or production, for managing eutrophication. Susceptibility may vary from estuary to estuary, depending on co-factors related to their physiography. Irrespective, the NOAA Assessment of Estuarine Trophic Status (ASSETS), based on the National Estuarine Eutrophication Assessment (NEEA) developed thresholds for chl-*a* with a group of regional experts: estuaries with highest annual chl-*a* less than 5 mg m^{-3} appear un-impacted; at 20 mg m^{-3} chl-*a* and above effects include declines in submerged aquatic vegetation, shifts in phytoplankton community structure, high turbidity and low bottom water oxygen. Within the European Union, the Water Directive Framework uses chl-*a* thresholds similar to ASSETS: with ranges from <5 mg m^{-3} in the undisturbed or slightly disturbed categories to >30 mg m^{-3} for highly disturbed or hypereutrophic.

It thus appears that a commonly used threshold for undisturbed systems is about 5 mg m^{-3}. In general, the Firth is below this level in its outer reaches (including the Firth of Thames monitoring site) although mooring data suggest it is exceeded on rare occasions. It is approached and often exceeded, in the inner Firth. As described previously (Nutrient section), it is likely that phytoplankton growth will maximise at DIN concentrations of about 14 to 70 mg m^{-3} DIN, depending on the taxa concerned. DIN concentrations later in the time series shown here were regularly within this range and were often exceeded in the inner Firth.

Returning to the discussion of eutrophication of Chesapeake Bay (Nutrient section), it is noted that its severe degradation is accompanied by mean chl-*a* levels in its mesohaline reaches of about 8 mg chlorophyll *a* m^{-3}(Kemp et al. 2005). These values are not much greater than the values often observed in the inner Firth today, or the values used in the ASSETS scoring for unimpacted estuaries (5 mg chlorophyll *a* m^{-3}). Chl-a *a* criteria for the mesohaline reach of Chesapeake Bay ranging from 2.2 to 8.7 mg chlorophyll *a* m^{-3} (depending on season and river flow) have been recommended (Harding et al. 2014) for achieving restoration targets. Values somewhat lower than these were recommended for the polyhaline (seaward) reaches. It is likely that the physiography of Chesapeake Bay (long residence time and stratification) renders it susceptible to blooms, and its similarity in that regard with the Firth implies that the Firth may be susceptible as well.

Reversibilty and remediation associated with phytoplankton and bacteria

Over the time scale of historical land use intensification in the Hauraki region, it is likely that there has been a progressive shift in trophic baseline involving a substantial increase in its dissolved nutrient stocks. In response to this catchment effect, primary production rates suggest that the Firth is now a mesotrophic coastal system. Reversing this trend would entail significant changes to land use which has developed over many decades.

More recent changes in phytoplankton and bacterial abundance and taxon composition, associated with elevated DIN, DON and N:P ratios (Nutrient section), suggest that trophic changes are continuing in the Firth. This may be a consequence of changes in denitrification efficiency, with similarity to the case of Chesapeake Bay, where reduced denitrification efficiency was driven by organic loading (Hagy et al. 2004). Although the Firth is sustaining compromised water quality, including increased phytoplankton and bacteria, it is not displaying the serious eutrophic conditions occurring in highly compromised systems such as Chesapeake Bay or other systems overseas. This may not be the case if current trends continue.

Oxygen and pH stressors

This section discusses dissolved oxygen and pH effects together, because they are linked by the underlying biogeochemistry of NEM and they are both consequences of eutrophication. They are also similar in being ‘acute stressors’, with their ecological effects manifested over relatively brief time scales. These relationships are summarised in Figure H-7.

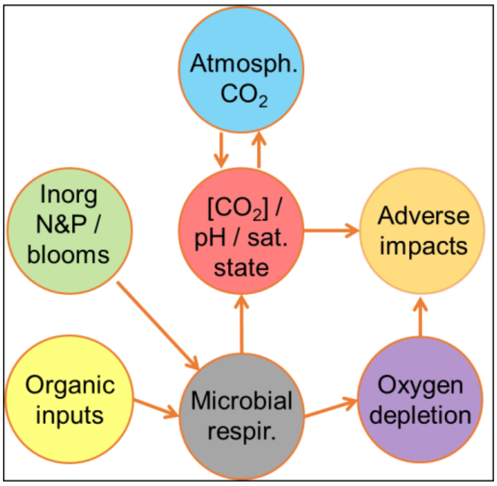


Figure H-7: Mechanisms of oxygen- and pH-related hazards related to eutrophication. Organic inputs and algal blooms stimulated by inorganic inputs fuel microbial respiration, depleting oxygen, generating CO2 and lowering pH and saturation state for calcification. These cause adverse impacts through hypoxia and acidification, against the background of acidification caused by atmospheric CO2 inputs. Adapted from Sunda and Cai (2012).

As shown in the Nutrient and Phytoplankton sections, net ecosystem metabolism (NEM) driven by plants, animals and microbes sets the balance between net-production and net-respiration in coastal waters. Oxygen is essential because it is the ultimate electron acceptor in aerobic respiration, enabling organisms to extract energy from organic matter (Lehninger 1975). Respiration turns oxygen into water, and therefore reduces oxygen concentration in the coastal environment. When respiratory consumption of oxygen becomes greater than replenishment by photosynthesis or hydrodynamic and atmospheric exchange, environmental oxygen concentrations are reduced and can become stressful for biota (Gray et al. 2002, Vaquer-Sunyer and Duarte 2008). In extreme cases, often in combination with physical processes such as water-column density stratification (Scully 2016), de-oxygenation can be catastrophic for biota and normal biogeochemical functioning of coastal ecosystems (Conley et al. 2009).

