David P. Hamilton · Kevin J. Collier
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Lake Restoration Handbook

A New Zealand Perspective

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In Memoriam: John Quinn, PhD, August 5, 1957–November 13, 2018

John M. Quinn took on the editorial role of this book with the renowned enthusiasm and competence that were the hallmarks of his involvement in scientific research. He was still editing chapters in the book a few days before his death, but he was also a passionate family man, musician and artist.

John was strongly dedicated to improving lakes and rivers in New Zealand. In 35 years, he progressed through the National Institute of Water and Atmospheric Research (NIWA) organisation to be the Freshwater and Estuaries Chief Scientist. During this time, he carried out important research in land–water connections, much focused on two important catchments, Whatawhata and Tukituki, in the North Island of New Zealand. In the Whatawhata catchment study, he worked with farmers to adopt new farming methods to improve stream water quality while maintaining profitable farming systems. Similarly, in the Tukituki catchment, he worked with the local community to try to find ways to increase stream health under intense pressure from increasing agricultural production. John was passionate about these projects, in part because they connected him to his roots—he was from a farm in the Waipukurau and as a child he swam in the local Tukituki River.

In 2016, John led a NIWA team that worked with the Waikato River Raupatu Trust, a Māori organisation responsible for implementing the Waikato River Settlement with the Crown on behalf of Waikato-Tainui iwi. A report card was developed for the Waikato and Waipa rivers. The report card received international acclaim and has been an important part of the engagement of indigenous tribes in the restoration of these two rivers.

John has worked with successive governments through the Ministry of Environment and the Ministry for Primary Industries to provide advice on freshwater issues in New Zealand. He was much respected for his impressive scientific publication record, balanced perspective and ability to actively engage with a broad cross section of the community to communicate complex scientific topics. John's methods were not always orthodox; his musical talent was sometimes used to "break the ice" and begin presentations with self-composed songs on the subject area of his presentation. Ultimately, he sought to bring focus to resolve some of the problems of land use and freshwater in a country where agricultural production is a critical component of the economy, but citizens value freshwater above all other resources. John loved the challenge of addressing diffuse source pollution, having seen the transition over past decades of removal of point source pollution from rivers throughout New Zealand. His PhD research had examined one of New Zealand's most polluted rivers, the Manawatu, where direct discharges of effluent from treated sewage, dairy factories and meat works were once commonplace.

In 2012, John won NIWA's Applied Science Excellence Award for using freshwater research to guide policymakers in management of water issues. He had previously won the Royal Society Bronze Award for his efforts in communicating science. In 2018, shortly before his death, John was awarded the highest accolade of the New Zealand Freshwater Sciences Society (NZFSS), the Medal of Excellence for outstanding contributions to freshwater science, recognising his lifetime contribution to research into improving stream health, helping to develop freshwater policy and more generally assisting in natural resource management. John left the world earlier than he should have but made a rich contribution to freshwater management for the benefit of all New Zealanders.

Foreword

In 1987, what was then New Zealand's national water and soil conservation agency published a *Lake Manager's Handbook*. I edited it, and the then Deputy Chairman of the agency, Basil Parkes, wrote the foreword. Some 20 government and university scientists and planners contributed the material, with a similar number of our colleagues providing peer reviews of it. Looking at the names of those involved, I see that while some have moved on and others have died, about one-third appear to be still active in freshwater science. And I gather that most of the copies of the handbook that are still on people's shelves are looking well-thumbed and a bit battered—not to mention being increasingly out-of-date.

So 30 years later, it's a pleasure and privilege for me to be asked to write the foreword of this welcome successor to the original handbook—the *Lake Restoration Handbook*. Professor David P. Hamilton arrived at the University of Waikato in 2002 and soon established an internationally respected centre for lake science there. Early on, he was successful in securing long-term government funding for the centre's studies; I was a member of a peer review committee established at the time and clearly recall that a key goal of the funders was the production of a new handbook. And this is it. In producing this new book, David and his co-editors have brought together nearly 40 of New Zealand's best scientists to pool and share their knowledge of the science underpinning lake restoration; three of these—Max M. Gibbs, Clive Howard-Williams and David Rowe—also contributed to the 1987 handbook, and I salute their dedication and perseverance.

It seems to me that there are three key areas where the new handbook clearly updates our understanding of our lakes and their management. Firstly, several chapters in the first part of the book describe how important catchment processes can be expected to change under alternative management regimes; they also describe how our ability to understand and predict these changes has improved with successive generations of computer models of lakes and their catchments. Secondly, although the original handbook did touch on the management of biological components of lake ecosystems—including phytoplankton, macrophytes and fish—the management of lake water quality was a major focus. In the new restoration handbook, however, biodiversity has become a dominant theme, with approaches to the control of invasive species and the re-establishment of native species being reviewed and evaluated. Finally, the handbook describes the development of several new technologies, ranging from the molecular level (environmental DNA) to the use of moored water quality sensors in lakes and remote sensing by satellite.

Changing social and cultural perspectives in New Zealand are also described in the restoration handbook. And these have undoubtedly played a part in the widespread and growing calls from the community to halt the deterioration of water quality here. The joint central and local government plan to protect the water quality of our largest lake, Lake Taupo, is a striking example of how we are able to respond to these calls. Much of the science underpinning this groundbreaking initiative had been described in the original Lake Managers Handbook. So I believe we can be confident that the *Lake Restoration Handbook* will also underpin future efforts to protect and enhance our many sadly-degraded lakes.

Can I finish by echoing Basil Parkes' blessing of his first handbook: May your use of it be a success.

Hamilton, New Zealand W. N. (Bill) Vant
March 2017

Preface

Freshwater is probably the most impacted category of natural resources on the planet. It bears the legacies of past human activities which include damming of rivers, intentional introductions of exotic species, point-source pollution and loss of connected ecosystems such as wetlands and peatlands. It must now be resilient in the face of new and emerging challenges as humans—mostly unintentionally—transport increasing numbers of exotic species to new freshwater habitats and export increasing diffuse-source pollutant loads as a response to increasing human populations and a drive for increasing agricultural productivity. Climate change looms as perhaps the greatest challenge of all, potentially changing lake mixing patterns, affecting the availability and quality of fluvial waters and altering habitats for organisms that have previously adapted to environmental change over time scales of centuries to millennia. These globally pressing issues are captured through Boxes contained within the chapters of this book. Authors from across the globe have contributed to these Boxes, which provide different perspectives on lake restoration related to their areas of subject specialisation.

Lakes are recognised as sentinels for human impacts. Unlike fluvial waters, water residence times of most natural lakes, and even many artificial lakes, allow their physico-chemical and biotic character to progressively approach an equilibrium condition in response to environmental forcings, sometimes only being substantially disturbed by a new species introduction or a major storm event. The composition and the organisms that lakes support therefore represent a complex integration of climatic, hydrological, geochemical and biotic forcings. The indigenous Māori people of New Zealand refer to natural lakes as tāonga (treasures), with a mauri (life force) that can be lost or degraded if it is not respected and nurtured. These viewpoints are increasingly enshrined into New Zealand environmental law; in 2017 the Whanganui River was given the legal status of a person—indivisible and a living whole being extending from the mountains to the sea—under a treaty settlement between the Crown and the local Māori tribe. A variety of settlements between the Crown and Māori tribes (e.g., lakebed ownership ceded to Māori) have taken place in recent years, and many such lake settlements are now mooted across

New Zealand. These settlements have provided a major impetus for lake restorative activities.

Lake restoration practitioners must now connect across local and international scales, that is, from local communities who may seek cultural redress as well as improved lake water quality, ecological status and amenity value, to international projects whose discoveries are leading to a revolution of new and accessible monitoring tools and greater diversity of in-lake treatment methods (e.g. geoengineering) to address nutrient legacies. In between is a complex array of environmental, social and policy imperatives. In New Zealand, these are captured in a framework known as the National Policy Statement for Freshwater Management (2014), which directs regional operational units (regional councils) to meet the water quality objectives derived from a collaborative community consultation process. The structures, policies and natural and social science disciplines involved in lake restoration reflect the complex and highly interdisciplinary nature of implementing successful restoration programmes. This book provides chapters which cover these subjects and provides numerous New Zealand case studies, as well as examining citizen participation and Māori roles in lake restoration. Many of the issues raised, the impediments encountered and the solutions are not unique to New Zealand, as reflected in the close connection between the chapter content and the Boxes contained therein.

This book provides a contemporary perspective and update on the Lake Manager's Handbook published in 1987. The Lake Manager's Handbook focused on implementing monitoring programmes and managing lakes in New Zealand as a resource (e.g. for hydroelectricity and trout fisheries). In the intervening 30 years, environmental pressures have intensified, and there has been a strong and enduring desire expressed by the New Zealand public to do better: that badly degraded lowland lakes must be restored and many other lakes urgently need to be protected. The New Zealand Resource Management Act (1991) provided the impetus to address much of the point-source pollution to New Zealand lakes in a manner not dissimilar to the United States' Clean Water Act (1992). Diffuse pollution from agriculture and urbanisation are more complex and nebulous, however, with major changes at the river basin scale also likely to have economic repercussions. The European Water Framework Directive (2000) and the New Zealand National Policy Statement for Freshwater Management (2014) seek to manage diffuse pollution at the river basin scale, but there are still relatively few working examples across the globe where addressing diffuse (as opposed to point-source) pollution has immediately brought about major improvements in water quality and ecological status of lakes. Lake ecosystems can have long response times that relate to a variety of human and natural factors. Lags can include the time required for policy and plan development, to address catchment and in-lake legacies of sediments, nutrients and exotic species and, in some cases, to transitioning away from a resilient degraded state after an ecological tipping point has occurred. The final chapter chapter in this book attempts to synthesise the challenges of lake restoration and makes recommendations on how to improve the quality of decision-making and the implementation of actions. This book in totality represents an important collation of contemporary knowledge of lake restoration which can be applied towards lake ecosystem restoration.

Brisbane, Australia	David P. Hamilton
Hamilton, New Zealand	Kevin J. Collier
Hamilton, New Zealand	John M. Quinn
Christchurch, New Zealand	Clive Howard-Williams

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David P. Hamilton is Deputy Director at the Australian Rivers Institute at Griffith University. He held the Bay of Plenty Chair in Lake Restoration at the University of Waikato, New Zealand for 15 years and led a major lake restoration programme designed to provide the underpinning research to address algal blooms and incursions of invasive fish. This programme provided the impetus for the current book. Hamilton is an associate editor on two international journals and has led several special issues of scientific journals. He has also previously been a faculty member at the University of Western Australia where he developed and applied models of lake water quality.

Kevin J. Collier is a freshwater ecologist at the University of Waikato who has worked across the science-management interface for over 30 years. He has previously co-edited books synthesising ecological knowledge of New Zealand freshwater invertebrates (in 2000) and the country's longest river, the Waikato (in 2010). He is associate editor on two international journals.

John M. Quinn (deceased) was an aquatic ecologist who was Chief Scientist for Freshwater and Estuaries at NIWA since 2015. Over the previous 20 years, he led cross-institute, interdisciplinary, research programmes on river, lake and estuary restoration, river ecosystems and land use interactions and managing forest harvest impacts on streams. He was a guest editor of two special issues of scientific journals focused on the outputs of these programmes.

Clive Howard-Williams is currently Chief Science Advisor (Natural Resources) at New Zealand's National Institute for Water and Atmospheric Research. He has published widely on freshwater ecosystems from tropical to polar regions, and has led research programmes on wetland, lake and estuarine ecosystems. He has also been involved in consultancy work on freshwater restoration. Clive has been the guest editor of several books and journal special issues related to aquatic ecology.

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Chapter 1
Context for Restoration

John M. Quinn, Kevin J. Collier, Clive Howard-Williams, and David P. Hamilton

Abstract New Zealand has made significant contributions to scientific knowledge underpinning lake restoration work and to the development and testing of management tools. This introductory chapter provides an overview of the history, current context, and policy framework for lake restoration in New Zealand and makes international comparisons. The book draws on recent advances in modelling and monitoring tools, and highlights how improved understanding of lake processes and their interaction with biological communities applies to restoration. Restoration solutions must address catchment as well as in-lake issues, and the socio-cultural considerations which influence the development, implementation and monitoring of lake restoration actions. Accordingly, Chaps. 2–16 are divided into five main sections: Management and modelling, Water quality restoration, Biodiversity restoration, Monitoring and indicators, and Socio-cultural considerations (including the roles of indigenous people and citizen science). New Zealand perspectives are complemented by international feature boxes which reinforce key concepts and highlight differences or similarities in restoration approaches. The feature boxes

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Brian Moss died soon after this contribution. The authors acknowledge his immense contributions to lake restoration and his encouragement in the preparation of this book.

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generally emphasise the ubiquity of the threats to lake ecosystems across the world but also the need to adapt restoration to the individual lake scale.

Keywords Lake management · Lake modelling · Water quality · Biodiversity · Monitoring · Indicators · Indigenous knowledge · Mātauranga Māori · Citizen science · Geochemical techniques · Lake destratification · Lake mixing · Invasive species management · Native fish · Remote sensing · High frequency monitoring · Ecological integrity · Genomics

1.1 Introduction

New Zealand lakes, like lakes across the globe, are subject to multiple pressures that continue to increase in severity and scale as land use intensifies, invasive species spread, and the global climate changes. This intersection of pressures elicits complex cause-effect pathways that can be challenging to address through management, especially given the significant lags and legacies that may constrain gains in water quality and biodiversity outcomes. Increasing awareness of the rate of water quality and biodiversity decline in New Zealand (Hughey et al. 2013; MfE and StatsNZ 2017) has focused attention on restoration of lakes which provide highly valued social, cultural, and economic resources.

The term 'restoration', applied in an ecological sense, is generally accepted to mean:

> The process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (Society for Ecological Restoration 2004).

Inherent in this definition is the assumption that environmental degradation is a reversible process that can be managed, engineered, or biomanipulated in some way to achieve a desired self-sustaining endpoint, although how this endpoint is defined can be a matter of considerable debate and the results of many projects have been disappointing (see Feature Box 1.1). Ecological endpoints may include pristine (minimally disturbed) reference states, some historic condition, a least disturbed condition taking account of pervading pressures, or a best attainable condition achievable within the constraints of human-dominated landscapes (Stoddard et al. 2006). Monitoring should be an essential component of lake restoration and rigorous statistical design should be woven into the restoration plan (see Feature Box 1.2).

Box 1.1 The Philosophy of Restoration: A Need for Ambition

Brian Moss
University of Liverpool, Liverpool, United Kingdom

'Restoration' of lakes and rivers has become common as freshwaters have suffered severe damage from farming and urban development in their catchments, river engineering, and introduction of alien species. The results of many projects have been disappointing, especially in terms of increase in biodiversity. Most have been adequate in terms of re-creation of geomorphological features or improvement of appearance, but in many, conditions have reverted, and most have not been properly monitored (Bernhardt et al. 2007; Gulati et al. 2008). Success is often assumed; awards are given, but the ecological gain is doubtful. Most projects are very small (a lake of a few hectares, at most a few km of river length); often only one, particularly phosphorus concentration or physical reconstruction, of several mutual impacts, is tackled; the land use of the catchment is largely unaltered and the aspirations of the project are subject to political limitations. Multiple use of the water is required, and multiple use generally means mediocrity in all of its several respects. Freshwater restorers are sidelined as servants of the developers. Since the 1960s, we have realised that living systems, through element cycling, maintain conditions in the biosphere that are favourable to our particular biochemical system. Why this should be so is controversial (Tyrrell 2013) but the fact is undisputed. Cycles of carbon and oxygen are particularly important, because of the roles that these elements play in determining climate. Carbon storage in lake and ocean sediments, wetland peats, and soils is very large and particularly important in maintaining relatively low carbon dioxide levels in the atmosphere. Between 50 and 75% of natural land biomes have been converted to agriculture or rangeland (Ellis 2011), and the carbon storage of such anthrome systems is negligible. The loss of natural biomes is reflected in the current inability of those that remain to prevent steadily increasing atmospheric carbon dioxide concentrations. All geoengineering proposals to cope with climate change are yet to be invented or potentially very risky. Earth surface temperature will not fall until carbon emissions are matched by carbon storage. Our current failure to curb emissions thus puts great emphasis on restoration of carbon-storing biomes and this requires attention to entire catchment systems and a system of zonation of excellent use (biome and anthrome) rather than of multiple mediocrity. We will need to restore biomes to the extent of around one third (see Fig. 1.1) of the existing anthromes (Moss 2014) if carbon emissions are not reduced by huge extents. This will require much greater cooperative ambition among all ecological restorers, a recognition that freshwater and land restoration are mutually inseparable, a grand political strategy to release the land necessary, and modifications of food production, distribution, and wastage, to use less land. But if climate change is as great a threat as it currently appears, we have no choice.

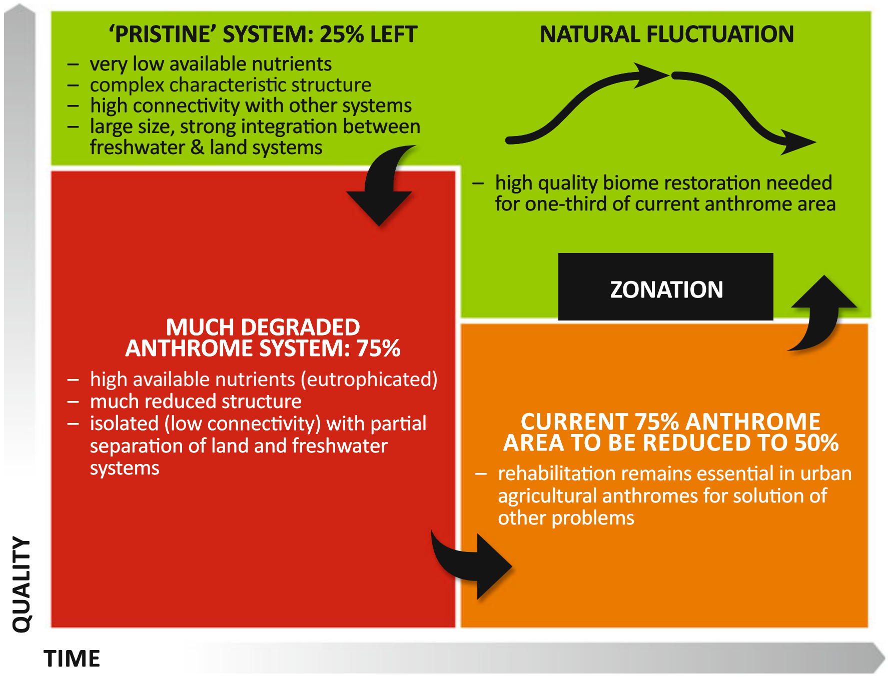

Fig. 1.1 Relationship between quality of habitat, time elapsing, and the nature of restoration. Green areas represent existing and proposed 'pristine' biome. Inverted commas indicate that nowhere do truly pristine habitats now exist, only close approximations to them. Red area indicates the current extent of anthromes. Orange outline represents conventional rehabilitation of the anthrome area. Indications are that one-third of this area needs to be fully restored to 'pristine' biome to mitigate climate change

Box 1.2 The Key Roles of a Monitoring Programme for Lake Restoration

Tamar Zohary
Kinneret Limnological Laboratory, Migdal, Israel

Lake restoration is an ambitious goal. Expensive operations are usually required, but not always do these operations lead to the expected outcome: at times lake restoration efforts result in only partial success or no positive outcome at all. Some lakes undergo restoration at huge expense, but no money is set aside to initiate and maintain a long-term monitoring programme to follow up the outcome. With little or no background or follow-up data, it is practically impossible to assess restoration success in objective terms. In cases of only partial or no success, lack of data will make it extremely difficult to

(continued)

Box 1.2 (continued)
decide whether further steps are required and, if so, which should be undertaken. Monitoring programmes are essential components of lake management in general (Sukenik et al. 1994) and lake restoration in particular. 'Before' and 'after' restoration monitoring data provide a basis for decision-making. 'Before' data are required for decisions about the desired changes to the ecosystem, the restoration objectives, and the design of the restoration plan. 'After' data are required for assessing the success of restoration and for following the behaviour of the restored ecosystem over time, to ensure that the improvements achieved are sustained. The longer the duration of the 'before' dataset, the better it is; basic monitoring has to begin 2–3 years before restoration begins; 1 year is too short. Similarly, the longer the 'after' dataset the better it is; minimally, it should span over twice the time that is relevant for observing the expected results. Considerable thought and expertise has to be invested in the planning of a monitoring programme. The planning should begin by specifying well-defined and tractable questions that will highlight clear short- and long-term objectives. These questions will then assist in agreeing on which parameters to monitor, where, at what time intervals, using which methods, and by whom. A rigorous statistical design should be woven into the plan. Responsible scientists should be allocated to oversee the programme and interpret its results. Monitoring activities include sample collection, laboratory analyses, data quality assurance, and data storage, organization, and analyses. The many parameters that affect the physical, chemical, and biological nature of the water body should be considered, although not necessarily all of them have to be monitored (Meybeck et al. 1996). Too often monitoring has been 'planned backwards on the collect now (data), think-later (of useful questions) principle' (Roberts 1991). Adopting the Adaptive Monitoring scheme of Lindenmayer and Likens (2009) is recommended. A crucial component of a monitoring programme is the researcher(s) who will be responsible for collecting and analysing the data. The scientist(s) in charge should examine the data regularly, be critical about its quality, be curious and ask questions emerging from the results, design research to answer those questions, apply for funding and conduct the research, come up with new insights and advise managers on how to manage the water bodies. In the long term, a well-planned and well-functioning monitoring programme will prove an invaluable asset in itself.

The notion of self-sustainability of a restored state, while desirable, is often difficult to achieve in lakes and is better considered as minimal management intervention. However, it must be recognised that there is a difference in fundamental approaches to lake management versus lake restoration. Lake management generally considers means of preventing (or slowing) deterioration through mitigations in the lake's catchment, engineering options, or biomanipulation. For instance, managing lakes to prevent harmful cyanobacteria blooms may require a combination of nutrient reduction, destratification, engineering (algal harvesting, diversions), and/or biomanipulation (Hamilton and Dada 2016). The most important difference between lake management and lake restoration is one of goal setting to not only manage deterioration but also, in the case of restoration, to reverse the trajectory to achieve some ecosystem recovery.

New Zealand has mapped approximately 3600 lakes >1 ha and, of those routinely monitored lakes, at least 45% are now considered eutrophic or hypereutrophic (Verburg et al. 2010) with potential for toxic cyanobacteria species to proliferate in extreme cases (Hamilton et al. 2014). Agriculture is the dominant modified land use and has had the most widespread impact on lake water quality through inputs of nitrogen, phosphorus, sediments, and faecal organisms (Abell et al. 2010; McDowell et al. 2013). Nutrients affect lake water quality by stimulating growth of primary producers, reducing water clarity and inducing loss of oxygen from bottom waters of deeper lakes, ultimately leading to accelerated in-lake nutrient cycling, catastrophic loss of macrophytes, and accelerated eutrophication causing regime shifts (Rowe 2004; Schallenberg and Sorrell 2009). Adding to this complexity is the use of many large New Zealand lakes, both 'natural' and artificial, for generating around 60% of the country's electricity. Hydropower affects water level variability, sediment regimes, connectivity with downstream waterways, and water quality (Young et al. 2004). Furthermore, New Zealand is considered one of six global hotspots for fish introductions, with non-indigenous species comprising over 25% of all freshwater fish species, many preferring to live in lakes (Leprieur et al. 2008). Additionally, 45% of aquatic plants are non-native (Johnson and Brooke 1998) and often out-compete native macrophyte species. Excessive invasive plant growths can become surface reaching and fill the water column of shallow lakes to the extent that weeds can become self-limiting, leading to deoxygenation of the water and plant collapse (Champion et al. 2002; Schallenberg and Sorrell 2009).

Restoration of lake quality requires an integrated strategy that addresses not only contaminants, in point-source wastewaters, urban storm waters, diffuse rural inputs, and industrial atmospheric fall-out, but also in-lake legacies from past pollution, lake-margin habitat condition, invasive species control, and connectivity pathways for migrating native biota (Hamilton et al. 2010, 2016). The spatial extent of interventions may range from whole catchment (e.g. lower-catchment wetlands) to property level (e.g. on-farm wetlands) and from whole-lake (e.g. lake sediment capping, biomanipulation) to habitat-specific (e.g. habitat creation) interventions. Most approaches to mitigating catchment issues as part of lake restoration involve: (i) land-based treatment of contaminants at source; (ii) interception of contaminants along hydrological pathways; and (iii) bottom-of-catchment methods that treat contaminants within receiving waters. Lake-margin habitat restoration may involve fencing, planting, or creation of wetlands, while impediments to connectivity require provision of fish passage at barriers. Control of invasive species, once established, can be more problematic, however, and may require targeted strategies that limit access to fish spawning habitats, for example (*see* Collier and Grainger 2015).

The wide range of origins, trophic states, catchment land uses, sizes, morphologies, hydraulic residence times and invasive species in New Zealand lakes requires targeted restoration strategies that are tailored to individual sites or types of lakes (Hamilton et al. 2016). Background information is critical to predict how each lake will respond to a specific intervention strategy, the magnitude (and costs) of the action that might be required, and the anticipated timescales of response. As part of developing restoration strategies, cause-effect pathways also need to be clearly resolved to identify factors that may constrain or delay water quality and ecological responses to restoration.

Disentangling causal factors in complex multiple stressor environments requires research and use of prognostic tools such as modelling to support lake restoration planning and outcome forecasting. New Zealand has made a significant contribution globally to the development and testing of such tools (see Table 1.1) and to scientific knowledge on lake restoration generally. This book provides an up-to-date synthesis of tools for, and approaches to, lake restoration applied in New Zealand with a view to enhancing future lake efforts. First, we briefly review the history of lake research that has supported recent restoration initiatives in New Zealand and describe the policy context for lake restoration.

1.2 History of Lake Research in New Zealand

In 1991, Professor Carolyn Burns reviewed 25 years of lake research in New Zealand (Burns 1991). The major emphasis of work up to that time had been on lake trophic state, with lake ecosystem deterioration and eutrophication forming a large number of papers in her review. A major finding was that, by world standards, New Zealand lakes were generally low in nitrogen. The potential for greater nitrogen limitation of phytoplankton growth in New Zealand lakes placed a different perspective on lake management than that advocated in northern hemisphere countries at the time. The management implications of much of the New Zealand research up until the early 1990s was, therefore, on management of nitrogen to halt or prevent deterioration (e.g. Vincent et al. 1984; Pridmore et al. 1985; Pridmore 1987; Vant 1987); restoration was not a major concept for the management of New Zealand lakes. In fact, there was only one such reference in Burns' (1991) comprehensive review of lakes research—to the restoration of marginal wetland vegetation.

In the 1970s and 1980s, it was recognised that invasions of non-indigenous aquatic weeds had caused significant deterioration in the native condition of New Zealand lakes (Coffey 1987; Howard-Williams et al. 1987), but the latter authors acknowledged that it was the effects of the invaders on human values (primarily hydropower, recreation, aesthetics) that were the principal cause of concern. Calls for aquatic weed management, therefore, took a lake management perspective rather than a focus on restoration. Howard-Williams et al.'s (1987) review did not consider restoration.

The first New Zealand review of aquatic restoration to include lakes (Collier 1994) pointed to the value at that time of New Zealand's recently promulgated

Table 1.1 Case studies of lake management in New Zealand. Adapted from Gibbs and Hickey (2012) and McDowell et al. (2013)

Stressor(s)	Strategy	Function	References for New Zealand
Multiple	Inflow diversion	Diverts nutrient-rich lake inflows directly to the lake outflow	Scholes and McIntosh (2010)
Nitrogen and phosphorus	Weed harvesting	Removes nutrients assimilated in excess weed growth	BoPRC et al. (2007)
Multiple	Hypolimnetic siphoning	Removes poor-quality bottom water of stratified lakes	McIntosh (2004)
Multiple	Dredging	Removes nutrients and sediments from the lake bed	Faithfull et al. (2006), Miller (2006)
Phosphorus (and nitrogen secondarily)	Sediment capping	Provides a capping layer to decrease nutrient releases from lake bed sediments	Hickey and Gibbs (2009), Özkundakci et al. (2010), Hamilton and Landman (2011)
Multiple	Increased flushing rate	Creates sufficient through-flow to physically remove phytoplankton before substantial growth response to prevailing nutrient concentrations	Howard-Williams et al. (1987), Hickey and Gibbs (2009)
Phosphorus and sediment	Wave barriers	Reduces resuspension of sediments and nutrients in shallow lakes through a physical barrier to reduce surface wave propagation	Jellyman et al. (2009), Gibbs and Norton (2013)
Nitrogen (primarily) and phosphorus	Floating wetlands	Uses wetland plants to take up nutrients, wetland environment to remove N via denitrification	Tanner and Headley (2011), Sukias et al. (2010)
Phosphorus	Phosphorus inactivation or flocculation	Uses chemicals (e.g. aluminium sulphate) to 'lock up' dissolved phosphorus in lakes via adsorption and precipitation processes	Paul et al. (2008), Hickey and Gibbs (2009), Hamilton and Landman (2011), Özkundakci et al. (2010), Scholes and McIntosh (2010)
Phosphorus (secondarily nitrogen, phytoplankton)	Oxygenation, destratification, or mixing propellers	Pumps air to the bottom of lakes to decrease redox-mediated nutrient releases	Howard-Williams et al. (1987), Hickey and Gibbs (2009)

Resource Management Act 1991 which required 'avoiding, remedying, or mitigating any adverse effects on the environment'. This Act, therefore, provided considerable scope for habitat restoration. Two case study lakes in Collier (1994) were related to restoration: contamination by sodium arsenate from weed-spraying in

Fig. 1.2 Lakes Rotoroa (photo: John M. Quinn) and Parkinson (photo: Tracey Edwards) that were the focus of early restoration studies in New Zealand

urban Lake Rotoroa in the North Island city of Hamilton (Fig. 1.2) (Clayton and de Winton 1994) and biomanipulation of invasive plants using fish in Lake Parkinson, northern North Island, (Rowe and Champion 1994). Subsequent update of the Lake Manager's Handbook in 2002 provided a follow-up series of reports intended to fill information gaps and synthesise recent research findings on invasive species (Champion et al. 2002; Elliott and Sorrell 2002; James et al. 2002; Rowe and Graynoth 2002).

The book Freshwaters of New Zealand, edited by Harding et al. (2004), provided the first national-scale review across all freshwaters that included chapters covering not only descriptions of lake state, ecosystem community structure, and lake management options, but also a chapter on lake restoration by Rowe (2004). Management options discussed included impacts of land use as they affected nutrient and sediment loss to streams and lakes (Parkyn and Wilcock 2004), with the emphasis on mitigation and prevention. Rowe (2004) identified six main causes of lake deterioration in New Zealand (eutrophication; increased turbidity; non-native plant invasions; non-native fish invasions; decline of fisheries; chemical contamination) and pointed out that before undertaking restoration it is necessary to identify the problems specific to each lake and their causes so that restoration goals can be set. He also emphasised the requirement for successful restoration to take into account a holistic approach that cuts across administrative boundaries for water management, lake bed control, recreational use, fishing, and customary uses, recognising that fragmented administrations with multiple boundaries can result in institutional (and legislative) barriers to restoration.

In a volume of the New Zealand Journal of Marine and Freshwater Research devoted entirely to restoration of aquatic ecosystems (Quinn 2009), Schallenberg and Sorrell (2009) reviewed lakes in New Zealand that had undergone regime shifts between macrophyte-dominated, clear-water states and human-induced de-vegetated, turbid states. Lake restoration in such degraded lakes was deemed tractable because of the strong correlations between lake state and both catchment land-use and presence of invasive aquatic weeds. In the same volume, Hickey and

Gibbs (2009) reviewed the implementation of lake management actions to reduce algal blooms and discussed these in relation to conditions in New Zealand's lakes. These authors provided a decision-support and risk management framework with strategies for reducing internal phosphorus loads to lakes and, therefore, reducing cyanobacterial bloom formation. In many cases, reductions in harmful algal blooms in lakes are seen as necessary early stages in restoration with an expectation that such reductions can be achieved by catchment or internal nutrient load reductions. Subsequently, Hamilton et al. (2010) discussed future prospects for the lowland lakes of the Waikato Region, northern New Zealand, in the context of both ongoing lake management and restoration and concluded that, to initiate regime shifts in degraded lakes, restoration will require concerted efforts to address high catchment and internal nutrient loads and prolific coarse fish populations.

1.3 Policy Framework for Lake Management and Restoration in New Zealand

Several recent policies promote the restoration of New Zealand lakes, increasing the need for this handbook. The Resource Management Act 1991 (RMA, New Zealand Government 1991) charges regional councils with responsibility for managing the environment, including lakes. Point-source discharges of nutrients, organic materials, sediments and harmful organisms are regulated under the RMA, and point-source inputs of these contaminants to most lakes have been reduced or removed under that legislation (e.g. Rutherford et al. 1996). However, the RMA was much less effective in addressing diffuse pollution from land use change and/or intensification, despite clear evidence that lake water quality and biodiversity have declined over the last three decades. The declines are in line with increasing diffuse inputs of nutrients from land-use intensification, particularly from agriculture (Hamilton 2003; Abell et al. 2010; Hamilton et al. 2010; Howard-Williams et al. 2011). Arguably, the decline in ecological integrity of lakes over the first 20 years of the RMA was facilitated by the delay in implementing its provisions for a National Policy Statement that set targets for freshwater water quality and ecosystem health.

The National Policy Statement for Freshwater Management (NPS-FM, MFE 2014) was introduced in 2011 and updated in 2014, building on reports of the consultative Land and Water Forum (2010, 2012a, b). It was designed specifically to arrest the decline of freshwater quality in New Zealand, with a major focus on diffuse pollution. It requires regional councils 'maintain or improve' water quality across 'freshwater management units'. This approach involves setting objectives for freshwaters to be achieved by limiting resource use (e.g. for nutrient loads to attain target lake condition). A key tenet of the NPS-FM is the National Objectives Framework (NOF, MFE 2014) involves setting values, attributes, and objectives for water quality so that bands (A to D) can be designated to freshwater management units. If a freshwater management unit falls into the D ('unacceptable') band, for example, then actions will be required to shift it out of this band and into a designated band defined by community aspirations (A, B, or C). Thus, the NOF places increased emphasis on lake restoration, particularly for the many lowland lakes that fail to meet the designated bands. These bands are not tied specifically to a reference condition (c.f. the European Union Water Framework Directive, European Union 2000) and, therefore, do not distinguish lakes on the basis of origin, morphology, or any other aspects of physiography. However, they do provide water quality targets for lake restoration projects. Implicit in their application is an improvement in biodiversity.

The Department of Conservation (New Zealand) is responsible for managing lakes on conservation land, while non-government organisations (e.g. Fish & Game NZ, Irrigation New Zealand, Waikato River Authority), tribal authorities, local councils and ratepayer groups, hydropower operators, and community groups (e.g. Rotorua Lakes Water Quality Society) can have varying degrees of influence on lake management strategies and objectives. In an increasing number of cases, specific lake management agreements have included Māori iwi (tribe) partners as the tāngata whenua (indigenous people) of New Zealand. The level of engagement of iwi and community in setting goals for freshwater systems, as envisaged under the NPS-FM, is now an important precursor to the implementation of rehabilitation actions especially compared with earlier considerations such as Rowe (2004).

In some areas, Māori have recently attained direct control or developed co-management arrangements over lakes through Treaty of Waitangi Settlement agreements. For example, in the Rotorua district, the regional council remains responsible for managing water quality, but the Te Arawa Lakes Trust is empowered by the Te Arawa Lakes Settlement Act 2006 to regulate the customary and recreational food gathering of indigenous fisheries species (e.g. kōura/freshwater crayfish, kākahi/freshwater mussels, and the fish species smelt, kōaro/galaxiids, tuna/eels). Iwi in the Waikato and Waipā catchments have co-management arrangements with Waikato Regional Council. The Waikato River Authority manages implementation of the Vision and Strategy for the Waikato River established by the Waikato-Tainui Raupatu Claims (Waikato River) Settlement Act 2010 (New Zealand Government 2010). The Vision and Strategy sets objectives to restore the health and well-being of the Waikato River (and lakes) and its people and underpins land and water policies in the Waikato Healthy Rivers Waiora Plan Change (WRC 2016). Together with a $220 million Clean-up Fund (over 30 years), this provides a strong need for lake and river restoration guidance.

Another set of policies promoting restoration of freshwater waterbodies provides funding mechanisms to restore degraded lakes or prevent iconic lakes from deteriorating further. The New Zealand Government's Freshwater Improvement Fund ($100 M over 10 years, MfE 2017) includes an additional set of clear eligibility criteria for funding support for freshwater restoration projects. Amongst these criteria is the requirement to manage within water quality and quantity limits and the need to clearly demonstrate environmental benefits. This funding provides a new impetus for restoration, together with treaty settlements between the Crown and Māori tribes. These settlements have allocated funding to restore water resources so that Māori may renew traditions of food harvesting and cultural practices, as well as benefiting communities in general.

In conclusion, a range of drivers and enablers has increased the urgency in New Zealand for this handbook on lake restoration.

1.4 International Context

While many of the issues confronting New Zealand lakes are similar to those in other parts of the developed world, the solutions applied here reflect the wide variability of New Zealand lake types present over small spatial scales, reinforcing the need for restoration to be tailored to individual lakes in order to be effective. Below we discuss some key differences and similarities between restoration approaches in New Zealand and internationally.

Murphy (2003) discussed lake degradation issues on a global scale that need to be incorporated in restoration approaches. These included ecosystem disruptions due to shipping and ballast water exchanges, acid rain, sulphur loading, and global toxicants such as some heavy metals and PCBs. New Zealand's isolation has kept our water bodies largely free from these pressures so approaches to lake restoration are essentially at local (catchment) scales, with eutrophication and aquatic biosecurity being the principal issues that need to be confronted in restoration planning. However, natural geothermal inputs into some lakes in New Zealand, particularly in the central volcanic region of the North Island, can create legacies that limit human use and biodiversity potential, particularly where hydro dams cause accumulation of mercury and arsenic in bottom sediments (Hickey and Martin 1996).

Jeppesen et al. (2003) stressed that biological restoration methods have been developed in Northern Hemisphere countries that have yielded major improvements in water quality and the ecological state of lakes. These include removal of planktivorous and benthivorous fish, stocking of piscivorous fish, protection and planting of submerged macrophytes, and introduction of artificial structures or addition of mussels. They stressed that these essentially northern hemisphere methods cannot readily be applied to sub-tropical or tropical lakes, and there is 'a major need for development and adaptation of methods focusing on south temperate, subtropical and tropical lakes' (Jeppesen et al. 2003). To date, such biomanipulation methods have had limited use in New Zealand (Burns et al. 2014), other than the stocking of grass carp (*Ctenopharyngodon idella*) to reduce macrophyte biomass (Hofstra et al. 2014).

Howard-Williams and Kelly (2003) identified three phases in the restoration of lakes from eutrophication: reduction or removal of point-source discharges, reduction of diffuse source discharges, and physical or biological manipulation. These authors also highlighted three major factors that reduce the applicability of general predictive models of the process of lake restoration following nutrient reduction: (1) the local geological context which determines the limiting nutrient(s) for plant growth and the extent of groundwater inputs to lakes; (2) the effect of regional or local climate (e.g. particularly wind speed) that affects lake stratification and internal loading; and (3) the local biogeography, such as regions that have no indigenous herbivorous fish (most New Zealand lakes) or where invasive plants modify lake ecosystems and compromise restoration goals.

On a more general note, Howard-Williams and Kelly (2003) argued that the local human community has to decide on what is a restored ecosystem. This decision depends on the regulatory framework, the cultural background of the community, its ability to afford restoration and its plans for waterbody use in the future. New Zealand's National Policy Statement for Freshwater Management 2014 provides a general framework for incorporating local community restoration aspirations above national 'bottom lines' set for nutrients and chlorophyll *a* (MFE 2014). In contrast, the European Water Framework Directive (European Union 2000) created general requirements for lake restoration to achieve 'good ecological status', independent of local interests, for which national water quality targets have been proposed (e.g. total phosphorus targets for deep and shallow Denmark lakes, Sondergaard et al. 2005).

Howard-Williams and Kelly (2003) suggested that failure to identify causes of lake deterioration, and hence failure to set realistic restoration goals, is a reason why restoration activities (and expenditure) sometimes fail to meet expectations. Communities undertaking restoration, particularly with regional government assistance, need to understand these issues. Moss et al. (2002) described a complete failure of a lake restoration project where, in spite of scientific advice and early restoration success, local competing financial and political interests interfered, and communication problems added to the failure of the restoration.

1.5 Introduction to Chapters

Research carried out on New Zealand lakes over the last 25 years operated in a climate of fluctuating funding, reflecting shifting priorities and variable appreciation of the complexities of lake management that science can help resolve (Prime Minister's Chief Science Advisor 2017). After several years of reduced funding, support was provided in 2005 for programmes of freshwater restoration research which have contributed much of the material presented in this book. The content of this book reflects recent advances in modelling and monitoring tools and better understanding of lake processes and their interaction with biological communities. The chapters and associated feature boxes acknowledge that restoration solutions must address catchment as well as in-lake issues and the rapid expansion of socio-cultural considerations which influence the development, implementation, and monitoring of lake restoration actions. Accordingly, the book is divided into five main sections: Management and modelling, Water quality restoration, Biodiversity restoration, Monitoring and indicators, and Socio-cultural considerations. The New Zealand perspectives presented in the following chapters are complemented by international feature boxes which reinforce key concepts and highlight

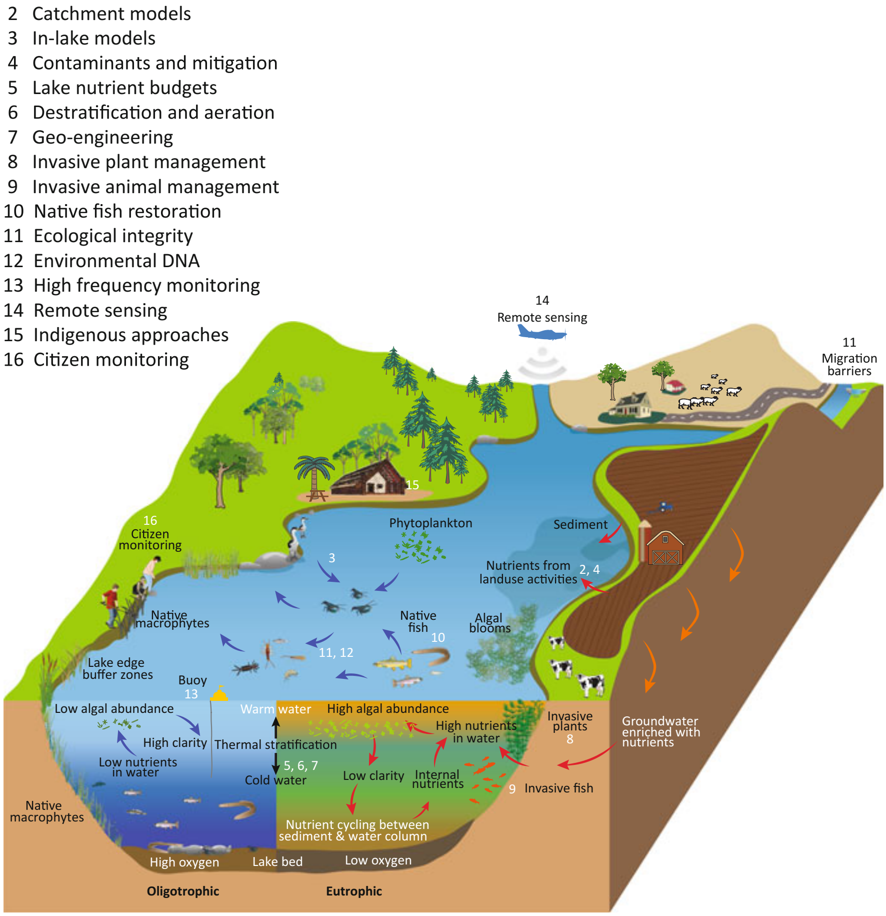

Fig. 1.3 Schematic of lake and catchment linkages and the numbered chapters of this book

differences or similarities in restoration approaches. Figure 1.3 provides a schematic of how the individual chapters contribute to the components of lake restoration.

The management and modelling chapters (Chaps. 2–3) cover the application of catchment and in-lake models to lake restoration.

Catchment models are key tools in lake restoration that describe and quantify the sources, transport, and fate of sediment, nutrients, and other constituents in a landscape. Chapter 2 describes how catchment models can be used to quantify and understand existing conditions, define areas with highest contaminant yields (hotspots, where actions would be most beneficial), and the relative importance of various sources (what types of actions would be most beneficial). It describes the continuum of catchment models, provides information for choosing specific models for various management applications, and provides examples of catchment models used to address a wide range of scientific and policy-driven issues in New Zealand and the USA. The widely used SWAT model is summarised in the international feature box.

In-lake models can support lake restoration by providing insight and understanding of lake processes and generating scenarios for hypothetical restoration actions or future environmental change that allow for more detailed mitigation investigations. Chapter 3 provides an overview of the types of in-lake models (empirical, neural network and deterministic), criteria for model selection and techniques for data assimilation, and assessing performance. It summarises the 40 applications of dynamic models to New Zealand lakes and presents example case studies. It concludes by providing a commentary on the ability of models to simulate aquatic biogeochemical variables and discusses how data assimilation provides a statistical framework for combining observations and models to improve their predictive capability. Two feature boxes discuss mechanistic lake modelling approaches in Europe and recent international advances in lake modelling.

Chapters 4–7 focus on catchment and in-lake water quality management for lake restoration. Managing contaminant losses from agricultural land use is often a key first step of lake restoration and relies on a good understanding of the sources of contaminants and the processes that facilitate and modify their downstream transport to the lake. Around the globe, focus has been brought to the mitigation of sediment and nutrients from agricultural land and especially so in New Zealand where greater than 50% of the land area is in agriculture. Chapter 4 provides an overview of contaminant transport pathways from agricultural land to water and the natural and manageable factors that modify these. The effectiveness of a range of mitigations on the land and at the land–water interface is discussed along with the findings of case studies involving catchment-scale application of bundles of mitigations. Feature boxes discuss assessing land cover changes with high spatio-temporal resolution to assist with catchment restoration and an international perspective on catchment processes affecting lakes.

A quantitative understanding of lake nutrient sources and sinks is essential for developing successful restoration options. Chapter 5 discusses how to develop this understanding through development of simple steady state lake nutrient budgets that account for sources, cycling and sinks of nutrients. Methods are reviewed, and examples are provided, for measurement and modelling of nutrient input budgets to lakes from streams, groundwater and the atmosphere, along with internal nutrient compartments (e.g. within macrophytes and fish) and fluxes, via flushing, denitrification, sediment nutrient release and sequestration. New Zealand examples of restoration interventions to reduce lake nutrient fluxes and stocks are discussed. The chapter concludes with a discussion of future prospects to improve simple steady-state budgets and extend them to include seasonal dynamics and better account for climate and climate change. A feature box on the Delavan Lake, Wisconsin, USA, case study provides an example of comprehensive catchment and in-lake approaches to long-term lake rehabilitation.

Managing stratification of lakes can be a powerful tool in restoration through manipulation of lake physical conditions that influence bottom-water deoxygenation, internal nutrient release from sediments, and phytoplankton growth. Chapter 6 describes the issues associated with thermal stratification and examines the possible options for removing or preventing thermal stratification by mixing as well as artificial destratification and bottom water re-oxygenation without destratification. Content in this chapter aligns with well-described actions often associated with engineering ‘solutions’ that require considerable capital investment and maintenance, often associated with high-value lakes where there are direct human uses (Cooke et al. 2005). Chapter 6 provides case study examples of destratification in New Zealand and evaluates their relative success. Flocculation and sediment capping are alternative in-lake restoration tools for control of phytoplankton and the recycling of legacy nutrients stored in the sediments, without managing stratification. Chapter 7 reviews the in-lake processes that underpin the effective use of these tools and discusses the application of a range of commercially available flocculation agents and passive and active sediment capping agents targeting phosphorus. Feature boxes discuss an international perspective on lake geo-engineering and review phosphorus geo-engineering binding agents.

Chapters 8–10 focus on restoration of lake biodiversity that is often a prerequisite to achieving desired water quality and is frequently the ultimate aim of restoration. The presence of invasive lake weed species invariably has a detrimental effect on native plant biodiversity, abundance, and depth range in the short term and on native plant seed bank in the longer term. Chapter 8 reviews the impacts of invasive plants and the range of tools for their control to enable restoration of native vegetation. These tools include habitat manipulation and/or weed control using biological, chemical, mechanical, manual, and integrated methods, as appropriate for the target weed species, lake characteristics, and restoration goals. Subsequent restoration of native vegetation can occur via passive regeneration of native plants from adjacent sites, seedbanks, and waterfowl-mediated dispersal, or actively through planting, and brings associated benefits of habitat for native fauna, with additional improved amenity and recreation values. Case studies provide practical examples of the application of invasive weed control and native plant reestablishment to meet restoration goals in lakes across New Zealand, and a feature box describes the effective use of V-blades and jute matting to control *Lagarosiphon* in Lough Corrib, Ireland.

Chapter 9 addresses the control or eradication of invasive freshwater animals, from zooplankton, through macroinvertebrates to fish. Eradication is often unachievable for invertebrate invaders so management focus needs to be on preventing their initial establishment. Control of vertebrate invaders is possible with ongoing effort, but complete eradication is typically difficult. Examples are discussed where fish eradication has been achieved, using piscicides and drainage, in New Zealand and overseas through concerted integrated management. Netting, trapping, electro-fishing, cages, baits, and one-way barriers have been used for fish control purposes in New Zealand, but the ecological outcomes have largely gone unmonitored. Feature boxes discuss experience in the Laurentian Great Lakes with successful management of invasive fish and wider invasive species threats posed by ship ballast water. The chapter concludes with discussion of emerging and future technologies for management and detection of non-indigenous fish and risks posed by climate change.

Restoring native fish communities is a common goal of lake restoration. Chapter 10 reviews New Zealand examples of native fish restoration in lakes, categorised by restoration method (viz. habitat improvement, connectivity restoration, predator control and translocation) and by species, and discusses the major characteristics of lake fish restoration efforts and the evidence for effectiveness. Methods used to restore native fish in New Zealand tend to be lake- and species-specific, because they are related to the specific vulnerabilities of the various species and to the type of lacustrine environment in which they occur. The legal framework also influences the emphasis of native fish restoration actions by providing stronger requirement for maintenance of natural connectivity than for habitat enhancement and control of invasive fish. Feature boxes summarise the Danish experience in lake restoration through stocking of predatory fish to control zooplanktivorous and benthivorous fish and two approaches for freshwater fisheries rehabilitation and management during times of rapid change: the Rehabilitation model and the Safe-operating Space concept.

Chapters 11–14 discuss the range of indicators and monitoring methods available for assessing lake restoration success.

Chapter 11 discusses the current development and future possibilities for using ecological integrity (EI) to assess the state of degraded lakes and help guide their restoration. It argues that the use of indicators of freshwater EI, incorporating the concepts of ecosystem 'health', unimpaired structure, composition and function, and a capacity for self-renewal, provides a holistic advance over standard water quality metrics for assessing lake condition. In the New Zealand freshwater context, EI has been defined as a composite of nativeness, pristineness, diversity, and resilience to perturbations and is closely aligned to the Māori concept of mauri or 'essential life force'. Measurable lake EI attributes have been proposed and calibrated against pre-impaired 'reference' conditions for different lake types. LakeSPI (Submerged Plant Indicators) is discussed as an example of a New Zealand method developed to assess lake ecological condition that has been calibrated for a wide range of New Zealand lakes. The EI approach measures departures from reference conditions (or other defined endpoints) making it beneficial for setting lake restoration goals or targets and for tracking restoration progress. Feature boxes discuss techniques for defining lake reference conditions and the range of methods used for assessing ecological condition of lakes across Europe.

The emerging use of molecular techniques, collectively referred to as biodiversity genomics, to support lake restoration by providing accurate assessment of species-level diversity is discussed in Chap. 12. Techniques such as DNA sequencing have allowed assessment of biodiversity to levels previously unattainable using traditional, morphological assessments alone. DNA barcoding uses small standardised fragments of DNA to identify species and has become an increasingly used tool for assessing changes in community composition following restoration efforts, often in conjunction with community metrics like the Rotifer Trophic Level Index. Next Generation Sequencing techniques enable multiple samples to be run simultaneously, greatly automating and streamlining the monitoring process. Developing reference libraries can be used to identify species ‘sight unseen’ through analyses of environmental DNA (DNA that is shed into the environment by plants and animals). This latter method has proven particularly useful for the monitoring of non-indigenous fish species, particularly following eradication efforts. These Next Generation Sequencing developments and further refinements of eDNA methods are predicted to produce a revolutionary change in biological monitoring of freshwaters that will enhance lake restoration monitoring. Feature boxes discuss the sensitivity and reliability of eDNA-based detection in aquatic restoration and a case study in the Laurentian Great Lakes.

Chapter 13 reviews how lake restoration can be enhanced through the use of high-frequency monitoring that applies advances in sensor and information technologies to enable autonomous measurement and analysis of the aquatic environment at ever-increasing spatial and temporal resolutions. Automated monitoring can complement and fill the gaps inherent in traditional sampling designs, detect rapid environmental change, and quantify transient processes that can have cascading impacts through time and space. Moreover, automated monitoring can inspire investigations and analyses that are impractical using more traditional methods. The chapter describes the fundamentals of automated high-frequency lake monitoring, including hardware and telemetry design, sensor types and measurement principles, maintenance requirements, and quality assurance/quality control of datasets required for robust monitoring and research of lake restoration. Examples are provided that demonstrate the value of high-frequency measurements and derived data products for analysing short- and long-term lake processes and underpinning restoration. Advances in high-frequency monitoring are being accelerated through the Global Lake Ecological Observatory Network (GLEON) with 60 lake observatories across 34 countries. Feature boxes further explore the roles of GLEON and the Networking Lake Observatories in Europe (NETLAKE) in advancing automated high frequency monitoring and enhancing lake monitoring and research globally.

Chapter 14 reviews the use of remote sensing (RS) techniques which gather information about a lake without direct contact to support lake restoration. Remote sensing vastly increases the temporal and spatial resolution of current monitoring techniques which typically involve collecting grab-samples at spot locations within lakes. Remote sensing of freshwater is generally passive, analysing reflected solar energy to characterise physical, chemical, and biological characteristics of water including temperature, turbidity and suspended particles, coloured dissolved organic matter, and chlorophyll *a* concentrations. For the RS of optically active water constituents, the method needs to be tailored to the lake size and optical complexity of the system, as well as the spectral and spatial resolution of the remote sensor. The general categories of sensors for space-borne remote sensing are described along with the factors determining the minimum lake size that can be monitored, the ability of the sensor to differentiate optically active constituents, and the analytical, semi-analytical, or empirical algorithms for RS of optically active water constituents. Feature boxes discuss contrasting uses of remote sensing: (i) the prospects for using satellite remote sensing for early warning of algal blooms in lakes and reservoirs as new satellite data become available; (ii) hyperspectral remote sensing using aircraft in the Chilean Lakes Network; and (iii) advanced in situ sensing technologies for remote sensing of deep lake processes in Lake Biwa, Japan, including autonomous underwater vehicles, remotely operated vehicles, side scan sonar, and multi-beam sonar.

Chapters 15 and 16 address two socio-cultural aspects of lake restoration that are increasingly recognised as crucial for success: indigenous values, knowledge and management, and citizen science.

Chapter 15 highlights that Āotearoa/New Zealand's founding Treaty of Waitangi forms the underlying foundation of the relationship between the government and the indigenous Māori people regarding freshwater resources, including lake restoration. While there is no 'one' Māori world view, there are principles and values that establish and reinforce identity, responsibilities, and rights to manage and use natural resources, including lakes. Consequently, co-designed lake restoration approaches that are grounded in Māori cultural practices, values and perspectives, and responsive to Māori aspirations are needed to produce beneficial outcomes to the participating indigenous community and strengthen existing community initiatives. This requires a commitment (by agencies and funders) to move beyond conventional understandings of who is 'qualified' to engage in lake research and restoration initiatives. Several case studies provide practical examples of the challenges and benefits of indigenous people's perspectives and practices in lake restoration. This chapter emphasises the need for a more holistic approach to lake restoration that recognises and empowers indigenous people to engage as co-governors, co-leaders, researchers, knowledge holders, and teachers. Truly collaborative lake restoration programmes will provide multiple roles for Māori, including the development and implementation of monitoring and evaluation approaches. An international case study is provided by a feature box on fisheries management by Shoshone-Paiute Tribes in Duck Valley Indian Reservation, USA.

Chapter 16 explores the growing role of citizen science in aquatic monitoring and restoration as the value of stronger relationships between the science community and public is recognised. Citizen science ranges from solely collecting environmental data to being fully engaged in project conception, design, and delivery. Case studies of lake citizen science in New Zealand and the USA are presented, along with a broader analysis of environmental citizen science in New Zealand. Community groups in New Zealand lead diverse environmental restoration projects but environmental monitoring is not generally prioritised. In contrast, a strong culture of volunteer water quality monitoring exists in the USA where programmes are designed to educate participants while also providing data for fundamental research and for government agency-led environmental decision-making. This is exemplified in the feature boxes on the 110-year history of the Lake Sunapee Protection Association of New Hampshire, USA, that has conducted citizen monitoring for the last 20 years, and the Florida LAKEWATCH established in 1991. Principles underpinning the development and implementation of long-term volunteer monitoring programmes are outlined. Stronger community participation in monitoring has the potential to improve both scientific and environmental literacy while building more complete data sets describing trends in freshwater resources, and it aligns with New Zealand Government's goal to increase public involvement in freshwater through participatory decision-making.

Finally, the future direction of New Zealand lake restoration is considered in a global context by synthesising the preceding chapters and critically evaluating opportunities.

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Chapter 2
Modelling for Catchment Management

Aroon Parshotam and Dale M. Robertson

Abstract Catchment models are useful tools to help describe and quantify the sources, transport, and fate of sediment, nutrients, and other constituents in a landscape. Results from catchment models are used to quantify and understand existing conditions and used in restoration efforts by defining areas with highest contributions (hotspots, where actions would be most beneficial) and describing the relative importance of various sources (what types of actions would be most beneficial). In practice, a continuum of models exists from simple empirical models to complex process-driven models, each requiring different types and amounts of information. Each of these models has its strengths and weaknesses, which should be considered when deciding which model to apply to a specific area. In many applications, a combination of models can be either coupled or run in series to help describe how nutrients and sediment are transported from the field to downstream receiving water bodies. In this chapter, we describe the continuum of catchment models that exist and provide information for choosing specific models for various management applications. We then provide examples of catchment models used to address a wide range of scientific and policy driven issues: two models commonly applied in New Zealand (CLUES and GLEAMS) and one model (SPARROW) applied to a large river basin in the United States (Mississippi River Basin).

Keywords Catchment modelling · Management · Regulation · Water quality models

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2.1 Introduction

Catchment models express a landscape process or combination of processes as a series of mathematical relations. These models are used for empirical prediction and often causal explanation (Shmueli 2010). In the catchment modelling discipline, mathematical models can take many forms but are usually developed in a manner that incorporates as much information about the process as is commonly accepted and is readily available. Catchment models are used to simulate specific aspects of the landscape describing and quantifying the sources, transport and fate of sediment, nutrients, or other constituents from where they originate or are applied on the landscape to some downstream receiving water body. Results from catchment models are not only used to quantify and understand existing conditions, but they are also used in catchment and downstream water body restoration efforts by defining areas with highest contributions (hotspots, where actions would be most beneficial) and describing the relative importance of various sources (what types of actions would be most beneficial). The process of developing and using catchment models is recognised as an important component of generating information necessary for catchment management and lies at the centre of water resources planning and management. Quantitative catchment models are among the best tools available for addressing water quality issues and are used extensively in research, management, and decision-making. The reliance on catchment modelling to assist in decision making is increasing in New Zealand and around the world to cope with emerging problems facing catchments and to exploit the many new data sources that have been developed with the onset of geographic information systems (GIS).

The term "catchment model" is a very broad term which can include groundwater and surface-water hydrological modelling, erosion modelling, fine sediment and nutrient modelling, and other constituent transport modelling. The emphasis of this chapter, however, is surface water quality models and primarily those dealing with sediment and nutrient transport. It should be noted that the term "catchment" is synonymous with the terms "watershed" and "drainage basin" (usually basin for short), which are commonly used in other countries to describe the land area contributing runoff to a stream or river, and that the term "sub-catchment" is synonymous with "sub-basin".

Catchment models can be powerful tools because they synthesise many different kinds of information and often allow the user to extrapolate data from a few monitoring locations to much larger unmonitored areas. Results from catchment models can help in making a variety of management decisions, including those related to developing nutrient-reduction and lake-protection strategies across wide geographic regions. Results from catchment modelling can advance our understanding of the relations between water quality, human activities, and the natural processes that affect water quality. Catchment models can project the consequences of alternative management strategies and planning or policy-level activities, which may substantially reduce the costs of managing water resources. Results from catchment modelling can also help guide decisions about future monitoring of streams, rivers, lakes, and estuaries that are highly vulnerable to environmental degradation.

Catchment models are used worldwide (see Feature Box 2.1). They can be used to (1) establish links between water quality and constituent sources (such as nutrients and sediment), (2) track the transport of constituents to streams and downstream receiving waters, such as lakes and estuaries, (3) assess the processes occurring that attenuate constituents as they are transported from land to streams and then to downstream water bodies, (4) identify parts of catchments that contribute the most sediment or nutrients, due perhaps to particular conditions, such as natural soils and sub-catchment slopes or anthropogenic factors, often referred to as "hotspots", (5) guide the selection of intervention measures (e.g. drainage wetlands or include buffer strips), and (6) provide information needed by other modellers to simulate existing and potential future water quality in downstream water bodies. Once calibrated, some catchment models can also be used to simulate the effects of changes in land management, land use, and climate. Modelling results are commonly incorporated into Integrated Catchment Management Plans (ICMPs) or used in deriving total maximum daily loads (TMDLs) to help make better decisions about land use, climate, water resources, and infrastructure.

Box 2.1 Catchment Modelling: Understanding the Land-to-Lake Connection

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Lake water quality may be strongly dependent on the properties of the surrounding catchment (Soranno et al. 1996). Establishing a land-to-lake connection through catchment modelling (eco-hydrological modelling) provides a means for quantifying nutrient losses to waterways and also helps identify critical nutrient source areas (Wellen et al. 2015). Catchment modelling may thereby help to better understand and predict effects of changes in climate or land use on receiving water bodies. Relevant management examples include assessment of the consequences of intensifying agriculture or urban development on nutrient exports to receiving waters. In practice, catchment modelling may aid decision making in relation to land-use regulation or quantification of mitigation measures required (in the catchment) to meet external nutrient loads targets or to achieve a given ecological state in a lake ecosystem (Nielsen et al. 2013). Catchment modelling has been widely employed across the world (Wellen et al. 2015), and several catchment models exist with varying complexities and conceptual approaches (see review of models in Schoumans et al. (2009) and Golmohammadi et al. (2014)). The Soil & Water Assessment Tool—SWAT (Arnold et al. 1998)—has been applied extensively in catchment management (see review by Gassman et al. (2007)). SWAT is a semi-distributed, physically based, river-basin scale hydrological model (Arnold et al. 1998; Neitsch et al. 2005)). It divides a catchment into sub-catchments (sub-basins) based on topography. Within each sub-basin, SWAT derives the unique combinations of land cover, soil type, and topographical slope into hydrologic response units (HRUs). In each HRU, hydrological and vegetation-growth processes are calculated based on the curve number rainfall-runoff method (Neitsch et al. 2005). SWAT takes into account

(continued)

Box 2.1 (continued)
surface runoff, evapotranspiration, percolation, lateral subsurface flow, flow through tile drains, fast and slow flowing groundwater flow, and river transmission losses. The SWAT model may be set up from free available data sources, e.g.:

- Land use: http://due.esrin.esa.int/page_globcover.php
- Soil properties: http://www.iiasa.ac.at/Research/LUC/External-World-soil-database/HTML/HWSD_Data.html?sb=4
- Digital elevation model: http://srtm.csi.cgiar.org/SELECTION/listImages.asp
- Weather: http://globalweather.tamu.edu/

The SWAT model operates on readily available data, which enables model applications for both well-monitored catchments (e.g. a Danish case: Thodsen et al. (2015)) and more data-sparse catchments (e.g. a Chinese case: Nielsen et al. (2013)). A recent development also allows SWAT model applications through the QGIS platform (Dile et al. 2016). The open source policy of QGIS and the ability to rapidly adapt input data into the model make SWAT the model of choice for a very broad user community.

Information from catchment models can be an important aid in guiding local decision making, such as specific management strategies (locations for stock management, fertiliser management, locations for riparian retirement, or change in land-use intensity), and can quantify the likely effects of the intervention measures. Models have been used in this way to understand catchment hydrology and constituent load in the stream network, evaluate impacts of development and urbanisation, determine the effectiveness of urban and agricultural detention ponds, evaluate the impacts of changing land-use management, evaluate erosion and deposition in streams, lakes and reservoirs, and provide guidelines to protect downstream waters.

Results from catchments models can be very important in driving national and regional actions. For example, New Zealand's National Policy Statement on Freshwater Management 2014 (NPS-FM 2014) and the Resource Management Act 1991 (RMA) provide the framework within which regional councils set regional policy statements and plan objectives to manage the effects of land-use practices on water quantity and quality. The RMA requires that applications for resource consent, authorisation for specific activities or uses of natural or physical resources, contain an assessment of the effects of management actions on the environment. These assessments are often informed by results of specific model simulations evaluating various scientific and economic assumptions. Two models developed in New Zealand have been linked together to help develop policy at a national scale—the Land Use in Rural New Zealand (LURNZ) model used to predict land-use change and the Catchment Land Use for Environmental Sustainability (CLUES) model which estimates nutrient loads from the landscape driven by the LURNZ predicted land-use changes. Results from these models have shed light on whether fresh water quality is getting better or worse in different parts of New Zealand (Parliamentary Commissioner for the Environment 2013).

In this chapter, we describe (1) the purpose of catchment modelling, (2) the wide range and types of catchment models, especially in New Zealand, (3) how to choose a model or combination of models for a specific purpose, and (4) methods for evaluating models and their results. We then provide two examples of catchment models being applied to address catchment and downstream issues in small- and large-scale catchments in New Zealand (CLUES and GLEAMS–Groundwater Loading Effects of Agricultural Management Systems) and an example of a catchment model applied to address a very large catchment in the United States (SPARROW–SPAtially Referenced Regression On Watershed attributes model as applied to the entire Mississippi River Basin).

2.2 The Purpose and Types of Catchment Models

2.2.1 The Purpose of Catchment Modelling

Catchment models are used to answer questions about how landscape processes affect water quality or quantity in streams and ultimately the downstream receiving waters and are frequently used to inform decision makers in regards to policy design, compliance, regulatory, and restoration purposes. Catchment models are typically used either for research or management; however, often these two purposes are not independent. Models used for fundamental or basic research are typically developed to help understand a specific observed behaviour (hypothesis testing), whereas models for management are typically developed to predict the outcome of specific manipulations to the landscape. Catchment models developed for management purposes are typically used for farm-level consulting, to inform policy design, to evaluate compliance, for regulatory purposes, and to develop strategies to improve water quality. The differences in models developed for research and those developed for management and regulatory purposes have led to different assumptions in how they represent what may be effectively equivalent systems, and how they are applied. Models used in research are often used to examine the effects of various processes that are not well understood. Models used for management and regulation typically require significantly more testing/validation than models for non-regulatory or research purposes, and are typically used to simulate the effects of processes that are well understood. Therefore, models used for management are often more transparent in terms of the assumptions made in the model and have less uncertainty in their predictions than models used for research. Furthermore, models used for regulatory purposes should include more public input in how they are developed and be more defensible in a legal situation than models used strictly for research.

2.2.2 Types of Catchment Models

There are many different types of catchment models. Catchment models can be classified in terms of what constituents are being simulated, how the processes are represented, the time and space scales that are used, and how the models are calibrated or validated. Some catchment models simulate changes in a specific constituent, such as total phosphorus, whereas other models simultaneously simulate the changes in a suite of interacting constituents, such as flow, sediment, and nutrients.

Processes involved in the transport of sediment and nutrients from the landscape to downstream water bodies are extremely complex, which leads to differences in how these processes are represented in the models. Catchment models are typically subdivided into (1) relatively simple empirically based statistical models, such as land-use export-coefficient models; (2) relatively complex dynamic, physically based process models that incorporate the effects of landscape slope, rainfall, hydrology, soil types, land uses, vegetation, and land-management practices; or (3) semi-mechanistic hybrid models that use a combination of statistical relations and physically based process relations (Fig. 2.1).

Some catchment models, especially statistical models, describe average conditions for a specific period (e.g. a long-term average), whereas other models, especially process models, simulate changes over a range of time scales, ranging from sub-daily, to daily, to annual, to long-term averages. Some models only simulate changes during a specific flow regime (i.e. base-flow conditions versus event conditions), and others simulate over a wide range of conditions.

Catchment models have been developed for spatial scales ranging from a paddock or farm scale to a sub-catchment, catchment, regional, or national scale. In practice, the scale of the application often drives which model can be used. Modelling for catchment management in New Zealand has generally been divided into two categories: farm or paddock-scale models that provide estimates of nutrient loss, and catchment-scale models that describe how nutrients and sediment travel across large-

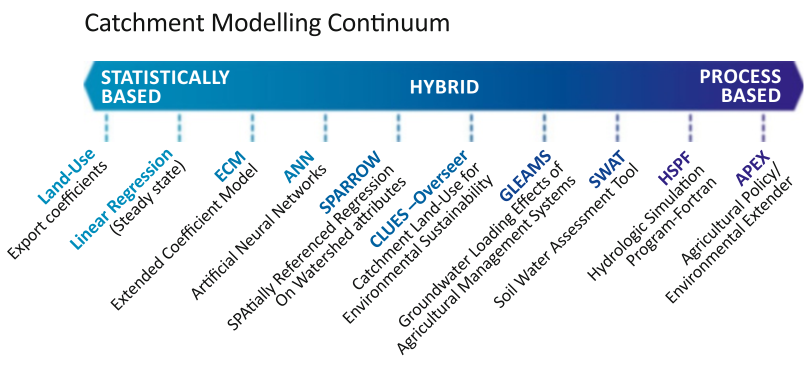

Fig. 2.1 Catchment modelling continuum (modified from Preston et al. (2009))

scale catchments to downstream water bodies. The latter models usually simulate both urban and rural conditions.

As noted above, models that use a combination of statistical and process-driven relations are referred to as hybrid models. Two examples of hybrid models are SPARROW and CLUES. SPARROW uses a linked statistical and process-based approach to predict long-term average loads of constituents, such as nutrients delivered to downstream receiving waters (Schwarz et al. 2006). CLUES is a hybrid model developed by the National Institute of Water and Atmospheric Research (NIWA) in New Zealand and is an amalgamation of existing modelling and mapping procedures contributed by various research organisations (Elliott et al. 2005, 2008). CLUES uses information from a simplified version of the farm-scale nutrient budgeting model OVERSEER® Nutrient Budgets ("OVERSEER"), which calculates nutrient budgets for management blocks within a farm, taking into account inputs and outputs and internal cycling of nutrients around the farm (Wheeler et al. 2006). CLUES then uses information from SPARROW for non-agricultural sources and to estimate the downstream transport of various constituents.

The spectrum of models that has been used over various spatial and temporal scales in New Zealand is described by Elliott and Sorrell (2002) and Fenton (2009). Catchment models are continually being improved and even updated, so any summary of the literature can quickly become outdated.

2.3 Detailed Descriptions of the Different Types of Catchment Models

2.3.1 Statistically Based Catchment Models

Statistically-based catchment models are driven by observed data and empirical relations that describe the observed data. These models are typically based on relations found between the constituent of interest, such as phosphorus (typically expressed as a yield, kg km^{-2}), and various characteristics describing the catchment, such as the percentage of agricultural area. The typical approach for developing a statistical catchment model consists of: (1) measuring the constituent of interest at as many locations in the catchment as possible, (2) calculating the attributes upstream of each monitoring sites (typically using GIS techniques), (3) developing statistical relations between the constituent of interest and selected attributes of the catchment, (4) calculating the attributes for all of the remaining unmonitored basins in the area of interest, and (5) using the statistical relations and the attributes of the unmonitored areas to predict the constituent of interest throughout the entire catchment. The step-by-step process of applying statistical catchment models is described by Elliott and Sorrell (2002).

Various statistical approaches have been used to develop the relations between a constituent of interest and specific aspects describing the catchment. The most common approaches are simple multiple regression or stepwise multiple regression, which can be developed with almost any statistical package. A few examples of simple statistical models include land-use export-coefficient models (Reckhow et al. 1980; Elliott and Sorrell 2002), the watershed regression for pesticides (WARP, Larson and Gilliom 2001), linear and nonlinear regression equations used to predict nutrient yields throughout the Mississippi River Basin (David et al. 2010; Jacobson et al. 2011); and the Extended Coefficient Model (ECM) that uses livestock numbers, population served, sewage treatment plants, septic tank numbers, road and track density, farming practices and location, and area of lakes and reservoirs to estimate nutrient exported from sub-catchments (Johnes 1996). Data-driven regression models can also be quite complex. Examples include non-linear time series models and artificial neural network (ANN) models.

Typically, the mathematical relations used in these statistical models are developed using regression techniques that minimise the variance in residuals between the observed and predicted values. Adding additional descriptor variables into these models continues to reduce the variance between the measured and predicted values; therefore, statistical fit criteria are commonly examined to try to minimise the possibility of overfitting the relations. The Akaike Information Criterion (AIC) and Bayesian Information Criterion (BIC) value, which penalise larger, more complicated models, with equal fit, are often used to aid in model evaluation and selection.

2.3.2 *Process-Based Catchment Models*

Process-based models are typically based on a series of mathematical relations describing each of the physical mechanisms involved in driving changes in the constituent(s) of interest. Like all models, process-based models involve some simplification and abstraction, and a range of complexity, with tradeoffs between the accurate representation of processes and the ability to obtain all of the information needed to estimate the unique set of parameters that describe those processes. Examples of highly complex process-based catchment models include the Agricultural Policy/Environmental eXtender (APEX) (Williams and Izaurralde 2005), the Hydrologic Simulation Program (HSPF) (Donigian et al. 1984), and the Soil and Water Assessment Tool (SWAT) (Arnold et al. 1998; Gassman et al. 2007; see Feature Box 2.1). Process-based catchment models typically have components or modules (a series of mathematical relations) that describe the routing of water, sediment, nutrients, and other constituents across complex landscapes and channel systems to the watershed outlet or a downstream water body, as well as components that describe the transport in groundwater and through reservoirs. In process-based models, each sub-catchment (in some case as small as fields) must be supplied with the data describing the source inputs of the constituent being modelled and all of the factors that affect its delivery, such as soil characteristics, management practices and meteorological conditions. Often highly complex process-based models are difficult to apply over large areas because of difficulty obtaining all of the detailed information required by the model.

Although process-based catchment models are based on a series of mathematical relations derived from theory and field studies, in application these models have coefficients that may be estimated and therefore the models have the potential to more accurately describe the changes in the constituent(s) of interest for a specified area than less complex models. These models are usually data intensive and frequently over-parameterised. In some simple applications, the unique parameters and their associated coefficients that describe each of the processes occurring in a sub-basin may be fit by trial and error. However, the manual trial-and-error approach is an imperfect process because even though some insight is gained in each attempt, feedback between sources and sinks and other correlated parameters and processes are very complex (Anderson et al. 2015), especially when trying to simultaneously fit many calibration sites. Therefore, automated calibration techniques have been developed to optimise the fit of the coefficients in models such as process-based models. A few commonly used automated calibration techniques include PEST (Doherty 2015), UCODE-2005 (Poeter et al. 2005), or MADS (Vesselinov and Harp 2012). Automated calibration codes are used to find the combination of coefficients that minimise a specific statistical index, such as the root mean square error (RMSE) between the predicted and measured values. In the calibration process, weights are often applied to each of the specific measurements to compensate for uncertainty in some of the observed measurements used in the calibration process.

After a process-based model is calibrated at selected locations in the basin, it is then applied to each sub-catchment with each sub-catchment having its own inputs of the constituent being modelled and all of the factors that affect its delivery, such as soil characteristics, management practices and meteorological conditions. In some applications, the coefficients found during the calibration process are used for all of the sub-catchments in the model; however, in other applications, coefficients are estimated based on the similarity of the unmonitored sub-catchment to those for which the coefficients were determined. The outputs from each sub-catchment are then usually routed down the stream network to the outlet of the catchment. Each process-based catchment model has its own routing mechanisms with its own techniques (series of mechanistic relations) that incorporate in-stream or in-reservoir decay. Routing through a stream network is not limited to process-based models.

2.3.3 Hybrid Catchment Models

Hybrid catchment models, such as CLUES and SPARROW, fall between the two extremes of full statistical and process-based models and often include mechanistic components that are empirically fit or calibrated. In SPARROW, mass contributions from measured sources are balanced with simulated losses (associated with transport to the stream network and in-stream decay) to optimally match the water quality measured at the monitoring locations. The land-to-water delivery and in-stream and in-reservoir decay rates used in each SPARROW model are empirically fit using statistical approaches, such as non-linear least squares regressions (Schwarz et al. 2006) or Bayesian nonlinear regression techniques (Qian et al. 2005). The spatially explicit, mass-balance, mechanistic framework in SPARROW enables individual nutrient sources to be tracked and quantified during downstream transport, assuming contributions from all of the sources are transported in a similar manner during downstream delivery. Typically, hybrid models do not attempt to describe all of the mechanisms that are included in process-based models. Instead, they are parsimonious and only try to explain the most important processes. For example, SPARROW models typically only include variables that are statistically significant in describing the variability in loading throughout the catchment (Schwarz et al. 2006).

2.3.4 Regression-Based Versus Process-Based Catchment Models

Simple regression-based models are good at describing the spatial distribution in concentrations, loads, and yields; however, results from simple regression relations may not accurately characterise cause-and-effect relations. The reason for this is that individual variables in a regression model can act as surrogates for various other variables if they are correlated with one another (co-linear); therefore, variables that are correlated with one another may not be significant in a regression relation (e.g. only one of the correlated variables might be statistically significant) but still be important in affecting sediment and nutrient delivery (Robertson and Saad 2013). In addition to this potential difficulty, if a sufficient number of sites is not monitored and thus included in the analysis, it is difficult to incorporate many explanatory variables into a regression relation. Therefore, results from simple statistical modelling techniques can often result in misleading results if coefficients are over-interpreted.

In contrast, process-based mechanistic models are based on the underlying mechanisms affecting sediment and nutrient delivery. Like all models, process-based models involve some simplification and abstraction, but they should better represent the behaviour of complex systems that simple linear description approaches may fail to capture, making them potentially more trustworthy for prediction. Process-based models fall along a continuum of complexity, with tradeoffs between the accurate representation of processes and the ability to estimate a unique set of precise parameters. Models, such as SWAT and APEX, are highly parameterised mechanistic models that attempt to simulate all of the important processes occurring in the watershed. Each catchment is typically provided with uniquely valued parameters that govern each of the numerous processes used to explain contaminant fate and transport. All of the data describing the inputs, management, and factors affecting the delivery of the constituent(s) being examined, however, are rarely available to estimate the catchment-specific parameters over the entire geography being examined. Therefore, some of the parameters in a process-based model are typically based on values derived from the literature, while others are estimated through calibration.

It should be noted that whether using a statistically-based model or a process-based model, it is still important to have monitoring sites describing the range in processes, sources, and environmental characteristics in the study area for model development and validation. In practice, however, for small-scale process-based models, one may have only one or two monitored locations or even none at all. These limited monitoring locations may not represent the entire study area and can result in the model not representing the unmonitored areas very well.

The availability of monitoring data and data describing the catchment often determines what types of modelling methods are implemented. Since the 1980s and generally in all the sciences, complex, mechanistic process-based models were developed to obtain as much understanding as possible about processes from the few data that were available. Today, with measurement instruments becoming less expensive, very large datasets have become available with new instrumentation, and methods for computation and data analysis have been developed to extract as much information as possible from these large datasets.

2.4 Choosing and Evaluating the Right Model

There are many models that describe sediment and nutrient losses from a catchment, and one is often faced with the problem of choosing the appropriate model from the wide range that are available, combining results of more than one model, modifying an existing model, or constructing a new model. The reasons for this range of models include the diverse range and complexity of catchments, the purposes for the modelling, which in turn implies an interest in different kinds of processes, and the availability of different kinds of information (i.e. data quantity and quality).

2.4.1 Choosing the Right Model or Combination of Models

The choice of model depends not only on the question being addressed and what data are available, but also depends on the training and background (science, applied mathematics, statistics, management, GIS, etc.) of the modeller. Depending on the modeller's training and background, they may consider the "model" as (1) a conceptual framework with numerous assumptions about the physical system, (2) a simple mathematical model with mathematical concepts and simple equations that describe the system, (3) a numerical model, which is a mathematical model that has used some sort of numerical procedure to obtain the behaviour of the mathematical model over space and time, or (4) a computer model, which combines the mathematical relations, is coded in specific software and could be ready to use. Formulating a mathematical model is regarded as one of the most important steps in the modelling process. However, many modellers prefer something previously developed and tested (off-the-shelf) and readily available that they can use with confidence. Other modellers prefer to develop models specifically for their application. Custom-made models are generally regarded as expensive but may be necessary to describe the unique landscape features of some areas, such as in New Zealand, or desirable to develop a group's modelling capability.

The choice of a model requires an alignment between the purpose for which the model was constructed and its intended use. A consequence of a model being developed for a specific purpose is that it may have limited use in other applications. Therefore, it often becomes preferable to limit model complexity, so that the model does not become site specific and not transferable to other sites.

Additional important considerations in choosing a model are its ease of use and how easily the results of the model can be presented. There is an increasing use in New Zealand of catchment models developed in other countries; however, these models are often perceived by some as not being easily applied to New Zealand catchments because of New Zealand's unique characteristics. It may be difficult to match the land uses and terrains in New Zealand with that typically used in most other models—for example, tussock grass is not prairie grass. Consequently, models such as SWAT, with their extensive database of land management, land cover, soils, etc., need to be populated with local parameter values to be readily useable in New Zealand and these parameter values (which are rarely presented in catchment modelling reports) need to be made available for others to use. In addition to choosing the best model to address a specific question, there are other considerations, such as data availability and availability of resources (time and money).

The Australian Cooperative Research Centre (CRC) for Catchment Hydrology has released a series of papers describing the fundamentals and practical considerations for modelling and guidelines in model selection (CRC for Catchment Hydrology 2005). These guidelines were thought to be necessary because of the multitude of models available and applied in Australia. The CRC for Catchment Hydrology suggests choosing the model that best fits the question being addressed and suggests choosing the simplest model that will do the job required. Adding unnecessary complexity can lead to reductions in the predictive performance. They provide a model choice decision loop to assist in choosing the right model for a particular purpose. This loop involves defining the objectives; assessing the availability of data, expertise, and resources; and then reassessing whether the objectives can be met. This loop also includes: usefulness and limitations of the models available, approaches being considered in the modelling, predictive performance of the model, and the temporal and spatial discretisation/resolution required. There are no such formal or informal guidelines in New Zealand where there is limited choice of models that have actually been developed and applied, largely because of the limited science structure and investments made in catchment modelling. To make more informed decisions about model selection and use, formal guidelines on model selection in New Zealand would be helpful. The Australian guidelines could be used as a template.

2.4.2 Evaluation of Catchment Models

Model output enables one to evaluate the assumptions, limitations, and the bounds and/or areas of uncertainty in the information being used. Most modellers evaluate model performance by analyzing how closely the model behaviour matches that of the real system and understanding the level of uncertainty in model results. Two of the most common indices for evaluating this type of performance are the RMSE and the Nash-Sutcliffe efficiency (NSE) statistic (Nash and Sutcliffe 1970). NSE can be interpreted as a classic skill score (Murphy 1988), where skill is interpreted as the comparative ability of a model with regard to a baseline model, which in the case of NSE is taken to be the mean of the observations. The evaluation of catchment models includes not only the evaluation of model performance but also the evaluation of uncertainty and realism (Wagener 2003). Characterising the full performance of models is described by Bennett et al. (2013), using a five-step procedure: (1) assessment of the original aim, scale, and scope; (2) characterisation of the data for calibration and testing; (3) evaluation of model bias; (4) evaluation of basic performance criteria; and (5) consideration of more advanced methods to handle problems such as systematic divergence between modelled and observed values. One key issue with evaluating the predictability of complex multi-parameter models is that there is not one evaluation criterion for all models. Hence, model performance statistics may not give a true representation of model accuracy (Bennett et al. 2013).

From a management point of view, uncertainty is quite simply the lack of providing exact knowledge regardless of its cause (Refsgaard et al. 2007). In practice, the evaluation of uncertainty has come to mean quantifying and understanding the magnitude of errors. Errors can come from various sources including randomness, measurement error, systematic error or natural variation, and subjective judgement due to the interpretation of data. In addition, however, there can be model structural uncertainty, parameter uncertainty, evaluation data uncertainty, and input data uncertainty (Krueger et al. 2009). Since models are only abstractions of the natural system, some less important variables and interactions are often left out. Also, there may be insufficient knowledge about certain processes. Uncertainty about the model's structure, i.e. uncertainty about the cause-and-effect relationships, is often difficult to quantify.