The respiration of organic matter generates CO_2, which enters the aqueous carbonate system of coastal waters and decreases pH (acidification). This lowers concentration of carbonate ions available for calcification, thus lowering calcium carbonate saturation state (Sunda and Cai 2012, Waldbusser and Salisbury 2014). From the perspective of an organism that requires calcium carbonate, decrease in saturation state is potentially deleterious as it demands that more energy be diverted to building shell at the expense of other key metabolic processes (Capson and Guinotte 2014, Law et al. 2018, Provoost et al. 2010). Negative effects of acidification also include direct effects on metabolism (Kroeker et al. 2010) and behaviour (Munday et al. 2010). At the global scale, ocean acidification is proceeding because of atmospheric injection of CO_2 (Borges and Gypens 2010) and, in some coastal areas, because of upwelling of historically acidified waters (Barton et al. 2012, Law et al. 2018) . However, in many developed coastal zones of the world acidification related to NEM is significantly outpacing atmospherically-driven acidification (Duarte et al. 2013, Law et al. 2018, Provoost et al. 2010).

In the Firth, water column profiles of dissolved oxygen (DO) were surveyed at the NIWA Firth of Thames monitoring site 3-monthly, from 1998-2013 (Figure 8), during the same surveys as used for nutrient and phytoplankton studies (refer Figure 5). In addition, DO was surveyed using moored oxygen sensors from 2005-2014 at the Firth monitoring site, with DO sensors mounted in the upper and lower water column sampling every 15 minutes (Figure 8). Spatial surveys of oxygen were also conducted across the Firth and Hauraki Gulf several times.

The 3-monthly time series of DO profiles at the Firth monitoring site showed:

- The upper 20 m of the water column was generally well oxygenated at about 7-9 mg O_2 L^{-1} and in near equilibrium (>90% saturation) with the atmosphere (Figure H-8).
- However, in most summer and autumn surveys (when temperature was maximal: Figure H-8) there was usually DO depression in the lower 20 m of the water column.
- These low DO conditions were usually between about 70% and 60% saturation although extreme events as low as <40% were recorded, corresponding to DO about 5.7, 4.9 and 2.7 mg L^{-1}, respectively.
- The water column tended to be density-stratified at these low DO times (Figure H-8).
- There were no long-term trends over 1998-2013 in average DO concentrations in either the upper (<20 m depth) or lower ≥ 20 m depth of the water column).

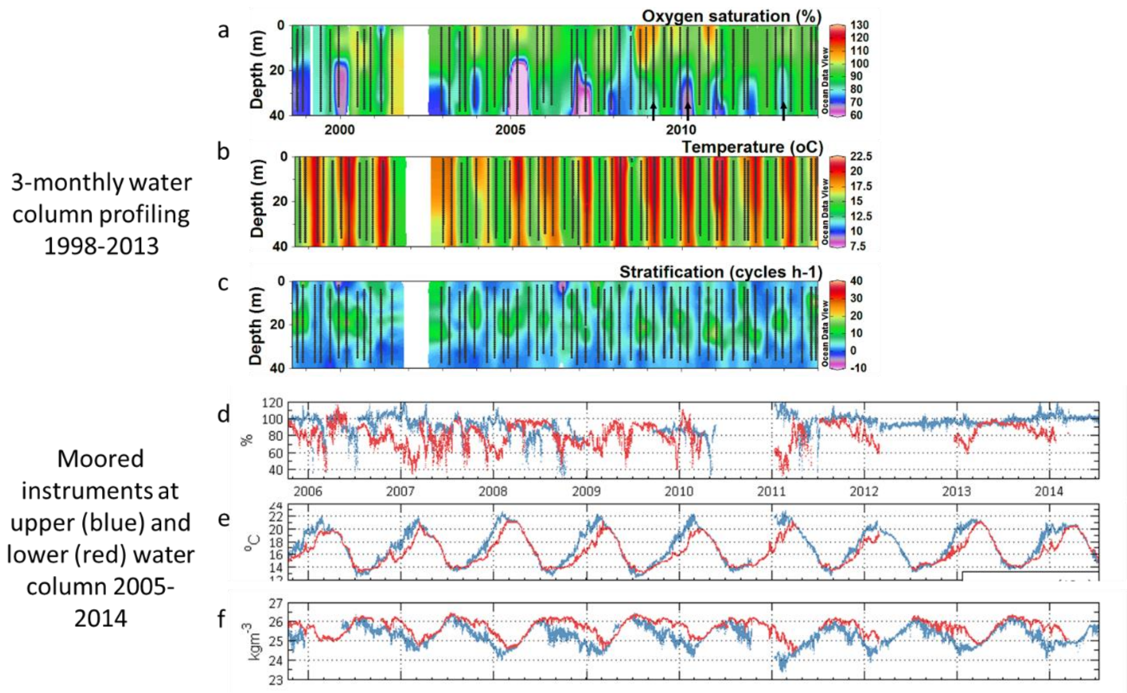


Figure H-8: Oxygen and physical properties from ship and mooring sampling at the NIWA Firth of Thames monitoring site, 1998-2013. (a-c) 3-monthly (1998-2013) water column profiles (1 m depth intervals) of oxygen saturation (% surface O2 saturation), temperature (oC) and water column vertical stratification (higher values mean more strongly stratified). Vertical lines show times of profiles and ticks on the x-axes are 1 January of each year. (d-f) continuous mooring sampling (2005-2014) of O2% saturation at upper and lower water column depths (10 and 33 m from the surface). Also given are temperature and water column stratification (densities at upper and lower water column).

The moored time series of DO at the Firth monitoring site showed:

- In the upper water column (10 m depth) oxygen was generally near saturation but events of lower oxygen (60% or 4.9 mg L^{-1}) occurred occasionally (Figure H-8).
- There were frequent DO depletion events (60% saturation) in the lower water column at 33 m, occasionally dropping as low as 40% or 2.7 mg L^{-1}. These low DO events were often missed by the temporally sparse 3-monthly profiling.
- The low DO periods generally corresponded to periods of warmest temperatures and greater water column stratification (Figure H-8).

Spatial plots of DO, extending from the inner Firth to outer Hauraki Gulf for 3 autumn surveys from separate years (Figure H-9), showed:

- Oxygen depletion was variable in strength between years, with more depletion in 2010 than in 2009 and 2013. This was consistent with the patterns seen in the DO record at the Firth monitoring site (see arrowed dates in Figure H-8).
- The low DO feature occupied a large area of the outer Firth and inner Hauraki Gulf near-seabed environment, in autumn.

- The spatial distribution of the strong autumn oxygen minimum in 2010 was like that of the deep phytoplankton (chl-*a*) maximum. The weaker oxygen minimum of 2013 had considerably less phytoplankton associated
- Waters with O_2 as low as 4.9 mg L^{-1} or 60% saturation were present at the Firth monitoring site in autumn 2010.

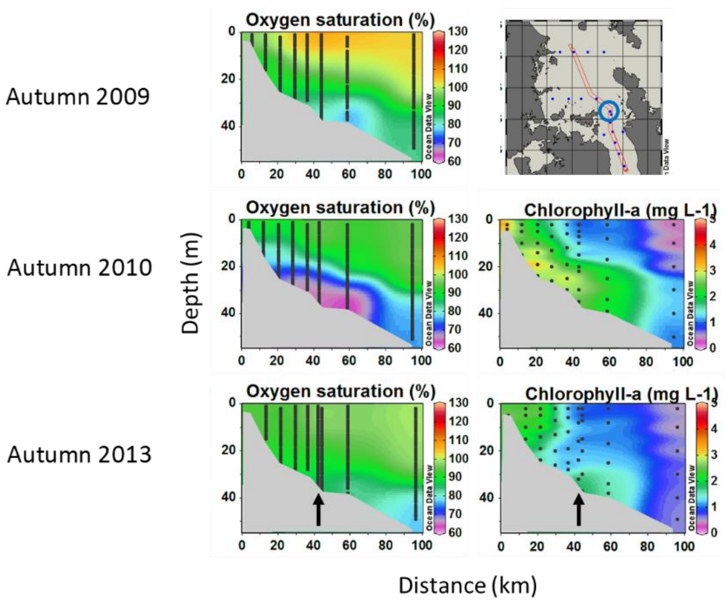


Figure H-9: Oxygen and phytoplankton in the Firth and Hauraki Gulf in autumn 2009, 2010 and 2013. Shown are oxygen saturation, seawater density and chlorophyll-a at stations on the orange transect in the inset map. Chlorophyll was not sampled in Autumn 2009. The Firth monitoring site location is arrowed in the panels and circled in the map.

More recent (2015-2017) O_2 and pH sampling using moorings at the inner Firth sites has shown strong O_2 depression in autumn to early winter lasting for periods of weeks near 60% saturation with minima as low as 40% saturation (J. Zeldis, NIWA unpubl. data).

Carbonate system parameters (partial pressure of CO_2 (pCO_2), dissolved inorganic carbon (DIC), and total alkalinity) were surveyed over the Firth and Hauraki Gulf on four seasonal voyages in 2012-13 (Figure 10 a). Underway mapping of the properties near surface (2m) was done while the ship steamed over the region, including temperature, salinity, chl-*a*, coloured dissolved organic matter (CDOM) and turbidity sampled at the same rates as pCO_2 (10 second intervals) (Figure H-10). During these surveys the ship was stopped at 17 sites to profile the water column for DIC and alkalinity, to collect depth-resolved carbonate data and salinity and nutrient samples for nutrient budgeting (Nutrient section). Results were:

- pCO_2 was highly seasonal, lowest in spring, reflecting spring bloom drawdown by actively growing phytoplankton, and highest in autumn, with maximum values in the Firth (about 550 µatm);
- pH (Figure 10 a) was highest in spring, reflecting CO_2 drawdown by actively growing phytoplankton, and lowest in autumn, when respiration and CO_2 production maximised. It was near oceanic pH (~8.05-8.1) in spring over the whole region, but

reached minimum values of ~7.92 in the inner Firth in autumn. There was a decreasing shoreward gradient in all seasons, strongest in autumn;

- in all seasons, there were shoreward increasing gradients of pCO_2, dissolved organic matter, turbidity and chl-*a* (Zeldis et al. (2015): shown for the autumn survey in Figure 10 b). This pattern mirrored the high rates of oxygen consumption measured in the water column and the benthos in the inner Firth, and the high riverine organic matter supply and higher inshore productivity patterns (Nutrient and Phytoplankton sections).