The evaluation of realism of a model's structure is an evaluation of how well it describes specific processes. Due to differences in behaviour among catchments, different models can be "right" for different catchments. Consistency is defined as the ability of a model structure to adequately reproduce several (hydrological) signatures simultaneously while using the same set of parameter values. Performance of a model's structure is its ability to mimic a specific part of the (hydrological) behaviour in a specific catchment. The consistency and performance of a model's structure can be determined independently, but are both important for its evaluation (Wagener 2003). An objective and diagnostic approach has been developed to identify whether a model structure is suitable for a certain catchment and has been applied in New Zealand to the Maimai catchment by Euser et al. (2013) and Vache and McDonnell (2006). Results show that some model structures have better performance and consistency than others; being aware of all of the various sources of uncertainty helps to fully evaluate a model.

2.4.3 The Need for Guidelines on How to Use the Model Selected

Prior to conducting a modelling study, specific guidelines or protocols on how the modelling should be conducted should be established. These protocols provide modellers with a roadmap to follow, reduce modellers' bias, enhance the reproducibility of model application studies, and eventually should improve the acceptance of modelling results. To date, most catchment modelling studies have not used formal protocols on how to use a specific model but instead have used ad hoc approaches. Catchment model application protocols have been presented by Engel et al. (2007) based on standard modelling applications and protocols adapted from U.S. Environmental Protection Agency. Australian groundwater modelling guidelines given by Barnett et al. (2012) describe the planning stage, model conceptualisation, design and construction of models, model calibration stage, predictive scenarios, and reporting. These guidelines are expected to be adopted by regulatory bodies, modellers, reviewers, and proponents of groundwater models as a nationally consistent guide to groundwater modelling. The role of these guidelines is a point of reference for best practice for all those involved in the development, application, and review of groundwater models and for surface water modellers who use the outputs from these models. Protocols, such as these, could be useful to develop guidelines on how catchment models may be used. Such protocols and guidelines are needed if models and model predictions are to be used for regulatory purposes (Parshotam 2015).

2.5 Catchment Modelling in New Zealand

2.5.1 Historical Context

The attitude towards catchment modelling in New Zealand has dramatically changed through time. In the 1980s, it was difficult to convince people of the importance of catchment modelling; however, today it is difficult to convince people that catchment modelling is not always required to manage watersheds and for downstream water body restoration. One of the most widely used catchment models in New Zealand has been the physically based Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) developed by the U.S. Department of Agriculture, Agricultural Research Service (USDA–ARS) to analyze the effects of pesticide and nutrient infiltration into the root zone of the soil profile (Knisel 1993). GLEAMS is an extension of the Chemicals, Runoff, and Erosion from Agricultural Systems (CREAMS) process-based model in that the hydrology, plant nutrient, and pesticide components of CREAMS were modified to simulate the movement of water, nutrients, and chemicals through the root zone. GLEAMS and its related family of models—Basin New Zealand (BNZ), GLEAMSHELL, and the Watershed Analysis Model (WAM)) (Cooper and Bottcher 1993; Bottcher et al. 1998)—have been used in many studies (Table 2.1). The BNZ model was developed in 1987−1988 and included components from CREAMS (Knisel 1980), but was converted to using components of GLEAMS when WAM was developed to take advantage of the spatial datasets that became available. GLEAMS is used to calculate the load from each sub-catchment in GLEAMSHELL and WAM models. Incorporation of GLEAMS into these models improved them by incorporating groundwater recharge and animal waste as a specific input. At the same time as components of GLEAMS were included in BNZ, four separate programs from CREAMS were combined into the integrated model WAM in 1996−1998. WAM (or its variants) have been used to address a variety of catchment and downstream water body restoration issues in New Zealand (Table 2.1). WAM and WAMView (the ArcView version of WAM) have been applied in many New Zealand catchments and in various catchments in the United States (Bottcher and James 2014). A more in-depth description of the use of GLEAMS-based models in New Zealand is given by Parshotam (2010). Williamson and Mills (2009) reviewed GLEAMS-based studies that examined the effects of storm water on the aquatic environment in the Auckland area.

Another commonly used model developed in New Zealand for catchment management is OVERSEER Nutrient Budgets (Wheeler et al. 2006), commonly referred to simply as OVERSEER. Edmeades (1995, 2012) described the early development of OVERSEER, which started as a tool to calculate a nutrient budget for a farm and management blocks within the farm, taking into account inputs and outputs and internal cycling of nutrients around the farm. Through time, OVERSEER has evolved to include nitrogen losses, description of the Fertilizer Code of Practice (in 2000), land application of effluent (in 2002), greenhouse emissions (in 2003), mitigation options and phosphorus runoff (in 2005), effects of wetlands, effects of dicyandiamide, monthly stock calculations and forage crops (in 2008), input parameter reports (in 2009), monthly time-step changes, and integration of pastoral and arable land uses (in 2012). The release of OVERSEER Nutrient Budgets version 6 in 2012 incorporated all farming types into the one model and incorporated a significant shift in the understanding of nitrogen immobilisation. At that time, OVERSEER also became an internet-based application making it easier for users to access. The OVERSEER model is used now for regulation and policy in quite a different way from its initial intended use as a farm nutrient-budget tool.

Table 2.1 Examples of GLEAMS, BNZ, and WAM studies in New Zealand

GLEAMS-based model	Study	Reference(s)
GLEAMSHELL	Managing water quality in agricultural landscapes and land-use conversion in Toenepi in Waikato	Rodda et al. (1997)
	Nutrient loads entering Lake Taupō	Elliott and Stroud (2001)
BNZ (Basin New Zealand)	Identifying sediment sources and potential effects of land-use change in the Mahurangi catchment	Stroud and Cooper (1997)
	Mahurangi land-use scenario modelling	Oldman et al. (1998)
	Sediment loss from vegetable growing fields in Pukekohe	Stroud and Cooper (1998)
	Catchment modelling of Oteramika, Southland	Thorrold et al. (1998)
	Estimating the effects of urbanisation on sediment loss in the Mangemangeroa catchment	Oldman and Swales (1999)
WAM (Watershed Assessment Model—Okura)	Impacts of urban and motorway development on sedimentation in Orewa Estuary	Williamson et al. (1998)
	Determining the effects of rural intensification options on sediment loads to the Okura Estuary	Stroud et al. (1999), Stroud and Cooper (1999)
	Risks of sediment runoff to estuarine biota under proposed Development in the Whitford catchment	Senior et al. (2003)
	Modelling long-term daily sediment loads to the Mahurangi Estuary	Stroud (2003)
GLEAMS	Predicting sediment loss under proposed development in the Waiarohia catchment	Collins (2003)
	Contaminant accumulation in the Upper Waitemata Harbour	Green et al. (2004a, b)
GLEAMS-CWH	Central Waitemata Harbour sediment study	Parshotam and Wadhwa (2007a, 2008)
GLEAMS-SEM	South-eastern Manukau Harbour sediment study	Parshotam (2008), Parshotam et al. (2008a, b, c)
GLEAMS-TAU	Tauranga climate change, land development, urbanisation, sediment runoff from urban areas	Parshotam et al. (2008d), Elliott et al. (2009)
GLEAMS + NIWA pond model	Assessment of the associated sediment and contaminant loads from the Western Ring Route—Waterview connection	Harper (2010)
GLEAMS-CWH with climate change scenarios	Evaluating the effects of climate change on sediment accumulation in the Central Waitemata Harbour	Green et al. (2010)

2.5.2 *Brief Summary of Catchment Models Commonly Used in New Zealand*

A wide range of models have been used in New Zealand for catchment management and downstream water body restoration, several of which are being used by Regional Councils and unitary authorities to inform policy development. Compiled inventories of the various decision support models used in New Zealand and the interactions between models and associated case studies are given on the Envirolink website (http://tools.envirolink.govt.nz/dsss) and software frameworks are available at (https://teamwork.niwa.co.nz/display/IFM). Some of the most widely used catchment models in New Zealand, particularly those emphasising surface water quality and agricultural management, are described briefly below.

2.5.2.1 GLEAMS

GLEAMS and its variations have been used extensively in New Zealand over the last 30 years (Table 2.1). GLEAMS was designed and used for solving catchment-scale problems, such as water and sediment transport to estuaries, pond, lake, and reservoir design and restoration, irrigation water transfer, and stream channel routing of sediment and agrichemicals. GLEAMS operates on a daily time step and accounts for spatial variability of soils, land use, climate, and topography in each sub-catchment.

2.5.2.2 ROTAN

The ROtorua TAupo Nitrogen (ROTAN) model is a GIS-based, statistically based, rainfall-runoff-groundwater model developed by NIWA. ROTAN simulates the effects of land-use change, rainfall scenarios, and nitrogen mitigation measures on nitrogen export to waterways (Rutherford et al. 2009, 2011; Palliser et al. 2011). ROTAN is based on Scandinavian Hydrologiska Byråns Vattenbalansavdelning (HBV) models in which soils and aquifers are modelled as a series of fully mixed reservoirs (buckets) connected in series and parallel and modified to simulate the transport of nitrogen and phosphorus. ROTAN has been used in policy formulation related to the Lake Rotorua catchment.

2.5.2.3 AquiferSim

AquiferSim is a region-level process-based model that simulates nitrogen transport via groundwater flows (Bidwell and Good 2007). AquiferSim was developed by Lincoln Ventures Ltd. as part of the multi-organisation Integrated Research for Aquifer Protection Project. AquiferSim produces spatial estimates of groundwater and nitrogen discharge to streams. Its inputs include maps of climate, topography, soil type, groundwater zones, land use, and production. AquiferSim integrates the effects of nitrate-contaminated recharge at the farm scale and provides information about the resulting horizontal and vertical distribution of nitrogen in an aquifer. The model was designed to help regional councils in New Zealand determine the long-term effects of land-use change on groundwater and estimate how long before these cumulative effects could become problematic. AquiferSim and its mathematical relations are becoming an integral part of many catchment models that require a groundwater component.

2.5.2.4 OVERSEER

The OVERSEER Nutrient Budget model (Wheeler et al. 2006) is an on-farm decision support model developed by AgResearch that helps farmers and growers make informed strategic management decisions about their nutrient use to improve performance and reduce losses to the environment. The model is based on nutrient budgets calculated for land management blocks within the farm, taking into account inputs, outputs and internal cycling of nutrients throughout the farm. The model is useful for solving management problems dealing with the transport of nutrients onto and off of a farm. The model has been developed to describe nutrient transport for the majority of farm types in New Zealand including dairy, sheep, beef, deer, dairy goats, kiwifruit, apples, grapes, avocados, peaches, and seed, grain, and vegetable crops. The model was initially used to determine inputs and outputs of nutrients from a farm; however, it is now also used as compliance tool by councils to assess effects in specific resource consent applications. Currently, OVERSEER is the only mathematical model used in New Zealand for regulatory purposes.

2.5.2.5 SPASMO

The Soil Plant Atmosphere Systems MOdel (SPASMO; Green et al. 2004a, b, 2012) is a process-based catchment model developed by Plant & Food Research of New Zealand that simulates the transport of water, microbes (viruses and bacteria), and solutes (nitrogen and phosphorus) through soils. The model incorporates the effects of climate, differences in soil types, and water uptake by plants in relation to farm and orchard practices. The model describes the movement of water and solutes through a one-dimensional soil profile and the overland transport of sediment and nutrients. The soil–water balance in the model is calculated by considering the inputs (rainfall and irrigation) and losses (plant uptake, evaporation, runoff, and drainage) of water from the soil profile. SPASMO has been shown to provide closely equivalent output to that generated by OVERSEER (Green et al. 2004a, b). SPASMO is currently used by six Regional Councils for the allocation of irrigation water.

2.5.2.6 SPARROW

The SPAtially-Referenced Regression On Watershed attributes (SPARROW) model is a GIS-based catchment model developed by the U.S. Geological Survey (Schwarz et al. 2006). SPARROW uses a mass-balance approach to estimate the non-conservative transport and transformation (i.e. losses) of nutrients under long-term steady-state conditions in relation to statistically significant landscape properties, such as climate, soils, and instream decay. SPARROW is a spatially explicit model and is calibrated using a hybrid statistical and process-based approach. Mass contributions from measured sources are balanced with modelled losses to optimally match water quality at monitoring locations. SPARROW models simulate long-term mean-annual transport given inputs similar to a specified base year. The spatially explicit, mass-balance, mechanistic framework of SPARROW enables individual sources of a constituent to be tracked and quantified during downstream transport. SPARROW models have been applied extensively in large basins in the United States. (Alexander et al. 2008; Preston et al. 2011) and New Zealand (Elliott et al. 2005).

2.5.2.7 CLUES

The Catchment Land Use for Environmental Sustainability (CLUES) model, described earlier, is a catchment model developed by NIWA. CLUES uses results from several other models, both process-based and statistical, including a simplified version of OVERSEER (losses from pastoral lands), SPARROW (losses from nonagricultural lands and downstream transport), SPASMO (losses from horticultural land and cropland), Triple-Bottom-Line (TBL, social, environmental and financial information), and EnSus (quantifying risk of nitrogen export). CLUES runs on a GIS platform and links outputs from several models and geo-spatial databases together into one software package. CLUES allows users to create both land-use and farm-practice change scenarios (stocking rates, mitigation) using a range of tools. Further details on the modelling framework can be found in Woods et al. (2006) and Elliott et al. (2011) and in the CLUES user manual (Semadeni-Davies et al. 2011). The CLUES modelling system was initially developed to provide a useful tool for councils to better manage catchments and downstream water bodies but has been used to prepare National Nutrient maps and study the effects of land-use intensification throughout New Zealand (Parliamentary Commissioner for the Environment 2013). The CLUES model has recently been used to assess the effects of implementing the National Policy Statement on Freshwater management in the Auckland Region (Semadeni-Davies et al. 2015). The CLUES model has been used to determine nutrient inputs to lakes (Verburg et al. 2012) and estuaries (Zeldis et al. 2012).

2.5.2.8 SWAT

The Soil and Water Assessment Tool (SWAT) model, described earlier, is a continuous-time step, semi-distributed, process-based catchment model developed by USDA–ARS (Arnold et al. 2012; Feature Box 2.1). SWAT includes key components contributed from several other USDA-ARS models. SWAT has been applied successfully to New Zealand catchments such as the Motueka River (Ekanayake and Davie 2004), and Puerenga Stream in Rotorua (Me et al. 2015). Parameter sets from many GLEAMS modelling studies in New Zealand are readily transferable to SWAT. SWAT-CUP (http://swat.tamu.edu/software/swat-cup/) is a public domain calibration and uncertainty program for SWAT. QSWAT (http://swat.tamu.edu/software/qswat/) is a GIS interface for SWAT using the open source GIS program, QGIS. The digital soil map for New Zealand, *S*-map (Lilburne et al. 2012), provides a seamless digital soil map coverage for New Zealand and should complement existing land use, topography, and climate layers needed as inputs by models such as SWAT.

2.5.2.9 DHI MIKE

Various modules of the integrated hydrologic modelling system MIKE, developed by the Danish Hydrological Institute (DHI), simulate flow, water level, water quality and sediment transport in rivers, flood plains, irrigation canals, reservoirs, and other inland water bodies (https://www.mikepoweredbydhi.com). The MIKE modules have been used for decades by New Zealand scientists, engineers, and private consultants. MIKE modules simulate water environments in a highly integrated manner. Modules describe groundwater, urban water, wastewater, flooding, water resources, rivers, ports, and coasts. MIKE models have been used over the last two decades in New Zealand to quantify the impacts of: (1) wastewater discharges into harbours and estuaries; (2) storm events on water quality in Okura Estuary, Auckland region (Pritchard et al. 2009a); (3) diverting rivers, such as the Kaituna River/ Maketu Estuary, Bay of Plenty (Tuckey 2014); (4) thermal discharges in Bream Bay, Northland for Mighty River Power (Lundquist et al. 2009); and (5) the effects of future land-use changes on sedimentation rates in the Tauranga Harbour, Bay of Plenty (Pritchard et al. 2009b), Central Waitemata Harbour (Pritchard et al. 2009a) and Upper Waitemata Harbour (Green et al. 2004a, b). Environment Canterbury Regional Council has developed coupled surface-water and groundwater hydrological models for the Hinds Catchment using MIKE, which was used to help develop the Land and Water Regional Plan focussed on setting allocation limits for protecting the quality and quantity of groundwater and surface-water resources (Durney et al. 2014).

2.5.2.10 Delft3D

The Delft3D process-based model was developed by Deltares, a Dutch-based research institute. Delft3D is a three-dimensional modelling suite that simulates the hydrodynamics, sediment transport, and morphology and water quality for fluvial, estuarine, and coastal environments. This model is increasingly being used to model channel dynamics in braided rivers, such as the Waimakariri River, and estuarine sedimentation (Green and Zeldis 2015). Delft3D has been applied to two riverine lakes: Lake Whangape and Lake Waahi (Jones and Hamilton 2014). A Delt3D modelling framework was set up by Deltares in collaboration with Dairy NZ for the Waituna Lagoon (Chrzanowski 2015), which consisted of a distributed hydrological model (WFLOW), OVERSEER, and a water quality model (Water Framework Directive Explorer, WFD-E). This framework was used to estimate in-river nutrient concentrations and total nutrient load contributions to the lagoon.

2.5.2.11 Hydroepidemiological Catchment Models

Two hydroepidemiological catchment models have been produced by NIWA (McBride and Chapra 2011): a dynamic catchment deterministic and process-based model and a catchment risk model. The dynamic deterministic model, developed for the dairy-dominated Waikato Toenepi catchment, describes the deposition of animal wastes from a range of animal types deposited directly to the ground or directly into streams and also estimates the fraction delivered as runoff to streams. These processes are simulated at an hourly or sub-hourly basis. The catchment risk model provides rapid assessment of various catchment loadings of *Campylobacter* and the associated water quality response. The risk model captures many of the key processes and microbial sources that are in the dynamic deterministic model, but uses a simplified representation of the processes and a coarser temporal and spatial resolution. The risk model includes consideration of parameter variation and uncertainty using a Monte Carlo simulation approach.

2.5.2.12 Nitrate and Phosphorus Leaching Maps

National nitrate and phosphorus leaching (export) maps have been produced by Landcare Research of New Zealand. These maps are similar to simple land-use export-coefficient models. These maps were constructed by first using the OVERSEER model to estimate nitrate and phosphorus leaching (export) rates from every soil and climate combination in New Zealand and then these results were combined with national maps of animal numbers to obtain total leaching, which represents the bulk of the nutrient loss from most of these areas (Dymond et al. 2013). A map of dissolved reactive phosphorus leaching was constructed from national maps of animal stock units and soil phosphorus retention. These spatially detailed maps enable planning for mitigation and restoration at local, regional, and national scales.

2.5.3 Availability of Model Formulation, Algorithms, and Source Code

More in-depth descriptions of most of the catchment models used in New Zealand are available in the literature or from the internet. The descriptions here are relatively brief and include a summary of the inputs and computer requirements needed for model application. However, these descriptions do not generally include complete model formulation nor the algorithms used in the model. A few commonly used catchment models, such as SWAT and SPARROW, have been thoroughly documented and their computer executable files are available upon request. For full model transparency, it would be beneficial to have the algorithms and assumptions used in catchment models, as well as the computer source code readily available. Without access to the code and its formulation, progress will be stalled as efforts are diverted to duplicate model development (e.g. AgSoft by Edmeades et al. (2016)).

2.5.4 Models for Compliance or Regulatory Purposes

OVERSEER has been reported to be one of the best tools to evaluate nutrient losses under various management practices in New Zealand (Williams et al. 2014). OVERSEER was initially developed to advise farmers on fertiliser application; however, today it is the only model used as a New Zealand compliance tool in a regulatory environment and so opens itself up to added scrutiny. OVERSEER can be used by both regulators and the pastoral industry to manage areas within environmental constraints because it provides comparative risks of management activities to the receiving environment. The dairy industry uses OVERSEER as a tool to assist farmers in comparing themselves with other farmers with respect to their sustainability accord; however, this requires that all farmers provide sufficient up-to-date OVERSEER information for their farms. Without OVERSEER, farmers would be faced with developing management plans to minimise their effects on the environment based only on their total input of nutrient sources. Environment Waikato uses OVERSEER to calculate Nitrogen Discharge Allowances for farmers in the Lake Taupo catchment. Similarly, the Bay of Plenty Regional Council uses OVERSEER to evaluate farmers in the Lake Rotorua catchment based on their relative nitrogen and phosphorus losses (Park 2014). As part of the Canterbury Land and Water Regional Plan, farmers are expected to use OVERSEER to calculate how much nitrogen will leach from their farms into groundwater. Other Regional Councils use OVERSEER to assess impacts of specific resource consent applications.

Various versions of OVERSEER have been used historically. The use of the various versions of OVERSEER has led to various consistency problems (Dewes 2015). This has led to several challenges for New Zealand councils, such as how to: (1) deal with model version changes, (2) manage input data quality and auditing, (3) establish user requirements, (4) understand the model averaging methods, (5) understand the level of uncertainty in the modelled outputs, and (6) obtain sufficient calibration data sets. Although OVERSEER has been extensively used in New Zealand, apart from internal reports and numerous conference presentations, there is little published on OVERSEER in the peer-reviewed journals and very little published on the formulation of the model. The source code is not available for commercially sensitive reasons. Parshotam (2015) explores the qualities a model used for regulatory purposes must have in order to be admissible (as scientific evidence) in the legal arena. Guidelines for Environment Court model admissibility conditions have not been used in New Zealand courts to assist courts assess whether model evidence should be admitted in the courts, but such guidelines have been used by courts overseas. These guidelines have been explored by Parshotam (2015), using the OVERSEER model as a test case.

2.5.5 A Small Catchment Context. A CLUES Example

The Best Practice Dairy Catchments Programme of New Zealand commenced in 2001 with the aim of integrating environmentally sustainable practices into dairy farming, while trying to increase farm business productivity. This project examined the effects of management practices in five catchments (Toenepi, Waikato region; Waiokura, Taranaki; Waikakahi, Canterbury; Pigeon Creek, Westland; and Bog Burn, Southland), which cover a range of soils, rainfall, topography, farming methods, and environmental concerns and range from 6 to 25 km^2 (Monaghan et al. 2006; Wilcock et al. 2007). These catchments were selected because they either had recognisable amenity and ecological values or were particularly good examples of highly impacted streams so that improvements arising from adoption of best management practices in their catchments might be most evident (Wilcock et al. 2007).

The CLUES model (v. 2.0.4) was applied to these five catchments to evaluate their nitrogen and phosphorus losses (Parshotam and Elliott 2009). Within this version of CLUES, the catchment and stream characteristics, fertiliser use, area in dairy operations, stocking rates, and irrigation statistics were obtained from farm surveys in Wilcock et al. (2007), land uses were obtained from New Zealand's landcover databases (LCDB2), licensed by the Ministry for the Environment, and the pastoral components were based on AgriBase 2003, New Zealand's farms database, maintained by AsureQuality. CLUES was first used to delineate the catchments for each of these sites using the coordinates of each of the catchment outlets. Then for each catchment, CLUES provided the stream reaches based on digital elevation models (DEMs), the nutrient yields and loads associated with each sub-catchment (Fig. 2.2, for a map of nitrogen loads), and the nitrogen loads in each stream reach.

To demonstrate how well CLUES simulated the loads from each of these catchments, the simulated annual nitrogen and phosphorus loads were compared with those measured for these sites (Fig. 2.3). This comparison demonstrated close agreement between measured and predicted nitrogen loads at the outlets from four of the five catchments; however, loads from Pigeon Creek were overestimated. Results from this study demonstrated the importance of including an additional source term for dairying, which has now been included in revised versions of CLUES. This study also demonstrated that CLUES was good at quantifying differences among catchments and parts of large catchments and the effects of changes in land use, rather than highlighting small-scale or subtle variations of yield within small sub-catchments of uniform land use.

2.5.6 A Large Catchment Context. A GLEAMS Example

Sedimentation is a continual problem for most estuaries and lakes. Land slope, land use, soil type, and rainfall all affect sediment loss from the landscape, and distributions of these factors usually result in a complex spatial pattern of sediment generation. In addition, various management activities can accelerate sediment loss. Therefore, there is a need to determine catchment sediment-loss hotspots and implement management controls in these areas to most effectively reduce sediment loss from the landscape and reduce downstream sedimentation in estuaries.

Several GLEAMS catchment models were developed to describe sediment loss from the landscape and transported to harbours and estuaries throughout New Zealand, including central Waitemata (Parshotam and Wadhwa 2007a, b, 2008), south-eastern Manukau (Parshotam 2008; Parshotam et al. 2008a, b, c), and Tauranga (Parshotam et al. 2008d; Elliott et al. 2009) harbours. These model variants were developed to address issues such as:

- Which areas, streams, soil types, and land uses contribute the most sediment to the harbours?
- What contribution do earthworks from urban development make to the sediment loads?
- What control measures are best at mitigating sediment loads from urban earthworks?
- How will the sediment loads increase in the future as a result of climate and land-use change?

The GLEAMS models developed for each region were used to simulate various climate and land-use scenarios based on regional growth management plans (Parshotam et al. 2008d). Model results were used to construct yield maps for each

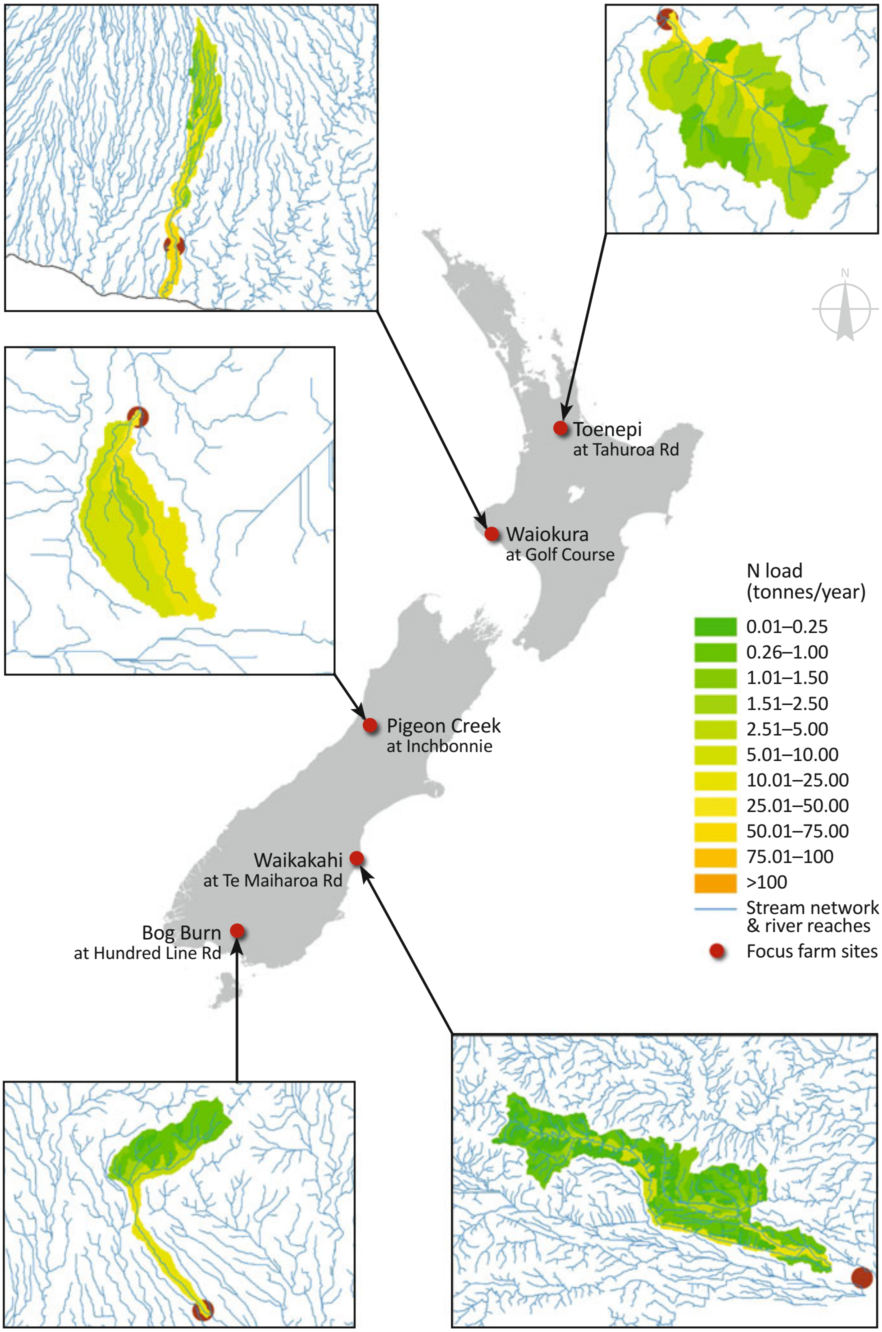

Fig. 2.2 Total nitrogen loads (tonnes/year) estimated using CLUES. Sub-catchments are coloured by the total nitrogen load in the corresponding reaches

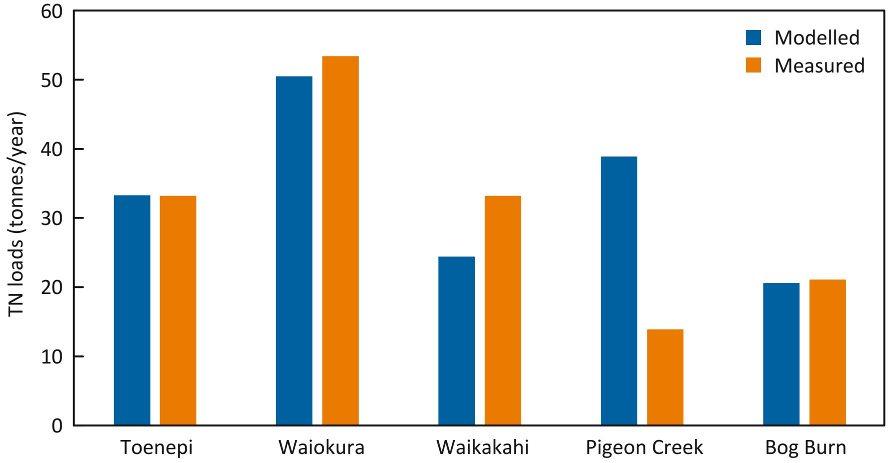

Fig. 2.3 Modelled total nitrogen (using CLUES) and measured annual total nitrogen loads

sub-catchment for current and future land use. An example of these simulations is given for the entire southern Tauranga Harbour catchment with Fig. 2.4, showing sediment yields for current land use.

The results of the Tauranga Harbour study demonstrated that pastoral land was the primary contributor of sediment to the harbour and that urban areas were only a minor contributor, contrary to what was previously thought. Pastoral land was estimated to generate the most sediment (64%), followed by bush, scrub, and native forest (28%). Urban areas were estimated to only generate 11% of sediment; urban earthworks and orchards and cropland contributed just 1.0 and 0.3%, respectively. This study demonstrated that stream bank erosion was a significant source of sediment to the Tauranga Harbour and that continued urban development should cause only a small change in sediment delivery.

2.6 Catchment Modelling in a Large Basin Outside of New Zealand: A SPARROW Example

Although only a few large-scale catchment models have been developed in New Zealand, there are several large-scale catchment models that have been developed in other areas of the world to address water quality issues important to an entire country or large parts of a country or countries and support restoration efforts. One of the largest-scale water quality issues in the world deals with coastal hypoxia. During the last few decades, excess nutrients from agricultural activities and wastewater delivered to the coastal environment have dramatically increased the occurrence of coastal eutrophication and hypoxia (WWF 2008; UNDP 2013). Worldwide there are now more than 500 dead zones, one of which is Cape Rodney,

Fig. 2.4 Sediment yields (tonnes/ha/y) for the current land use, mapped for every 30 m × 30 m model cell (over 1.1 million cells in total), estimated using GLEAMS

New Zealand. In these areas, excessive nutrients, especially nitrogen and phosphorus, fuel phytoplankton growth. As the phytoplankton die and sink to the bottom, oxygen is consumed by the bacteria decomposing this material. Hypoxia in the Gulf of Mexico, offshore of the United States, is the second largest zone of hypoxia caused by humans (Scavia and Evans 2012) resulting from high biological productivity driven by excess nutrient contributions from its very large watershed, primarily from the Mississippi River Basin. The Mississippi River Basin drains over 3,200,000 km^2 or about 41% of the conterminous United States (USGS 2000). The Mississippi River Basin has some of the most productive farming regions in the world, referred to as the Corn Belt, large forested areas, and many large cities. Therefore, definition of where and what causes the high nutrient loads is needed to guide restoration activities and would be extremely difficult without using large-scale catchment modelling techniques.

National and regional SPARROW models have been developed to describe nutrient losses from catchments throughout the entire United States (Smith et al. 1997; Preston et al. 2011) and help address widespread regional issues. To help address the hypoxic conditions in the Gulf of Mexico, SPARROW models were constructed and used to estimate total nitrogen and phosphorus loading from areas throughout the entire Mississippi River Basin, rank all of the contributing areas based on their respective loads and yields to the Gulf, and determine the relative importance of each of the major sources of nitrogen and phosphorus to the loadings over various spatial scales (Robertson and Saad 2013, Robertson et al. 2014). SPARROW models were developed using spatially explicit nutrient input information from wastewater treatment plants, fertilisers, manure, atmospheric deposition, and urban and forested areas and calibrated using nitrogen and phosphorus load measurements throughout the entire basin.

The nitrogen and phosphorus SPARROW models were both fully evaluated (Robertson and Saad 2013). The coefficients for all sources and transport factors in the models were highly significant ($p \leq 0.05$), indicating that each source and transport factor was important in describing the measured loads. All of the coefficients in the models, estimated using nonlinear least-square regression techniques, were also robust (small standard errors compared to magnitude of the coefficient, narrow 90% confidence intervals, relatively small variance inflation factors, and generally within 5% of estimates made using alternative bootstrap estimation techniques). The nitrogen model explained 94 and 88% of the variances in the 856 monitored nitrogen loads and yields, respectively, and had a RMSE of 0.53, while the phosphorus model explained 90 and 75% of the variances in the 988 monitored phosphorus loads and yields, respectively, and had an overall RMSE of 0.67 (all in natural log units). Model residuals for both models were also examined at all of the monitored locations and demonstrated no consistent regional biases. This type of evaluation should be conducted for all catchment models and is important for understanding the predictability of the models.

Results from the SPARROW models delineated the spatial patterns in nitrogen and phosphorus lost to streams throughout the entire Basin (Fig. 2.5a, b) and quantified the amount delivered to the Gulf of Mexico from each of these sub-catchments (Fig. 2.5c, d). The delivered loads incorporate all instream processes affecting the downstream delivery of nutrients, such as denitrification and in-stream/in-reservoir deposition. The results demonstrated where the highest nitrogen and phosphorus yields originate: primarily from areas throughout the central part of the Basin, along the Mississippi River, and from areas draining large population centres.

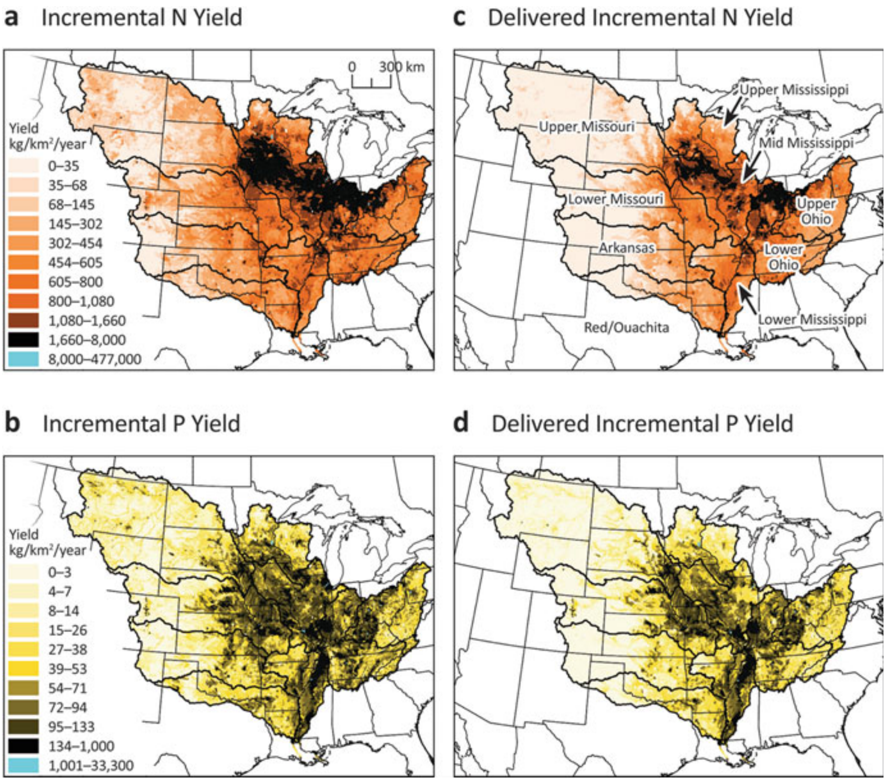

Fig. 2.5 SPARROW model estimates of catchment and delivered incremental (sub-catchment) nitrogen and phosphorus yields to the Gulf of Mexico (Robertson and Saad 2013)

Reducing nutrient loadings throughout the entire Mississippi River Basin would provide a comprehensive way to reduce loading to the Gulf of Mexico and reduce Gulf hypoxia; however, it would not be the most efficient restoration strategy because not all areas contribute equal quantities of nutrients. SPARROW results can be used to target where reduction efforts would provide the largest potential benefit, that being in areas with the largest relative contributions or highest contributing areas (hotspots). Therefore, various scales of sub-basins and political boundaries (states) were ranked on the basis of their respective aggregated loads, yields (by compensating for watershed size), incremental (sub-catchment) yields, and delivered incremental (sub-catchment) yields (by compensating for losses during downstream transport) (Robertson et al. 2014). An example of the ranking is demonstrated by state in Fig. 2.6. The highest contributions of the nitrogen load came from Iowa; therefore, efforts in this area may have a large impact on the overall nitrogen loading to the Gulf of Mexico.

SPARROW model results also demonstrated the dominant sources of nitrogen and phosphorus at each of the various spatial scales (Robertson and Saad 2013). Agricultural input (fertiliser, manure, and additional inputs from legume crops) was

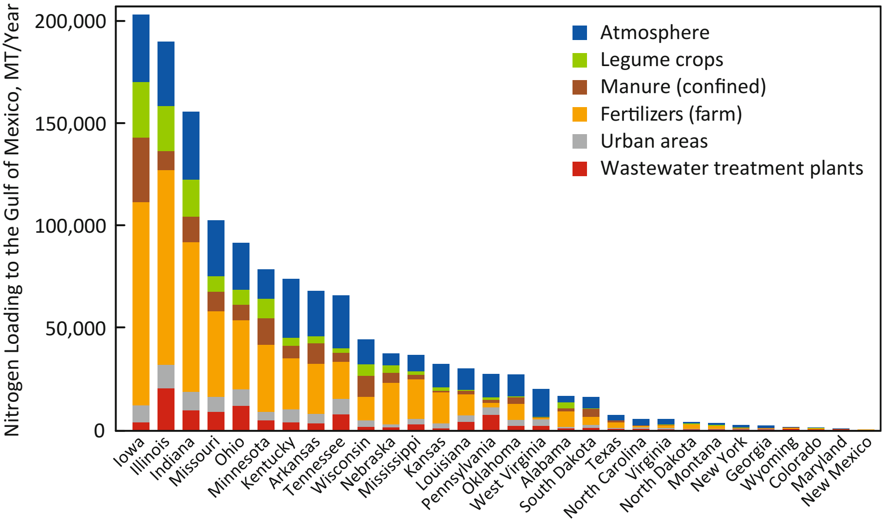

Fig. 2.6 SPARROW model results ranking each state in the Mississippi River Basin, USA, based on their relative contributions of nitrogen to the Gulf of Mexico. The relative importance of each of the nitrogen sources is shown for each state (modified from Robertson et al. 2014)

the largest source of nitrogen to the Gulf of Mexico (60% of the total), followed by atmospheric deposition (26%) and urban areas (14%) (Fig. 2.6). Model results were also used to describe the dominant source of nitrogen and phosphorus for various sub-basin and political boundary scales.

SPARROW results demonstrated that the areas contributing the highest amounts of nitrogen were not necessarily contributing the highest amounts of phosphorus. Highest nitrogen and phosphorus yields were from sub-catchments dominated by wastewater treatment plants; however, in agricultural areas, the nitrogen yields were highest in areas with crop-oriented agriculture (primarily associated with high farm fertiliser input rates) in the Corn Belt, whereas highest phosphorus yields were in areas with crop- and animal-oriented agriculture, which are more widely distributed. The model also demonstrated the differential effects in how the nutrients are delivered to streams. Nitrogen is more efficiently delivered to streams in areas with tile drains than is phosphorus.

Information from large catchment models, such as shown here from SPARROW, may assist managers in guiding restoration efforts to reduce nutrient loading from very large areas by describing where efforts could have the largest effects (based on ranking results) and what types of actions would have the largest impact (based on source-type allocation results).

In general, most large-scale catchment models are regression-based models, such as the Watershed Regression for Pesticides, WARP (Larson et al. 2004), or hybrid models, such as SPARROW, because it is difficult to obtain all of the detailed information required in a process-based model, such as SWAT. Process-based models, such as SWAT models for the Mississippi River Basin, however, can also be developed for large catchments if the input information can be estimated (White et al. 2014). Estimating the information required for process models applied to large areas, such as the exact timing of nutrient inputs, specific management actions, and meteorological forcing, can be difficult in the spatially explicit manner needed to obtain accurate model results.

2.7 Use of Catchment Model Output for Lake and Reservoir Modelling

Output from catchment models is commonly used as input into lake and reservoir models to understand how the landscape may be influencing water quality of standing waters or to project how future changes may affect their water quality and potentially guide restoration efforts (see Chap. 3). If catchment modelling results are planned to be used in combination with lake modelling, it is important to know the requirements for the lake or reservoir model prior to choosing the catchment model. Many statistically based lake models, such as the Canfield–Bachmann equation used to estimate in-lake phosphorus concentrations (Canfield and Bachmann 1981) or the empirical relations contained in the U.S. Army Corps of Engineers BATHTUB model (Walker 1996) used to estimate in-reservoir phosphorus and chlorophyll-*a* concentrations and water clarity, use annual or seasonal inputs that may be obtained from statistically based catchment models. The BATHTUB model, for example, has been applied to Lake Waikare in New Zealand (Cox 2015).

Seasonal or annual results from statistically based catchment models or daily results from processed-based catchment models that have been summarised into seasonal or annual summaries may be linked and used with the statistically based lake models. However, process-based lake/reservoir models, such as the one-dimensional DYRESM-CAEDYM model (formed by two different components: the hydrodynamic model DYRESM (DYnamic REservoir Simulation Model) and the water quality modules in CAEDYM (Computational Aquatic Ecosystem Dynamics Model) (http://ecobas.org/www-server/rem/mdb/caedym.html), the one-dimensional GLM (General Lake Model) (http://aed.see.uwa.edu.au/research/models/GLM/) and the two-dimensional CE-QUAL-W2 model (http://www.ce.pdx.edu/w2/) that simulate temporal and spatial changes in various water quality parameters on a daily time-step, typically require daily inputs from the catchment. Therefore, process-based lake models typically need to be linked with process-based catchment models, such as SWAT, which provide daily load estimates. However, some process-based lake or estuary models run at an annual time step, such as the USC-3 (Urban Stormwater Contaminant) model (Green 2008) that estimates sedimentation and accumulation of contaminants in the bed sediments of estuaries, and can use input from either statistically based catchment models or summarised daily inputs from process-based catchment models, such as GLEAMS or SWAT.

The combined use of catchment models and lake/reservoir water quality models is a common approach to determine what actions are needed in a catchment to obtain a specified water quality goal, such as defined in lake management plans (SEWRPC 2015) or TMDL investigations (MPCA 2007, 2010).

2.8 Benefits of Integrated Monitoring and Modelling

Management of water resources around the world requires integrated monitoring and modelling efforts. Monitoring provides direct observations of water quality properties and characteristics, whereas modelling provides the ability to interpret and extrapolate those observations and project how various actions should affect water quality.

Monitoring efforts are typically aimed at quantifying the condition of selected catchments and the downstream estuaries, reservoirs, and lakes. Monitoring is usually aimed at: (1) characterising current conditions and describing historical changes in water quality, (2) identifying specific existing or emerging water quality problems, (3) gathering information to design specific pollution prevention or restoration programmes, (4) determining compliance with regulations or implementation of control measures, and (5) responding to emergencies such as spills and floods. Conditions monitored may be physical (e.g. temperature, flow, sediments), chemical (e.g. dissolved oxygen, nutrients, metals, oils, and pesticides), or biological (e.g. abundance and variety of aquatic plant and animal life, and the ability of test organisms to survive in sample water). Monitoring can be conducted at fixed stations, random sites throughout an area, or at sites chosen on an emergency basis (such as after a spill or discharge) and conducted on a continuous basis, seasonal basis (e.g. summer swimming), or on an as-needed basis to answer specific questions.

In New Zealand, regional councils and unitary bodies are primarily responsible for monitoring and reporting of water quality information. However, other organisations, such as universities, environmental groups, private citizens, research communities, and permitted dischargers (e.g. local councils), also conduct water quality monitoring. The National River Water Quality Network (NRWQN), which began in 1989, provides reliable scientific information describing many important physical, chemical, and biological characteristics of selected rivers in New Zealand. Land, Air, Water Aotearoa (LAWA) provides results of water quality monitoring throughout New Zealand. Initially formed as a collaboration of New Zealand's 16 regional and unitary councils, LAWA is now a partnership between the councils, Cawthron Institute, Ministry for the Environment, and Massey University. LERNZdb (2015), developed by Lake Ecosystem Restoration NZ, is a data repository that supports the storage and retrieval of water quality and biological data collected from lakes, rivers, and wetlands in New Zealand.

The future of water quality monitoring in New Zealand, as well as other places in the world, is likely to involve citizen science (Chap. 16). Citizen science provides additional data collected by the community, which should lead to additional water quality success stories. One example is NZ Landcare Trust's Aorere River project. Local marine farmers in the Aorere River Basin faced closure due to deteriorating local water quality. The dairy farming community recognised the need to reduce bacteria being delivered to waterways. Therefore, monitoring was done by local farmers, supervised by scientists, and intensive modelling of nutrient and pathogen delivery from the catchment was performed. Results of these efforts were then presented to the Tasman District Council and local marine farmers. As a result of the modelling supported by citizen-based monitoring, best management practices were implemented to eliminate stock access to waterways, stop effluent irrigation to saturated soils, and reduce effluent application rates, especially in the model-directed hotspots. All of the farmers in the catchment took ownership of their environmental performance and did not just see this effort as simply a compliance issue. In 2002, the local shellfish harvest rates were as low as 28% of the historical rates. In 2006, when the project was officially started, harvest rates were around 50% of the historical rates. But by 2009, the harvest rates reached 80% of their historical rates. In 2015, NZ Landcare Trust's Aorere Project won the prestigious Morgan Foundation NZ River Prize.

Together monitoring and modelling are key to the management of water quality and the restoration of impaired downstream water bodies. Monitoring provides the information to develop and validate catchment models. Modelling provides an understanding of the causes of deteriorated water quality and helps guide restoration efforts to improve water quality. Modelling results can also identify gaps and prioritise future monitoring, as well as identify monitoring that might be unnecessary.

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Chapter 3
Modelling Water Quality to Support Lake Restoration

Moritz K. Lehmann and David P. Hamilton

Abstract Numerous applications of deterministic models have been used to support decision making in relation to lake restoration actions in New Zealand. The most widely used are one-dimensional, coupled hydrodynamic-ecological models suitable for long-term (multi-year) simulations to explore inter-annual variability and progressive changes in response to restoration actions and global change drivers (e.g. climate change). Three-dimensional models have also been used to examine, for example, spatial variability associated with inter-basin circulation transfers in a deep hydrolake, dispersion of geothermally heated waters in a shallow volcanic lake and a double gyre circulation pattern influencing dispersion of inflows, including a wastewater discharge, to a large volcanic lake. We provide a framework for categorising these applications based on theoretical, heuristic and predictive considerations. Information on model selection, data assimilation and calibration processes presented in this chapter are designed to support increasingly sophisticated modelling approaches, many of which will be supported by autonomous sensor data. The need for these types of models is likely to increase in the future as they are used to support the goals of the National Policy Statement for Freshwater Management (MfE 2014) to maintain or improve water quality. The models will be used to assess the ecological outcomes of potential restoration actions as part of an integrated assessment that includes expert knowledge, economic considerations and social outcomes.

Keywords DYRESM-CAEDYM · ELCOM-CAEDYM · Restoration scenarios · Theoretical model · Heuristic model · Predictive model · Climate change · Calibration · Data assimilation

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3.1 Introduction to Lake Models

The modelling of water quality to support lake restoration is a multi-task activity which involves coalescence of several major factors. These factors include (1) close interaction with stakeholders and clients who will have specific expectations of model outputs, (2) collation of background information and available data for input to a model, (3) model formulation with calibration and validation, (4) documentation of methodologies and assumptions so that model simulation results are transparent and reproducible and (5) generation of scenarios to inform decision making. Modelling may be viewed as a workflow in which setting up and running a computer simulation is one component of a larger scope involving shared understanding of the problem and agreement on the approach, development or selection of an appropriate model, definition of acceptable performance criteria and quantification of limitations and uncertainties associated with the model outputs. In this chapter, we begin with a broad description of the modelling process in order to provide a foundation for summarising the main features of about 40 lake modelling studies carried out in New Zealand including categorising the model applications based on fundamental questions asked of them (see Table 3.1). Many of these modelling studies have been aligned with addressing options to manage lake water

Table 3.1 Applications of models to New Zealand lakes, describing the lake, reference, objectives and key result of simulations

Lake	References	Objective of model application	Key simulation result
Benmore	Norton et al. (2009)	Predict relationships between nutrient loads under different land use scenarios and lake condition indices (e.g. TLI) using 1-D and 3-D models	Chlorophyll *a* in the epilimnion increased and dissolved oxygen decreased in the hypolimnion as nutrient loads increased. A four- to six-fold increase in nutrient loads (for the two major inflows) resulted in a shift from oligotrophic to eutrophic status
	Trolle et al. (2014b)	Examine spatial variability in lake response to increased nutrient loads associated with agricultural intensification using a 3-D model	Lake trophic status changed from oligotrophic (TLI = 2–3) to eutrophic (TLI = 4–5) when external loadings increased eight-fold. The shallow basin responded most strongly to increased nutrient loads
Brunner	Spigel and McKerchar (2008)	Model patterns of thermal stratification so that a reliable hydrodynamic model could set a basis for further water quality simulations	Patterns of stratification and mixing were generally well represented but occasional mis-matches of temperature profiles were attributed to problems with input data and limitations of the model

(continued)

Table 3.1 (continued)

Lake	References	Objective of model application	Key simulation result
Ellesmere/ Te Waihora	Trolle et al. (2011)	Quantify the effects of a future climate and land use change to examine long-term planning options and implementation of lake management strategies using a 1-D model	The lake became increasingly eutrophic and summer phytoplankton blooms intensified in the presence of a warmer climate and without management interventions. Effects of major catchment nutrient load reductions are delayed by internal legacies
	Allan (2014)	Examine spatial distributions of suspended minerals using 1-D and 3-D models and model validation with satellite remote sensing observations	Temporal variations in suspended minerals simulated by 1-D and 3-D models were similar and conformed broadly to suspended mineral distributions from MODIS data. 3-D simulations tended to over-estimate suspended mineral concentrations in shallow lake margins
	Hamilton et al. (2017)	Achieve a simultaneous water and salinity balance for the lake and examine effects of possible future land and lake water level management regimes using a 1-D model	Strong dominance of internal over external nutrient loads due to frequent sediment resuspension. Lake opening scenarios indicate water quality benefits from higher water levels and reduced sediment resuspension
Hakanoa	Paul et al. (2011)	Provide assessments of the water quality effects of different restoration scenarios using a 1-D model	The combination of land use change from pastoral to forestry, expansion of wetlands and removal of septic tanks significantly improved lake trophic state
Martha Pit Mine	Castendyk and Webster-Brown (2007a, b)	Environmental risk assessment for a mining void to evaluate sensitivity of water quality to pit morphology, circulation patterns and pH	Circulation in the filled mining void was not particularly sensitive to changes in basin morphology due to the strongly seasonal patterns of stratification. pH is highly important in geochemical dynamics

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Table 3.1 (continued)

Lake	References	Objective of model application	Key simulation result
Ngāroto	Paul et al. (2008)	Assess restoration scenarios using a 1-D model: diverting a major inflow, changing lake water level, reducing external loads and dredging lake sediments	External nutrient load reduction equating to land use change had the most benefit to water quality. Diversion of the major inflow and raising the lake level resulted in only small changes in water quality
	Lehmann et al. (2017)	Assess whether inflow diversion, geochemical engineering of bottom sediments, water level increases and catchment nutrient load reductions would help address the degraded state of this shallow lake	No scenario resulted in the status of the lake improving from a fail (D band) to acceptable (C band) in terms of the NPS-FM (2014). The most effective scenario equated to geochemical engineering and catchment nutrient load reductions by more than 50%
Ohinewai	Allan (2016)	Examine effects of riparian buffer development, catchment nutrient reductions and pest fish removal on lake nutrient budgets and sediment resuspension using a 1-D model	Pest fish contribute significantly to internal loads of sediments and nutrients but a combined strategy, also involving external nutrient load reductions, is required for successful lake restoration
Okāreka	Trolle et al. (2011)	Study the effects of a future climate on Lake Okāreka in order to support long-term planning and implementation of lake management plans using a 1-D model	A warmer climate will make it more difficult to achieve water quality targets. Increases in external nutrient loads could shift the lake from oligo-mesotrophic to eutrophic state under a warmer climate
Ōkaro	Özkundakci et al. (2010, 2011)	Investigate whether further nutrient loading reductions are required despite a recent intensive catchment management programme, complemented by in-lake geoengineering	The trophic status of the lake, measured quantitatively with the Trophic Level Index (TLI), may shift from highly eutrophic to mesotrophic with external and internal loads of both N and P reduced by 75–90%
Ototoa	Spigel (2007)	A hydrodynamic modelling study to examine opportunities created by good thermal profile simulations and extend the modelling to including climate change and water quality simulations	Stratification and mixing patterns (timing, strength of stratification, thermocline depth) were well captured by the model. Limited duration of meteorological data collection was problematic for long-term model runs

(continued)

Table 3.1 (continued)

Lake	References	Objective of model application	Key simulation result
Rotoehu	Trolle et al. (2011)	To quantify the effects of future climate on Lake Rotoehu with a view to supporting long-term planning and implementation of lake management plans	Future climate scenarios predicted a 9% and >25% increase, respectively, in summer average chlorophyll *a* concentrations and summer biomass of cyanophytes relative to the base scenario. If left unmanaged, Lake Rotoehu will become increasingly eutrophic and likely be subject to more frequent harmful algal blooms
	Allan et al. (2016)	To assess the spatial heterogeneity of Lake Rotoehu surface water temperature simulated with a 3-D hydrodynamic model using surface water temperature retrieved from Landsat	Simulations reproduced the dominant horizontal variations in surface water temperature in the lake. Accurate satellite-based thermal monitoring could potentially be used to validate water surface temperature simulated by 3-D hydrodynamic models
Rotoiti	Priscu et al. (1986)	To use a 1-D model to explore effects of heating from a geothermal vent in the bottom of the lake	Hypolimnetic stirring resulted in this layer being very homogeneous, with solutes dispersed relatively evenly through the layer
	Hamilton et al. (2005)	Use a 1-D model to simulate changes in temperature, dissolved oxygen and trophic status of Lake Rotoiti in response to a proposed diversion wall to redirect the main inflow directly to the lake outflow	Cyanobacterial biomass in the lake was reduced by about 40%. The improvements in water quality predicted from the model provided part of the justification for implementation of the diversion wall
	Stephens (2004)	A 3-D hydrodynamic study to inform design of a wall in Lake Rotoiti to divert a high-nutrient inflow (Ohau Channel) directly to the lake outlet	The results informed the design length of the wall required to prevent backflow into Lake Rotoiti

(continued)

Table 3.1 (continued)

Lake	References	Objective of model application	Key simulation result
	Von Western-hagen (2010)	Analyse temporal and spatial variations (horizontal and vertical) in water quality in Lake Rotoiti using a 3-D model	Extensive field measurements (temperature, dissolved oxygen, chlorophyll) allowed for comprehensive model validation. They showed a morphologically complex, geothermally influenced lake. Observed nutrient concentrations were relatively poorly simulated
	Jones et al. (2013)	Quantify, using a 3-D model, whether effectiveness of a diversion wall may be compromised by variable outflows which facilitate rafting operations in the outflow stream	At times of low inflow when outflows are artificially reduced to hold back water for alternate periods of high outflow when rafting operations take place, the inflow may flow around a diversion wall and enter the main basin of Lake Rotoiti; a minimum operating outflow is required to avoid such an occurrence
Rotokakahi	Jones et al. (2014)	Evaluate using a 1-D model the causes of decline of water quality that may have been associated with forest harvesting operations and increased sediment and nutrients in ephemeral inflows	Large error terms in modelled variables may have been associated with difficulty in assessing discharge, sediment and nutrient loads in ephemeral streams and groundwater, which are the dominant hydrological contributors to the lake
Rotokauri	Sharma (2011)	Determine how staged urban development affects the water quality of Lake Rotokauri	Protection of lake water quality during urban infill requires sound management practices such as infiltration at source, and natural and constructed riparian wetlands
Rotomānuka	Lehmann et al. (2017)	Examine with a 1-D model possible management actions to meet goals in the NPS-FM (2014) to improve water quality from a D ('fail') to a C band ('pass'; NPS-FM (2014))	Water quality goals (C band, NPS-FM (2014)) may be attained with good land management practices, despite anoxia of bottom waters during seasonal stratification. Enhancing riparian wetlands will assist with meeting water quality goals

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Table 3.1 (continued)

Lake	References	Objective of model application	Key simulation result
Rotorua	Rutherford et al. (1996)	Estimate with a 1-D model the rate of recovery of water column P, hypolimnion DO and nutrients in bottom sediments following removal of treated wastewater discharge to the lake	Reductions in lake water TP concentration take place over approximately 20 years following sewage diversion but sediments take about 10 times longer for P to recover to pre-wastewater discharge levels
	Burger et al. (2008)	Use a 1-D model to examine benthic nutrient fluxes in relation to temporary stratification and anoxia, as well as cyanobacteria dominance	Showed a massive internal flux of N and P associated with anoxia from temporary stratification, which was similar in magnitude to external loads. This was identified as a driver for summer blooms of cyanobacteria
	Özkundakci et al. (2012)	Carry out historical (starting 1920s), current and future simulations with a 1-D model to quantify effects of land use change and future climate change	Simulations showed that sediment nutrient release rates are sensitive to changes of bottom sediment composition and external nutrient loads through time
	Hamilton et al. (2012b)	Provide quantitative assessments of water quality using a 1-D model to evaluate effects of land use change, climate warming and in-lake management actions	Removal of 350 t N y^{-1} and a correspondingly large fraction of P from the lake would be required to achieve a sustained reduction of TLI to the target value of 4.2. Load reductions required to meet the TLI target increased with climate warming
	Abell and Hamilton (2014)	To study how dynamic variations in water, nutrient and sediment transport in a stream inflows interacted with circulation processes and affected water quality during a 5-day period of high rainfall in the summer	This study highlighted how characteristics of littoral-pelagic coupling can influence fine-scale spatial and temporal variations in the water quality of large lakes
	Hamilton et al. (2014b)	Apply a 1-D model to examine effectiveness of alum dosing of two inflows to the lake and to compare these with external nutrient load reductions	Alum dosing was associated with greater sedimentation of particulate organic nutrients and reduced nutrient releases from bottom sediments, in addition to direct effects in reducing P in the two inflows

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Table 3.1 (continued)

Lake	References	Objective of model application	Key simulation result
	Abell et al. (2015a)	Environmental effects study using 1-D and 3-D models to assess the discharge of treated wastewater on nutrient and bacteria levels in Lake Rotorua	Highly treated wastewater discharge had little impact on TLI in Lake Rotorua based on 1–D model simulations. 3-D simulations showed dispersion of treated wastewater was strongly affected by lake-scale circulation patterns
Taupo	Spigel et al. (2005)	Use of field data to validate the entrance of cold inflows into the largest lake in New Zealand	Field data from large, cold, plunging inflows was used to adjust the way that a 1-D model simulated the inflow path. The model proved satisfactory for tracking inflow insertion depths and entrainment of lake water
Tikitapu	McBride et al. (2014)	Apply a 1-D model to evaluate septic tank removal and replacement with a reticulated sewage system and to evaluate phytoplankton silica limitation	Simulation of septic tank reticulation resulted in substantial water quality improvements within a time frame of 2 years
Tutira	McBride and Hamilton (2016)	Coupled catchment and lake (1-D) model applications for the assessment of management options for Lake Tutira, e.g. re-diversion of inflows, land use change, aeration, geoengineering, sediment capping	Modelling results prompted more detailed investigations into aeration and destratification, as well as providing impetus to improve land use practices to reduce external nutrient loads
Waahi	Jones and Hamilton (2014)	Assess inundation of the lake bed and surrounding floodplain for reinstatement of higher water levels using 3-D model	A substantial area of floodplain around the lake is currently protected by stopbanks, limiting the intended benefits from raising water levels
	Lehmann et al. (2017)	Evaluate restoration options to enable water quality to meet the 'C band' in the NPS-FM (MfE 2014) using a 1-D model	The simulated restoration options were not sufficient to meet the bottom line in the NPS-FM (MfE 2014). Scenarios driven primarily by a large reduction in the internal nutrient load (hypothetically through geochemical engineering) showed the greatest improvement

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Table 3.1 (continued)

Lake	References	Objective of model application	Key simulation result
Waikare	Cox and Cooke (2014)	Assess effectiveness of lake flushing with water from the Waikato River using the Simplified Lake Analysis Model	Trophic status improved considerably with flushing but displacement of pollutant load to downstream waterways and inability to address high catchment loads represented a major risk for this action
Wainono Lagoon	Abell et al. (2015b)	Assess changes in trophic state from altered external nutrient loads and augmentation with high-quality river water using 1-D and 3-D models	Flow augmentation could offset increases in TLI from nutrient load increases from catchment land use intensification. Large internal sediment and nutrient legacy is perpetuated with wind resuspension
Waituna Lagoon	Hamilton et al. (2012a)	Determine whether various management scenarios can prevent a regime shift and sustain *Ruppia* sp. populations using 1-D and 3-D models	Timing and duration of opening of the lagoon had major impacts on water quality and ecology and could partially offset major increases in nutrient loads from land use intensification
Wakatipu	Bayer et al. (2013)	To assess sensitivity to changes in climate in a deep lake subject to strong wind forcing using 1-D model	Projected increased air temperature produced higher lake temperatures and stronger stratification that was not offset by cooling effects of projected increases in rainfall, river flows and wind speeds
Wanaka	Bayer et al. (2013)	See Lake Wakatipu	See Lake Wakatipu
Whangape	Jones and Hamilton (2014)	To assess potential inundation of areas around the lake under a regime of higher water levels using 3-D model	Stopbanks around the lake restrict floodplain habitat by approximately 25% compared with historical inundation at a given water level

TLI Trophic Level Index

quality and mitigate eutrophication problems. Four case studies are used to provide technical detail on some of the lake models. International perspectives (Feature Boxes 3.1 and 3.2) emphasise many of the attributes that have been sought in the New Zealand model applications:

- Good model documentation, transparency and, where possible, open source model code—see Feature Box 3.1.
- Models of sufficient complexity to capture feedback loops and non-linear pathways that are common in lake ecosystems.
- Ability to quantitatively measure the accuracy of model outputs (e.g. using statistical metrics)—see Feature Box 3.2.
- Opportunities for code development to incorporate important variables and processes.

Models are routinely developed for a wide range of complex environmental systems and may include processes to represent climate, hydrology, ecology, biogeochemistry, fate and transport of pollutants and population dynamics. In the context of lake water quality and its restoration, a model that can be usefully applied must describe relevant indicators, such as chlorophyll and nutrient concentrations, and how these change in response to potential restoration actions. A model needs to demonstrate that it can simulate observed data to acceptable levels of accuracy before it can be useful for generating predictions of the future state of an ecosystem and its response to perturbations such as management actions, extreme weather events or long-term environmental change. Model accuracy is commonly quantified with a suite of statistical metrics that indicate closeness of fit of simulated to observed data and any bias (i.e. under- or over-prediction) (Bennett et al. 2013).

Ecological models that output a large number of state variables will have a number of parameters that require adjustment and there are several methods to achieve this adjustment, such as setting acceptable values from parameter ranges in the literature, trial-and-error calibration or autocalibration where parameter values are repeatedly and automatically adjusted within preset ranges. Some combination of all three methods is often useful to balance acquisition of knowledge of processes and model output sensitivity to parameter adjustments. The calibration process is generally considered to be complete when statistical error between observed data and model output data has been reduced to some acceptable level. A formalised sensitivity analysis (e.g. Schladow and Hamilton 1997; Makler-Pick et al. 2011) is sometimes used to identify parameters that are the focus of calibration efforts, but the value of sensitivity analyses may extend much further, i.e. to inform future monitoring efforts or experimental studies (Flynn 2005).

Box 3.1 Mechanistic Lake Modelling Approaches in Europe

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Mathematical models are widely used in the management of lake water quality, particularly for setting maximum allowable nutrient load targets. Simple empirical models, such as those by Vollenweider (1976), have been and still remain widely used for management purposes, whereas more complex mechanistic (also known as process-based or deterministic) models have traditionally been used for research purposes. In recent years, however, mechanistic models have also found increasing use in management (e.g., Trolle et al. 2008), particularly since they are able to account for multiple stressors, such as the effects of changes in nitrogen and phosphorus loads and climate (e.g., Nielsen et al. 2014). One of the earliest examples of mechanistic lake modelling in Europe is the Danish Lake Glumsø model by Jørgensen (1976), which involved dynamics of three trophic levels, including phytoplankton, zooplankton and fish. Additional mechanistic models have since been developed in Europe, such as SALMO (Recknagel and Benndorf 1982), PCLake (Janse 1997) and PROTECH (Reynolds et al. 2001). Since its origin, PCLake has undergone further development and today represents one of the most complex lake ecosystem models available. It describes four trophic levels, and state variables include multiple phytoplankton groups: macrophytes, zooplankton, zoobenthos, and zooplanktivorous, benthivorous and piscivorous fish. A new development by Hu et al. (2016) has also enabled PCLake model applications to physically heterogeneous water bodies, in contrast to the original fully mixed box model. Recent developments in the field of mechanistic lake modelling tend to focus on new modelling tools and different application approaches, rather than implementation of additional and new process understanding. Important examples are ensemble modelling (Nielsen et al. 2014; Trolle et al. 2014a), automatic parameter estimation techniques and databases (Mooij et al. 2014) and frameworks for enhanced model development (e.g. Bruggeman and Bolding 2014, see Fig. 3.1). Collectively, these tools and approaches are meant to stimulate model development and also support model inter-comparisons and thereby increase the transparency and reliability of model predictions.

Box 3.2 Lake Modelling: An International Perspective

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Lake modelling is widely used around the world as an aid to lake management, in many cases to help plot a course towards restoration. Food web models are used to understand the impacts of invasive species on lake ecosystems (Langseth et al. 2012), fisheries stock assessment models are used to set catch limits (Tsehaye et al. 2014) and complex biogeochemical models are used to understand heavy metal contamination and lake acidification (Hipsey et al. 2014). By far the most popular application of lake models, however, has been to shed light on the drivers and triggers of harmful algal blooms and predict how they might respond to planned or inevitable environmental change. The last decade has seen a renaissance in lake modelling research, following several paths of improvement. These have included better integration of lake and catchment models, better representation of ecophysiological processes at an algorithm level, improved practices in model evaluation and an increased awareness of model uncertainty (Robson 2014a). Sources of model uncertainty include observational error, algorithm and structural uncertainty and uncertainty in specification of parameter values. Disentangling the effects of these interacting sources is not an easy task. Understanding uncertainty is, however, essential if we are to more confidently use models to guide lake management decisions. A model that performs well in some circumstances may become unreliable if applied in a context where its assumptions may not be valid and sources of uncertainty are increased. For example, a model that can reliably predict the onset of a harmful algal bloom under current conditions is not necessarily suited to predicting the long-term impacts of climate change on the occurrence and severity of such blooms. There are several steps that the lake modellers can take to address this. These include careful documentation of the choices and assumptions made in developing and evaluating models, improved use of scientific literature and community datasets to specify the values and probability distributions of model parameters and adoption of methods such as Bayesian hierarchical modelling that allow explicit consideration of uncertainty (Robson 2014b). Natural resource managers, for their part, can demand such documentation and can adopt adaptive management protocols that are designed to facilitate decision-making with uncertain predictions.

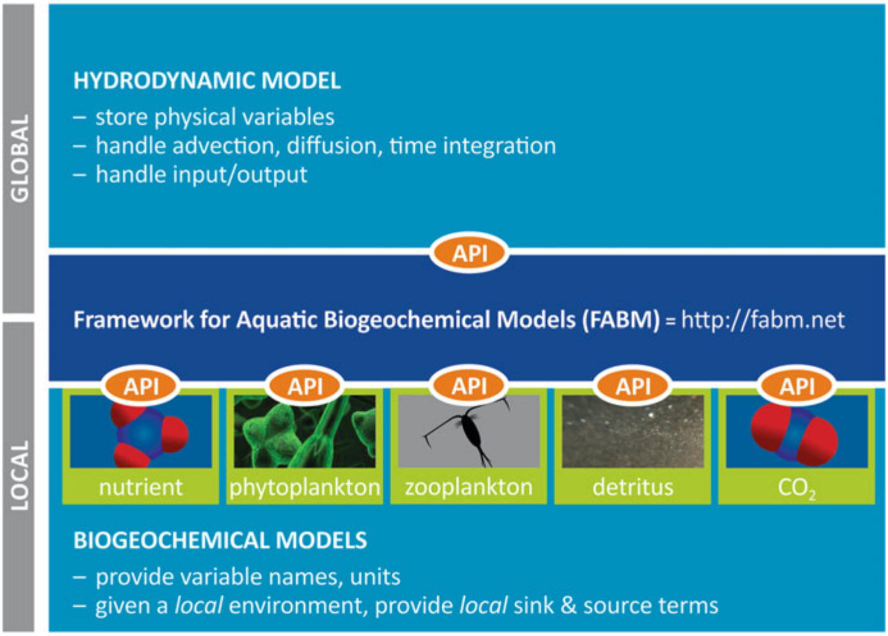

Fig. 3.1 Example from Bruggeman and Bolding (2014) describing the Framework for Aquatic Biogeochemical Models (FABM), where FABM provides the 'glue' between hydrodynamic and biogeochemical processes and API is the Application Programming Interface. The aim of FABM is to ease the integration of physical and biogeochemical processes and thereby enhance code development and quality, reducing the time and cross-disciplinary knowledge required by an individual model developer (for more information, see fabm.net)

3.2 Types of Models

A model is a simplified abstraction of a complex system. For the model to be useful, it must describe or replicate with reasonable accuracy, phenomena or processes that occur in environmental systems. Fundamentally, a model will improve knowledge, allow for new insights or hypotheses to be generated and provide useful and accurate predictions. It may also give an opportunity to thoroughly evaluate, and possibly question, the relevance and accuracy of data that are usually collected routinely for a system, e.g. as part of a lake monitoring programme. In a more formal sense, an environmental model uses a series of equations to represent and codify some aspect of natural phenomena, using equations compiled as a program on a computer (Zeckoski et al. 2015). Models cannot perfectly describe all aspects of complex phenomena and modellers need to be aware of, and clearly communicate, the limitations and errors that are inherent in the models they apply.

3.2.1 Empirical Models

Empirical lake models represent a group of statistical models and have formed the basis of eutrophication predictions for many decades (e.g. Vollenweider 1976, Brett and Benjamin 2008), with several applications to New Zealand lakes (Pridmore 1987; Abell et al. 2010). These models generally predict phytoplankton biomass (commonly using chlorophyll *a* as a proxy) from water column total phosphorus concentrations using regression relationships developed for a single lake or multiple lakes. Total phosphorus concentrations can be derived from a simple equation which is generally based on a steady-state response to external phosphorus loads, lake water residence time and a net phosphorus sedimentation rate (see Chap. 5). These models are often statistically reliable, i.e. the phosphorus–chlorophyll relationship or the net phosphorus sedimentation rate is derived for a large number of lakes; however, they are often of limited value for investigating responses to transient or acute events that lake managers may face (Harris 1997). For example, algal blooms, fish kills or regime shifts are often transient events but can be a major driver for decision-making in relation to lake management actions.

3.2.2 Neural Networks

Artificial Neural Networks (ANNs) are a type of model which commonly emphasises an automated mode of feedback (‘learning’) based on improved statistical fit using successive iterative comparisons with observed data. Artificial Neural Networks can have many key variables (nodes) which are linked together by statistical, probabilistic (commonly binary) or process functions. There can be many nodes as well as many layers, including input, output and ‘hidden’ (processing) layers. Multiple layers can result in a highly computationally intensive model. Artificial Neural Networks have been used for short-term predictions of lake water quality associated with occurrences of algal blooms or bottom-water anoxia (Recknagel et al. 2015). Bayesian Belief Networks (BBNs) are a subset of ANNs which use a graphical representation of key variables (nodes) and link these variables through a set of probabilistic functions (i.e. conditional dependencies). The Bayesian element of these models reflects their mix of understanding and belief formulations. Bayesian Belief Networks have been used to support a diverse set of water management decisions (e.g. Arhonditsis et al. 2007; Stewart-Koster et al. 2010) and can be of value in encouraging co-development (e.g. between scientists, water managers and community members). They may also be readily updated and enhanced by refining the probabilistic functions that connect nodes and do not necessarily require highly technical modelling skills. On the other hand, BBNs may have limitations which relate to their complexity, especially when required to capture temporal and spatial variability (e.g. as separate nodes) of a system. It may also be difficult to describe conditional dependencies in terms of discrete probabilities when they are actually continuous variables. The most comprehensive demonstration of their utility in New Zealand is the integration of environmental, economic and social decision making under considerations of the environmental effects of irrigation-driven agricultural intensification (Quinn et al. 2013).

3.2.3 Deterministic models

In this chapter, we focus on a subset of models (hereafter referred to as deterministic) which have variously been termed dynamic (i.e. time-resolved), mechanistic (i.e. process representations) or deterministic. The term deterministic refers to a single model simulation output which arises from the initial conditions and parameter values entered as input to the model. The processes in a deterministic model are most often represented as a series of mathematical equations of varying complexity. Many of these equations are numerical or empirical solutions for key processes; hence, many of the parameters in these equations require calibration. Deterministic environmental models offer opportunities for exploratory or predictive management applications that extend beyond the bounds of input data, to different time periods (e.g. for climate change applications) or to a range of systems (Robson et al. 2008).

Modellers generally use existing deterministic models for lake applications as most questions asked of the models require simulation of several state variables and it is too time consuming to construct a new model for each individual application. Feature Box 3.1 discusses the concept of developing utility and capability in existing models through using the collective knowledge and expertise of a global lake modelling community. Conceptualisation of processes and functions is still an important part of the modelling procedure, however, and processes that operate within a system are often summarised in simplified diagrams. Figure 3.2 shows a conceptual model of the spatial representation of a lake using a one-dimensional (1-D) and a three-dimensional (3-D) hydrodynamic base model, as well as a highly simplified food web representation.

From a conceptual model, a process-based model can be formulated in which specific physical, chemical or ecological processes are represented as mathematical equations and coded and linked through a computer language (e.g. Fortran, C++ or in an Excel spreadsheet) to form a working computer model. Modern computers have capacity to perform thousands of calculations per second and therefore enable a level of integration of physical, chemical and biological processes that is not otherwise possible from simple empirical or statistical relationships. Complex process-based models may, therefore, offer opportunities for exploratory or predictive management applications that extend beyond the bounds of input data and to different times (e.g. climate change applications) or a range of systems (Robson et al. 2008).

The level of complexity of process models varies widely depending on: the size and desired degree of resolution of the system (e.g. zero through three dimensions); the number and type of state variables simulated (e.g. temperature, dissolved

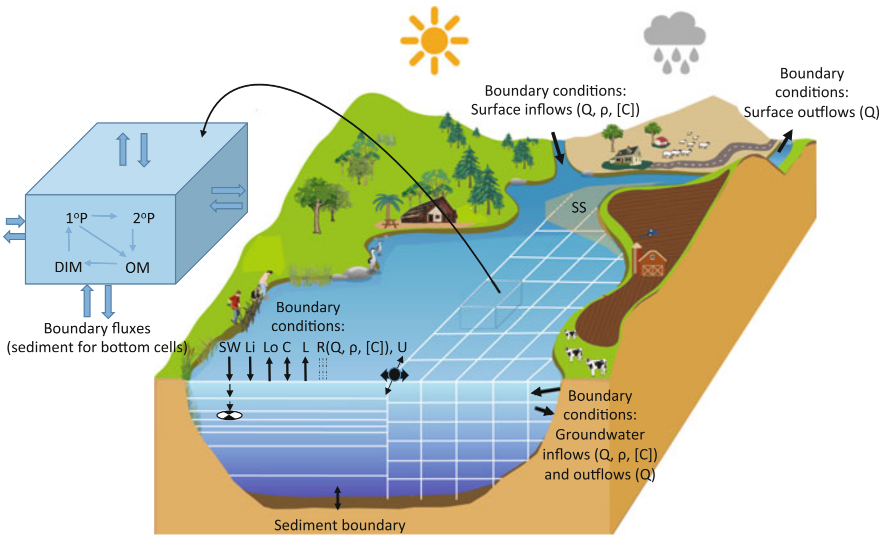

Fig. 3.2 One-dimensional (left-hand side) and three-dimensional (right-hand side) representation of a lake. The white lines in the lake represent model cells. Boundary conditions include discharge (Q), density (ρ) and composition [C] for inflows, Q for outflows, and shortwave radiation (SW), longwave input (Li), longwave emission (Lo), conductive (C) and latent (L) heat exchanges, rainfall (R) for which Q, ρ and [C] can be defined (e.g. to represent atmospheric deposition where [C] would be nutrient concentrations in rainfall), and wind (U). Note that shortwave radiation is represented to penetrate below the water–atmosphere boundary layer. SS represents suspended sediment (e.g. from sediment resuspension). A single cell in a three-dimensional model (left-hand box) represents biogeochemical processes in very simple terms, showing relationships amongst phytoplankton (taking up dissolved inorganic matter, i.e. nutrients), zooplankton (consuming phytoplankton) and organic matter (e.g., nutrients in organic form) that take place at each time step. Boundary fluxes would include atmospheric exchanges if the cell was at the surface or sediment-water fluxes if the cell was at the bottom (e.g. sedimentation of nutrients out of the water column, sediment resuspension and/or dissolved nutrient exchanges)

oxygen); the degree to which the models attempt to simulate ‘reality’ (e.g., simulating a community, population or individual); and whether the model is steady state (i.e., at equilibrium) or dynamic (i.e., time-varying). The choice of spatial dimension (e.g., 1-D or 3-D; see Fig. 3.2) has important implications for the complexity and computer run-times of a model simulation. Highly resolved 3-D models, for example, can potentially take many hours of computer run time on a PC to complete a 1-year simulation of an aquatic system.

Critics of these complex process models suggest that they are either too simple to properly represent the enormous complexity of natural ecosystems or too complex and, therefore, cannot be calibrated sufficiently well to be insightful. The quote credited to statistician George E. P. Box that ‘essentially, all models are wrong, but some are useful’ (Box and Draper 1987) may be attributed in part to misalignment of available model input data, model complexity and questions asked of a model. For example, most 3-D models do not contribute predictions on inter-annual variations in a lake ecosystem because the duration of the simulation period is restricted by lengthy computer run times and high grid resolution; however, these models can reveal important information on spatial variability over short (e.g. diel) to intermediate (e.g. seasonal) time scales.

3.3 Selecting an Appropriate Model

For the purpose of managing aquatic ecosystems, careful model selection is required. Simple models generally do not have extensive data input requirements. The dedication of scientific resources to assimilate data, run simulations and present output data can, therefore, be greatly reduced compared with more complex deterministic models. These models may be useful when rapid turnaround is required so that managers and stakeholders can quickly estimate directional changes and order of magnitude effects for a suite of possible ecosystem mitigation actions (i.e. scenarios). By contrast, applications of complex deterministic models commonly necessitate a high level of modeller skill and expertise, have longer turnaround times and give output data that require careful graphical or numerical analysis. In many cases, the extensive output data generated from these models is simplified to allow comparisons with indicator metrics (e.g., an index for trophic state in the case of water quality) or to predict threshold values in response to environmental perturbations such as climate change or habitat loss (Stillman et al. 2016). Careful analysis, manipulation and presentation of output data are, therefore, critical for these models to be useful to inform management and policy.

To non-modellers, the distinction of model types, details of implementation, dimensionality and the number of modelled variables can be daunting (Trolle et al. 2012). In some instances, different deterministic models may be used in an ensemble modelling approach to generate a statistical distribution to reflect elements of uncertainty in the modelling; an approach well developed for climate models but rarely used for water quality models (see Trolle et al. 2014a for such an exception). It might also be valuable to commence more complex modelling projects with empirical approaches to provide a ‘reality check’ on results that may be expected from more complex model output. Also, in some instances, a combination of deterministic and neural network approaches may be used (Arhonditsis et al. 2007). In many cases, however, time constrains the amount of model exploration that is possible and most applications simply utilise ‘off-the-shelf’ models or include modest adaptations or extensions of a set of fundamental and well-established equations (Janssen et al. 2015).

It may also be instructive to step back from details of model formulations and classify models into theoretical, heuristic and predictive categories (Franks 1995) where (1) theoretical models address the question: ‘what would/could happen if...?’, (2) heuristic models suggest mechanisms to explain the past: ‘how did this happen?’ and (3) predictive models address the future state of the system: ‘what will happen. . .?’. Theoretical models are sometimes used to explore problems that are difficult to investigate given technological, logistical or monetary constraints (Franks 1995). These models include the classic theoretical work on population oscillations due to predator–prey interactions (Lotka 1925; Volterra 1926) and explorations of stability or resilience of ecosystems (May 1974).

Heuristic models may be used to gain insights into what caused particular changes in a system. From the perspective of lake restoration, these models are important because successful hindcasts (i.e., achieving satisfactory fit between simulations and observations) are usually an important criterion for generating confidence that the model will behave realistically when input forcing data are outside the range of past conditions (e.g., in response to restoration actions).

Predictive models go beyond the scope of heuristic models and are used to extrapolate beyond the sampled space. In some cases, there may be a need to predict the state of the ecosystem for days or weeks into the future in a way analogous to a weather forecast, for example, to issue health warnings to recreational or commercial users of water. In such cases, a forecast model is desired to predict ecosystem variables such as water temperature, chlorophyll concentration, cyanobacteria biomass or coliform bacteria counts over time scales of days to weeks. Forecast models are a type of predictive model that uses numerical techniques applied during simulation runs and as new observations are made (e.g., using data assimilation techniques—see Sect. 3.5).

3.4 Model Performance Criteria and Skill Assessment

In order to use models effectively to guide lake management and restoration activities, it is vital to establish an acceptable level of confidence in the model performance. Assessing model performance typically includes calculating skill scores from a range of pairwise statistical approaches that compare simulation output with observed results from the system on which the project is focussed (Lynch et al. 2009; Stow et al. 2009; Bennett et al. 2013). Identifying the variables to be used to assess model performance and agreeing on the metrics to quantify model skill are important components of the modelling workflow. Problems can arise from intrinsic discrepancies between the data collected by biologists and the state-variable outputs of models (Flynn 2005). It is, therefore, important to reinforce the need for field and lab practitioners, scientists generally, as well as managers, to be more closely involved with modellers to ensure that the data collected are comparable to the state-variable outputs of the models (Kara et al. 2012). Flynn (2005) provides an example where models may commonly use a ‘currency’ for phytoplankton biomass expressed in terms of C, N and P whereas those undertaking field measurements may seldom express their measurements as variables other than chlorophyll or organism numbers. Another issue identified by Kara et al. (2012) is a lack of direct comparability of in-vivo sensor data (e.g., chlorophyll fluorescence) with model output variables (e.g., chlorophyll *a*). This is far less of a problem for sensors which measure physical variables (e.g., temperature) for which the main issues are precision, accuracy and drift and which can usually be quantified and corrected. Hamilton et al. (2014a) outline the opportunities enabled by use of sensor data for model calibration and validation, specifically using measurements of chlorophyll fluorescence and phycocyanin for Lake Rotoehu, Bay of Plenty, New Zealand.

The discrepancy between observed and modelled variables is part of the reason why aquatic ecosystem models tend to have low statistical power, especially for biological variables such as phytoplankton biomass. Arhonditsis and Brett (2004) summarised 153 studies to derive estimates of percentage of variation explained (r^2) and relative error (RE) for comparisons of different state-variable outputs from aquatic biogeochemical models against observed data. For a physical variable such as temperature, the fit was good ($r^2 = 0.90$, RE = 7%) but it became substantially weaker for nutrients such as nitrate ($r^2 = 0.68$, RE = 36%), ammonium ($r^2 = 0.39$, RE = 48%) and phosphate ($r^2 = 0.47$, RE = 42%) and for biological variables such as biomass of zooplankton b ($r^2 = 0.24$, RE = 70%) and bacteria ($r^2 = 0.06$, RE = 36%). To counter the low statistical power of ecosystem models, investigators have begun to apply more advanced techniques of data assimilation (Gregg et al. 2009; Dowd et al. 2014) and ensemble modelling (Trolle et al. 2014a).