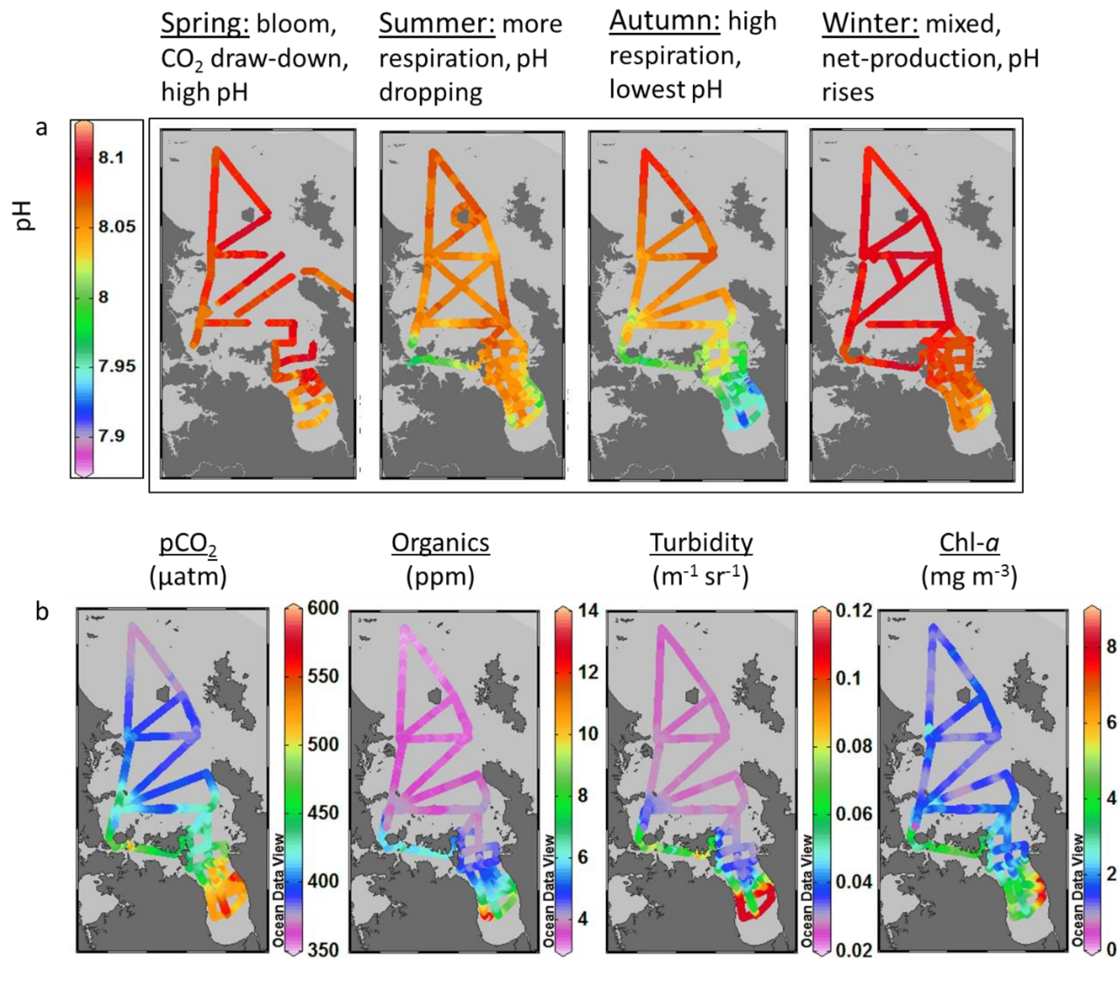


Figure H-10: (a) pH seasonality from underway surveys, showing decreasing gradient of pH toward the Firth and lowest values in autumn. (b) Autumn survey pCO2, dissolved organic matter, turbidity, and chl-a in surface waters showing increasing spatial gradients of all parameters toward the Firth.

Sampling done near-surface and near the sea bed during the 4-season surveys allowed calculation of carbonate parameters, including $Omega_{Ar}$ (Ω_{Ar}), the saturation state of aragonite (Zeldis et al. 2015). Discrete pCO_2 was highest (and pH lowest) in autumn in the inner Firth and was higher (lower) near the sea bed than near the surface in spring and summer, but more equal in autumn and winter. These vertical contrasts were probably sustained by higher phytoplankton growth rates in spring and summer, which reduced CO_2 (and increased pH) in the upper water column. Productivity was lower in autumn and winter, and mineralization was more evenly distributed in the water column. There was a decreasing gradient in saturation state from offshore into the Firth. Values offshore were mainly Ω_{Ar} = 2.8-3.0 and in the Firth, were usually about Ω_{Ar} = 2.3-2.4.