3.5 Data Assimilation for Modelling

Data assimilation is a generic term given to a variety of techniques to methodologically combine observations and models to obtain the 'best' estimate of the state of a system or in some cases to obtain the 'best' set of parameters for a model (Evensen 2007). The key aspect of data assimilation is that the term 'best' is mathematically defined. Data assimilation techniques are likely to become increasingly important as more real-time data streams from automated high-frequency monitoring stations (Chap. 13) and remote sensing by satellites (Chap. 14) become available.

Techniques to improve operational forecasts by assimilating real-time observations have been used in meteorology and oceanography since the 1970s and 1980s, respectively (Ghil and Malanotte-Rizzoli 1991). Applications to coupled physical–biological models of aquatic ecosystems are more recent and have been advanced primarily for marine systems (e.g. Dowd and Meyer 2003; Mattern et al. 2013) including by New Zealand investigators (e.g. Briggs et al. 2013). Despite active research on assimilation of ecological observations into models, very few operational data assimilation systems currently exist which forecast ecological components including water quality. Exceptions include: an early warning system for the aquaculture industry for harmful *Phaeocystis* blooms in Dutch coastal waters (Twigt et al. 2011); predictions of chlorophyll and nutrient levels in South Korea through the Four Rivers Restoration Project (Kim et al. 2014a, b); and the Aquatic Real Time Monitoring System (ARMS) for predicting water quality variables for reservoirs in New South Wales and Victoria, Australia (Mills et al. 2013).

3.6 Synthesising Ecological Model Applications: The Importance of Calibration

Many of the applications of lake models in New Zealand (Table 3.1) have used the Computational Aquatic Ecosystem Model (CAEDYM; developed originally at the Centre for Water Research, University of Western Australia, Hipsey et al. 2007). A feature of this model is the large number of parameters which can be calibrated or 'tuned' to varying degrees. This process can be quite daunting to those unfamiliar with CAEDYM, its underlying ecological process equations, and the interactive nature by which state variables respond to individual parameter adjustments. The large number of ecological parameters in CAEDYM is associated with numerous state variables, many of which relate directly to measurements made using profiles or grab samples from a lake (e.g. dissolved oxygen, total chlorophyll *a*, total nitrogen, nitrate, ammonium, total phosphorus, dissolved reactive phosphorus, suspended solids and light extinction), while others (e.g. different zooplankton taxa and chlorophyll *a* associated with different phytoplankton taxa) generally do not have measureable counterparts or are less frequently part of a routine monitoring programme. CAEDYM can be coupled to models which describe the distributions of temperature, salinity and density taking account of mixing processes. Most commonly, CAEDYM is coupled to the 1-D model DYRESM resolving vertical processses, or ELCOM, a 3-D hydrodynamic model (Hodges et al. 2000, also developed at the Centre for Water Research, University of Western Australia).

Over many years, knowledge has been acquired about the ranges of CAEDYM parameters based on relevant international literature, local lake studies within New Zealand, theoretical reasoning and repeated calibration to minimise errors between model output and observed data. In a number of cases, fluxes from cells or layers in the CAEDYM model (e.g., rates of mineralisation of organic nutrients, uptake by phytoplankton, etc.) have been used at daily or sub-daily time scales to inform the user of the processes driving changes in state variables (Özkundakci et al. 2010). The output from the model also includes limiting factors for phytoplankton growth (e.g. Özkundakci et al. 2011; Abell and Hamilton 2014). This output is extremely valuable for calibration, providing information to the user about the dominant fluxes (i.e., the kinetic parameters controlling rates). It has also been used to inform managers about the frequency of phytoplankton nutrient limitation and the primary limiting nutrient (e.g., N vs. P). This information is rarely used in other modelling studies where the calibration has almost solely focused on minimising error between model outputs of state variables and observations, as opposed to ensuring the validity of individual fluxes that affect the concentrations of a state variable.

The calibration of DYRESM-CAEDYM has generally involved a manual process, mostly because of difficulties with weighting and prioritising state variables using an autocalibration procedure. However, automated sensitivity analysis involving repeated model runs with parameters adjusted within fixed ranges has been used to assist with prioritising parameters for manual calibration. The sequence of calibration has generally involved minimising error between modelled and observed temperature, followed by

Table 3.2 Subset of parameters in CAEDYM showing how values may be set a priori from theoretical considerations, specific studies (e.g. for nutrients) and universal physiological values (e.g. for phytoplankton) as a preliminary step in a calibration procedure applied to a specific lake, Rotorua, Bay of Plenty, New Zealand

Parameter	Value	Units	Justification
Dissolved oxygen (DO)			
Temperature multiplier, sediment oxygen demand	1.07	–	Approximates to Q10 value; doubling of rate per 10 °C temperature increase
Half-saturation constant, sediment oxygen demand	0.4	g m^{-3}	Measured value: in situ benthic chamber experiments (Burger et al. 2007)
Sediment oxygen demand	2.8	g m^{-2} d^{-1}	Measured value: benthic chambers (Burger et al. 2007)
Nitrogen			
Denitrification rate coefficient	0.1	d^{-1}	Values estimated from adjacent Lake Rotoiti when bottom waters become anoxic (Priscu et al. 1986)
DO half-saturation constant for denitrification	0.2	g m^{-3}	Estimated from adjacent Lake Rotoiti observations
Temperature multiplier for denitrification	1.07	–	Approximates to a Q10 value of 2 (i.e., rate doubling per 10 °C)
Sediment release rate of ammonium	0.28	g m^{-2} d^{-1}	Measured value from benthic chambers (Burger et al. 2007)
Particulate nitrogen sedimentation rate	0.3	m d^{-1}	Measured value from in situ sediment trap deployments (Burger 2006)
Phytoplankton			
Cyanobacteria growth rate	0.7	d^{-1}	Approximates maximum growth rate of *Anabaena* sp., the dominant cyanobacteria genus in Rotorua which represents cyanobacteria state variable
Diatom growth rate	1.4	d^{-1}	Approximates maximum growth rate of the diatom *Aulacoseira* sp. used to represent the second ('other') phytoplankton state variable
Cyanobacteria respiration rate	0.07	d^{-1}	Represents 10% of maximum growth rate (Reynolds 1997)
Diatom respiration rate	0.14	d^{-1}	Represents 10% of maximum growth rate (Reynolds 1997)
Cyanobacteria temperature multiplier	1.09	–	Cyanobacteria growth rate increases rapidly with temperature (Reynolds 1997)
Diatom temperature multiplier	1.06	–	Diatom growth rate increases relatively slowly with temperature (Reynolds 1997)
Temperature where cyanobacteria growth rate begins to decrease	33	°C	Cyanobacteria are generally tolerant of high water temperature (Reynolds 1997)
Temperature where diatom growth rate begins to decrease	25	°C	Diatoms have diminished growth at moderate temperature

(continued)

Table 3.2 (continued)

Parameter	Value	Units	Justification
Cyanobacteria saturating light (I_k)	170	μmol m^{-2} s^{-1}	*Anabaena* sp. have tolerance to high--light environments
Diatom saturating light (I_s)	15	μmol m^{-2} s^{-1}	Diatoms such as *Aulacoseira* are found in low-light environments such as in the deep chlorophyll maximum (Hamilton et al. 2010)

dissolved oxygen, nutrients and phytoplankton (as well as zooplankton when this state variable is included in the model simulation). The logic of this sequence is that the physical model should reproduce the dominant mixing and transport processes as drivers for the advection and dispersion of the ecological state variables. The initial focus for calibration of the biological model is on dissolved oxygen, particularly the sediment oxygen uptake, and sediment nutrient fluxes (e.g., from dissolved nutrient releases or resuspension) as these are effectively non-conservative, reflecting the fact that CAEDYM does not explicitly simulate bottom sediment composition but instead prescribes release rates dependent on properties of the overlying water layer in the model. In some cases, sediment nutrient release rates have been adjusted empirically through the course of long-term (decadal) simulations to reflect progressive organic enrichment of the bottom sediments associated with eutrophication (Özkundakci et al. 2012).

Table 3.2 provides an abbreviated list of parameters in the CAEDYM model and gives a justification for selecting specific numerical values based on modelling of Lake Rotorua that has extended over several years (e.g. Burger et al. 2008; Abell and Hamilton 2014). Our experience is that an initial theoretical basis for selecting parameter values (i.e., based on literature, local studies and knowledge of the system) generally provides a 'jump start' for the calibration process, and that the calibration can then focus on refining model fit to observed fluxes and individual state variables.

3.7 Introduction to New Zealand Lake Model Applications

As mentioned in Sect. 3.2.3, we have focused attention on deterministic models. This focus reflects the fact that these models have been extensively applied to lakes throughout New Zealand (Table 3.1) and have often been used in a theoretical, heuristic or predictive contexts to assist with considerations of lake restoration. In many cases, model applications cannot be separated strictly into one of these categories. The models have also often been used to address multiple questions, some of which may relate to restoration actions and others to global forcing (e.g., climate change).

Several lake models applied in New Zealand systems fit the definition of theoretical models by virtue of generating predictions which are currently beyond a standard definition of validation. For example, Allan et al. (2016, see Sect. 3.8.3) extended output from an existing coupled hydrodynamic-ecological model (DYRESM-

CAEDYM) by incorporating empirical relationships to describe the effect of invasive koi carp (*Cyprinus carpio*) on water column suspended sediment and nutrient concentrations. The inclusion of these empirical relationships allowed hypotheses to be developed about impacts on lake water quality of the relative importance of nutrient loads from the catchment (i.e. external loads) and from benthivorous feeding activities by koi carp (i.e. internal loads). Actual testing of these predictions, however, is difficult, because altering catchment nutrient loads is a process acting on a time scale of decades. Furthermore, only moderate success has been achieved in efforts to remove koi carp from the lake. This type of information can be combined with expert knowledge to provide managers with guidance on where to focus future restoration efforts (Schallenberg et al. 2017). Another example of a theoretical model is the application of DYRESM-CAEDYM to predict the effects of climate change on lakes (e.g., Trolle et al. 2011). This application may be considered to be theoretical as the simulation output cannot be verified for a specific lake.

The majority of deterministic water quality models applied to New Zealand lakes have used a heuristic approach, i.e. models are calibrated using an observed set of lake water quality data and validated using observations not used in calibration. For example, Özkundakci et al. (2011) investigated why Lake Okaro remained highly eutrophic despite an intensive in-lake and catchment restoration programme. The model suggested that the restoration effort has been compromised because while it addressed external nutrient loads, water quality problems appeared to be driven mostly by internal loads of both nitrogen and phosphorus, which accumulated in the hypolimnion during periods of stratification. Efforts have been made to deal with the internal loads in this lake through geochemical engineering (see Chap. 7), but the modelling indicates they have not adequately mitigated internal and external loads to have resulted in re-oligotrophication.

A predictive modelling approach is illustrated in the Lake Rotorua example given in Sect. 3.8.4, where the authors (Abell et al. 2015a) used a 3-D hydrodynamic model (ELCOM, Hodges et al. 2000) to simulate mixing and transport processes in the lake. The model was validated with current meter and temperature measurements at a central lake station (Gibbs et al. 2016). Assuming the meteorological forcing in model hindcasts is sufficiently representative of meteorological conditions in the future, the model simulations can be considered to be predictive, providing insights into a future state of lake water quality under different wastewater discharge conditions.

3.8 New Zealand Examples

3.8.1 Modelling to Inform Potential for Regime Shifts and Macrophyte Collapse in Waituna Lagoon, Southland

Waituna Lagoon and its wetland complex are located at the southern end of the South Island of New Zealand. Due to its largely natural character and high diversity of endemic species, the lagoon has been designated as a wetland of international significance according to the criteria of the Ramsar Convention in 1976. It also holds important historical and cultural values for local Māori (Ngāi Tahu) and is highly valued for its abundance and diversity of traditional food sources. The lagoon is separated from the ocean by a beach barrier which historically breached naturally when the water level of the lagoon exceeded approximately 4 m above sea level (asl). The current management of water levels involves creating an artificial connection between the lake and the ocean when water levels exceed about 2 m asl. The opening may persist for periods of hours to several months before closure occurs naturally.

Conversion of relatively low-intensity sheep and beef farms to dairy since the late 1990s, as well as establishment of pasture in areas of peat and native scrub (mostly manuka, *Leptospermum scoparium*), has increased nutrient concentrations in tributary and groundwater inputs to Waituna Lagoon. The specific concern is that *Ruppia*, a macrophyte identified as a keystone species critical for sustaining biodiversity of higher trophic levels, may be threatened by the increasing nutrient loads. A possible outcome of increasing nutrient loads is a regime shift (Schallenberg and Sorrell 2009) which has been identified as having the potential for loss of *Ruppia* from the lagoon due to nutrient-stimulated increases in epiphytic algae and macroalgae. This increase in benthic producer biomass could shade and stress *Ruppia* populations and might be exacerbated by further increases in nutrient loads leading to increases in phytoplankton biomass and additional shading of *Ruppia* populations.

A study was commissioned by Environment Southland to use an ecological model to simulate the dynamics of Waituna Lagoon so that it could be used as part of a decision support system for the assessment of restoration options, alongside expert assessment and knowledge of relationships of vegetation dynamics and nutrient loading for other Intermittently Closed and Open Lakes and Lagoons (ICOLLs) from around the world (Schallenberg et al. 2017). The catchment and lagoon management scenarios designed as inputs to the model were intended to inform scientists, stakeholders and local government about whether the natural values of the Waituna Lagoon ecosystem could be maintained or enhanced through a limited number of available restoration options. Specifically, guidance was required on the most effective options to prevent the decline and potential collapse of *Ruppia* beds (Hamilton et al. 2012a). The questions relevant to development of specific model scenarios for Waituna Lagoon included:

- How do different strategies for the timing and duration of opening of the lagoon to the ocean affect the lagoon water quality and ecology?
- What level of reduction of catchment nutrient and sediment inputs to the lagoon is required to avoid a regime shift to a turbid state?
- How does climate (e.g. rainfall and temperature) modulate responses of the lagoon?

Addressing these questions required a modelling approach that covered a range of time scales (tidal, seasonal and interannual) and included interconnected physical, chemical and biological processes. Simulation of the biomass of both planktonic and benthic primary producers was also required. The DYRESM-CAEDYM model (Hipsey et al. 2007) was further developed to include *Ruppia* and macroalgae as state variables, including algorithms specific for self-shading and desiccation—both critical processes relevant to considerations of eutrophication and water level variations, respectively. DYRESM-CAEDYM is one-dimensional in its spatial representation, i.e. it represents the lagoon as a series of horizontally homogeneous layers that are stacked on top of each other. Simulations with DYRESM-CAEDYM extended over several years to capture interannual variations in inflows and outflows, as well as meteorological forcing. In order to better understand the spatial extent of saline intrusion and tidal forcing of marine water when the lagoon was open, a three-dimensional model, ELCOM, was used to complement the DYRESM-CAEDYM simulations and provide high-resolution temporal and spatial dynamics of the marine water intrusion and circulation within the lagoon.

These model applications to Waituna Lagoon highlighted complex interactions and processes that have a significant effect on the ecology of the lagoon. The model simulation results are consistent with previous research that suggests that increasing eutrophication in shallow coastal ecosystems may initially result in dominance of macroalgae over *Ruppia* (e.g., see status quo simulation of *Ruppia* in Fig. 3.3a), followed by phytoplankton dominance over macroalgae. They also indicated that

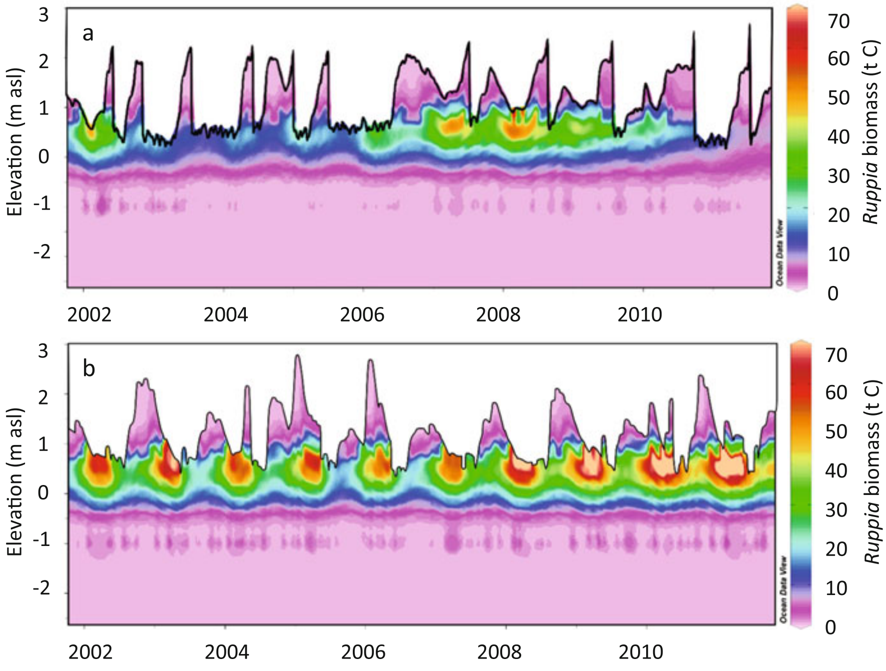

Fig. 3.3 Water levels (as elevation above mean sea level) and *Ruppia* biomass simulated with DYRESM-CAEDYM for the period October 2001–October 2011. Low biomass at low elevation relates to growth limits for *Ruppia* due to light limitation. Low biomass at high elevations relates to the upper growth limits due to desiccation stress. (**a**) A base scenario representing current conditions including actual barrier openings, and (**b**) a scenario with winter opening (3 months) and reductions of 50% in external nitrogen load and 25% of phosphorus load

adjusting the opening regime of the lagoon alone would not maintain an abundant and stable *Ruppia* population under the current nutrient loading regime. If the lagoon was not opened artificially at all, substantial (70–90%) nutrient load reductions would be required to maintain *Ruppia*. Alternatively, winter openings in combination with nutrient load reductions of 50% for total nitrogen and 25% for total phosphorus could be expected to maintain *Ruppia* (Fig. 3.3b), consistent with other research on nutrient load thresholds for macrophyte health in coastal ecosystems (Schallenberg et al. 2017). Time series of spatial distributions of salinity in the lagoon simulated by ELCOM (Fig. 3.4) may be able to be used to test different

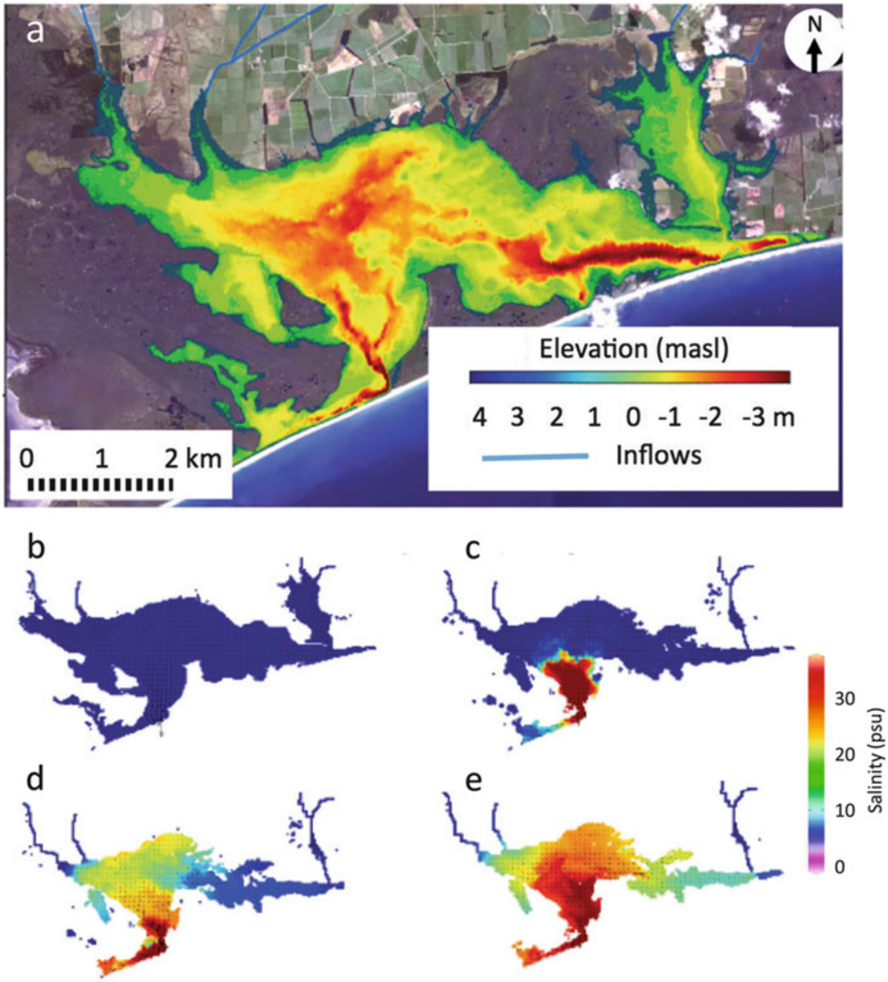

Fig. 3.4 Bathymetry of Waituna Lagoon (**a**) and distributions of salinity simulated with the ELCOM model and associated with a barrier opening event on 12 July 2007 for the day of opening (**b**) and 2 days (**c**), 4 days (**d**) and 16 days (**e**) following opening. Note: the lagoon area in the simulations varies with the balance of freshwater inputs and marine intrusion

scenarios of freshwater input and tidal forcing affecting salinity-sensitive organisms as well as informing the design of sampling programmes aligned with salinity distributions. The 3-D model is not going to be appropriate, however, for simulations over several years, particularly with a coupled ecological model, because of the lengthy computer run times required using standard computing resources.

3.8.2 Modelling to Support an Engineered Inflow Diversion for Restoration of Water Quality in Lake Rotoiti

Lake Rotorua (area = 80 km^2, mean depth = 10 m) and Lake Rotoiti (area = 34.4 km^2, mean depth = 33 m) are adjoining lakes that are part of a complex of volcanically formed lakes known as the Te Arawa or Rotorua Lakes, in the central North Island of New Zealand. Water flowed from Lake Rotorua into the western basin of Lake Rotoiti (Fig. 3.5a) through the Ohau Channel until its course was altered in 2008 following the completion of a diversion wall near the entrance of the channel entrance to Lake Rotoiti (Fig. 3.5b, c). The diversion wall restricts the flow of water to the north-west shoreline and towards the Kaituna River, the only surface outflow of Lake Rotoiti.

From the 1960s to the mid-2000s, there was marked decline in the water quality of Lake Rotoiti (Vincent et al. 1984; Von Westernhagen 2010). Eutrophication of Lake Rotorua through the same period of time was identified as a primary cause of the declining water quality of Lake Rotoiti, partly attributable to wastewater discharge to Lake Rotorua until 1991 (Rutherford et al. 1996) and latterly to land use change and intensification leading to increased nutrient loads to the lake (Smith et al. 2016). The Lake Rotorua inflow was estimated to contribute about 70% of the total nutrient load to Lake Rotoiti (Hamilton et al. 2005), and construction of a diversion wall was proposed as a means to isolate Lake Rotoiti from the discharge originating from Lake Rotorua, short circuiting the discharge towards the Kaituna River outflow (Gibbs et al. 2003).

A three-dimensional hydrodynamic model (Stephens 2004) and the 1-D DYRESM-CAEDYM model (Hamilton et al. 2005) aided the planning and environmental consenting process for the wall. The models assisted with testing the effectiveness of different design (primarily length) options for the diversion wall, with focus on whether there would be 'short circuiting' of the inflow around the wall that would compromise its effectiveness. Additionally, relatively complex hydrodynamic models (i.e. ELCOM and DYRESM) were required to capture the insertion dynamics of the inflow, which varied according to the temperature of the inflow relative to that of the vertical profile in Lake Rotoiti.

The CAEDYM model conceptually had a nutrients–phytoplankton–zooplankton (N-P-Z) trophic cascade including comprehensive budgets for carbon, dissolved oxygen, inorganic particulates and nutrients. The application of this model was important in assessing whether the loss of the deep insertions of oxygenated water

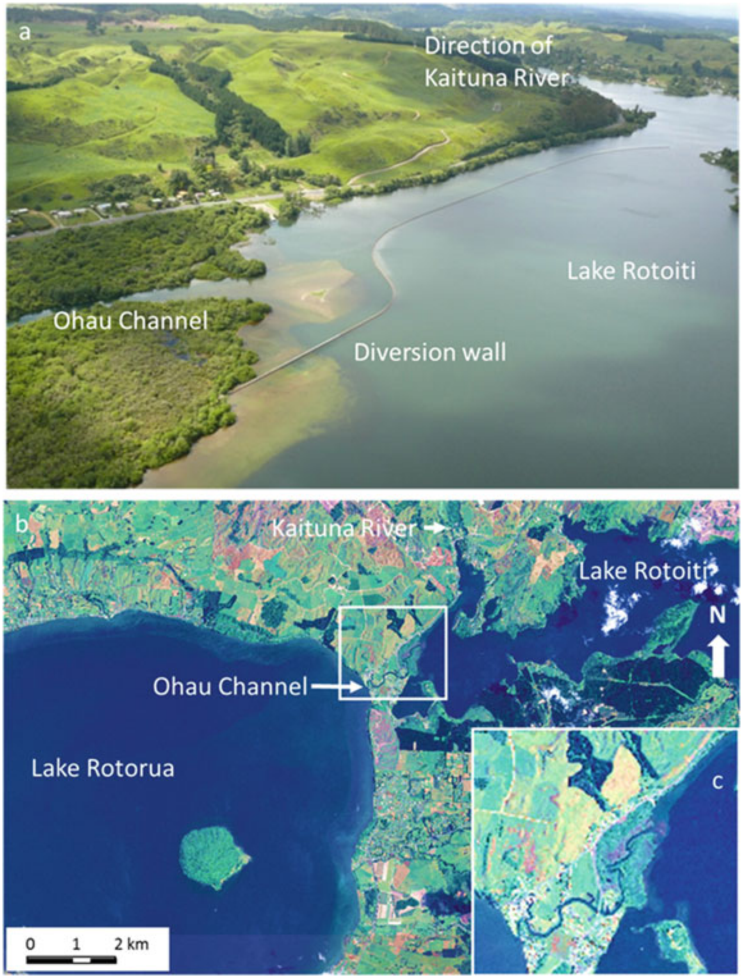

Fig. 3.5 Ohau Channel diversion wall: (**a**) aerial photo, (**b**) SPOT 6 satellite image on 17 January 2014 with the box area expanded in (**c**) to show detail of the Ohau Channel and the diversion wall separating relatively turbid water arising from Lake Rotorua (adjacent to north-west shore) and clear water of Lake Rotoiti

from the Ohau Channel, which occurred predominantly in winter, would extend the period of deoxygenation in the hypolimnion of Lake Rotoiti. Flow regimes of 50, 10, 5 and 0% of Rotorua inflow were examined by changing the inflow time series while leaving the meteorological input and remaining inflows unchanged and running the model again. The simulations indicated that there would be minimal impact on hypolimnetic deoxygenation, and there would be some net improvement in water quality, including an estimated ~40% reduction in cyanobacteria biomass 4 years following the inflow diversion (Fig. 3.6). The modelling studies formed an important

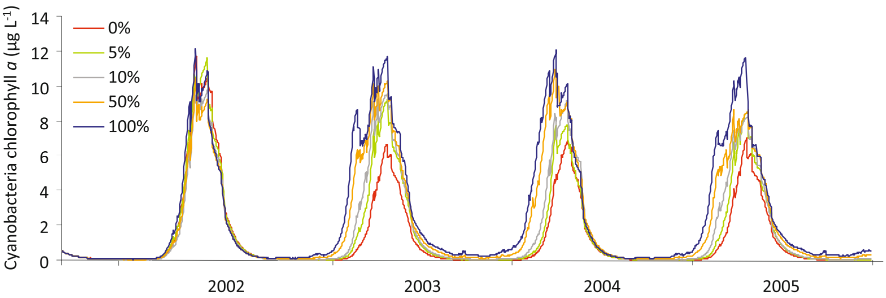

Fig. 3.6 Key model output of cyanobacteria biomass (expressed as chlorophyll *a*-equivalents) which underpinned a decision to construct the Ohau Channel diversion wall. The percentages represent different levels of diversion of the Ohau Channel away from Lake Rotoiti and towards the Kaituna River outflow. The simulation period was for 4 years and 3 months. The progressive reduction in cyanobacteria biomass partly reflects the adjustment in lake water residence time with the diversion; from approximately 1.5 to 5 years

part of the justification for this engineering solution and supported a decision to build the wall in 2008 at a cost of $NZ10 million. Reviews of the diversion wall have shown that water quality of Lake Rotoiti has indeed improved markedly following its completion, with the lake attaining its Trophic Level Index goal set in the Water and Land Plan of the Bay of Plenty Regional Council (McIntosh 2014), but the causes of the improvement remain equivocal. For example, there appeared to be some improvement in water quality of Lake Rotoiti even before the completion of the diversion wall 2008, and this improvement may have been related to alum dosing of Lake Rotorua, which commenced in 2006 and has resulted in a dramatic improvement in water quality and substantially reduced cyanobacteria biomass in this lake (Smith et al. 2016).

3.8.3 Modelling of Pest Fish and Catchment Effects on Lake Water Quality

Lake Ohinewai is a polymictic, riverine lake in the Waikato region of the North Island of New Zealand. It has a surface area of 16.2 ha and a maximum depth of 4.5 m. A marked decline of the water quality of the lake became clearly evident in the 1980s. There has been debate, however, about the factors leading to the decline; nutrient inputs from farming activities in the catchment have increased but koi carp (*Cyprinus carpio*) were first reported in significant numbers in the lower Waikato River in the 1980s (Collier and Grainger 2015), and this fish can access the lake via its outlet to the river. Koi carp are one of the ecologically most destructive invasive freshwater species (Weber and Brown 2015) and degrade shallow aquatic ecosystems mostly through their benthivorous feeding which resuspends bottom sediments and nutrients, as well as excreting nutrients into the water column. Lake Ohinewai had lost all its submerged vegetation by 1991 and today has hypertrophic status despite intensive carp removal and riparian planting programmes. A modelling approach was used to inform the planning of restoration activities, particularly the relative focus on catchment (e.g., agricultural best management practices to reduce external nutrient loads) vs. in-lake mitigation approaches (i.e., koi carp control to reduce internal loads).

To address these goals, a coupled catchment-lake modelling approach was developed. The process-based catchment model INCA (INtegrated CAtchment model, Whitehead et al. 1998; Wade et al. 2002) was used to provide stream discharges and nutrient loads to Lake Ohinewai which included simulation of physical and biogeochemical processes with DYRESM-CAEDYM. A benthivorous fish component has never been applied in CAEDYM, and empirical mathematical relationships were developed to describe their effects on sediment resuspension, nutrient release and translocation from sediment pore waters to the water column, as well as nutrient excretion, based on findings of previous studies (Breukelaar et al. 1994; Morgan and Hicks 2013; Allan 2016).

Nutrient load scenarios approximated different levels of catchment intervention, from conversion of the entire catchment to native bush to constructing wetlands near the inflows, as well as removal of koi carp biomass to different levels according to hypothetical control efforts. The model simulations indicate that removal of a high biomass of koi carp (from 374 kg ha^{-1} to 10 kg ha^{-1} wet weight) would reduce the percentage of the total nutrient load that is internal from around 46 to 33% for total phosphorus (Fig. 3.7) and from 47 to 42% for total nitrogen, primarily as a result of reducing excretory fluxes (Allan 2016). While this modelling study indicated that

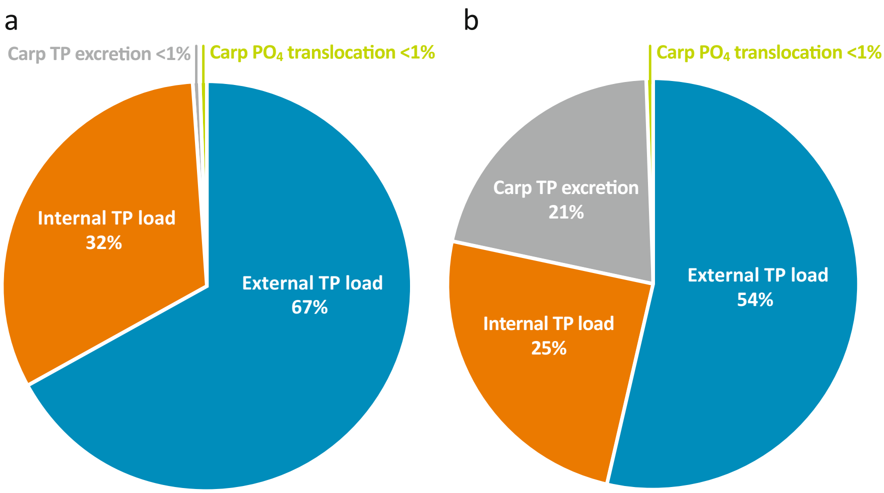

Fig. 3.7 Phosphorus loads for Lake Ohinewai differentiated by external, internal and koi carp TP excretion and PO_4-P translocation from the bottom sediments: (**a**) carp biomass of 10 kg ha^{-1} and (**b**) 274 kg ha^{-1} wet weight

koi carp introduction to Lake Ohinewai has contributed to its current hypertrophic state, it also shows that koi carp removal alone is not sufficient to effect substantial improvements in water quality. Additional model scenarios indicated that to return the system to a stable, clear-water state associated with macrophyte re-establishment, an integrated catchment and in-lake management programme is necessary, reducing inputs from both sources by at least 50%.

3.8.4 Wastewater Dispersion Modelling in Lake Rotorua

ELCOM-CAEDYM has been used to support a decision-making process on how treated municipal wastewater from the city of Rotorua (central North Island, New Zealand) should be discharged after 2019, when the operation of a land treatment system is scheduled to cease (Abell et al. 2015a). The proposed discharge locations include a stream flowing into Lake Rotorua, several sites along the lake's shoreline and an offshore lake-bed location. The model was used to predict the potential effect of mixing processes on dilution and dispersal of treated wastewater throughout the lake using scenarios of different discharge locations.

Wastewater introductions to the lake were represented using a conservative tracer to examine dispersion. Two separate periods with contrasting conditions were simulated: December–January (summer) 2013/2014 and June–July (winter) 2014. Each simulation was preceded by a 2-week 'spin up' period so that numerical lag of water velocities was avoided. The performance of the model was validated by comparing simulated water temperatures with high-frequency temperature measurements collected from a thermistor chain at a centrally located monitoring buoy in the lake. ELCOM simulations of Lake Rotorua had also been validated with mid-lake water column velocity profiles (Gibbs et al. 2016).

Twelve model scenarios were run, differing in the wastewater discharge location and wind forcing. While some simulations used hourly observed wind, the model was also run using constant wind forcing to generate more general predictions based on the predominant wind directions experienced at the lake. The simulations highlighted the potential for wind-driven basin-scale circulation processes to greatly influence how treated wastewater mixes throughout the lake, depending on prior wind conditions and the location of the outfall. Specifically, south-west winds were predicted to reduce dispersion of the wastewater, with higher concentrations of the tracer along the eastern shore for the scenarios of discharge to a nearby stream (Puarenga) or the lake shore site (Fig 3.8a). North-east winds were predicted to produce greater dispersion from transport west towards Rotorua city lakefront, with this effect less pronounced for the scenario of discharge along the shoreline to the northeast of Puarenga Stream, near the mouth of Sulphur Bay (Fig. 3.8b). Surface concentrations were still predicted to be low ($<1\%$ of tracer input concentrations) in these near-shore areas. Offshore discharge to the lake bed was predicted to result in the lowest tracer concentrations in near-shore areas due to increased advective transport and mixing of treated wastewater (Fig. 3.8c).

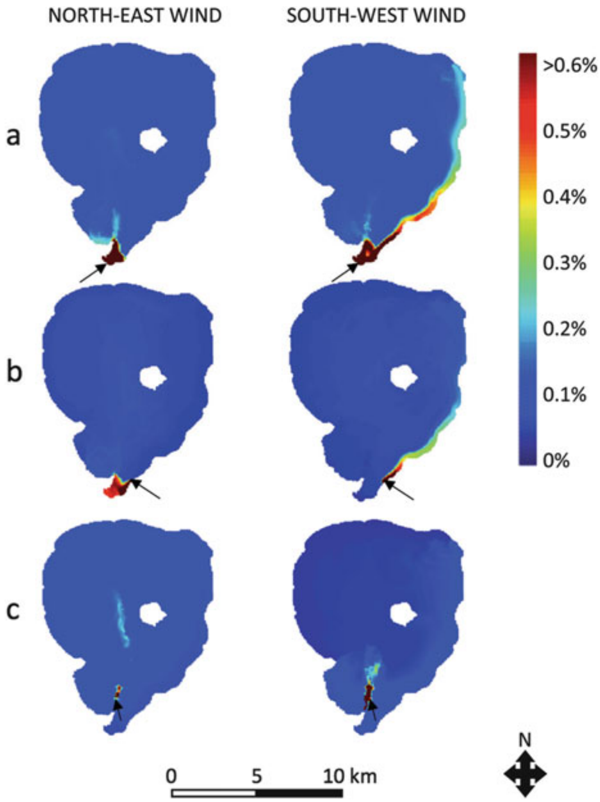

Fig. 3.8 Modelled concentration of a passive tracer 30 days after its release in idealised north-west and south-west wind forcing, respectively. Arrows mark simulated discharge sites: (**a**) to the Puarenga Stream, (**b**) to the south-east lake shore and (**c**) to the lake bed

This 3-D model application used only a short duration of simulation, excluded ecological processes and had a limited amount of model validation, but is nevertheless useful for examining the potential for localised or acute events in response to effects of wind-induced lake circulation patterns on wastewater concentrations. The work has been complemented by a 1-D model application which includes ecological considerations and allows for long-term (multi-year) assessments of changes in lake trophic state in response to different wastewater discharge options (27 different scenarios; Abell et al. 2015a). The combination of models with different levels of complexity has provided a comprehensive assessment of the discharge options and effects to support considerations of levels of wastewater treatment and discharge location and volume for a consenting process.

3.9 Future Prospects

Applications of deterministic numerical models are now well established to support environmental management at the catchment scale, with models used as both a scientific tool to understand why a change in water quality occurred (i.e. hindcasts) and as a decision support system to generate scenarios (i.e. forecasts) for a range of potential restoration strategies. Model applications are often extremely valuable because they provide a level of interrogation of measured data that goes beyond routine quality assurance and quality control procedures. Therefore, a model study often leads to improvements in the design of sampling programmes, measurement protocols and the choice of attributes measured. Modelling has the potential to be highly useful as water quality monitoring evolves to address the goals of the National Policy Statement for Freshwater Management (MfE 2014) to maintain or improve water quality of lakes and freshwater generally. Data assimilation techniques for autonomous in situ sensors and remote sensing will also improve the availability and quality of information to derive model boundary conditions and support more rigorous calibration and validation procedures (Luo et al. 2011).

Considering modelling as a workflow which includes stakeholder communication is important to ensure that what is asked of the models is within the capabilities of the model, to address model errors and to ensure that the model is not excessively complex or over-parameterised. For example, managers may wish to know why a particular species of phytoplankton bloomed or to have some measure of resilience to understand the susceptibility of a system to a regime shift. The former dictates a level of complexity and species-specific information which may not be readily available while the latter may not be easily quantified either from observations or from model output. Thus, it is critical to communicate a priori the balance between the complexity required to simulate specific acute events that are often key factors in generating ecological shifts and changes in community structure and simplifications necessary to derive statistically robust and validated comparisons of model output against measured data. Associated with this is a need to expand modelling of higher trophic levels, including zooplankton and fish, and possibly also to model lower trophic levels such as bacteria, including *E. coli*.

This chapter has provided a synopsis of lake ecosystem modelling, including a review of the many applications throughout New Zealand and details on selected case studies. Many of the applications have a recurrent theme related to eutrophication, and they are, therefore, also closely tied to restoration options including testing catchment nutrient load reductions, in-lake nutrient reductions arising from removal of pest fish or geochemical engineering, inflow diversions and water level variations. A synthesis of the substantial body of New Zealand studies available to date reinforces good modelling practice of careful planning and communication to achieve consensus with respect to expectations of model capability and limitations.

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Chapter 4
Agricultural Catchment Restoration

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Abstract A good understanding of the sources of contaminants and the processes that facilitate and modify their transport to and within freshwaters is the first step in the restoration of good water quality in agricultural catchments. This understanding needs to be combined with knowledge on how different agricultural systems and practices influence contaminant sources and processes, and this knowledge embedded within tools that enable robust decisions to be made to meet desired water quality objectives. Such information and tools will enable us to explore the consequences of mitigation decisions, trade-offs, and ultimately optimise where agriculture and desired water quality best coexist.

Keywords *Escherichia coli* · Good management practices · Groundwater · Mitigation · Nitrogen · Phosphorus · Runoff · Sediment

4.1 Introduction

The quality of water leaving agricultural catchments, as assessed by a variety of indicator analytes (nitrogen, N; phosphorus, P; suspended sediment, SS; *Escherichia coli*, *E. coli*), is often worse than in forested (native or exotic) catchments (e.g. Quinn and Stroud 2002). These analytes, hereafter called contaminants, indicate directly or indirectly the potential for a variety of detrimental effects in flowing streams, rivers, lakes, and reservoirs. Much research has quantified the factors involved in

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influencing the loss of contaminants from land to water. Unfortunately, due to a wide array of sources and transport pathways into flowing waters, management (viz. mitigation) of contaminant losses is often complex. Added to this complexity are the changes to contaminant concentrations that occur once in the water column in streams and variable in-lake legacy and recycling effects (see Feature Box 4.1). The ability to link effects in lakes and reservoirs to causes (or sources of contaminants) within a catchment is therefore often fraught with uncertainty. This uncertainty can lead to various combinations of ineffective policy, mistrust between landowners and policy-makers, and consequently little effective action to remediate poor water quality. On a positive note, while we recognise that our understanding of the connectivity between scales within a catchment may be lacking, research into specific catchment processes is more advanced. Perhaps best understood, and hence able to be quantified in response to management changes, are those processes that control contaminant losses from land into flowing waters. We, therefore, outline in this chapter some of these processes for N, P, SS and *E. coli* losses, but more importantly comment on our ability to manage the loss of each and demonstrate some case studies where management aimed at mitigating contaminant loss has altered catchment water quality outcomes.

Box 4.1 Catchment Processes Affecting Lakes: An International Perspective

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Many of our lakes receive high levels of pollutants from their catchments; these lead to poor water quality and potentially toxic algal blooms. Sources of these pollutants include agricultural runoff, and discharges from industrial sources and wastewater treatment works. However, this is not a recent problem; where long-term data exist, it is clear that many lakes, and the rivers that drain into them, have been adversely affected by pollution for many years (e.g. Howden et al. 2010). In addition to the effects of pollution, some lakes have also been affected by hydro-morphological modifications within their catchments, such as dam construction (Zhang et al. 2012) and increased abstraction to supply irrigation and drinking water (Jeppesen et al. 2015). Lakes can respond to changes in external pressures relatively quickly and, as such, they have often been flagged as sentinels of environmental change (Adrian et al. 2009; Schindler 2009; Williamson et al. 2014). As a result, it has often been assumed that lake water quality problems can be solved quickly if external pressures are reduced. However, whilst there is a realistic expectation that lake water quality will improve if pressures from the catchment are

(continued)

Box 4.1 (continued)
reduced (Jeppesen et al. 2005), expectations of the speed of recovery are often less realistic (Jeppesen et al. 2007). This is because pollutants that have accumulated in the system over long periods of time are released slowly from catchment soils and lake sediments (Sharpley et al. 2013). This legacy effect can prolong recovery times by many years (Jarvie et al. 2013; Fig. 4.1). Although in-lake measures, such as geo-engineering, aeration/mixing, and/or bio-manipulation, can be used to decrease lake recovery times in the short term, sustainable recovery can only be achieved if the main sources of external pressures are identified, quantified, and addressed. At the local scale, this includes reducing agricultural runoff and controlling pollutant discharges. However, at the broader scale, some pressures (e.g. climate change) are beyond the control of lake restoration projects, and lake managers may instead need to mitigate for their impacts (Rolighed et al. 2016). For example, Rolighed et al. (2016) predict that a 6 °C rise in temperature in Lake Søbygaard would result in an increase in total chlorophyll-*a* concentrations of about 65%. The authors conclude that, to mitigate for this effect and maintain chlorophyll-*a* values similar to those of the present day, the external nutrient loading would need to be reduced by 60%.

Fig. 4.1 Loch Leven, Scotland, UK—a rapidly warming shallow lake with an intensively farmed catchment and phosphorus legacy issues

4.2 Sources of Contaminants

The loss of contaminants from land to surface water is a function of the availability of contaminants, a transport pathway to get them from their source to streams and rivers, and intervening attenuation processes along the transport pathway (Fig. 4.2). Both availability and transport can be influenced by land use and land management, for instance, switching nutrient forms and concentrations between dissolved and particulate forms (Fig. 4.2).

A surplus of N increases the availability of N for loss—irrespective of the enterprise involved. Most N in soil is contained in soil organic matter that can be mineralised and released whenever the soil is tilled. Planning crop rotations to consider the likelihood for drainage and the timing of fertiliser inputs are important factors that help to minimise the size of the N pool available for loss (McDowell et al. 2008). However, for grazing systems, the inefficient conversion of dietary protein to milk, meat, and fibre by ruminants results in urine patches that can have equivalent N concentrations of up to 1000 kg N ha^{-1} (Haynes and Williams 1993). This is well in excess of the N requirement of most pastures leaving the rest available

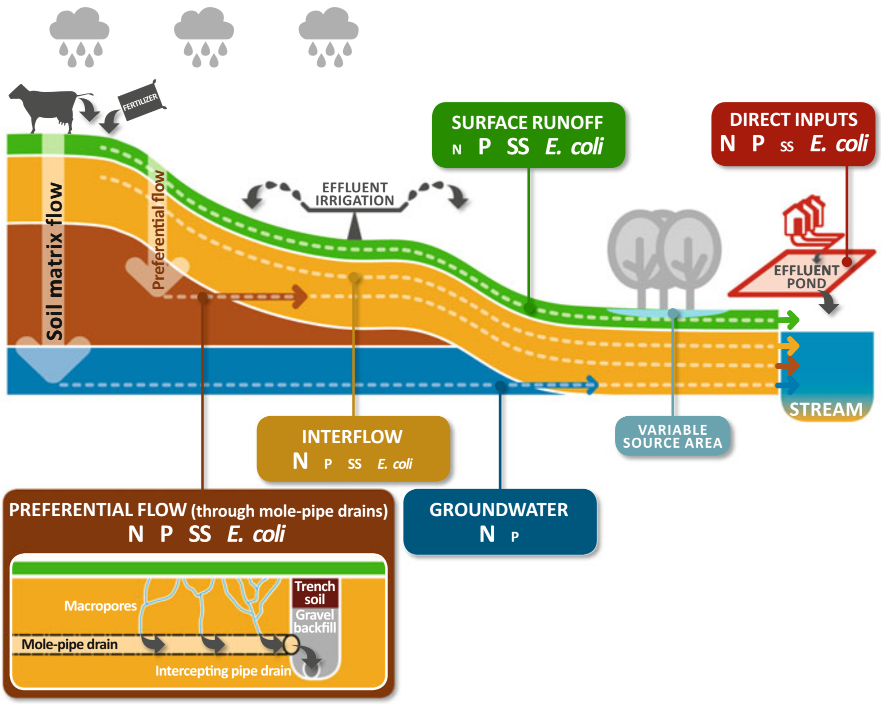

Fig. 4.2 Conceptual diagram of the transport pathways involved in the transfer of contaminants (nitrogen, N; phosphorus P; suspended sediment, SS; *E. coli*) from land to water. The presence and relative size of each of the contaminants indicates the importance of the pathway to contaminant-specific loss

for loss. A key management factor for decreasing the amounts of N potentially available for loss from grazing systems is, therefore, manipulation of the amounts and timing of urinary N excreted and deposited to pastures by the animal. This can be achieved by considering animal stocking rate, dietary composition, and the duration of pasture grazing (Monaghan et al. 2007a).

Another potential source of N loss (and other contaminants) on dairy farms is excreta either via livestock access to streams, or runoff of faecal deposits on land, or the application of farm dairy effluent (FDE). In the 1970s, a two-pond FDE system was introduced that combined an anaerobic pond with a facultative pond (Sukias et al. 2001). This decreased the biological oxygen demand of raw effluent before discharging "treated" effluent to streams, but was inefficient at removing N and P (Hickey et al. 1989). Such systems are now less common in most regions of New Zealand with application of effluent to land being preferred. Land application systems allow the soil to filter out much of the N, P, SS, and *E. coli* in effluent, but can be leaky if too much is applied, especially to wet soils, increasing the availability of contaminants to transport via preferential subsurface and surface flow pathways (Monaghan and Smith 2004; Houlbrooke et al. 2008).

Sources of P loss, in addition to those associated with poor FDE management practices, include: soil, fertiliser, plant residues, and animal dung (McDowell et al. 2007). As P surpluses increase, it is likely that soil P concentrations increase. This, especially in soils with lower anion storage capacity, increases the availability of P to surface runoff, sub-surface flow, and possibly into groundwater (McDowell et al. 2015). The erosion of particulate-associated P and loss via surface runoff can be enhanced by soil compaction and pugging following treading by grazing animals. Direct applications of P can also occur via animal dung (not urine) and fertiliser, although the availability of P in fertiliser is inversely proportional to the fertiliser's water solubility (McDowell et al. 2003a). Plant residues can be an important source of P in arable systems. This can also be the case for a short time immediately after grazing by ruminants, but is enhanced by dung deposition whose availability decreases rapidly as a crust forms on the dung, thus impairing interaction with rainfall (McDowell 2006).

Sediment losses in New Zealand vary widely due to rainfall and geology (Hicks et al. 2011). Large sediment loads are associated with areas of high rainfall and soft rocks. In contrast, areas with high rainfall, but hard, erosion-resistant rocks like Fiordland, deliver little sediment to streams (Hicks et al. 2004). Besides rainfall and geology, agricultural practices influence erosion by changing runoff characteristics (see Feature Box 4.2) and the likelihood of sediment loss from topsoil and stream-banks. For instance, Matthaei et al. (2006) found that stream beds surrounded by ungrazed tussock had significantly less fine sediment than those with pasture grazed by dairy cattle or deer. Much of this sediment in grazed catchments may originate from the treading of grazing animals on stream banks, which destabilises soils, causing slumping and loss of soil into stream channels (McDowell et al. 2003b). This effect is exacerbated by shallow-rooted pasture species that do not stabilise soil as well as the roots of larger trees and shrubs. In some catchments, subsurface drains are important conduits of sediment transport (Walling et al. 2002). Sediment inputs to streams in agricultural catchments without extensive drainage networks or grazing animals may be dominated by sheet and rill erosion from tilled systems (Wischmeier and Smith 1978).

Box 4.2 Assessing Land Cover Changes with High Spatio-temporal Resolution to Assist with Catchment Restoration

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The water quality of lakes and rivers reflects all that has happened within the catchment over both space and time. Of all the catchment features, land use/cover has the greatest impact on water quality, particularly agriculture due to mobilisation of sediment, nutrients and microbial contaminants, and removal of vegetation that would normally filter these same pollutants. The two broadest agricultural land uses in New Zealand are livestock grazing and plantation forestry, both of which can convert a densely vegetated area of land into bare soil at short time scales (see Fig. 4.3). Thus, to capture how the landscape is actually changing and to use effective restoration strategies to improve water quality, land cover maps are required that capture rapid changes at fine spatial resolution (e.g. 30 m). Unfortunately, the current national-scale land cover maps for New Zealand are semi-decadal. We are currently developing an 8-day, 30-m land cover time series (expressed as disturbed pixels that are mostly bare soil) for all New Zealand grasslands and forests from 2000 to 2015. We work with the MODIS Nadir BRDF-adjusted reflectance (NBAR) data at 8-day and 500-m resolution (MCD43A4) because it standardises the reflectance values to nadir view, which results in the minimisation of view angle artefacts. We transform these data into normalised brightness, greenness, and wetness values, which we then use to calculate a disturbance index where index values greater than three are considered disturbed (i.e. bare soil). In order to obtain 30-m spatial resolution, we fuse this time series with Landsat satellite imagery. This high resolution land cover time series can be used to detect the exact date (within a week) of a forest cutting or the relative intensity of livestock grazing (i.e. how often a paddock becomes bare soil). This is useful data when planning catchment restoration because it gives a catchment-scale perspective of land disturbance with the added information of timing and frequency of disturbances.

Dung and manure from grazing animals may contain pathogenic microbes such as *Giardia*, *Salmonella*, *Cryptosporidium*, and *Campylobacter*. The preferred indicator of faecal pollution is *E. coli*. Major sources of *E. coli* in agricultural catchments

Fig. 4.3 Example of strip grazing in the central Waikato region during winter 2010, where a vegetated surface is converted to bare soil in less than a week by dairy cow grazing. Winters in NZ are characterised by moderate sunlight, high rainfall, and lower pasture recovery rates. The combined effects can lead to high sediment and nutrient runoff and thus degraded water quality of receiving waters

include surface runoff from grazed pasture, deer wallows, poorly designed animal feeding areas and point sources such as purpose-built waste treatment systems such as oxidation ponds for FDE (McDowell et al. 2008). Other diffuse sources include runoff from farm-stock tracks, livestock accessing unfenced streams, and stock crossings of streams (Collins et al. 2007). Farm-scale modelling shows that mitigating inputs such as direct stream access by grazing animals and inputs from FDE would substantively decrease *E. coli* concentrations in a dairy-farmed catchment (Muirhead et al. 2011). Losses of *E. coli* are dominated by stormflow events (e.g. Davies-Colley et al. 2007). However, concentrations during baseflow, presumably sourced from point sources or in-stream stores, are important when considering health risk from pathogens to downstream water users, including bathers and other recreational users, and drinking water for livestock. Storm-flow loads are more important to downstream water uses such as shellfish harvesting and aquaculture (e.g. Davies-Colley et al. 2007).

4.3 Pathways of Contaminant Loss

The yield (e.g. kg ha^{-1} year^{-1}) and form (dissolved or particulate) of contaminants reaching freshwaters is driven by the activation of transport pathways such as runoff through and over topsoil and leaching to groundwater and the attenuation that occurs during transport. The confined pore size of soils, the vadose zone and aquifers imparts a much greater filtration effect in transport through groundwater to surface water than likely in direct transport via runoff. This restricts contaminant transfer via groundwater to those contaminants that exhibit a dissolved phase. The exception to this general rule is the fraction of runoff that is transported as subsurface flow (viz. interflow) and intercepted by artificial drainage. Here, macropores can provide a rapid conduit between surface sources of contaminants and transport them via subsurface drains to open drains or directly into streams. In pastoral systems where surface runoff has been identified as a significant pathway of contaminant loss, evidence exists to show that losses may simply be transferred in subsurface flow when artificial drainage is installed, although often exported in a dissolved rather than particulate form (Monaghan et al. 2016). Over time, artificial drainage networks may also serve as a source of contaminants where, for instance, the banks of surface drains collapse and erode or mole channels linked to pipe drains collapse.

Filtration of contaminants in the vadose (unsaturated) zone varies with the degree of saturation. There is much greater transport of particulate contaminants (SS and microbes) and sediment-associated contaminants (P) in saturated flow conditions through the unsaturated (vadose) zone. In unsaturated flow, transport occurs through the finer pores where there is greatest filtration. However, in saturated flow conditions, all pores can transmit water and transport is much faster (orders of magnitude) with less filtration. Saturated flow can occur with flood irrigation, excessive spray irrigation, possibly combined with rainfall, high intensity rainfall events, and redistribution of rainfall due to micro-topography resulting in transient saturated flow conditions. Studies of microbial transport under flood (Close et al. 2008) and spray irrigation (Close et al. 2010) have demonstrated that significant transport of microbes can occur with flood irrigation but minimal microbial transport takes place with properly managed spray irrigation.

Aquifer properties and biogeochemistry can also affect the transport and transformations of dissolved contaminants in groundwater. For instance, McDowell et al. (2015) found that groundwater P concentrations were enriched where there was a land use regularly supplying P, a soil type of low P sorption capacity and sufficient drainage (rainfall or irrigation induced) to transport P to groundwater. Furthermore, in aquifers of low P sorption capacity (e.g. sand and gravel), good connectivity to surface waters meant that in some cases P concentrations were also enriched in baseflow. This could result in a legacy of increasing P inputs to surface water via groundwater even if all surface runoff sources of P are mitigated. In contrast to P, nitrate is negatively charged and not sorbed or retarded in most subsurface systems. The main loss process for nitrate is microbially mediated denitrification, which occurs once oxygen levels are low or absent (anoxic, reducing). The rate of N removal depends on the amount of nitrate-containing water that flows through reducing zones. This can also vary seasonally with rises and falls in groundwater levels. Woodward et al. (2013) demonstrated how nitrate concentrations varied according to flow pathways in the Toenepi catchment, Waikato (Fig. 4.4). Summer flows were sustained by deeper, slower flowing groundwater from a reducing environment that is essentially nitrate-free. During winter, shallow fast flowing groundwater, flowing through a partially reduced zone, dominated stream flow and was responsible for 91% of the annual nitrate yield (Fig. 4.4). Nitrate concentrations in this shallow fast groundwater were significantly lower than expected. The amount

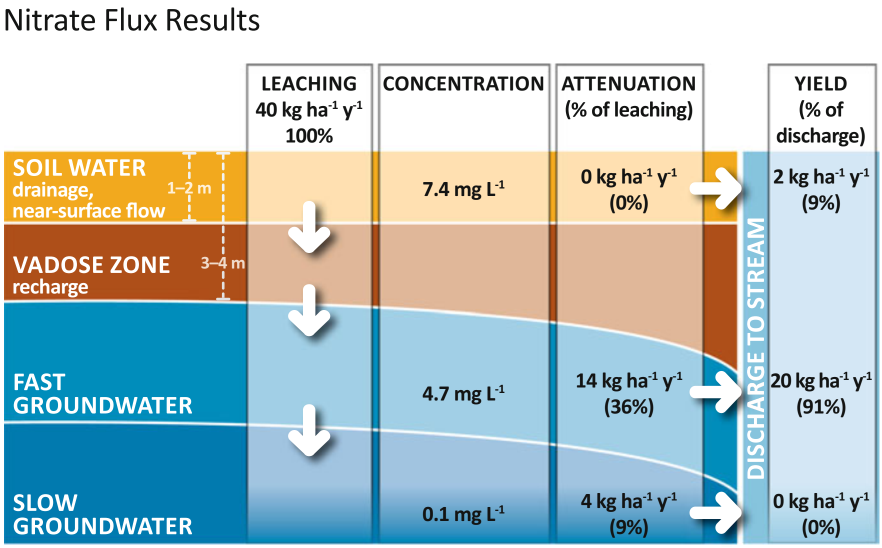

Fig. 4.4 Schematic of N fluxes in the Toenepi catchment, based on Woodward et al. (2013). Figure courtesy of Roland Stenger, Lincoln Agritech Ltd

of denitrification calculated (36%) was greatest in this zone because of the high N flux combined with the partially reduced environment.

For surface runoff, contaminant losses can occur via infiltration-excess and saturation-excess mechanisms. Under infiltration-excess conditions, the infiltration capacity of the soil is exceeded resulting in runoff. In New Zealand, this usually occurs under high-intensity rainfall or hydrophobic soil conditions at any time of year (Bretherton et al. 2011), whereas saturation-excess surface runoff only occurs when soils are saturated (largely in winter and spring) resulting in any excess rainfall running off. Due to topography, the areas affected by saturation-excess surface runoff are generally located near the stream channel and expand and contract in response to rainfall events and evapotranspiration. Due to the energy of high-intensity rainfall events, infiltration-excess surface runoff can contain more particles than saturation-excess surface runoff (Buda et al. 2009). However, most runoff and contaminant loss occurs in winter and spring. This does not necessarily translate to impact as small events occurring in summer and autumn can have greater impact. For instance, McDowell and Srinivasan (2009) found P losses in winter and spring accounted for 75% of losses, but of the remaining storms more P was in a dissolved form and, therefore, available to periphyton at a time when the seasonal risk of nuisance growths was highest (low flows and elevated temperatures).

Attenuation refers to the loss or temporary storage of contaminants as they are transported from the site of generation to where they impact water quality (i.e. a stream, lake or estuary). Generic attenuation processes include flow buffering, filtering and deposition, adsorption and precipitation, microbial immobilisation and transformations (e.g. denitrification), vegetation assimilation and other physical and biogeochemical processes. These natural processes can decrease contaminant concentrations and loads, and modify availability (e.g. particulate organic cf. dissolved inorganic), but should not to be confused with catchment restoration achieved via human-implemented mitigations. Natural landscape features that can promote attenuation include wetlands and vegetated riparian zones and channels.

Agricultural practices such as unrestricted grazing and trampling of riparian zones or the drainage of wetlands and channelisation of natural waterways often decrease natural attenuation functions. However, in some cases attenuation can be enhanced by rehabilitating degraded systems or constructing surrogate artificial systems, such as denitrification walls (Schipper et al. 2010) or constructed wetlands (Tanner and Sukias 2011). The installation and maintenance of riparian zones can be highly effective at attenuating sediments and attached contaminants from surface run-off, but may also facilitate contaminant removal by plant uptake or denitrification (Sweeney and Newbold 2014; McKergow et al. 2016). Studies of riparian buffers reviewed by Parkyn (2004) show sediment and TP removal rates of 53–98% that tend to increase with buffer width. However, removal of dissolved contaminants from surface and subsurface flows can be poor, varying with the hydrogeological characteristics of the site, the upslope gradient, buffer width, and vegetation type (Collier et al. 1995). In contrast, seepage wetlands in the headwaters and on the edges of streams in New Zealand have shown nitrate removal, due to denitrification, often exceeds 75% under base-flow conditions (Cooper 1990; Burns and Nguyen 2002).

4.4 Mitigation Strategies

There are many options and strategies available to mitigate the loss of contaminants from land to surface waters. Many are contaminant-, source-, or pathway-specific, while others can simultaneously tackle multiple contaminants, contaminant sources, or transport pathways. Mitigations can be applied at different scales from field to farm to catchment-scale. For a full explanation of the range of options available, the reader is directed to review articles such as McDowell et al. (2013) or Monaghan et al. (2007a). Many of the farm-scale mitigations involve increased efficiency of resource use (e.g. tailoring nutrient applications to optimal levels) and can decrease farm costs, thereby improving farm profitability. If these options are insufficient to achieve desired targets, then further improvements often have economic implications for the profitability of farming enterprises and/or require co-ordination and action beyond individual farms.

Because of the practical difficulties of directly measuring the contaminant loss responses of different mitigation scenarios across different landscapes and time-scales, models that can simulate farm, catchment and regional responses are crucial for predicting cumulative impacts to guide land management and regulation. These models may operate at different spatial (farm to catchment to region) and temporal (event to seasonal to annual time steps) scales. When choosing the appropriate model to guide and regulate actions, it is important to understand uncertainties and therefore limitations of each model (Anastasiadis et al. 2013). This is especially relevant when multiple models are linked together and problems around disparate data sources and model compatibility are also encountered.

The majority of contaminant modelling assessments in New Zealand have used a combination of farm and catchment-scale models that estimate annual loads. Focusing on an annual time step minimises variability and is often easy to allocate, but may require long time frames to detect meaningful change (Snelder et al. 2014) and miss large losses due to short-term events. Farm-scale models include, but are not limited to, APSIM, Overseer®, and SPASMO, while catchment-scale models include CLUES, MODFLOW, ROTAN, and SedNet (Anastasiadis et al. 2013). To these are linked either commercially available (e.g. Farmax Pro; www.farmax.co.nz) or bespoke economic models (Daigneault et al. 2013). However, while most farm-scale models tend to be non-spatial, some are doing so with the aim of optimising land use (according to profitability, e.g. MyLand, West and Jovner 2012) or balancing contaminant losses with the cost-effectiveness of mitigation strategies at the farm- (e.g. MitAgator™, McDowell et al. 2014a), catchment- (e.g. Daigneault et al. 2013; Elliott et al. 2014), regional- (Kaye-Blake et al. 2013), or national scales (e.g. Snelder et al. 2013).

The applicability and cost-effectiveness of different mitigation options may vary substantially for different landscapes and farm systems. They may also fit differently with the farming styles, knowledge-base, financial status, and preferences of specific farmers. Some options, such as the revegetation of erodible land or restoration/construction of wetlands, may have ancillary benefits (e.g. biodiversity; carbon sequestration; aesthetic, cultural, or recreation values) that provide value to the wider community. In these cases, cost sharing between farmers, other interested parties, and the community may be appropriate to achieve multiple goals.

Management at farm scale can include both strategic approaches that move an enterprise towards a goal of lower contaminant loss (strategic) and tactical approaches that make the right steps in the short term to implement the goal. One example combining both approaches would see the use of lower rates of fertiliser inputs to a dairy farm to lower soil Olsen P concentrations and targeting time of application to periods when surface runoff was unlikely—thereby also decreasing the likelihood of direct fertiliser-P losses (McDowell et al. 2003a).

Strategic decisions (e.g. investment) on-farm will consider catchment objectives such as decreasing a load of N or P at the outlet of the catchment. After establishing the need to mitigate contaminant losses, regulatory or voluntary approaches (or a mix of both) can be considered for implementing mitigations. In many countries, a regulatory approach is preferred by agencies (e.g. England, DEFRA 2014). However, this relies on regulation being applicable across all possible soil and climatic permutations, which is seldom the case. This approach can also stymy innovation on new ways and systems to decrease contaminant losses. However, a strictly voluntary approach is also seldom effective. While a proportion of land owners and producers will lead by taking up or adopting new technologies and practices to mitigate contaminant losses, others may continue with the status quo which is often to

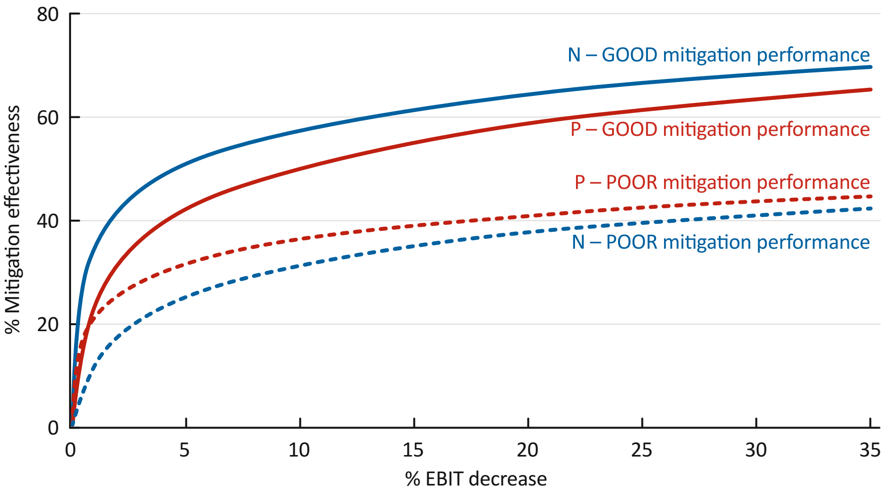

Fig. 4.5 Cost curve showing the percentage change in earnings before interest and tax (EBIT) against the percentage N and P mitigation effectiveness, for example Canterbury dairy farm—provided mitigations are applied on the basis of cost-effectiveness. Curves are generated to account for good and poor mitigation performance associated with spatial and temporal variation. Data calculated from McDowell et al. (2013)

intensify production at a rate often greater than the mitigation of contaminant losses. The key may be to adopt a mixed approach whereby regulation (backed by science to prove the spatial extent of the problem) prohibits practices that cause substantial contaminant losses. Limits can then be set that allow flexibility (outside of prohibited practices in specific areas) to do what the landowner or producer wishes provided they cause no additional losses.

A major problem with a mixed approach is that it requires a good understanding of where and when contaminant losses are occurring and then what to do about mitigating them. Recent research has made substantial steps in defining the areas that account for the majority of contaminant losses but such losses are produced from only a minor portion of a field, farm, or catchment area. These areas of loss, termed critical source areas, can be targeted with mitigation strategies that result in far less cost to the landowner or producer than strategies that are imposed across all areas (McDowell et al. 2014b).

The cost of mitigating contaminant losses can be minimised where the cost-effectiveness of potential strategies is fully assessed and the opportunity given to the producer to mix and match those best suited to their farm (McDowell 2014). This often leads to cost curves that are farm-, catchment-, or regionally specific (e.g. Fig. 4.5). If taking ease of implementation into account, another adaptation is to bundle mitigations by ease-of-use from those that require little investment and can be quickly implemented to those that require system changes and much longer planning before implementation. Bundling also helps to negate variation caused by differences in, for example, soil types and climates associated where the mitigation practice was developed and where it is applied (Table 4.1).

4.4.1 Implementing Mitigations to Decrease Catchment-Scale Losses

Case studies can be used to determine the likely uptake and efficacy of farm-scale mitigations in achieving catchment-scale objectives. In response to public concern over greater land use pressure and subsequent trends in water quality, a nation-wide dairy catchments project was commenced in New Zealand in 2001 with the aim of integrating environmentally sustainable dairy farming practices with the industry's policy of increasing farm productivity. Four catchments were originally selected, two located in traditional dairy farming areas in the North Island (Toenepi in Waikato and Waiokura in Taranaki; see Fig. 4.6) and two in areas in the South Island that had recently undergone conversion to dairy farming (Waikakahi in Canterbury and Bog Burn in Southland). Monitoring of a fifth catchment (Inchbonnie) located on the West Coast commenced in 2004 (denoted as Piegon Creek in Fig. 4.6). These catchments contained characteristics that were considered widespread in dairy-farmed areas of New Zealand. Detailed descriptions of the selected catchments and farming systems within each have been documented elsewhere (Wilcock et al. 2006, 2007, 2009, 2013a; Monaghan et al. 2007b, 2009). A catchment management planning process was followed that strategically directed science activities and extension aimed at voluntary farmer adoption of key land management practices that would improve catchment-specific water quality objectives. A key feature of the study was that it was a collaboration involving the dairy industry, farmers in each of the catchments (and later, others from further afield), local regulatory agencies (notably the scientific and extension staff dealing with water resources), and scientific and technical advisory/extension agencies.

Early in the study it was recognised that the development of farm plans would be helpful tools for farmers tackling environmental initiatives on their properties, such as upgrading effluent management systems or fencing and planting riparian strips. Farm plans are helpful for defining some of the most obvious issues to be tackled first and for developing a schedule of when important tasks should be completed.

The farm plans adopted in the case study varied according to the expertise available and the initiatives that were already underway in each catchment. In the case of the Waiokura catchment, a comprehensive riparian fencing and planting planning initiative had been supported by the local regional council and farm planning efforts accordingly incorporated this. In the case of the Toenepi catchment, comprehensive farm plans dealing with effluent, silage, riparian, and nutrient management issues were developed with the support of local industry, regulatory agency extension specialists, and research providers. For the Waikakahi catchment, additional attention was paid to improved practices for water irrigation, such as greater

Table 4.1 Bundles of mitigation strategies based on ease-of-use and cost-effectiveness for dairy and lowland sheep farms targeted to specific or multiple contaminants

Bundle	Measure	N	P
Dairy			
Easy	Stock exclusion from streams[a]	Y	Y
	Infrastructure for better FDE practice[a]	Y	Y
	Laneway runoff diverted[a]	Y	Y
	Decreasing Olsen P to agronomic optima[a]		Y
	Using low water solubility P fertilisers[a]		Y
	More efficient irrigation[a]	Y	
Cost (effectiveness)	Minimum	5 (0.1)	8 (12)
	Median	7 (1)	10 (21)
	Maximum	12 (2)	15 (75)
Medium	Less fertiliser N applied[b]	Y	
	Installing wetlands and/or sediment traps[a]	Y	Y
	Autumn substitution of N-fertilised pasture[b]	Y	
	Using winter-active pasture species	Y	
	Split grass-clover pastures[c]		Y
	Tile drain amendments to sorb P[a]		Y
Cost (effectiveness)	Minimum	95 (15)	30 (39)
	Median	230 (25)	70 (54)
	Maximum	450 (35)	125 (85)
Hard	Off-paddock wintering[a,b]	Y	Y
	Restricted grazing of pastures[a]	Y	Y
	Restricted grazing of cropland[a,b]	Y	Y
	Alum application to pasture[a]		Y
	Alum application to crop[a]		Y
Cost (effectiveness)	Minimum	395 (46)	330 (65)
	Median	750 (58)	640 (76)
	Maximum	1195 (70)	970 (94)
Sheep (lowland)			
Easy	Stock exclusion from streams[a]	Y	Y
	Using low water solubility P fertilisers[a]		Y
Cost (effectiveness)	Minimum	0 (0)	5 (5)
	Median	5 (0.1)	10 (9)
	Maximum	11 (0.2)	17 (42)
Hard	Installing wetlands and/or sediment traps[a]	Y	Y
	Using winter-active pasture species	Y	
	Tile drain amendments to sorb P[a]		Y
	Split grass-clover pastures[c]		Y
Cost (effectiveness)	Minimum	12 (4)	25 (36)
	Median	25 (8)	70 (52)
	Maximum	90 (19)	140 (65)

Annualised costs are given in \$ ha^{-1} year^{-1}, and effectiveness (in parentheses) is the percentage expected decrease in anthropogenic contaminant loss

[a]Outlined in McDowell and Nash (2012)

[b]Outlined in Monaghan et al. (2007a, b)

[c]Outlined in McDowell et al. (2013)

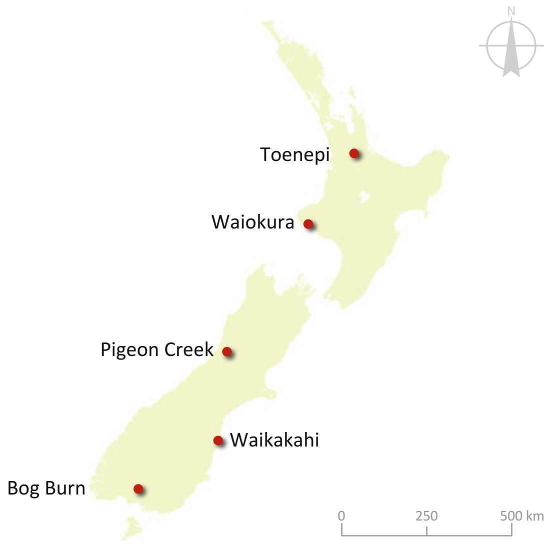

Fig. 4.6 Location of the dairy catchments studied in New Zealand

precision of application depths, bunding of border ends to prevent excessive flow of border dyke outwash to streams, and maintenance of irrigation races. For the Bog Burn catchment, improved management of FDE application to land was a particular focus, for example, deferred irrigation and/or low rate application methods (Houlbrooke et al. 2008).

Water quality outcomes, listed in Table 4.2, show SS concentrations (and loads) had significant ($P < 0.05$) decreasing trends in all five streams, while *E. coli* concentrations exhibited decreasing trends in two streams. Similarly, four streams show improving visual clarity. These improvements in physical and microbial water quality were attributed mainly to riparian protection. On average, stream fencing improved from less than 50% in 2001 to over 80% in 2011, with the result that less livestock-induced erosion occurred. The nutrient outcomes are more mixed (Table 4.2). Some measures of success with the farm planning were reflected in the nutrient budgets developed for most of the catchment farms. These showed how maintenance fertiliser P inputs were decreased over time to allow the relatively high soil Olsen P concentrations measured on many of the farms to decrease to more economically optimal levels and thus minimise the risk of increased enrichment of P in water flow leaving farmland (Wilcock et al. 2013b). There was also a marked decrease in the number and proportion of farms discharging treated FDE to streams in the Toenepi and Waiokura catchments (Wilcock et al. 2006, 2009), although analysis of trends in measured water quality contaminants over the monitoring periods were mixed (Table 4.2).

Farm surveys completed at the beginning and near the end of the study showed that land-use intensity and milk production increased in all catchments. The most obvious indicator of this was the increased milk production recorded per hectare, which increased by between 7% (Inchbonnie) and 31% (Waikakahi) over the 7-year

Table 4.2 Summary of catchment water quality trends and estimates of changes in average milk production from Wilcock et al. (2013a)

Catchment	TN	$NO_X - N$	FRP	TP	*E. coli*	SS	Black disc	Milk production[a]
Toenepi	NSC	NSC	NSC	NSC	NSC	↓	↑	↑
Waiokura	↓	↓	NSC	NSC	↓	↓	↑	↑
Waikakahi	↑	↑	NSC	NSC	↓	↓	↑	↑
Bog Burn	↑	↑	NSC	NSC	↑	↓	NSC	↑
Inchbonnie	↓	↓	↓	↓	NSC	↓	↑	↑

TN total nitrogen, *NO-N* oxidised nitrogen, *FRP* filterable reactive phosphorus, *TP* total phosphorus, *SS* total suspended solids

Arrows indicate significant ($P < 0.05$) increasing or decreasing trends (as per arrow direction). NSC is either no significant trend or a zero trend slope. Where appropriate, data were flow-adjusted to remove trends caused by changing flow regimes

[a]Estimates of percentage increases in milk production on farms within each of the catchments was based on farm survey information

(Inchbonnie) or 10-year (other catchments) monitoring periods. When combined with changes in the area in dairy farming within the catchments, we estimate that milk yields increased by 24, 26, 57, 23, and 18% for the Toenepi, Waiokura, Waikakahi, Bog Burn, and Inchbonnie catchments, respectively (Wilcock et al. 2013b). This increased milk production per catchment was also supported by significant increases in N fertiliser inputs and purchased supplementary feed in most catchments. The trends noted for water quality contaminants (Table 4.2) are, therefore, mostly favourable given increasing farm intensification. Furthermore, this suggests that farm productivity can be uncoupled from farm contaminant losses, especially for SS, P, clarity, and *E. coli*, where mitigation efforts can be more spatially targeted, for example, to effluent-treated paddocks, and near-stream areas, including riparian margins. However, the goal of decreasing N losses from dairy catchments against a backdrop of on-going farm intensification remains challenging. As noted earlier, research has shown that animal urine patches are usually the major source of N losses to water from grazed pasture systems. The trends of increasing N concentration in the Waikakahi and Bog Burn catchments (Table 4.2) are, therefore, not unexpected given the observed increases in cow numbers and milk production in these catchments.

The dairy catchment project highlighted a number of lessons pertinent to the success of water quality initiatives in the catchments. Clearly defined catchment values and environmental goals were important prerequisites that shaped the policy, research, and extension activities undertaken. Field days followed by one-on-one interactions via farm planning initiatives showed some success in improved farm environmental performance, as evidenced by changes in farm management practices and improvements in water quality for some catchments. However, the adoption of other more costly or complex management practices occurred at a much slower pace and has failed to halt increasing trends in nitrate-N concentrations in catchments that had seen recent conversions from sheep to dairy farming.

4.5 Conclusions and Future Prospects

The restoration of water quality within agricultural catchments requires a good understanding of the sources of contaminants and the processes that lead to their transport and modification en-route to freshwaters. Layered on top of these processes is a complex range of agricultural systems and practices. Capturing processes and practices in decision support tools will enable us to make better choices to mitigate farm-scale contaminant losses at least cost. Furthermore, these tools can be used to explore the trade-offs between competing land uses, help inform investment decisions, and link farm-scale decisions to catchment-scale water quality objectives.

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Chapter 5
Nutrient Budgets in Lakes

Piet Verburg, Marc Schallenberg, Sandy Elliott, and Chris G. McBride

Abstract This chapter discusses the sources and ultimate fate of nutrients into downstream lakes. It includes processes such as microbial denitrification, internal nutrient loading derived both from in situ measurements and from a mass-balance approach, and the loss of nutrients by flushing. Net and gross internal loads can be estimated using mass-balance equations. While the gross load is of interest because it contributes to the total load that drives algal growth, usually a large part of the gross load ends up sequestered in the sediment. Lake restoration efforts to reduce nutrients available to phytoplankton can focus on reducing both the inputs of nutrients and the cycling of nutrients within the lake. Various in-lake restoration techniques exist to target different fluxes within lake nutrient cycles. Therefore, understanding nutrient dynamics in the lake and sources and sinks of nutrients in lakes helps identify restoration approaches that are most likely to successfully reduce nutrient availability, phytoplankton blooms, and other related problems.

Keywords Internal loads · External loads · Land use · Nutrient burial · Denitrification · Residence time · Flushing · Macrophyte harvest

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5.1 Sources and Fates of Nutrients in Lakes

5.1.1 The Nutrient Mass Balance

Lake nutrient budgets account for sources and sinks of nutrients, as well as nutrient cycling and sequestration. Nutrient budgets quantify external loading, outflow rates, and lake nutrient concentrations. The main nutrients that sustain algal biomass (Schindler et al. 2008), nitrogen (N), and phosphorus (P) enter lakes primarily from their catchment and from the air by atmospheric deposition. In some lakes, nutrients may enter from geothermal sources. In addition, nitrogen can be added to lakes by fixation of atmospheric nitrogen (N_2), mostly by cyanobacteria and can be shed as N_2 gas by microbial denitrification and anaerobic ammonium oxidation (Fig. 5.1). There are no analogous processes that biologically augment or shed phosphorus from lakes. A proportion of the nutrients in lakes is deposited and buried in the sediment; however, some of this may be returned to the water column by internal loading, in particular, phosphorus under anoxic bottom-water conditions or by wind-driven resuspension in shallow lakes or littoral zones. Thus, there is a fraction that is temporarily buried and then recycled and a fraction that is permanently buried (sequestered) in the lake bed. The balance of the remaining nutrients is flushed out of the lake via the outflow. The outflow may be a surface flow such as a river or water may seep out of lakes as groundwater. All these sources and sinks together make up the nutrient budget.

Most of the nutrients enter a lake with the input of water; therefore, it is important to understand the lake water balance of the lake:

$$\begin{aligned}\text{Change in Storage} = {} & (\text{Total surface inflow} + \text{Precipitation on the lake} + \text{Groundwater in}) \\ & - (\text{Outflow} + \text{Evaporation} + \text{Groundwater out})\end{aligned}$$

Over multiannual time scales, the change in lake level and lake volume can be ignored, because the mean annual change in water volume is typically small relative to the other components of the water balance, especially in deep lakes.

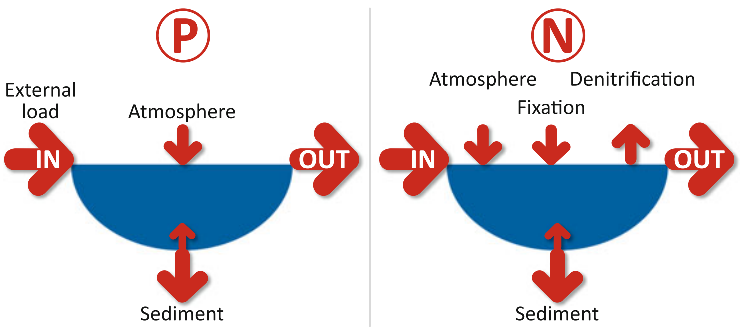

Fig. 5.1 A mass balance for nitrogen and phosphorus, showing inputs from the catchment, by atmospheric deposition, and by internal loading from the sediment, and outputs by flushing through the outlet and burial in the sediment. Microbial processes (e.g. N fixation and denitrification) can add complexity to the nitrogen balance

The nutrient mass balance is given by:

$$\text{Phosphorus : Catchment inputs} + \text{Atmospheric inputs} = \text{Change in contents} + \text{Sequestered} + \text{Outflow}$$

$$\text{Nitrogen : Catchment inputs} + \text{Atmospheric inputs} + \text{N-Fixation} = \text{Change in contents} + \text{Sequestered} + \text{Denitrification} + \text{Outflow}$$

The external nutrient load is the sum of nutrient inputs from the catchment including groundwater, by atmospheric deposition, and by nitrogen fixation in the lake. In lakes, nutrient cycling and fluxes of nutrients between the sediment and the water column occur and where a net flux from the sediments to the water column exists, this "internal load" is an additional input on the left side of the equation, usually altering the amount of nutrients sequestered and flowing out of the lake.

The above steady-state formulation of a nutrient budget is applicable to annual or longer time scales. Such an example is provided in Feature Box 5.1 for Lake Delevan, Wisconsin, USA, where restoration requirements for nutrient load reductions were assessed over long time scales independent of short-term events that negate some of the principles of steady-state conditions. Short-term changes in a nutrient budget depend on a combination of temporal changes in nutrient loads, hydraulic inputs, hydraulic residence times, mixing regimes, burial and sequestration in the sediment, and microbial nitrogen transformations. Furthermore, spatial variation in nutrient dynamics within lakes often may need to be considered. Spatial variations can be substantial in elongated lakes and those with embayments.