An underway survey in autumn 2010 showed higher pCO_2 than in autumn 2013, but otherwise the two surveys showed almost identical increasing shoreward pCO_2 (Zeldis et al. 2015). pH values in autumn 2010 were lower than in 2013, reaching minimum values of 7.88 and with values less than 7.95 over a large area of the inner Firth. The ΩAr values sampled in autumn 2010 were lower than in autumn 2013, reaching minimal values of ~1.8 and were typically ~ 2.2-2.3 in the Firth. The autumn 2010 survey also had lower oxygen in deeper waters of the mid-outer Firth, and a more intense near-bed maximum of phytoplankton than the autumn 2013 survey.

The recent (2015-2017) moored O_2 and pH sampling in the inner and outer Firth has shown strong pH depression in autumn to early winter in the inner Firth (< 7.7: J. Zeldis, NIWA unpubl. Data: Figure 11). This corresponds to the strong, coincident O_2 depression observed.

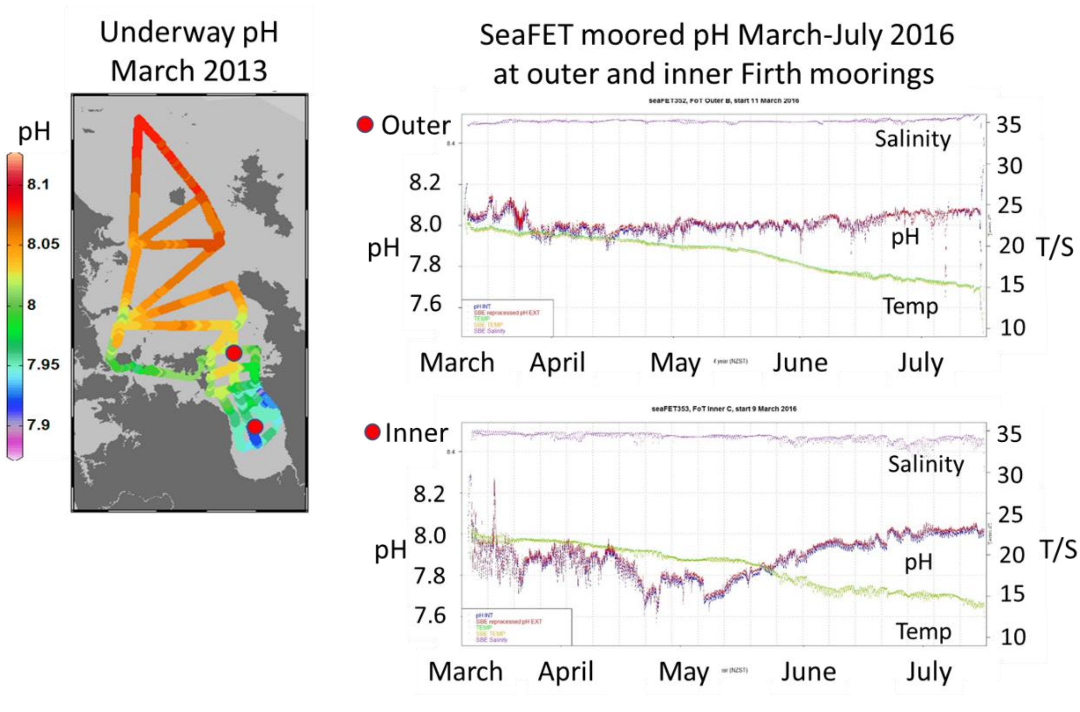


Figure H-11: Sampling with moored pH sensors (SeaFETs) at NIWA Firth Monitoring site (outer) and inner Firth sites March-July 2016, showing large pH depression in autumn early winter at the inner Firth. The map shows the mooring site locations and pH patterns found in the underway survey of autumn 2013 (refer Figure H-9).

The results of this O_2 and pH section together with the Nutrient and Phytoplankton sections show how the cycles of carbon, oxygen and nutrients involved in NEM are linked (Howarth et al. 2011) (Figure H-7). Oxygen reaches annual minima as the primary production of spring and early summer is consumed in late summer/autumn by microbial respiration. pH minimises in autumn as CO_2 maximises from the respiration. The carbonate mapping (Figure H-10) revealed strong temporal and spatial correlations of pCO_2 with other indices of NEM including primary production (Phytoplankton section), oxygen consumption in the water column and benthos (Nutrient section), chl-*a*, organic matter and turbidity (this section).

Oxygen minima of varying intensities form in the lower 20 m of the water column at the outer Firth monitoring site, as the water column stratifies in summer and autumn each year. The minima are largely confined to the deeper water but do, on occasion, penetrate into the upper water column. This is significant because of the proximity of the monitoring site to the zone designated for fish farming off Coromandel (see 'thresholds', below). The minima extend well into the Firth, including waters adjacent to the large mussel farms at Wilson Bay Areas A and B. The oxygen minima also extend into the inner Hauraki Gulf, largely congruent with the seaward extent of high phytoplankton densities extending into the Gulf from the Firth. The intensity of oxygen minima correlated with the concentrations of this phytoplankton near the seabed, consistent with the interpretation that they are driven by respiration of senescent primary biomass, are metabolic in origin and local to the Firth.

Higher CO_2 production in the inner Firth than further seaward was consistent with the benthic and pelagic oxygen uptake measurements which showed the inner Firth sustains net-heterotrophy about twice that of the outer Firth. In all seasons (except winter when the water column was mixed), pCO_2 was higher and pH and saturation state were lower near the sea-bed than near surface, and higher toward the inner Firth than outer Firth and Hauraki Gulf, indicating a metabolic association with oxygen consumption. This has implications for benthic animals which, in addition to experiencing potential oxygen minima, could also be exposed to more extreme pH effects.

Formation of physical stratification allows the deep O_2 minimum to form in the outer Firth. This is similar to formation of seasonal hypoxia in Chesapeake Bay, which is set by the interaction seasonal physical stratification and the extent of eutrophication (Hagy et al. 2004, Scully 2016). Although oxygen depression is much more severe in Chesapeake Bay than in the Firth, the two systems probably share this mechanism.

Further seaward, it is likely that low O_2 in the outer Hauraki Gulf originates in a mixture of local oxygen consumption and upwelled, lower oxygen waters advected from offshore. This upwelled water appears in the outer Hauraki Gulf in spring and early summer forming a reduced-oxygen, near-bed layer (Zeldis 2004). Upwelling frequency declines markedly in the region after December, after which oxygen depletion is driven more by local remineralisation than by advection. This is when heterotrophy maximises in the Firth with oxygen utilisation levels about four times that found in the outer Hauraki Gulf (J. Zeldis pers. obs.).

It is unlikely that Firth pH is strongly affected by anthropogenic CO_2 injected from the atmosphere, as it is exhibiting pH variation much larger than the open ocean, and its offshore waters currently have pH close to pre-industrial levels. Modification by river runoff with differing pH and alkalinity was also unlikely, as maximum depression of Firth pH occurred in autumn when river flows were at their annual minima. The large depression in pH in autumn relative to other seasons was not related to depressed salinity. These points favour the proposition that dominant drivers of the Firth carbonate system are metabolic rather than oceanographic or hydrographic (Duarte et al. 2013, Law et al. 2018, Provoost et al. 2010).

There was no multi-year trend in oxygen depression at the Firth monitoring site in the 1998-2013 ship sampling nor the 2005-2014 mooring data. The lack of trend could be because of the complex factors affecting oxygen concentrations and distribution, including intensity of stratification and degree of organic loading over the growth season and, in the case of the ship sampling, the periodic nature with which the environment is being sampled. In this regard, the oxygen data are different from the trends in nutrients and phytoplankton and bacteria which have shown large increases over 1998 to 2013.

The time series for pH is brief in comparison to that for oxygen. The observation that the Firth's net heterotrophy is driven strongly by nutrient loading and that such loading has increased historically with land development (Nutrient section) leads to the conclusion that the 'carbonate climate' in the Firth has been altered over decades. This would apply also to the oxygen climate. As noted above, it is characteristic that coastal acidification related to catchment loading is proceeding considerably faster than atmospherically driven acidification. The maximum pH reductions presently reached in the Firth (often 0.2 units and up to 0.4) already approach the 0.4 reduction expected by 2100.

Thresholds associated with oxygen and pH

Limits setting by coastal managers with respect to oxygen has been more certain than for nutrients and phytoplankton (Sutula 2011). Oxygen indicators are considered to have strong relationships between the indicator and nutrient management, and acceptable measurement precision, for eutrophication assessment. For example, within the the NEEA assessment for US estuaries a limit of 3 mg O_2 L^{-1} (~40% saturation, at 19°C) was set for the fair/poor' threshold and 5.5 mg O_2 L_{-1} (~60% saturation) for the 'good/fair threshold. Criteria for Chesapeake Bay were developed with limits including a 30-day mean of 5 mg L^{-1} applied to open-water habitats, with 7-day means of 4 mg L^{-1}(~45% saturation) and instantaneous minima of 3.2 mg L^{-1}.

Published meta-analyses have outlined a taxonomic progression of decreasing sensitivities to DO stress, progressing from fish → crustaceans→ annelids → bivalves (Gray et al. 2002, Vaquer-Sunyer and Duarte 2008)(Figure H-12). Fish and crustaceans had the highest (i.e., were most susceptible) lethal concentration thresholds, followed by bivalves. Sublethal thresholds, associated with live-giving factors such as reduced growth and reproduction, increased physiologic stress, forced migration, reduction of suitable habitat, increased vulnerability to predation, and disruption of life-cycles were found to be highest for fish and crustacea, followed by molluscs. Lethal times (after exposure to acute hypoxia) were shortest for crustacea and fish (order of few hours to a few days) while times for molluscs were order of a few hundred hours.

The meta-analysis of Vaquer-Sunyer and Duarte (2008) has questioned the widespread use of the threshold of 2 mg O_2 L^{-1} and recommended its upward revision. They showed that the 2 O_2 L^{-1} threshold is below the empirical sublethal and lethal O_2 thresholds for half the species they tested. A level of 4.6 mg L^{-1} (~50% saturation) was recommended by those authors as 'a precautionary limit to avoid catastrophic mortality events, except for the most sensitive (e.g., crab) species, and effectively preserve biodiversity'. Within New Zealand and based on this information it was recommended in the Horizon's Regional Council 'One Plan' that dissolved oxygen saturation standards proposed for its estuary management subzone should be 70% DO saturation (Zeldis 2009). A level of 80% saturation is recommended in Waikato Regional Council standards to avoid unsatisfactory conditions. As shown here, these standards for minimum O_2 levels are often breached, both at the outer Firth site in the lower water column and especially the inner Firth site. They are also approaching the 'fair/poor' thresholds in overseas limits settings given above.