Box 5.1 Long-Term Rehabilitation of Delavan Lake, Wisconsin, USA

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Excessive nutrient loading to Delavan Lake, Wisconsin, USA, similar to what has occurred to many lakes throughout the world, resulted in it becoming hypereutrophic with severe blue-green algae blooms and a fishery dominated by rough fish. Based on a preliminary study, phosphorus (P) was identified as the nutrient of primary concern and almost 80% of the external P loading was attributed to sewage-treatment effluent and leakage from septic tanks (Fig. 5.2; USEPA 1974). Therefore, a new more complete sanitary sewer system was installed and its effluent was diverted around the lake in 1981. Despite this diversion, water quality in the lake still ranked among the worst in Wisconsin, and the cyanobacteria bloom during 1983 was among the densest on record. Therefore, a study was conducted by the U.S. Geological Survey, from which a refined detailed P budget was constructed for the lake (Fig. 5.2; Field and Duerk 1988). From results of this study, one of the most comprehensive lake

(continued)

Box 5.1 (continued)
rehabilitation plans ever developed was implemented from 1990 to 1993 to shift the lake from a hypereutrophic condition back to a mesotrophic condition and improve the fishery in the lake (Robertson et al. 2000). The plan was to treat the sources of the problems rather than treat the individual symptoms observed. To reduce external P loading, Best Management Practices were implemented in the watershed, wetlands were created upstream of the major tributary of the lake, and a peninsula was built to divert inflow through the lake. To reduce internal P loading, alum (aluminium sulphate) was applied throughout the lake. To rehabilitate the fishery, rotenone was applied to the lake and its tributaries to remove all coarse fish from the system, and the lake was restocked with game fish.

Rehabilitation efforts were extremely successful and resulted in the lake becoming oligotrophic and dominated by game fish (Robertson et al. 2000). Continued monitoring of P loading and in-lake water quality demonstrated that much of the improved water quality was short-lived because P loading could not be reduced to that estimated to be needed by the Canfield–Bachmann eutrophication model (Canfield and Bachmann 1981), and P concentrations again exceeded the established goals as early as 1993 (Fig. 5.3). Even with the high P loading, however, water clarity remained relatively good (with summer average Secchi depths typically exceeding 3 m) because of the biomanipulation and changes in the food web (Robertson et al. 2000). Changes in water quality continue to occur because of changes in P loading, invasion of zebra mussels, and a continually changing food web and reductions in large zooplankton. With continued efforts to improve the water quality (continued dredging of upstream wetlands and inlet), P concentrations in the lake (Fig. 5.3) gradually decreased to the original goals for the rehabilitation. Only through scientifically guided efforts could Delavan Lake reach the original goals of the rehabilitation programme for P and water clarity.

Lakes are most often sinks for nutrients in the landscape but, under certain (non-steady-state) conditions, they may be net nutrient sources to downstream environments. Simple mass-balance models have been established to relate the nutrient load to nutrient concentration in lakes, generally where no significant internal loading is expected to occur, and assuming steady-state conditions (Vollenweider 1975, 1976; OECD 1982). Such mass-balance models estimate concentrations of nutrients in lakes from nutrient loading rates, lake volume, and by estimating the retention (sequestration) of nutrients. Estimates of retention can be carried out in numerous ways, but the most common is to rely on statistical relationships between measured retention rates and either the hydraulic residence time or the annual areal hydraulic load (Dillon and Kirchner 1975). When retention coefficients are estimated from lakes with minimal internal loads, then discrepancies between the thus-predicted mean nutrient concentrations and those that are measured in lakes can reveal the extent of internal loading as anomalies in the nutrient retention parameter.

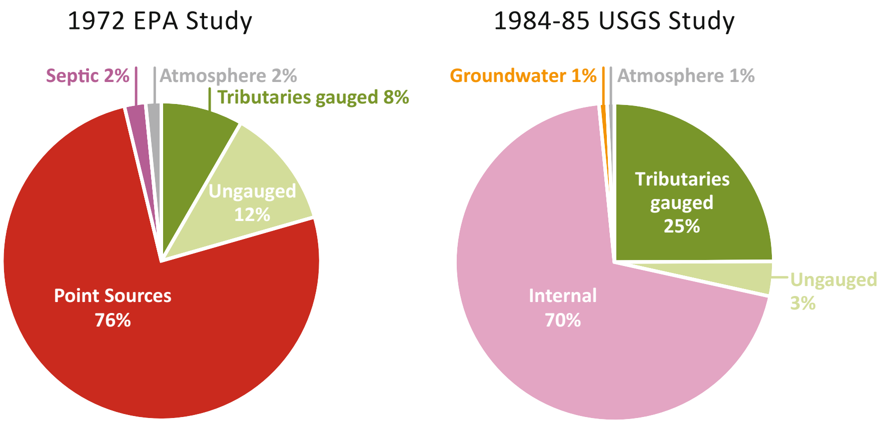

Fig. 5.2 Phosphorus budgets for Delavan Lake from a 1972 U.S. Environmental Protection Agency study (USEPA 1974) and a 1984–1985 U.S. Geological Survey study (Field and Duerk 1988)

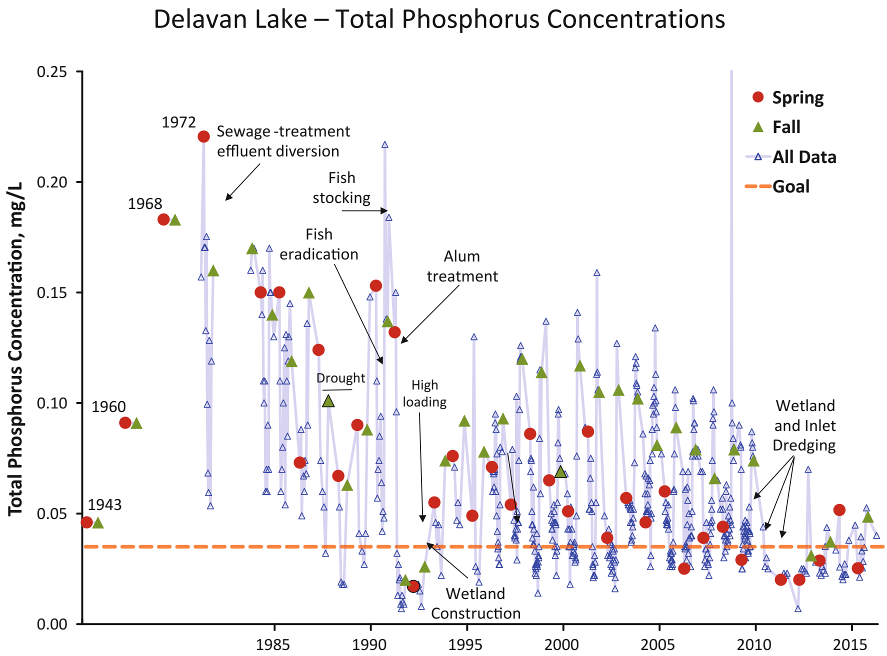

Fig. 5.3 Total phosphorus concentrations measured near the surface of Delavan Lake, Wisconsin, USA. Early data, phosphorus goal, and labelled management actions taken in the lake are described in Robertson et al. (2000), and recent data are from USGS (2016)

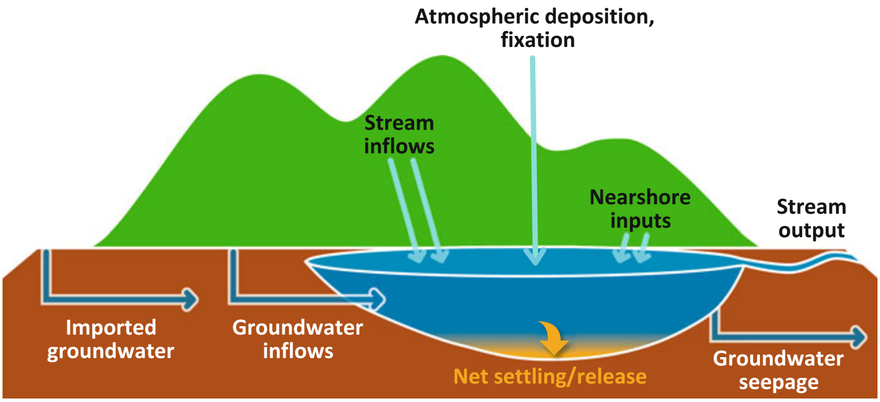

Fig. 5.4 Schematic of sources and sinks of nutrients in lakes

5.1.2 Sources of Nutrients

Quantifying the various nutrient sources (Fig. 5.4) serves as a first step to identifying key foci and opportunities for lake remediation. Usually nutrient budgets are assembled based on long-term average loading, evening out shorter-term variations and storages. Catchment-derived nutrients can enter lakes either via stream inflows or by groundwater.

5.1.2.1 Stream Inflows

Nutrient input (loading) from streams entering the lake may be measured based on monitoring of stream concentrations and flow rates. Usually, this entails setting up flow monitoring stations to capture the variation of flow over time and determining nutrient concentrations in stream samples. Ideally, storm events will be sampled (manually or by automatic samplers) because a large proportion of the load can arrive during storm events (Danz et al. 2010). A range of methods are available for estimating the stream loading from these measurements including the use of daily interpolation and rating curves (Quilbé et al. 2006; Snelder et al. 2014). Uncertainty in measured loads can be considerable, and several of the statistical methods enable estimation of some aspects of the uncertainty in load. Typically, it will not be possible to sample from all the tributaries so the inputs from other tributaries can be estimated on an area pro-rata basis, especially if the catchment is fairly homogenous and the 'missing' areas are small, or with catchment models.

Modelling serves as an alternative to (or complement to) measurements, enabling more insight into the contribution of various source types. The simplest type of model is based on export coefficients, whereby the area of each land use (assessed from spatial analysis) is multiplied by an export coefficient (an estimated average load per unit area attributable to a land cover, land use, slope, climate, etc.) and then summed over the different land uses for the full lake catchment. Nutrient point sources can be added to this estimate. Export coefficients for different land uses are available (Elliott and Sorrell 2002; McDowell and Wilcock 2008). Whole catchment estimates extracted from compilations of land parcels and point sources involve considerable uncertainty so it is desirable to use carefully measured export coefficients for relevant land uses obtained in the vicinity of the lake for which the budget is being constructed. Source coefficients for pastoral catchments may also be estimated from the farm-scale OVERSEER Nutrient Budget model (Selbie et al. 2013) which is used widely in New Zealand. The OVERSEER losses for P are calibrated to measurements from small streams, and so already account for some degree of catchment nutrient attenuation, while the losses for N are based largely on leaching from soils (estimated from lysimeters) and so need to be corrected to estimate losses to groundwater and streams (e.g. Elwan et al. 2015). It should be noted that the OVERSEER model assumes best management practice at the land phase. Export coefficients can be modified to take account of additional mitigation measures [see McDowell et al. (2013) for a summary], and such approaches have been applied recently to some lakes in the Bay of Plenty (e.g. McIntosh 2012).

The catchment nutrient export model CLUES (Catchment Land Use Environmental Sustainability) can be used to estimate the flux of nutrients in New Zealand streams under different land uses (Woods et al. 2006a).[1] CLUES uses a simplified version of OVERSEER to estimate losses from pasture areas together with other source coefficients and decay coefficients obtained by calibration to measured loads from around New Zealand. The loads into a lake can be determined from the sum of the tributary catchment inputs or from the lake outlet load adjusted for modelled in-lake losses (Verburg et al. 2012) and are annual means without temporal resolution.

A range of more sophisticated dynamic, process models is also available for assessing external loadings to lakes. These offer the prospect of decomposing loads over time and into different nutrient fractions, but entail considerable effort to set up and run. Generally, the uncertainties of the model outputs are not well characterised in these models and so they should be used with caution in setting nutrient load limits. Anastasiadis et al. (2013) reviews a number of models used in New Zealand.[2,3] A detailed summary of the use of catchment models to estimate nutrient loading into lakes is given in Chap. 2.

5.1.2.2 Groundwater Inputs

Nutrients, especially N, may enter or leave lakes through groundwater (Gibbs 1987). For example, Lake Horowhenua (Manawatu region) receives a large part of its N

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[1] https://www.niwa.co.nz/freshwater-and-estuaries/our-services/catchment-modelling/clues-%E2%80%93-catchment-land-use-for-environmental-sustainability-model

[2] https://teamwork.niwa.co.nz/display/IFM/Framework+for+Interoperable+Freshwater+Models

[3] http://tools.envirolink.govt.nz/about/

inputs from groundwater (Gibbs and White 1994). One implication of nutrients coming in with groundwater is that the true catchment of a lake can be larger than the surface water catchment (e.g., Lake Rotorua, White et al. 2014). Groundwater measurements may be required to infer the entire hydrological catchment area as opposed to surface water catchment area. In extreme cases, such as some dune lakes in Northland, there are minimal surface inflows and outflows and the lake nutrient budget is dominated by perched or regional groundwater systems (Champion and de Winton 2012). Groundwater flows are usually difficult to measure directly but can be estimated from water budgets, lake and groundwater measurements, and groundwater models. For example, groundwater outflows are unlikely if groundwater contours all slope towards a lake or if the lake is perched above the regional groundwater table. If measured, total tributary inputs are less than expected based on catchment rainfall and evapotranspiration; this indicates that additional inputs are likely to enter the lake via groundwater. Similarly, net groundwater inflows can be inferred from a water budget of the lake using measured tributary flows and lake surface outflow, along with estimated net inputs to the surface of the lake (precipitation less evaporation). The concentrations associated with the groundwater can be measured using bores, piezometers, and seepage meters, or they can be estimated from catchment models. While typically groundwater can have elevated levels of nitrogen (as nitrate or ammonium), under some circumstances (old or deoxygenated groundwater), it can also carry significant quantities of phosphorus (McDowell et al. 2015; Moreau and Daughney 2015; Morgenstern et al. 2015).

5.1.2.3 Atmospheric Inputs

While water quality in most lakes is strongly influenced by the nutrient loading from their catchments, sources from further away can contribute nutrients directly on the lake surface by atmospheric deposition. The proportion of the nutrient load from atmospheric sources can be substantial in lakes situated in relatively pristine catchments. The main nutrients in atmospheric deposition are nitrate (NO_3) and ammonia/ammonium (NH_3/NH_4) and to a lesser extent also phosphate (PO_4). The main sources of atmospheric deposition are derived from the combustion of burning of fossil fuels and from intensive farming practices. The latter is especially important in New Zealand because of low population density and lack of heavy industry. Atmospheric deposition occurs as dry (dust and aerosols) or wet (precipitation) deposition. Wet deposition is usually larger than dry deposition.

New Zealand's atmosphere is considered relatively pristine because it is surrounded by ocean, but confirmation of this is weak because little monitoring of atmospheric concentrations and atmospheric deposition of nutrients has been carried out in the past two decades. The deposition of N and P (wet and dry combined) averaged across studies of atmospheric deposition carried out on the North Island was 6.37 ($\pm$2.38 standard deviation) kg N ha^{-1} y^{-1} and 0.34 ($\pm$0.17) kg P ha^{-1} y^{-1}. The average of studies on the South Island was 2.35 ($\pm$0.65) kg N ha^{-1} y^{-1} and 0.23 ($\pm$0.18) kg P ha^{-1} y^{-1}, respectively (Verburg unpublished data). Atmospheric deposition may be a higher proportion of the total nutrient load in lakes with very small catchments and in oligotrophic lakes where nutrient inputs from the catchment are low. For instance, in Lake Taupo, atmospheric deposition accounts for about 20% of both N and of P loads (Vant and Gibbs 2006). Dry deposition was 37% of total deposition for inorganic phosphorus and 12% for inorganic nitrogen at Lake Taupo according to White and Downes (1977). These proportions were similar to those reported by Fish (1976) for PO_4 (21%), NO_3 (5%), and NH_4 (7%) near Lake Rotorua (Bay of Plenty). Vant and Gibbs (2006) found that 42% of N and 54% of P in atmospheric deposition was inorganic. There are few recent studies estimating the average contributions of atmospheric deposition to lake nutrient loads and in view of recent agricultural intensification in many parts of the country, the proportion of lake nutrient budgets contributed by atmospheric deposition has probably increased.

5.1.3 Internal Sources of Nutrients to the Water Column

5.1.3.1 Geochemistry Effects on Nutrient Release From the Sediment

In addition to external nutrient loading from the catchment and atmospheric deposition, internal loading can be important in determining the in-lake nutrient concentrations, in particular in lakes which have undergone a prior period of cultural eutrophication. This can result in 'legacy loads'. Most attention has been focused on internal loading of P because of its importance in causing eutrophication and its facultative binding to minerals in lake sediments. However, internal N loading can be substantial as well.

Environmental conditions can favour the release of nutrients from the sediment on the lake bottom, especially when in contact with anoxic bottom waters in lakes that thermally stratify for a substantial part of the year. Anoxia in bottom waters occurs when there is an excess of organic matter and its decomposition after sinking out consumes oxygen faster than it can be replenished from the surface layer while the lake is stratified. Phosphorus is a component of sedimented particulate organic matter in lakes, and sulphur and various minerals including iron and manganese oxyhydroxides, aluminium, and calcium can effectively bind phosphorus and thereby control the rate of P release from the sediment (Sondergaard et al. 2001; Gächter and Müller 2003). Under anoxic conditions, iron and manganese oxyhydroxides (if present) become chemically reduced, dissolving and releasing phosphate into sediment pore waters where it can diffuse into the overlying water column. P concentrations in the hypolimnion often increase in lakes during stratified periods, in particular in the more eutrophic lakes (Nürnberg 1984; Søndergaard et al. 2003; Hamilton et al. 2004; Özkundakci et al. 2011), as nutrients are released from the sediment. Potentially large amounts of P can be released back to the water column which can eventually be entrained into the productive euphotic zone of the lakes following the breakdown of stratification. These nutrient pulses can stimulate phytoplankton production during winter in monomictic New Zealand lakes. In polymictic systems such as Lake Rotorua (Bay of Plenty), pulses of nutrients supplied to the euphotic zone following episodic periods of stratification during warmer months can contribute to rapid phytoplankton bloom formation. Such conditions have become more prevalent in many lakes in recent years.

5.1.3.2 Accumulation of Dissolved P in Hypolimnia

Several studies have estimated internal loading from the accumulation of P in the hypolimnion in stratified lakes during the summer. Nürnberg (1984) found thus estimated internal P loading to be similar to results for the net internal load obtained from the mass balance. In Lake Okaro, the P concentration at 14 m depth increases by about 600 mg m^{-3} during summer (Özkundakci et al. 2011). Another way to estimate internal P loads is to compare surface layer total phosphorus (TP) concentrations in the winter mixing period and at the end of the stratified season and multiplying the difference by mean depth (Dillon and Molot 1996). In lakes where part of the P accumulated in the hypolimnion is primarily derived from external sources, the hypolimnetic accumulation rate may not be a good estimator for the internal load from sediments. The accumulation of nutrients in the hypolimnion may, apart from internal loading, also result from sinking out and decomposition of algal matter below the thermocline and, without access to the surface layer, it is prevented from flushing through the outlet. While a literature review carried out by Nürnberg (1984) found on average there was no P release in oxic sediment core tubes, P and N may accumulate in the hypolimnion in lakes with an oxic sediment–water interface (White et al. 1980; Vincent and Downes 1981; Sondergaard et al. 2001). For instance, in oligotrophic Lake Taupo (Verburg and Albert 2016) where dissolved oxygen in bottom water rarely drops below 7 mg L^{-1}, DRP near the maximum depth at 150 m increases by about 10 mg m^{-3} over summer. Under oxic conditions, P release is mainly governed by microbial cycling of particulate organic P. In deep lakes that are stratified throughout summer such as Lake Taupo, nutrients accumulating in the hypolimnion, either by internal loading or by microbial cycling, only affect the concentrations in the surface layer during mixing in autumn and winter, apart from low rates of entrainment across the thermocline during summer.

5.1.3.3 pH-Dependent P Release

Elevated water pH levels above about 9, as can occur in response to intense algal blooms, can also enhance P fluxes from sediments to water (Seitzinger 1991; Jin et al. 2006). High pH caused by planktonic cyanobacterial blooms has been shown to dissolve metal-hydroxide-bound P in lake sediments while dissolved inorganic P was being taken up by the cyanobacteria (Hao et al. 2016).

5.1.3.4 Sediment N Release

Few studies have examined the effects of dissolved oxygen concentration and variations in pH on the release of N from lake sediments into overlying waters. However, Zhang et al. (2014) showed that conditions of low dissolved oxygen concentration (<1 g m^{-3}) coupled with a pH < 6 promoted the release of NH_4 from sediments while oxic conditions with a pH > 8 promoted the release of NO_3 from the sediments of Erhai Lake, China.

5.1.3.5 Wind-Induced Nutrient Resuspension

In shallow lakes, P release can also be caused by wind mixing and resuspension (Sondergaard et al. 2001) as is probably the case in Lake Omapere in Northland (Verburg et al. 2012). Lake Omapere is a shallow, polymictic lake (maximum depth = 2.6 m, surface area 1250 ha) which has undergone severe human-induced eutrophication. It is estimated that the net internal P load during the period 2005–2009 was 4.1 t y^{-1}, which was 160% of the annual external load estimated using CLUES (Verburg et al. 2012). The most likely explanation for the high internal loading is that nutrients deposited in the sediments in the past were recycled back into the water column either by release under anoxic bottom-water conditions or by sediment resuspension and pore water entrainment during windy conditions.

The effect of sediment resuspension on concentrations of bioavailable forms of N and P in the water column is complex. In Lake Waihola (Otago), sediment resuspension increased N and P concentrations in the water column, but entrainment of dissolved N and P was relatively small (Schallenberg and Burns 2004), similar to observations during resuspension events in seven shallow lakes in the South Island of New Zealand (Hamilton and Mitchell 1997). It was also noted that changes in water column turbidity did not correlate with dissolved N and P concentrations, implying that the concentrations in the pore waters were nearly in equilibrium with the water column and/or that partitioning of water column nutrients (adsorption/desorption) complicates simple entrainment dynamics and nutrient availability to phytoplankton. Evidence for frequent equilibration between P in pore waters and the water column was reported from a shallow Danish lake, where the extent to which the water column dissolved P concentrations were elevated by sediment resuspension events was influenced by the time since the previous resuspension events (Søndergaard et al. 1992).

In Lake Waihola, nutrient fluxes from resuspension often stimulated phytoplankton production, indicating that resuspension events often alleviated nutrient deficiency in the phytoplankton (Schallenberg and Burns 2004). Sediment resuspension in Lake Tuakitoto (Otago) sometimes alleviated phytoplankton N deficiency (Ogilvie and Mitchell 1998).

5.1.3.6 N-Fixation

Terrestrial nitrogen fixers such as gorse and broom fix and leak substantial amounts of N (Drake 2011) and may contribute up to 200 kg N ha^{-1} y^{-1} to the nutrient load from New Zealand catchments (Magesan et al. 2012). Nitrogen fixation can also occur in lakes, by floating weeds such as *Azolla* [fixation is carried out by its symbiont, a N-fixing cyanobacteria *Dolichospermum* (formerly *Anabaena*) *azollae*] or planktonic N-fixing cyanobacteria such as *Dolichospermum* sp. (Kellar and Goldman 1979). In stratified lakes, planktonic N-fixing cyanobacteria are often most abundant at the end of summer when bioavailable forms of N have been depleted from the surface layer and warm stratified conditions favour cyanobacteria (Wood et al. 2010). Nitrogen-fixing species of cyanobacteria may then enhance N concentrations, but N-fixation is rarely taken into account in lake N budgets because of lack of information on fixation rates. MacKenzie (1984) concluded that in two eutrophic lakes bioavailable N derived from N-fixation in lake sediment was small compared to inputs of N from other sources. A benthic N-fixation rate of 160 kg N y^{-1} in oligotrophic Lake Taharoa (Northland region; Lam et al. 1979), which has low N concentrations, amounts to 4% of the catchment load estimated with CLUES (Verburg unpublished data). The contribution by pelagic N-fixation may be more substantial, especially in lakes that have a low TN:TP ratio (total nitrogen : total phosphorus, Schindler et al. 2008). Viner (1987) mentioned that in Lake Rotongaio (Waikato region), the amount of N supplied by fixation was thought to be about 25% of the dissolved N brought in by the inflows, although no data were reported.

5.1.4 Fates of Nutrients in Lakes

5.1.4.1 Sequestration of N and P, and Conversion of Bioavailable Forms of N to N_2

For P only two loss pathways exist: it is buried in the sediment or is flushed from the lake via the outlet. The proportion of N not exported via the outlet is either retained by sedimentation and sequestration in the lake bed or is removed from the lake by denitrification, anammox, and other microbial processes (Burgin and Hamilton 2007). In New Zealand, rates of denitrification and other microbial processes which shed N from lakes have received little attention. However, these undefined losses of N from lakes can be represented within the N retention coefficient (R) in simple mass-balance models (Saunders and Kalff 2001). The N retention coefficient is the proportion of the external load that does not leave the lake in its outflow which accounts for burial in the sediment and, in the case of N, permanent loss of fixed N from lakes by microbial processes. On average about two-thirds of N retention in lakes is accounted for by denitrification and the remainder by uptake by aquatic plants and sedimentation resulting in permanent burial (Saunders and Kalff 2001) but these proportions are highly variable.

The retention coefficient can be estimated from the difference between nutrients entering lakes and nutrients leaving the lake through the outlet. The observed retention is expressed as a proportion of the external nutrient load by:

$$R_{\mathrm{obs}} = 1 - \frac{O[\mathrm{P}]_{\mathrm{lake}}}{L_{\mathrm{ext}}} \tag{5.1}$$

where O is the annual outflow rate (m^3 y^{-1}), $[\mathrm{P}]_{\mathrm{lake}}$ is the observed average total P concentration in the lake (mg m^{-3}), and L_{ext} is the total annual external loading rate of P (mg y^{-1}).

In lakes where N-fixation is a large component of the N mass balance but not taken into account, the total load of N is higher than estimated. It follows that in that case Eq. 5.1 will result in too low of an estimate of the retention coefficient for N.

Retention by permanent burial (sequestration) in the sediment can also be estimated by examining sediment cores and, after careful dating of sections of the core, calculation of mass accumulation rates of N and P. This method can provide a record of long-term changes in nutrient loading in the past decades and centuries, but because P geochemistry is complex, P mobility in the sediments may confound simple inferences about historical P loads derived from the analysis of sediment core profiles.

The retention coefficient has been estimated in a large number of lakes. The proportion of nutrient retained in the lake is mainly a function of the residence time with a greater proportion retained in lakes with long hydraulic residence times. The following equations allow prediction of retention of P and N in lakes under oxic conditions, but not in lakes where internal loading from the sediment may occur, which reduces retention especially when bottom waters become periodically anoxic.

5.1.4.2 Estimation of the Phosphorus Retention Coefficient

Vollenweider (1976) found empirically that

$$R_{\mathrm{pred}} = \frac{\sqrt{T_w}}{\left(1 + \sqrt{T_w}\right)} \tag{5.2}$$

where R_{pred} is the predicted lake retention coefficient for P, T_w is the hydraulic residence time (y) given by $V/O = z/q_s$, q_s ($= O/A$) is the annual hydraulic load (m y^{-1}), A is the lake surface area (m^2), V is the lake volume (m^3), and z the mean depth (m).

Several authors derived R_{pred} from q_s using an equation of the form $R_{\mathrm{pred}} = v/(v + q_s)$, where v (m y^{-1}) is the settling velocity of P (Dillon and Kirchner 1975). Assuming v to be a constant, the value of v was found by fitting the equation to data from a set of lakes (Kirchner and Dillon 1975; Chapra 1975; Dillon and Kirchner 1975; Vollenweider 1975). However, Dillon and Molot (1996) showed from empirical results that v can vary substantially between lakes and with time. Also Ahlgren et al. (1988) pointed out that v should not be treated as a constant. Nürnberg (1984) adapted $R_{\text{pred}} = v/(v + q_s)$ into the following expression for predicted P retention: $R_{\text{pred}} = 15/(18 + q_s)$. Nürnberg's (1984) method sets unrealistic bounds for R_{pred} and v. This model for P retention suggests that v is a monotonically increasing function of q_s with an asymptotic maximum of $v = 15$ m y^{-1}. However, in the empirical results of Dillon and Molot (1996), v varied from 3 to 27 m y^{-1} in their set of 8 lakes. From rearranging equations (not discussed in detail here), it follows that instead of a function of q_s, v is controlled by the ratio of z and $\sqrt{T_w}$. This allows for a greater range in v and more variability in the relationships between v and T_w and v and q_s. In 12 of the Bay of Plenty lakes, v, estimated as the ratio $z{:}\sqrt{T_w}$, varied from 6 to 25 m y^{-1}, consistent with the results of Dillon and Molot (1996), whereas the maximum v that follows from Nürnberg's (1984) equation was 13 m y^{-1}. In addition, the theoretical maximum value of R_{pred} according to Eq. 5.2 is 1.0, while with Nürnberg's (1984) equation R_{pred} can be no higher than 0.83. There is no logical reason for a maximum limit to R less than 1.0. Among 12 Bay of Plenty lakes, R_{obs} was >0.83 in four lakes (maximum 0.95). In these lakes, R would be underestimated by the method of Nürnberg (1984).

The method of Nürnberg (1984) to estimate P retention is commonly used (e.g. Rutherford and Cooper 2002) because it was described in the Lake Managers Handbook by Vant and Hoare (1987). However, it is not commonly used in other places of the world (Prairie 1989; Brett and Benjamin 2008). For instance, the two most influential limnology textbooks of the past decade (Wetzel 2001; Kalff 2003) do not mention the method of Nürnberg (1984) but instead advocate Vollenweider's equations (Eqs. 5.2 and 5.4) to estimate retention and predict lake P concentrations from catchment loading. Brett and Benjamin (2008) carried out an assessment of lake P retention and lake P concentration models using data from 305 lakes, the largest data base examined in such comparative studies thus far. Brett and Benjamin (2008) showed that expressions for retention based on residence time, such as the Vollenweider expression (Eq. 5.2), give better results than expressions that effectively assume a constant particle settling velocity (Dillon and Kirchner 1975; Vollenweider 1975) or a limited range in particle settling velocity, such as is inherent to the method which Nürnberg (1984) derived from fewer lakes (54 lakes).

5.1.4.3 Estimation of the Nitrogen Retention Coefficient

The prediction of N retention requires a different expression than used for P because N retention in the sediment is affected differently from P retention by binding properties of other constituents and because N is lost from the lake by denitrification. In deep and fully oxic lakes, N retention is usually less efficient than P retention (Wetzel 2001). The average N retention found in lakes is 34% (Saunders and Kalff 2001). The predicted retention coefficient, using an equation of Harrison et al. (2009) for total nitrogen removal in lakes, is

$$R_{\text{Npred}} = 1 - \exp\frac{-aT_w}{z} \tag{5.3}$$

with $a = 6.83$. The value of a is higher in reservoirs ($a = 13.66$), which results in a higher predicted retention. The N retention coefficient increases with water residence time (Windolf et al. 1996) as does the P retention coefficient. The differences in retention of N and P can lead to changes in the TN:TP ratio in the lake compared with the external inputs (Verburg et al. 2013), potentially affecting the key drivers of phytoplankton growth.

5.1.4.4 Flushing and Seepage

A proportion of the nutrients that enter the lake leave again through the outflow. The nutrients exported can be estimated as the product of the outflow discharge and the concentrations in the outlet. If the hydraulic discharge from the outflow isn't known, the long-term average outflow can be estimated for New Zealand lakes using the model of Woods et al. (2006b), which uses the mean annual rainfall and potential evapotranspiration to determine the mean annual runoff. The estimated outflow is the total loss of water from the lake after accounting for rainfall and potential evapotranspiration and, therefore, includes losses to groundwater systems (Woods pers. comm.).

To estimate the amount of nutrients flushed annually by the outlet, if good direct measurements of the concentrations in the outlet do not exist, the nutrient concentrations in the outflow can be assumed equal to those in the surface waters of the lake. The difference between the amounts of nutrients entering and leaving the lake generally increases with residence time. In a lake with a long residence time such as Lake Taupo (11 years), the proportion of the input of P that leaves the lake is expected to be only in the order of 20% (Eq. 5.2), while almost 50% of N is predicted to leave the lake via the outlet (Eq. 5.3).

5.2 Flipping Lakes

At least 37 lakes in New Zealand have undergone rapid regime shifts between two apparently stable states (Schallenberg and Sorrell 2009): (1) a clear water state in which macrophytes are present and P net accumulates in the sediment, building a P legacy potentially available for later release and (2) a turbid water state in which macrophytes are absent and sediment P may be depleted by high rates of internal loading (Verburg et al. 2012). Some of the lakes, e.g. Lake Omapere (Northland), Lake Forsyth/Wairewa (Canterbury), Upper Tomahawk Lagoon (Otago) exhibit a flipping behaviour, alternating between these two states.

In Lake Omapere, Verburg et al. (2012) observed average losses through the outlet of TN and TP (using Eq. 5.1) that exceeded the external load by 151% and

213%, respectively, indicating that the net internal load of these nutrients exceeded the external load, resulting in a negative retention. Because of high internal loads, nutrient concentrations in the lake and in the outlet were higher than the average in the inflows from the catchment. Although it is possible to have high net internal loadings which result in no net retention for a period of time, in the long run the nutrients in the sediment will become depleted and losses through the outlet can't exceed external loads (at least for P, N fixation may augment external N loads from the catchment). Therefore, the observations from Lake Omapere show that the lake was in a transitional state (i.e. net loss of P must mean depletion of the sediment nutrient pool) and is consistent with the behaviour of "flipping" between alternative stable states. In Lake Omapere, the clear water and turbid water phase each lasted about 8 years and in recent years the lake has been improving and is likely to enter a new clear water phase.

During clear water phases, weed beds prevent the resuspension of nutrients in decomposing organic matter, lake water nutrient concentrations are low, and the nutrient content of the sediment is expected to increase. This phase ends when the weed beds collapse as nutrients start to increase in the water column, enhancing phytoplankton growth which reduces the amount of light available for re-establishment and growth of the weed beds. In the following turbid phase, the nutrient concentrations in the lake are much higher than can be explained from terrestrial inputs because during this phase nutrients sequestered in the sediment are returned to the water column by release when lake bed sediments become anoxic or by resuspension. A portion of these recycled nutrients is flushed from the lake through the outflow and, as a result, after several years, nutrient concentrations in the lake decline eventually allowing the return of macrophytes.

The existence of flipping lakes highlights that the nutrient budgets of many lakes may not be in equilibrium and that net fluxes of nutrients may occur both from sediment to water and vice versa. However, in theory, net fluxes from sediment to water should not be sustainable in the long term and the existence of such a flux exceeding the external load on an annual basis indicates that the lake is not in a steady state.

5.3 Nutrients Contained in Biota: Fish, Macrophytes, and Phytoplankton

Generally, biota are not considered as part of lake nutrient budgets because nutrients flow in and out of organisms but the amount of nutrients they contain are generally assumed to be constant over time. However, from the point of view of nutrient harvesting as a nutrient mitigation strategy, it may be of interest to account for biomasses and harvest rates of different biotic components in mass-balance models.

Table 7.1 shows an estimate of the nutrients contained in phytoplankton in Lake Horowhenua (Horizons region), based on chlorophyll *a* concentrations (Verburg et al. 2010), and assuming an algal N: chlorophyll *a* ratio of 7.5 and an algal P: chlorophyll *a* ratio of 1 (middle of the range of weight ratios under moderately nutrient limiting conditions, Healey and Hendzel 1979). This amounted to 2253 kg N and 300 kg P, or 17% and 29% (Table 7.1), respectively, of the N and P in the water column (Verburg et al. 2010). The amount of nutrients contained in phytoplankton was higher than contained in macrophytes or fish.

In Lake Horowhenua, macrophyte development measured by hydroacoustics was highest in spring. In the part of the lake where macrophyte harvesting is being considered (118 ha, or 39% of the full lake area), 0.7 g N m^{-2} and 0.1 g P m^{-2} were contained in the macrophyte biomass (de Winton et al. 2015). This amounted to 827 kg N and 118 kg P, or 6% and 12% (Table 7.1) respectively, of N and P in the water column (Verburg et al. 2010). De Winton et al. (2015) estimated, based on the seasonal standing crop and growth rates, that up to 1.6 t of N and 0.25 t of P could be removed over summer and 3.60 t of N and 0.55 t of P in spring. The sum of summer and spring harvests could account for 2.5% and 3.4% of the external N and P loads, respectively, estimated with CLUES (Table 7.2), which is modest in comparison with the amount of nutrients flushed through the outlet (annual average 24 and 15%, respectively). However, nutrient removal would not be the primary incentive of macrophyte harvesting in Lake Horowhenua, which would be to reduce cyanobacteria blooms by reducing the pH (by reducing photosynthesis by macrophytes) and by reducing macrophyte biomass that dies off during summer, decomposing and causing anoxia. High pH and anoxia both enhance P release from the sediment during spring, summer, and autumn in Lake Horowhenua, stimulating cyanobacteria blooms. Moreover, at the very pH reached in Lake Horowhenua, enough NH_4 is converted to NH_3 to cause toxicity.

In Lake Rotoehu (Bay of Plenty region), up to about 4 t N and 0.6 t P has been removed per year by harvesting about 3.5 kt of hornwort which consisted of 4.0% dry matter with 3.0% of N and 0.4% P (P. Scholes, Bay of Plenty Regional Council, pers. comm.). This would account for 12% and 18% of the external N and P loads, respectively, estimated with CLUES (Table 5.1). Matheson and Clayton (2002) found similar proportions (17% and 19% for the external N and P load respectively). The Bay of Plenty Regional Council has annual removal targets of 2.4 t N and 0.32 t P

Table 5.1 Nutrients in biota and as a percentage relative to the total nutrients contained in the water column (Verburg et al. 2010) in four shallow lakes. Nutrients in fish were estimated based on an assumed 75% water content in wet weights, 12% N in dry weight (based on 29 fish specimens, standard deviation 1.4%, data Verburg), and 3% P in dry weight (Vanni et al. 2002)

Lake	Biota	kg N	kg P	% N	% P
Horowhenua	Fish	341	85	2.6	8.4
	Macrophytes (spring)	827	118	6.3	11.6
	Algae	2253	300	17.2	29.4
Serpentine North	Fish	7.3	1.8	4.9	50.1
Serpentine East	Fish	2.2	0.6	5.0	53.1
Whangape	Fish	6116	1529	19.2	95.4

Table 5.2 Nutrients removed by harvesting weeds or fish in three lakes and the proportion of the external load accounted for

Lake	Biota	P (t y^{-1})	N (t y^{-1})	% N	% P
Horowhenua	Weed	0.8	5.2	2.5	3.4
Rotoehu	Weed	0.6	4	12.3	17.9
Waikare	Fish	0.05	0.18	0.1	0.2

to be removed by hornwort harvesting (Bay of Plenty Regional Council 2011). The greater proportion of the nutrient load that can be removed by weed harvesting in Lake Rotoehu, compared with Lake Horowhenua, is probably explained by the smaller external nutrient load (about 18 times less per m^2 lake surface than in Lake Horowhenua) and by the greater mean depth (about 6 times more), allowing for more macrophyte biomass.

Table 5.1 also shows that fish biomass (excluding eels) estimated for Lake Horowhenua (Tempero 2013) accounted for a smaller proportion of nutrients than in macrophytes. However, in two of the Serpentine Lakes (Waikato region), the estimated proportion of P contained in fish biomass relative to P in the water column was much higher, around 50% (Hicks et al. 2015). In Lake Whangape (Waikato region), the proportion of P in fish biomass was 95% (non-indigenous fish amount to 93% of total fish biomass, Hicks et al. 2015). Fish harvesting has been carried out in Lake Waikare (Waikato region) and, like Lake Whangape, Lake Waikare is infested with non-indigenous fish. Around 6 t of fish have been harvested per year in the past 6 years (B. David, Waikato Regional Council, pers. comm), which amounts to about 0.1% of the annual nitrogen load and 0.2% of the external phosphorus load in Lake Waikare as estimated with CLUES (Table 5.2). Thus, fish and macrophyte harvesting, while contributing to nutrient removal, is not likely to play a major role in reducing bioavailable nutrients in most shallow lakes, but it may speed recovery from eutrophication if combined with other more effective nutrient reduction strategies. Some non-indigenous fish species can prevent macrophyte growth by disturbing the sediment or by grazing, and their removal has indirect positive effects on nutrient concentrations in lakes.

5.4 Lake Restoration by Adjusting Components of the Nutrient Budget

In this chapter, we have discussed how nutrient budgets can be used to understand the many ways that N and P stocks and fluxes affect lake trophic state. Table 5.3 summarises some of the key aspects of lake nutrient budgets that have been highlighted in terms of their potentials for lake restoration. Intervention points in the nutrient budgets are classified in terms of whether they reduce the amount of external nutrient load, alter lake hydrology, target internal nutrient loading, or focus

Table 5.3 Lake restoration efforts that affect nutrient fluxes and stocks to improve the trophic state of lakes. New Zealand references are preferentially provided, where available

Restoration method	Effect on nutrient budget	Reference(s)
External load reduction		
Land use mitigation	Reduce soil losses of N and P	Elliott and Sorrell (2002)
Riparian restoration	Attenuate N and P fluxes from land	Parkyn (2004)
Hydrological alteration		
Increased flushing	Water diversion and/or barrier bar opening to flush nutrients downstream	Sutherland and Norton (2011) and Schallenberg et al. (2010)
In-lake nutrient manipulation		
P flocculation	Bind phosphate in water column, increase P sedimentation, increase P sequestration	Hickey and Gibbs (2009), Gibbs et al. (2008) and de Winton et al. (2013)
Sediment capping	Prevent sediment N and P release, increase sequestration	Hickey and Gibbs (2009)
Destratification/ aeration	Prevent anoxic P and N releases (internal loading)	Hickey and Gibbs (2009) and de Winton et al. (2013)
Inhibition of resuspension	Prevent entrainment of sediment and pore water N and P into water column	Stephens et al. (2004)
Manipulation of biota		
Macrophyte harvesting	Remove N and P from lake	de Winton et al. (2013), Lake Rotoehu Action Plan—Amended (2011) and Matheson and Clayton (2002)
Fish harvesting	Remove N and P from lake, remove non-indigenous fish that disturb the sediment and prevent macrophyte growth	Waikato Regional Council
Phytoplankton harvesting	Remove N and P from lake	Hao et al. (2016)
Enhancement of macrophyte biomass	Increase N and P removal from lake water, increase sediment P binding, increase sequestration	Jellyman et al. (2009) and Gerbeaux (1989)

on altering biotic components of the lake to achieve the desired restoration outcome of reducing nutrient available for phytoplankton growth.

The restoration interventions listed in Table 5.1 haven't all been successfully used in lake restoration, but in some cases they are being trialled (e.g. macrophyte and fish harvesting) or suggested (phytoplankton harvesting) as interventions to facilitate restoration.

Strategies for reducing lake nutrient concentrations should consider the relative contributions and dynamics of these nutrient sources, and estimating nutrient budgets is a useful tool for determining the relative importance of each source. In the long term, however, internal and external loads are linked (e.g. Rutherford 1988) and restoration efforts, while they may include interventions to reduce internal nutrient loading and/or in-lake nutrient stocks, generally succeed in the long term only when they also succeed in reducing external nutrient loads (Søndergaard et al. 2007).

Knowledge of internal loads can improve the prediction of how changes in the external loads could influence lake nutrient status and trophic state. Specifically, estimates of internal load must be accounted for when determining the total P loading to a lake and ultimately setting limits on external loads or when planning in-lake nutrient remediation such as P flocculation and sediment P capping. As described below (Sects. 5.5.2–5.5.4), the *net* internal load does not account for all P released from sediment to the water column, because much of the *gross* internal load (i.e. the recycled internal P load) ends up sequestered in the sediment only after multiple cycles of fuelling primary productivity. The amount of biotically cycled P lies between the net and gross amounts of internal loads. The amount of internally recyclable P is constrained by the amount of available P in the sediment, but this can be difficult to ascertain as P geochemistry is complex and P can be bound in many forms (depending on the sediment geochemistry) and can be mobilised under certain conditions (see Sect. 5.1.3).

5.5 Estimating Nutrient Concentrations and Internal Loading Using Empirical Models

5.5.1 Estimating Nutrient Concentrations Using Empirical Models

For 119 monitored lakes in New Zealand, the annual mean surface water concentrations of TN ranged from 35 to 3476 mg m^{-3} and TP ranged from 3 to 303 mg m^{-3} (Verburg et al. 2010).

Mean nutrient concentrations in lakes depend on their nutrient inputs and nutrient retention, which depends on algal growth and sedimentation as well as hydraulic residence time. Nutrient concentrations can, therefore, be predicted from simple mass-balance equations which incorporate key aspects of both nutrient dynamics and the water budget. One of the first mass-balance models was developed by Vollenweider (1976) and relates in-lake P concentrations to external P loading, the P retention coefficient (R), and the outflow rate (O)

Mean TP concentration in the lake (mg m^{-3}) can be estimated from lake volume, the outflow rate (which together give residence time), and total external P loading, using the Vollenweider (1976) equation:

$$[\mathrm{P}]_{\mathrm{lakepred}} = (1-R)\frac{L_p}{q_s} = \frac{L_p/q_s}{\left(1+\sqrt{T_w}\right)} = (1-R)\frac{L_{\mathrm{ext}}}{O} \tag{5.4}$$

where L_p is the annual external loading rate of P per area of the lake surface (mg m^{-2} y^{-1}). If P retention is not measured, R_{pred} (Eq. 5.2) can be used. Equation 5.4 can be applied for TN as well, by using Eq. 5.3 for R_{pred} and replacing L_p with L_n, the annual external loading rate of N per area of the lake surface (mg m^{-2} y^{-1}).

OECD (1982) presented modifications of the earlier Vollenweider model to predict nutrient concentrations:

$$[\mathrm{P}]_{\mathrm{lakepred}} = 1.55\left[\frac{L_p/q_s}{\left(1+\sqrt{T_w}\right)}\right]^{0.82} \tag{5.5}$$

And for TN:

$$[\mathrm{N}]_{\mathrm{lakepred}} = 5.34\left[\frac{L_n/q_s}{\left(1+\sqrt{T_w}\right)}\right]^{0.78} \tag{5.6}$$

OECD (1982) also offers equations to predict chlorophyll *a* concentrations from nutrient concentrations or from external nutrient loads.

These equations were based on data sets of a large number of lakes where no internal loading was considered to occur (OECD 1982), although internal loading may actually have been present in some of the included lakes in view of their trophic state. In lakes where internal loading occurs at levels higher than in the data set underlying Eqs. 5.4–5.6, mean nutrient concentrations would be enhanced. Therefore, these equations will not predict accurate nutrient concentrations when internal loading is high and in extreme cases nutrient concentrations in a lake may be higher than the average in the inflows from the catchment. In the latter case, retention is negative and more nutrients exit a lake than enter it.

Equations 5.4–5.6 can also be used to predict how nutrient concentrations in a lake might change as a result of different scenarios of land use with associated external loads and of manipulation of the flow into and out of the lake (Verburg and Semadeni-Davies 2016). However, when undertaking such scenario testing, it is important to remember that the models reflect a steady-state condition, reflecting the lake in equilibrium with its catchment. Thus, consideration must be given to the response time, which will be related to the hydraulic retention time of the lake.

5.5.2 Net Versus Gross Internal Loading

Gross internal nutrient loading to a lake represents the entire mass of N and P that is mobilised from lake sediments to the water column through chemical and physical processes including sediment resuspension, ORP (oxidation–reduction potential), or pH-dependent desorption and diffusion. Events of high P release (gross loads) may drive temporary increases in phytoplankton productivity and bloom frequency when entrained into the euphotic zone. Gross internal loads determined in situ by measuring release rates from the sediment can be large compared to external loads, especially in shallow lakes. For instance, Ekholm et al. (1997) report a gross internal load of P that was 356 times the external load and of N that was 57 times the external load during the ice free period (223 days) in Lake Pyhajarvi. In a literature review of 49 shallow lakes, Van der Molen and Boers (1994) found a median gross internal load of 3000 mg P m^{-2} y^{-1}. The average gross internal load was 22% higher than the average external load. Given the high temporal and spatial variation, it is difficult to derive a lake-wide estimate of gross internal loading by measuring P release using in situ sensors or incubated cores at a number of sites.

Over time part of the gross internal load will bind to particulate matter, chemically reprecipitate, or resediment to the bottom incorporated into seston. The difference between the gross internal load and that part which is again retained in the lake is the net internal load; thus, the net internal load is the portion of the internal load that is eventually flushed from the lake via the outflow.

There is a net sedimentation rate of P in a lake when losses of P through the outlet are less than the external inputs. Most lakes have gross sedimentation rates greater than gross release rates and, therefore, have net retention of P. A net release may occur during summer, when bottom water is more likely to be anoxic, but on annual scale there is usually a net deposition. In lakes where observed retention is less than predicted from external loading, based on relationships in lakes where no anoxia occurs in the hypolimnion, the difference may be explained by a net internal load.

5.5.3 Estimating the Net Internal Nutrient Load

Net internal nutrient loading into a lake can be estimated from the difference between measured and predicted nutrient retention. Empirical P loading models have been developed for lakes with ‘normal’ sediment P retention (Ahlgren et al. 1988), in lakes where the hypolimnion does not become anoxic and where no internal load was expected. In these lakes, observed retention of P usually agrees well with retention predicted by Eq. 5.2. However, Eq. 5.4 for the predicted P concentration does not apply to lakes where internal loading is present. Internal loading results in a lower observed proportional retention and a lake P concentration higher than expected given the external load and given the retention expected from the lake’s volume, area, and outflow rate. The net internal loading is estimated as the difference between observed retention and predicted retention:

$$\Sigma P_{\text{int-net}} = \Sigma P_{\text{Rpred}} - \Sigma P_{\text{Robs}} \tag{5.7}$$

where ΣP_{Rpred} is the predicted mass of P (t y^{-1}) retained in the lake without internal loading given by

$$\Sigma P_{\text{Rpred}} = R_{\text{pred}} \Sigma P_{\text{ext}} \tag{5.8}$$

and the actual observed retained mass of P (t y^{-1}) is given by

$$\Sigma P_{\text{Robs}} = R_{\text{obs}} \Sigma P_{\text{ext}} = \Sigma P_{\text{ext}} - O[P]_{\text{lake}} \tag{5.9}$$

where O is the annual outflow rate ($m^3\ y^{-1}$), and $[P]_{\text{lake}}$ is the observed average total P concentration in the lake (mg m^{-3}).

The observed net retention can be negative when more P leaves a lake than enters. However, the net internal load cannot exceed the loss though the outlet (Eqs. 5.10 and 5.11). Rearranging Eq. 5.9 shows the net internal load to be equal to the difference between the observed and predicted mass of TP lost through the outlet:

$$\Sigma P_{\text{int-net}} = O\left([P]_{\text{lake}} - [P]_{\text{lakepred}}\right) \tag{5.10}$$

$$\Sigma P_{\text{int-net}} = O[P]_{\text{lake}} - \left(1 - R_{\text{pred}}\right)\Sigma P_{\text{ext}} \tag{5.11}$$

The TP concentration used to estimate the mean $[P]_{\text{lake}}$ is the mean concentration in the epilimnion, and not the average whole lake concentration or the spring overturn concentration, because $[P]_{\text{lake}}$ is used in Eqs. 5.7–5.11 to estimate the loss through the outlet and the concentration in the outflow is expected to be similar to the surface water or epilimnion concentration. This method to estimate the net internal load is essentially the same as the method used by Nürnberg (1984).

The methods to estimate net internal loading have an advantage over measuring release rates in situ because release rates can be highly variable in time and space and in addition can be offset by sedimentation rates. It is important to realise that $\Sigma P_{\text{int-net}}$ (Eqs. 5.7, 5.10 and 5.11) is the net internal load and not the gross internal load, meaning it is what remains of the internal load after sedimentation is taken into account. In the long term, beyond the time scale of the establishment of a nutrient legacy load in the sediment, net internal loads cannot on average exceed external loads.

The predicted lake P concentration, taking both external and net internal loads into account, is

$$[P]_{\text{lakepred}} = \left(1 - R_{\text{pred}}\right)\frac{\Sigma P_{\text{ext}}}{O} + \frac{\Sigma P_{\text{int-net}}}{O} \tag{5.12}$$

where the net internal load can be estimated confidently by other means (Dillon and Molot 1996) and the external load can be estimated by rearranging Eq. 5.12 (Nürnberg 2009).

5.5.4 Estimating Gross Internal Loading

The equation for the net internal load (Eq. 5.7) considers retention for only the external load and not for the net internal load. Applying the predicted retention of the external load (as in Eqs. 5.8 and 5.12) to the net internal load as well results in an estimate of the gross internal load. Part of the gross internal load is indeed retained upon re-sedimentation, which can be expressed by taking the P load in Eq. 5.4 as the sum of the external and the gross internal loads, with $\Sigma P_{\text{int-gross}}$ as an estimate of the gross internal load, and rewriting Eq. 5.4 as

$$[P]_{\text{lakepred}} = \left(1 - R_{\text{pred}}\right) \frac{\Sigma P_{\text{ext}} + \Sigma P_{\text{int-gross}}}{O} \tag{5.13}$$

For lakes where internal loading occurs, Eq. 5.13 shows that back-calculating catchment nutrient loads from lake nutrient concentrations using Eq. 5.4 (e.g. Rutherford and Cooper 2002) can result in overestimated catchment loads due to the inclusion of the (theoretical) gross internal nutrient load.

Rearranging Eq. 5.13 results in

$$\Sigma P_{\text{int-gross}} = \frac{O[P]_{\text{lake}}}{1 - R_{\text{pred}}} - \Sigma P_{\text{ext}} \tag{5.14a}$$

$$= \frac{\Sigma P_{\text{int-net}}}{1 - R_{\text{pred}}} \tag{5.14b}$$

$$= \Sigma P_{\text{ext}} \sum_{n} \left[\left(R_{\text{pred}} - R_{\text{obs}}\right) R_{\text{pred}}{}^{n}\right] \text{ for } n \text{ is } 0, 1, 2, \ldots, \infty - 1, \infty \tag{5.14c}$$

Equations 5.13 and 5.14a–c consider predicted retention to be the same for the external and internal load. As a result, the estimate provided by Eq. 5.14a–c for the gross internal load is always greater by a factor equal to $(1 - R_{\text{pred}})^{-1}$ than the net internal load given by Eq. 5.7. In other words, for an $R_{\text{pred}} = 0.5$, Eq. 5.14a–c would result in a gross internal load double the estimate of the net internal load, and for $R_{\text{pred}} = 0.75$ the gross internal load is fourfold the net internal load. The result is that the estimate of the gross internal load will be higher relative to the estimated net internal load in lakes where the predicted retention is high, typically large deep lakes with long residence times. However, those lakes are more likely to be oligotrophic and to have low rates of internal loading.

Because the method of Eqs. 5.13 and 5.14a–c takes into account that part of the P released from the sediment retained by redeposition in the sediment, Eq. 5.14a–c considers retention of the same P atom to occur multiple times (see Eq. 5.14c), after its first release from the sediment, and again after subsequent release from the sediment, and so on. Therefore, Eq. 5.14a–c represents a cycling loop between P sedimentation and release from the sediment, and gross internal loads can exceed external nutrient loads even where no legacy load build-up in the sediment exists. The approach of Eq. 5.14a–c to estimate the gross internal load, based on the assumption that retention is the same for the gross internal load as for the external load, was first used by Nürnberg (2009) and may result in a reasonable estimate of the actual gross internal load. However, retention of gross internal loads may be dissimilar to retention of external loads because internal loading may be variously dominated by either dissolved (released from sediment, bioavailable and potentially less subject to sedimentation) or particulate (resuspended and potentially more subject to sedimentation) nutrients. Therefore, estimates of gross internal loading by Eqs. 5.13 and 5.14a–c should be treated only as indicative. This is also clearly suggested by the rigid relationship between net and gross internal loads as estimated by these equations, in view of the fact that it only depends on the predicted retention.

5.5.5 Uncertainty in Mass-Balance Estimates

The mass-balance models described above were derived from large empirical datasets of nutrient loads, concentrations, and physiography for tens to hundreds of lakes, with inputs of very different quality. Acknowledging sources of uncertainty is important for useful interpretation of these models (Table 5.4).

Estimates of retention and nutrient concentrations based on generalised relationships across large numbers of lakes should not be assumed to be precise for any single system. Various studies have derived alternative methods for estimating retention (e.g. Vollenweider 1976; OECD 1982; Nürnberg 1984; Brett and Benjamin 2008) resulting in different mass-balance models. These methods can yield differing estimates of retention, which could, therefore, yield a variety of estimates of lake concentrations or net internal loading. Thus, the choice of empirical retention coefficient to use in estimating nutrient budgets for lakes is an important consideration and could affect conclusions drawn concerning nutrient flows and fates.

Nitrogen retention may be less predictable than P retention due to the cumulative effect of multiple first-order processes including N-fixation and denitrification. Furthermore, Harrison et al. (2009) found that the N-settling velocity differed between lakes and reservoirs and also among latitudinal zones. The retention of N and P may be particularly high in lakes in which the availability of either of these nutrients limits phytoplankton growth. In contrast, where either of these nutrients is in excess, relative to demand, retention efficiency may be reduced.

A degree of uncertainty also applies to estimates of catchment nutrient loads and nutrient export from lakes (Johnes 2007) because rarely are fluxes into and out of lakes measured continuously. For example, catchment nutrient loads estimated from areal land use export coefficients might differ substantially from estimates obtained empirically by measurements of stream discharge and concentration. Empirical load

Table 5.4 Some sources of uncertainty in the use of nutrient mass-balance models for estimated nutrient budgets for lakes

Source of uncertainty	Suggested action
1. Statistical error in parameters derived from large empirical datasets	• Account for parameter statistical error
2. Nutrient retention coefficient estimation methodology	• Justify methodology used • Account for assumptions in methodology
3. Complexity of N fluxes in lakes (e.g. denitrification, N-fixation)	• Account for N fluxes and their effects on N retention
4. Variation in nutrient settling velocities	• Account for variation in particle sizes and flux rates
5. Nutrient status (limitation, saturation)	• Account for the nutrient status of the lake and its likely effect on N and/or P retention
6. Lake-catchment system is non-steady state	• Verify that the residence time of the lake is similar to the time scale of the mass-balance model • Verify that catchment nutrient loading rates have not undergone recent substantial changes • Verify that surface and groundwater nutrient fluxes are not subject to excessive time lags
7. Catchment nutrient load estimates	• Account for potential underestimation of catchment nutrient loads (e.g. lack of measurement of fluxes during flood flows, etc.)
8. Benthic-pelagic coupling in shallow lakes	• Account for effects of macrophytes, if dominant, and sediment resuspension, if common
9. Mixing regimes	• Use mass-balance models and parameters derived from lakes with the same mixing regime

estimates may also differ depending on the inclusion or exclusion of particulate nutrients exported during episodic stormflows. Errors in estimated catchment loads or lake exports will in turn influence estimates of internal loading and nutrient concentrations derived from mass-balance equations. These aspects should be carefully considered when applying these methods to specific lakes for management or restoration purposes.

Nutrient concentrations in shallow lakes are considered to be difficult to predict using mass-balance models because they often have a large macrophyte biomass (sequestering and cycling nutrients) and are often subject to wind-induced sediment resuspension (Brylinsky 2004). Disturbance and resuspension typically result in higher nutrient concentrations than predicted by mass-balance models. In addition, the mass-balance models may work less well in lakes with residence times <1 year (OECD 1982). In lakes with short residence times, uncertainty in estimates of nutrient retention increases, reducing the accuracy of the prediction of lake nutrient concentrations. Finally, because of the relatively long growing season at the latitude of New Zealand, it is also possible that nutrient retention capacity in New Zealand's warm, monomictic lakes is fundamentally different to that of dimictic, Northern Hemisphere lakes from which many of the mass-balance relationships and their parameters were derived. At the same time, many of the fundamental concepts of mass-balance models have been shown to hold across a wide diversity of lake types (Feature Box 5.2).

Box 5.2 Nutrient Mass Balance in Impoundments in Temperate and Monsoon-Dominated Regions

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Limnologists generally accept that lake chemistry, and consequently lake productivity, is determined by external loading of plant nutrients from the catchment and air shed as modified by morphology, hydrology, and internal processes. Understanding and quantification of this paradigm has roots in the detailed study of abrupt changes in the nutrient income of Lake Washington (USA, Edmondson 1961) and the cross-system, mass-balance models initiated by Vollenweider (1975). This rich body of research assumed steady-state conditions and largely focused on processes in temperate, glacial lakes. The outcome resulted in restoration measures to reduce nutrient-related problems and improve the ecological status of treated lakes (Cooke et al. 2005). Case studies show long-term recovery was most successful with drastic nutrient reductions achieved by diverting or treating point-source municipal sewage inputs. In contrast, non-point controls based on watershed best management practices, which rely on physical or biological processes to achieve nutrient reduction, have been inadequate to achieve measurable lake improvements (Osgood 2017).

In many areas of the world, constructed impoundments deliver ecological services similar to those provided by natural lakes. Reservoir water storage currently exceeds seven times the volume of global rivers. Impoundments are typically located at the lower end of valleys to ensure stable water levels, which results in greater annual water flow than most natural lakes. Despite differences in hydrology, mass-balance processes determine nutrient levels similarly in both lake types (Canfield and Bachmann 1981). Inter-annual difference in water flow accounts for the extreme temporal variation in reservoir nutrients measured in some impoundments. For example, long-term data from Lake of the Ozarks (USA, Fig. 5.5) show nearly a fivefold range in summer total phosphorus values over 35 years and that outflow (which matches inflow) from the reservoir accounts for some 82% of the variation. After a prolonged drought, summer TP in a large US reservoir averaged 18 μg/L, which was followed by a period of extreme rains and a summer TP of 163 μg/L (Knowlton and Jones 1995). These extremes, in sequential seasons in an individual

(continued)

Box 5.2 (continued)
reservoir, closely match the range of TP values in reservoirs throughout the region.

A summer monsoon dominates the annual hydrologic cycle, and consequently the nutrient budget, of many Asian reservoirs (Jones et al. 2009). Available data from moderately productive Asian lakes and reservoirs show nutrients peak during monsoon inflow, which is consistent with theory that non-point-source inputs determine external loading and in-lake conditions. In contrast, values measurably decreased during the monsoon in nutrient-rich reservoirs, suggesting municipal point-source inputs provide the principal nutrient load during the non-monsoon period. In these reservoirs, monsoon flow from the watershed dilutes in-reservoir nutrient concentrations rather than being a major nutrient source. The mechanisms of nutrient mass balance differ between these extreme examples but are consistent with concepts underpinning limnological models.

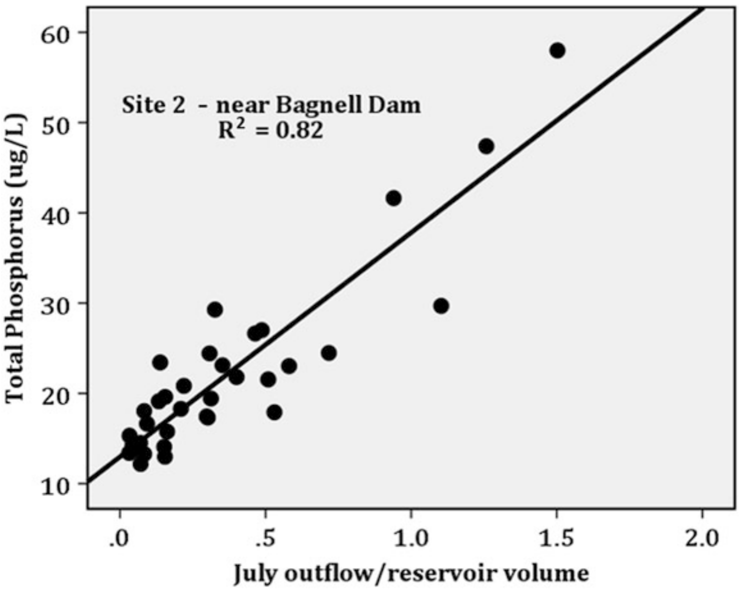

Fig. 5.5 Average summer total phosphorus values (μg/L) measured during 1980–2014 at a site near the dam of Lake of the Ozarks, Missouri (USA). Values averaged 20 μg/L (range 12–58 μg/L). Regression analysis suggests water flow accounts for 82% of inter-annual variation in summer TP

5.6 Future Prospects

The type of simple, steady-state nutrient mass-balance model described here focuses on annual nutrient budgets and is not appropriate to examine seasonal or episodic aspects such as the importance of seasonal (gross) internal loads or sequestration into macrophyte biomass. These models have been used to understand and manage lake eutrophication since the late 1960s (e.g., Vollenweider 1968; Canfield and Bachmann 1981; Nürnberg 1984; Brett and Benjamin 2008), and their use in many applications is undeniable. However, more work remains to be done to refine and improve and extend the use of such models.

A key area of refinement relates to the estimation of external nutrient influxes and in-lake nutrient retention. Catchment input loads used in nutrient budgets are often modelled (see Chap. 2) but may not be accurately estimated on appropriate time scales. Measured loads often suffer from inadequate sampling frequency. Nutrient retention coefficients are usually estimated from statistical models and are not measured in specific lakes. So refinements to input data and model parameters should improve the accuracy of most nutrient budgets.

Basic mass-balance models usually focus on N and/or P and in some lakes other factors may drive or limit phytoplankton productivity and biomass, including factors such as light availability in the mixed layer, grazing, temperature, and micronutrient availability (Schallenberg 2004; Downs et al. 2008). While these factors usually exert less influence on phytoplankton growth and biomass than N and P concentrations, in some lakes such factors may control phytoplankton biomass and productivity to the extent that annual average biomass is affected. Modelling approaches could benefit from taking such factors into account, which may lead to a refinement in model parameters for certain types of lakes such as shallow lakes, humic stained lakes, and lakes affected by glacial sediment.

The simple models outlined here are generally most applicable to deeper monomictic or dimictic lakes. They do not specifically account for temperature, mixing regime or other climate factors. Thus, impacts of climate and climate change on lakes (e.g. Hamilton et al. 2013) are not accounted for in the basic nutrient budget approach. Further refinements to these models could explicitly account for different mixing regimes as well as the influence of climate in general on lake nutrient budgets.

Simple mass-balance models such as those described in this chapter are attractive as lake management and restoration tools, partly because of their simplicity. Until now, there has been little effort to explicitly define sensitivities and uncertainties when employing such models. Given the assumptions and limitations of simple, steady-state mass-balance models, responsible use of such models for lake restoration and management should explicitly represent errors and uncertainties in the model predictions and outputs (see Loucks and van Been 2005).

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Chapter 6
Physical Processes for In-Lake Restoration: Destratification and Mixing

Max M. Gibbs and Clive Howard-Williams

Abstract Stratification in lakes is a natural phenomenon caused by solar heating resulting in development of a thermocline. This becomes a barrier to heat, dissolved oxygen and dissolved nutrient transfer between the upper and lower water column. Nutrient runoff from land and phytoplankton growth in lakes can cause bottom waters to become oxygen depleted and potentially unsuitable as a habitat for aquatic biota. Conversely, reduced depth of mixing above the thermocline provides a high light field that enhances algal and cyanobacteria growth and, in nutrient rich conditions, the lake becomes degraded, the natural lake eutrophication process. Reducing the nutrient load on the lake may not be possible and alternative actions are required such as minimising or removing the magnifying effects of thermal stratification on the eutrophication process. This chapter describes issues associated with thermal stratification in lakes and examines the possible options for removing or preventing thermal stratification by mixing as a management strategy for the rehabilitation of degraded lakes. The most common mixing device is aeration, using bubble plumes to induce vertical movement of the water column. Sparge line aeration systems can be designed to suit most water bodies from small ponds to large reservoirs and natural lakes, with the maximum size being determined by economics: a generic design is described in detail. Also described are alternative water column mixing systems and the option of bottom water re-oxygenation without destratification. Discussions are based around strategies adopted for the ten water supply reservoirs for Auckland City (New Zealand), used as a case study.

Keywords Thermal stratification · Destratification · Lake mixing devices · Bubble plumes · Hypolimnetic oxygenation

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6.1 Introduction

Restoration of lakes involves multiple processes, many of them biological, but there are several engineering interventions that have been successfully used, such as diversions of eutrophic water sources away from less eutrophic lakes as was done with the Ohau Channel diversion wall between Lake Rotorua and Lake Rotoiti (BOPRC 2008). This chapter discusses two groups of engineering options that influence the in-lake processes of stratification and oxygen depletion. Alterations to these processes have been shown to be successful for lake management and restoration.

6.1.1 Thermal Stratification

Stratification in a lake water body is caused by density differences between water in the upper and lower water column (Fig. 6.1a). Thermal stratification is the process whereby such differences are due to temperature (rather than salinity) effects and results in the separation of the lake water column into three layers definable by the temperature-induced density differences: the epilimnion at the top is the warmest, the metalimnion in the middle is cooler than the epilimnion and the hypolimnion at the bottom is the coldest. The thermocline within the metalimnion is the zone is where the largest temperature change occurs with change in depth (Fig. 6.1b). As the temperature gradient across the thermocline increases, it becomes a barrier to the rapid mixing of heat and oxygen down to the hypolimnion and transporting nutrients

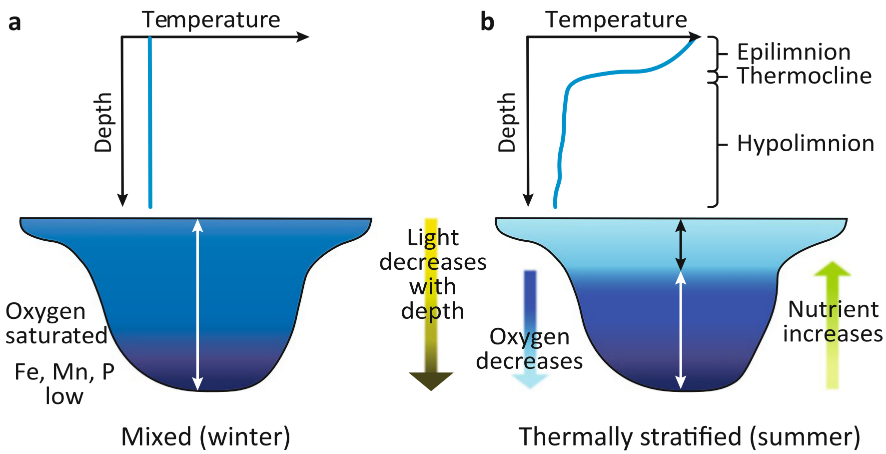

Fig. 6.1 Schematic diagram of a lake showing the concept of (**a**) fully mixed and (**b**) thermally stratified lake (redrawn from Gibbs and Hickey 2012)

released from the sediment up to the epilimnion. Eventually, the thermocline reduces these flux rates to the rate of diffusion.

In New Zealand's warm-temperate climate, thermal stratification is a seasonal phenomenon with most lakes being fully mixed in winter and thermally stratified in summer. Such lakes are termed monomictic as they only stratify and then mix once per year. In countries of the Northern Hemisphere, lakes may also become ice covered in winter and so stratify and mix twice in a year (dimictic), a process that occurs in a very few high altitude South Island lakes. In this chapter, we only consider monomictic lakes.

6.1.2 Consequences of Thermal Stratification

6.1.2.1 Deoxygenation

Decomposition processes in the hypolimnion of the lake water column and sediments consume dissolved oxygen (DO). This process is referred to as hypolimnetic oxygen demand (HOD). HOD comprises oxygen loss from the water column plus sediment oxygen demand (SOD). If the loss of oxygen from the water column is greater than the rate of oxygen diffusion across the thermocline, the hypolimnion begins to become oxygen depleted. As the DO concentration reduces, electrochemical reduction and oxidation (REDOX) potentials in the water column reduce from positive to negative with the zero point coinciding with zero DO, a condition called anoxic. In this anoxic state, there is no free oxygen in the water column but there are oxygen-containing anions such as nitrate (NO_3^-) and sulphate (SO_4^{2-}), and oxygen in the organic molecules (e.g. sugars and fatty acids) in organic detritus settling out of the upper water column. As the REDOX potential becomes increasingly negative, bacteria in the sediment strip the oxygen from these molecules releasing ammonium (NH_4+), hydrogen sulphide (H_2S) and methane (CH_4). When there are no more oxygen atoms available in the system, the condition is termed anaerobic. In the process of becoming oxygen depleted, there are several threshold DO concentrations where biological and geochemical changes occur. In general, the presence of most fish species and other non-microbial aquatic biota declines markedly when the DO concentration falls below 5 mg L^{-1}.

6.1.2.2 Nutrient Release

At a DO concentration of about 5 mg L^{-1}, manganese changes from insoluble manganic (Mn^{3+}) form to soluble manganous (Mn^{2+}) form and at an approximate DO concentration of <2 mg L^{-1} iron changes from insoluble ferric (Fe^{3+}) form to soluble ferrous (Fe^{2+}) form. Both iron and manganese can sequester (adsorb) soluble reactive phosphorus (SRP) (mostly PO_4^{3-}) from the water in their insoluble forms, but release it when they dissolve under reduced oxygen concentrations (Fig. 6.2). This process is

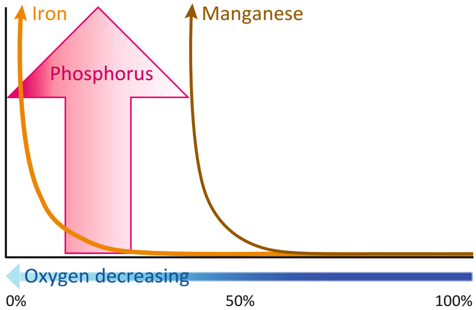

Fig. 6.2 Biogeochemical processes in a lake that are directly controlled by oxygen. Upward arrows indicate increasing concentration of the soluble form while their horizontal position indicates the level of oxygen required for these transitions (redrawn from Gibbs and Hickey 2012)

reversible, with SRP being sequestered or released depending on the DO concentration in the water column. The sequestration of SRP from the water column is not instantaneous, and there is a kinetic reaction time which allows a short window of time (hours to days, depending on algal abundance) when there is SRP free in the water column after the iron and manganese oxides have precipitated to the lake bed.

Decomposition processes in the sediment render the sediment anaerobic from a few mm below the sediment surface. This means the Fe and Mn in the sediment are in their soluble forms. However, as these metals diffuse out of the sediment, they may encounter oxygen in the overlying water and precipitate as their insoluble forms at the sediment surface. Decomposition processes also release dissolved inorganic nitrogen (DIN) from the sediment in the form of ammonium-N (NH_4-N). The NH_4-N is continuously released from the sediment at all oxygen levels and it diffuses up out of the sediment. In the process, it must pass through the surface layer of sediment where, if the overlying water is aerobic, nitrifying bacteria can oxidise it to nitrate-N (NO_3-N). If the water column is anoxic or anaerobic, nitrification does not occur and the water column accumulates elevated concentrations of NH_4-N with no NO_3-N. As oxygen levels in the overlying water increase from the anoxic state, the bacterially driven nitrification rate increases producing NO_3-N, and both NH_4-N and NO_3-N can be present in the water column. At the point that nitrification rates exceed the rate of NH_4-N release from the sediment, only NO_3-N remains in the water column. If the NO_3-N being produced is still within the surface sediment layer, denitrifying bacteria can reduce it to nitrogen gas (N_2), which is lost from the water to the atmosphere. There is no similar loss mechanism for phosphorus (P).

6.1.2.3 Light Penetration

The release of SRP, NH_4-N and NO_3-N into the lake water column from both decomposition in the hypolimnion and in the sediments provides these nutrients for algal (phytoplankton) growth. The rate of phytoplankton growth at a specific depth in the water column depends, amongst other things, on the species, the water temperature and the depth of light penetration. The upper water column between the surface and the depth at which the light level is 1% of the surface illumination is called the euphotic zone. In this zone, there is sufficient light for photosynthesis and hence phytoplankton growth. With thermal stratification in summer, phytoplankton can circulate in the high light environment of the surface epilimnion and mixing may not carry them into low light conditions (i.e. Fig. 6.1). This positively influences growth rates. Thermal stratification, therefore, produces conditions that can enhance phytoplankton growth by constraining growth to the upper water column where they experience longer periods in the euphotic zone. The euphotic depth (1% light level) need not coincide with the thermocline. In high water quality lakes, the euphotic depth may be deeper than the thermocline while in degraded lakes it may be shallower. However, if there is no stratification, or the lake has been destratified, wind-induced mixing of the water column will generate vertical mixing currents that may carry phytoplankton cells through the depth of the water column (i.e. Fig. 6.1a condition). During that deep mixing cycle, they may experience considerable periods of time in low light conditions, and the growth rates will be reduced (Gibbs and Hickey 2012).

The length of time spent in the euphotic zone determines the rate at which phytoplankton biomass increases while the time spent in the dark determines the loss of phytoplankton biomass through respiration, which may ultimately lead to cell death. Zooplankton grazing also results in phytoplankton biomass reduction. The theoretical depth at which phytoplankton growth in the euphotic zone is matched by respiratory losses of phytoplankton biomass in the dark is known as the ‘critical depth’ (Sverdrup 1953). Phytoplankton biomass declines when the critical depth is exceeded. Artificially mixing the water column in stratified lakes by a process of ‘destratification’ can produce conditions where the critical depth is exceeded. This technique can, therefore, be used to limit algal growth and even control nuisance cyanobacteria blooms (Visser et al. 2015).

6.2 Lake Mixing

6.2.1 Natural Processes

Natural mixing of lakes occurs regularly in shallow lakes by strong winds. However, wind energy may generally not be strong enough to cause full depth mixing of deeper lakes and reservoirs during summer stratification (Lossow et al. 1998). This is because energy transfer from a surface wind wave down into a lake is through orbital rotation currents produced by the wave. The orbital velocities generally decrease with increasing depth and, in lakes of small to medium depth, and may only extend to a depth equivalent to *ca.* nine times the amplitude of the surface wave (Kumagai 1988). This effect is an important driver of thermal stratification in summer when wind speeds tend to be lower than at other times of year. In autumn, the cooling surface water will reduce the temperature gradient across the thermocline making it easier for wind-induced currents to mix the lake to the bottom (Gibbs et al. 2016). This event is called ‘turn-over’.

The potential for wind mixing may be assessed by using the Osgood index (US Department of Agriculture 1999) or the Wedderburn number (Wedderburn 1912). The Osgood Index is defined as the mean depth (z) of a waterbody in meters divided by the square root of the surface area (A) in km^2, or $z/A^{0.5}$. It reflects the degree to which a lake or reservoir will mix because of forces of wind. Low numbers indicate a shallow, large lake that is readily mixed by wind, although during a period of calm days, it may become temporarily stratified. The Wedderburn number W gives an indication of how likely the lake is to mix and can be interpreted by:

- $W \geq (L/4)h_e$—Strong thermal stratification, little mixing, small internal seiche amplitudes.
- $\frac{1}{2} < W < (L/4)h_e$—Wind-induced mixing stronger that thermal stratification, more surface mixing than instability at the thermocline, large internal seiche amplitudes.
- $h_e/L < W \leq \frac{1}{2}$—High degree of mixing between the epilimnion and the hypolimnion, much upwelling at the thermocline (unstable) surface at the upwind end of the basin.
- $W \leq h_e/L$—complete overturn (mixing).

Where h_e is the epilimnion thickness at rest condition and L is the length of the lake basin.

Partial mixing of a thermally stratified lake will transfer nutrients from the hypolimnion into the epilimnion where they can support phytoplankton growth and in some cases even result in autumnal blooms. Full depth mixing in autumn disperses nutrients from the hypolimnion throughout the lake. These nutrients support phytoplankton growth and typically result in a spring growth that may reach bloom proportions. If the lake is deep and the hypolimnion has become anoxic or anaerobic, full depth mixing at turn-over may result in a drop in DO concentration throughout the lake water column, reflecting the volumetric mixing ratio of the hypolimnion/epilimnion and the conservation of mass of the DO concentration in each layer. This effect may last for a short period until the lake becomes fully re-oxygenated again, but may have adverse effects on aquatic biota that were living in the epilimnion.

6.2.2 *Artificial Processes*

The process of artificially mixing a lake is called destratification (Feature Box 6.1). The main reasons for artificially mixing a lake or reservoir are to raise the DO concentration in the bottom water and to improve water quality by preventing the release of Fe, Mn and SRP from the sediment. Therefore, the term aeration is used together with artificial mixing as the two processes act in concert. Achievement of aeration to the bottom of a lake requires full depth mixing of the water column to raise the DO concentration at the sediment–water interface above 5 mg L^{-1}. Artificial mixing is achieved by establishing a circulation current that draws surface oxygenated water down to the bottom of the lake to replace bottom water raised to the surface by the mixing device. DO in the downward-circulation current will raise the DO concentration in the bottom water. This level of mixing may also allow nitrification of any released NH_4-N to NO_3-N at the sediment–water interface, thereby reducing NH_4-N and DIN concentrations in the lake.

Artificial mixing to increase lake bottom water DO concentrations has the added advantage of increasing the aerobic habitat for fish and benthic biota. It may, in some cases, also have a potentially negative effect in raising the bottom water temperature thereby reducing cold water refuges for fish such as trout, forcing them to move into the colder inflow streams.

A further common reason for mixing a lake or reservoir is to disperse or prevent algal (and particularly cyanobacterial) blooms developing (Visser et al. 2015) by making use of the critical depth factor (see Sect. 6.1.2.3). Consequently, in a eutrophic lake or reservoir, full depth mixing may result in a change in the phytoplankton species composition from buoyant cyanobacteria, which can form surface blooms and scums in calm conditions, to diatoms and chlorophytes, which require turbulence to keep them suspended in the water column.

There is likely to be a financial constraint to the size of lake or reservoir that can be reasonably managed using this turbulent mixing technique. There are also issues associated with the lake water column structure to be considered when the mixing device is first turned on. These issues are described later in this chapter.

Box 6.1 Artificial Destratification

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Artificial destratification is the mechanical manipulation of water circulation within a reservoir with the aim of weakening or eliminating the density stratification of the water column. Density stratification is a nearly ubiquitous feature of reservoirs globally and most commonly is the result of thermal stratification caused by seasonal warming of the upper regions of the water column in response to solar heating. It is quite common in temperate and tropical climates for thermal stratification to commence in early spring as both day length and air temperatures increase and to persist through late autumn.

Left undisturbed, density stratification can suppress vertical transport of heat, dissolved nutrients and dissolved oxygen to rates approaching as little as 1–10 times molecular diffusion levels. Vertical transport can be 10–100 times faster in actively mixing zones of the water column such as the surface layer, where wind mixing and cooling by evaporation and conduction supply mixing energy (turbulent kinetic energy, or TKE) and along the bottom boundaries

(continued)

Box 6.1 (continued)
where velocity shear (a change in velocity with distance from the boundary) introduces TKE.

The suppression of vertical transport arising from density stratification facilitates oxygen depletion of deeper waters as well as the release of nutrients and reduced forms of iron and manganese from anaerobic sediments. The severity of these water quality problems are reservoir-specific and reflect differences in local climate, inflows and outflows and pollutant loading. In addition, density stratification is frequently accompanied by relatively shallow ($<$ 6 m deep) surface mixing layers (SMLs) (just 0.1 °C can be a sufficient temperature change to suppress transport downwards through the bottom of the SML) which provide support for the development of harmful algal blooms of buoyant algae by increasing the amount of light experienced by the algae within the SML. As a general rule, if the SML depth is less than three times the euphotic depth (the depth to which 1% of the light incident at the water surface penetrates), the light environment will be conducive to blue-green algal bloom formation.

Artificial destratification typically uses bubble plumes or mechanical mixers to induce vertical currents in a reservoir. A bubble plume introduced at the bottom of the water column will rise to the surface and lift the surrounding water with it. As the plume rises, it entrains water laterally from the reservoir so that by the time the plume reaches the water surface it contains a mixture of water drawn from nearly the full height of the water column. At the surface, the plume will have a temperature intermediate to that observed at the top and bottom of the water column. The plume spreads radially along the surface for only a few metres before it plunges back down through the water column until it reaches its level of neutral buoyancy, i.e. the level at which the temperature of the reservoir away from the bubble plume matches the temperature of the descending water. At the level of neutral buoyancy, an intrusion forms and the plume spreads horizontally until it reaches the boundaries of the reservoir. In its simplest form, this sets up two counter-rotating circulation gyres with flow moving away from the plumes at the level of the intrusion and moving towards the plume at the top and bottom of the water column.

It is important to remember that away from the bubble plumes themselves, and in the absence of strong winds or cooling, the water column of the reservoir will frequently not be turbulent apart from the boundaries of the intrusion flow generated by the plumes. In other words, the destratifier does not mix the reservoir directly. Instead, by inducing a net downwards movement of the water column away from the plume (to balance the amount of water entrained into the plume), the circulation that is established effectively shifts heat and oxygen vertically downwards at some characteristic velocity ‘*w*’. As the number of plumes increases, w increases along with the rate of

(continued)

Box 6.1 (continued)
reduction of density stratification. Eventually, the bottom temperature might increase enough so that normal nighttime cooling (it is generally penetrative convection and not wind mixing that causes deepening of the SML) or wind mixing is able to overcome the residual stratification and cause full mixing of the reservoir.

In theory, an artificial destratification system can be engineered to eliminate density stratification in almost any reservoir. However, effective destratification rapidly becomes more expensive as the depth and volume of a reservoir increase. When considering the use of artificial destratification to improve water quality, it is crucial to carefully consider what the problem is. Past reviews of artificial destratification in Australia (McAuliffe and Rosich 1989) have generally found that this technique is more effective at managing dissolved oxygen concentrations and reducing nutrient release from sediments and dissolved reduced forms of iron and manganese than it is at preventing the occurrence of blue-green algal blooms. In the first case, the design criteria is to ensure that the downwards transport of oxygen is sufficient to meet the in situ demand (which will increase if the temperature increases) whereas the latter case generally requires production of a deeper SML. Producing a deeper SML is a much more challenging problem, especially for storages $>$ 10–20 GL in volume, because SML dynamics are much more sensitive to local climatic conditions. Fortunately, the physics of artificial destratification is well understood and there are several numerical models that can be employed to simulate the performance of the method for any particular reservoir.

6.2.3 Influence of Lake Size and Shape

The size and shape of the lake or reservoir may determine the type and number of mixing devices required to achieve a specific rehabilitation or water quality goal. Lakes with low shoreline to surface area ratios, i.e. lakes tending towards a circular shape, will generally be easier to mix than lakes with high shoreline to surface area ratios such as elongated drowned river valleys with side arms. Thus, small 'round' ponds ($<$1 ha) may be efficiently mixed with a single centrally located mixing device, such as motor driven pumps and fountains, mechanical paddles, air lift bubble plumes and solar-powered mixing devices. Water depth influences the efficacy of some mixing techniques, some are designed specifically for shallow water and others work best in deep water.

In contrast, larger lakes and lakes with complex basin morphometry, including elongated side arms, may require several mixing devices to achieve a particular level of mixing. There will be an upper limit to the size of the lake that it is possible to artificially mix cost effectively. Consequently, the economic returns of having a recreational facility provided by the lake should also be considered. An example of a large reservoir

(72 m deep, surface area 622 ha or 6.22 km^2 and capacity 139 million m^3) that has successfully been mixed using aeration is El Capitan, a water supply reservoir for San Diego city in California (Fast 1968). This reservoir is elongated and is >3.5 times larger than the largest reservoir in Auckland city's water supply network (Mangatawhiri Reservoir). Part of the economic benefits in El Capitan include recreational amenities such as sailing, water skiing, kayaking, fishing and swimming that offset the aeration costs.

6.2.4 Destratification and Aeration Mechanisms

There are several options for lake mixing (e.g. Singleton and Little 2006a, b) and destratification (Feature Box 6.2). Most of these either use air-lift techniques or propellers to induce vertical mixing currents to the surface or water jets to carry surface water down into the lake. In addition, there are several options for aerating or oxygenating the hypolimnion without mixing and destratification that have also been used. The first option we discuss is the Bubble-Plume Diffuser and its operation in non-stratified and stratified lake conditions.

Box 6.2 Artificial Destratification, Surface Mixers and Hypolimnetic Oxygenation

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The thermal or density stratification that results from differential heating of surface waters can often lead to the deterioration of water quality. As water at the surface is heated, it becomes less dense (more buoyant) and floats on the cooler denser water below. A diurnally stratified lake or reservoir is typically characterised by a surface layer that stratifies during daytime and mixes nocturnally with convection as heat is lost to the atmosphere. A seasonally stratified lake or reservoir typically has a metalimnion that marks the zone of sharp temperature change between the epilimnion (surface water) and the hypolimnion (cooler bottom water) and persists over some months.

The bloom-forming cyanobacteria tend to dominate phytoplankton communities when lakes or reservoirs are stratified as they contain gas vesicles which provide buoyancy and overcome losses from sedimentation. Under these conditions of relative water column stability, algae that have a density greater than water, such as diatoms, will tend to settle out. Buoyant cyanobacteria, however, will float up and concentrate their biomass in the

(continued)

Box 6.2 (continued)
illuminated, near-surface mixed layer. The degree of mixing, therefore, plays a key role in determining phytoplankton competition and succession (Visser et al. 2015).

Density stratification within a lake or reservoir indicates there is no or restricted vertical mixing, limiting gas exchange with the atmosphere. Microbial activity in the sediments, termed the sediment oxygen demand, can utilise the available oxygen, significantly reducing the dissolved oxygen concentrations. This can lead to hypoxic or anoxic conditions in the diurnally stratified lower water layer or seasonal hypolimnion, with redox conditions that promote the flux of nutrients and metals from the bottom sediments. Reduced manganese will be released from sediments when dissolved oxygen concentrations in the overlying water are less than 4 mg L^{-1}. Iron and phosphorus are released from the sediments at still lower oxygen concentrations. Iron and manganese are problematic as they can persist in reduced form during water treatment and cause 'dirty water' issues as they oxidise upon exiting the consumer's tap. Nutrients released from the sediment are a concern as they can increase algal growth, with the cyanobacteria being of particular concern as they produce toxins and compounds that cause taste and odour issues for potable water.

Artificial destratification is a commonly used technique to disrupt or prevent the initiation or persistence of stratification, making the water column of a lake or reservoir more vulnerable to overturn from wind or convection, and increasing gas transfer to the hypolimnion. There are two distinct types of devices used for artificial destratification: bubble-plume aerators and surface mounted mechanical mixers. Bubble-plume aerators disrupt the stratification (Fig. 6.3) and transport gas and heat deeper in the water column. There are many examples of successful bubble-plume aerators. In Chaffey Dam (Australia), the years when the bubble-plume aerator was operated had considerably lower phosphorus concentrations in the hypolimnion than in years when the aerator was not operated (Sherman et al. 2000).

Artificial destratifiers are often deployed to control cyanobacteria. The theory is that phytoplankton will be mixed deeper in the water column and so become light limited. Furthermore, in a fully mixed water column, the advantage of cyanobacteria buoyancy is nullified. It is evident that stratification still occurs outside of the immediate influence of the bubble plume. Visser et al. (1996) demonstrated that *Microcystis aeruginosa* colonies remained positively buoyant close to a bubble plume in Lake Nieuwe Meer (The Netherlands) as the mean light that colonies experienced was low due to deep mixing and photosynthate (sugar) accumulation insufficient to overcome the buoyancy provided by gas vesicles. In contrast, *Microcystis* colonies in the more stratified regions further away from the aerator cells were not fully

(continued)

Box 6.2 (continued)
mixed, floated up and experienced higher mean light near the surface. Artificial mixing to control cyanobacteria is not always successful and should be carefully designed, taking into account key characteristics of the lake—like average and maximum depth—and characteristics of the dominant cyanobacteria—like size and flotation velocity (Visser et al. 2015).

The fact that intense temperature stratification can occur in the surface layers and support cyanobacterial populations, even when bubble-plume aerators are used, prompted trials using surface mounted mechanical mixers. These consist of large impellers that draw water through a draft tube in either an upward or downward flow configuration (see Fig. 6.4). Lewis et al. (2010) measured flow in a downward impeller configuration and showed that a small temperature difference between the surface water and the water at the depth of exit meant that the warmer buoyant water returned rapidly to the surface and mixing with the adjacent water was minimal. While the energy use of these systems is considerably less than bubble-plume aerators, the mixing is inefficient and the surface mounted mixers need considerably more maintenance. Small solar-powered mixers, for example, are ineffective at mixing lakes and have an extremely small zone of influence (Upadhay et al. 2013).

Artificial destratification not only mixes dissolved gas that diffuses into the sediment, but also transfers heat. The warmer temperatures may stimulate microbial activity, which will increase oxygen demand and may also increase the bottom-sediment contaminant fluxes if oxygen is in short supply. Hypolimnetic oxygenation is a possible alternative to artificial destratification as a method to limit metal and nutrient fluxes from sediment. A major benefit is that oxygen is delivered to the bottom waters and diffuses to the bottom sediment, but the lake is not destratified and so a cooler hypolimnion is maintained. Hypolimnetic oxygenation is becoming more widely used and may present a sound alterative to artificial destratification as the climate warms and vertical density stratification increases. It has less potential, however, to specifically remove the physical conditions that promote buoyant colonial cyanobacteria.

6.2.4.1 Bubble-Plume Diffuser

The simplest aerator is the bubble-plume system where compressed air is blown through pipes to one or more diffusers on the lake bed. The design of the diffuser produces a rising column of air bubbles which entrains bottom lake water into the plume and induces a vertical current in the water column (Schladow 1993). The depth of the air outlet is important—the greater the depth, the more efficient the mixing (Cooke et al. 1993).

A common misconception for rising air bubble plumes is that aeration is achieved by the dissolution of the oxygen from the bubbles into the water. In practice, this is

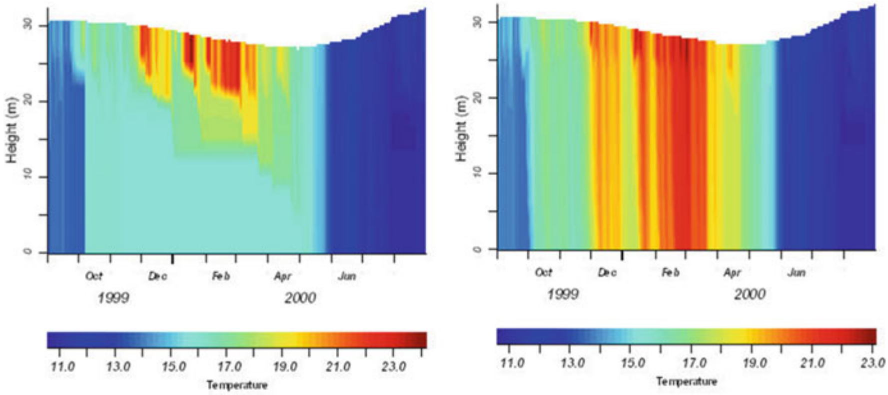

Fig. 6.3 Model simulation of Myponga Reservoir over 12 months using the one-dimensional hydrodynamic model DYRESM. The left panel is the temperature profile under natural conditions and the right panel is the temperature profile using artificial destratification

Fig. 6.4 A surface mounted mixer being deployed in Myponga Reservoir in South Australia

only a small component of oxygen transfer. The major transfer of oxygen into water occurs at the lake surface when the oxygen-depleted water is brought into contact with air. Air bubble plumes are produced by forcing compressed air to the bottom of a lake or reservoir and allowing it to escape from a diffuser. A compressed air flow rate of 9.2 $m^3 \, min^{-1} \, km^{-2}$ lake area should be sufficient to achieve adequate surface re-aeration in most lakes (Lorenzen and Fast 1977).

For some reservoirs, it is common to use an aeration sparge line design where a long diffuser is attached to a weighted cable (ballast rope) deployed in a straight line along the deepest part of the water body and attached to an anchor block at each end. The length of the diffuser depends on the size of the water body and is designed to provide sufficient air lift in the bubble plume to mix the water body (Fig. 6.5).

The diffuser design (sparge pipe, Fig. 6.5) comprises a series of small 0.5–1 mm diameter holes (ports) through the top or sides of an Alkathene pipe attached to the ballast rope by tethers to keep it above the sediment (Fig. 6.6a). Because Alkathene is less dense than water, it will float and the tether length determines how high it will be above the sediment. The spacing between diffuser ports in the sparge pipe should be 0.1–0.2 times the depth of water above the pipe, e.g. in 30 m water depth the spacing would be around 3 m but in practice the maximum spacing is 300–600 mm apart, allowing the formation of a coherent bubble curtain.

In the example shown (Fig. 6.6a, cross section (left) and transverse section (right)), the sparge pipe is tethered about 1 m above the ballast rope. The end of the pipe is sealed to allow compressed air pressure inside the pipe to be forced air through the diffuser ports to form the bubble plume.

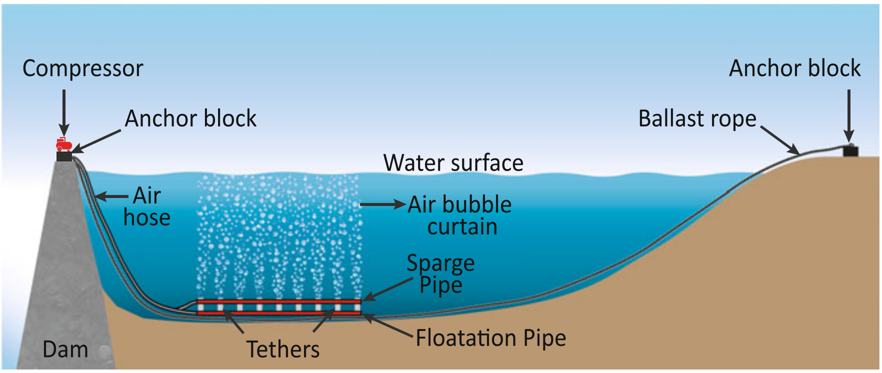

Fig. 6.5 Schematic diagram of a bottom mounted tubular aerator system. The length of the sparge pipe is typically 50–100 m and is positioned in the deepest part of the reservoir near the dam. The ballast rope is anchored at both ends and orientated along the deep axis of the reservoir so that it lies on top of the bed. The ballast rope can be 32 mm trawler rope or a similar sized plastic sheathed multi-core steel cable

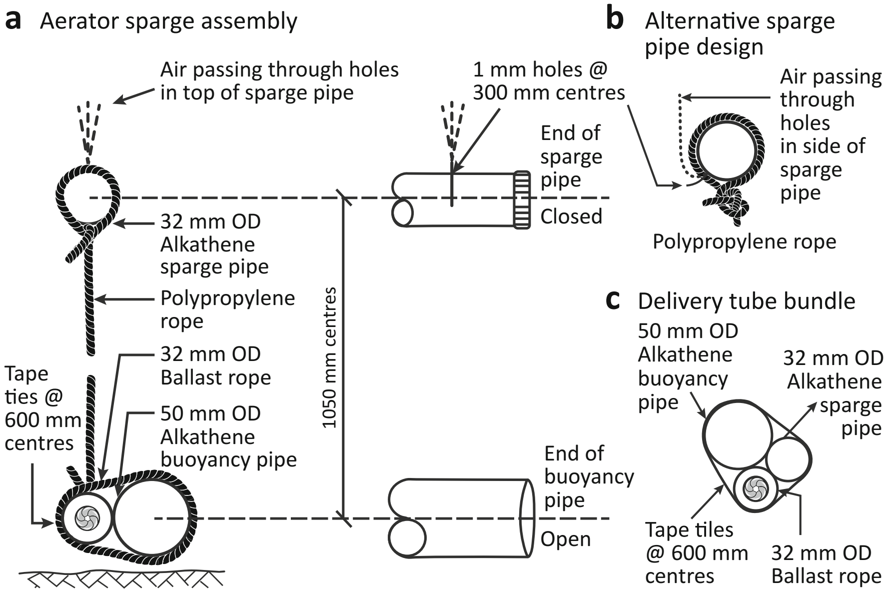

Fig. 6.6 Tubular aerator/sparge system showing (**a**) dual line sparge pipe and ballast rope/buoyancy pipe assembly, (**b**) an alternative sparge pipe configuration with the holes on the side rather than on top, (**c**) delivery tube bundle to the sparge tube

Also attached to the ballast rope directly is a buoyancy pipe which has an open end. In operation, this buoyancy pipe is full of water and is held down by the ballast rope. For servicing and maintenance, air is pumped into the buoyancy pipe and it will lift the ballast rope and sparge pipe to the surface. To allow the aeration system to settle back to the bottom, water is pumped through the buoyancy pipe to flush out the air. The valve manifold used at Opuha Reservoir near Timaru is shown in Fig. 6.7a.

There are several different designs for a sparge pipe diffuser. One alternative is to have the holes through the side of the pipe (Fig. 6.6b), so that there is always additional residual buoyancy due to air trapped in the upper part of the pipe when the system is not in use.

It is not necessary to empty the reservoir to install a sparge line aeration system. With the sparge line system assembled, it can be lain across the surface of the water body and the ends attached to the anchor blocks with sufficient slack to account for the water depth and bottom topography. Then the buoyancy tube is pumped full of water and the ballast rope will carry the assembly to the lake bed. Diver or drop-camera inspection is recommended to allow correct adjustment of the ballast rope length at the non-manifold end.

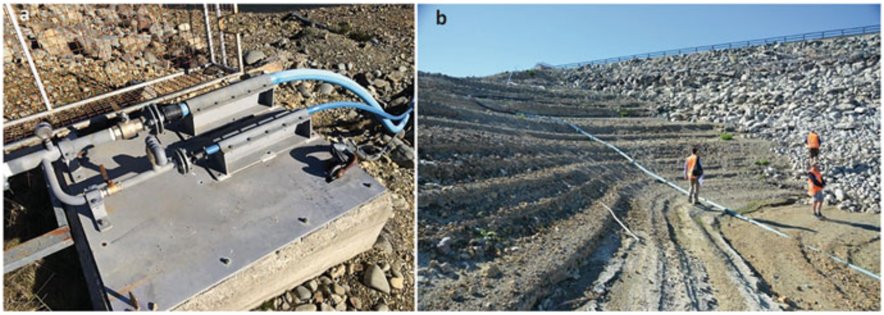

Fig. 6.7 Opuha Dam aerator delivery system. (**a**) Manifold and anchor block, (**b**) setting out the lines to the aerator across the reservoir bed (photos: Tony McCormick, Opuha Water Ltd)

The size or length of the bubble plume required to mix a water body depends on the size of the water body and on the degree and strength of stratification. They all operate by the same principles but non-stratified and stratified lakes are discussed separately below.

6.2.4.2 Bubble-Plume Diffuser in a Non-stratified Water Body

Bubble-plume aerators operate by pumping air through a diffuser (sparge pipe) near the bottom of the reservoir. As the small bubbles rise to the surface, they entrain water and a rising plume of bubbles develops (Fig. 6.8a). As this plume rises to the surface the bubbles expand and entrain more ambient water into the plume and the volume of water in the plume increases. At the surface, the bubbles dissipate into the air and water from the plume spreads across the surface absorbing oxygen from the air before plunging back to a level of equivalent density. If the water column is almost isothermal (uniform temperature at all depths) and hence non-stratified, the water will continue moving horizontally away from the plume until it meets the edge of the lake, a barrier, or it gets caught in the downwards-moving water replacing the water entrained into the bubble plume. This flow pattern is similar to a conveyer belt (Fig. 6.9a) and will maintain vertical mixing in the water body (e.g. Fig. 6.1a). This flow pattern means the orientation of the aerator sparge pipe is important and can be used to focus the energy of the aerator along the axis of the lake or reservoir to best advantage.

The vertical flux (amount of water moved) in the plume is found by solving differential equations describing conservation of flux and momentum for vertical velocity and plume radius (Fig. 6.8b). Once established, the 'conveyer-belt' circulation can be maintained with less energy than was required to establish it, due to the hydraulic inertia of the water in the circulation current. Consequently, it is possible to turn the aerator system off for a period of several days and then back on for at least 2 days. For example, a 4-day-off/2-day-on cycle currently works well in the Auckland water supply reservoirs (observations from NIWA practical advice to Watercare

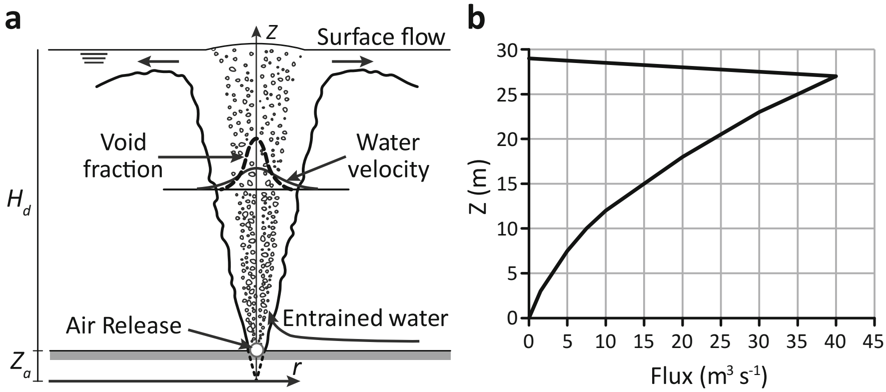

Fig. 6.8 Bubble-plume representation. (**a**) Near isothermal water column, (**b**) estimation of vertical water movement (flux) in m^3 s^{-1} (stylised images redrawn from internet images)

Services Ltd for their reservoir management.) This mode of operation reduces running costs. The timing of this strategy is given in Sect. 6.2.5.

If the water column is non-stratified and relatively well mixed, the time to set up the conveyer-belt circulation pattern may be 5–10 days. It will take longer if the water column is thermally stratified.

6.2.4.3 Bubble-Plume Diffuser in a Stratified Water Body

If the water column is thermally stratified, a detrainment point (Schladow and Fisher 1995) will develop about the thermocline. This results in an intrusion current which will propagate horizontally away from the aerator plume at about the depth of neutral buoyancy generating a separate circulation current in the hypolimnion mirroring the circulation current in the upper water column (Fig. 6.10). As the intrusion moves through the reservoir, there is a return flow above and below the intrusion and these circulation cells may cause mixing between the surface layer and the hypolimnion and reduce the thickness of the thermocline. Given sufficient time and sufficient air, typically 20–30 days, the thermocline may disappear and the reservoir would then become fully mixed. Until full mixing is achieved, this indirect mixing circulation in a stratified lake is less efficient than direct mixing circulation in a non-stratified lake.

6.2.5 Morphology and Timing

The operation of a bubble-plume diffuser in a stratified lake requires consideration of: (a) the morphology of the lake, (b) the amount of air required, and (c) the timing of the aeration process.

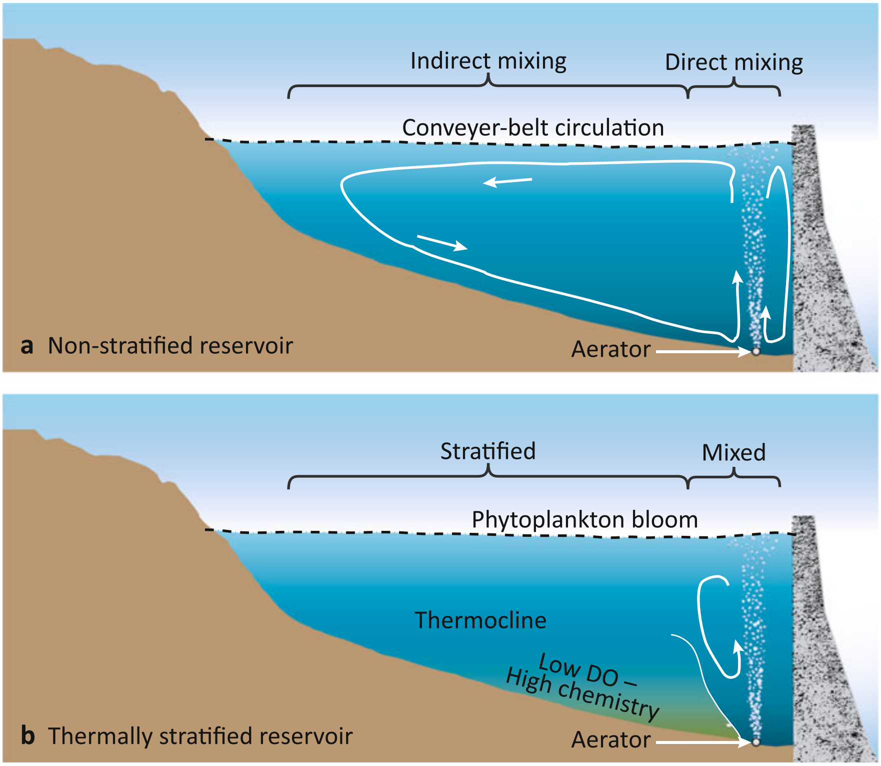

Fig. 6.9 Stylised representation of (**a**) the 'conveyer-belt' circulation pattern in an unstratified reservoir. Aeration occurs mostly in the indirect mixing zone. In (**b**) circulation pattern develops in a thermally stratified reservoir when the aeration system is first turned on. Nutrients from below the thermocline will be entrained by the rising plume and will stimulate algal growth in the upper water column. Because it may take 20–30 days of continuous aeration to mix the water body, there is sufficient time to develop an algal bloom, which may spread throughout the lake (redrawn from Gibbs and Hickey 2012)

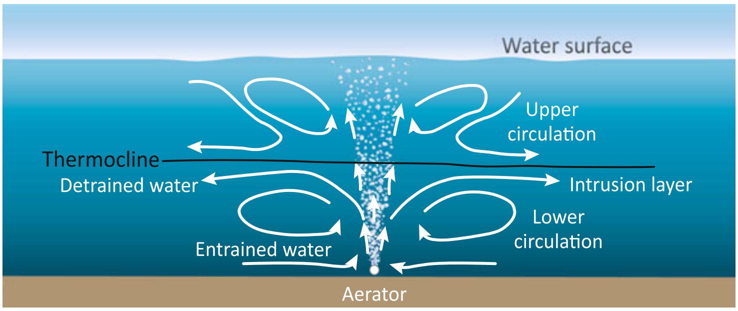

Fig. 6.10 Bubble-plume representation in a thermally stratified water body. Detrained water leaves the bubble plume at the thermocline, reducing the overall efficiency of mixing (stylised image redrawn from Kirke 2000)

(a) The morphology (size and shape) of the lake is important in determining the efficacy of the mixing process in reservoirs and lakes in drowned river valleys. These valleys often have elongated side arms that are not fully or even partially mixed by the circulation currents from the main aeration system. The main consequence of having incomplete mixing or residual thermal stratification in the elongated side arms is that these locations may have smaller wind fetches than the main body of the lake or reservoir. Small fetches allow for calmer conditions that encourage the development of surface-forming algal or cyanobacterial blooms. To mitigate this problem, the side arms may require additional mixing devices installed.

(b) The amount of air required is a function of the size of the waterbody to be mixed. Commercially available bubble-plume aerators delivering around 4.7 L s^{-1}, which is 10 cubic feet per minute (cfm), are suitable for smaller lakes and medium-sized ponds. Using the rule of thumb for calculating compressed air flow rate requirements of 9.2 m^3 min^{-1} km^{-2} lake area (Lorenzen and Fast 1977), the 4.7 L s^{-1} device would mix a water body with a lake area of around 3 ha. A flow rate of 9.2 m^3 min^{-1} is equivalent to 153 L s^{-1}, which is suitable for a lake of 100 ha. The actual air flow rate used in any particular lake will depend on the depth of the lake and the size of the compressor available which can exceed the hydrostatic pressure associated with that depth, i.e. at 40 m depth the hydrostatic pressure is 4 atmospheres or 3.86 bar. As a general rule, a small over capacity is better than not enough air. However, while there is merit in having over capacity, too much air may prevent stable bubble formation resulting in fizzing where the bubbles are too small to have effective lifting power and the mixing will be slower. Installation of a longer sparge line or additional sparge lines is more appropriate for larger lakes and reservoirs, e.g. El Capitan (Fast 1968).

(c) The timing for turning on aeration is important (Chowdhury et al. 2014). Early aeration in a large reservoir, just before stratification establishes, may be a way to develop the conveyer-belt circulation pattern with minimal energy expenditure. However, if the aeration is too early, it may not be cost-effective and aeration that is too late, after the lake has thermally stratified and the hypolimnion has become oxygen depleted, will require considerably more energy than if started at the optimum time. Note that, if aeration is started after nutrients have accumulated in the hypolimnion, these may be dispersed into the upper water column with the unintended consequence of stimulating algal growth and potential for producing cyanobacteria blooms (Fig. 6.9b).

An example of how timing affects a water supply reservoir is shown in Fig. 6.11. A sparge pipe aerator system was installed in the Nihotupu reservoir at the end of 1999 and it was turned on whenever the reservoir showed thermal stratification and hypolimnetic deoxygenation (i.e. the conditions illustrated in Fig. 6.9b).

This mode of operation is referred to as ‘turn-on by calendar date’. In 2006, data from temperature/DO profiles and the timing guidelines (see Sect. 6.2.6) were used to inform aeration turn-on points. Although this reservoir is notorious for developing

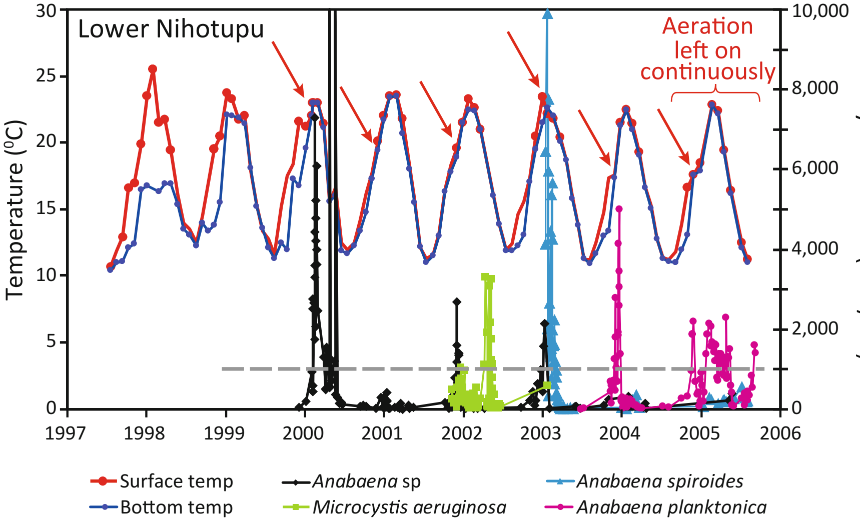

Fig. 6.11 Time-series data from Lower Nihotupu reservoir in Waitakere Ranges, Auckland, showing surface and bottom temperatures, turn-on points for aeration (red arrows) and the cyanobacteria species abundance in cell ml^{-1}. The horizontal line at 1000 cells ml^{-1} represents the drinking water standard for planktonic cyanobacterial cells. From 2005 to 2006, the aeration system was run continuously (data from Watercare Services Ltd.)

algal proliferations, the results were immediate and the cyanobacteria blooms disappeared (Fig. 6.12). The short-term exceptions in 2013 and 2014 occurred when there was a failure of the compressor during a hot calm period.

6.2.6 Aeration Protocols

When using temperature and DO data to control aeration, there is a recommended procedure to ensure aeration is turned on at the appropriate times ensuring cost-effective management (revised from Gibbs and Hickey 2012).

Oxygen concentrations and/or surface-to-bottom temperature differences are used as the triggers for the different stages for activating the aerator through spring and summer. An appropriate procedure is as follows:

1. Temperature low and vertical profile uniform with DO near 100% saturation throughout: *Aerator off; monitor monthly*. (This represents the winter mixed condition.)
2. Temperature higher by >1°C in surface than bottom water with DO in bottom water below 100% saturation but >8 g m^{-3}: *Aerator off; monitor fortnightly*. (The lake is developing a weak thermal stratification.)

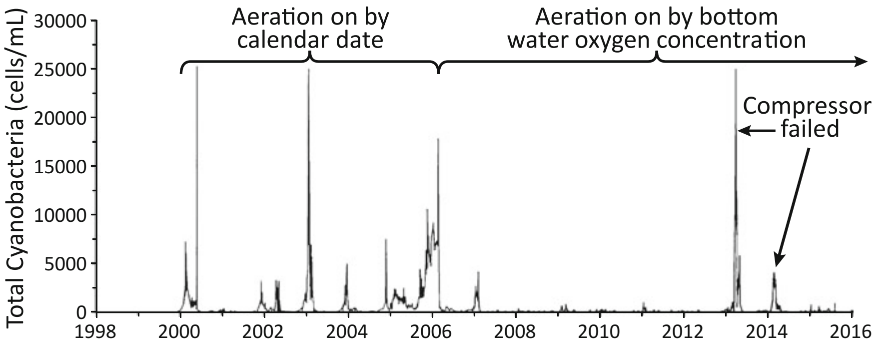

Fig. 6.12 Time-series occurrence of cyanobacteria blooms in Lower Nihotupu reservoir using turn-on by calendar date (before 2006) and information from temperature and DO profiles after 2006. Short-term blooms in 2013 and 2014 were associated with compressor failures (data from Watercare Services Ltd.)

3. Temperature higher by >1°C in surface than bottom water with DO in bottom water between 7 and 8 g m^{-3}: *Aerator off; monitor weekly.* (The lake is thermally stratified and developing bottom water oxygen depletion but DO is above the action aeration threshold.) A caveat for this case is that if the temperature rises to >2°C warmer in surface than bottom water with DO in the bottom water <8 g m^{-3} but above 7 g m^{-3} (i.e. there is strong thermal stratification), then remedial action is required: *Aerator on continuously; monitor weekly.*
4. Temperature higher by >1°C in surface than bottom water, with DO in bottom water <7 g m^{-3}: *Aerator on continuously; monitor weekly.* (With DO > 6 g m^{-3}, the aerator can rapidly mix the water column causing re-aeration to occur and this level is above the minimum of 5 g m^{-3} required for most fish species.)
5. Temperature high and vertical profile uniform with DO concentrations uniform and >7 g m^{-3}: *Aerator cycled at 2 days on and 2 days off; monitor weekly.* (The lake is mixed and aeration is in maintenance mode. The most appropriate cycle can be determined experimentally—if 2-days off does not cause a measurable decrease in the DO concentration and the development of thermal stratification in the water column, the off period can be increased by a day—if it does, then the off period should be reduced by a day. Minimum on period is 2 days and this can be increased if required to maintain a mixed water column and DO concentrations >7 g m^{-3}.)
6. Temperature lower than previous measurement and profile uniform with DO concentrations uniform and >7 g m^{-3}: *Aerator off; monitor weekly for a month then monthly.* (This represents the autumn cooling phase of the annual cycle. The monitoring is continued at weekly intervals for a month after switch-off to ensure that the falling water temperature was not just a transient change in late summer.)

The bottom line is that if DO falls below the threshold of 7 g m^{-3} at any time of year, the aerator should be turned on until the threshold is exceeded again.

6.3 Mechanical Mechanisms

Where incomplete mixing or residual thermal stratification in elongated side arms of lakes and reservoirs occurs, separate and independent mixing devices may need to be installed at one or more locations within these side arms. There are a range of options for these smaller mixing devices, other than bubble-plume diffusers, that can be used to mix smaller lakes and ponds, as well as the side arms of large lakes.

These include mechanical mechanisms such as: low energy mixing impellors that draw bottom water up or forces surface water down through constraining draft tubes; high energy pumps that jet the water down at an angle, entraining air into the jet; and the classic vertical aeration pumps and fast splashing paddles used to aerate waste water treatment plant ponds. These latter are not discussed here.

6.3.1 Solar-Powered Water Mixers

Because lake side arms are sometimes remote and may be difficult to provide power to compressors, there is an opportunity to use solar-powered water mixer (SWM) devices (Upadhay et al. 2013). These devices are cost-effective where the cost of installing electricity to the remote site is greater than the cost of the SWM. The SWM is typically floated on the surface of the lake, with a low energy solar-powered electric motor turning the impellors. There are also mains-powered versions.

A commonly used SWM is the SolarBee® which floats on the lake surface and draws water up through a constraining tube, or 'draft' tube, and discharges it across the surface of the lake. The operating principle is similar to an air-lift bubble plume but without the bubbles. The main difference is that, because the propeller that moves the water is at the surface, a draft tube is required to determine the depth from which the water is drawn up. Typically, the draft tube is adjustable and can be positioned to draw water from almost any depth in the waterbody. Draft tubes range from 0.2 to 0.9 m diameter and have a maximum draw depth of 18–30 m (depending on the model). They are fitted with a base plate that only allows water entry horizontally thereby preventing scouring of the lake bed, if the draft tube intake has been positioned near the lake bed or allowing entry of water from a specific depth in the water column.

In operation, the SolarBee® draws deep water horizontally into the bottom of the draft tube, pulls it up the centre of the draft tube and discharges it across the surface of the reservoir, thereby generating a circulation flow pattern similar to the bubble-plume 'conveyer belt' current. This circulation pattern is claimed to be able to move from 284 to 2272 $m^3\ h^{-1}$ and can slowly destratify a thermally stratified lake with an area of up to 14 ha. The mean water depth and thus the water volume are likely to be crucial design factors.

If the reservoir is thermally stratified, the bottom of the device can be set below the thermocline. The colder bottom water then mixes with warmer surface water as it is discharged and reoxygenated. This results in water of an intermediate temperature and density, which remains in the upper water column at a depth of equal density. By increasing the amount of water above the thermocline in this way, the thermocline is displaced downwards and will eventually reach the lake bed or the draw depth of the SWM device. Consequently, destratification is caused by downwards depression of the thermocline. If the bottom waters are anoxic and nutrient enriched, this option is not recommended.

Mixing processes with SWM devices are slow but efficient as the battery-powered motor runs continuously and the device keeps the circulation pattern operating. It works well in smaller ponds and lakes and should perform well in the confined shallow side arms of reservoirs. In larger ponds and lakes, several SWM devices may be required although a single unit may be sufficient to destratify a narrow side arm.

However, because the SWM is a low energy device, the destratification system using these devices must be designed to have sufficient water transfer to exceed the re-stratifying effects of solar heating. A significant problem with artificial mixing will arise if insufficient energy is used and the waterbody fails to mix within the time taken for natural re-stratification. This could be exacerbated by an inappropriately positioned mixing device that fails to deliver appropriate circulation. For example, if the lake or reservoir has become thermally stratified and the draft tube is set below the thermocline, the low energy SWM device will lift the nutrients accumulating in the hypolimnion and disperse these through the upper water column thereby stimulating algal growth. Alternatively, if the draft tube is set above the thermocline, the mixing currents developed will erode the top of the thermocline. If the velocity and turbulence of the mixing current are great enough compared with the strength and depth of the thermocline, the entire thermocline may eventually be eroded. In contrast, for weak currents only limited erosion of the thermocline would occur, possibly creating even sharper temperature gradients at the base of the mixing layer. Because sharp temperature gradients inhibit vertical turbulence, this would create greater resistance to further mixing (Kortmann et al. 1982). The latter outcome may be more likely for SWM devices.

The thicker the metalimnion, the shallower the gradients of temperature, oxygen and nutrients. This would facilitate the transfer heat and dissolved oxygen down through the water column and nutrients up from the hypolimnion. Conversely, the thinner the metalimnion and the stronger the temperature gradient, the harder it is for these transfer processes to occur, and the greater the resistance to mixing. Note that these gradients may also be affected by the concentration gradient.

6.3.2 Constrained Bubble Plumes

These devices use an air-lift bubble plume instead of the propeller inside the draft tube to induce upwards entrainment of bottom water. They have the same limitations as the SWM but can have an advantage where the compressed air supply provides

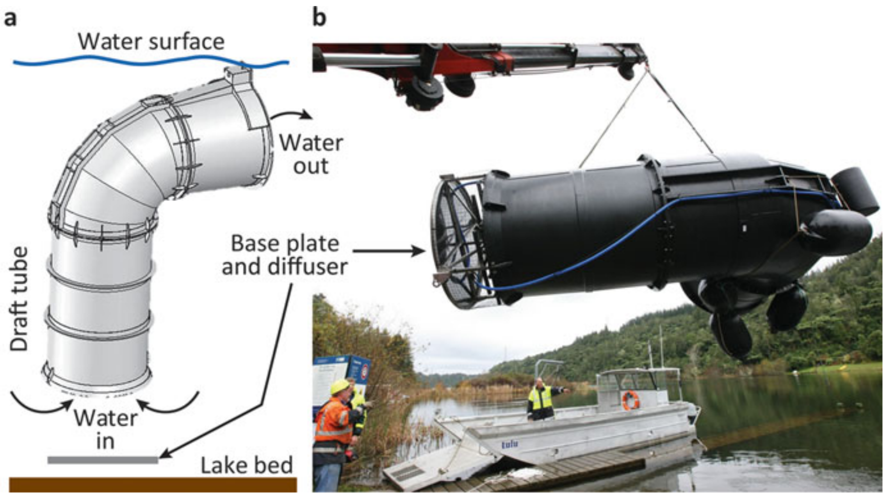

Fig. 6.13 (**a**) Schematic of the draft tube and mixing engine (provided by Hans Burggraaf) showing the relative deployment position below the lake surface and the water flow directions, (**b**) a photo of the draft tube and mixing engine (on its side) being loaded onto a barge for deployment. For size perspective, the black floats are 900 mm diameter mussel buoys (photo by Andy Bruere, Bay of Plenty Regional Council)

greater mixing energy than the SWM propeller. However, in a situation where there is insufficient energy provided by the mixing device to mix the whole waterbody, the re-aerated bottom water may settle on or above the thermocline and move away from the mixing device as an intrusion layer.

An example of this occurred in Lake Rotoehu (an 8 km^2 lake with multiple side arms) where an experimental air-lift bubble-plume design was being tested. The design comprised a large diameter (~3 m) tubular structure (mixing engine) (Fig. 6.13), which constrained the bubble plume causing the bubbles to draw water from below the thermocline in a horizontal current caused by the base plate and discharging them horizontally below the lake surface from the 90-degree bend at the top.

In Lake Rotoehu, three mixing engines were deployed together upright, in a clover leaf pattern around a central axis, with the outflow openings pointing horizontally out into the lake at 120° from each neighbour. This produced a triangular flow pattern across the lake surface for a short distance from the mixing engines.

To determine the efficacy of the mixing engines, Rhodamine WT dye was injected into the bottom of one rising bubble plume and the path of the dye leaving the mixing engine was tracked using a fluorimeter. These results were collated with concentration and contoured to produce a visual representation of the water movement from the mixing engines (Fig. 6.14). The horizontal flow plume was lost at about 100 m from the mixing engine but was still detected by a bottom mounted acoustic doppler current profiler. The results imply that, to mix a large lake such as Lake Rotoehu with this design of mixing engine, there would be a requirement for

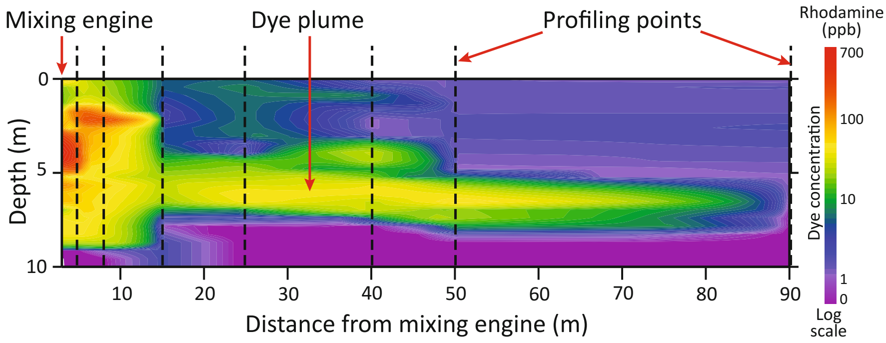

Fig. 6.14 Contour plot of the dye plume from the mixing engine in Lake Rotoehu using Surfer32 by Golden Software. Vertical lines are profiling points. Extrapolation between points used triangulation with a ratio of 12:1. The contour plot is a representation of the likely dye flow path based on the available data (redrawn from McBride et al. 2015)

multiple mixing engines set probably about 200–300 m apart all using the same amount of air. This would be consistent with the use of multiple aerator/oxygenation devices in larger lakes and reservoirs overseas (e.g. Beutel and Horne 1999).

The data from the Lake Rotoehu dye study (Fig. 6.14) indicated that the exit velocity from the mixing engine was insufficient to produce a strong lateral plume away from the mixing engine, and, therefore, there was insufficient mixing with warmer surface water to raise the temperature of the exit plume. Consequently, the exit water plunged within 10–15 m from the mixing engine and was entrained back through the thermocline to be drawn back into the mixing engine again. The vertical profiles through the dye plume showed a sharp cut off at the thermocline with no dye found below the thermocline away from the mixing engine (Fig. 6.14). This is consistent with a sharp thermocline at the time of the experiment. This phenomenon would be less likely to occur in a deeper lake where the added depth would allow the water velocity in the draft tube to increase and produce a stronger plume out of the exit tube.

By way of explanation, as the bubbles expand, they increase in cross-sectional area and volume. An increase in depth from 10 m (1 atmosphere pressure) to 20 m (2 atmospheres pressure) will require double the compressed air pressure to the diffuser to get the same initial bubble plume. This means the bubbles will increase in volume by a factor of four relative to the 10-m level and become even larger as they approach the surface. The original volume of water entering the draft tube remains unchanged and no water can be entrained through the sides of the tube. Because the volume (water plus air) is increasing inside a confining tube, the velocity must increase inside the draft tube towards the surface, i.e. a 20 m draft tube will have a faster exit velocity than a 10 m draft tube, and a 40 m draft tube should be faster still.

6.4 Selective Draw-Induced Stratification

A phenomenon that can be associated with constrained upwards circulation devices is the generation of a sharp artificial 'metalimnion' at the depth of the intake. With a base plate fitted, the draw is horizontal and confined to the gap between the base plate and the draft tube. If the water column is thermally stratified at the level of the draw, then upward movement of denser water below that depth will be suppressed and water entering the tube will tend to come from a horizontal layer at the level of the draw. As this water is removed, lighter, warmer water from higher levels in the water column will drop down to replace it, creating a temperature discontinuity at the level of the draw. The deeper water then becomes isolated and does not mix. That water will have the same characteristics as the hypolimnion of a thermally stratified waterbody.

This phenomenon, called draw-induced stratification (Fig. 6.15), is seen in reservoirs without mixing/destratification devices but where the offtake water is drawn constantly from the same depth (selective withdrawal). Selective withdrawal is commonly used for managing the quality of water discharged from a reservoir (Çalışkan and Elçi 2009).

6.4.1 *Case Study: Upper Huia Reservoir*

While selective draw-induced stratification might seem to be a problem, it can be used to advantage where there is no artificial mixing device available. A management strategy for selective withdrawal was developed on the upper Huia reservoir by Spigel and Ogilvie (1985) and demonstrates this. They examined the effect of prolonged draw from the mid-water column offtake valve over 4 years from 1980

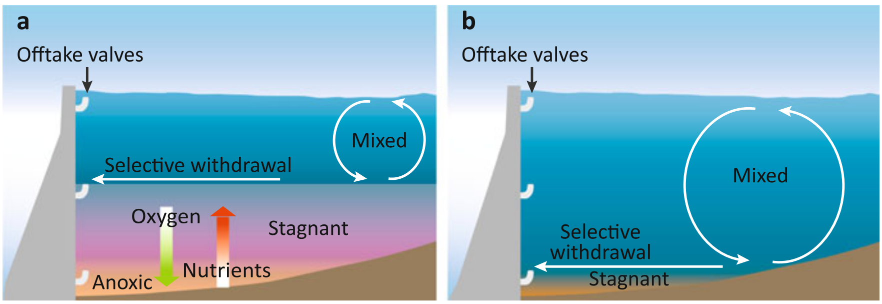

Fig. 6.15 Stylised diagram of a reservoir with three offtake depths. Continuous draw from a mid-depth in (**a**) causes stagnation or dead volume in the water below the draw depth because there is no mechanism for mixing oxygen below this depth. This would also happen if there was continuous draw from the lowest depth in (**b**), but then the dead volume would be minimal (redrawn from Gibbs and Hickey 2012)

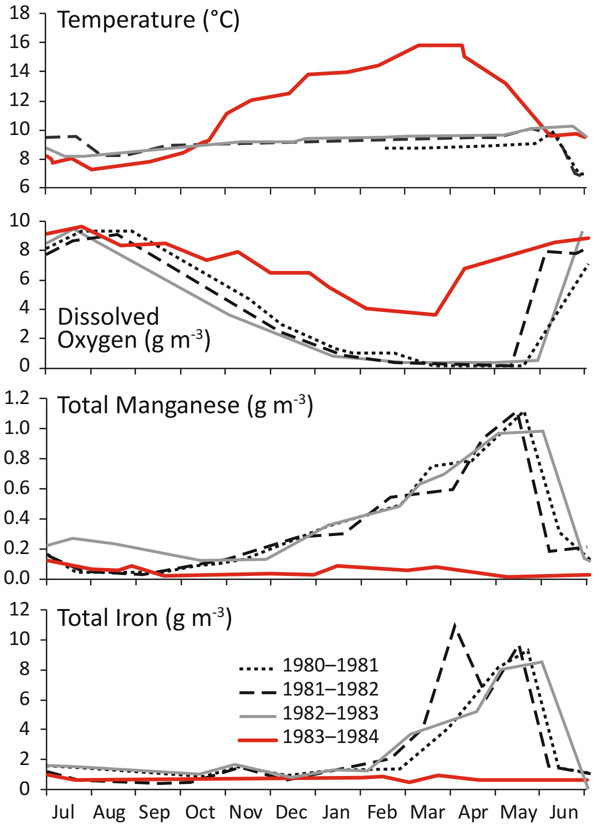

Fig. 6.16 Time series results of bottom water (**a**) temperature, (**b**) dissolved oxygen, (**c**) total manganese (Mn), (**d**) total iron (Fe) in the Upper Huia reservoir from 1980 to 1984. For explanation, see text (graph redrawn from Spigel and Ogilvie 1985)

to 1984. Spigel and Ogilvie (1985) found that continuous selective withdrawal from a fixed depth-induced density stratification at that depth. Essentially, water from below that depth was not drawn while the water above that depth was and became progressively warmer as the lake level fell bringing the warmer surface water closer to the draw depth. The resulting density stratification produced the same geochemical effects as normal thermal stratification, with water below the offtake valve depth becoming anoxic and nutrients and minerals released from the sediments accumulated in high concentrations (Fig. 6.16; years 1980–82).

From 1980 to 1983, the draw depth was the middle offtake valve at a depth of 17 m. The data (years 1980–1982) show slight warming in the bottom water through the summer and autumn as well as oxygen depletion beginning as early August/September (Fig. 6.16). As oxygen depletion progressed to a DO concentration of around 6 g m^{-3}, manganese insoluble oxides began to reduce from manganic form (Mn^{3+}) to manganous form (Mn^{2+}), which is soluble, and Total Manganese concentrations began to rise (Fig. 6.16). As oxygen depletion progressed to a DO concentration of around 2 g m^{-3}, iron in insoluble oxides began to reduce from ferric form

(Fe^{3+}) to ferrous form (Fe^{2+}), which is soluble, and total iron concentrations began to rise (Fig. 6.16). These changes were consistent with the oxygen driven processes shown in Fig. 6.2.

Shifting the draw depth to the bottom offtake valve at a depth of 29 m in 1984 caused the water column above the draw depth to remain oxygenated and eliminated the mineral release from the sediments (Fig. 6.16; years 1983–4). This technique was used until summer 2000 when aerators were installed in the reservoirs. The disadvantage of destratifying and mixing a reservoir is that the whole water column warms to near-surface temperature (Fig. 6.16; years 1983–4 in temperature graph) thereby removing the cold-water refuge required for some aquatic biota and fish as well as the ability of the water supply company to provide a cold water supply to domestic and industrial clients. In addition, the warmer water increases the oxygen demand, and a balance between in-situ demand and supply from the atmosphere is needed. Where these effects pose a serious problem, techniques have been developed for oxygenating the hypolimnion without destratification, a process known as hypolimnetic aeration.

6.5 Oxygenation of the Hypolimnion Without Mixing

Where the cold bottom water layer in a lake or reservoir is important, i.e. needed as a refuge for some fish species or a town cold water supply, special aeration equipment that aerates just the hypolimnion is used (Beutel and Horne 1999). There are two main types of hypolimnetic aeration (Fig. 6.17):

1. The equipment is mounted in the hypolimnion so that it aerates the hypolimnion only but has a small vent tube extending to the surface. The entrained water with dissolved oxygen is dispersed back into the hypolimnion below the thermocline without breaking the thermal stratification. Any bubbles rising from the aerator are directed to the surface through the vent tube (Fig. 6.17).
2. The equipment is mounted at the surface and incorporates a return tube to carry the oxygenated water back down to the hypolimnion before it is dispersed. There are many designs for this 'full-lift' type hypolimnetic aeration system (e.g. Ashley 2000).

In both of these systems, aeration with pure oxygen is more effective than compressed air, which has only 20% oxygen. Using pure oxygen raises the running costs.

In both systems, if they are used in deep lakes, the aerator component needs to be a heavy walled tube to prevent ambient lake water pressure crushing the tube. Air bubbles inside the tube occupy some of the volume thereby reducing water density. Consequently, the pressure inside the tube is less than outside. Placement of the hypolimnetic aerator should be central at the deepest part of the lake. That will allow water to be drawn into the aerator from all directions and be dispersed back into the hypolimnion in all directions.

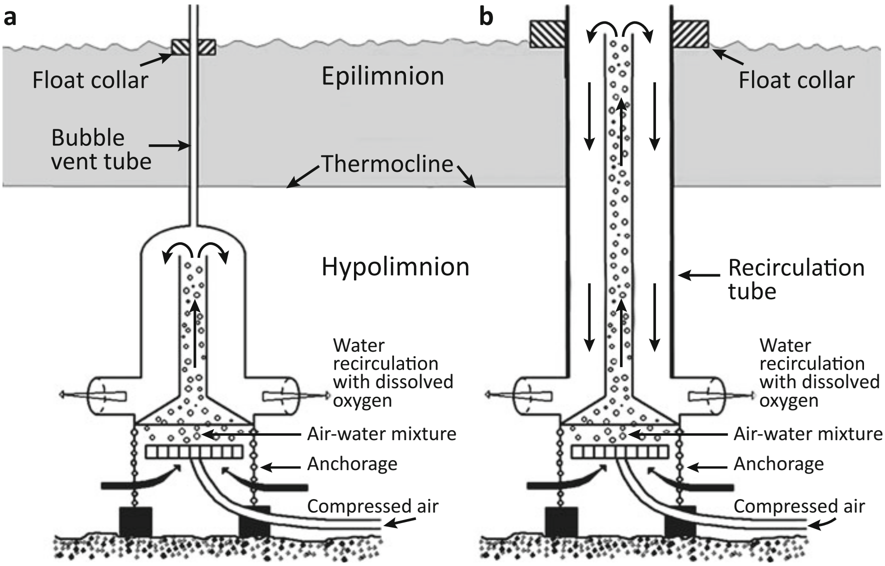

Fig. 6.17 Hypolimnetic aerators (**a**) mounted in the hypolimnion (Martinez 1995) and (**b**) suspended from the surface. This is a full lift aeration system. Both systems do not warm the hypolimnion but re-oxygenate the water (redrawn from Gibbs and Hickey 2012)

Much research has been undertaken on hypolimnetic aeration systems overseas and many devices have been designed (Prepas and Burke 1997; Lindeschmidt and Hamblin 1997; Ashley and Nordin 1999; Singleton and Little 2006a, b; Ashley et al. 2008, 2009, 2014; Barber et al. 2015) and reviewed (Beutel and Horne 1999; Gafsi et al. 2009; Ashley 2000). However, while hypolimnetic aeration works, it is more expensive to operate than most other systems for re-oxygenating the hypolimnion of a lake or reservoir. The major cost is equipment that is needed to confine the air or oxygen and prevent it from destratifying the lake. Few lakes in New Zealand would require this type of mixing device.

The main advantage of hypolimnetic aeration is that, because the hypolimnetic water is returned to the hypolimnion, there is no nutrient enrichment of the epilimnion and the hypolimnion is mixed and oxygenated but is not warmed. This provides a safe, cool refuge for fish species such a salmonids, which prefer temperatures below 22 °C. As most fish species can tolerate DO concentrations down to about 5 g m^{-3}, it is not necessary to fully oxygenate the hypolimnion provided that this level of re-oxygenation is achieved. As stated earlier, a DO concentration of 5 g m^{-3} and above is also sufficient to control the release of minerals (Mn and Fe) and associated P from the sediments (Pakhomova et al. 2007; Hupfer and Lewandowski 2008).

Other hypolimnetic oxygenation system options include injection of micro oxygen bubbles or liquid oxygen directly into the water at the bottom (Beutel and Horne 1999). Applied at the correct rate, pure oxygen dissolves in the water without developing a bubble plume that would destratify the lake. This was the concept behind the original oxygen injection system (Speece et al. 1973).

More recently, a line diffuser system has been developed where pure oxygen is delivered to the hypolimnion through a porous hose (Gerling et al. 2014). The line diffuser system uses side stream supersaturation (SSS) and is designed to deliver large quantities of oxygen with maximum oxygen transfer efficiency through diffuser lines that may be 1–2 km long. The SSS system is claimed to have increased hypolimnetic dissolved oxygen concentrations at a rate of ~1 g m^{-3} $week^{-1}$ without weakening stratification or warming the sediments. A nano-bubble technique is also being developed but has not been published yet.

6.6 Down-Flow Mixing Devices

Most of the devices discussed above for lake water quality improvement involve the vertical movement of water upwards. There are also a range of mixing devices that move water downwards. These include pump systems, which jet the water down through the thermocline (Fast and Lorenzen 1976; Lorenzen and Fast 1977; Gu and Stefan 1990; Cooke et al. 1993; Kirke 2000; Michele and Michele 2002). Such mechanical mixing devices can be small high speed pumps as used in oxidation ponds through to large diameter (>5 m in width), low speed and low energy impeller fans, with or without draft tubes, designed to push warmer surface water down through the thermocline.

These downward mixing devices work well in shallow waterbodies, but their operation works against the intrinsic buoyancy of warm water. Consequently, even though they entrain water in much the same way as a bubble plume, the plume may not extend to the lake bottom, whereas a bottom mounted bubble plume will always entrain hypolimnetic water up to the surface. If used when the lake is strongly stratified, the buoyant plume from a down-flow mixing device will return to the surface close to the intake and will be recycled, together with any nutrients entrained in the returning water. Consequently, these devices are best used in a pre-emptive mode where they maintain the waterbody in a mixed state before it begins to thermally stratify. In this way, they can be used to prevent thermal stratification developing if activated early in spring (e.g. Chowdhury et al. 2014; see also Sects. 6.2.5 and 6.2.6). In practice, these devices are used more to improve water quality by controlling algal blooms than to destratify or aerate a lake.

6.7 Computer Modelling

If the mixing device is too small to mix the whole lake water body then several devices may be needed. The efficacy of a mixing device can be simulated in a lake hydrodynamics model such as DYRESM (Dynamic Reservoir Simulation Model—Centre for inland Waters, University of Western Australia) (Moshfeghi et al. 2005), which will allow an aeration system to be designed for the specific water body. An air bubble destratification sub-model of DYRESM can be used to derive the design specifications of a bubbler system, and DYRESM can be run to predict the effect of bubbler on the reservoir destratification. The design of the aeration system requires information on a number of parameters, which can be provided from routine monitoring programmes or the use of high frequency data collection used in lake monitoring buoys (Read et al. 2011).

6.8 Aeration Case Studies

Aeration can be used in the restoration of degraded lakes by accelerating decomposition processes that reduce the organic content of the sediment. Over time (i.e. several years), continuous aeration during summer has been shown to reduce the sediment oxygen demand and eliminate periods of anoxia that would normally occur during summer stratification without aeration (Grochowska and Gawrońska 2004). Here, we discuss two contrasting New Zealand case studies where aeration has made significant improvements to lake quality: Opuha Dam and Virginia Lake.

6.8.1 Opuha Dam

The Opuha River flows into the Opihi River which is used for irrigation. Because water flows in the Opihi River system are unreliable in summer, local interests combined to construct the Opuha water supply scheme to provide water for urban consumption in Timaru and for farm irrigation. The scheme relies on the fact that the largest water flows into the Opuha River catchment occur during winter and early spring. This water could be stored in a reservoir and used to achieve more reliable monthly environmental flows and to satisfy the high water demand for irrigation in late spring and summer. Although the Opuha Dam and reservoir was originally conceived as an irrigation scheme for water-short areas of South Canterbury, the economics of the scheme as a stand-alone urban water supply/irrigation proposal were marginal so to ensure a more viable scheme, a power station was added, to utilise the energy available in the head of water held by the dam.

6.8.1.1 Background

The Opuha dam is an earth dam (see Fig. 6.7) constructed at the confluence of the North and South Opuha Rivers in South Canterbury. The lake behind the dam is 3.3 km long, has an area of 710 ha, a maximum depth of about 40 m and a storage capacity of 91 million m^3. The off take is at a depth of 30 m, 10 m above the lake bed, discharging water through the power station at a constant 16 m^3 s^{-1}, when operating. The power station is used for peak power generation in the morning and evening and is off during the middle of the day and at night, a total of about 10 h day^{-1}, producing unstable flows in the Opuha River downstream.

About a kilometre below the dam is a weir, which is designed to act as a regulation pond that is used to balance the intermittent discharge flow from the power station. This produces a less variable discharge of water released into the Opuha River throughout the day for irrigation and environmental flow downstream of the dam. All water for irrigation must pass through the power station and is drawn from the Opuha River below the lower weir.

The scheme is subject to constraints, in respect of minimum water flows down the Opuha and Opihi Rivers and the use of water stored behind the dam, by conditions applying to the granting of consents under the Resource Management Act (RMA) 1991. The RMA consents require that the Opuha scheme must be managed to ensure (in the following order of priority):

- A minimum water flow downstream of the dam weir of 1.5 m^3 s^{-1} (to ensure satisfactory conditions for fish life in the Opuha River).
- A minimum specified environmental flow in the Opihi River at Pleasant Point (varying from 3.5 to 8.5 m^3 s^{-1} specified for particular months) which may require the release of water from the dam and which requires the continuous calculation of a river system flow balance for the Te Ngawai, Opihi and Opuha Rivers matched with offtakes for irrigation both upstream and downstream of Pleasant Point and urban water supply for Timaru.
- The rate of filling of the dam is maximised by restricting winter electricity generation to achieving only the minimum flows above, to ensure that it is full at 1 October each year, the start of the drought season.

Only after meeting the consent requirements can Opuha Water Limited manage water flows from the dam.

6.8.1.2 Water Quality Problems

The water quality in the lake initially deteriorated, with high dissolved colour, high nutrient concentrations and no oxygen in the bottom waters in summer (Hawes and Spigel 1999; Meredith 1999). Consequently, water released from the lake in summer was adversely impacting on the downstream river environment. Subsequent formal review of dam consent conditions required installation of continuous monitoring of stratification, deoxygenation and operation of an in lake aeration system. These were operational by 2001, and the degraded water quality conditions have been avoided and effectively managed since.

Excessive nuisance growths of algal mats in the Opuha River below the downstream weir have been a reoccurring problem since the dam was commissioned. These have included prolific growths of toxic cyanobacteria mats and later the introduced algae *Didymosphenia geminata*. These growths have also proliferated in reaches of the Opihi River below the confluence with the Opuha River (Lessard et al. 2013) and result from excessive flow regulation, a lack of 'flushing flows' and an armoured or embedded river bed material. There is on-going examination of mechanisms to address these issues, including elements of redesign of the downstream weir.

6.8.1.3 Design Issues

There are four major design issues with the Opuha Dam:

1. Having a single offtake valve near the bottom of the lake. This restricts the quality of the water being discharged to the worst possible quality in the hypolimnion, when the lake is stratified, without the option of discharging cleaner oxygenated water from higher up the water column (Lessard et al. 2013).
2. It is primarily an irrigation dam, not a hydroelectric reservoir. This means that, although the outlet valve always discharges at 16 $m^3\ s^{-1}$ because it flows through the turbine, it only operates for brief periods of time. Consequently, the deep offtake, which the designers anticipated would act as a deep withdrawal system to entrain the oxygenated surface waters with that flow (see Sect. 6.4), has little effect on the lake water quality. This management strategy is only effective where there is continuous flow. With an area of 710 ha, the short duration flow of 16 $m^3\ s^{-1}$ is unlikely to have a substantial effect on the oxygen levels in the deep water basin near the dam wall.
3. Stratification. Large deep reservoirs such as Opuha Dam are likely to thermally stratify due to solar heating in summer and deoxygenate such that active lake water quality management systems need to be built into the design. Primary consideration should be given to installing an aeration system to keep the water column mixed during summer and thus prevent bottom water anoxia developing. Stratification in a lake is not always driven by solar heating. If the inflow streams are colder than the main body of the lake, they can form density currents that flow down the lake bed and form a cold water layer at the bottom of the lake. This will initially be well-oxygenated water but will rapidly lose that oxygen if the density current has a high biochemical oxygen demand (BOD).

   In Lake Opuha, the two sub-alpine river inflows are almost always colder than the lake and always form density currents. They are also highly turbid after rainfall events and, therefore, have high BOD loads. Because of the location of the offtake, the turbid water overwhelms the offtake and turbid water is discharged from the power station into the downstream environment.
4. Flushing flows. There is no requirement for periodic discharge of flushing flows from the dam to maintain downstream river habitats. Allowing only regulated low flows from the downstream weir eliminates flushing flows and has generated the problems of regulated flow downstream (nuisance algae, armoured bed etc.). Flushing flows 'rumble' the river bed stones, sloughing off periphyton and

redistributing fine sediment rather than letting it consolidate and smother benthic invertebrate habitats between and beneath stones.

6.8.1.4 Mitigation Measures

After initial problems of deoxygenation in the hypolimnion were identified, an aeration system was installed in Opuha Dam (illustrated in Fig. 6.7). Ideally, the offtake system should have multiple valves at different levels to allow management of the water quality being discharged to the downstream environment. This requires a water quality monitoring programme to provide the information needed to allow adaptive management of the discharge water quality. A tower used to raise and lower the control plug on the offtake valve has been fitted with monitoring instruments to provide information for adaptive management of the lake.

A bubble-plume aeration system was installed in Opuha Dam in October 2003 using air supplied with a 11.3 m^3 min^{-1} (400 cfm) compressor. This has the theoretical capacity to mix a lake with a surface area of 1.22 km^2 based on the 9.2 m^3 min^{-1} km^{-2} lake area guideline of Lorenzen and Fast (1977). The design of this aeration system has been described above (Figs. 6.6 and 6.7). Although the engineers have been enthusiastic about the results, they acknowledge that they may have overstated the case and the rate of aeration is slow. Actual data indicates that a period of 14 days aeration increased the DO saturation concentration by 14%.

Overall, operation of the aeration system in Opuha reservoir has been successful, and the incidence of thermal stratification with concomitant anoxic events is rare rather than common. A larger capacity air compressor system is being installed to provide un-interrupted aeration during the summer months, and the aeration mixing protocols are proactive using the monitoring data for management decision making.

6.8.2 Virginia Lake

6.8.2.1 Background

Virginia Lake in Whanganui was known to Māori as Rotokawau and was an important eeling reserve. Kawau, the name of the black shag, refers to the large number of shags that used to frequent the lake. Europeans settled around the lake in the 1850s, and the Borough Council bought the lake in 1874 as a water supply reservoir. A dam and offtake structure controlled the water level in the lake, which was a spring-fed with high quality water. It was fashionable to have a house overlooking the lake, and the native evergreen vegetation was replaced with deciduous European trees. The gardens developed around the lake form a popular recreational park where visitors can picnic and feed the multitude of ducks, geese and swans.

Virginia Lake has an area of 4.5 ha, a volume of 540,000 m^3, a maximum depth of around 20 m and a residence time of around 1.6 years. The outflow is via a surface skimmer drop shaft at the offtake structure and this serves to control the lake level. The lake thermally stratifies at a depth of about 6 m and only mixes briefly due to the shelter provided by the tall trees around the lake shore.

6.8.2.2 Water Quality Problems

Virginia Lake thermally stratifies in winter (Fig. 6.18) and the hypolimnion becomes oxygen depleted to the point of anoxia in spring (Gibbs 2004). Phosphate and ammonium released from the sediments drive the algal blooms in spring. The intensity of the algal bloom reduces the nutrient concentrations in the surface waters as photosynthesis causes oxygen super-saturation and high pH in the epilimnion as CO_2 and bicarbonate are removed from the water. With pH values above 9.2, on occasion, concentrations of ammonia develop to toxic levels (MfE 2014). When the algal bloom is comprised of cyanobacteria, it will drift inshore and the ammonia may result in fish kills. As the cyanobacteria senesce, they may also release toxins.

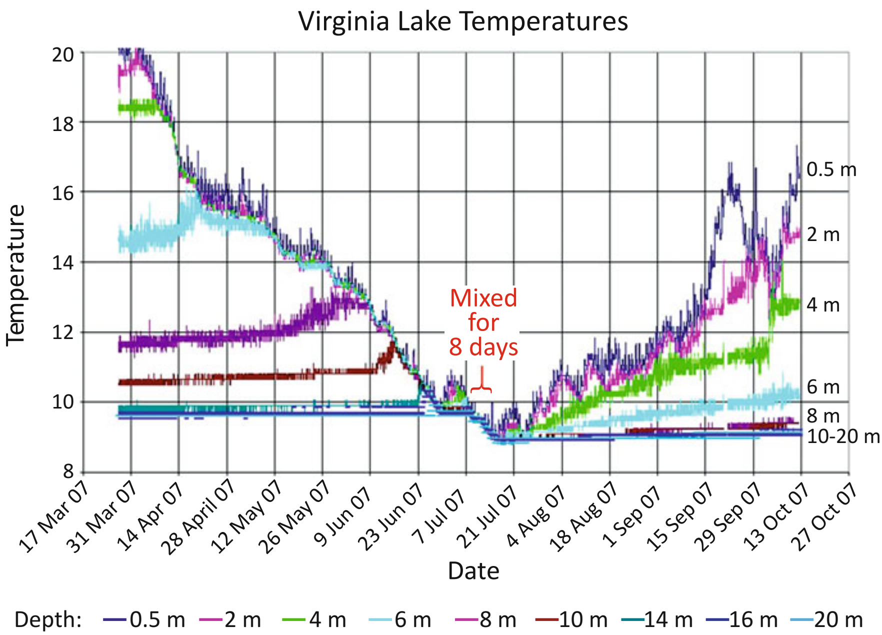

Fig. 6.18 Temperature structure in the water column at the critical winter mixing time. Each line represents the temperature at a specific depth. The water column is mixed when all the temperature lines merge into one. In this 2007 data set, the lake was mixed for 8 days (data from Colin Hovey, Whanganui District Council)

Winter mixing in 2007 occurred just before the 30 July profile indicating some re-oxygenation of the hypolimnion before it became anoxic again by 20 September. A cyanobacteria bloom in September caused DO super-saturation of the surface waters. Photosynthesis by cyanobacteria removed the carbonate and bicarbonate from the upper water column causing the pH to rise above 9.2, a level where ammoniacal nitrogen in the water column is in the form of ammonia (NH_3) which is toxic to aquatic biota including fish.

There are several factors that have combined to cause the water quality problems in Virginia Lake. These include:

- Changing the vegetation from evergreen natives to deciduous exotic species has deposited a large amount of leaf carbon in the lake as a pulse in autumn, which may be one of the drivers of sediment oxygen demand as it decomposes. The very short mixing period in winter may not provide sufficient re-oxygenation of the bottom water to balance this seasonal input.
- Development of housing serviced by septic tanks in close proximity to the lake has contaminated the surface groundwater aquifer with nutrient-rich leachate (e.g. Gibbs 1977a, b) which can enter the lake.
- Introduction of geese and swans to the lake has destroyed the marginal vegetation that would otherwise act as a buffer zone to remove the nutrients from the groundwater. Without the buffer zones shore line erosion is occurring, reducing the clarity of the lake through increased suspended solids.
- The introduction of coarse fish (pest species) will be impacting the regrowth of these buffer zone plants.
- Feeding of large numbers of ducks by the public introduces large amounts of bread, which settles to the sediments either directly or via faecal material from the water fowl and adds to the sediment oxygen demand. It also adds to the nutrient load on the lake. For example, Burns and Singleton (1994) determined that the TP input from ducks represented 34% of the annual external load on 54 ha Lake Rotoroa, Hamilton.
- Draining storm water away from the lake and having a surface outflow has reduced the flushing rate.

6.8.2.3 Design Issues and Mitigation Measures

The lake has a water fountain which is likely to degas the surface water and reduce the super-saturation slightly. A Solar Bee mixer has also been used as a mitigation measure. While this appeared to reduce the extent of the algal bloom, the water column profiles suggest that it may have sharpened the thermocline reducing the diffusive nutrient flux from the hypolimnion.

Restoration action for Virginia Lake could include

- Aeration from near the bottom to keep the lake fully mixed throughout the year. Turn-on would need to coincide with the brief period of winter mixing to reduce the energy otherwise required to break the thermal stratification. At that time, it

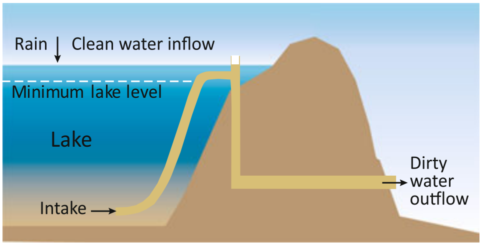

Fig. 6.19 Schematic of a hypolimnetic siphon for Virginia Lake. This uses the existing offtake drop shaft outflow pipes but has an intake at the bottom of the lake. The open head tube sets the water level so that bottom water is drawn continuously. The height of the head tube could be set at the maximum lake level under heavy rain events to prevent flooding

would be unlikely to alter the magnitude of the spring algal bloom that occurs after mixing. By pumping air through the aeration mixer, the water column would have more CO_2 which would tend to reduce the pH below 9, thereby eliminating the production of toxic ammonia. Because the water column would be fully mixed, cyanobacteria would be less likely to become dominant and the magnitude of algal growth could be reduced due to critical depth (see Sect. 6.1.2.3) effects.

- Increase bottom water exchange. In the absence of an aeration system, the outflow could be engineered to include a hypolimnetic siphon (Fig. 6.19) which would discharge the anoxic, nutrient-rich bottom water from the lake instead of the nutrient-depleted aerated surface water. The outflow water would need to be treated to introduce oxygen and reduce the nutrient concentrations that would otherwise impact the downstream environment.
- The diversion/piping of additional clean water into the lake as a surface inflow would enhance the flushing via a hypolimnetic syphon.
- Reducing road runoff would reduce the nutrient load on the lake.
- Reducing the numbers of geese and swans on the lake would allow the riparian buffer zones to re-establish and would reduce the nutrient load from groundwater on the lake. Even though the area now has reticulated waste water, the historical legacy of septic tank effluent will still be travelling in the groundwater to the lake, along with more recent nutrients from lawn and garden fertilisers.
- Educate the public that feeding the ducks so much bread is adversely affecting the lake water quality.
- The sediments could be treated to reduce nutrient release. Sediment capping to retain the P in the sediment has the potential to reduce the abundance of algae and dominance of cyanobacteria due to P-limitation.

Virginia Lake is a classic example of how a lake (natural or artificial) can become degraded through small apparently innocuous changes in the lake catchment. It is also a prime example of where an aeration/destratification system would make an immediate improvement to the lake water quality. The Whanganui District Council observes in their Lake Management Plan (WDC 2009) that 'the water quality of Virginia Lake has been improved since the installation of a Solar Bee lake aeration device in December 2007'. However, 'whilst there has been a considerable reduction in the amount of toxic blue-green algae (cyanobacteria) blooms, rich algae populations are still present in the Lake' (WDC 2009).

6.9 Future Developments

Techniques for destratification and mixing of lakes to manage water quality have improved rapidly over the last few decades as the processes driving deoxygenation and lake degradation become better understood. The rehabilitation potential of traditional and new engineering devices needs to be tempered with the need to meet the requirements of the end user of the water, be it the biota living in the lake or the external users of the lake water for drinking, irrigation or recreation. The bottom line is that, whatever device is used, it must be fit-for-purpose.

Key points to consider will include:

- Size and depth: the mechanical devices need to match the size of the lake and problem. Insufficient mixing power may exacerbate the problem while too much power is a waste of energy and may cause enhanced turbidity. There will be an upper limit to the size of lake that can be mixed economically.
- The end use of the water should be decided as part of the engineering design phase. It may not be necessary to fully re-oxygenate a lake to achieve a water quality that meets the requirement for fish, irrigation or many recreational pursuits. Conversely, full mixing and oxygenation will be required for municipal drinking water supplies.
- Understanding the seasonal cycle and succession of phytoplankton species will enable an aeration system to be activated pro-actively to prevent cyanobacteria growth and proliferation. Getting the timing correct by using feedback from monitoring programmes is a relatively new concept that has proved very effective in medium to large water supply lakes.
- Using multiple mechanical devices to mix larger lakes is a possibility that needs further investigation.
- Combinations of different mechanical devices may be needed to mix lakes with complex morphology, and solar-powered devices with storage batteries will improve with the rapidly advancing technologies in these areas.
- Pre-emptive mixing is likely to be more cost effective than destratification.
- Destratifying a lake to begin the rehabilitation process can have unexpected short-term effects. These should be discussed with a competent lake water quality person before any action is taken.
- Modelling should be used as a tool to check whether the equipment suggested for a specific restoration project is fit-for-purpose, i.e. is it feasible that it will perform as anticipated or could it have unexpected results.

It is likely that future developments in this area for lake restoration will be a closely coordinated combination of engineering and ecological approaches.

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Chapter 7
Flocculants and Sediment Capping for Phosphorus Management

Max M. Gibbs and Chris W. Hickey

Abstract Flocculation and sediment capping are two of the tools in the manager's toolbox for lake restoration. For them to be used effectively requires an understanding of the in-lake processes that control the recycling of nutrients from the legacy stored in the sediments and the interactions with microbial communities and aquatic plants. The form of the nutrient is important, and there are natural in-lake processes that can influence the chemical intervention applied. For example, the chemical control of phosphorus (P) with P-inactivation agents in a lake is only applicable to soluble reactive phosphorus (SRP), i.e. phosphate, and not particulate or organic forms of P. Consequently, particulate forms, including detritus, fine sediment, and algal cells, need to be managed with flocculants which remove them from the water column to the sediments. A range of commercially available flocculation agents as well as passive and active sediment capping agents targeting P are discussed. Future prospects for chemical lake restoration methods are likely to include flocking and locking, i.e. using flocculation to settle the insoluble forms of P then locking that P in the sediments with an active barrier of P-inactivation agent as a sediment cap. Chemical lake restoration methods are not a panacea, but they are an important part of an integrated management plan that includes remedial action in the catchment.

Keywords Flocculation · Sediment capping · Lake restoration · Polyacrylamide · PAM · Phosphorus · Nitrogen · Alumfbox · Phoslock™ · Allophane · Calcite · Aqual-P® · Iron · pH · Alkalinity

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7.1 Introduction

This chapter examines the use of flocculants and sediment capping for phosphorus (P) management from a New Zealand perspective with cross-referencing to overseas research. Some commonly used materials overseas cannot be used in New Zealand because the products have not been registered and permitted for use and, in some cases, there have been issues associated with cultural acceptability of use in lakes owned or co-managed by Māori.

In lake restoration studies, the case has been made for the management of a single nutrient—usually phosphorus (P) rather than nitrogen (N) (e.g. Schindler et al. 2008; Wang and Wang 2009)—instead of reducing both N and P, to drive the system into a nutrient-limited state for phytoplankton growth. The general argument for reducing P is that it is easier to manage P in the aquatic environment than N, even when the lake may be N-limited to phytoplankton growth. However, just removing P from a lake, without regard to N, leaves the potential for algal blooms if P management is not maintained. Consequently, it is the consensus of other researchers in this field that, while reducing P can provide a 'quick fix' in many situations, both N and P should be managed to produce a long-term sustainable improvement in lake water quality (Lewis and Wurtsbaugh 2008; Abell et al. 2010; Paerl et al. 2011). In this chapter, we present the background information and strategies required to achieve a nutrient-limited state for the management of phytoplankton in order to restore a lake to a previous higher water quality state, but not to its original pristine condition.

7.1.1 Nutrient Limitation of Phytoplankton

Natural phytoplankton communities are usually comprised of populations of different species. For these species to persist through time, their cellular growth rate must exceed or equal losses to dilution, sedimentation, physiological death, and grazing. As a general rule, for growth they require carbon, nitrogen and phosphorus in the atomic C:N:P ratio of 106:16:1 which equates to a mass ratio of 42:7.2:1 (Redfield 1958). If any one of these key nutrients is in short supply, that shortage will limit phytoplankton growth. Carbon dioxide (CO_2) is rarely a growth-limiting nutrient because the atmosphere readily replenishes the dissolved inorganic carbon (DIC) in water consumed during photosynthesis. Consequently, we normally refer to P-limitation or N-limitation.

These nutrient limitation states have been defined with reference to the total N (TN) and total P (TP) concentrations in the water in many papers (e.g. Abell et al. 2011) as:

- P-limitation: a mean ratio of TN:TP by mass of $>15:1$.
- N-limitation: a mean ratio of TN:TP by mass of $<7:1$.

Between these two mean ratios, the phytoplankton are likely to be nutrient replete and co-nutrient limitation may occur, i.e. the control of either N or P could reduce phytoplankton growth.

The form of the N and P is important. Nutrients need to be in a soluble form before they can be assimilated (taken up) by plants for growth. The main forms of utilisable N are nitrate-N (NO_3-N) and total ammoniacal-N, which is the preferred nutrient for plant growth. Total ammoniacal-N includes the non-toxic form as ammonium ions (NH_4-N) and unionised ammonia (NH_3), which is toxic to fish and other aquatic biota. The proportion of each component present in the total ammoniacal-N pool is a function of pH, with NH_4-N being the dominant form at pH < 8 while NH_3 is the dominant form at pH > 9. Since natural biological processes, such as photosynthesis, can change the pH from <8 to >9, it is important to measure pH when measuring total ammoniacal-N.

In freshwater systems, nitrite-N (NO_2-N) is usually at very low concentrations as it is a short-lived, transitional component in some of the processes in the nitrogen cycle. While it can be measured separately, it is also included in the total NO_3-N analysis. Consequently, it may be specified either as included in the NO_3-N value or as part of the total oxidisable N (NO_x-N). If NO_2-N is included in the NO_3-N value, that information should be stated. In this chapter, NO_2-N is included in the NO_3-N value. Where the form of the N is not important, we may refer to dissolved inorganic N (DIN) which includes NO_2-N, NO_3-N, and total ammoniacal-N, or simply N.

The main form of utilisable P is soluble reactive P (SRP), which may also be described in some papers as dissolved reactive P (DRP) or filterable reactive P (FRP). The main component of SRP is phosphate (PO_4^-) but the analytical method will also breakdown and include some of the low molecular weight dissolved organic P (DOP) in the water. The term SRP is, therefore, more precise than using the term phosphate (PO_4^-). Researchers often refer in general terms to N and P rather than specifying their forms.

7.1.2 Sources of Nutrients in Freshwater Lakes and Reservoirs

Both N and P come from both external and internal sources.

- **External sources** include point source inputs from rivers and streams, storm water and wastewater discharges, road runoff, diffuse source inputs from the catchment surrounding the lake, and rainfall. Point sources and rainfall enter directly into the lake while diffuse sources may enter the lake during rainfall events as overland flow or via groundwater inflows. Nitrogen can be in any form i.e. DIN, dissolved organic N (DON), and particulate N (PN). Phosphorus is mainly particulate P (PP) because SRP binds to the soil minerals iron (Fe), manganese (Mn), and aluminium (Al). However, biogeochemical processes can release P from mineral binding and P appears as SRP at relatively high

concentrations in spring waters of the central volcanic plateau of New Zealand (Timperley 1983).
- **Internal sources** include decomposition of organic matter in the water column and sediment and release from the legacy of nutrients stored in the sediment from past inflows (Burger et al. 2005).

There are general relationships between the external input and output nutrient loads, where a proportion of the external load is retained in the lake. For example, Burger et al. (2007) found that about half the external load into Lake Rotorua (North Island, New Zealand) was retained in the lake, adding to the legacy in the sediments, and that the internal load was usually much larger than the external load (Fig. 7.1). Consequently, for lake restoration, both the external and the internal loads need to be managed for effective lake restoration (Burger et al. 2008). Management of external loads is considered in detail in Chap. 4.

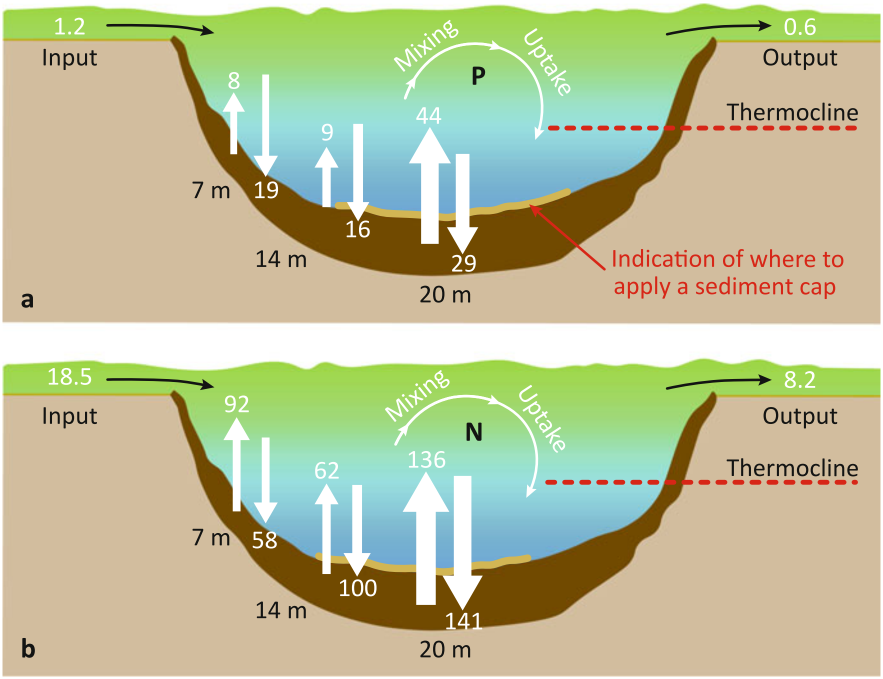

Fig. 7.1 Schematic of fluxes (mg m^{-2} day^{-1}) of (**a**) phosphorus and (**b**) nitrogen measured in Lake Rotorua showing that about half of the external P and N loads are retained in the lake (i.e. Input minus Output). The relative magnitude of each flux is indicated by the size of the arrow. The position where a sediment cap would be applied is indicated (see Sect. 7.3.2). Redrawn from Burger et al. (2007)

7.1.3 Internal Nutrient Cycling

Nutrients retained in a lake are a legacy of catchment inputs over time (Nürnberg 1996; Feature Box 7.1). To understand how a lake is functioning, before restoration measures are applied, it is advisable to construct a simple nutrient budget to determine whether the internal load is increasing or decreasing (Burger et al. 2008; see Chap. 5). With a hydraulic retention time of less than a year, it is possible for a lake to gradually lose nutrients due to flushing, under normal flow conditions. However, a single flood event may replenish nutrients lost over the remainder of the year. Consequently, managing the external loads is important.