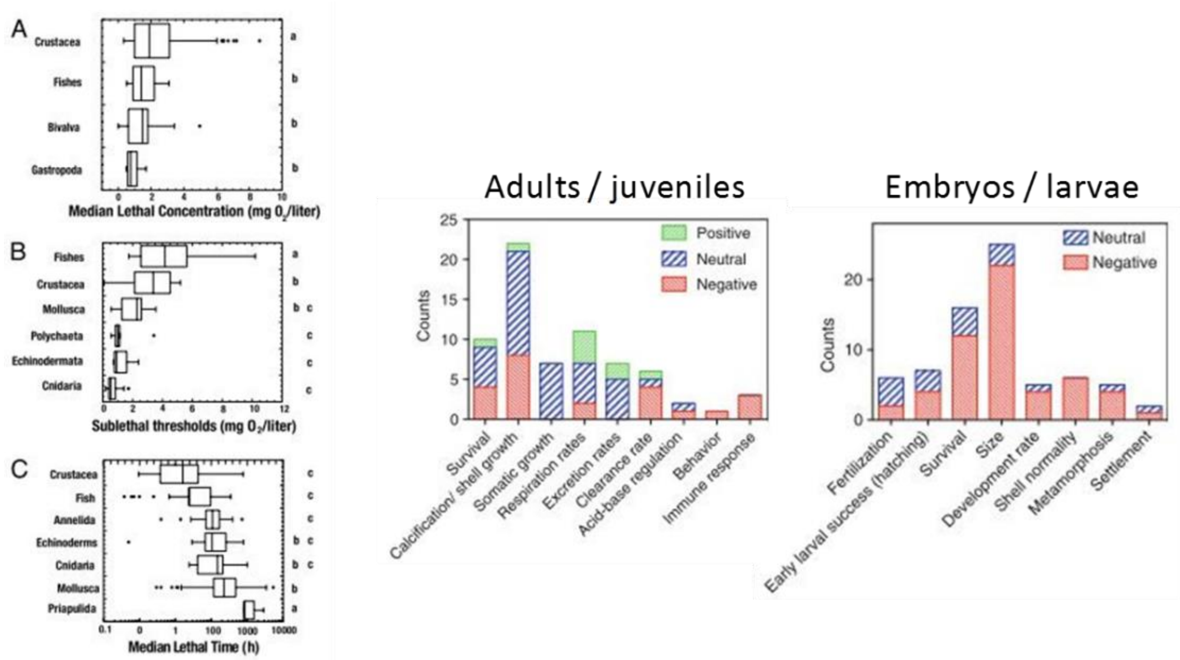


Figure H-12: Left panels: box plots showing the distributions of oxygen thresholds among taxa. (A) Median lethal concentration (mg O2 L-1). (B) Median sublethal concentration (mg O2 L-1). (C) Median lethal time (h). Adapted from Vaquer-Sunyer and Duarte (2008). Right panels: Summary of the literature describing impacts of ocean acidification on marine-shelled molluscs for studies considering a pH decrease lower than 0.4. Results are given separately for adults and juveniles, and embryos and larvae. Adapted from Gazeau et al. (2013).

The meta-analyses cited above indicate that bivalve molluscs are among the most resilient invertebrates to low O_2. This is relevant to the observations of low O_2 in the vicinity of the Wilson Bay Areas A and B Marine Farm Zones (Zeldis et al. 2015). From the limited data available to date, O_2 conditions appear to be sufficient for bivalve farming. However, there is no information on O_2 tolerances of cultured green-lipped mussels (*Perna canaliculus*) adults. For larvae, data for *P. canaliculus* showed large negative effects on survival and settlement at 6 mg O_2 L^{-1} (~65% saturation), but did not affect spat survival or settlement ((Alfaro 2005). Such levels have been measured at Wilson Area B near surface (Zeldis et al. 2015) and values as low as 4.2 mg L^{-1} have been measured near the sea-bed. Given that settled spat, rather than larvae, are used in farming operations these results may indicate a reduced risk to operations, although effects on farmed spat should be further investigated.

The most sensitive group in terms of sub-lethal effects is fish, particularly active swimmers. This is relevant to the fish farming in the Coromandel Fish Farm Zone, for which farming of yellowtail kingfish (*Seriola lalandi*) and hapuku (*Polyprion oxygeneios*) has been proposed. Studies on kingfish juveniles showed a 13% decrease in specific growth rate under hypoxic conditions at 21^{o}C (Bowyer et al. 2014). We know of no data for adult kingfish performance or for hapuku. In comparison, for salmon the recommended minimum dissolved oxygen concentration is 6 mg L^{-1} and concentrations below that are defined as hypoxic (Sim-Smith and Forsythe 2013).