N mostly enters the lake as DIN and most of that will be in the form of NO_3-N. During a flood event, the NO_3-N associated with overland flow is diluted with rainwater and the concentration decreases. In contrast, P mostly enters the lake in particulate form bound to the iron oxides in fine soil particles. Consequently, during a flood event, the concentration of TP will increase as the sediment load increases in the floodwater. For example, Hoare (1982) measured TP loads in the Ngongotaha Stream flowing into Lake Rotorua and found that TP was highly correlated with flow. He also found that 42% of the total annual P load was carried over a period of 3 days in 1976, and that 25% of the total annual P load in that year was carried at flow rates greater than 20 $m^3 s^{-1}$, even though that flow rate was exceeded only for 16 h. The P bound to fine sediment disperses around the lake and gradually settles coating the lake bed and any mitigation strategies with a layer of 'new' P.

While the P in the fine sediment is not readily available for plant growth, the fine sediment is a pollutant in its own right, blocking light and causing light stress or limitation to benthic organisms and plants. Because fine sediment is very slow to settle, it becomes a major problem in drinking water supply reservoirs by clogging filters in water treatment plants drawing that water. The treatment plant solution is to add a flocculant which rapidly coagulates the fine sediment causing it to aggregate into larger particles that settle rapidly. The use of flocculants in lakes is discussed below.

The release of nutrients from lake sediments is largely controlled by biogeochemical processes that are directly controlled by dissolved oxygen (DO) concentration (Fig. 7.2) and pH, with temperature increasing the process rates in summer. As DO concentrations reduce, the reduction–oxidation (redox) potential across the sediment–water interface declines and thereby reduces the oxidation state of the minerals binding P. The boundary between aerobic and anoxic/anaerobic conditions at the sediment–water interface can be very thin with a very large gradient over a few millimetres change in depth (Fig. 7.2).

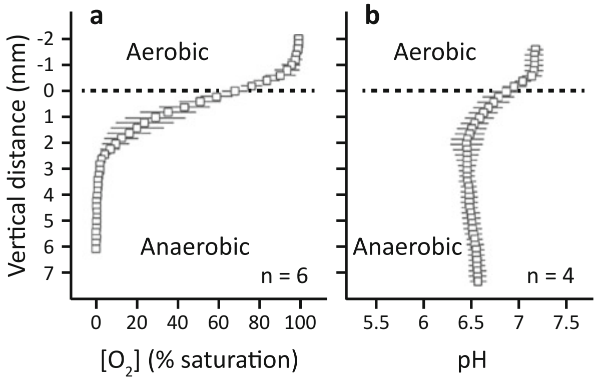

Fig. 7.2 Micro-profiles through the sediment–water interface with DO saturated water above anaerobic lower sediments. (**a**) DO profile and (**b**) pH profile. Note that the redox boundary is only about 3 mm thick (redrawn from Vopel et al. 2008)

Box 7.1. Geo-engineering in Lakes—An International Perspective

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'Geo-engineering in lakes' was defined by Mackay et al. (2014) as '. . .the deliberate manipulation of lake processes using natural and engineered amendments to induce a desired chemical or ecological outcome' (see Fig. 7.3 for an example). The approach is not new, for example, the use of aluminium salts as a topical treatment for phosphorus control in lakes and reservoirs has continued for nearly half a century, albeit, with mixed results (Huser et al. 2016a). In recent years, however, our understanding of legacy pollutants in lakes and their catchments has improved. We know now that catchment management, alone, without also controlling the release of legacy nutrient (particularly phosphorus) stores often in combination with other factors (e.g. restoring biological connectivity) is unlikely to result in ecological recovery within the short time scales demanded by society. That is, typically within the lifetime of a 3 to 5 year funding or political programme. However, it can take decades to centuries for legacy phosphorus to be naturally relinquished from eutrophic catchments and lakes, if at all, slowing ecological recovery and risking little apparent reward for significant investments (Jeppesen et al. 2005; Sharpley et al. 2013). This latter point is particularly problematic for wide-reaching environmental directives which demand that ecological improvements are achieved within a few decades or shorter. There is, therefore, a need to (1) explore methods to deliver rapid mitigation of the symptoms of eutrophication, (2) to develop approaches for managing recovery following catchment

(continued)

Box 7.1 (continued)

management, and (3) to educate the public and water managers on the time-scales and likelihood of recovery in lakes. In this context, geo-engineering has been demonstrated to reduce the release of legacy phosphorus stores in lake bed sediments (e.g. using lanthanum bentonite (Spears et al. 2016) or aluminium salts (Huser et al. 2016a)) and to achieve rapid reduction of human health risks posed by toxin-producing cyanobacteria using short-term or continuous treatment for lakes and reservoirs (e.g. using flocculants; Noyma et al. 2015; Shi et al. 2016).

So if the approach is so promising why does it remain contentious? The approach has, in some instances, been misused in the absence of robust site-specific evidence with which to assess the candidacy of lakes and the data produced from trials have, at times, been unconvincing. This has led researchers to call for (1) improvement in the procedures used to assess the efficacy of various materials and approaches in candidate lakes, the 'systems analysis' approach (Lürling et al. 2016), (2) longer term funding to support more comprehensive monitoring of 'experimental lakes', and (3) the establishment of a recognised international authority on geo-engineering to provide independent guidance on its use.

As in climate geo-engineering, alarmist sentiment can lead to unfounded caution in the face of strong promotion of products by a growing 'green industry'. The potential for harmful effects on flora and fauna, or at least the perception of risk associated with the addition of chemicals to natural waters, is a major limiting factor. Similarly, the addition of 'chemicals' in drinking water supply reservoirs is met with caution. Comprehensive assessment and data availability on ecotoxicological effects following lake applications are growing but rare (Spears et al. 2013b). Ecotoxicological data on one of the most widely used compounds, lanthanum-modified bentonite, is one notable exception (Copetti et al. 2016). Further complicating matters is the wide range of potential treatment strategies that have been reported. These approaches include, for example, the 'field of dreams' approach where repeated low-dose applications are used to create clear-water windows of opportunity for macrophyte recolonisation in shallow lakes and the 'shock and awe' approach where materials are dosed heavily in an attempt to control all potentially 'mobile phosphorus' in the system (Meis et al. 2013). Materials have also been applied to inflows to intercept phosphorus as a supplement to catchment management measures (Smith et al. 2016). Collectively, however, these approaches represent an impressive arsenal, but their relative benefits and costs should be more effectively communicated to water managers to support decision making.

Given the commercial interests in geo-engineering in lakes, both in terms of off-setting costs of eutrophication management (e.g. $2.2 billion per year cost

(continued)

Box 7.1 (continued)
of eutrophication of freshwaters in USA; Dodds et al. 2009) and direct commercial income to suppliers (cost to customer of about \$0.3 million to \$0.8 million per km^2 lake surface area; Spears et al. 2013a), new materials continue to be developed and proposed for use (Douglas et al. 2016). Hickey and Gibbs (2009) provide decision support, including cost analysis, for use of various materials in New Zealand. Hupfer et al. (2016) demonstrate a promising modelling approach with which the relative cost-effectiveness of catchment loading control coupled with 'repeated measures' geo-engineering for control of in-lake legacy phosphorus release from bed sediments can be predicted.

The variable success of the approach provides an opportunity for us to learn from our mistakes. Knowledge of the factors underlying variable longevity of treatments (Huser et al. 2016a, b) and that confound operational performance of materials (Dithmer et al. 2016a; Reitzel et al. 2013a, b) is growing, leading to greater confidence in dose estimation techniques. Finally, debate continues on ecotoxicological risk associated with established and emerging materials. This issue will be overcome if a central authority assumes responsibility for producing high quality validation data on established and emerging materials (Spears et al. 2013a).

The time scales associated with achieving adaptive change (i.e. changing resource use in food production) are long and we predict an increase in the use of geo-engineering in combination with other techniques to achieve topical mitigation of the symptoms of rapid environmental change. In this context, we expect the emergence of materials for the control of nitrogen, phosphorus, and carbon in the next five to ten years to satisfy the need to control eutrophication and greenhouse gas emissions in freshwaters, simultaneously. Over similar time frames, geo-engineering will be utilised to supplement other approaches including methods to overcome constraints on biological connectivity (e.g. macrophyte propagule dispersal techniques) which currently restrict biodiversity. Finally, the approach remains one of the only options for rapid topical treatment of water quality to mitigate human health risks associated with cyanobacteria and other waterborne pathogens.

Because the sediment is anoxic/anaerobic below the surface, N and P in the pore water are in the form of NH_4-N and SRP. The NH_4-N diffuses up through the sediment surface into the overlying water continuously with the rate of diffusion out of the sediment (release) being determined by the concentration gradient between the pore water and the overlying water. The release rate is usually highest in summer, i.e. release is due to microbial activity which is faster when warm.

Considering nitrogen processes (Fig. 7.4, left), organic matter is decomposed and mineralised into ammoniacal-N in the sediment. As this NH_4-N diffuses up to the sediment surface, it passes through a zone where nitrifying bacteria can convert it to

Fig. 7.3 Geo-engineering trial of a modified lanthanum bentonite product called 'Phoslock' at Mere Mere, UK. Image courtesy of Bryan Spears, Centre for Ecology & Hydrology, UK

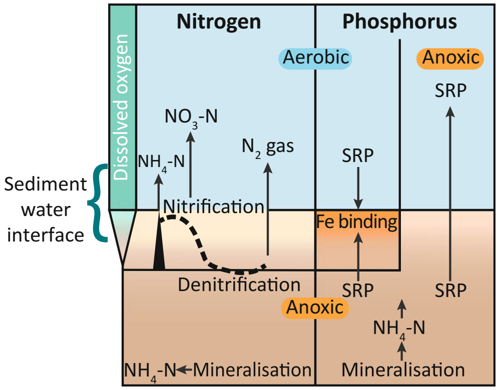

Fig. 7.4 Schematic of N and P processes across the sediment–water interface. Low concentrations of SRP can diffuse out of the sediment under aerobic conditions or can be released at pH > 9

NO_3-N so that the amount of NH_4-N escaping into the overlying water is minimal or zero. In this zone, denitrifying bacteria can convert some or all of the NO_3-N to nitrogen gas (N_2), which escapes to the atmosphere. These processes provide a pathway for reducing the N concentrations in a lake. Note that if the overlying water is anoxic, NH_4-N will escape into the overlying water directly because the nitrification step does not occur. If NO_3-N is not produced, denitrification stops. Consequently, in anoxic water normally only the NH_4-N form of N is present.

In contrast to the nitrogen processes, there is no air phase mechanism for reducing phosphorus in the lake (Fig. 7.4, right), even though the decomposition and mineralisation processes are comparable. When the overlying water is aerobic, the SRP from mineralisation in the sediments must diffuse up through a thin layer of iron oxide (ferric oxide) at the sediment–water interface. Some of this SRP will escape into the overlying water as a low SRP concentration flux but most of the P gets bound to the insoluble ferric oxide and is held in the sediments. If the overlying water becomes anoxic, the redox potential falls and the ferric oxide binding the P dissolves into soluble ferrous form releasing the SRP into the overlying water column. This process is reversible and when the overlying water becomes aerobic again, the ferrous ions in the water oxidise and precipitate as ferric oxides, which sequester the P from the water column again. The kinetics of this process have a time lag so that there is a residual amount of SRP in the water column for a short period of time after re-oxygenation when phytoplankton can uptake the P for growth. The phytoplankton eventually die and sediment to the lake bed completing the cycle without loss of P.

The redox kinetics of the iron cycle also apply to the manganese cycle (Fig. 7.5), with the transition between insoluble manganic oxide and soluble manganous ions occurring at a higher DO concentration. Because these geochemical transformations occur at specific redox potentials, they can be linked to the DO concentration in the lake water at a pH of around 7 (Stumm and Morgan 1995). For example, the dissolution of manganese oxide to Mn^{2+} begins when the redox potential falls below 0.55 Eh Volts, which equates to a DO concentration of about 5 g m^{-3} (Chap. 6, Fig. 6.2), but the dissolution of iron oxide to Fe^{2+} begins when the redox potential falls below 0.1 Eh Volts, which equates to a DO concentration of about 2 g m^{-3} (Achterberg et al. 1995; Stumm and Morgan 1995).

To break these cycles, P-binding or P-inactivation agents, which are not affected by redox changes at normal 'near neutral' pH, can be introduced as thin layers across the sediment surface as a sediment cap to replace the iron and manganese binding. As with iron, these P-inactivation agents only bind SRP and not particulate or organic forms. Once bound to these agents, the P cannot by utilised by phytoplankton and the bound P is buried in the sediment (e.g. Lewandowski et al. 2003). In practice, the capping material placed over the sediment surface causes the sediment immediately below the layer to become anoxic (Vopel et al. 2008), thereby releasing the P from the iron and manganese allowing it to diffuse up and be sequestered by the capping agent.

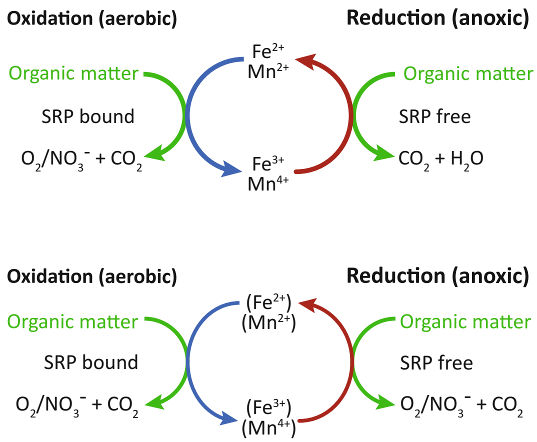

Fig. 7.5 Schematic of the iron and manganese cycles at neutral pH. The transition point for iron occurs at a dissolved oxygen concentration around 2 g m^{-3} and for manganese at around 5 g m^{-3} (image redrawn from Roden 2012)

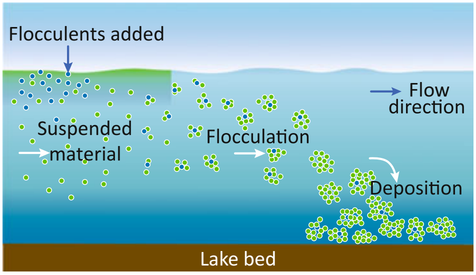

Fig. 7.6 Schematic of concept of flocculation. Redrawn from internet site www.ewisa.co.za/eWISAWaterworks/misc/WaterTreatment/defaultcoagulation.htm

7.1.4 Definitions

- **Flocculent**: Resembling 'wool' especially in loose fluffy organisation. Also, containing, consisting of, or occurring in the form of loosely aggregated particles or soft flakes.
- **Flocculation**: The process by which individual particles of clay aggregate into clot-like masses or precipitate into small lumps. Flocculation occurs as a result of a chemical reaction between the clay particles and another substance.
- **Flocculant**: Something that rapidly traps the fine suspended particles in a liquid in its 'fluffy structure' causing them to aggregate into larger particles that have increased mass and settle rapidly (Fig. 7.6).
- **Sediment capping**: A non-removal remediation technique for contaminated sediment that involves leaving the contaminants in place and isolating them from the environment by placing a layer of soil and/or other material over the sediment to prevent further spread of the contaminant. In lakes, the issue is legacy sediment from the catchment, and the key contaminant targeted by the sediment cap is P.
- **Passive capping agent**: A stable layer of inert material, such as sand, clay, sediment, gravel, rock, and/or synthetic material, which effectively isolates the contaminant from the environment due to the thickness of the layer. Layer thicknesses are measured in centimetres and form a physical barrier.
- **Active capping agent**: A chemically active material that targets the contaminant using only a thin layer. Layer thicknesses are measured in millimetres. Active capping agents may be natural, e.g. zeolite, allophane, red clay, precipitated lime, or synthetic, e.g. aluminium sulphate (alum), aluminium modified zeolite (Aqual-P®), and lanthanum-modified bentonite (Phoslock™), with new products being designed and tested at the time of writing. For example, bioactive multilayer capping (BMC) with biozeolite and sand has been tested for removal of N released from the sediment (Huang et al. 2013).

7.2 Flocculation

Once a cyanobacterial bloom has developed, it is too late to use many of the restoration techniques such as sediment capping, but the use of flocculants is an option. The advantage of using flocculants in lakes is that an algal bloom or high turbidity can be treated directly to get immediate results. To that end, much research has focused on the use of clay to floc the bloom so that it settles out of the water column (Sengco and Anderson 2004; Beaulieu et al. 2005; Padilla et al. 2006; Zou et al. 2006; Biyu et al. 2010; Chen and Pan 2012; Pan et al. 2012). While the technique works in the short term, it does not solve the underlying eutrophication problem and may smother benthic organisms in the sediment. This process may be used prior to other treatments to obtain better control (see Sect. 7.5 below).

As well as treatment of the lake water, flocculants are often used to treat water flowing into streams and lakes to remove fine sediment carrying P. While aluminium sulphate (alum) and poly aluminium chloride (PAC) are well known for removing sediment from swimming pools and drinking water in water treatment plants, they can also bind P in the sediment of a lake and can be used to manage internal P loads (Feature Boxes 7.1, 7.2). Both alum and PAC contain aluminium which, under specific conditions of pH, can be toxic to aquatic organisms (*see* Sect. 7.4 below). Alum is acidic and must be used with an appropriate buffer to hold the water at a circumneutral pH to prevent toxic trivalent Al^{3+} ions forming. These two products are also used to actively bind P from the water column (*see* Sect. 7.3.2 below). In New Zealand, neither product can be used in a natural lake without a permit.

Another flocculant, commonly used in agriculture to stop erosion of fine soil particles from irrigated crop land in United States and in China, is anionic polyacrylamide (PAM). Polyacrylamide is the generic name for a group of very high molecular weight macromolecules produced by the free-radical polymerisation of acrylamide and an anionically charged co-monomer, sodium acrylate. The combination of molecular weight and ionic charge results in extremely viscous aqueous solutions, one of the main properties of these polymers. Both the charge density (ionicity) and the molecular weight can be varied to suit the application. Whereas anionic PAM is essentially non-toxic, cationic PAM is toxic (Table 7.1) and, although it is a very effective flocculant, cationic PAM should not be used directly in natural waters. Site-specific evaluations should be undertaken in situations where it may enter natural water in streams or lakes to ensure effective toxicity reduction—generally by flocculation, but dissolved organic carbon also reduces toxicity.

Table 7.1 Summary table of anionic and cationic polymer flocculants with their toxicity (LC_{50}) values for fish and invertebrates as reported from studies or manufacturers' MSD sheets

Anionic polyacrylamides (PAMs)				Organism				
Manufacturer	Trade name	Chemical name	CAS No.	Common name	Scientific name	Test duration (h) (acute duration unless stated)	LC_{50} (mg L^{-1})	Reference
Applied Polymer Systems	APS-700 Series Silt Stop							1
Applied Polymer Systems	APS-712			Cladoceran	*Daphnia magna*	48	17,450	1
Applied Polymer Systems	APS-705			Cladoceran	*Daphnia magna*	48	869	1
Applied Polymer Systems	APS-712			Fathead minnow	*Pimephales promleas*			1
Applied Polymer Systems	APS-705			Fathead minnow	*Pimephales promleas*	96	3250	1
SNF, Inc.	GeoScrub 10			Cladoceran	*Daphnia magna*	48	1337	1
Ixom (previously Orica)	PAC	Polyaluminium chloride	1327-41-9	Cladoceran	*Daphnia magna*	48	>5000	MSDS
Ixom (previously Orica)	PAC	Polyaluminium chloride	1327-41-9	Rainbow trout	*Oncorhynchus mykiss*	96	390	MSDS
Ixom (previously Orica)	PAC	Polyaluminium chloride	1327-41-9	Fathead minnow	*Pimephales promleas*	96	517	MSDS
Applied Polymer Systems	APS 706b Floc Log			Cladoceran	*Daphnia magna*	48	>420	MSDS
Applied Polymer Systems	APS 706b Floc Log			Rainbow trout	*Oncorhynchus mykiss*	96	637	MSDS

(continued)

Table 7.1 (continued)

Anionic polyacrylamides (PAMs)				Organism		Test duration (h) (acute duration unless stated)	LC_{50} ($mg\ L^{-1}$)	Reference
Manufacturer	Trade name	Chemical name	CAS No.	Common name	Scientific name			
Applied Polymer Systems	APS 706b Floc Log			Fathead minnow	*Pimephales promleas*	96	>1680	MSDS
Applied Polymer Systems	APS 706d Floc Log			Cladoceran	*Daphnia magna*	48	383	MSDS
Applied Polymer Systems	APS 706d Floc Log			Rainbow trout	*Oncorhynchus mykiss*	96	1900	MSDS
Applied Polymer Systems	APS 706d Floc Log			Fathead minnow	*Pimephales promleas*		ND	MSDS
Applied Polymer Systems	APS 712 powder			Cladoceran	*Daphnia magna*	48	1617	MSDS
Applied Polymer Systems	APS 712 powder			Rainbow trout	*Oncorhynchus mykiss*		ND	MSDS
Applied Polymer Systems	APS 712 powder			Fathead minnow	*Pimephales promleas*	96	>6720	MSDS
Ixom (previously Orica)	Crystalfloc B610 (Anionic PAM)	Acrylamide/ Sodium acrylate copolymer		Crustacean	*Daphnia magna*	48	>100	MSDS
Ixom (previously Orica)	Crystalfloc B610 (Anionic PAM)	Acrylamide/ Sodium acrylate copolymer		Zebrafish	*Danio rerio*	96	>100	MSDS
Ciba Specialty Chemicals, Australia	Magnasol AN2			Northern trout gudgeon	*Mogumda mogumda*	96	>1020 (as TOC)	2

Ciba Specialty Chemicals, Australia	Magnasol AN2			Crustacean	*Moinodaphnia macleayi*	120–144 (growth, chronic)	3 (as TOC)	2
Ciba Specialty Chemicals, Australia	Magnasol AN2			Green hydra	*Hydra viridissima*	72 (growth, chronic)	170 to >205 (as TOC)	2
Ciba Specialty Chemicals, Australia	Magnasol AN2			Tropical duckweed	*Lemna aequinoctialis*	96 (growth, chronic)	190 (as TOC)	2
Ciba Specialty Chemicals, Australia	Magnasol AN2			Alga	*Chlorella* sp.	72 (growth, chronic)	220 (as TOC)	2
Ciba Specialty Chemicals, Suffolk, VA, USA	Magnasol AN2			Lake trout	*Salvelinus namaycush*	96	>600	3
BASF	Magnasol AN2			Rainbow trout	*Oncorhynchus mykiss*	96	>100	MSDS
BASF	Magnasol AN2			Crustacean	*Daphnia magna*	48	>100	MSDS
SNF Holding Co.	Flopam AN923			Zebrafish	*Danio rerio*	96	>100	4
SNF Holding Co.	Flopam AN923			Cladoceran	*Daphnia magna*	48	>100	4
SNF Holding Co.	Flopam AN923			Alga	*Scenedesmus subspicatus*	72	>100	4
SNF Holding Co.	Flopam AN923			Mussel glochidia	*Lampsilis cariosa; Alasmidonta raveneliana; Megalonaias nervosa*	24	412–1000	4
SNF Holding Co.	Flopam AN923			Juvenile mussels	As above	96	129 to >1000	4

(continued)

Table 7.1 (continued)

Cationic polyacrylamides (PAM)				Organism		Test duration (h)	LC50 (mg L^{-1})	Reference
Manufacturer	Trade name	Chemical name	CAS No	Common name	Scientific name			
HaloSource, Inc	HaloKlear Series							
HaloSource, Inc	DBP-2100 MB			Fish (Fathead minnow)	*Pimephales promleas*	96	825	1
HaloSource, Inc	GelFloc MB				–		–	1
Degusa	Praestol 186K	Polydiallyldimethyl ammonium chloride	26062-79-3	Crustacean	*Daphnia magna*	48	0.57 in 100 mg L^{-1} $CaCO_3$	MSDS
Degusa	Praestol 186K	Polydiallyldimethyl ammonium chloride	26062-79-3	Fathead minnow	*Pimephales promleas*	96	16.3 in 180 mg L^{-1} $CaCO_3$; with 10 mg L^{-1} TOC	MSDS
Degusa	Praestol 186K	Polydiallyldimethyl ammonium chloride	26062-79-3	Fathead minnow	*Pimephales promleas*	96	0.75 mg L^{-1} in 100 mg L^{-1} $CaCO_3$	MSDS
Ixom (previously Orica)	Crystalfloc B400 (Cationic PAM)	Acrylamide/ Sodium acrylate copolymer		Crustacean	*Daphnia magna*	48	20–50	MSDS
Ixom	Crystalfloc B400 (Cationic PAM)	Acrylamide/ Sodium acrylate copolymer		Fish (Zebrafish)	*Danio rerio*	96	5–10	MSDS
Ixom	Crystalfloc B3H (polyDADMAC)	Polydiallyldimethyl ammonium chloride	26062-79-3	Crustacean	*Daphnia magna*	48	>10	MSDS
Ixom	Crystalfloc B3H (polyDADMAC)	Polydiallyldimethyl ammonium chloride	26062-79-3	Zebrafish	*Danio rerio*	96	>10	MSDS
Ixom	Crystalfloc L3RC (polyDADMAC)	Polydiallyldimethyl ammonium chloride	26062-79-3	Crustacean	*Daphnia magna*	48	10–100	MSDS
Ixom	Crystalfloc L3RC (polyDADMAC)	Polydiallyldimethyl ammonium chloride	26062-79-3	Zebrafish	*Danio rerio*	96	10–100	MSDS
Ciba Specialty Chemicals, Suffock, VA, USA	MagnaFloc368			Fish (Lake trout)	*Salvelinus namaycush*	96	2.08	3
BASF	MagnaFloc368			Crustacean	*Ceriodaphnia dubia*	48	>10	MSDS

Chitosans				Organism		Test duration (h)	LC50 ($mg\ L^{-1}$)	Reference
Manufacturer	Trade name	Chemical name	CAS No	Common name	Scientific name			
KML Inc	Chitosan (KML V2; 2% active ingredient (AI))	Poly-*N*-acetyl-D-glucosamine		Rainbow trout	*Oncorhynchus mykiss*	48	0.38 (on AI basis)	5
KML Inc	Chitosan (KML V2; 2% AI)	Poly-*N*-acetyl-D-glucosamine		Channel Catfish	*Ictalurus punctatus*	48	0.37 (on AI basis)	5
KML Inc	Chitosan (KML V2; 2% AI)	Poly-*N*-acetyl-D-glucosamine		Mussel (Threehorn Wartyback)	*Obliquaria reflexa*	48	>100 (on AI basis)	5
KML Inc	Chitosan (V54; 5.4% AI)	Poly-*N*-acetyl-D-glucosamine		Rainbow trout	*Oncorhynchus mykiss*	48	0.5 (on AI basis)	5
KML Inc	Chitosan (V54; 5.4% AI)	Poly-*N*-acetyl-D-glucosamine		Channel Catfish	*Ictalurus punctatus*	48	0.92 (on AI basis)	5
KML Inc	Chitosan (V54; 5.4% AI)	Poly-*N*-acetyl-D-glucosamine		Threehorn Wartyback	*Obliquaria reflexa*	48	>100 (on AI basis)	5
Sigma-Aldrich	Chitosan	Poly-*N*-acetyl-D-glucosamine	9012-76-4	Crustacean	*Daphnia pulex*	48	13.7	MSDS
Sigma-Aldrich	Chitosan	Poly-*N*-acetyl-D-glucosamine	9012-76-4	Rainbow trout	*Oncorhynchus mykiss*	96	1.7	MSDS

Abbreviations: LC_{50}, concentration causing a 50% lethality or growth effect; MSDS, Material Safety Data Sheet; AI, Active Ingredient; TOC, Total Organic Carbon.
References: (1) Johnson and Klaine (2014), (2) Harford et al. (2011), (3) Liber et al. (2005), (4) Buczek et al. (2017), (5) Waller et al. (1993)

Box 7.2. Lake Geoengineering Using Phosphorus-Binding Agents

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The cycling of phosphorus from bottom sediments to the water column and lake eutrophication are often linked; however, it is the watershed that controls the delivery of P to lakes and accumulation in sediment (Huser et al. 2016a, b). Changes in land use cause increase transport of P to lakes, with most of this P being stored as a legacy in the sediment for decades or longer (Sas 1990). Whether, and to what degree, legacy P in sediment leads to internal P cycling largely depends upon the abundance and type of phosphorus-binding elements (e.g. Mn, Fe, Mg, Ca, and Al) that co-deposit with phosphorus and in-lake chemical conditions (e.g. redox potential).

Several studies (discussed below) have reported that ratios of phosphorus to binding elements can explain some of the phosphorus variability normally observed in lakes. In Michigan, United States, calcium in a lake water column was shown to co-precipitate with phosphorus during phytoplankton blooms, thereby regulating phosphorus concentration in the studied lakes (Hamilton et al. 2009). Jensen et al. (1992), demonstrated that the ratio of iron to phosphorus in Danish lake bottom sediments was negatively correlated with phosphorus concentrations in the water column of these lakes. Hu and Huser (2014) reported lower phosphorus in the water column of 65 lakes in Sweden treated with lime for the purpose of raising pH. This was linked to increased Al-P formation in acidified lakes, which was magnified in limed systems (elevated Al-P formation and Ca-P formation, Huser and Rydin 2005). Kopáček et al. (2007) showed that in an untreated Czech Republic lake, sediment with higher ratios of Al to Fe were associated with higher ratios of aluminium bound phosphorus (Al-P) to iron bound phosphorus (Fe-P).

Previous work linking P-binding metals and sediment sorption capacity suggests P management using a geoengineering approach—in this case the addition of phosphorus-binding elements (e.g. Al, Fe, Ca) to lake sediment—is a viable approach to control internal phosphorus loading. These metals are found naturally in soils and sediments in the form of minerals that bind P. Aluminium salts (aluminium sulphate, aluminium chloride, etc.) are the most commonly used sediment amendments as aluminium bound phosphorus (Al-P) is not sensitive to reducing conditions that are often found in eutrophic systems. Aluminium is also the most common metal in the Earth's crust and has been used for nearly half a century to manage internal P loading in lakes (Cooke et al. 2005). The Al-P complex is stable in pH conditions predominant in most lakes (6–9), dosing can be adjusted to reach a desired result (Huser and Pilgrim 2014; Rydin and Welch 1999; Reitzel et al. 2005) and the factors that affect

(continued)

Box 7.2 (continued)

longevity of treatment are becoming better understood (Huser et al. 2016a, b). To date, aluminium is one of only a few elements that has been reported to eliminate excess internal phosphorus loading for decades or longer after treatment (Huser et al. 2016a, b).

Iron salts have not been used as widely as aluminium, mainly due to their limited binding under low-oxygen conditions, and because dosing methods are not as well defined. Of the published studies, full-scale applications of iron and iron combined with aeration (Engstrom 2005) in lakes have been shown to reduce internal cycling of P. Recent research (Orihel et al. 2016) with mesocosms in a hypereutrophic lake in Alberta, Canada, has demonstrated that high Fe doses, ranging from 64 to 225 g Fe m^{-2}, reduced summer phosphorus release rates from 19.0 to as low as 8.1 mg m^{-2} day^{-1}. Using iron and phosphorus masses reported for the treated sediment in this study, the highest iron treated sediment had a Fe to P w/w ratio of 50 (Fe for the sediment solids plus pore water). Jensen et al. (1992) identified a Fe to P w/w ratio above 15 as a potential management target for internal phosphorus load control, though this was only for well oxygenated, aerobic conditions. It is not clear what the longevity of Fe addition may be, and long-term viability of this approach may be limited where concentrations of sulphur allow for significant formation of non-P-binding minerals such as FeS/FeS_2.

Calcium, applied as calcium hydroxide or mixtures including calcium carbonate, can lead to increased phosphorus binding and formation of calcium phosphate (Ca-P) in lake sediments (Dittrich et al. 2011). Repeated full-scale lake treatments in Canada (Prepas et al. 2001) demonstrated that phosphorus in two eutrophic to hypereutrophic lakes (Halfmoon and Figure Eight Lakes) was reduced measurably in the lake water column but the reduction was not enough to change the trophic state of these systems. Because the formation of Ca-based P-binding minerals is generally more efficient at pH > 8, the use of Ca is more likely to be effective in alkaline systems.

It should be recognised that for all of these elements, there will be non-specific binding of sediment constituents (e.g. de Vicente et al. 2008a) as well as geochemical transformations (e.g. crystallisation, Berkowitz et al. 2006) that will lead to reduced surface area, and thus binding sites, of these types of amendments following treatment. Longevity of any geochemical treatment will generally be based on the ratio of internal to external load, lake mixing regime (controls the transfer of P from sediment to surface water), and the amount of mineral added to bind excess sediment P and reduce internal P loading (Huser et al. 2016a, b).

There are some newly developed products, mainly modified clay materials that can control internal P cycling. One of these products (Phoslock™) that has lanthanum (La) inserted into the clay matrix has been used in a number of

(continued)

Box 7.2 (continued)
lakes in Europe (Mackay et al. 2014, see Feature Box 7.1 of this chapter). The main binding pathways include $LaPO_4$ formation and edge binding to the clay material. These materials are relatively new with longevity, dosing, and the ultimate effects on sediment P pools currently being investigated. Some of these investigations have reported on delayed sediment P uptake (Dithmer et al. 2016a, b), difficulty in reducing the sediment pools available for release (Meis et al. 2012, 2013), binding of P pools that do not contribute to sediment P release (Meis et al. 2012), and P uptake reduction and release of La when high levels of dissolved organic carbon are present (Lürling et al. 2014). These engineered products are also likely to be more expensive due to the additional handling needed to create the product. Because aluminium treatment is the most commonly used sediment amendment to manage internal P loading in lakes, it has the longest record of use. Thus, a table of treatment considerations applicable to aluminium is provided, but these will generally apply to other amendments as well (Table 7.2).

The cause of the toxicity of cationic PAM is its ability to form an 'ionic bridge' directly with the clay particles (Fig. 7.7) and any other negatively charged particles including aquatic organisms, but excluding plants. In contrast to the direct ionic bridge formation by cationic PAM, anionic PAM needs a cation such as a calcium ion (Ca^{2+}) to form the ionic bridge and consequently cannot attach directly to aquatic biota.

Anionic PAM is also used in cosmetics, food production (e.g. clarification of fruit juices), and stock food thickening. It has been tested in New Zealand for removing sediment from construction site sediment detention ponds (ARC 2004a, b), and the anionic PAM 'Crystalfloc B610' (Ixom 2011) was utilised as a risk management tool on an 'as required' basis to help achieve the necessary water quality targets during the construction of the Mackays to Peka Peka Expressway near Wellington (NZTA 2012).

Apart from reducing soil erosion from cultivated agricultural land under irrigation, polyacrylamides are also widely used as a water treatment for storm water discharges and as a pre-emptive treatment for any runoff situation where fine sediment may enter a waterway. The US EPA Office of Water has produced a handout providing information on best management practices for storm water (US EPA 2013). Advice on the use and application of PAMs is freely available from the internet (e.g. APS 2002).

Because of the long history of use in the USA agriculture industry, it is generally accepted that anionic PAMs are 'safe' to use in all situations for application to soil. However, not all anionic PAMs are equal. Some work better on different soils (the supplier will advise), and the various formulations available have different LC_{50} values (Table 7.1). When used properly, anionic PAM has no measurable toxicity to humans, aquatic organisms, or plants.

Table 7.2 Considerations for aluminium treatment of lake sediment

Consideration	Effect	Compensating measure	Further reading
Alkalinity and pH	• Potential aquatic life toxicity starting at low pH: 5.5 to 6.0 and below • Highly soluble and poor flocculation at high pH-8.5 and above	• Use of buffered Al forms or split treatment into multiple applications to limit pH change • Timing of treatment when lake pH is closest to 6	Pilgrim and Brezonik (2005), Jensen et al. (2015)
Watershed to lake area	• High watershed to lake area leading to high phosphorus loading to sediments	• Split dose into lower doses and apply over several years • Implement watershed sediment and phosphorus load controls before treatment	Welch and Cooke (1999), Huser et al. (2016a, b)
Sediment bed slope and dosing effects on crystallisation and binding effectiveness	• Binding effectiveness limited when crystallisation occurs in absence of P	• Split treatments over several years • Treat areas outside of anaerobic zones with highest P release	Huser (2012, 2016a), Egemose et al. (2013)
Mobile and labile organic P forms	• High Mobile P • High Org-P	• Split doses over time to limit crystallisation (without P) and increase binding efficiency	Reitzel et al. (2005), de Vicente et al. (2008b), Huser (2016a)
Sediment mixing from fish (e.g. carp) and wave action effects on available sediment P and Al-floc distribution	• Sediment mixing from coarse fish or waves dilutes added Al over greater sediment depth and/or translocates Al-floc after treatment	• Use coring and sediment P data to determine sediment mixing (and treatment) depth • Split doses over several years to achieve even distribution in sediment	Egemose et al. (2010), Huser et al. (2016a, b)
High variability of mobile P and Org-P in lake sediment	• High variability in sediment complicates dosing approach	• Conduct intensive coring across lake to map the spatial distribution of labile P forms	James and Bischoff (2015)

A recent study by Buczek et al. (2017) of five different anionic PAMs and one non-ionic compound found median lethal concentrations (LC_{50}) ranged from 411.7 to >1000 mg L^{-1} for glochidia and from 128.7 to >1000 mg L^{-1} for juveniles from three freshwater mussel species. All LC_{50} values were orders of magnitude greater (2–3) than concentrations typically recommended for turbidity control (1–5 mg L^{-1}), regardless of their molecular weight or charge density. They concluded that the PAM compounds tested were not acutely toxic to the mussel species and life stages tested,

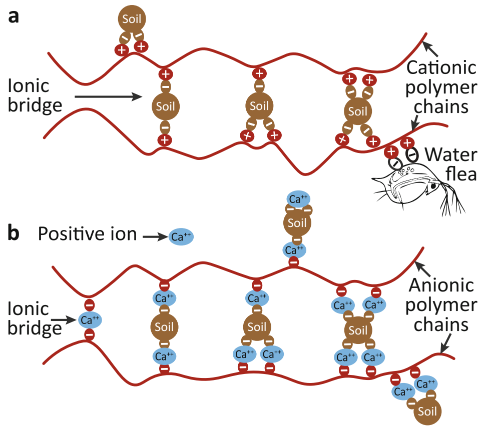


Fig. 7.7 Schematic diagram of the difference between (**a**) cationic PAM, which is toxic to aquatic biota, and (**b**) anionic PAM, which is non-toxic (redrawn from US EPA 2013 with added Daphnia)

indicating minimal risk of short-term exposure from PAM applications in the environment. A similar study by Kerr et al. (2014) compared the effects of ten different PAMs at concentration levels from 3 to 300 mg L^{-1} on juvenile rainbow trout (*Oncorhynchus mykiss*) and concluded that, while there was evidence of some gill irritation with some anionic PAMs at the highest concentrations, the cationic PAMs produced a similar level of irritation at concentrations 1000 fold lower than the anionic PAMs tested.

Toxicity data for chitosan flocculants is limited, with LC_{50} values varying widely depending on species and formulations (Table 7.1). The data indicates that fish, particularly rainbow trout, may be highly sensitive based on the active ingredient content of the products.

There are also a number of application restrictions that have been developed in the USA to ensure, safe, effective, and environmentally friendly application of anionic PAM to construction sites (e.g. VDEQ 1992, 2002). The PAM copolymer formulation:

- Must be anionic (negatively charged), with a charge density of 8–35% by weight, (15–18% is typical).
- Must have an ultra-high molecular weight of 6–24 mg mol^{-1} (preferably 12–-15 mg mol^{-1}).
- Must be water-soluble, 'linear', or 'non-crosslinked'.
- Must have a residual acrylamide monomer content of <0.05%.

There are New Zealand standards for the supply of polyacrylamides for use in drinking water treatment (NZWWA 1999).

The high molecular weight anionic PAM suggested has a low toxicity risk, as the molecules are so ultra large that they do not penetrate biological membranes. Additionally, when anionic PAM is introduced into waters containing sediments, humic acids, or other impurities, its effects are buffered to an even greater degree (Anon. 2002).

Putting this in perspective, toxicity limits were derived for clean water for short-term (acute) and long-term (chronic) protection. If applied to turbid water, the flocculation action further reduces the amount of dosed product left in the water. Ideally, this should result in a very low residual amount of the product such that there is negligible toxicity and residual concentrations are reduced below the environmental water quality guideline. To achieve this result, the suspended sediment load needs to be measured and an appropriate application rate established.

An ecotoxicological assessment of an anionic PAM (Magnasol AN2; formulated as 60% PAM with polyethylene glycol in a block form; Magnafloc®, Ciba Specialty Chemicals, Australia) was established for five species representative of a range of trophic levels by Harford et al. (2011). Their results demonstrated that the sensitivity of freshwater species to anionic polyacrylamide can vary considerably, with the most sensitive being reproductive effects on a cladoceran species. These and other toxicity testing results using chronic and sub-lethal endpoints have shown that cladocerans are adversely affected by PAM at concentrations ~1–2 g m^{-3} (nominal PAM).

A key challenge with respect to the application of water quality guideline values for flocculant compounds is the ability of a standard water quality monitoring programme to measure and detect concentrations of dissolved flocculant components in receiving waters where such products have been used. Analytical methods for specific PAMs are not standardly available, so non-specific methods, such as total organic carbon (TOC), must be used. However, the low concentrations of PAM which may be required to be measured for higher value ecosystems will often be below the background level of TOC in many freshwaters.

PAM is one of several polymer flocculants available and each has different properties. These products include Polydiallyldimethylammonium chloride (PolyDADMAC), a high charge density cationic polymer marketed in New Zealand by IXOM (formerly Orica Ltd) as 'Crystalfloc B3H' or Crystalfloc L3RC (Ixom 2012). While it is claimed by the suppliers to be fit for use in drinking water treatment as a direct replacement for alum, the product Safety Data Sheet has a warning 'Harmful to aquatic life with long lasting effects'. Consequently, while this product is effective in storm water and sediment settling in construction site detention ponds, it may not be useful in the restoration of lakes and a site-specific risk assessment should be undertaken prior to use in a natural environment. Consequently, we recommend use of an anionic PAM such as Crystalfloc B610 (Ixom 2011) in preference.

The disadvantages of using flocculants in lakes are that, unless they are also an active capping agent such as alum, they will have no lasting effect on the algal bloom and may stimulate greater nutrient release from the sediment by increasing the carbon content from the algal cells, thereby raising the sediment oxygen demand (SOD), which drives the sediments anoxic.

A new product being developed in China as a flocculant uses nano-bubble technology to activate natural local soils for use as flocculants. This technique may overcome some of these issues and provides a new approach to the restoration of shallow lakes (Pan et al. 2011, 2012; Chen and Pan 2012). In essence, the seeds of desirable macrophytes are included in the mixture that comprises the flocculant. This means that when the water clears, the seeds are on the sediment surface and ready to germinate (Pan et al. 2011). This process could also be used with passive sediment capping.

7.3 Sediment Capping

The advantages of using a sediment capping agent are that the contaminant in the legacy sediment doesn't have to be removed from the lake (Fig. 7.8). There are two types of sediment capping agents: 'passive' which has no chemical reaction with the target nutrients or contaminants and will have a general impact on the lake sediments and 'active' which has a chemical bond with the targeted nutrient and, therefore, can be applied to specific sites to achieve their efficacy. A broad range of application techniques have been developed (e.g. Filshill and Loux 2014).

7.3.1 Passive Sediment Capping Agents

Most products used for passive sediment capping are natural and rely on the thickness of the layer applied to prevent sediment disturbance or bioturbation from releasing the contaminant. The result is that the contaminant flux from the sediment is reduced to molecular diffusion rates (e.g. 10^{-4} to 10^{-5} mg m^{-2} day^{-1}) and it will have minimal effect on water quality. The disadvantage of using passive sediment capping is the amount and, therefore, the cost of capping agent that has to be applied to the lake to achieve the cap. Effective layers can be 4–8 cm (Kim et al. 2007) with thicker layers up to 36 cm (14 in.) being tested for local capping of toxic waste (Sevenson 2006). Thinner layers may be effective provided the thickness of the cap layer exceeds the depth of organism interaction with the sediments (bioturbation) and the pore water concentrations within the biologically active zone remain low,

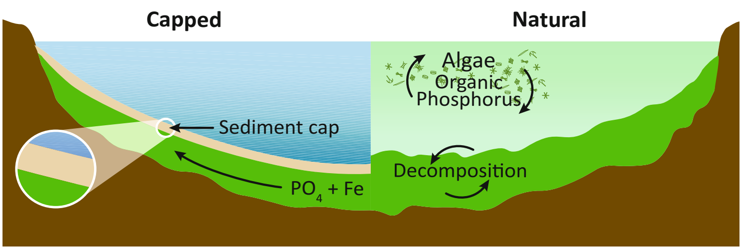


Fig. 7.8 Schematic diagram of sediment capping to manage internal P loads. The sediment capping is only required in the zone below the thermocline to block P release from the sediment when the hypolimnion goes anoxic

i.e. molecular diffusion controls transport from the underlying sediment layer (Lampert et al. 2011).

For small ponds, geotextiles or latex films can be used as passive capping barriers. While these are impractical for larger waterbodies, they can be used to isolate localised areas of contaminated sediment from the rest of the waterbody in larger lakes, e.g. in marinas.

7.3.2 Active Sediment Capping Agents

Active sediment capping agents targeting P are also known as P-inactivation agents and are used to make that P unavailable for algal growth by binding it to a metal such as Fe, Mn, Al and lanthanum (La). New P-inactivation agents are also being tested (Spears et al. 2013a).

The most popular and well-studied P-inactivation agent is alum, which has become the go-to method for reducing internal loading in eutrophic lakes in the Northern Hemisphere. Its effectiveness at reducing internal loading is usually around 80% and adequately dosed treatments last around 10 years (Hupfer et al. 2016; Huser et al. 2016a). Some of these results are described in Cooke et al. (2005) and in Welch and Cooke (1999). A more recent comprehensive assessment by Huser et al. (2016b) examined 114 lakes across the world in terms of longevity and effectiveness of aluminium addition to reduce sediment P release and restore lake water quality. Although alum is a very good P-inactivation agent, it is not a sediment capping agent. It settles onto the lake bed in a thin layer like a cap but is easily disturbed by water column turbulence (e.g. Gibbs et al. 2016) and bioturbation. Given sufficient time, it may become incorporated into a cohesive layer across the sediment surface.

The result of P-inactivation is the reduction in magnitude or elimination of cyanobacteria blooms (Cooke et al. 2005). In many treated lakes, the frequency of bloom occurrence is greatly reduced but there may still be occasional small blooms. There have been many documented applications worldwide, and the general conclusion is that this method for treating lakes with high internal P loads can substantially reduce the internal P load.

Phosphorus inactivation is a naturally occurring phenomenon where, under aerobic (oxidising) conditions, P can be sequestered onto the surface of metal oxides such as naturally occurring Fe and Mn in the sediments (Fig. 7.4). Unfortunately, the process is reversible. Under anoxic (reducing) conditions, Fe and Mn oxides dissolve and release the previously bound P back into the water column.

Products used for active sediment capping have usually been manufactured to target a specific nutrient in the lake, either N or P, or sometimes both. These include natural materials that have been processed into a finer grade than naturally occurs. Common active capping agents (Table 7.3) rely on geochemical interactions between the active ingredient and the targeted nutrient. This includes ionic exchange to allow the target nutrient to bind to the product surface. The products are best applied at a critical part of nutrient cycling when the greatest effect can be achieved

Table 7.3 Summary of active sediment capping agents

Capping agent	Active ingredient	Targeting	Comment	References
Iron	Fe_2O_3	PO_4^{3-}	Releases P under anoxic or high pH >9	Cooke et al. (1993), Gunnars et al. (2002), Immers et al. (2013), Bakker et al. (2016)
Alum	$Al_2(SO_4)_3$	PO_4^{3-}	Not affected by anoxia but releases P under high pH > 8.5, releases toxic Al^{3+} at low pH (<5)[a]	Cooke et al. (1993, 2005), Egemose et al. (2010), Gibbs et al. (2011), Reitzel et al. (2013b)
Aqual-P®	Al modified zeolite	PO_4^{3-} & NH_4-N	Stable to anoxia and pH (not tested above pH 8.9)	Gibbs et al. (2011)
Al/ Bentonite	Al modified bentonite	PO_4^{3-}		Egemose et al. (2010)
Phoslock™	La modified bentonite	PO_4^{3-}	Stable to anoxia and pH (<9.5), leaks low level La in low alkalinity (soft) water	Egemose et al. (2010), Gibbs et al. (2011), Spears et al. (2013b)
Allophane	Fe & Al oxides	PO_4^{-}	Natural clay product	Gibbs et al. (2011)
Zeolite	Molecular sieve	NH_4-N	Natural clay product	Huang et al. (2011)
Biozeolite	Zeolite biofilm	N	Modified zeolite	Huang et al. (2013)
Calcite	$CaCO_3$	PO_4^{3-}	Natural product begins binding P at pH > 8.5	Cooke et al. (1993)

[a]ANZECC (2000) water quality guidelines for aluminium have a 95% protection value for pH >6.5 of 55 mg m^{-3}. Only a 'low reliability' guideline value is provided for pH values <6.5

for the least effort and/or cost. For example, a granular P-inactivation agent that settles quickly, and therefore has little ability to bind SRP from the water column, will be ineffective if applied at the end of seasonal stratification even though there may be high concentrations of SRP in the anoxic hypolimnetic water. Applying the same product ground to a fine powder would result in slower settling and, therefore, longer contact time allowing the product to sequester and bind SRP from the water column. Applying the granular product after lake mixing, when the natural sequestration process with Fe and Mn oxides has precipitated the SRP into the sediments leaving very little in the water column, would be very effective as a sediment cap to block future P release. However, there needs to be a balance around particle size and particle density so that the capping material doesn't sink into the sediment, thereby reducing its efficacy as a P-inactivation agent (Gibbs and Özkundakci 2010). A finer grain size will be more effective at binding P because of the larger surface area per unit volume of the product applied. A P-inactivation agent such as alum that forms a floc has the ability to remove particulate P as well as SRP from the water column by trapping it in the floc.

While P-binding to Fe and Mn oxides is reversible depending on the redox state (Fig. 7.5), the P-binding process to Al and La is irreversible under normal lake conditions and the P is not released under reducing conditions in the anoxic hypolimnion. The consequence of this is that there is very little SRP available for algal growth (i.e. the water column phytoplankton has been driven to a P-limitation state), and any SRP that appears from dying algal cells is rapidly assimilated by all algae. In these circumstances, cyanobacteria no longer have a competitive advantage and, ignoring turbulence effects, they are less likely to become the dominant species under ‘normal lake conditions’ i.e. pH around neutral (7). Large shifts in pH can change the P-binding ability of metal oxy-hydroxides. This is discussed below in Sect. 7.4.

The other important consideration is that, except for flocs, P-inactivation agents only remove SRP from the water column, not particulate or organic P. This means that monitoring of the efficacy of a P-inactivation treatment has to focus on the change in SRP and not TP, because P attached to the P-inactivation agent becomes part of the TP and there will be no change until the sediment capping agent settles out of the water column. This is especially important when calculating the treatment load for a P-inactivation agent treatment. If there is no SRP in the water column, just TP, the most appropriate action would be to use a flocculant to settle the TP and then apply a sediment cap to lock the P in the sediment (see Sect. 7.5 below).

7.3.3 Specific P-Inactivation Agents

7.3.3.1 Allophane

Allophane is a natural P-inactivation agent. The active ingredients are iron and aluminium. It is produced during volcanic eruptions and is typically mined from ash shower deposits across the landscape in volcanic areas. Laboratory testing of allophane (Gibbs et al. 2011) found that it has a maximum P-binding capacity of around 17 g P kg^{-1} at high P concentrations but only around 5 g P kg^{-1} at ecologically relevant P concentrations (around 5–10 mg m^{-3}). This may be because the binding capacity is partially used by P leached from the surface soils and carried downwards by percolating rainwater loading the upper layers of the ash deposits with P. Being a natural product, there are minimal restrictions on its use in lakes. An example of a whole lake treatment effect is the response to about 2.2 million tonnes of allophanic ash deposited across Lake Taupo during the 1995–1996 eruptions of Mount Ruapehu (Fig. 7.9). This produced an average ash thickness of around 2–3 mm across the bed of the lake (Cronin 1996; Cronin et al. 1996).

Ash from the 18 September 1995 eruption had an immediate and long-term effect on the nutrients in Lake Taupo. SRP concentrations decreased to <0.5 mg m^{-3} for about 2 months before beginning to increase again (Fig. 7.10a). Concentrations of

Fig. 7.9 Ash cloud from Mount Ruapehu eruption on 17 June 1996, which drifted across Lake Taupo depositing around 2.2 million tonnes ash into the lake [photo: Anon.]

NO_3-N and NH_4-N also decreased to below detection levels, then NH_4-N increased while NO_3-N remained low for about a year (Fig. 7.10b), implying that nitrification had been suppressed. Ash from the second eruption series in July 1996 had almost no further effect on the lake water chemistry (Fig. 7.10) indicating that the allophane had come from the 6.5 million m^3 of acid water ejected from the volcanic crater lake in the initial 1995 eruptions, and no more acidic water with allophane was ejected in 1996.

The suppression of NO_3-N for a year after the first eruption while NH_4-N increased above expected concentrations implies that nitrifying bacteria were adversely affected by the ash or the allophane. An increase in NH_4-N release from sediment in cores treated with active sediment capping agents in laboratory studies (Gibbs et al. 2011) indicates that these agents may have an adverse effect on the microbial nitrifying communities in the sediment.

7.3.3.2 Iron

Although the P- to Fe-binding process is reversible under anoxic conditions, Fe can be a very effective P-inactivation agent in shallow, well-oxygenated lakes with low natural Fe content (Immers et al. 2013; Bakker et al. 2016). The main mechanism for release of SRP from the sediments is through disturbance by wind waves. SRP

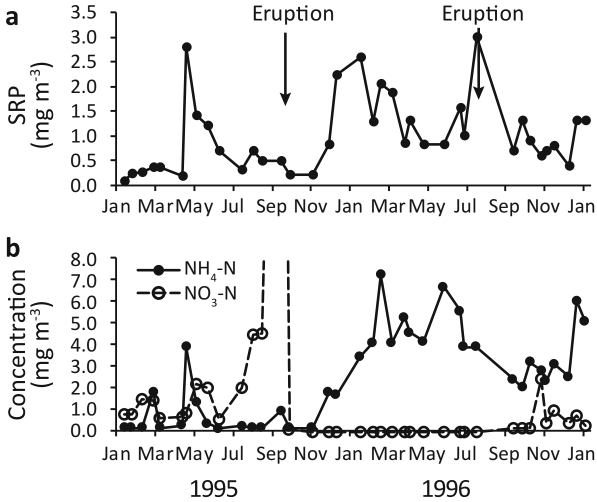


Fig. 7.10 Effect of volcanic ash showers on the water chemistry of Lake Taupo after the first eruption. (**a**) SRP concentration time-series data and (**b**) NH_4-N and NO_3-N concentration time-series data (Gibbs 2013)

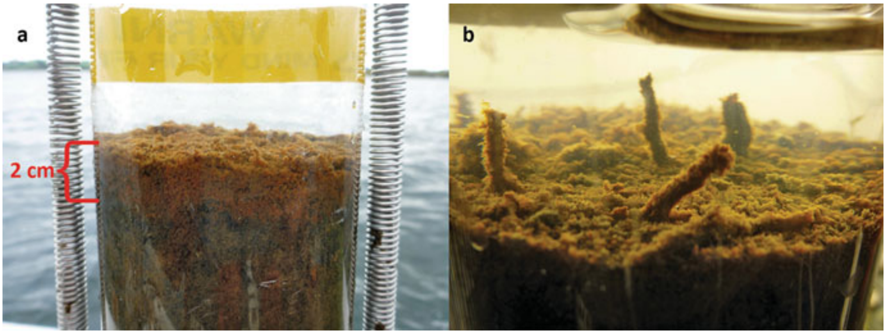


Fig. 7.11 Sediment–water interface in a 7-cm ID core tube (**a**) immediately on collection showing a 2-cm-thick layer of iron oxy-hydroxide gelatinous floc and (**b**) a closer view of the same core surface a few minutes later after the floc had collapsed, exposing the tubes of chironomid larvae [photos by M. Gibbs]

released by disturbance from the anoxic sediment below the sediment surface will be rapidly sequestered by the Fe sediment cap.

In lakes with high natural Fe, the Fe covers the lake bed with an oxy-hydroxide gelatinous floc layer which can be several mm to cm thick (Fig. 7.11) (authors observations from Lake Rerewhakaaitu, New Zealand). The floc is very delicate and easily disturbed: just removing the tube from a corer can cause the floc to collapse and expose the more rigid tubes used by chironomid worms to breathe and move through the floc layer. Because the floc has a consistency of a nepheloid layer, benthic biota would not survive if they could not lift themselves above the floc.

7.3.3.3 Alum

The most commonly used P-inactivation agent is aluminium sulphate, commercially known as alum (Table 7.3)—formula $Al_2(SO_4)_3 \cdot 14H_2O$. It has a maximum P-binding capacity of about 85 g P kg^{-1} product in saturating high concentration P solutions and needs to be used with a buffer in low-alkalinity water to maintain the pH at >6.3 during alum treatments. This is a flocculant in common use for the treatment of swimming pools to clarify water and to remove suspended solids from domestic water supplies in water treatment plants.

In lakes, alum performs a triple function of (1) binding any SRP in the water column, (2) flocculating suspended solids, and (3) becoming an active sediment cap, i.e. when the alum floc settles to the lake bed, any unused P-binding capacity remains active and will bind any P released from the sediments. There is an ageing effect on alum floc in the sediment (Berkowitz et al. 2006) which reduces the amount of P bound due to the reduction in surface area of the amorphous aluminium oxy-hydroxide as it transforms to crystalline Gibbsite. Note that because phosphate and arsenate have similar chemical structures and one can replace the other, alum will also bind with arsenic (As) thereby reducing its binding capacity for P uptake when there are elevated concentrations of As present, such as in the geothermally influenced lakes around the central volcanic plateau in the North Island of New Zealand.

Alum is typically supplied as a solution that can be diluted in water to a strength appropriate for the type of water being treated or the amount of P per unit area of the lake through the whole water column. For example, the areal mass of P in the top 4 cm of Lake Rotorua sediments is around 3–3.5 g P m^{-2}. To inactivate this amount of P would require sufficient alum to bind that much P at the binding capacity of the product. Calculations based on laboratory testing and a nominal P-binding capacity of 10 (10 g alum binds 1 g P) estimated that between 30 and 60 g alum m^{-2} would be required to treat Lake Rotorua (Gibbs and Özkundakci 2010; Gibbs et al. 2011).

In contact with water, alum forms a white aluminium oxy-hydroxide floc which slowly settles through the water column, removing suspended solids, including cyanobacteria and other algae, and provides an amorphous surface onto which the SRP can bind. When it reaches the sediment surface, it gradually forms a cohesive capping layer through which any P released from the sediment must pass and is, therefore, bound. The success of the treatment critically depends on the formation of that floc, and formation of the floc depends on the alkalinity of the water. Many upland South Island lakes have an alkalinity of >150 gEq (gram equivalents measured as g $CaCO_3$ m^{-3}) (Reid et al. 1999) and rapidly form a floc (Fig. 7.12). The floc in a mesocosm in a South Island lake took an average of 5 h ($n = 3$) to sink to the lake bed at 10 m depth, based on measurements made with a turbidity sensor on a vertical profiler (authors' observations, unpublished data). This is because as the floc coagulated the particles from the water column, the resultant aggregation became denser and settled faster. Without the aggregation effect, the floc has been observed to settle slowly over >10 days. The high alkalinity buffers the acidity of

Fig. 7.12 Floc formation after dosing at a rate of 70 g alum m^{-2} into a 10-m-deep mesocosm in a high alkalinity lake (alkalinity >400 gEq) [photo by C. Hickey, NIWA]

the alum so that the pH of the lake water does not change substantially during the dosing.

In contrast to the South Island lakes, many of the North Island lakes are essentially rainwater and have low alkalinity with values of <30 gEq (McColl 1972). Under these conditions, the formation of the floc is difficult and it is recommended that a buffer such as sodium bicarbonate or sodium aluminate is added with the alum. The use of a buffer has the additional benefit of keeping the pH of the lake water above 5.5. Below this pH, the alum dissociates and releases trivalent Al^{3+} ions, which are highly toxic to aquatic biota (see Sect. 7.4). If a buffer is not used, the alum dose rate must be reduced so that the pH does not fall below 5.5 (Paul et al. 2008).

Application of alum can be done by spraying the diluted raw solution onto the lake surface (Fig. 7.13). In a trial on Lake Okaro near Rotorua, an airboat was used so that the alum floc would form and not be broken up with the wash from an outboard motor propeller.

7.3.4 *Surface Application Issue*

A potential issue with surface application of active sediment capping agents is that there will be a period of time immediately after application when the product concentration potentially exceeds the maximum concentration permitted in the resource consent or regulatory guidelines. For example, if the lake is treated with 70 g alum m^{-2}, which is equivalent to 5.5 g Al m^{-2}, then as the alum settles through the first metre depth of water, the theoretical Al concentration is 5.5 g Al m^{-3}, seven times more than the acute water quality guideline for Al (0.75 g m^{-3}, US EPA 2009).

Fig. 7.13 Spraying alum onto the surface of Lake Okaro. The airboat was fitted with GPS navigation to control where the spray was applied [photo by M. Gibbs, NIWA]

This is generally recognised as an artefact of the application, and the subsequent monitoring would normally show the fully mixed Al concentrations below the acute and chronic toxicity guideline values.

7.3.5 Does Brief Exceedance of the Permitted Maximum Have an Effect?

Data obtained from monitoring the low alum dose experiment (0.6 g Al m^{-2}) on Lake Okaro (Paul et al. 2008) (Fig. 7.14) showed a temporary chlorophyll *a* increase, which implies that there may be an unexpected effect of the surface treatment. Explanations for the sudden increase in chlorophyll *a* concentration include:

(i) Growth stimulated by the alum dose. This is unlikely because the SRP concentrations in the epilimnion decreased to almost zero and the DIN:SRP ratio rose above 15, indicating possible P-limitation.
(ii) Reduction in grazing pressure. This is more likely because laboratory tests showed that zooplankton became trapped in the alum floc and were removed from the water column.

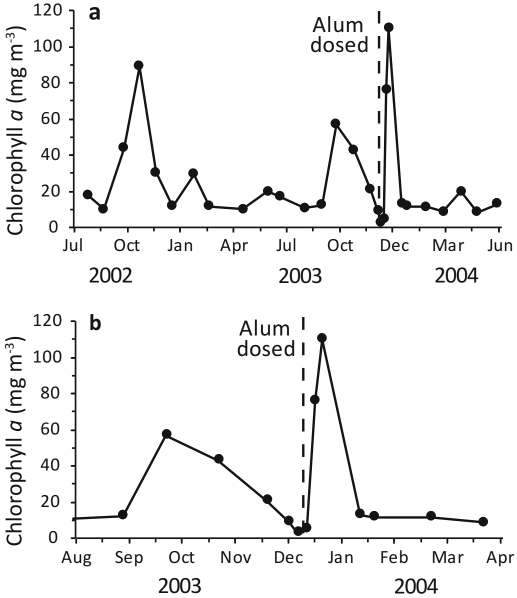


Fig. 7.14 Time-series chlorophyll a concentrations in the epilimnion of Lake Okaro (**a**) over the 2-year period around the low dose alum experiment and (**b**) showing an expanded view of chlorophyll a anomaly observed immediately after the dosing (Bay of Plenty Regional Council data)

(iii) Change in phytoplankton species. The dominant phytoplankton species changed from *Anabaena* sp. to *Ceratium hirundinella* at the same time (Paul et al. 2008), possibly in association with improved light climate.

Time-series data (Fig. 7.14a) for Lake Okaro show the typical seasonality of a spring algal bloom (Bay of Plenty Regional Council (BOPRC), unpublished data) with the uncharacteristic chlorophyll *a* spike immediately after the alum dose (Fig. 7.14b). The sudden rise and fall of the spike suggests removal of grazing pressure rather than the development of a nutrient induced algal bloom.

7.3.6 *Aqual-P*®

Aqual-P® was formerly known as Z2G1 in the testing process and publications (Vopel et al. 2008; Hickey and Gibbs 2009; Gibbs 2010; Gibbs and Özkundakci 2010; Parkyn et al. 2010; Gibbs et al. 2011; Clearwater et al. 2014). Aqual-P® is a zeolite-based product with an aluminium salt as the active ingredient (Table 7.3). It has a maximum P-binding capacity of around 23 g P kg^{-1} product in high concentrations but a markedly lower capacity of 1–1.7 g P kg^{-1} at ecological concentrations (10–100 mg m^{-3} in water column, as measured using binding isotherms, Olsen and Hickey 2015) and does not require a buffer. The P-binding affinity of Aqual-P® is high with flow through reactor columns containing granular material (2–4 mm), with capability to reduce P concentrations to <10 mg m^{-3} (Olsen et al. 2013). Zeolite has a natural affinity for NH_4-N meaning that an application of Aqual-P® has the capacity to adsorb both SRP and NH_4-N. Applied as a fine grain powder in a slurry, the product settles relatively quickly to the lake bed where it can form a cohesive capping layer. Because of the short water column contact time (a few hours), the product has a limited effect on any SRP in the water column, binding up to 25% of the free phosphate (Gibbs et al. 2011) which leaves it with a high capacity for binding P released from the sediment (Gibbs 2010; Gibbs et al. 2011; Gibbs and Özkundakci 2010). Consequently, Aqual-P® is designed to work on the sediment as a sediment capping agent and should be applied in winter or early spring before the sediment becomes anoxic and the P begins to be released from the lake bed sediments.

Aqual-P® has been subjected to testing for potential toxicity covering a wide range of expected sediment application rates (Parkyn et al. 2010; Clearwater et al. 2014). These studies have focused on four native species which are ecologically important and widespread in New Zealand lakes (freshwater crayfish, *Paranephrops planifrons*; freshwater mussels, *Echyridella menziesii* (previously *Hyridella menziesi*); common bullies, *Gobiomorphus cotidianus*; and fingernail clams, *Sphaerium novaezelandiae*). Each of these studies also included an alum floc treatment to benchmark test results against an alternative P-inactivation treatment. The results showed that application of Aqual-P® at rates up to 200 g Aqual-P® m^{-2} (1.7 mm cap thickness) or alum 344 g alum m^{-2} (>7 mm cap thickness) would have no significant adverse effects on freshwater fish or crayfish growth and survival, or mussel survival over a 2-month period. However, the results indicated that fingernail clam survival would be adversely affected by the highest alum application rate of 344 g alum m^{-2}, but application rates of 115 g alum m^{-2} (~7 mm cap thickness) or lower would not have any detectable effect on survival rates or morbidity (i.e. reburial activity).

Aqual-P® has been used as a sediment capping agent on Lake Okaro where it was applied from a barge initially as a 1–3 mm diameter granule using a fertiliser spreader (Fig. 7.15). This grain size was too large and the heavy granules sank into the soft sediment, reducing its efficacy to cap the sediments. Subsequently, it was applied as a slurry of <1 mm particle size using sub-surface lake injection hoses (Fig. 7.16), which effectively resulted in capping of the sediment. Sedimentation plates with artificial turf surfaces (Fig. 7.16d) to trap the particles (mat traps) were deployed at several locations across the lake to assess the thickness of the capping layer and evenness of the application across the lake bed.

The sequential trialing of different P-inactivation agents and application methods on Lake Okaro achieved a substantial reduction in the internal P load from the sediments (Fig. 7.17). Aqual-P® was also used pre-emptively on Okawa Bay, Lake Rotoiti, where it was applied as a slurry from a helicopter (Fig. 7.18) and may have prevented cyanobacteria from developing into a bloom.

Fig. 7.15 Applying granular Aqual-P® using a fertiliser spreader on a barge [photo Andy Bruere, BOPRC]

7.3.7 Phoslock™

Phoslock™ is a bentonite-based product with a lanthanum (La) salt as the active P-binding agent. Phoslock™ has a maximum P-binding capacity of around 12 g P kg^{-1} product and does not require a buffer. It disperses rapidly in water but settles slowly and, consequently, has a long contact time. Because its main action is in the water column, it should be applied in summer when the DRP concentrations in the lake water are maximal. Under these conditions, Phoslock™ has the capability of binding up to 95% of the DRP in the water column (Gibbs et al. 2011). Once it settles to the lake bed, any unused binding capacity remains available to sequester any P released from the sediment under anoxic conditions. Under normal pH conditions (pH ~7), the P is strongly bound to the lanthanum and is not available to algae for growth. Phoslock™ is beginning to be widely used overseas (Spears et al. 2013b; Copetti et al. 2016) and has been used in one lake trial in New Zealand, Lake Okareka (Landman and Ling 2006; Landman et al. 2007; McIntosh 2007).

Because it is a relatively new commercial P-inactivation product on the market, Phoslock™ has been subjected to much scrutiny as to its efficacy and potential toxicity over and above the expected independent ecotoxicity testing for registration (Barry and Meehan 2000; NICNAS 2001, 2014; Clearwater 2004; Clearwater and Hickey 2004; Martin and Hickey 2004; Watson-Leung 2009). Apart from these data, very little is known about the toxicity of La in the aquatic environment and much of

Fig. 7.16 (**a–c**) Applying <0.5 mm Aqual-P® as a sub-surface slurry injection from a barge. (**d**) Aqual-P® coverage of an artificial turf substrate used to check the evenness and thickness of the Aqual-P® cover across the lake bed [photos **a** and **b** by Andy Bruere, BOPRC; photos **c** and **d** by M. Gibbs, NIWA]

what is known is summarised in tables in Herrmann et al. (2016). A potential issue with interpretation of the LC50 results depends on whether the tests included an addition of SRP or ethylenediaminetetraacetic acid (EDTA), both of which will sequester any free La^{3+} ions in the water column (Martin and Hickey 2004). With low SRP concentrations and little humic material, free toxic La^{3+} ions remain in the water column comparable with toxic Al^{3+} ions but at a higher pH (Fig. 7.19). Consequently, Phoslock™ is likely to be less suitable for use in the low alkalinity waters of many of the North Island lakes in New Zealand, where it might have otherwise been used.

This conclusion is consistent with the Spears et al. (2013b) evaluation of 16 case study lakes. Their modelling indicated that the concentrations of La^{3+} ions will be very low (<0.0004 mg L^{-1}) in lakes of moderately low-to-high alkalinity (>800 gEq), but higher (up to 0.12 mg L^{-1}) in lakes characterised by very low alkalinity (<80 gEq). They also concluded that, while the effects of elevated La^{3+} concentrations following Phoslock™ applications in lakes of very low alkalinity requires further evaluation, Phoslock™ was a useful addition to the lake restoration toolbox.

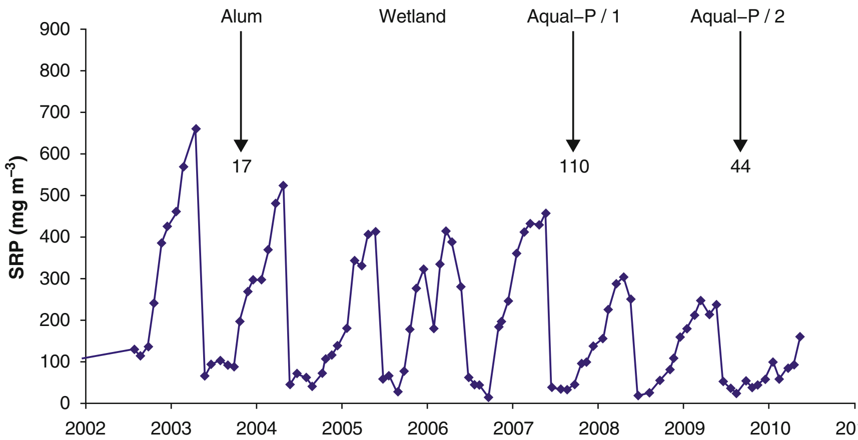


Fig. 7.17 Time series of hypolimnetic SRP concentrations in Lake Okaro covering the experimental period starting with the low dose alum treatment (December 2003), the installation of a wetland on the main inflow and the two Aqual-P® treatments: 1 = 1–3 mm granules, 2 = <1 mm slurry. The numbers below the arrows are the mass in tonnes applied at those times (Bay of Plenty Regional Council data)

Fig. 7.18 Application of Aqual-P® <1 mm slurry by helicopter from a monsoon bucket fitted with a fertiliser spreader. The helicopter position was controlled using GPS navigation [photo: Graham Timpany, NIWA]

There are three potential disadvantages of using Phoslock™: (1) Being a bentonite clay-based product, it is very slow to settle and may take up to a month before the water clears completely. However, where the lake has high turbidity this may not be an issue. (2) If the lake has large weed beds, Phoslock™ can settle on the leaves and

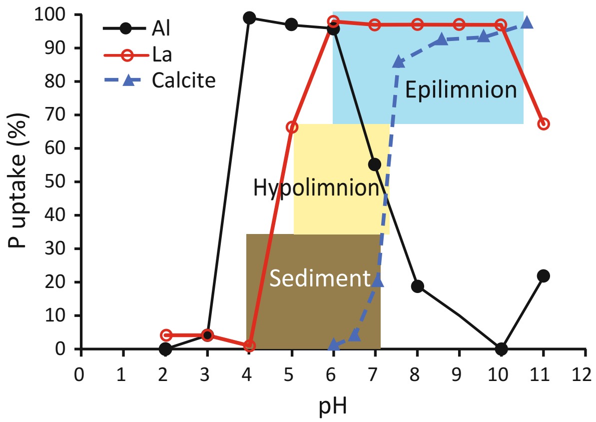


Fig. 7.19 pH effects on SRP precipitation with Al and La salts, and calcite adsorption (Al and La data redrawn from Peterson et al. 1976, calcite data extracted from Olila and Reddy 1995). Coloured blocks indicate the typical pH ranges found in different parts of a lake

not reach the sediment. (3) There are questions about the potential toxicity of Phoslock™ to zooplankton and fish in low alkalinity waters (NICNAS 2001, 2014; Martin and Hickey 2004; Chapman et al. 2009).

Phoslock™ has been found to leak La ions from the bentonite carrier in low alkalinity water (Clearwater 2004; Martin and Hickey 2004; Gibbs et al. 2011). Because La binds rapidly with SRP in the water to form rhabdophane, a hydrous phosphate of La ($La(PO_4){\cdot}H_2O$), which is insoluble and non-toxic, this leakage has been considered to be not a problem. However, if there is no SRP in the water for the La to bind with, then La^{3+} ions may cause toxicity to juvenile fish (Martin and Hickey 2004) and zooplankton.

The surface application issue raised for alum (*see* Sect. 7.3.4 above) also applies to Phoslock™ treatments in low alkalinity lakes. For example, Deep Creek water supply dam in New South Wales, Australia, experienced a fish kill of 2–3 cm sized fish after an approved treatment with Phoslock™ (Chapman et al. 2009; Pablo et al. 2009).

7.3.8 Calcite

Calcium carbonate (calcite, $CaCO_3$) or calcium hydroxide (lime, $Ca(OH)_2$) can be added to lakes as phosphorus precipitants (Dittrich et al. 1997, 2011). Calcite sorbs phosphorus especially when pH exceeds 9.0 and results in significant phosphorus removal from the water column. Phosphate adsorbs at the calcite surface or binds inside a crystal during the formation of $CaCO_3$ precipitate when calcium hydroxide is applied (Kleiner 1988; House 1990). Various calcite forms have also been reported for their potential use as active barriers for sediment capping to reduce phosphorus release from sediments (Hart et al. 2003).

Olsen et al. (2016) undertook adsorption isotherm measurements to screen a range of regionally available calcium carbonate materials from around New Zealand. The materials included crushed limestones and shells (Pacific oysters, flat oysters, and green-lipped mussels). Adsorption isotherm testing of materials using a soft water containing P at 10 mg L^{-1} via batch experiments provided thermodynamic data for comparing a range of different materials with potential for removal of phosphate from natural waters. The initial pH was 9.0 in order to standardise P-binding in the optimal range. Mussel shells showed better P-uptake capacities (0.018–0.12 g kg^{-1}) at low phosphate concentrations of $\leq$1 g m^{-3} compared with oyster shells (0.007–0.060 g kg^{-1}). P-uptake capacity was significantly reduced when larger particle size fractions were tested and reflects the change in surface area for P-adsorption. A selection of four ground limestones from the North and South Islands displayed very similar performance with P-uptake ranging between 0.029 and 0.110 g kg^{-1} at low phosphate concentrations of $\leq$1 g m^{-3}.

Olsen et al. (2016) concluded that all of the regionally sourced calcium carbonate materials (i.e. shells, limestones) tested had a markedly lower P-binding capacity compared with commercial products such as Aqual-P® and Allophane (Olsen et al. 2013, 2016). These commercial products have P-binding capacities ranging between 1.2 and 1.3 g kg^{-1} at equilibrium [P] of 0.10 g m^{-3}, which is approximately 30 times more effective than raw shells. At phosphate concentrations of 1.0 g m^{-3}, the P-uptake capacity of Aqual-P® and Allophane are 1.7–1.8 g kg^{-1} and 2.7 g kg^{-1}, respectively. The commercial products were also more effective at removing dissolved phosphate at elevated phosphate concentrations, compared with raw shell materials.

The application of a calcite active sediment barrier has been investigated to prevent P release from eutrophic lake sediments (Berg et al. 2004). The calcite saturation of the water predetermines how the P is bound. In waters supersaturated with calcite (as $CaCO_3$), adsorption and simultaneous co-precipitation with calcite are the main processes for P-binding. In addition, precipitation of Ca–P compounds occurs on the surface of calcite seed crystals due to the decrease in the interfacial nucleation energy. The capacity of P-binding is greatly influenced by the physical properties of the calcite grains; this includes grain size, specific surface area, and roughness of the grain surface. In laboratory short-term batch and long-term sediment incubation experiments, Berg et al. (2004) found that a 1-cm-thick barrier of Rohrbach calcite (a crushed Jurassic limestone with a specific surface area of 4.3 m^2 g^{-1} and a micro porosity of 7%) resulted in an 80% reduction of the P flux from sediment for at least 2–3 months, whereas a quartz sand barrier, containing 2 wt% of highly active calcite such as precipitated lime 2–4.5 cm thick, prevented the release of P from eutrophic lake sediments for at least 7–10 months.

Lin et al. (2011) also used a series of batch and sediment incubation experiments to evaluate the use of active barrier systems with calcite/zeolite mixtures for sediment capping to simultaneously prevent SRP and ammonium (NH_4^+) release from eutrophic lake sediments under anaerobic conditions. This study tested natural calcite and various zeolites (natural, NaCl-pretreated and $CaCl_2$-pretreated zeolites) and found that the calcite was efficient for the removal of SRP in aqueous solution and the zeolite was an efficient adsorbent for the removal of NH_4^+ from aqueous solution. This study also found that, while the mixture of calcite and natural zeolite could block the release of SRP from sediment, the mixture of calcite and $CaCl_2$-pretreated zeolite was the most effective capping material. They concluded that this mixture supplied additional Ca^{2+} ions through a Ca^{2+}/NH_4^+exchange to improve the ability of the capping material to immobilise P release from the sediments.

Hupfer et al. (2000) investigated the mechanical resuspension of autochthonous calcite (seekreide), which was effective in laboratory incubations but failed to control internal phosphorus cycle in a eutrophic lake at field scale. Their results showed that there was insufficient calcite to bind the P release from the sediment and that the majority of the P removal was associated with iron oxyhydroxides.

Unlike alum and lanthanum, where P-binding occurs at circum-neutral to moderately high pH, calcium binding of P occurs at high pH. At high pH, and concentrations of Ca^{2+} and soluble P, hydroxyapatite is formed.

$$10\,CaCO_3 + 6\,HPO_4^{2-} + 2\,H_2O \rightarrow Ca_{10}(PO_4)_6(OH)_2 + 10\,HCO_3^-$$

Hydroxyapatite has its lowest solubility at pH $>$ 9.5 and binds phosphorus strongly at high pH (Cooke et al. 2005). However, if the pH falls, the solubility sharply increases releasing the bound P. This occurs especially in zones with intense bacterial respiration near the sediments (Driscoll et al. 1993).

Typical application doses of lime are in a range of 25–300 g Ca m^{-2} (Søndergaard et al. 2002). The advantage of lime is its low price and non-toxicity. However, adverse effects to aquatic organisms may occur because application of lime increases the pH of water (Miskimmin et al. 1995; Yee et al. 2000). In soft-water lakes, the pH may easily exceed 11 (Zhang and Prepas 1996). The lime treatment also temporarily increases turbidity in the overlying water column as the calcite precipitates. During the precipitation, phase calcite can effectively sequester P from the water column and may continue to do so in the sediments, depending on the pH (Fig. 7.19) (Olila and Reddy 1995).

From a practical perspective, the longevity of Ca is short and requires retreatment yearly (Zhang and Prepas 1996; Cooke et al. 2005). In low alkalinity water, this treatment may not be effective.

7.4 pH Effects and Speciation

Redox and pH potential have major effects on the transformation of minerals that bind P in the sediment between solid phase and soluble phase (Moore and Reddy 1994; Reitzel et al. 2013b). Consequently, there are specific requirements for the use of alum in natural waters. Because alum is acidic (the 47% solution has a pH of 2.1), it requires to be buffered to a pH of around 6.5–7 where the waters have low alkalinity (Hickey and Gibbs 2009) to allow the floc to form. The alkalinity of many of New Zealand's largely rainwater filled North Island lakes is less than 40 gEq. The nominal value above which little or no buffer is required for normal dosing is 80 gEq. If alum is used on a low alkalinity lake, a buffer such as soda ash (sodium carbonate) or sodium bicarbonate needs to be mixed with the alum to ensure the formation of a floc before dosing. Without the buffer, there is a high likelihood that the pH would fall below 6, no floc would form and toxic trivalent aluminium ions (Al^{3+}) could be released into the water column. A comparable problem exists for the La in Phoslock™ but with different pH thresholds (Fig. 7.20, Peterson et al. 1976).

If there is no SRP to bind to the trivalent Al or La ions in the lower pH water, these ions will remain in the water column causing a potentially toxic environment. The pH ranges indicated for the sediment and hypolimnion are largely determined by decomposition processes and the production of hydrogen sulphide (H_2S) and carbon dioxide (CO_2). Figure 7.19 might be interpreted to indicate that it would be unwise to apply Phoslock™ as a sediment capping agent in a thermally stratified lake with low pH. The data in Fig. 7.19 might also be interpreted to indicate that alum should not be applied in a lake where the pH in the epilimnion is above 9, because there will be very little P-binding capacity at that pH. While it is true that the treatment solution can be buffered to the pH range where the floc will form, as the floc disperses into the high pH water it will dissolve and have little or no binding capacity. This dissolution is irreversible, and a floc does not form again as the pH lowers in winter and gibbsite

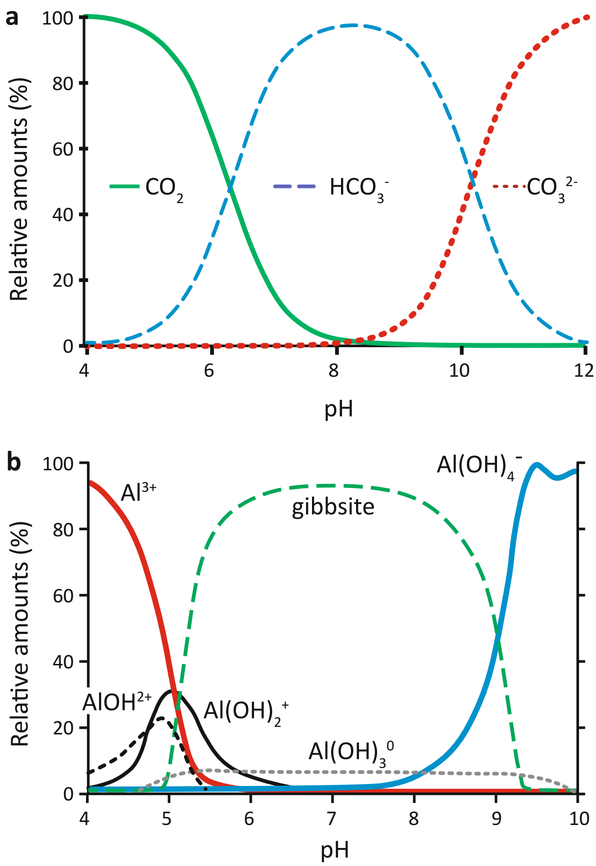


Fig. 7.20 (**a**) The relative proportions and transition points of carbon dioxide, bicarbonate and carbonate in water at different pH. (**b**) Speciation of aluminium in water at different pH with relative proportions and transition points [redrawn from Pedersen et al. (2013), (**a**); and Gensemer and Playle (1999), (**b**)]

is formed instead (Fig. 7.20b). Reitzel et al. (2013b) suggest that pH can influence aluminium dissolution from sediment treated with aluminium.

The inverse relationship for P uptake between Al and calcite also indicates that treatment of lake sediment with both binding agents might provide a broad pH spectrum treatment to suppress P release from the internal load in the sediments (Fig. 7.19), although retreatment with calcite may be needed annually.

7.4.1 Causes of High pH

How likely is it that the pH of the epilimnion would exceed 9?

Just as adding CO_2 to water will lower the pH due to the production of carbonic acid, removing the carbonate and bicarbonate will cause the pH to rise.

$$\underset{\text{Dissolved } CO_2}{CO_2 + H_2O} \leftrightarrow \underset{\text{Carbonic acid}}{H_2CO_3} \leftrightarrow H^+ + \underset{\text{Bicarbonate}}{HCO_3^-} \leftrightarrow H^+ + \underset{\text{Carbonate}}{CO_3^{2-}}$$

For example, as aquatic plants, both algae and macrophytes (water weeds), photosynthesise, they consume the carbonate from the water and the pH rises (Fig. 7.20a). Plants that can only utilise CO_2 will raise the pH to a maximum of around 8.5–8.7. Many of New Zealand's native plants and green algae are in this category. Plants that are bicarbonate adapted, such as most exotic macrophyte weed species and cyanobacteria, can push the pH almost to 11 (Fig. 7.20a). Under these conditions, native species become carbon limited, can no longer photosynthesise for growth, and may die.

As the pH changes from 4 to 10, the solubility of aluminium (and lanthanum, Fig. 7.19) changes from nearly all as toxic trivalent Al^{3+} to all as insoluble non-toxic aluminium oxy-hydroxides then back to non-toxic soluble aluminium hydroxides (Fig. 7.20b). At high pH > 9, other chemical species such as ammonia (NH_3) become toxic and the high pH itself has a toxic effect on aquatic biota.

The importance of knowing the pH before dosing a lake with alum is illustrated in the time-series data from Lake Okaro examining changes that occurred following a flood event which by-passed the wetland sediment trapping system and the subsequent further treatment with small doses of alum and Aqual-P® (Fig. 7.21). Following the second Aqual-P® treatment in 2009, hypolimnetic SRP concentrations were low, as were the chlorophyll *a* concentrations in the lake. This successful management intervention was destroyed by a storm event in 2011 which took sediment-laden flood water past the wetland sediment detention system directly into the lake. This covered the lake bed with P-rich fine sediment burying the active sediment capping layer negating its ability to sequester P release from the sediment in summer 2011 and 2012. In winter 2011, an application of Aqual-P® was made at the beginning of the stratified period (Fig. 7.21, Aqual-P®). This small dose slightly reduced the release of SRP from the sediment (Fig. 7.21A) but had little or no effect on the cyanobacteria concentrations during spring (Fig. 7.21B). This was because the treatment had been applied when the epilimnion pH was >9 and Al has no P-binding capacity above pH 9. A dual buffered alum treatment in winter 2012 also had little effect on the cyanobacteria because it was applied after the pH had risen. It did have an effect on the SRP release from the sediment and presumably had settled through the thermocline to form a weak sediment cap (Fig. 7.21, Alum 1).

At the start of that study, the estimated releasable P (measured as total P by persulphate digestion) in the sediment was 3.2 g P m^{-2}. After the first dose of 110 t of Aqual-P®, the amount of releasable P had reduced to 0.94 g P m^{-2} (Gibbs 2010). The treatment dose of 44 t Aqual-P® was calculated to be sufficient to sequester that amount of P. However, the subsequent flood event added an estimated 280 kg P to the ~30 ha lake, which equates to approximately 0.9 g P m^{-2}, effectively negating that treatment. This demonstrates the need for in-lake remedial actions to be coupled with effective catchment management strategies.

In 2013, two alum doses were applied before the pH rose above 8.5 (Fig. 7.21, Alum 2). Although the small dose had minimal effect on the SRP release from the sediment, it stripped all the SRP from the epilimnion, turning the water P-limited water to phytoplankton growth, and the cyanobacteria bloom was only a minor event

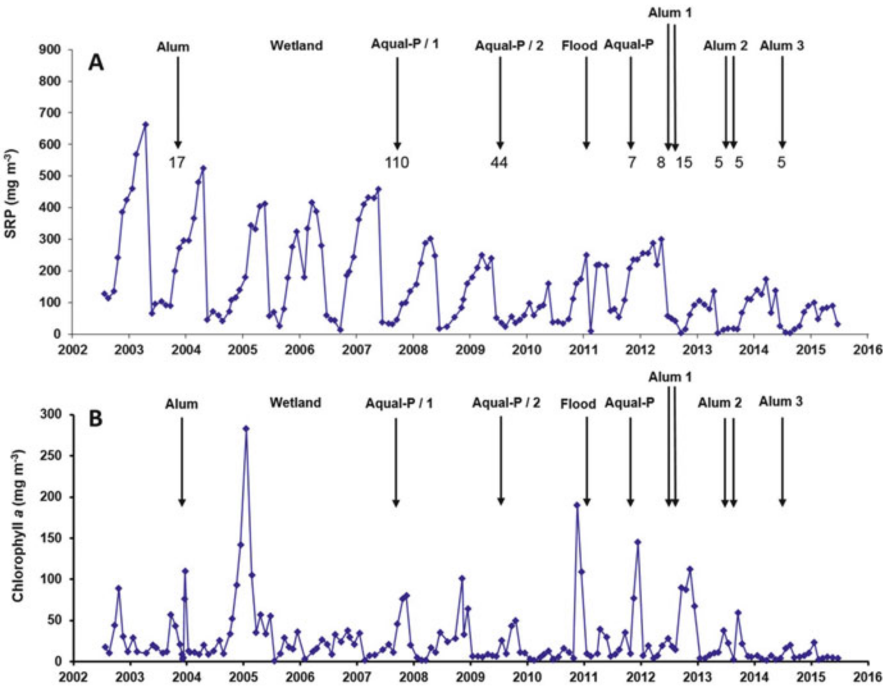
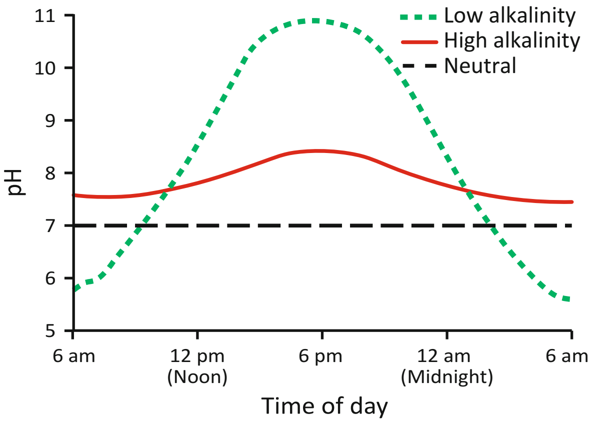


Fig. 7.21 Time series of (**a**) hypolimnetic SRP concentrations and (**b**) epilimnetic chlorophyll a concentrations in Lake Okaro, covering the full experimental period and showing the effects of each treatment. The numbers below the arrows in part (**a**) are the tonnes of specified material applied at that time (Bay of Plenty Regional Council unpublished data)

Fig. 7.22 Stylised representation of relative pH changes in low and high alkalinity lake water due to plant photosynthesis over a 24-h period in summer



(Fig. 7.21B). This treatment timing was tried again in 2014 with the same result—the cyanobacteria bloom did not develop. The water quality in the lake improved from eutrophic to meso-eutrophic without significant cyanobacteria blooms in summer.

7.4.2 Timing

The Lake Okaro case study demonstrates that the management of cyanobacteria blooms is likely to depend on intervening at the correct time. In this case, timing was guided by watching the pH rise (an indication of algal growth) and dosing the lake when the pH reached 8.5. Unbuffered alum was used and this lowered the pH slightly improving the efficacy of the P-binding.

While changes in the pH of the lake may be followed over a time scale of weeks to months, the pH can also change on a daily cycle such that the pH is lowest in the morning after a long dark period without photosynthesis and rises during the day to a maximum around mid-afternoon due to photosynthesis (Fig. 7.22). These changes are greatest in low alkalinity lake waters, which have little buffering capacity. Consequently, applying alum as a flocculant/P-inactivation agent would best be done in the morning and not in the afternoon.

7.5 Flocking and Locking

Recent management strategies using in-lake interventions are including different approaches to the use of the flocculation and sediment capping procedures. In combination, flocculating can precipitate the suspended solids which contain the P and algal matter; then these can be locked in the sediment with an appropriate sediment capping agent, for example, Lake Rauwbraken, the Netherlands (van

Oosterhout and Lürling 2011; Lürling and van Oosterhout 2013), Funnel Reservoir, Brazil (Noyma et al. 2015), and Lake De Kuil, the Netherlands (Waajen et al. 2015). In these studies, the suspended material including P was stripped from the water column with a low dose of poly aluminium chloride (PAC), or the cyanobacteria was removed with red clay and then the sediments were locked with a capping layer of Phoslock™. This floc and lock treatment also strongly reduced P-release from the sediment and had a strong positive effect on water quality: filamentous cyanobacteria were removed and chlorophyll *a* concentrations dropped to very low levels (2 mg m^{-3}). However, Al, La and suspended clay concentrations were temporally markedly elevated. The elevated La concentrations may have been the result of applying Phoslock™ to water with very low SRP concentrations while the increase in Al would be associated with the bentonite clay support material in Phoslock™. Within 1 week following the treatment, *Daphnia galeata* disappeared from the water column for 3 months. This was attributed to a combination of the physical effects due to flocs, grazing inhibition by flocs and clay, very low food concentrations and absence of refuge from predation. The effects were temporary and *Daphnia* recovered from the treatment.

In another case study by Mason et al. (2005), PAM and alum were used to reduce phosphorus-laden sediments and SRP from the inflows of predominately nutrient-rich agricultural wastewater to the Salton Sea in California (area 980 km^2) by treating the tributaries. Preliminary laboratory jar tests determined PAM effectiveness as 2 mg L^{-1} for turbidity reduction with cationic > anionic = nonionic. Cationic PAM was the most effective at reducing turbidity at higher speeds. Alum at 4 mg Al L^{-1} reduced turbidity in low energy systems by 95% and was necessary to reduce SRP, which comprised 47–100% of the total P concentration in the tributaries.

When PAM was added with alum, the anionic PAM became ineffective in aiding flocculation. Mason et al. (2005) recommended the use of nonionic PAM at 2 mg L^{-1} with alum at 4 mg Al L^{-1} to reduce suspended solids in higher energy systems and reduce soluble P by 93%. Note: cationic PAM is highly toxic to aquatic organisms whereas anionic and nonionic PAMs are essentially non-toxic. Dose concentrations of PAMs need to be tailored to the concentrations of suspended sediment present in the system being treated.

As both anionic (or nonionic) PAM and alum are flocculants, the addition of both together into the inflow tributary would give a strong flocculation effect as well as P-binding effect.

In contrast to the Salton Sea study, alum at 1 mg Al L^{-1} has been added to two inflow streams to Lake Rotorua, a large (80 km^2) shallow (mean depth 12 m) temperate polymictic lake in the central volcanic plateau of New Zealand, to reduce the SRP input load on the lake from the naturally elevated SRP in these streams. Treatment began in 2007 with one stream; then a second stream treatment was started in 2009. The floc formed in the streams and reduced the SRP concentrations to very low concentrations. As this input load was estimated to be around 30% of the SRP load on the whole lake, that reduction had a major effect on the water quality in Lake Rotorua improving it from supertrophic–eutrophic, with a trophic level index

(TLI, Burns et al. 2000) of around five, to mesotrophic–eutrophic with a TLI of 4.2. The SRP concentrations in the upper and lower lake water column decreased substantially, and the overall total chlorophyll *a* concentrations in the upper water column decreased (Fig. 7.23).

The key mechanism(s) for the success of this treatment technique is unclear with several possibilities including:

- The 30% decrease in SRP input may have reduced the algal biomass to below a tipping point where there is insufficient organic matter settling to the sediment to cause anoxic SRP releases during the short periods of summer stratification.
- Catchment mitigation strategies may have augmented the SRP reduction from the alum dosing to reduce the internal P load released from the sediment.
- Residual alum floc may have been carried out into the lake and sequestered additional SRP from the lake water column.
- Residual alum floc may have been carried from the stream inflows into the lake and deposited in the deeper parts effectively capping those sediments.

While the hydraulic mechanism is feasible, as demonstrated by Gibbs et al. (2016), the occurrence in summer 2014–15 of a substantial sediment P release during an extended period of thermal stratification that resulted in an anoxic hypolimnion indicates that it is taking longer to establish anoxia after thermal stratification. This means that either of the first two mechanisms above are more likely as the sediment oxygen demand has reduced substantially and the lake is recovering.

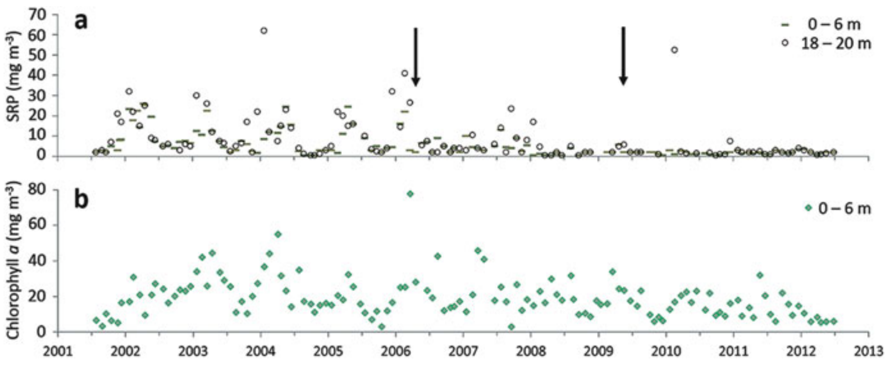


Fig. 7.23 Time series of (**a**) SRP concentrations in the upper (0–6 m) and lower (18–20 m) depth layers; and (**b**) total chlorophyll *a* concentrations in the upper (0–6 m) depth layer at a central monitoring site in Lake Rotorua. Arrows indicate approximately when dosing of the two streams began: the first in 2006 and the second in 2009. Data from Bay of Plenty Regional Council, redrawn from Hamilton et al. (2014)

7.6 P-Binding Capacity

The P-binding capacity of a P-inactivation agent is the maximum amount of P that can be sorbed by P-inactivation agent from a solution of SRP and has the units g P kg^{-1}. The P-binding capacity depends on a number of factors, including concentration of SRP, temperature, pH, reaction time, and particle size of the P-binding material. The maximum P-binding capacity may be determined by agitating the agent for several days in a strong SRP solution, which saturates all the uptake sites on and inside the P-inactivation agent structural matrix. This is not the same as P-binding efficacy which reflects the ability of the P-inactivation agent to sorb SRP at ecologically relevant concentrations and ambient conditions of pH. For example, allophane has a maximum P-binding capacity of 16.3 g P kg^{-1} in strong SRP solutions (Table 7.4) but can only sorb about 2–5 g P kg^{-1} at ecological SRP concentrations (Gibbs et al. 2011; Olsen and Hickey 2015). A batch isotherm adsorption technique can be used to characterise the product and determine both the P-binding capacity and the adsorption affinity in relation to ambient P concentration and effects of other key factors (Olsen and Hickey 2015; Olsen et al. 2016).

Gibbs et al. (2011) determined the maximum P-binding capacity of each granular product by suspending 5 g of each in a litre of 100 g m^{-3} solution of phosphate measured as P and buffered to specific pH. Rather than measuring the reduction in P in the solution, the granules were collected by filtration and washed briefly several times with deionised water to remove any surplus P solution. The granules were then dried and the maximum amount of P adsorbed by each product was measured by difference from original material by ICP-mass spectrometry after digestion in aqua regia. The P-binding capacity of alum was taken from the literature.

Note that the binding capacity of all of these P-inactivation agents is markedly lower at P concentrations relevant to natural waters—as discussed earlier. The changes in binding capacity with pH at high P concentration (Table 7.4) are likely to be indicative of those which will occur at P concentrations present in natural waters; however, tests should be undertaken for specific applications using site-specific water quality and physico-chemical data.

Table 7.4 Maximum P-binding capacity (g P kg^{-1}) of four P-inactivation agents measured in 100 g P m^{-3} solution at three different pH values

	pH			
P-inactivation agent	6.1	7	8.9	Active ingredient
Alum	95[a]	47[a]	12[a]	Aluminium sulphate
Phoslock™ (flakes)	11.6	11.3	15.1	La salt on bentonite
Aqual-P (1–3 mm)	21.5	21.5	11.6	Al salt on zeolite
Allophane (<2 mm)	16.3	16.0	9.5	Al + Fe crystalline

Data from Gibbs et al. (2011)
[a]Literature estimates

7.7 Calculating Dose Rates

The method for calculating the treatment rate for the application of alum is described in detail in several publications (Kennedy and Cooke 1982; Cooke et al. 1986, 2005). An adaptation of this method was used for estimating treatment rates for the four P-inactivation agents (products) discussed above, i.e., alum, Phoslock™, Aqual-P®, and allophane, in a sediment core incubation study (Gibbs et al. 2011).

The treatment rates (dose) are calculated based on the areal mass of total releasable phosphorus in the top 4 cm of sediment plus the areal mass of SRP in the overlying water. The total releasable P is almost the same as the TP content as measured by persulphate digestion, or by ICP-mass spectrometry using a total recoverable extraction (e.g. US EPA total recoverable method 200.2 Method B). Consequently, the sum of concentrations in the top four 1-cm thick slices from a sediment core of known cross-sectional area can be used to estimate a first approximation for the areal sediment P mass. This will be an over estimate with the excess accounting for the small amount of SRP that is likely to diffuse up from the deeper sediment. The TP content of the top 10 cm could also be used (Cooke et al. 2005).

Calculation of the treatment rate is as follows: If, in a hypothetical shallow lake, the SRP concentrations in a 10 m mean depth water column averages 0.06 g m^{-3} and the top 4 cm of sediment hold 2.7 g P m^{-2}, the total areal mass would be 3.3 g P m^{-2}. Because the Al–P-binding has a stoichiometric molar ratio of 1:1, the amount of alum required must be estimated from the aluminium content of the alum. In New Zealand, the actual application rate should allow for a binding efficiency factor of about four but in North America, where there are other parameters that can affect the binding capacity in the sediment, this factor may be as high as 12 (Cooke et al. 2005). For the New Zealand condition, at a binding efficiency factor of four and an alum:aluminium molar ratio of 12, it would require 13.2 g aluminium m^{-2} to sequester the 3.3 g P m^{-2}, which equates to 158.4 g alum m^{-2}. Alum is supplied in the form of a 47% liquid, which would take the dose rate to 337 g m^{-2} of the liquid alum. Therefore, to treat a 30 ha lake (e.g. Lake Okaro) with an areal P mass of 3.3 g P m^{-2} would require about 100 tonnes of liquid alum. If the Cooke et al. (2005) binding efficiency factor of 12 is used, it would require about 300 tonnes of liquid alum to sequester the estimated 990 kg P.

A buffer addition to prevent excessive pH reduction with the alum addition could be provided with soda ash (sodium carbonate) or sodium bicarbonate, which is also used in water treatment plants. In New Zealand, sodium carbonate is preferred to using sodium aluminate buffer as it minimises the amount of aluminium being added to the low alkalinity lake water. The amount of buffer required is normally about twice the amount of alum, i.e. about 200 tonnes in the Lake Okaro example, but the actual amount needed should be checked using the alkalinity of the lake water before treating the lake. The preferred time for treatment would be mid-summer when SRP concentrations in the hypolimnion of the lake water were at maximum levels. For Lake Okaro, the application would need to be directly into the hypolimnion (below

6 m) because the epilimnion (0–5 m) has a pH > 9 at that time of year and the alum floc would not form, or would dissolve releasing the SRP (Fig. 7.20).

The laboratory incubation of the four sediment capping agents on sediment cores from Lake Rotorua, which has a similar areal P mass to Lake Okaro, found that the P-binding capacity (Table 7.4) also needed to be included in the treatment calculation (Gibbs et al. 2011). The results of this correction show that proportionally more P-inactivation agent is required as the P-binding capacity decreases (Fig. 7.24).

More information on these four P-inactivation agents and their use can be found in Hickey and Gibbs (2009).

Lake Rotorua has not received a direct alum or Aqual-P® treatment as applied in Lake Okaro. However, as described above, an alternative indirect approach was to treat two major inflow streams, the Utuhina and Puarenga, which have high natural SRP concentrations derived from the local geology (Timperley 1983) and, in combination, account for about 30% of the SRP load on the lake.

Consequently, calculation of the dosing rates considers only the SRP concentrations in these streams rather than the areal mass of P in the sediments. Even this calculation was not straight forward, and a dosing strategy was gradually developed that achieved the required SRP reduction and water quality improvement in Lake Rotorua. Dosing alum in the Utuhina Stream began at a rate of about 1 g Al m^{-3} s^{-1} and the dose was varied daily to account for changes in the TP in the lake. The mean daily dose rates were around 100 kg Al day^{-1} in the Utuhina Stream and 75 kg Al day^{-1} in the Puarenga Stream (Hamilton et al. 2014). This means that a total of around 63.6 t Al were used each year and this equates to 1630 t liquid alum each year.

Based on the sediment P mass of 3.2 g P m^{-2} and Lake Rotorua's area of around 80 km^2, the amount of releasable P in the sediments was about 256 t. At a 1:1 molar binding ratio for Al:P, it would require a minimum of 6536 t liquid alum to sequester the initial mass of P in the sediments as at 2010. At the current dose rate in the two inflow streams, it would take about 4 years to supply sufficient alum to sequester the initial mass of P. However, it would still require that amount of alum to manage the SRP in the inflows.

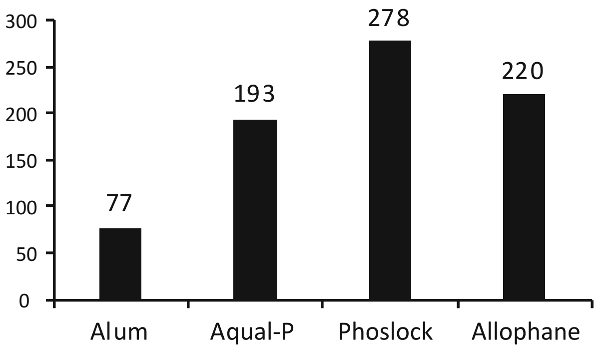


Fig. 7.24 Application rates of four P-inactivation agents (g agent m^{-2}) required to sequester all releasable P from the top 4 cm of sediment in Lake Rotorua (3.2 g P m^2) (from Gibbs et al. 2011)

7.8 Other Issues

In the discussions and examples presented above, potential issues of effects on non-target species have been noted. While the intention is to prevent or reduce the freely available SRP in the water column from sediment release or external inflows to a lake in order to reduce cyanobacterial blooms, the flocculants and sediment capping agents may also adversely affect other biota or microbial processes in the ecosystem, albeit on a short term basis. The noted effects include:

- Reduction in zooplankton grazing pressure as adult zooplankton caught in the flocking agent are removed from the upper water column. Recovery of zooplankton biomass is most likely by the growth of next generation. The effect is seen as a sudden but short-duration increase in phytoplankton biomass as indicated by chlorophyll *a* in Lake Okaro (e.g. Fig. 7.14). Sediment capping agents, which are not flocculants, may not have this effect (Özkundakci et al. 2011).
- Release of toxic trivalent Al^{3+} and La^{3+} ions when these P-inactivation agents are applied to low alkalinity water without an appropriate buffer (pH goes below 5.5) or there is insufficient SRP in the water column to bind to the Al^{3+} or La^{3+} ions released from these agents.
- Suppression of nitrification after the sediment capping agent is applied to the sediment. This means that NH_4-N released from the sediment is not oxidised to NO_3-N as it appears in the water column and denitrification cannot occur. The mechanism is not clear but it has been observed in Lake Taupo following a volcanic eruption (Fig. 7.10) and was observed in a sediment core incubation experiment (Gibbs et al. 2011).
- Suppression of nitrification also occurs as the pH rises above 8 and nitrification stops when the pH exceeds 10 (Ruiz et al. 2003). This could happen if calcite is used as an active barrier or lime is used to co-precipitate P with Ca from the water column.

7.9 Future Prospects

In 2008, at the culmination of a 37-year whole-ecosystem experiment, Schindler et al. (2008) made the controversial statement that 'Eutrophication of lakes cannot be controlled by reducing nitrogen input'. While this may be true for many lakes, the counter argument was that both N and P should be managed wherever possible (Lewis and Wurtsbaugh 2008; Abell et al. 2010; Paerl et al. 2011).

7.9.1 Sediment Capping Targeting N

In this chapter, we have investigated the use of flocculants, which can remove particulate N and P from the water column, and sediment capping agents which incorporate P-inactivation agents and, therefore, target the removal of P. While this is in line with the Schindler et al. (2008) concept, recently developed products, such as Aqual-P®, target both N and P. Current research internationally is investigating ways to reduce the nitrogen loads in lakes and rivers using sediment capping techniques. A promising further development in the use of zeolite, which has a nature sorption affinity for ammonia released from the sediment, is to load the zeolite with a biofilm 'biozeolite' that can remove nitrate from the overlying water column (Huang et al. 2011, 2012, 2013). The active barrier with the biozeolite was developed for use in lakes and canals in China. Continuing research has found that biozeolite also works with a protective sand covering in the flowing water (Huang et al. 2013). Development and use of this product as a thin layer sediment capping agent for mitigation of N in lakes is continuing (Zhou et al. 2016).

7.10 Conclusions

Eutrophication has become the primary water quality issue for most freshwater ecosystems in the world, and its acceleration is one of the most visible examples of the biosphere's alteration due to human activities. The main driving forces of eutrophication are high concentrations of both nitrogen (N) and phosphorus (P) inputs directly into surface waters and indirectly via groundwater. Without in-lake intervention, nutrients in the lake sediments become a legacy of human activity in the catchment that will remain to fuel internal loads long after catchment remediation strategies have reduced external loads. While there is some debate as to whether N or P should be managed, this chapter has followed the pragmatic approach of reducing the P internal load in lakes through the use of passive and active sediment capping barriers to reduce recycling of legacy P stored in the sediments. Capping is an appropriate short-term measure that will improve the water quality in the lake while the long-term strategies for sustainable management and reduction of external nutrient loads come into effect. However, given the potential for sediment toxicity and reduction in food quality for benthic-dwelling species, it is preferable that the application of sediment capping agents should be targeted and avoid littoral and epilimnion areas where biological diversity is highest in lake ecosystems.

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Chapter 8
Control of Invasive Aquatic Plants

Deborah Hofstra, John Clayton, Paul Champion, and Mary D. de Winton

Abstract The presence of invasive weed species invariably has a detrimental effect on native plant biodiversity, abundance and depth range in the short term and on native seed bank in the longer term. Removal of invasive weed beds may enable the recovery and restoration of native vegetation. The method or tools available for weed control include habitat manipulation and biological, chemical, mechanical, manual and integrated weed control. The selection of tools utilised for a particular weed issue are primarily dictated by the target weed species, characteristics of the lake or water body and the management goals. Management goals that target a significant reduction in weed biomass or eradication (in the longer term) and tools that result in selective target weed control provide opportunities for restoration of native aquatic plants. Restoration can occur via passive regeneration of native plants from adjacent sites, seed banks and waterfowl-mediated dispersal, or actively through planting, and brings associated benefits of habitat for native fauna, with additional improved amenity and recreation values.

Keywords Life-form types · Weed control · Weed eradication · Native values · Seed bank

8.1 Introduction

New Zealand has a multitude of freshwater bodies and lakes, many of which are integral to its international reputation as a land of scenic beauty, recreation and wonder. Our legacy of native species and unique opportunities for their protection and restoration is well known, as are the threats from invasive species that readily displace native plants and animals.

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8.1.1 Native Flora

The native aquatic flora is relatively diverse with over 100 species recognised to date, although most species are low growing. Unlike New Zealand's terrestrial flora, there is a relatively low rate of endemism with only 35 native aquatic species restricted solely to New Zealand. Migratory waterfowl seem responsible for introducing most of the plants that are native to both New Zealand and Australia, with some additional wind-dispersed species such as raupo (*Typha orientalis*) likely to have been introduced via the prevailing westerly winds (Champion and Clayton 2000).

Prior to European settlement submerged, native vegetation was dominated by macro-algae, known as charophytes, in the genera *Chara* (4 species) and *Nitella* (11 species) with seven common flowering plants (three water milfoils—genus *Myriophyllum*, three pondweeds—genera *Potamogeton* and *Stuckenia*, and an unattached bladderwort—*Utricularia australis*) and two species of quillworts—genus *Isoëtes*. All of these species are normally submerged, with the exception of a pondweed, *P. cheesemanii*, which has both floating and submerged leaves. A much larger assemblage of plants (over 40 species belonging to many different plant families) form low turfs in the shallow margins of water bodies in areas which often dry out during summer. Free-floating plants were found in sheltered areas and included the fern azolla (*Azolla rubra*), duckweed (*Lemna disperma*), watermeal (*Wolffia australiana*) and two species of liverwort (*Ricciocarpos natans* and *Riccia fluitans*). A diverse range of tall sedges (greater than 1 m tall) belonging to many different genera and raupo (*Typha orientalis*) commonly grow on the shallow fringes of water bodies with leaves emergent above the water's surface. Swamp willow weed (*Persicaria decipiens*) and swamp millet (*Isachne globosa*) are also emergent plants but they have a sprawling, rather than erect, growth form. These emergent plants may grow in water as deep as 3 m in sheltered areas.

8.1.2 Invasive Aquatic Plants and Their Impacts

A succession of new aquatic plants and animals (Chap. 9) have been introduced primarily for ornamental and aesthetic reasons through the aquarium trade and subsequently spread either deliberately or accidentally by people (Champion et al. 2002). Aquatic plant species have had spectacular success in invading New Zealand lakes, and their continued introduction and spread has resulted in few water bodies retaining their natural or original indigenous aquatic vegetation (Champion 2014a). A total of 89 introduced aquatic plant species have naturalised in New Zealand since the 1850s with an average of six new species naturalising every 10 years (Champion 2014a). The number of introduced and native aquatic plants can be compared based on their growth form classification of amphibious turf, submerged, free-floating, waterlily-type, sprawling emergent and erect emergent (Table 8.1). Growth forms

Table 8.1 The number of introduced and native aquatic plants classified by life form type

Life-form	Native	Naturalised	% of life-form naturalised
Amphibious turf	46	4	8
Submerged	32	18	36
Free-floating	5	6	55
Waterlily-type	2	14	88
Sprawling emergent	3	30	98
Erect emergent	23	17	43
Total	111	89	45

poorly represented in the native flora (sprawling emergent, waterlily-type and free-floating) tend to have a large representation of naturalised plants.

The characteristics and growth forms of New Zealand's native aquatic plants result in communities that tend toward an open structure, providing wide diversity and complexity of habitat for aquatic fauna (Hofstra and Clayton 2014). However, this open structure and low stature of most native aquatic species, compared with the dense growth forms of many of the introduced species (e.g. monocultures of species from the families Hydrocharitaceae and Ceratophyllaceae), has led to either a complete elimination of native species from some water bodies or (more commonly) exclusion from more favourable sites (to either deeper more light-limited or shallower more exposed sites). Invariably, the presence of invasive weed species has a detrimental effect on native plant biodiversity, abundance and depth range (Howard-Williams et al. 1987; Clayton and Edwards 2006). The displacement of native plant species represents a loss in habitat for native animals and genetic diversity due to replacement and degradation of seed banks that follows from the isolation of seed sources and changes in burial rate (Rowe and Champion 1994; de Winton and Clayton 1996). In addition, in shallow water bodies, where submerged weeds become surface reaching and fill the water column, the weeds can become self-limiting, leading to deoxygenation of the water and plant collapse. This in turn can flip the lake into an algal dominated turbid state (Champion 2002; Schallenberg and Sorrell 2009).

The impacts of surface floating [e.g. salvinia (*Salvinia molesta*), water hyacinth (*Eichhornia crassipes*)], sprawling emergent [e.g. alligator weed (*Alternanthera philoxeroides*) and reed sweet grass (*Glyceria maxima*)] weed species may be more apparent initially to the observer, as floating weeds are seen to spread across the water surface. Plants of these life-form types exclude light to submerged plant species growing beneath them, which reduces wildlife habitat, and leads to further decline of the aquatic system (Champion et al. 2002). Alongside reduced native and biodiversity values, other well-documented impacts associated with invasive aquatic plants include reduced amenity values and the economic utility of waterways for hydroelectric generation and the fact that water extraction can be compromised (Champion et al. 2002).

8.1.3 Recognising a Weed

Some plants are described as weeds because of their prolific growth and the adverse impacts they have on the local environment, such as displacing native flora and fauna. Amongst aquatic plants, particular attributes including growth form, capacity for reproduction and tolerance to a wide range of environmental conditions can be used to assess the risk that a species may pose, in terms of its ability to result in adverse or detrimental environmental impacts (Champion and Clayton 2000). Based on these attributes and a Weed Risk Assessment (Champion and Clayton 2000), examples of the six worst submerged pest plants in New Zealand are hydrilla (*Hydrilla verticillata*), hornwort (*Ceratophyllum demersum*), egeria (*Egeria densa*), lagarosiphon (*Lagarosiphon major), Utricularia gibba* and *Vallisneria australis* (Champion and Clayton 2000). These species are recognised as 'transformer species' because they transform (severely modify and impact) the systems that they invade (Richardson et al. 2000). In 2009, approximately 28% of the 344 New Zealand lakes that had been surveyed were impacted by one or more of the six worse submerged pest plants (de Winton et al. 2009), and there have been several notable incursions of these species since that time (e.g. *L. major* in Lakes Ngakapua, Ianthe and Aviemore; *E. densa* in Lake Carrot and Duddings Lake).

However, all submerged aquatic plants may have important ecological roles in stabilising sediments, improving water clarity, nutrient uptake and providing food and habitat for fauna (Clayton and Wells 1999). Even the recognised transformer species listed in the previous paragraph provide ecosystem services that are preferable to an algal dominated turbid state without any submerged aquatic vegetation. This is particularly apparent in severely degraded ecosystems with poor water clarity and quality and can lead to different perceptions of the 'weed' label. In general terms, a 'weed' may refer to any plant species perceived to be causing unwanted impacts. These unwanted impacts can be species based, site or lake specific or a combination of both species and location-based impacts.

8.2 How Weeds Can Be Managed to Restore Biodiversity in Lakes

8.2.1 Control Tools

Once a weed species or weed issue has been recognised, control options are often sought to protect amenity and utility functions of aquatic systems. However, there is also a growing desire to control invasive plants to support the restoration of lakes, improving biodiversity and habitat values. The tools or methods that can be utilised for the control of invasive aquatic plants can be broadly described by the following categories:

- Habitat manipulation
- Mechanical and manual
- Biological
- Chemical
- Integrated weed control

8.2.1.1 Habitat Manipulation

Habitat manipulation may be defined as altering the aquatic environment to favour desirable vegetation at the detriment of the target 'weed' vegetation. This includes techniques such as controlling the water level (e.g. draw-down) or installing benthic barriers or bottom lining to shade out target plants (see Feature Box 8.1). For example, lake level manipulation can vary in depth range, duration and frequency of draw-down, each having different impacts. If draw-down exceeds the photic zone, there will be no vegetation however small (2–3 m depth) but frequent fluctuations can increase habitat diversity by creating an amphibious zone that favours either emergent species or prostrate turf-forming species that are tolerant of periodic exposure or emersion (Clayton et al. 1986). Lake level draw-down was historically used for agricultural and power generation purposes in New Zealand, but for most hydro-lakes it is not generally considered an economically acceptable option due to loss of water storage and flexibility for power generation. Submerged rooted aquatic plants can grow to a depth of c. 10 m in clear water, with water pressure preventing deeper growth, while non-vascular native charophytes can grow to over 40 m depth depending on water clarity. If water levels fluctuate around 8–10 m (e.g. Lake Hawea) during the course of a year, a de-vegetated zone is created with disturbance limitations minimising biodiversity potential; however, charophytes can still form diverse meadows down to c. 35 m due to high water clarity (Clayton et al. 1986). Native aquatic plant species are well adapted to disturbance (e.g. desiccation from exposure or wave erosion during lowering and cattle grazing around lake margins) as most reproduce sexually and are able to recover from seed banks. For example, amphibious turf plant species around lake margins were quickly outcompeted by taller growing invasive species on sheltered shorelines. When the domination of tall-growing emergent plant species was prevented, the survival of an endangered turf species *Trithuria inconspicua* ensued (Tanner 1992). Stable water levels on the other hand can also have a detrimental impact on native aquatic plant diversity and abundance where invasive plant species coexist.

Box 8.1 Control of *Lagarosiphon major* in Lough Corrib, Ireland, and Biodiversity Restoration

Joseph Caffrey
Inland Fisheries, Dublin, Ireland

In 2005, the presence of an aggressive invasive species, lagarosiphon, was confirmed in Lough Corrib, Ireland (Fig. 8.1). Lough Corrib is the second largest lake in Ireland (17,800 ha), is of considerable ecological and conservation importance and is an internationally renowned Atlantic wild salmon (*Salmo salar*) and wild brown trout (*Salmo trutta*) recreational fishery. Lagarosiphon is a perennial submerged aquatic plant, native to southern Africa, which reproduces solely by fragmentation or vegetative reproduction. It probably escaped from a garden pond to the lake. In 2005, the weed was confined to a number of small sheltered bays but by 2010 it was present, often with considerable biomass, in more than 150 bays and littoral areas (see Fig. 8.1). At sites where lagarosiphon becomes established it creates dense, canopy-forming stands that exclude native macrophytes, altering habitat conditions for indigenous macroinvertebrate and fish communities, and directly impacting recreational exploitation of the affected watercourses.

With assistance from a European Life program and funding from Ireland's National Parks and Wildlife Service, Inland Fisheries Ireland commenced a programme of lagarosiphon control and lake restoration in 2009. This involved studying the biology and ecology of the plant in Lough Corrib in order to expose weak links in its life cycle that could be targeted for control. In addition, the efficacy of a range of practical control methods was investigated. Principal among these methods were mechanical removal using deep-cutting V-blades or trailing knives and light exclusion using jute or hessian matting.

Where the lagarosiphon grew to the lake surface (during the colder winter months in Lough Corrib), V-blades were employed. Unlike traditional weed cutting methods that can rarely cut deeper than 1.8 m below the water surface (while lagarosiphon optimally grows in 1–4 m water depth), V-blades cut or rip through the weed at lake bed level. This traumatises the plant and removes large volumes of rooting material from the soft substrate. The cut material is harvested and brought ashore. Subsequent regrowth of lagarosiphon at cut sites is closely monitored and has been demonstrated to be minimal.

Initial trials using sheets of black plastic to exclude light from lagarosiphon proved to be excessively onerous, environmentally damaging, and expensive. Subsequent trials using mats of natural fibre (jute or hessian) proved to be highly effective (Fig. 8.2). The open-weave jute material is durable, saturates and sinks rapidly when laid on the water surface, is relatively inexpensive,

(continued)

Box 8.1 (continued)
decomposes completely on-site and effectively kills lagarosiphon. What was not expected from the trials was that, once the lagarosiphon had been killed, the native charophyte vegetation that had been extirpated by this invasive species germinated from settled oospores and grew through the pores in the jute matting. Within 7–10 months of laying the jute, the percentage cover with native charophyte vegetation varied between 37 and 85% (see figure below). No lagarosiphon regrew through the jute.

The wide-scale use of V-blades and jute matting throughout Lough Corrib between 2009 and 2014 has reduced the level of infestation with lagarosiphon from about 92 ha to below 20 ha. No surface canopies of the invasive weed are present and all areas of the lake are again open for angling and boating. Native macrophyte and macroinvertebrate communities have naturally re-established in all treated areas. The lagarosiphon in Lough Corrib has not been eradicated but can now be maintained at manageable levels using the control methods developed during this project.

Fig. 8.1 Rinerroon Bay, Lough Corrib, pre-lagarosiphon invasion (left) and post-lagarosiphon invasion in 2005 (right)

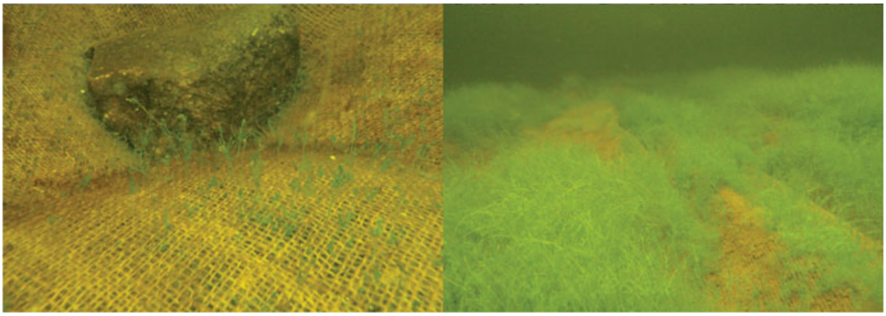

Fig. 8.2 Newly emerging *Nitella* spp. 8 weeks (left) and charophyte meadow 8 months (right) after the geotextile was laid

8.2.1.2 Mechanical and Manual Control

Mechanical and manual control of invasive aquatic plants in New Zealand lakes involves a diverse array of tools or methods from hand weeding to harvesters and suction-dredges for cutting and dredging submerged weeds, respectively (Fig. 8.3). For control of low weed biomass, hand weeding may appear to be an unlikely control tool, but for submerged weed species it often represents the removal of an early stage weed incursion or the final stages of a containment or eradication programme (Clayton 1996), such as for lagarosiphon in Lakes Wanaka and Waikaremoana. Diver operated suction-dredging (venturi suction pump) is also used in several lakes in New Zealand either for one-off removal involving a limited weed incursion that has a high risk of weed fragmentation and escape (e.g. from hand weeding) or as part of an ongoing management plan where weed biomass is progressively controlled through a succession of stages using tools appropriate to the biomass and stage of control (Fig. 8.3).

A small number of harvesters have been imported into New Zealand to remove invasive aquatic weeds, such as in the slow-flowing Avon River, Christchurch, where harvesters cut and remove nuisance weed growths twice a year (Wells and Clayton 2005), and for the removal of *C. demersum* in Lake Rotoehu in the Rotorua Lakes region. The principal reasons for using harvesters have been removal of nuisance weeds and/or removal of nutrients to achieve desired management targets. Elsewhere, a variety of locally designed cutting machines are operational, mostly purpose-built for use in small water bodies, canals and drainage systems (Wells and Clayton 2005; de Winton et al. 2013). Mechanical harvesters are best suited to areas free of underwater obstacles and water bodies where the target weed is already

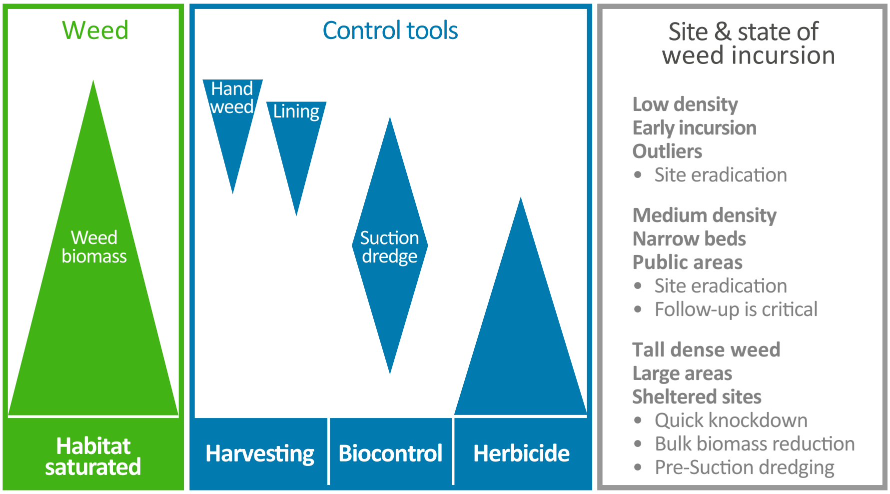

Fig. 8.3 Key factors for consideration when selecting an appropriate weed control tool. Weed biomass (green, left), site and state of the weed incursion (grey, right) determine the selection of control tool or tools (blue, centre)

widespread so as to avoid further establishment from drift fragments. Often the effects of cutting are short-lived, especially where tall growing weed beds are cut to only 2 m below the surface. In some cases, repetitive harvesting can result in reduced regrowth rates and when exotic weed beds are cut close to the sediment, a change in weed species dominance from lagarosiphon to native *Nitella* spp. has been reported (Howard-Williams 1993). Hornwort weed beds on Waikato hydroelectric lakes have a pronounced seasonal cycle where autumn efflux can lead to unacceptably high levels of drift towards power station intakes. These weed drifts have traditionally been managed by using interception booms, as well as a harvester to pick up weed accumulations and transport to shore (Howard-Williams et al. 1996). In recent years, research trials have demonstrated that pulverising of weed rafts floating downstream or accumulations on the boom was an acceptable management strategy.

8.2.1.3 Biological Control

Biological control refers to the use of one biological organism to control another. Biological control (or biocontrol) may be further defined in terms of classical biological control (CBC) where the target weed, in this case, and the organism have a very specific association such that abundance of the weed and agent closely follow one another or, generally, where there may be multiple weeds targeted by a generalist agent. In New Zealand, CBC agents have been introduced for the sprawling emergent plant, alligator weed (Julien 1981). The herbivorous fish grass carp (*Ctenopharyngodon idella*) was introduced into New Zealand to control aquatic weeds as a generalist, which means that a wide range of plants are consumed in order of the feeding preference of the grass carp (Rowe and Schipper 1985; Hofstra et al. 2014) (see Sect. 8.4.3.2). The development of a mycoherbicide (fungal based herbicide) for submerged aquatic plants is under investigation both in the USA and New Zealand (Shearer 1996, 2009; Hofstra et al. 2004a; Shearer and Jackson 2006).

8.2.1.4 Chemical Control

Two herbicides registered for use on submerged aquatic plants in New Zealand are diquat (dibromide) and endothall (dipotassium salt). Glyphosate (isopropylamine) is a broad-spectrum, non-selective, systemic herbicide that may be used to control plants around waterways. Restricted use of additional herbicides [metsulfuron methyl, haloxyfop methyl, imazapyr isopropylamine and triclopyr triethylamine (TEA)] under regulated conditions is possible to enable the eradication and control of priority national and regional pest plants (de Winton et al. 2013).

Herbicides can provide selective and targeted weed control (see Sect. 8.4). Products can be applied via hand held sprayers, submerged trailing hoses (e.g. by boat) or from the air (e.g. by helicopter). Product placement can be improved by using dye tracers (e.g. rhodamine dye) to predict water movement, by the addition of gelling agents in the product to alter viscosity in the water and with GPS tracking to improve accuracy and cover of large weed beds.

8.2.1.5 Integrated Control

An integrated approach, as the name suggests, refers to a combination of tools to provide the weed control outcome that is sought.

Selecting an appropriate control tool or tools requires consideration of key factors such as the amount of weed biomass, both in terms of the density and area, and the utility of the control tool (Fig. 8.3). For example, hand weeding and bottom lining are feasible tools for low density and early weed incursions, whereas herbicides, harvesting and grass carp are tools needed to reduce large weed volumes (Fig. 8.3). Species factors, such as the propensity of the plant to fragment and form new plants, or the presence of seed may mean that harvesting (where plant fragments are created) is only suitable in water bodies that are already habitat saturated by that species and where downstream spread is not an issue or can be mitigated. Site factors such as water depth, prevailing winds, wave fetch or substrate type in the littoral zone influence the suitability of mechanical methods or the placement of bottom lining fabrics. Whereas the use of grass carp is subject to statutory approvals and is site dependent. For example, site base factors that limit the use of grass carp include sites with connected watersheds in which grass carp cannot be contained or with poor water quality or that are very shallow (Hofstra et al. 2014).

Other considerations include a legislative requirement to control some pest plants in regional pest management plans [e.g. Waikato Regional Pest Management Plan 2014–2024 (Waikato Regional Council 2014)]. It is also noteworthy that doing nothing (no action) to control aquatic transformer species is likely to lead to increased environmental challenges in the future. As invasive transformer species continue to persist and dominate the vegetation at a site, local impacts may compound, resulting in legacy issues that make future restoration more difficult. For example, the deterioration of native seed banks under dense weed beds of submerged aquatic plants has been well established (de Winton and Clayton 1996; de Winton et al. 2000). In the absence of control, downstream impacts and the probability of expansion to new adjacent sites increase over time (Lockwood et al. 2009; Simberloff 2009).

The control of invasive aquatic plants is likely to result in trade-offs between the ability to control the weed species, the time frames, costs and limitations of what can be achieved for the water body or lake in question with the method or tool used. The decision process requires an overview, with goals for managing the weed, the site and/or the region.

8.3 Management Targets: Restoring Function and Indigenous Values

While the impact of invasive weed species is huge in many New Zealand lakes, there are still a number of water bodies relatively unimpacted by invasive species. In these cases, the management focus should be proactive protection (e.g. prevention of weed spread and incursion response to eradicate newly detected invasion by those species) rather than restoration. A region-wide strategy to prioritise management of lakes has been developed for the Northland region. This covers an area supporting a large proportion of warm, lowland New Zealand lakes still with relatively good water quality, supporting a range of endemic endangered species with limited impact of invasive species on their native biota, which is unparalleled in any other region of mainland New Zealand (Champion and Wells 2014). The use of management strategies is discussed with the Case Studies later in this chapter.

When a weed issue has been identified beyond the level where incursion response management is deemed feasible, the choice of tool or tools used to control invasive aquatic plants are determined primarily by the target weed, the water body type and the management goals (Fig. 8.3). The management goals for weed control are intrinsically linked to the goals for restoration (Fig. 8.4). In general terms, removal of invasive aquatic plants enables restoration, either passively via re-establishment of native plants from seed banks or actively through planting of native species. To enable restoration, a significant reduction (or eradication) of the invasive plants are usually required, and additional activities or tools may be required to achieve that reduction over and above the activities required to achieve control of nuisance level weed infestations (Fig. 8.4). Managing specifically for restoration, the feasibility of restoration must be taken into account (i.e. what is being restored at that site, and what is the restoration goal?).

8.3.1 *Nuisance Control*

Nuisance control is practiced widely in New Zealand, particularly where aquatic weed growth compromises safety (e.g. swimming, boating), impedes drainage (e.g. affecting land productivity) or reduces hydropower generation. In these situations, weed control is mostly seasonal, such as in summer for public recreation or from autumn to early winter during weed bed senescence associated with increased efflux from higher water flows as in hydrolakes. The primary drivers in this case are either public safety or economic benefits, with nuisance weed control costing less than the benefits gained (e.g. land productivity or electricity sold). A common outcome from the use of surface-applied herbicide or using a harvester to cut submerged weed is that viable buds on remaining bottom rooted stems can enable weed bed recovery within a year (Fig. 8.5).

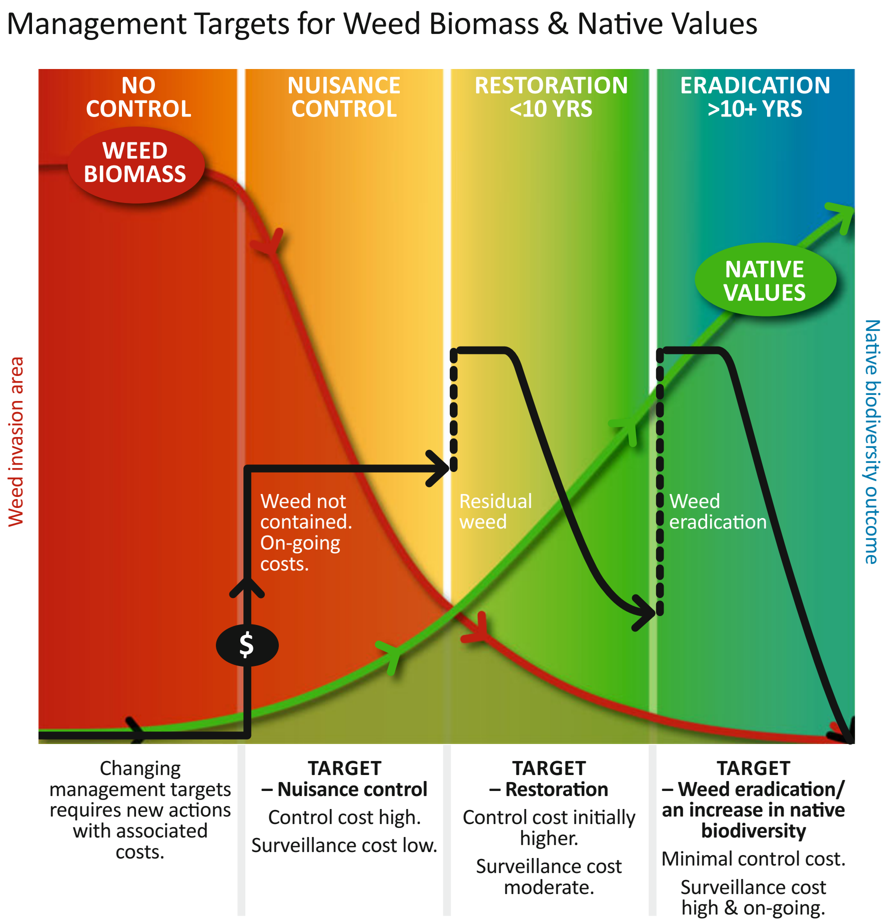

Fig. 8.4 Diagrammatic illustration of management targets for weed biomass reduction and outcomes for native biodiversity restoration. The four vertical panels represent key decision points and targets for weed control; the black, red and green lines indicate (respectively) the relative cost (dollars), weed biomass and increasing native values as weeds diminish

8.3.2 Restoration

Restoration is increasingly pursued as a management goal in New Zealand, especially where weed growths can be cost-effectively suppressed to avoid seasonal 'bounce-back'. 'Bounce-back' is common when nuisance weed control removes only the biomass lying at or near the water surface (previous section). However, restoration requires additional follow-up control appropriate for the growth characteristics of the target weed species and the habitat. For example, as shown in Fig. 8.5, spraying with diquat effectively defoliated lagarosiphon and also destroyed the upper portions of each stem. The subsequent recovery of the native pondweed

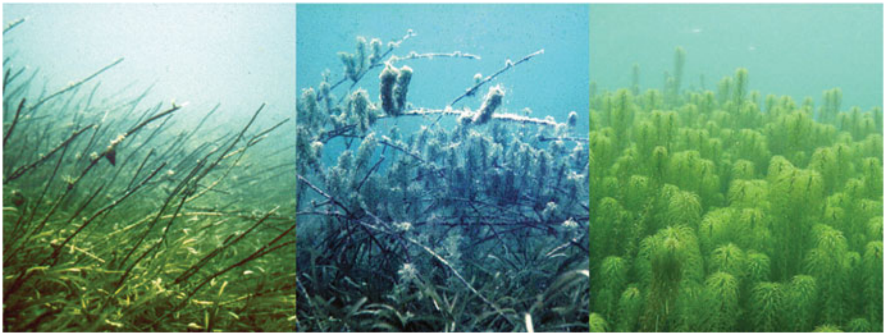

Fig. 8.5 Weed bed recovery (*Lagarosiphon major*) following annual control with herbicide in Lake Wanaka. Upper weed bed biomass is destroyed by initial treatment (left), enabling seed bed recovery of native pondweeds (*Potamogeton ochreatus*) but new lagarosiphon shoots emerged from buds of remaining lower stems. Lagarosiphon shoots form a new dense canopy returning to the surface and displacing all native macrophytes within the year (right)

Potamogeton ochreatus from exposed sediment was only possible due to the opening up of the lagarosiphon weed bed canopy allowing light to reach the lake bed. However, the remaining defoliated lagarosiphon stems still retained numerous dormant vegetative buds, which started sprouting and before long outcompeted native plant regeneration and re-established a monospecific surface-reaching canopy.

Although this annual cycle of weed destruction and regrowth is considered effective as a nuisance control strategy, follow-up herbicide treatment of the recovering weed bed is required to achieve restoration. The recovering weed bed has soft new stem growth that is more vulnerable to diquat than mature 'woody' stems. Treatment at this stage not only destroys the recovering stems but also the young newly formed buds that have not yet sprouted. Such a follow-up treatment can suppress further weed regrowth for several years, as in Paddock Bay in Lake Wanaka. The opportunity for restoration is dependent upon the presence of a viable seed bank of desirable native species in the sediment. Very few of the introduced submerged aquatic plants produce seed, while most native plant species contribute to the seed bank.

The viability of a seed bank is dependent upon the density and duration of invasive weed bed presence. Where no nuisance weed bed control occurs, native seed banks are progressively buried by sedimentation of particulates and organic matter suspended in the water column. Furthermore, without seasonal replenishment seed bank viability becomes impoverished. One of the advantages of regular nuisance weed control is that the opportunity for seed bank replenishment continues; whereas without any weed control restoration potential is compromised (de Winton and Clayton 1996; de Winton et al. 2000).

Key benefits of restoration as a management target for weed control include (1) more enduring control of target weed species leading to reduced weed control

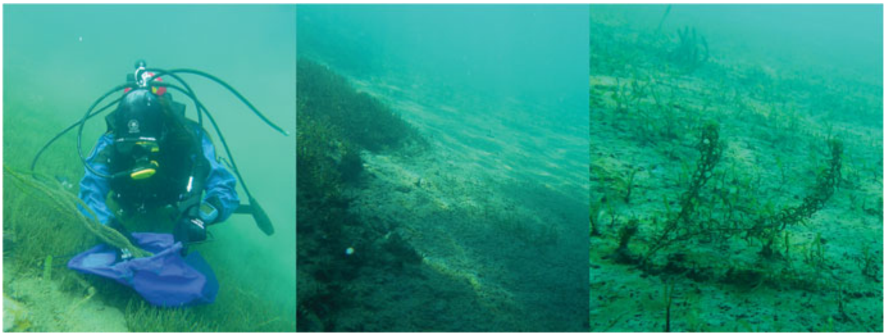

Fig. 8.6 Eradication and restoration. A single invasive lagarosiphon plant is removed from a dense native sward of *Isoetes alpina* (left). In deeper water (c. 3–6 m) below the maximum depth of *I. alpina*, several kilometres of lagarosiphon have been removed by scuba divers and suction pump (centre); several months later, the same site shows the start of widespread recovery of native pondweeds (*Potamogeton ochreatus*), but follow-up removal of occasional lagarosiphon plants is required to ensure eradication and restoration (right)

costs and (2) increased native plant recovery, species diversity and replenishment of native seed banks (Fig. 8.6). In freshwater environments, invasive plant species often form dense monocultures that become deoxygenated at night, while a build-up of low density, flocculent, organic sediment can exclude desirable aquatic fauna such as freshwater mussels (*Echyridella* spp.) and koura or crayfish (*Paranephrops* spp.).

8.3.3 Eradication

Eradication as a management target for invasive aquatic plants is an ideal or ultimate goal requiring careful assessment as to its achievability and likely cost. The scale of infestation, area of coverage, risks posed, method of plant reproduction, dispersal mechanisms and transparency of habitat are all relevant factors to consider. Mammalian pest eradication (e.g. rabbits, stoats and rats) has enabled endangered native bird species to be restored on many off-shore islands around New Zealand and these restoration initiatives are internationally recognised (e.g. Clout and Russell 2006). The same principles apply to lakes within New Zealand, which function essentially as isolated 'islands' of water. Eradication of targeted aquatic weed species has been achieved on a national scale (e.g. *Pistia stratiotes*, *Potamogeton perfoliatus*); island scale (e.g. hornwort from the South Island) (Champion et al. 2014); and water body scale (e.g. egeria from Parkinson's Lake—see Sect. 8.4.3.2). The advantage of eradication is that once achieved all further costs are substantially reduced or eliminated. Although eradication is the ultimate goal, the cost of ongoing surveillance can be high where there is a risk of re-infestation.

8.3.4 Monitoring and Surveillance of Aquatic Weeds for Lake Restoration

To identify management targets and assess the feasibility for managing invasive aquatic weeds at a water body, requires knowledge on what weeds are present, where and how abundant they are (see previous sections). Equally if not more important, is consideration of which weeds may threaten the system in the future. Proactive management, where targeted investigations or surveillance can be used to prevent or intercept invasions, is much less costly than reactive management of weeds once they have established at a site (Champion and Wells 2008). Early detection is also the key to successful eradication (see case studies) or cost-effective intervention. The main steps for monitoring and surveillance of weeds are outlined in Fig. 8.7.

Weed inventory or information gathering is an important first step. The existence of weed records at the site or nearby locality may be searched from New Zealand databases (e.g. National Institute of Water and Atmospheric Research (NIWA) Environmental Monitoring and Observations (NEMO), Department of Conservation (DOC) BioWeb, Plant Conservation Network, New Zealand Virtual Herbarium, New Zealand Biodiversity Recording Network, New Zealand National Vegetation Survey Databank). Another source of information is contained in Regional Pest

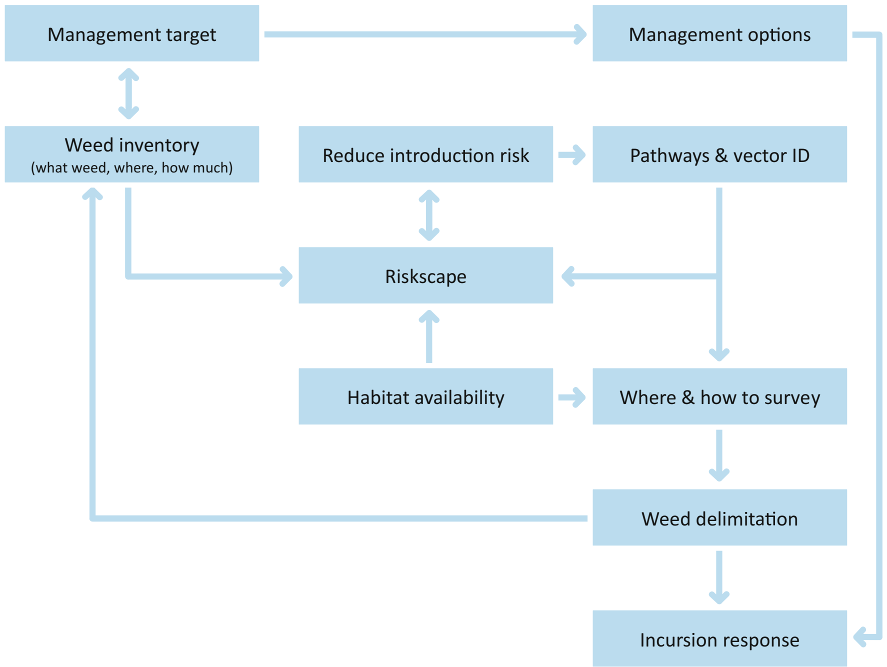

Fig. 8.7 Initial steps for gaining aquatic weed information via monitoring or surveillance for management assessment

Management Plans (e.g. Waikato Regional Council 2014). As well as uncovering latent weeds at a site, the information collected contributes to an overview of the 'risk profile' for a lake (including risk of introduction of weeds in the area).

Additional assessment of the routes by which pest plants can enter lake environments (pathways) and the mechanisms by which they are spread (vectors) are necessary to gauge risk and to tailor surveys to those priority lake sites with the highest invasion potential (Fig. 8.7). Recognition of, and management of, potential pathways and vectors is key to removing invasion risk to water bodies (Fig. 8.7). However, it is also important to consider the habitat availability for particular types of aquatic weeds and their likely impacts.

Because the majority of freshwater submerged weeds are spread between lake catchments by human activities in New Zealand (Compton et al. 2012; Johnstone et al. 1985), recreational access points in lakes are of key consideration for surveillance for new weeds (Champion and Wells 2008). These include road access, ramps and public boating amenities, but also private jetties and property waterfronts, popular beaches and anchorages. Trailered boating traffic is likely to be the lead vector in weed spread (Johnstone et al. 1985). However, plant transfers with contaminated barges, mechanical weed harvesters or weed cutting boats are some unfortunate examples of management actions gone wrong in New Zealand.

Invasive weeds have been introduced during efforts to enhance lake environments, with examples of weeds established in ornamental ponds in lakeside parks, with out-plantings of water lilies (de Winton et al. 2009), or as plant nursery contaminants. Some weeds have been found in association with mai–mai (shooting hides) of duck shooters, either as a contaminant of building materials and equipment or introduced as waterfowl food/habitat. Weeds have migrated along drainage canals, aided by contaminated diggers, with similar potential for weed spread associated with restoration of hydrological connectivity for lakes.

Surveys to inventory weed presence and extent should target suitable habitat areas (Fig. 8.7); for example, invasions were preferentially recorded at river inflow deltas in a North American lake (Eichler et al. 2001). Usually, weed habitat considerations for lakes are level fluctuations, shoreline wave exposure, water depth and clarity, slope and existing vegetation type and development.

Survey methods to detect or delimit invasive aquatic plants depend upon the weed (emergent or submerged) and environmental conditions (e.g. water clarity, background vegetation). Detection of submerged weeds within lakes, especially when present at low abundance immediately after introduction, is particularly challenging and it is difficult to confirm a 'true absence'. Methods employed for detection of low density submerged weeds include surface-viewing from boat or glass-bottomed boat, remote camera, snorkel and SCUBA (boat tows, underwater scooters or free-swimming). Shoreline searches for fragments are rarely effective for picking up weeds before they are well established. The most efficient surveillance method for early detection of weed surrogates under moderate water clarity (<5 m Secchi disc depth) was found to be snorkel or scuba diving, depending on water depth (de Winton et al. 2014). Similarly, boat-towed snorkelling and scuba diving was best for detection of lagarosiphon after control works in an Irish lough (Caffrey et al. 2011).

Information on weed extent and abundance feeds directly to re-consideration of the management objective (e.g. nuisance control, restoration or eradication). In the case of early detection for a recently introduced weed, design of an incursion response may be the most appropriate management action towards containment of the weed or the ultimate goal of eradication.

Reassessment of the steps in Fig. 8.7 may be required if lake restoration alters the Risk Profile. For instance, improved water clarity may enhance habitat for weeds, or littoral space be created by hydrological modification. The invasion status of adjacent or connected water bodies may also change over time.

In the event that weed risk can be seen as very high or increasing, preplanning for incursions allows a swift response in the event of invasion. This may be as basic as having a contingency budget, exploring feasibility of different control techniques, through to arranging consents for herbicide use.

8.4 Examples of Controlling Invasive Aquatic Weeds and Restoration Outcomes for Lakes

8.4.1 Management Strategies for Large Lakes or Regions

8.4.1.1 Lake Wanaka

Lake Wanaka in New Zealand's South Island is an example of how management strategies can be developed for a large complex lake. This is the fourth largest lake (192 km^2) in New Zealand; 42 km long and 10 km wide. It has islands and side arms, resulting in substantial habitat complexity. It is a popular summer and winter tourist area, and the lake is protected under the Lake Wanaka Preservation Act 1973 with an official oversight committee "Guardians of Lake Wanaka" appointed by the Minister of Conservation to oversee compliance with the Act and matters that may affect the welfare of the lake.

Lagarosiphon was first recorded on the Roys Bay foreshore in about 1972, having entered the lake from Bullock Creek that runs through the Wanaka Township. Although significant effort was made to eradicate this weed, there were several periods when funding and commitment stalled and lagarosiphon spread escalated. In 2005, the Lake Wanaka Guardians held a 2-day workshop that led to the establishment of a multi-stakeholder management committee and the development of a Ten Year Management Plan for the weed. This provided a guiding document that was critical to the ongoing commitment required to manage lagarosiphon in Lake Wanaka.

Four primary weed management objectives were considered appropriate for specific shorelines and discreet areas around Lake Wanaka: Eradication, Minimum Biomass, Containment and Nuisance Control. Prioritisation of control works and due diligence on expenditure of the weed management budget was essential. In response, the Government agency, Land Information New Zealand (LINZ), in conjunction with NIWA and an environmental economist established a weed control prioritisation model that could be applied to each distinctive spatial management unit (e.g. islands, bays). The prioritisation process was based on assessing each management unit for:

- *Pest Containment*—Objective to minimise intra- and inter-lake weed spread
- *Pest Control*—Objective being maximum control for minimum cost
- *Habitat influences*—Objective being to take account of environmental, habitat and plant factors that can influence the nature of the final outcome
- *External Influences*—Objective being to minimise negative community and political effects from the implementation of, or failure to carry out, mitigation plans.

Value Factors (or 'indicators') were identified for each of these management units, including: Political sensitivity, Public perception, Public safety, Commercial impact, Amenity value, Risk of intra-lake spread, Risk of inter-lake spread, Biodiversity values, Remoteness, Ease of control, Cost of control, Chance of eradication, Time to achieve goal, Recovery Rate and Habitat limitation. All of these factors were scored and compared for each of the management units. This then formed the basis of prioritising control works between sites and allocating a budget for contractors, subject to final endorsement by a multi-agency management committee that included the Lake Wanaka Guardians. Contractors engaged to carry out control works filled out a spreadsheet based on works completed, weed removed and actual costs. This iterative model was applied twice each year with refinements based on outcomes and contractor experiences.

The control tools used in the Lake Wanaka lagarosiphon management programme include: suction-dredging, hand weeding, bottom lining using hessian (see Feature Box 8.1) and diquat herbicide application either by helicopter or from a submerged boom beneath a barge. Suction-dredging has been critical for removing areas of lagarosiphon that were too large to remove completely by hand weeding but which lay within a designated 'Eradication Zone'. Hand weeding has been effective for follow-up clearance of lagarosiphon shoots or regrowth following suction-dredging. Diquat application has had mixed results ranging from minimal impact to highly effective control of extensive nuisance weed beds. The risk of inadequate control following aerial application of diquat has been reduced by the use of gel additives to help minimise diquat dispersion from the target zone. Additionally, the use of rhodamine dye provided a visual marker on water movement thereby enabling refined placement to better target the designated weed beds. Another refinement developed during the lagarosiphon control programme in Lake Wanaka has been the use of a 'dirtiness' scale to assess weed condition, which is used shortly before any decision is made to apply diquat to target weed beds. Clean to slightly silty shoots are considered appropriate for diquat treatment, while 'dirty' shoots present a risk of failure (Clayton and Matheson 2010).

Commitment to the Lake Wanaka lagarosiphon control programme over the last 10 years has led to substantial claw-back of lagarosiphon spread and dense weed growths. A significant achievement has been made in maintaining and expanding the Eradication Zone. Clearance of lagarosiphon from immediately outside this zone has continued to progress each year, for example, by 2015 the lagarosiphon along several hundreds of metres of shoreline had been reduced by management to the point where removal by hand weeding is now feasible. The goals and targets identified in the 2005–2015 Lake Wanaka Lagarosiphon Management Plan (2005) have been achieved and a revised 2016–2025 Management Plan has been developed.

8.4.1.2 Northland Region: Lakes Prioritisation, Surveillance, Incursion Response, and Eradication Progress

The Northland region has over 400 freshwater lakes, the majority of which are coastal dune lakes. Some are in a relatively pristine condition, with minimal impacts from pest species and catchment activities. However, recent declines in many of these water bodies have been detected, with the spread of invasive aquatic weeds a major driver of these changes and a coordinated approach to lakes management was identified as a first step to effective management. A strategy was devised to guide monitoring and management (including restoration) by regional council and other organisations (Champion and de Winton 2012). This Northland lake strategy provided an overview of the significance of the lakes, a classification system for these lakes, measurement of lake values and ranking based on these values, identification of threats and pressures and assessment of these for each lake.

Ecological values were assessed for each lake including habitat size, buffering, water quality, aquatic vegetation diversity and integrity, presence of endangered and key species and connectivity. Thirty-four lakes were classed as Outstanding, High or High–Moderate. The Outstanding lakes were Taharoa, Humuhumu, Waikere, Rotokawau (Pouto), Mokeno, Kai-Iwi, Ngatu, Wahakari, Kanono, Waiparera, Waihopo and Morehurehu. Northland Regional Council endorsed the management of the 12 priority lakes (Champion 2014b).

Despite the high ecological values provided by many Northland lakes, the status of these water bodies is not secure. Pressures and threats identified and evaluated include biosecurity threats (aquatic weeds, pest fish and their risk of spread), eutrophication pressures from land use such as pasture and pine forestry and predominance of planktonic algal blooms and water level fluctuations, especially dropping lake levels. There have been numerous incursion responses to submerged aquatic weeds, and additional eradication of newly detected emergent weeds (Champion and Wells 2014; NIWA unpublished data) has reduced the risk of further weed spread and will allow for restoration of indigenous vegetation.

To date, no new submerged plant incursions have been detected at the six surveillance lakes, although viable fragments of the weed *Elodea canadensis* were found on the shore of Lake Taharoa at a popular boat launching area in 2007. Surveillance at Lake Ngatu led to the detection and successful eradication of the emergent yellow flag iris (*Iris pseudacorus*) by hand weeding in 2007 and progress towards eradication of alligator weed (detected at Lake Ngatu in 2012) using the herbicide metsulfuron methyl is well advanced. The wetland weed Christmas berry (*Schinus terebinthifolius*) was detected in April 2014 and has been eradicated by cutting and stump swabbing with triclopyr ester (as X Tree Basal®). The submerged weed lagarosiphon was detected in the high value Lake Ngakapua in October 2014, and was subsequently eradicated.

The vegetation of two lakes, Lake Rotootuauru (also known as Lake Swan) and Lake Heather, were completely dominated by the weeds hornwort and egeria. Lake Rotootuauru is situated on Northland's Pouto Peninsula amongst a group of other lakes that are relatively unimpacted by introduced weeds, with lakes Humuhumu, Rotokawau and Kanono all classified as regionally outstanding water bodies. Lake Heather is situated within a group of lakes on the southern Aupouri Peninsula including the regionally outstanding Lake Ngatu. Targeted eradication of the problem species hornwort and egeria from two lakes using grass carp is well advanced, with no sign of either submerged weed detected on the last inspection (October 2014, NIWA unpubl. info.). In addition, the abundance of a desirable sprawling emergent native plant *Myriophyllum robustum* has increased in Lake Heather (Fig. 8.8) (Wells and Champion 2015).

Lagarosiphon major formed a dense band of submerged vegetation between 1 and 2.5 m deep in Lake Phoebe (Champion and Wells 2014). Selective herbicidal control of lagarosiphon was achieved using endothall and now, there is only indigenous submerged vegetation present in this lake. Endothall will also be used to attempt eradication of the new lagarosiphon incursion in Lake Ngakapua (I. Middleton, Northland Regional Council pers. comm.).

8.4.2 Selective Target Weed Control

8.4.2.1 Lake Okataina

Lake Okataina in the Rotorua Lakes District of the Central North Island has been the focus of a hornwort eradication programme. One isolated plant was initially found in 2007 near the lake access site adjacent to a jetty. This was thought to be an isolated incident with introduction by boat or trailer at the ramp. Subsequent searches failed to find any more hornwort plants, but in 2009 several drifting plants were found by research divers 1.2 km from the boat ramp. This led to extensive shoreline and diver searches to try and locate the source of drift fragments. In 2010, the original source was located at the opposite end of the lake in south west arm. The regional management authority, Bay of Plenty Regional Council, initiated measures to prevent establishment of hornwort along the public access area at the north-east end of the lake (Tauranganui Bay), since not only would hornwort be a public nuisance, this weed would also be at much greater risk of transfer out of the catchment to other nearby lakes such as Rotoma. One factor confounding efforts

Fig. 8.8 A large patch of *Myriophyllum robustum* in Lake Heather

to locate the source of hornwort or even find where else it might be establishing was the tall dense growths of lagarosiphon occupying much of the littoral zone. Lagarosiphon was also the dominant plant along Tauranganui Bay, so in order to prevent discreet hornwort invasion of lagarosiphon beds of Tauranganui Bay, this stretch of open shoreline was treated regularly with diquat to keep lagarosiphon weed beds under control. This programme resulted in almost total elimination of about 400 m of lagarosiphon weed bed along the lake littoral zone and subsequent recovery of native species from sediment seed banks. A variety of charophyte species and native plants (mostly *Potamogeton ochreatus*) replaced the band of invasive lagarosiphon, and mixed turf species were common in the shallow water margin. This example fits the model (Fig. 8.4) where localised restoration has been achieved, but in this case as collateral benefit to weed control measures taken to improve weed surveillance capability.

8.4.2.2 Yellow Flag Iris in Lake Rotoroa

An eradication programme for yellow flag iris was implemented at Lake Rotoroa (Hamilton Lake), and within 2 years few iris plants remained and restoration of emergent sedges was facilitated (Fig. 8.9).

Fig. 8.9 Yellow flag iris (*Iris pseudacorus*) at Lake Rotoroa (Hamilton Lake) before 2003 (left) and after metsulfuron methyl control and re-planting in 2005 (right)

In the late 1980s, yellow flag iris first appeared as a few clumps on the margins of Lake Rotoroa. Yellow flag iris has a very attractive large yellow flower (and was grown as a garden ornamental), but has a thick creeping rhizome and can grow to 2–3 m tall in northern New Zealand blocking access to waterways, reducing water movement and affecting bird habitat by facilitating predator access to nesting sites. In 2002, it was listed in New Zealand's National Pest Plant Accord banning its sale, propagation and distribution (www.biosecurity.govt.nz/nppa).

At Lake Rotoroa, yellow flag iris was initially slow in spreading but invaded the limited areas of native emergent aquatic plants, displacing raupo, bamboo sedge (*Machaerina articulata*) and even kuta (*Eleocharis sphacelata*) in deeper water and forming extensive floating mats over water as deep as 1.7 m. In 1988, glyphosate and metsulfuron methyl were trialled at different rates and times of year at the lake. Glyphosate applied at 3% (10.8 g ai/L) plus 0.1% surfactant applied in spring arrested growth and flower development, but when applied in early winter it killed about 80% of the rhizome. However, metsulfuron methyl 0.3 g ai/L plus 0.1% surfactant in late autumn gave a much higher success rate (100% where good coverage was achieved) and had the advantage of causing little off target damage to native sedges and raupo. Again the timing of application in late autumn was a critical factor.

It was not until 2003 that lake residents and the regional council recognised yellow flag iris as a significant problem as it had grown to dominate the wetland margins of Lake Rotoroa and had spread further downstream to infest the margins of the Waikato River. An eradication programme for the lake was then implemented based on research results. Within 2 years, few iris plants remained, and restoration of emergent sedges through natural spread and enhancement of marginal areas with a range of native species sourced from local Waikato wetlands and lakes was undertaken. A walkway was constructed through this area allowing public access. Annual follow-up to prevent new infestations of yellow flag iris developing and seeding is an ongoing (but diminishing) maintenance requirement.

8.4.3 Eradication of Invasive Aquatic Plants to Enable Native Plant Regeneration

Where possible eradication of invasive aquatic weeds, followed by the restoration of beneficial aquatic plants in the longer term, to enhance ecosystem values and functions is considered desirable for most aquatic systems. The following case studies demonstrate the use of different tools to eradicate target weeds and enable restoration of native flora.

8.4.3.1 Endothall to Eradicate Hornwort and Lagarosiphon

The introduced submerged aquatic plant hornwort (*Ceratophyllum demersum*) ranks as one of our worst aquatic weeds (Clayton and Champion 2006), causing large-scale weed problems by forming weed beds up to 10 m tall, weed drift that blocks power station intakes and displacing native aquatic vegetation over the 1–14 m depth range (de Winton et al. 2009). Hornwort is listed as an Unwanted Organism throughout New Zealand, meaning it is an offence under the Biosecurity Act 1993 to sell, propagate or distribute it. It is also listed by the Ministry of Primary Industries (MPI) as a National Interest Pest Response (NIPR) for the South Island only with the aim of eradication and exclusion there. Endothall [dipotassium salt (DPS)] has been the main tool to achieve this goal. Endothall was used to successfully eradicate hornwort from 0.7 ha Centennial Lake, Timaru, the last known hornwort South Island field site in 2006–2007. This was achieved with only one application of endothall at 5 mg L^{-1} total water body application (Wells and Champion 2012) with no hornwort found since, despite all other species remaining in good condition.

Similarly, lagarosiphon is an invasive aquatic weed listed as an Unwanted Organism. It was eradicated from five water bodies (up to 3.9 ha) with rates as low as 0.1 mg L^{-1} (50 times less than label rate), under cool (c. 16 °C) summer temperatures in the Southland region of New Zealand (Wells and Champion 2012; Wells et al. 2014). Five years after the treatment, no lagarosiphon was present, and no effects were seen in native species.

8.4.3.2 Grass Carp to Eradicate Egeria and Hydrilla

In Parkinson's Lake, grass carp eradicated the target plant egeria and removed most other aquatic vegetation (non-target species). The removal of the grass carp was followed by the re-establishment of native macrophyte species reproduced from seeds, spores and buried rhizomes (Rowe and Champion 1994; Tanner et al. 1990).

Parkinson's Lake is a small (ca 1.9 ha) eutrophic dune lake located south of Auckland, and it is one of the earliest lake examples where grass carp were evaluated for their ability to control aquatic weeds in New Zealand (Mitchell 1980; Rowe 1984). Although the history of the lake's deterioration, pest introductions and subsequent bio-manipulation of plants and fish to restore Parkinson's Lake is described in detail by Rowe and Champion (1994), it is included briefly in this chapter because the lake provides a valuable example of invasive weed eradication and consequent native vegetation recovery (i.e. restoration of the lakes indigenous character).

The invasive macrophyte egeria was introduced into the lake in the 1960s and by 1976 had replaced the low-growing native plant species around the littoral zone and formed surface reaching weed beds from depths of 5 m (Rowe and Champion 1994), occupying a third of the lake area (Mitchell 1980). The native emergent aquatic plant kuta occupied about 44% of the lake (Mitchell 1980).

Grass carp were incrementally stocked between May 1976 and November 1977 to reach a density of 44 fish/ha (Mitchell 1980). By January 1979, the grass carp had totally removed all egeria and shoots of kuta (Mitchell 1980). The only remaining vegetation was comprised of low-growing turf plants (*Glossostigma elatinoides* and *Myriophyllum propinquum*) in shallow water (<2 m deep) (Rowe and Champion 1994).

In October 1981, 4 years after their first release in the Parkinson's Lake, the grass carp were removed. Subsequent monitoring of the lake's vegetation illustrated that native aquatic plants (primarily *Potamogeton ochreatus* and *Nitella* sp. aff. *cristata*) had re-established from the seed bank (Tanner et al. 1990; Rowe and Champion 1994).

Grass carp were introduced into Elands Lake in the Hawkes Bay region of the North Island as a trial in 1988 to determine their effectiveness to control and potential to eradicate hydrilla. Unlike egeria, hydrilla has long-lived propagules (tubers) that necessitated a longer term trial to determine whether or not hydrilla could be eradicated and to establish the restoration potential of the lake (Clayton et al. 1995).

Elands Lake is a 4 ha spring-fed dam on a privately owned farm. In the 1980s hydrilla covered about 1 ha of the lake down to around 4.5 m depth of this shallow (max depth 7 m) water body, having displaced the native flora (Clayton et al. 1995). Initially 100 fish/ha (400 fish in total) each about 270 mm in length were stocked in December 1988 (Clayton et al. 1995). An assessment of vegetation in April 1990 revealed a major reduction in hydrilla 17 months after grass carp were released. In 1991, further grass carp were released to provide grazing pressure by smaller, younger fish amongst obstacles along the shoreline that may have impeded access by the now larger fish from the initial stocking. In November 1991, extensive searches at depths of 1–1.5 m revealed occasional hydrilla plants regrowing from tubers or buried stems, predominantly in areas supporting low growing turf plants and amongst fallen tree branches (Clayton et al. 1995). Sediment sampling down to 3 m water depth also revealed viable tubers. However, no plants or regrowth occurred in areas of the lake between 1.5 and 4.5 m deep, the predominant range of hydrilla before grass carp introduction (Clayton et al. 1995). Annual (April) vegetation survey of Elands Lake has continued since then, with a hydrilla plant last found in 2003, and more recent surveys reporting only the continued presence of the turf plant community (Hofstra et al. 2008) and young raupo (Hofstra et al. 2004b).

The Elands Lake grass carp trial demonstrated the effectiveness of grass carp at removing hydrilla, providing proof of concept for the use of grass carp as a tool in the MPI hydrilla eradication response (MAF 2008).

Although the native plants that were present in the lake had a limited distribution and abundance with hydrilla dominating the littoral zone (Neale unpublished report ca 1988), they were further reduced by grass carp browsing. This included erect emergent plants such as raupo and *Schoenoplectus tabernaemontanii* and submerged plants such as *Potamogeton ochreatus, P. cheesemanii*, *Chara australis, Nitella* sp. aff. *cristata* and *Myriophyllum propinquum.* Only plants that were inaccessible to the grass carp persisted, such as low growing (turf) species (e.g. *Myriophyllum propinquum, Elatine gratioloides* and *Glossostigma elatinoides*) on gently shelving slopes with shallow (c. 0.5 m) water. With the concept of restricting grass carp access in mind, in 2004, DOC and NIWA installed cages in shallow water that could exclude grass carp grazing and provide an opportunity to access the regeneration of native plants (Hofstra et al. 2004b). Despite additional challenges of aerial top dressing, and continued stock access to the lake, which is believed to have resulted in compromised water quality, submerged native plants of *P. ochreatus* have been recorded from two of the six cages. These results have highlighted the potential of exclusion cages to allow native flora to establish in select areas while grass carp are still present for longer term weed control.