Another relevant issue is that oxygen levels below fish pens near the seabed can become hypoxic because of organic matter deposition from the pens. The likelihood of this is highly dependent on the rate of supply of oxygen, which, in turn, determines the rate of mineralisation of waste. This rate will be determined by the local deposition, and by the ambient field of oxygen in the wider environment. The regular occurrence of oxygen minima below 6 mg L^{-1} in the lower 20 m of the water column at the Firth monitoring site and the propensity for this condition to penetrate to shallower depths are of concern for management of prospective fish farming in the Firth and require further investigation. The biogeochemical environment itself is also subject to identifiable thresholds of response to oxygen minima. Studies have shown that denitrification is sensitive to surficial sediment and epibenthic oxic conditions, with less N_2 produced per unit N effluxed) as benthic DO decreased (Nutrients section). Important effects occur at bottom water DO of 3-5 mg L^{-1}. Such levels occur in Firth bottom waters (e.g., 2005 and 2007), especially in the inner Firth.

Limits setting with respect to pH is less well developed, reflecting a less certain scientific understanding of the effects (Sheldon and Alber 2011). For calcifiers, at $\Omega < 1$, $CaCO_3$ shell dissolves and at $\Omega >> 1$, it is easier for an animal to build and maintain a shell. For most calcifiers however, growth and survival can be compromised at Ω values considerably higher than 1 because of the increasing physiological cost building shell as Ω drops. This is most marked for corals for which survival can be compromised at Ω values as high as 3.5.

Published meta-analyses (Gazeau et al. 2013, Kroeker et al. 2010, Law et al. 2018) have found biological effects of acidification to be large and negative but with important variation across taxa. It was found that larvae were more sensitive than adults (Figure H-12). Experiments have often been done using carbonate conditions considerably more extreme than those presently observed in the Firth. In a more relevant example (in terms of similarity to conditions in the Firth), oyster larval production in US Pacific Northwest hatcheries has been shown to decline at Ω_{Ar} values below 2.0 (*cf* minimal values of 1.8 in the Firth and commonly observed values of 2.2-2.3) (Capson and Guinotte 2014) .

In addition to direct effects on the calcification process, fertilisation, physiological and behavioural impairments associated with carbonate system parameters (pCO_2, pH). Elevated CO_2 was shown to have dramatic effects on behaviours and sensory responses of reef fishes with detected at 700 ppm CO_2, with many individuals becoming attracted to the smell of predators at pCO_2 of 700 µatm. At 850 µatm, the ability to sense predators was completely impaired (Munday et al. 2010). These pCO_2 values are not greatly above present-day Firth values in autumn.

Work in New Zealand on local molluscan responses to acidification was reported at recent workshops (Capson and Guinotte 2014) and by (Law et al. 2018). *P. canaliculus* larvae underwent significant (nearly 50%) reductions in growth rate in pH = 7.7 vs 8.0 treatments (Law et al. 2018). Large effects on shell thickness were also demonstrated, though pH ranges were not detailed. Testing on adult (5-14-month-old) cockles, abalone, and flat oysters, in terms of survival, respiration (abalone), growth, reburial (cockles), physical condition, weight loss, physiological condition and righting behavior (abalone) showed impairment at pH declines of 0.3—0.4 pH units and Ω_{Ar} states above 1.

To summarise the information regarding O_2 and pH-related stressor thresholds and aquaculture species in the Firth, it is not known if these stressors have reached levels deleterious to aquaculture operations. However, it is likely that such levels are being ‘approached’. Oxygen levels have already reached deleterious levels for prospective finfish culture during autumnal, low oxygen events. pH – related effects for larval stages of wild fish stocks may be getting approached.

Reversibilty and remediation associated with oxygen and pH

Although the Firth is sustaining compromised water quality, including seasonal oxygen and pH minima, it is not displaying the serious eutrophic conditions occurring in highly compromised systems such as Chesapeake Bay or other systems overseas. This may not be the case if current trends continue.

The trends over the last decade of declining water quality (Nutrient and Phytoplankton sections) in the Firth are probably adding to the 'shifted baseline' that has occurred in the Firth over decades, associated with the historical changes in trophic state. Conley et al. (2009) and Kemp et al. (2005) have shown that recovery from such changes can be slow and unpredictable. Regime shifts to alternative stable states can occur in response to extreme events – such reactions to intense oxygen minimum events have been documented in Danish Straits and Chesapeake Bay (Ibid.). Conceivably, this could have occurred in the Firth, with persistently lessened water quality (in terms of increased nutrients and phytoplankton) over the last decade. One of the largest oxygen minimum events was in autumn 2005, with <40% saturation. Another event occurred in 2007 with 40-60% saturation for about 4 months. Overseas examples (Chesapeake Bay, Danish Strait estuaries, Gulf of Mexico) show that relatively sudden events have led to systemic changes in oxygen conditions (Conley et al. 2009). In the case of Danish Straits, an identified change point occurred following a particularly strong hypoxic event. These are examples of non-linear responses to pressures that can resist return to the original state, upon remediation (hysteresis: Muradian (2001)).

Catchment nutrient load reduction (of both inorganic and organic N) would be expected to reduce organic loading to the Firth system (external and internal). However, it is not yet known what load reduction this would require. It is possible that changes to the internal N processing (denitrification efficiency) within the Firth is driving water quality change. If this is the case, remediation through load reduction may not be immediate. Understanding the cause of N enrichment over the last decade is important for assessing future risk, and capacity for remediation – does it represent a shift in the ecosystem's capacity to assimilate nitrogen?

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