8.5 Future Prospects

The future of invasive aquatic plants and the threats they pose to the lakes and waterways of New Zealand are linked to the continuing arrival of new plants, and new incursions of invasive species and the role that climate change may play in the ability of these pest species to establish, thrive and expand their range. The tool box for weed control must continually be developed to meet the new challenges of invasive aquatic plants in an effective and sustainable way, through the smarter use of existing products and new techniques validated by scientific research.

8.5.1 New Species Detections and 'Sleeping Giants'

Any potential importer of aquatic plants not already present in New Zealand is required to apply to the New Zealand Environmental Protection Authority outlining the potential effects of the species on the environment, human health, society, Māori culture and traditions and the market economy under the Hazardous Substances and New Organisms Act (1996). Costs for the provision of information and assessment are borne by the proposed importer. This is an expensive process compared with any potential gain the importer may accrue from importation and no aquatic plants have been assessed for importation in the past two decades (Champion et al. 2014).

Unfortunately, illegal importation apparently continues, with imported aquatic plants not being screened via relevant Import Health Standards with biosecurity risks posed both by the imported plants and any associated organisms (as documented by Keller and Lodge 2007; Duggan 2010). Champion and Clayton (2000) found that 27% of aquatic plants available from aquarists and nurseries were unknown at the last census of species in the 1980s and were unlikely to have been legally imported. Since that time, a number of consignments of aquarium plants including species new to New Zealand have been intercepted at the International Mail Centre and two successful prosecutions under the Biosecurity Act have resulted: one contained the weed hydrilla (Champion et al. 2014).

In addition to interceptions at the border, on-line plant sales regularly list species on the National Pest Plant Accord (2002) and in 2014 Amazon sponge plant (*Limnobium stolonifera*), a plant unknown from New Zealand, was offered for sale. Subsequent culture of this plant resulted in its identification as water lettuce (*Pistia stratiotes*) a Notifiable Organism thought to have been eradicated from New Zealand (Champion et al. 2014). *Eichhornia crassipes* and *Salvinia molesta* are two other free-floating Notifiable Organisms that have been targeted for eradication for at least the last 27 years (Champion and Clayton 2003). Approximately 250 field populations of these plants are now considered eradicated but new field populations are continually discovered. New infestations presumably originate from plants maintained in cultivation for ornamental purposes and in order to breed tropical fish, despite their cultivation being illegal under the Biosecurity Act and previous noxious plant legislation (Champion and Clayton 2003; Yamoah et al. 2013).

Despite New Zealand being, quite rightly, heralded as a leading nation in the field of biosecurity best practice (Meyerson and Reaser 2002; Caffrey et al. 2014) illegal importation and non-compliance amongst residents still short-circuit management efforts to reduce biosecurity risks posed by cultivated aquatic plants.

8.5.2 Climate Change

Conservative estimates of climate change on New Zealand temperatures would see a mean increase of 2.1 °C by the end of the century. Other impacts include increased salinisation of coastal water bodies, increased periods of extreme winds, longer and more extreme dry periods, with various studies suggesting that climate warming can stimulate eutrophication (Hamilton et al. 2013).

Globally, potential impacts of climate change on invasive aquatic weeds include increased likelihood of new species introductions, increased likelihood of naturalisation and reduced resistance of ecosystems to invasive species. In the worst case scenario, climate-driven invasions could lead to completely transformed ecosystems where alien species dominate, leading to reduced diversity of native species (Walther et al. 2009).

In New Zealand, the most immediate threat to lake ecosystems comes from the increased likelihood of naturalisation of aquatic plants already present in the country, e.g. in the aquarium trade (Champion and Clayton 2001), as well as expansion of current naturalised populations of invasive weeds (Champion 2014a). However, colonisation opportunity for potential pests has been limited by management, such as by bans on sale and distribution, including a National Pest Plant Accord (Champion et al. 2014) and also through ongoing management programmes effectively preventing range expansion of weeds such as water hyacinth and salvinia (Clayton 1990).

Potential habitat availability for invasive weed species such as egeria and hornwort are likely to be increased with increased water temperature and eutrophication. Hyldgaard and Brix (2012) found that invasive hornwort from New Zealand acclimated better to elevated temperatures than non-invasive European hornwort plants, suggesting weed performance could be enhanced under climate change. Egeria appears most invasive in eutrophic conditions and has also been associated with weed bed collapses and regime shift to turbid state in shallow northern lakes (Champion 2002; Schallenberg and Sorrell 2009). Hamilton et al. (2013) consider that more southern shallow water bodies are at risk of regime shift under climate warming. However, distribution of these weeds that reproduce vegetatively is best explained by variables characterising human access and use (Compton et al. 2012) and human-mediated actions are still the main driver of their spread.

Increases in water storage structures and distributional infrastructure driven by climate change and limited water resources are likely to increase the opportunities for spread and establishment of aquatic weeds and a greater frequency of extreme rainfall events under climate change scenarios may lead to increased flood-mediated transfer between water bodies (Hamilton et al. 2013).

Continued biosecurity management will be required to reduce the impacts of climate change on invasive weed establishment and spread including identification, assessment and pre-emptive management of new threats (both internationally and within New Zealand), surveillance and incursion response.

8.5.3 The Tool Box

The use of herbicides to control aquatic weeds has always been controversial, with many perceived risks posed by their use including toxicity to humans and other animals, non-target plants and algae and the environment in general, bioaccumulation through the food chain and environmental persistence. Many of the gains made in the management of aquatic weeds have been due to the availability of selective herbicides permitting their control without damage to native species. The Hazardous Substances and New Organisms Act (HSNO 1996) regulates these substances and applies controls that manage the risks associated with their application onto or into water (EPA 2013).

The number of herbicides available for aquatic use are limited, with diquat available since the 1960s and glyphosate (for emergent species only) in the 1970s being the only products registered for this use until 2005. That year, the herbicide endothall was permitted for use in New Zealand based on extensive trial work undertaken by NIWA (Hofstra and Clayton 2001; Hofstra et al. 2001), driven by the requirement to find a product that would control hydrilla and would also be effective in turbid waters for the management of pest plants such as hornwort. Funding for registration was provided by a consortium of regional and central government authorities and power generation companies.

In 2013, four additional herbicides (triclopyr trimethylamine, imazapyr isopropylamine, metsulfuron methyl and haloxyfop-R-methyl) were reassessed for aquatic use (to control emergent weeds) by the New Zealand Environmental Protection Authority (EPA). Part of this process involved a national EPA environmental meeting to discuss the ecotoxicology and environmental fate of these herbicides and a Hazardous Substances and Organisms Act (1996) hearing in 2012 (New Zealand Herald 2012). Based on the evidence presented at this hearing, the EPA modified the approvals for those herbicides so they can now be applied onto or into water as herbicides to control aquatic pest plants. The EPA considers these substances beneficial in the control of aquatic pest plants and more effective than other methods of control (EPA 2013). A number of controls including notification of affected persons (e.g. adjacent landowners, iwi, irrigators etc.), appropriate signage, maximum application rates and monitoring to ensure environmental exposure limits (EEL) and tolerable exposure limits (TEL) are not exceeded in the receiving environment (EPA 2013).

With the approval of aquatic use of these herbicides and EPA recognition that such herbicides provide the best currently available control techniques, their continued use as restoration tools for the reduction of aquatic pest plant impacts appears secure.

Classical biological control (CBC) has had limited application on aquatic plants in New Zealand. Alligator weed is a semi-aquatic plant that also grows on dry land and this is the only ‘aquatic’ species to date that has had a CBC agent released, with mixed results (Stewart et al. 1999). The potential for further release of biocontrol agents on submerged aquatic plants is under consideration, however, the challenges are significant and the potential outcomes unknown at this stage. An alternative to CBC is the use of a bioherbicide, which is a plant pathogen grown in culture and inundatively applied to target submerged weed species (similar to chemical application). A fungal mycoherbicide (*Mycoleptodiscus terrestris*) has been cultured and applied successfully to kill submerged plants in laboratory and outdoor mesocosm settings (Hofstra et al. 2004a; Shearer 2009), but field trials to date have not been as successful. The use of ‘living’ organisms to control aquatic pest plants offers some promise for the future, but requires further research.

In conclusion, control of invasive aquatic plants is a dynamic and advancing field. One only needs to reflect on history to appreciate how much progress has been made to provide selective, safe and effective tools or methods to control aquatic weeds. Research should continue to refine existing tools and explore new opportunities that help better manage invasive species, while also enabling and facilitating restoration of native values.

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Chapter 9
Management of Non-indigenous Lacustrine Animals

Ian C. Duggan and Kevin J. Collier

Abstract Numerous non-indigenous species have established populations in New Zealand lake ecosystems, including some relatively recently, and many others are likely unrecognised. Control or eradication as part of lake restoration programmes is more achievable for vertebrate than invertebrate species once established. Focus for management of smaller species must, therefore, be placed on preventing initial establishment rather than post-establishment control. Eradication of the Australian marron (*Cherax tenuimanus*) from New Zealand is a rare example of eradication success for an invertebrate. Even for vertebrates, while control is possible with ongoing effort, complete eradication is typically difficult, even within a single waterbody. This has been achieved using piscicides and drainage in some parts of New Zealand, although examples exist overseas of successful eradication of fish through concerted integrated management. Netting, trapping, electro-fishing and use of cages, baits and one-way barriers have been used for control purposes in New Zealand, but the ecological outcomes have largely gone unmonitored. Emerging and future technologies that may assist in the management of non-indigenous fish include the use of pheromones to enhance capture rates, the introduction of taxon-specific pathogens and the genetic modification of fish to produce single-sex (male-only) progeny. Environmental DNA (eDNA) techniques show promise for early detection of vertebrate invaders. Warming of lakes due to climate change may increase the number of species able to establish populations and also the potential for serious parasites to be carried by some invaders.

Keywords Non-indigenous species · Eradication · Invasive species · Biosecurity · New Zealand

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9.1 Introduction

Invasions by non-indigenous species are a multistep process, involving entrainment in a vector, transportation to a new site, introduction, followed by establishment and spread (Williamson 1996; Feature Box 9.1). Management of non-indigenous aquatic species can occur at any stage along this process and may involve prevention of establishment, control of numbers once established, or eradication of species (Feature Box 9.2). Such management is essential for the conservation of native flora and fauna, and numerous methods are employed in New Zealand for control or eradication of terrestrial pests. Control of non-indigenous species is also important for achieving biodiversity and water quality outcomes in lake restoration, but methods are much less advanced for freshwater compared to terrestrial ecosystems.

Invasions are commonly self-perpetuating with reproduction allowing the problem to potentially grow and spread through time, unlike water quality issues which will typically abate once the source activities cease. In the same way that the water quality of a lake cannot be restored long-term without considering the catchment, management of invasions cannot be considered without taking into account species not yet introduced to a site. Thus, in-lake control or eradication of non-indigenous animals makes little sense without reducing the probabilities of invasion or re-invasion. Eradication of smaller species in lakes, particularly invertebrates that produce resistant diapausing stages, is typically impossible once establishment has occurred, making restoration to a predetermined biological state out of the question (Galil 2002; Duggan et al. 2006). As such, we take a more holistic view of ‘restoration’, not just focussing on in-lake restoration, but also on prevention—as the old adage goes, ‘an ounce of prevention is worth a pound of cure’ (Leung et al. 2002).

In this chapter, we review invasions of animals into New Zealand lakes, beginning with inventories of non-indigenous species, including their distributions and known effects. We limit ourselves where possible to free-living animal species that have been accidentally or deliberately released by humans into standing freshwaters of New Zealand’s main islands. Thus, we do not consider species living in saline coastal lakes, species found to date only in flowing waters, introduced parasites of freshwater species, offshore or sub-Antarctic islands, or species seemingly self-introduced since human arrival. We also limit ourselves to species that spend a significant amount of their lives within freshwater (i.e. non-native waterfowl, which primarily live on the water, are not considered). Translocations of native species are also not considered, although they may compromise lake restoration outcomes through effects on food webs; for example, the native common smelt (*Retropinna retropinna*) has been introduced into many North Island lakes (see Chap. 10) to provide food for introduced trout. We review management techniques for non-indigenous species, including pre-border, border and post-border controls, and in-lake restoration approaches involving invasive species management.

Box 9.1 Ballast Water Management in the Laurentian Great Lakes

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Ballast water has been a principal vector for introduction of non-indigenous species (NIS) since its advent approximately one century ago. Ballast water is rather unique among vectors transmitting NIS in that it may harbour an enormous diversity of species, each present at different abundance. In the Great Lakes, ballast water accounts for at least 55% of established NIS reported since the modern Seaway opened in 1959. Species introduced originate primarily from Europe, with a strong contingent from the Black-Caspian Sea region. This contingent includes highly invasive molluscs (zebra and quagga mussels), fishes (round goby) and crustaceans (fishhook waterfleas), all of which entered the lakes in the last 30 years. Benthic invertebrate biomass in regions of the Great Lakes may be heavily dominated by these species (e.g. >99% dreissenid mussels in Lake Erie).

Global trade has resulted in new opportunities for NIS to colonise areas that were previously inaccessible. This pattern may result from hub-and-spoke or stepping stone invasions patterns. In addition, invasions by NIS are unlike traditional forms of pollution which, when countered by an appropriate management action, may abate in severity. By contrast, self-replicating NIS may continue to plague affected systems even after vector management has been implemented. It is for this reason that preventing further invasions is essential to reducing spread of even more NIS. Regulatory ballast water management was first implemented in the Great Lakes in 1993 for vessels arriving from overseas with filled ballast tanks. This policy was then extended to loaded vessels carrying only ballast residuals, beginning in 2006. Currently, all vessels arriving from foreign destinations must ensure that ballast water or residuals have been exchanged or flushed on the open ocean, thereby minimising opportunities for freshwater NIS to remain in tanks upon entry to the system. The policy appears to be successful in that no new ballast-mediated NIS have been reported in the system since 2006. This dramatic reduction in apparent invasion rate seemingly occurred as a result of a reduction in propagule pressure for freshwater invaders carried rather than the number of high risk species carried in the tanks. In this regard, the current ballast water issue for the Great Lakes resembles that in New Zealand, where vessels do not enter freshwater and, thus, are incapable of discharging NIS into vulnerable ecosystems.

Box 9.2 Carp Eradication: Tasmania

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After *Cyprinus carpio* (European carp) were found in lakes Crescent and Sorell in the central highlands of Tasmania, the Inland Fisheries Service (IFS) established the Carp Management Program (CMP). The CMP has successfully eradicated carp from Lake Crescent, using a range of techniques that continue to be used in Lake Sorell today. In January 1995, an angler found a fish being eaten by a sea eagle on a bank at Lake Crescent. He handed the remains of the fish to the IFS which confirmed it was carp. The Service then surveyed the lake using backpack electrofishing and confirmed the presence of carp. Soon after, carp were also found in Lake Sorell. Both lakes were closed to the public using provisions of the Inland Fisheries Act 1959. As part of the CMP, outflows from both lakes were modified to enable screening of water to 1 mm. Twenty years of surveys show that the screens have been successful as no carp have been found downstream from the lakes. The CMP uses persistent physical effort through a range of techniques including biotelemetry, trapping, netting, electrofishing, spot poisoning, spawning sabotage and pheromone attractants. 7797 carp were removed from Lake Crescent. The last wild carp, a mature female, was caught in December 2007. This is a significant result given the 12 years of effort including the development of the approach that integrates such a broad range of techniques. Lake Crescent was declared carp free in 2009 (see Fig. 9.1). In 2009, estimates indicated that carp numbers in Lake Sorell had been reduced to below 50. However, efforts were setback when the carp were able to spawn. The CMP's current priority is to prevent any further spawning while continuing to fish out the 2009 cohort. About 14 km of purpose built barrier netting is being used to block the extensive wetlands that are preferred spawning sites. Over 40,000 carp have been removed from Lake Sorell. A capture/mark/recapture population study that started in 2011 indicates that around 7000 maturing carp remain. These fish grow slowly in Tasmanian conditions with some males beginning to mature in their fourth year with the first of the females at least a year behind. The CMP continues to refine old methods and develop new in its integrated approach to carp eradication.

9.2 Inventories and Distributions

Numerous aquatic species have established non-indigenous populations in New Zealand lakes, while many others are currently entrained in vectors such as aquaria (e.g. numerous fish species) or freshwater aquaculture facilities (e.g. Malaysian freshwater prawns, *Macrobrachium rosenbergii*, and grass carp,

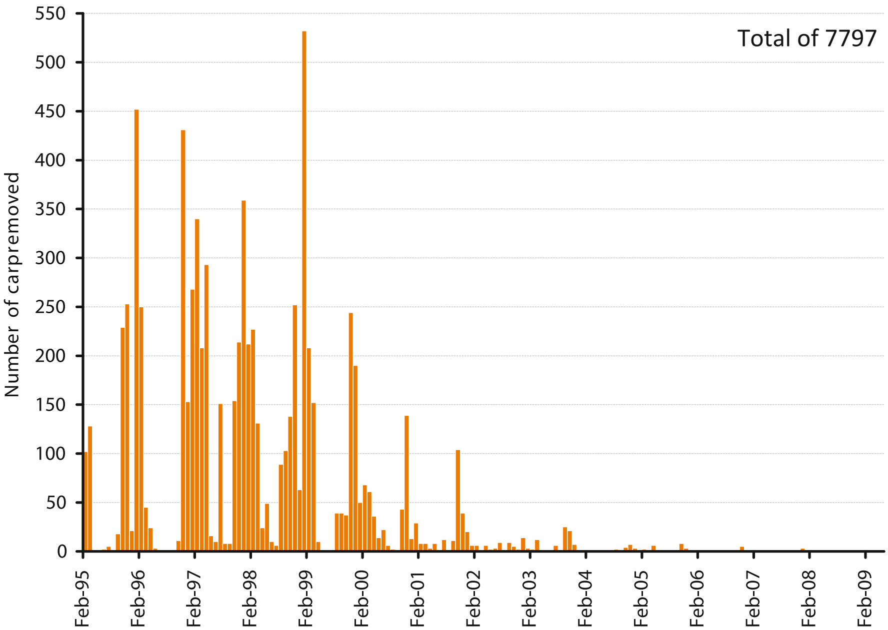

Fig. 9.1 Monthly carp captures from Lake Crescent

Ctenopharyngodon idella). Some species have been introduced to natural waters for specific purposes (e.g. the grass carp for aquatic weed control) but fail to establish self-sustaining populations because they are unable to reproduce in the recipient environments. The following section provides an overview of non-indigenous aquatic invertebrate and vertebrate species that have established populations in standing freshwaters in New Zealand.

9.2.1 Invertebrates

At least 23 species of non-indigenous invertebrate have established populations in New Zealand freshwater lakes (Table 9.1), and it is likely a number of others are currently unrecognised. The vast majority of these non-indigenous invertebrates have been introduced unintentionally, with many entering incidentally through the aquarium trade. The exceptions are *Lymnaea stagnalis* introduced as trout food (Pullan et al. 1972) and marron (*Cherax tenuimanus*) introduced for aquaculture purposes. In general, the more recent species recorded as established are smaller in size than those introduced earlier, most likely reflecting more stringent border controls.

At least eight zooplankton species have established non-indigenous populations in New Zealand lakes, four of which have been discovered since 2000 (Table 9.1).

Table 9.1 Non-indigenous invertebrate species established in standing waters

Class/Subclass Species	Common name	Establishment/ discovery in wild	Native area	Reason
Hydrozoa				
Craspedacusta sowerbii	Freshwater jellyfish	1950s	China	Accidental/ Unknown
Crustacea				
Cherax tenuimanus	Marron	2005[a]	Western Australia	Aquaculture
Daphnia galeata	Daphnia	1993	North America	Accidental/ Unknown
Daphnia obtusa	Daphnia	<1976	Widespread in Northern hemisphere, South America and Africa	Accidental/ Unknown
Daphnia 'pulex'	Daphnia	2005	North America	Accidental/ Unknown
Boeckella symmetrica	Calanoid copepod	<1967	Australia	Accidental/ Unknown
Boeckella minuta	Calanoid copepod	1966	Australia	Accidental/ Unknown
Calamoecia ampulla	Calanoid copepod	<1984	Australia	Accidental/ Unknown
Sinodiaptomus valkanovi	Calanoid copepod	2000	Japan	Accidental/ Unknown
Skistodiaptomus pallidus	Calanoid copepod	2000	North America	Accidental/ Unknown
Mollusca				
Physa acuta	Left-handed pond snail	<<1956[b]	Holarctic	Accidental: Aquarium trade
Planorbarius corneus	Great rams-horn snail	1968	Palearctic	Accidental: Aquarium trade
Planorbella sp.		2007	North America	Accidental/ Unknown
Pseudosuccinea columella	American ribbed fluke snail	1940	Holarctic	Accidental: Aquarium trade
Lymnaea stagnalis	Great pond snail	1864	Holarctic	Intentional: Trout food
Galba (*Lymnaea*) *truncatula*	Dwarf pond snail	1890	Europe	Accidental: Aquarium trade
Radix (*Lymnaea*) *auricularia*	Ear pond snail	1977	Holarctic	Accidental: Aquarium trade

(continued)

Table 9.1 (continued)

Class/Subclass Species	Common name	Establishment/ discovery in wild	Native area	Reason
Hippeutis complanatus	Flat rams-horn snail	1978	Palearctic	Accidental/ Unknown
Pisidium casertanum	Pea cockle	1862	Cosmopolitan	Accidental/ Unknown
Hirudinea				
Barbronia weberi	Asian fresh-water leech	1976	Asia	Aquarium trade
Oligochaeta				
Lumbriculus variegatus	Blackworm	<1971	Holarctic	Accidental/ Unknown
Branchiura sowerbyi	Annelid worm	<1975	Asia	Accidental/ Unknown
Insecta				
Culex (Culex) quinquefasciatus	Brown mosquito	1830s	Cosmopolitan	Accidental/ Unknown

[a]The species is likely no longer present in New Zealand
[b]Confused taxonomy prior to this time

All are likely to have been unintentionally introduced with flora or fauna intentionally imported or in association with contaminated equipment (e.g. fishing gear, dam building equipment). Three are *Daphnia* species, adding to our two currently recognised native *Daphnia* species (Burns et al. 2017). The North American *Daphnia galeata* has become the most common and widespread *Daphnia* species in the North Island since its first record in 2000, and it is now also spreading through the South Island (early records of *Daphnia dentifera* reported by Duggan et al. (2006) have since been confirmed as *D. galeata*). Another North American species, *Daphnia* '*pulex*', was first observed in South Island lakes in 2005 and is now spreading rapidly throughout that island (Duggan et al. 2012). More recently, this species has also been found to be common in Auckland ponds (Branford and Duggan 2017). *Daphnia obtusa* has been recorded near Dunedin (Chapman et al. 2011), but has not been observed for a number of years, either through lack of sampling or because it has been extirpated. If present, the identity of this species requires re-evaluation, as the taxonomy of this and closely related species has long been confused and is only being clarified through recent genetic advances (Benzie 2005). However, *D. obtusa* has also been recognised in Australia from a dam impoundment constructed in 1964, indicating it is non-indigenous there also (Benzie and Hodges 1996). Further, a record of *Daphnia lumholtzi*, which is native to Africa, Australia and Asia, in Lake Rotoroa, South Island (Jolly 1955), is doubtful.

The effects of non-native *Daphnia* species in New Zealand are unclear. However, *Daphnia*, including the native species, were uncommon inhabitants of New Zealand lakes prior to 1990, but they are now extremely widespread and abundant and, as noted earlier, populations are dominated by non-indigenous species. Balvert et al. (2009) noted a large reduction in rotifer densities, likely resulting from the superior competitive abilities of *Daphnia galeata* following its invasion in a North Island lake. In addition, due to their high efficiency as filter-feeders relative to other zooplankton, a large increase in water clarity followed, at least in the short-term (Balvert et al. 2009). Being larger than most native zooplankton, the new *Daphnia* species may alter energy flows to higher consumer levels through, for example, greater susceptibilities to predation than smaller species.

Five non-indigenous calanoid copepod species have been recorded, primarily associated with constructed waters (e.g. dams, retired quarries, ornamental ponds; Banks and Duggan 2009). Due to distinct morphologies and geographical distributions within the group, species can be readily identified and their origin determined. The Australian species *Boeckella symmetrica* and *Boeckella minuta* have been known for many years from the North Island, confined to water supply and hydroelectricity reservoirs, a retired quarry and an ornamental pond. The North American species *Skistodiaptomus pallidus* and the Japanese *Sinodiaptomus valkanovi* were both found in 2000, but likely existed in New Zealand for longer, being found initially in constructed urban ponds not typically sampled by New Zealand freshwater biologists (Duggan et al. 2006). All of these calanoid copepods were, until recently, thought to be confined to constructed waters.

The populations of *S. valkanovi* can be traced genetically to the north of Honshu, Japan (Makino et al. 2010), but as with other calanoid copepod species the localities provide little clue as to how this species came to be in New Zealand. This species, first described from a botanic garden in Bulgaria, was recorded in New Zealand from the Auckland Wintergardens glasshouse, suggesting that it may have been moved in association with aquatic macrophytes. *Skistodiaptomus pallidus* was also first recorded in a botanic garden (Auckland Regional Botanic Garden ponds) and was found soon after among *Daphnia carinata* cultures being sold as live food in aquarium stores (Duggan et al. 2006), highlighting another potential vector of zooplankton invasion. *Skistodiaptomus pallidus* has recently spread into natural lakes, and this has been linked to movement of grass carp for the control of macrophytes (Duggan et al. 2014; Branford and Duggan 2017). Although all recent records have been from lakes where grass carp have been released, some of the earliest records are not, indicating this species may have been introduced by other means and is being spread by different vectors. Finally, the Australian species *Calamoecia ampulla* was recorded once at an unknown South Island location. Overall, little is known of the ecological effects of these calanoid copepod species, although in the longitudinal study of Duggan et al. (2014) from Lake Kereta, Auckland region, *S. pallidus* was observed to reach high abundances following establishment and suppressed the abundances of the native calanoid *Calamoecia lucasi*.

The majority (nine) of the remaining non-indigenous invertebrate species are molluscs, and all but one of these are gastropod species (Table 9.1). Most will have arrived in New Zealand in association with aquatic plants and many have spread widely in lakes. The snail species all feed on benthic algae and as such may compete with other grazers in New Zealand. For example, *Physa acuta* is thought to have displaced the native *Glyptophysa variabilis* in much of its range (Winterbourn 1973; Collier 1993). Also, aquatic snails are intermediate hosts for a number of diseases.

Lymnaea stagnalis is an intermediate host of the fluke *Echinostoma*, the cause of echinostomiasis in humans, while *Planorbarius corneus* can carry the gapeworm nematode (*Syngamus trachea*) which affects birds, and *P. acuta* is likely an intermediate host for swimmer's itch (Champion et al. 2012). In addition to species listed from standing waters in Table 9.1, *Melanoides tuberculata* is established in a geothermal stream at Golden Springs (Duggan 2002), and an individual apple snail *Pomacea diffusa* has been recorded from the Waikato River (Collier et al. 2011); neither species would be likely to form self-sustaining populations in lakes, except under circumstances where water is heated through geothermal activity or by industrial cooling-water discharges.

The freshwater jellyfish *Craspedacusta sowerbii*, native to the Yangtze Valley in China, was first recorded in New Zealand in the 1950s. Observations to date have primarily been of the medusa (jellyfish) stage, which periodically blooms in some lakes. Boothroyd et al. (2002) examined the feeding of the medusa and determined that although zooplanktivorous, they are unlikely to have a major impact on the zooplankton community except possibly a temporally limited impact during bloom periods. A recent survey of *Craspedacusta* polyps showed the species to be in a high proportion of North Island lakes, in places where medusa have never been observed (Duggan and Eastwood 2012); nothing is known of the feeding effects of this life stage in New Zealand.

At least two annelid species are considered potentially non-indigenous in standing waters in New Zealand, although they occur in a broad range of habitats internationally; the leech *Barbronia weberi* (Mason 1976—from man-made fish ponds) and the oligochaete *Lumbriculus variegatus* (Brinkhurst 1971; Talbot and Ward 1987—from lakes). When first observed in New Zealand, Mason (1976) noted *B. weberi* was found only in artificial environments such as man-made fish ponds or aquaria, giving clues to its vector for introduction. Where introduced elsewhere, it is also noted that it has a close association with common aquarium plants (e.g. Genoni and Fazzone 2008). *Branchiura sowerbyi* has been found in flowing waters in New Zealand, but elsewhere is commonly found in stagnant ponds (e.g. Aston 1968; Georgieva et al. 2012); as such, it likely inhabits New Zealand ponds where few benthic macroinvertebrate studies have been conducted. Another non-indigenous leech, *Helobdella europaea* (Collier et al. 2014), and the non-indigenous oligochaetes, *Eiseniella tetraedra* (Marshall and Winterbourn 1978) and *Stylodrilus heringianus* (Marshall 1978), are also known only from New Zealand flowing waters; a number of Naididae species are possibly also non-indigenous in New Zealand (Glasby et al. 2007). All these species currently occupying lotic habitats in New Zealand also have the potential to invade standing waters.

Finally, several non-indigenous insects are considered established or vagrant in New Zealand. The mosquito *Culex quinquefasciatus* has been recorded widely in New Zealand from small artificial standing waters (e.g. tyres, gully traps) and is a known vector elsewhere for the parasitic roundworm *Wuchereria bancrofti* (the major cause of lymphatic filariasis), *Plasmodium* (avian malaria), myxomatosis, and other diseases (Holder 1999). Other insect species, including the damselfly *Ischnura aurora*; the dragonflies *Hemianax papuensis, Tramea loewii* and

Hemicordulia australiae (Rowe 1987); waterboatman *Agraptocorixa hirtifrons* (Young 2010); the whirligig beetle *Gyrinus convexiusculus* (Wise 1989); and the lacewing *Sisyra rufistigma* (Wise 1998), are likely self-introduced from Australia and are thus not included in Table 9.1.

A number of other invertebrate species are likely to have invaded freshwaters, but have not yet been discovered or recognised. For example, Duggan (2010) found two species of harpacticoid copepod, *Nitokra pietschmanni* and *Elaphoidella sewelli*, amongst the bottom sediments of home aquaria. Although these are both likely to survive in New Zealand lakes, neither has yet been recorded in the wild, although perhaps due only to a lack of expertise and surveys for these taxa. The ostracod *Eucypris virens*, otherwise considered native to the Holarctic, is considered an invader in Australia (Koenders et al. 2012) and has been recorded in New Zealand from temporary waterbodies and farm ponds around Auckland (Barclay 1968). Genetic analysis of this species, and other small invertebrates shared between New Zealand and elsewhere, is required to better determine their status as native or non-indigenous.

9.2.2 Fish

Of the 63 freshwater fish species known to occur in New Zealand (i.e. outside of captivity), 21 are introduced and 19 are (or were) considered established; 17 of these species spend a significant period in freshwater lakes (Table 9.2). All of these non-indigenous fish have been deliberately introduced, either legally, particularly by acclimatisation societies as sport fish, or illegally with a range of intentions. Several of these fish have the potential to compromise lake restoration goals when present in high numbers, for example, by exacerbating other pressures and adversely affecting water quality, habitat and/or native biodiversity. Mechanisms for these effects include (1) bioturbation that can decrease water clarity and, along with excretion, transfer bioavailable nutrients to the water column, (2) degradation of habitat through mobilisation of sediment and direct consumption of aquatic plants and (3) food-web effects through top-down or bottom-up control of other biota (Collier et al. 2015). The larvae of all invasive fish species feed on zooplankton and the high fecundity of coarse fish in particular means there is considerable potential for their larvae to have cascading effects on algal grazing by zooplankton and, therefore, water quality.

Salmonids, mainly brown trout (*Salmo trutta*) and rainbow trout (*Oncorhynchus mykiss*), occur throughout the country, but are less common in warmer parts of northern North Island. There are four species of trout and two species of salmon present in New Zealand, but only brown and rainbow trout are widespread in lakes; feral stocks of *Salmo salar* are considered close to extinction. Trout typically spawn in lake tributaries but can also use exposed shorelines where conditions are suitable, although generally spawning habitat is limited and stocking occurs for recreational fishing purposes (Rowe and Graynoth 2000). Adult trout are opportunistic feeders in lakes, including on native fish and large invertebrates. The other trout species and

Table 9.2 Non-indigenous vertebrate species established in standing waters

Species	Common name	Establishment/ discovery in wild	Native area	Reason
Salmo salar	Atlantic salmon	1868[a]	Northern Atlantic	Sport
Salmo trutta	Brown trout	1867	Europe and North Africa	Sport
Oncorhynchus mykiss	Rainbow trout	1883	Northern Pacific Ocean	Sport
Oncorhynchus nerka	Sockeye salmon	1902	Northern Pacific Ocean	Sport
Oncorhynchus tshawytscha	Chinook (quinnat) salmon	1875	Northern Pacific	Sport
Salvelinus fontinalis	Brook char	1877	Eastern North America	Sport
Salvelinus namaycush	Mackinaw/lake char	1906	Northern North America	Sport
Carassius auratus	Goldfish	1860s	Eastern Asia	Ornamental
Cyprinus carpio	Common carp	Late-1960s	Central Europe, Asia	Illegally smuggled
Tinca tinca	Tench	1868	Temperate Europe and Asia	Sport
Scardinius erythrophthalmus	Rudd	1967[b]	Western Europe to central Asia	Illegally smuggled
Leuciscus idus	Orfe	1980s	Western Europe to Asia	Illegal
Perca fluviatilis	Perch	1860s	Europe	Sport
Ameiurus nebulosus	Brown bullhead catfish	1877	North America	Unknown
Gambusia affinis	Gambusia (mosquitofish)	1930	Northern Gulf of Mexico	Biocontrol
Poecilia latipinna	Sailfin molly	<1967	Gulf of Mexico, south-east USA	Ornamental
Litoria aurea	Green frog	1867	Australia	Biocontrol
Litoria raniformis	Golden bell frog	1867	Australia	Biocontrol

[a]The species is likely no longer present in New Zealand
[b]First introduced in 1868, but did not establish populations at that time

salmon have restricted distributions and are rarely abundant, although tiger trout (brown trout-brook trout hybrids) are stocked into some Rotorua lakes (D.K. Rowe, personal communication).

Common carp (*Cyprinus carpio*) are widespread in northern New Zealand, but do not occur in the South Island following eradication of an isolated incursion. Carp can be considered as ‘nutrient pumps’, consuming organic sediments and excreting bioavailable nutrients into the water column, contributing to elevated levels of chlorophyll *a* and cyanobacteria (Weber and Brown 2015). Carp rarely consume macrophytes directly but dislodge roots from fine sediment during feeding, and the increased turbidity they create attenuates light for macrophyte growth (Weber and Brown 2009). Carp feeding also undermines banks and degrades benthic habitats, partly accounting for an observed negative relationship between carp biomass and benthic invertebrate taxa richness (Vilizzi et al. 2014). Carp move into interconnected shallow lakes, as well as inundated floodplains and wetlands, in spring and summer where they congregate in spawning aggregations in shallow, weedy areas (Tempero et al. 2006; Hicks and Ling 2015).

Goldfish (*Carassius auratus*) occur throughout both the North and South Islands but are much more numerous in the north. Goldfish can withstand temperatures from freezing to over 35 °C, very low oxygen levels and have been associated with adverse impacts on water quality through resuspension of sediment and nutrients during feeding, in a similar way to carp. It has been shown that growth of cyanobacteria is stimulated by passage through goldfish intestines (Kolmakov and Gladyshev 2003), suggesting high numbers may contribute to the development of algal blooms in enriched waterbodies (Morgan and Beatty 2007). Their main impact on native fish is likely to be through competition for food and other resources. Spawning occurs in spring and summer among macrophytes in shallow margins of lakes (Hicks et al. 2010).

The catfish brown bullhead (*Ameiurus nebulosus*) is widespread throughout the middle of the North Island with scattered records from the South Island, and it has been associated with impacts on benthic and littoral food-webs in lakes. In Lake Taupo, for example, catfish from weedy habitats were found to feed predominantly on gastropods, cadddisflies, cladocerans and chironomids (Barnes and Hicks 2003), and elsewhere they have also been reported to feed on eggs of native fish and extensively on crayfish (Rowe and Graynoth 2000; Clearwater et al. 2014). The wide-ranging diet of catfish makes them likely to both compete with other benthivorous native fish such as eels and they can include native fish in their diets (Collier et al. 2018). They can survive long periods out of water if the skin is kept moist and are, therefore, extremely resilient to adverse conditions, difficult to eradicate and easy to spread inadvertently. As well as modifying invertebrate communities, their benthic feeding has the potential to affect ecosystem processes and nutrient status through stirring up bottom sediments. Thus, increased rates of nutrient cycling caused by catfish may contribute to higher productivity in lakes (e.g. Hicks et al. 2001).

Gambusia (*Gambusia affinis*) was introduced into New Zealand for mosquito control but is ineffective at this. The species occurs mainly in upper half of the North Island (north of Taupo), with populations also in Taranaki, Hawkes Bay, Manawatu and Wellington, but they are absent from the South Island except at a few sites in and around Nelson where eradication is underway (Grainger 2015). Gambusia proliferates in shallow margins, mainly around aquatic plants in summer and autumn months. Rapid growth and turnover makes them capable of colonising new habitats rapidly. Each female is capable of producing up to 130 live young and 2–3 broods between November and April each year, although this varies with size and up to 9 broods over 6–7 months have been reported (Pyke 2005). High numbers of gambusia have been associated with negative effects upon a range of fish, invertebrate and amphibian species worldwide, through direct predation or competition (e.g. Lloyd et al. 1986). In New Zealand, there are reports of gambusia feeding upon juveniles and fry of threatened black mudfish (*Neochanna diversus*) which can inhabit lake margins, aggressively attacking dwarf inanga (*Galaxias gracilis*) in a dune lake, forcing small indigenous fish species into deeper waters, and competing with native fish (e.g. bully, smelt) for space and food in shallow lake margins when they occur at high densities (e.g. Barrier and Hicks 1994; Rowe 1998; Ling 2004; Rowe et al. 2007). Gambusia is also responsible for the extinction of *Galaxias gracilis* in Lake Kai Iwi, Northland (Rowe 2003; Pingram 2005).

Rudd (*Scardinius erythrophthalmus*) have been introduced to many lakes in the North Island where they are a concern for native macrophyte populations because of the preference by adults for feeding on native aquatic plants, giving introduced macrophytes a competitive advantage (Wells 1999). Their feeding may also suppress the regeneration of macrophytes in turbid lakes and thereby prevent re-establishment of native *Nitella* and *Potamogeton* species in restoration efforts because they preferentially graze on growing tips or young plants (Lake et al. 2002). High densities of rudd may contribute to reduced water clarity in shallow lakes through increased wind disturbance following macrophyte loss and a reduction in large zooplankton from larval feeding. Because they only digest about 30% of plant material and damage plants while feeding, rudd release phosphorus to the water (Ravera and Jamet 1991), potentially promoting the succession of submerged macrophytes to algal dominance and eutrophication in lakes. Diet of small rudd (56–65 mm fork length) overlaps significantly with common smelt and shoaling galaxiids [e.g. dwarf inanga and koaro (*Galaxias brevipinnis*)], which feed at mid-water and from the surface, suggesting the possibility of competition for food resources with some native fish (Cadwallader 1977; Lake et al. 2002). Large fish can also prey on native bullies and populations increase rapidly as a result of multiple spawnings, making them difficult to control (Rowe and Graynoth 2000). Rudd can also have sociocultural impacts on trout angling as they can take lures more readily than trout leading to angler disinterest (see Rowe and Champion 1994).

Adult tench (*Tinca tinca*) inhabit shallow lakes, reservoirs, ponds and wetlands in the northern and central parts of the North Island and in scattered locations around the South Island. They are benthivorous and feed mainly at night, consuming benthic invertebrates and detritus, although some fish can feed solely on zooplankton (Rowe and Graynoth 2000). Tench thrive in both clear and turbid waters, so the high suspended solids levels occurring in turbid lowland New Zealand lakes are unlikely to affect them. Generally, tench seem unlikely to have direct effects on other fish which they do not eat, but have been implicated in reduced densities of some benthic invertebrates (Rowe 2004). Tench are also known to be selective planktivores and so may exert top-down effects (i.e. a reduction in zooplankton) on some lake ecosystems, thereby increasing phytoplankton and reducing water clarity (Rowe and Champion 1994). Tench prefer temperatures 20–21 °C and are rarely found in waters over 25 °C.

Perch (*Perca fluviatilis*) occur throughout most of the west coast of the North Island and the east coast of the South Island of New Zealand. In lakes and ponds, they are found mostly around margins or shallow water close to large beds of macrophytes and/or emergent plants such as rushes. In lakes, larval perch are generally pelagic zooplankton feeders that form shoals in shallow, open waters and along littoral zones. Adults are aggressive piscivores, and they have been implicated in the decline of native fish and crayfish populations in ponds, lakes and small South Island tarns and a northern New Zealand dune lake (Rowe and Smith 2001; Closs et al. 2003; Ludgate and Closs 2003). Stunted populations of perch can contribute to reduced water clarity through consumption of zooplankton and consequent reduction in algal grazing (Rowe 2007). However, reduction in adult perch numbers can lead to the proliferation of zooplanktivorous fish, and overseas perch stocking has been utilised to suppress these with the aim of improving water quality. Perch spawn during late winter–spring and eggs die when water temperatures rapidly increase above 12 °C; consequently, temperature is a key factor limiting perch distribution and growth in freshwaters.

Many lakes have multiple species of invasive fish and, in combination with native species, form novel communities with potentially complex and unknown interactions that may have additive, synergistic or even antagonistic effects on lake ecology and water quality (Feature Box 9.3). For example, Rowe (2007) reported that the number of fish introduced into small North Island lakes affected the relationship between water clarity and lake depth, and he concluded that control of just one species may not result in an improvement in water clarity because of the interacting effects of multi-species assemblages on lake trophic processes. Thus, where invasive species control forms part of a restoration action plan in lowland lakes, a knowledge of community-level interactions and trophic linkages is required to predict ecological and water quality outcomes.

9.2.3 Other Vertebrates

Of the three species of frog introduced from Australia, two bell frogs spend part of their lives in standing waters. *Litoria aurea* is restricted to the top half of the North Island, whereas *Litoria raniformis* is widespread throughout the country. Both were introduced for mosquito control. A major cause of concern regarding these bell frogs is that they serve as vectors of amphibian diseases, such as Chytridiomycosis, which was responsible for a dramatic decline in populations of the native frog *Leiopelma archeyi* on the Coromandel Peninsula in 2001 (Bishop 2008). *Litoria* tadpoles largely feed on algae and dead insects (Pyke and White 2001). Red-eared slider terrapins (*Trachemys scripta elegans*) and snake-neck turtles (*Chelodina longicollis*) have been recorded periodically in New Zealand lakes, but are not known to have established self-sustaining populations due to temperature requirements for breeding. Both species are long-lived and feed on a range of aquatic plants, small invertebrates and fish (e.g. Dreslik 1999).

Box 9.3 Managing Invasive Fish in the Laurentian Great Lakes; 'Darwin's Dreampond' or Nightmare?

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The Laurentian Great Lakes are likely the most heavily invaded freshwater ecosystem globally, with food webs dominated by non-native species that have altered ecosystem services and functions with huge consequential reductions in native biodiversity (Pagnucco et al. 2015; Ricciardi 2006). Yet, some have claimed that continual manipulation of the fishery has maintained a productive equilibrium, in contrast to devastation wrought by introductions in other great lakes ecosystems (e.g. Nile Perch in Lake Victoria, sensu Goldschmidt 1996). Management of these binational North American waters has certainly resulted in invasive species control success stories, notably the ongoing suppression of Sea Lamprey (*Petromyzon marinus*). A combination of traps, barriers, systematic monitoring and larval lampricide treatment has successfully sustained a 90% reduction in sea lamprey abundance (Christie and Goddard 2003). Similarly, stocking of Pacific salmon has significantly reduced populations of Alewife (*Alosa pseudoharengus*) and Rainbow Smelt (O*smerus mordax*) (Bunnell et al. 2013). This strategy has allowed Great Lakes fisheries managers to develop an economically important salmon fishery, while simultaneously controlling these two invasive prey fish, whose high abundance adversely affected native fisheries and in the case of the former regularly fouled beaches and water intakes during major die-off events.

However, over the last 15 years, the fish forage base has become increasingly unpredictable following invasive quagga mussel (*Dreissena bugensis*) proliferation, which has resulted in concentrations of nutrients in nearshore and benthic zones (Cuhel and Aguilar 2013) and increasing risk of fishery collapse. The recent failure of the Chinook Salmon (*Oncorhynchus tshawytscha*) fishery in Lake Huron (O'Keefe and Miller 2011) following a crash of the alewife population is case-in-point to this daunting management challenge. This nutrient disruption and the subsequent loss in offshore alewife and salmon production is observed across the Great Lakes in varying degrees, but has opened the door for the recovery of native fish such as Cisco (*Coregonus artedi*), Lake Trout (*Salvelinus namaycush*), Lake Whitefish (*C. clupeaformis*) and Walleye (*Sander vitreus*). Unfortunately, gains in native fish recovery linked with Alewife and Rainbow Smelt declines now appear to be hampered by a new suite of invasive species that consume native fish eggs and cause recruitment

(continued)

Box 9.3 (continued)
failures (Claramunt et al. 2005). The round goby (*Neogobius melanostomus*) and rusty crayfish (*Orconectes rusticus*) have now invaded the Great Lakes coastal zone. These benthic egg predators (see trout-egg-baited traps in Fig. 9.2) are reaching abundance levels three to five times that of the native species which previously occupied coastal habitats (Bronte et al. 2003; Jonas et al. 2005). Round goby have become the most abundant and conspicuous near shore benthic predator in the Great Lakes, and to complicate matters further is also an increasingly important part of the prey base (Crane et al. 2015).

The tension between coping with the complex and evolving synergistic interactions between native and invasive species, responding to the next looming invasion crisis (e.g. Asian carp), coupled with effects of other anthropogenic stressors (Pagnucco et al. 2015; Allan et al. 2013) continues to be a monumental challenge for Great Lakes protection and management. For Darwin's dreampond, the Laurentian Great Lakes is an important testing ground globally, for both the concepts of species adaptation in a rapidly changing environment as well as development of tools and approaches to prevent or manage species invasions (Bailey et al. 2011; Jerde et al. 2011; Kolar and Lodge 2002; Christie and Goddard 2003). Darwin's nightmare, however, may very well be that the species most unable to adapt to the rapid environmental changes in the Great Lakes, to which they are most critically linked and dependent on, are not the fish but the humans.

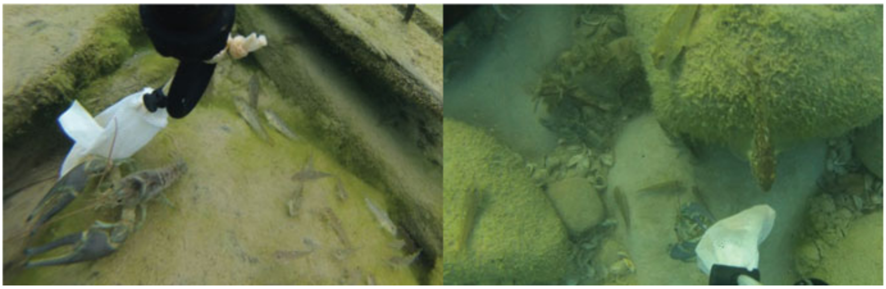

Fig. 9.2 Baited underwater images taken on shallow lake trout spawning reef habitat in Grand Traverse Bay, Lake Michigan, showing round goby and rusty crayfish congregating around a bait bag containing lake trout eggs. Note also live and dead dressenid mussels in interstitial spaces of cobble habitat in the right image. Images by Krista Robinson, Central Michigan University

9.3 Review of Control Techniques

Management and control of non-indigenous species can occur at a number of stages in the invasion process. Management can occur prior to the transportation phase (pre-border), during the transportation phase (i.e. border control or post-border) or following introduction or establishment to a new waterbody (post-border). Pre-establishment management is covered here as in-lake restoration is largely impossible for most invaders once established. Indeed, ignoring the possibility for reducing the probability of establishment of new non-indigenous species in lakes is akin to disregarding the catchment of a lake when restoring the effects of nutrient additions.

9.3.1 Pre-border Controls

A number of laws and regulations exist in New Zealand to reduce the number of species able, or allowed, to enter the country. This pre-border control is the first line of defence in reducing the establishment rate of non-indigenous species into lakes, attempting to allow entry of only a subset of species that have low probability of survival, reproduction or potential for effects. The primary pieces of legislation with respect to the importation of living things, intentionally or accidentally, are the Biosecurity Act 1993 and the Hazardous Substances and New Organisms Act 1996. The Biosecurity Act states that importers of risk goods (including live animals) must take all reasonable steps to ensure that the goods comply with applicable 'Import Health Standards'. These standards are based on risk assessments (Hayden and Whyte 2003), and specify requirements that need to be met before importation, providing explicit directions on what measures need to be taken before goods can be imported.

Several standards are applicable to animals that might establish populations in New Zealand lakes. One standard that allows for the deliberate importation of many freshwater species is the 'Import Health Standard for Ornamental Fish and Marine Invertebrates from all Countries'. This is essentially a 'white list', naming 1010 permissible freshwater aquarium fish species. Previously, this list comprised 280 names, of which 178 were genera, some of which contained multitudes of species (McDowall 2004). A subset of the 1010 permissible species (44 species) is identified as 'high risk', due to being susceptible to one or more diseases, and must meet additional pre-quarantine or quarantine measures before clearance.

The Hazardous Substances and New Organisms Act 1996 regulates the importation of 'new organisms' into New Zealand. In this Act, the definition of a new organism is one that was not present in New Zealand prior to 29 July 1998; any present before that date do not represent new organisms. If an importer wants to introduce a species, or a genetically modified organism, that is not listed on an import health standard, or was not officially present prior to 29 July 1998, an import permit must be obtained from the Environmental Protection Agency (EPA). The EPA undertakes a risk assessment of potential adverse effects of importing that organism and takes a precautionary approach. As a minimum standard, the EPA must be satisfied that, after entry to the country, the species could not: (1) form self-sustaining populations anywhere in New Zealand; (2) displace or reduce a valued species; (3) cause deterioration of natural habitats; (4) be a parasite, or be a vector or reservoir for human, plant or animal disease; or (5) have any adverse effects on human health and safety or the environment. Biosecurity inspectors are able to seize any organism that they have reason to believe may be a new organism. The Biosecurity Act 1993 also states a person must not provide false, misleading, or incomplete information about goods to be imported or undeclared goods, or take steps that are likely to hinder the detection of undeclared goods by an official.

Several standards also are aimed at preventing aquatic fauna being introduced to the country accidentally. Aquarium plants, which may be contaminated with small invertebrates, including worms, copepods and snails (Duggan 2010), must meet the standard for 'Importation of Nursery Stock'. Aquarium plants such as *Anubius* spp., *Hygrophila difformis* (wisteria) and *Echinodorus amazonicus* (Amazon sword plant), for example, have had a requirement that they be inspected immediately prior to export for the presence of snails, snail eggs, worms or leeches, and be transported in an aquarium containing a dilute solution of copper sulphate. There is also an Import Health Standard for 'Used Equipment Associated with Animals or Water'. With respect to freshwaters, used equipment associated with freshwater aquatic animals or activities may be allowed provided they are cleaned, checked and dried prior to arrival into New Zealand, and the equipment needs to be visibly clean and free from contamination with organic material including sediment, soil, weeds or other aquatic organisms. As such, this should also reduce the potential for small aquatic invertebrates to enter, live or as diapausing stages.

Although New Zealand primarily works on a 'white-list' system, the Freshwater Fisheries Regulations 1983 also provide a 'black list' for some species. The regulation states that no person shall have in their possession or under their control, or rear, raise, hatch or consign, any 'noxious' fish. The regulations provide a short list of 'noxious' taxa, including walking catfish (*Clarias batrachus*), pike (*Esox lucius*), piranha (*Pygocentrus* spp., *Rooseveltiella* spp., *Serrasalmus* spp.) and tilapia (*Tilapia* spp. or *Sarotherodon* sp.). Further, the Hazardous Substances and New Organisms Act 1996 specifically prohibits entry of the family Esocidae (e.g. pikes, muskellunge) or stickleback (*Gasterosteus* spp.). The rationale for listing these in the legislation, and not others that might potentially establish, is not provided. Nevertheless, the provision of white lists for species within Import Health Standards effectively make these listings redundant.

9.3.2 Border Control

Since New Zealand comprises a series of islands, it is in a better position to implement protection at the border to reduce the rate of invasions than many countries elsewhere. At international airports, passengers must complete a 'Passenger Arrival Card', which includes a question regarding any animals (or animal products) that may be carried. Further, passengers are typically questioned, and luggage is commonly checked, for organisms that are not allowed into New Zealand. Aquatic animals are occasionally intercepted at the border. For example, in 2013 a Vietnamese man was caught attempting to smuggle seven live tropical cichlids through Auckland airport, hidden in plastic bags in his pockets; he was only caught when water was seen dripping down his trouser leg.

All items of international mail are also checked for biological material. In 2008, six Siamese fighting fish (*Betta splendens*) sent from Thailand were intercepted at the International Mail Centre, Auckland, with the owner having provided an incorrect declaration; ironically, this species can be bought legally in New Zealand or can be legally imported with a permit. In October 2010, the secretary of the New Zealand Killifish Association was similarly caught importing fish eggs from Sweden and Denmark. An intercepted parcel to the man contained three small plastic bags containing fresh peat moss with killifish eggs. Interceptions of freshwater species have also been made on shipping vessels, such as the Asian tiger mosquito (*Aedes albopictus*) on a cargo ship arriving from Vanuatu in 2011. These examples highlight the importance of border control to prevent the incursion of non-indigenous species that could compromise lake restoration outcomes in the future.

9.3.3 Post-border, Pre-establishment Management

Post-border legislation is complicated, with a number of different Acts and Regulations being relevant depending on the species involved. These statutes include the Biosecurity Act 1993, the Freshwater Fisheries Regulations 1983 and the Conservation Act 1987. The Biosecurity Act 1993 is the main piece of legislation providing for post-border management or eradication of non-native species.

Post-border control can include species that have passed through the border, but have not yet been released to the wild, or the management of species after they have been introduced or have established in natural waters. Despite pre-border and border control, there is ample evidence that the border is leaky, with a number of incursions of prohibited freshwater animals detected. One species that made it through border controls, but was eradicated prior to entering natural waters, is the North American tadpole shrimp *Triops longicaudatus*, a species sold elsewhere as diapausing eggs and hatched as pets. In 2005, *Triops*, received as a present by a 6-year-old boy, was seized at Warkworth Primary School and subsequently destroyed. In 2014, the species was again found being advertised on a New Zealand online auction site; the winner voluntarily surrendered the kit to authorities. In 2007, illegally imported, genetically modified individuals of the zebra fish species *Danio danio* were found at four properties in Christchurch, from which they were sold and/or bred. Although this is an allowable species in New Zealand and unlikely to establish populations in the wild, being genetically modified designated these individuals as 'new organisms' in breach of the Hazardous Substances and New Organisms Act.

Deliberate illegal introduction of species into New Zealand has occurred for some time. In the 1960s, carp were illegally imported by a fish breeder, by incorrectly labelling them as shubunkin (multi-coloured goldfish), and these are now a major problem in many northern New Zealand lakes. Similarly, in 1967, rudd were smuggled into New Zealand in the vegetable room of the liner *Rangitoto* (Winters 2012). More recently, in 1999, a plastic tube arrangement was used to smuggle gudgeon eggs into New Zealand, passing through customs undetected (Winters 2012).

9.3.4 Post-establishment Control Measures: Invertebrates

Once established, different sets of laws are intended to reduce the spread of species. The Conservation Act 1987 states that 'no person shall transfer live aquatic life or release live aquatic life into any freshwater in a new location where the species does not already exist, including fish farms'. The Biosecurity Act 1993 also provides for post-border management, allowing for the eradication or effective management of harmful organisms. Strangely, the Freshwater Fisheries Regulations 1983 states more explicitly that no person shall 'place, liberate, or introduce into any lake, river, or stream any indigenous or exotic species of Mollusca, Crustacea, Protozoa, Insecta, or of annelid, nematode, platyhelminth worm, or oligochaete worm'. While this covers most non-indigenous invertebrate species in New Zealand, it does not include all of them, such as the freshwater jellyfish *Craspedacusta sowerbii*.

For invertebrates, and particularly for zooplankton, little can be done once a species becomes established other than reducing the rate of spread. By this stage, eradication will typically be impossible, and to date little effort has, therefore, been made to do so. One exception, however, has been the large freshwater crayfish, Western Australian marron, which was originally introduced to New Zealand in 1986 for aquaculture. However, in the early 1990s, a change in government policy led to the commercial marron farm being closed down and animals being destroyed, following concerns about the impact these crayfish could have on native fauna, particularly native crayfish (*Paranephrops* spp.). The major threat posed by marron was the possibility that the smaller female koura might prefer to mate with the larger male marron (D.K. Rowe, personal communication).

In 1993, an estimated 500,000 marron were destroyed at the Warkworth farm after the fish farmer had his permit revoked. In 2005, however, a lone marron was found on a footpath outside a disused West Auckland service station and an inspection of an adjacent property revealed more marron and also the gudgeon (*Gobio gobio*). Subsequent investigations revealed two ponds at South Head, Kaipara, containing more specimens of both species. About 500 marron were destroyed by draining the ponds. These marron may have originated from the farm at Warkworth; it is suspected some were removed prior to authorities taking over management of the farm. Elsewhere in the world, non-native crayfish have been eradicated from ponds using pyrethroids (Peay et al. 2006; Sandodden and Johnsen 2010). While chemicals such as pyrethroids or other insecticides are also applicable to crustacean zooplankton (e.g. Wendt-Rasch et al. 2003), the utilisation of resistant diapausing stages by that group, which lie dormant within sediments until conditions are appropriate, will render these techniques temporary at best for restoring indigenous lake zooplankton communities.

A trend among invertebrate invasions in inland waters, both in New Zealand and elsewhere, is that the first sites of establishment are commonly constructed waters (e.g. retired mines and quarries, and dammed river impoundments for hydroelectricity or water supply (e.g. Johnson et al. 2008; Banks and Duggan 2009)). These constructed waters typically have zooplankton species less specialised for pelagic conditions than natural waters, as noted by Parkes and Duggan (2012). Indeed, these authors suggested that newly constructed waterbodies should have native species introduced, so that they develop 'biotic resistance' more rapidly as a means to reduce establishment rates of non-indigenous species. Taylor and Duggan (2012) demonstrated this effect experimentally, whereby tanks seeded with sediments from natural lakes developed zooplankton communities more rapidly, and after 1 year more readily repelled the invasion of new species introduced to experimental tanks. In particular, establishment rates of the North American calanoid copepod *S. pallidus* were reduced, particularly when native calanoid copepods were present, indicating that some species may be key to reducing establishment of particular taxa. *Skistodiaptomus pallidus* has recently been observed in natural lakes and a link was made by Duggan et al. (2014) between its establishment and the legal releases of grass carp into lakes for aquatic macrophyte control. This link has since been supported by the finding of this copepod in grass carp aquaculture ponds (Duggan and Pullan 2017) and in further ponds that have received grass carp from these farms (Branford and Duggan 2017). The Ministry for Primary Industries is thus in the process of strengthening its requirements for the release of grass carp and silver carp from farming operations, in an attempt to reduce the rate of incidental invasions from this vector.

Management strategies designed to reduce the rate of movement of macrophytes also will aid in reducing the inadvertent movement of invertebrates. For example, signage at popular recreational lakes for boat operators to check and remove weeds from their vessels, motors and trailers is aimed at slowing the rate of invasion of these weeds and the many invertebrates, particularly snails, associated with them. Attempts have apparently not been made to eradicate non-native molluscs in New Zealand to date, likely due to their seemingly negligible current effects on the economy and human or other animal health. Elsewhere molluscicides, such as niclosamide, have been used to eradicate disease carrying snails (Clearwater et al. 2008). With warming of New Zealand due to climate change, and an associated increased risk of tropical diseases being carried by these snails, toxins may need to be considered in the future. Another alternative is complete destruction of habitats in which the species are found if distribution is restricted and the habitats do not have high native biodiversity values. This approach is not desirable or feasible for most lakes, but may be done for small constructed ponds. For example, the calanoid copepod *S. valkanovi* was eradicated from Bulgaria following the destruction of the pond in Sofia Botanic Gardens from which this species was first described (Duggan et al. 2006). In New Zealand, one population of the copepod *Boeckella symmetrica* has been eradicated following the infilling of Wiri Quarry (Branford and Duggan 2017).

9.3.5 Control Measures: Vertebrates

A range of options is available for control of non-indigenous fish as part of lake restoration. However, some species are easier to target than others and complete eradication is difficult (Collier and Grainger 2015). To achieve effective control, a combination of methods is typically required and re-invasion pathways need to be blocked. The high fecundity of most invasive fish means that control measures must be either total (i.e. eradication by piscicide) or sustained over the long term to achieve desired ecological outcomes, as spawning by only a few remaining fish can rapidly reinstate population densities. In addition, single species control may lead to ecological surprises, such as releasing other species from competitive suppression or top-down control. An understanding of community-level food-web interactions is, therefore, required to predict the outcomes of invasive fish control and its relative importance in terms of meeting lake restoration targets compared to other factors such as managing catchment land-use practices. In addition, a knowledge of species biology, behaviour and environmental cues is important so key life-cycle bottlenecks can be exploited, such as spawning aggregations and migration routes (Collier and Grainger 2015; David 2015).

Eradication of invasive fish is most cost-effectively achieved through the use of piscicides and drainage. Two types of toxin have been used in New Zealand for invasive fish control: lime [calcium hydroxide ($Ca(OH)_2$) and rotenone (cube root powder)]. Use of lime was restricted to small ponds and involved drainage to reduce water volume before spraying a mixture of water and hydrated lime to achieve a pH of greater than 9 (West 2015). This method is no longer used now that rotenone is more readily available in New Zealand. Rotenone was first used in New Zealand to eradicate rudd, tench and grass carp in Parkinson's Lake (near Auckland), and native fish were then restocked into the lake (see Rowe and Champion 1994). It is now applied as a cube root slurry (1.2–1.8% rotenone) using a helicopter for large ponds and lakes; to date there have been 81 rotenone operations across the country targeting gambusia, carp, rudd and tench, with three applications on lakes of 10–17 ha (West 2015). Eradication is easier on small waterbodies where outlets can be closed off, although outcomes may be compromised by inputs from springs providing clean-water refugia for fish during poisoning operations. Rotenone was successfully used to eradicate trout from Upper Karori Reservoir and its tributary streams, Wellington, resulting in a dramatic improvement in native galaxiid fish and crayfish numbers in the inlet stream and a shift in zooplankton composition (Pham et al. 2013; Duggan et al. 2015). However, using poison in high quality lakes may result in collateral damage to native species, which is an important consideration when deciding on which control methods to deploy.

Various combinations of nets and other fishing methods can be used to reduce invasive fish biomass in lakes [see Case Study—Rotopiko (Serpentine)], although netting is not effective for all species (e.g. carp). In Lake Ohinewai, the combination of multiple other large nets and traps, seining and electrofishing over several months achieved a reduction in carp biomass from 373 to c.100 kg/ha (see Case Study—Ohinewai). The nets employed included ‘pod traps’, which have one-way doors that prevent fish from escaping and automated feeders that frequently add fresh bait to the trap to improve catch rates (Hicks et al. 2015). Baiting of nets and traps has been shown to improve catch rates of carp but baits lose most of their attraction properties within an hour. Baits can be laced with toxins such as rotenone, although sodium nitrate has been shown to be just as effective (Morgan et al. 2014). However, there can be some aversion to toxins requiring the use of masking agents (Morgan et al. 2013). Boat electrofishing was also used in Lower Karori Reservoir, Wellington, to reduce top-down control of perch populations on zooplankton, leading to improvements in water quality (Hicks et al. 2015). However, once fishing ceased, perch numbers increased reinforcing the need for sustained control of invasive fish numbers to meet lake restoration targets.

Case Study 9.1 Rotopiko (Serpentine) Lakes

This lake complex is located to the south of Hamilton in the central North Island and comprises three small connected peat lakes that have high conservation value due to healthy indigenous macrophyte communities and intact native marginal terrestrial and wetland vegetation (de Winton 2014; Price and Gumbley 2015). In addition to several native fish species, including smelt which do not naturally occur there, the lakes also contain the catfish brown bullhead, goldfish, gambusia and rudd, with the latter species of most concern because of its propensity to consume aquatic plants (Lake et al. 2002). A restoration programme for the lakes has been ongoing for several years and, since 2001, has included a netting programme for pest fish targeted at rudd control. Fine mesh monofilament gill nets are deployed annually in spring prior to spawning, and submerged aquatic vegetation is monitored twice yearly (de Winton et al. 2006).

Gill netting appears to have been effective for suppressing rudd numbers, but the effect on indigenous fish populations is difficult to define due to the lack of observations before netting started. However, the indigenous fish populations of the Rotopiko (Serpentine) Lakes are relatively large compared to pest fish population (Wu et al. 2013), pointing towards a positive effect of the netting programme. Notwithstanding the decline in rudd populations, compensatory changes in catfish and goldfish populations may continue to pose a threat to macrophyte populations in these lakes, but their effects are as

(continued)

Case Study 9.1 (continued)
yet unknown. No improvement in scores of a macrophyte health index (LakeSPI; see Chap. 8) was evident following initiation of fishing in 2001, although the absence of data collected before fish removal makes drawing conclusions difficult. It is possible that planktonic algae in the lake water or self-shading by luxuriant macrophyte growths, promoted by enrichment, limits the amount of light available for macrophytes. Under this situation, a certain level fish grazing may, in fact, help to enhance macrophyte cover and diversity, suggesting that management of rudd numbers rather than eradication could be considered for restoration purposes.

Case Study 9.2 Lake Ohinewai
This is a shallow (4.5 m depth), 16.8 ha lake on the floodplain of the Waikato River, central North Island, where catchments are highly developed for agriculture and extensive drainage, and flood control measures regulate river and lake levels. Lake Ohinewai deteriorated from a stable oligotrophic (macrophyte-dominated) state to a stable eutrophic (algal-dominated) state during the early 1990s. This lake was selected for invasive fish removal with the ultimate goal of improving water quality and restoring indigenous biodiversity. The main fish species of concern is carp, but the lake also supports high numbers of goldfish, brown bullhead and gambusia, along with some rudd.

The immediate goal was to reduce carp to a level at or below 100 kg/ha with the intention of improving water quality and macrophyte cover. Mark-recapture fishing, using a combination of fyke nets, minnow traps, electrofishing, beach seining and baited traps, was successful in reducing carp biomass from 373 kg/ha to below 100 kg/ha (Tempero et al. 2015). To limit reinvasion, a one-way barrier was installed in the lake outlet allowing adult common carp to leave but not re-enter (Fig. 9.3b), and this resulted in a further 50% drop in biomass to approximately 45 kg/ha as fish exited the lake but could not return. The barrier was designed with horizontal bars to allow debris <30 mm to pass through unobstructed and was hinged at the top to allow for easy cleaning in the case of blockage. The bar spacing of the one-way gate installed in the barrier was based on the fish trap design of Thwaites et al. (2010) and included a set of weighted swinging bars at the base of the trap that would allow adult carp and eels to push through the trap when moving downstream but would not allow adult carp to return to the lake for spawning.

Water quality (Secchi disk transparency and total suspended solids) showed early signs of improvement in Lake Ohinewai, and aquatic macrophytes are reappearing around the margins, although these are mainly introduced

(continued)

Case Study 9.2 (continued)
Ludwigia and *Myriophyllum* species. Thus, while regarded as a success in terms of carp control, the ecological outcomes in terms of macrophyte recovery have been compromised by the dominance of invasive plant species, although there may be longer term recovery in native species if seed banks on the lake bed are still viable. Modelling of land management and invasive fish control scenarios showed that integrated catchment management of tributary stream riparian zones and constructed wetlands would be required along with carp control to achieve desired improvements in lake health (Allan 2016).

Recent overseas developments in cage and barrier design for point source control and restriction of movement have been aimed at controlling carp or preventing silver carp (*Hypophthalmic molitrix*) from accessing the Great Lakes, USA. Barriers can be physical, visual, acoustic or electrical obstructions and may limit movement in one direction or both upstream and downstream directions. Zielinskia et al. (2014) found that curtains of graded and coarse bubbles reduced directional movement of carp by 75–85%, and they concluded that sound and fluid motion were more important than visual cues for restricting movement past the curtains. However, bio-acoustic fish fences can be expensive to implement (Brammeier et al. 2008), and neither these nor electric barriers (e.g. Verrill et al. 1995) have been implemented in New Zealand. Rather, vertical physical barriers $\geq$1 m high have proven ideal for invasive fish exclusion in this country due to the climbing ability of many migratory native fish and the inability for the current suite of invasive fish to pass such barriers. Vertical barriers can be as simple as a timber weir or a perched culvert, and if necessary weirs can include metal bars or plunge pools to limit upstream ingress by jumping (e.g. Fig. 9.3a; see also Gumbley and Daniel 2015). Native fish passes can be fitted if necessary using ramps, ropes or other devices to ensure populations of most indigenous fish are sustained in restored lakes (David et al. 2014; see also Chap. 10).

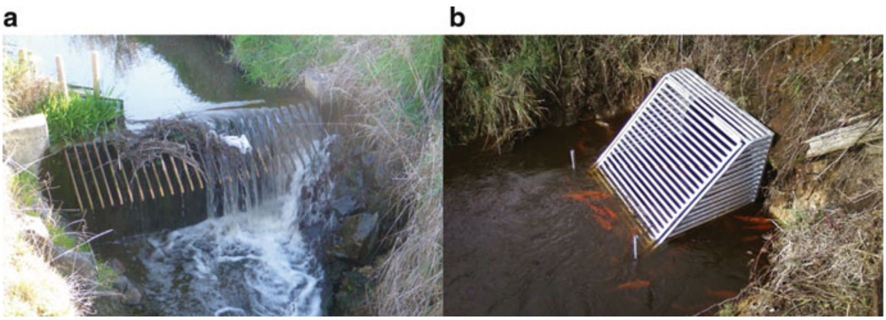

Fig. 9.3 Fish barriers deployed downstream of lakes to prevent upstream movement of invasive fish. (**a**) Lake D—rods prevent fish jumping over concrete weir; (**b**) Lake Ohinewai carp screen with one-way finger gate below water surface (photo: A. Daniel)

Fig. 9.4 Carp-N neutral showing the carp cage on the left and Lake Waikare in the background (photo: B. David)

One-way barriers exploit the pushing behaviour of carp, and a finger-trap design has been successfully deployed on the outlet of Lake Ohinewai to prevent re-invasion by carp following biomass reduction (Tempero et al. 2015; see Case Study—Lake Ohinewai; Fig. 9.3b). These devices have also been effective in trapping non-native fish at the Lake Waikare separation cage (Fig. 9.4), which removed around 10 tonnes of carp, goldfish and catfish in 2 years (David 2015). Less than 20 individual non-target fish were captured during that period. Upstream netting indicated that large numbers of smaller native fish, but also invasive species (particularly small catfish and goldfish <200 mm), passed through the cage design unharmed, indicating that the fish trap is an effective and reliable tool for mass adult invasive fish harvest with minimal by-catch, but it does not exclude all invasive fish (David 2015). While barriers and exclusion screens can prevent access to target habitats such as high-value lakes (e.g. Thwaites et al. 2010), separation cages can be used to isolate carp in containment areas by capitalising on their tendency to jump out of the water when they encounter a barrier (e.g. Stuart et al. 2006).

9.4 Conclusions and Future Prospects

While pre-border and border controls should reduce the rate of new invasions to New Zealand lakes, they clearly still occur and will continue to do so in the future, particularly for smaller invertebrates. Once established, it can be difficult or impossible to eradicate species from the system, making restoration to a pristine state problematic. Fewer control options are available for smaller species than larger species, reinforcing the need for continued border and post-border surveillance so as not to compromise the outcomes of lake restoration efforts. Vectors have yet to be identified for a number of recent invaders, meaning that re-invasion may be difficult to prevent, rendering restoration of native biotic communities futile (i.e. the aim to achieve non-native free sites).

A large number of non-indigenous animals have invaded New Zealand lakes to date, but the current census is likely to be an underestimate, particularly for smaller species. For example, non-indigenous species of harpacticoid copepod are known from amongst the bottom sediments of home aquaria, but they are not yet recognised in the wild. Other non-indigenous species potentially present may simply not be recognised, highlighting a need for genetic screening. Further, many non-indigenous fish species intentionally carried by the aquarium trade are not yet known to be established in the wild but could pose a future threat of establishment with rising temperatures resulting from climate change. Similarly, as a number of non-indigenous invertebrate species in New Zealand are carriers of serious parasites and diseases elsewhere, the potential for some invaders to have effects on human and other animal health correspondingly increases.

Emerging and future technologies for controlling or eradicating non-indigenous fish include the use of pheromones to enhance capture rates, the introduction of taxon-specific pathogens and the genetic modification of fish to produce single-sex (male-only) progeny. As such, opportunities for the control or eradication of some species will increase through time. Additionally, the methods for detecting species are also improving. Techniques that are able to detect an incursion at an early stage, following introduction or early establishment, may greatly increase the possibilities for eradication. For example, 'environmental DNA' (eDNA) techniques are an emerging method to detect the presence of vertebrate species, where DNA from material sloughed from animals is detected from water samples (Jerde et al. 2011; see also Chap. 12). The usefulness of eDNA was evaluated during the eradication of brown trout in streams flowing into a Wellington reservoir, with PCR product specific to brown trout able to be amplified from samples before, but not after, eradication (Wood et al. 2013). Continued development of new detection and control technologies will be essential to keep pace with the increasing threat of invasive aquatic animals establishing in New Zealand lakes.

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Chapter 10
Restoration of Native Fish in New Zealand Lakes and Reservoirs

David K. Rowe, Gerry Closs, and David W. West

Abstract Since 1900, a range of anthropogenic stressors has steadily depleted the indigenous fish fauna in New Zealand lakes. Efforts to restore these fish populations have involved translocations of small indigenous fish species to replace those reduced by introduced piscivorous fish and elver trap-and-transfer operations over high dams to restore elver recruitment to commercial and customary eel fisheries above the dams. Habitat restoration for small benthic fish (through removal of invasive macrophytes) and for pelagic species (through water quality improvement) has also occurred in lakes, but this was incidental to other management goals and not specifically targeted at native fish restoration. Control of introduced pest fish in lakes is now increasing in importance but is limited in both scope and methods and so is still in its infancy. Overall, there is currently much less focus on indigenous fish restoration in New Zealand lakes and reservoirs than in rivers and streams. Information on the success of methods used in lakes is sparse because effective monitoring is often lacking or too limited to provide reliable data on success rate.

Keywords Restoration methods · Translocation · Connectivity · Invasive species

10.1 Introduction

The global decline in freshwater biodiversity (Dudgeon et al. 2006) has affected fish species more than other aquatic taxa (Lévêque et al. 2008) and this is especially so in New Zealand. In 2007, the International Union for Conservation of Nature (IUCN) noted that there were more extinct and threatened species in New Zealand than in

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any other nation with 68% of all freshwater fish under threat (IUCN 2007). In a separate study carried out in 2008, Allibone et al. (2010) concluded that 67% of indigenous freshwater fish taxa in New Zealand were either threatened or at risk. Goodman et al. (2013) repeated the Allibone et al. (2010) assessment in 2012 and found that the proportion of threatened and at risk freshwater fish taxa had increased to 74%. The decline in indigenous freshwater fish species in New Zealand waters is, therefore, severe and restoration of the fauna is of increasing concern. The intensification of fish decline in New Zealand inland waters reflects that in North America (see Feature Box 10.1) and is more intense in lakes and wetlands than in rivers and streams because lacustrine environments are subject to greater levels of human pressure.

Box 10.1 Rehabilitation and a Safe-Operating Space for Freshwater Fisheries

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Freshwater ecosystems and fisheries are threatened globally by humans (Carpenter et al. 1992; Ficke et al. 2007), and impacts have only intensified with recent time (Vitousek et al. 1997; Post et al. 2002). Past approaches to restoring inland fisheries have been mostly local and species-specific (Daugherty et al. 2008; Arlinghaus et al. 2015). For example, stocking has historically been prioritised as the preferred management tool to repair degraded populations (Sass et al. 2017). Yet it has become increasingly apparent that new approaches are needed that consider more than one ecosystem or one species at a time (Magnuson 1991, 2007; Jacobson et al. 2016). Central to this perspective is the reality that landscapes are comprised of ecosystems maintaining diverse combinations of interacting species (i.e. communities), but also that these resources are shifting with varying uncertainty due to global anthropogenic impacts.

We offer a short-hand view of approaches to fisheries rehabilitation and management in an era of rapid change. Two conceptual models seem especially germane: (a) the rehabilitation model (Fig. 10.1) and (b) the safe-operating space (SOS) concept (Fig. 10.2). We also offer a series of concrete steps for consideration by managers interested in conducting a rehabilitation approach (Table 10.1). In developing this table, we drew from our own experiences forging, implementing and monitoring large fishery adaptation plans including: (1) a watershed-scale water clarity management plan via biomanipulation (Vanni et al. 1990; Lathrop et al. 2002) and (2) a landscape-scale adaptive fishery management plan to improve panfish-sized structure (Hansen et al.

(continued)

Box 10.1 (continued)
2015; Rypel 2015; Rypel et al. 2016). As the field continues to move towards comprehensive approaches to sustainable and resilient fisheries, we encourage managers to consider how their fisheries and ecosystems are shifting with time. Importantly they need to ask: are there opportunities to improve the resilience of fisheries by taking a rehabilitation approach?

The rehabilitation model (Fig. 10.2) is useful for clarifying a cluster of concepts and terminologies (Francis et al. 1979; Magnuson 1991, 1992). For example, ecosystems degrade from an initial condition to a current state because of various anthropogenic impacts operating at different scales. Resource managers must recognise that, in a great many cases, ecosystems have been altered irrevocably, e.g. by a species extinction, invasion, a large dam or urban sprawl (Hobbs et al. 2014; Moyle 2014). Indeed, biologists increasingly are abandoning the notion of total restoration in favour of more realistic and reconciled management goals (Rosenzweig 2003; Kimball et al. 2015; Lin et al. 2015; Liu et al. 2016).

Once the sources of degradation have been identified, managers should develop a comprehensive adaptation strategy that considers all available restorative and enhancement tools (Table 10.1). Strategies might include a mix of reducing environmental stresses, stocking, biomanipulation, watershed management and engineering solutions. "Rehabilitation"—the direction towards which the fishery moves with management actions rarely would result in a system being restored to original conditions, but hopefully it would represent a positive direction. For example, in challenging fisheries scenarios, simply abating environmental degradation might be the principal initial goal for the manager.

The safe-operating space (SOS) is another useful conceptual model (Fig. 10.2) rooted in the spirit of rehabilitation and resilience (Rockström and Karlberg 2010; Bennett et al. 2016). Ultimately, the SOS concept derives from a common observation: that ecosystems show few signs of change up until a threshold or "tipping point" is reached, after which point recovery becomes slow, difficult and expensive (Rockström et al. 2009; Scheffer et al. 2015). The SOS, therefore, can be conceptualised as a series of conditions that maintains fishery sustainability in the face of broad-scale change (Fig. 10.2, Carpenter et al. 2017). In a highly simplified example, the SOS might even be thought of as the relationship between one important variable under local management control (e.g. harvest) versus one that is not (e.g. climate change). At low amounts of climate change, a large amount of harvest might be possible on a climate-sensitive species because high quality habitat engenders high biomass production, blue scenario in Fig. 10.2 (Rypel 2013; Rypel and David 2017). Yet when climate effects become intense, a substantial reduction in harvest would be required to conserve the SOS, red scenario in Fig. 10.2 (Grafton 2010; Carpenter et al. 2017). We use the relationship between harvest and climate as one example, but in reality, an adaptive mix of options (aside from only harvest) would be most effective at maximising SOS area.

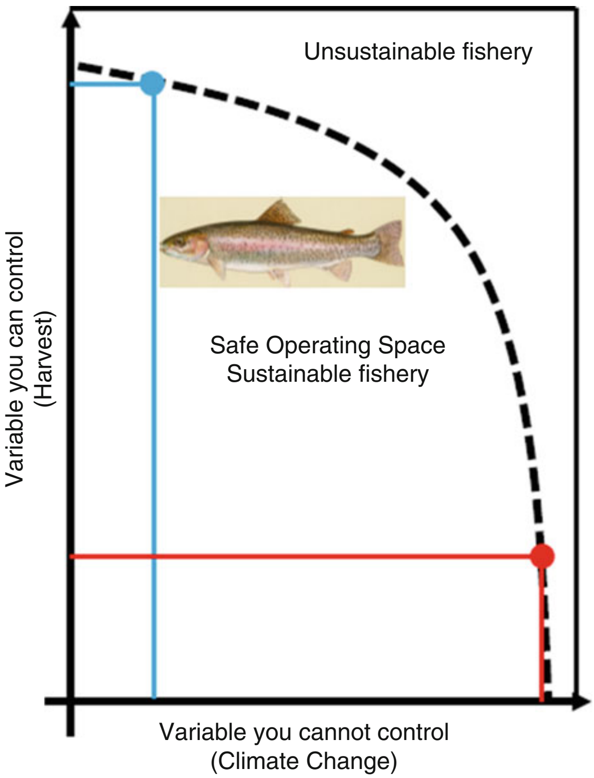

Fig. 10.1 The safe-operating space (SOS) concept using a native steelhead *Oncorhynchus mykiss* fishery as an example. Fish artwork via Wikimedia Commons (US Fish and Wildlife Service)

Some indigenous fish species in lakes have been more affected than others. For example, between 1900 and 1950, kōaro (*Galaxias brevipinnis*), whose juveniles (length 30–60 mm) once formed large schools in the pelagic zones of lakes, declined rapidly in many of New Zealand's larger lakes because of heavy predation from introduced trout (McDowall 1990; Rowe 1993; Rowe et al. 2003). In contrast, the abundance of common bully (*Gobiomorphus cotidianus*), a similar-sized fish species also subject to heavy predation by trout in lakes, was unaffected (Rowe 1999). Many populations of longfin (*Anguilla dieffenbachii*) and shortfin (*Anguilla australis*) eel in lakes have also declined (McDowall 1990; Jellyman 2012) because, although they are more tolerant of low water quality than other indigenous fish species (Richardson et al. 1994; Richardson 1997; Dean and Richardson 1999), they are vulnerable to both over-fishing and recruitment failure caused by barriers to their upstream migration (Boubée and Jellyman 2009). Smelt (*Retropinna retropinna*) is the least tolerant indigenous species to water quality decline (Ward et al. 2005), and its lacustrine populations have declined in eutrophic lakes (Rowe and Taumoepeau 2004). In contrast, populations of common bully increase in eutrophic lakes (Rowe 1999).

These species-specific differences in response to anthropogenic impacts depend mainly on variations in life-history characteristics and on vulnerabilities to habitat degradation. Such differences also depend on the susceptibility of the species to

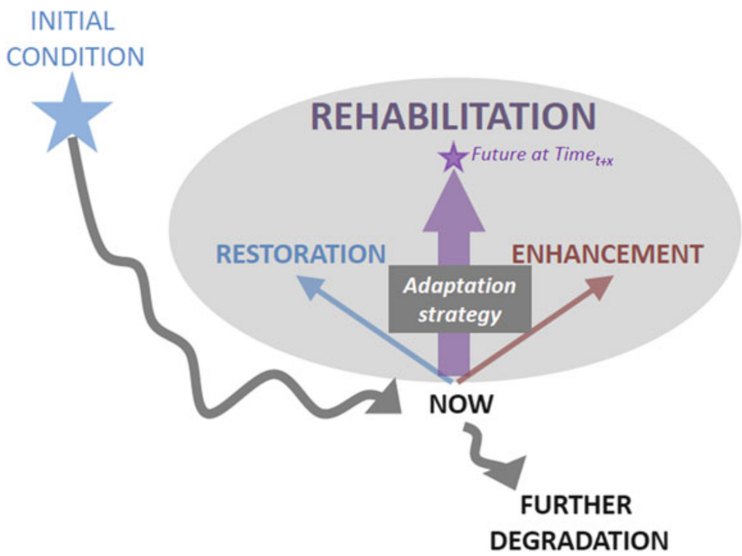

Fig. 10.2 Conceptual figure illustrating the rehabilitation approach and its key concepts. Ecosystems degrade from initial conditions to a current time point where resource managers must make important decisions that can produce divergent results. For example, decisions can send the system towards long-term restoration, reconciled enhancement, no additional loss, or further degradation

predation and competition from invasive fish species. Consequently, successful restoration of native fish in New Zealand lakes depends on the identification of the species-specific stressors operating in each lake and then application of appropriate restoration tools to reduce these.

Globally, the principal causes of fish decline in lakes have included over-fishing, loss of key habitat, loss of connectivity (e.g. to spawning grounds) and eutrophication. Of these, eutrophication is perhaps the most widespread reason for fish population decline (Tammi et al. 1999; McKinnon and Taylor 2012; Vonlanthen et al. 2012). This is because it degrades fish habitat, reduces water quality and simplifies food webs. However, the main mechanisms by which fish are affected can vary from lake to lake and include fish kills (due to algal blooms and de-oxygenation), reduced spawning and hence recruitment (from habitat loss), reduced growth (due to food web changes) and increased mortality (from predators).

In some Danish lakes, biomanipulation of fish stocks has been used recently to overcome some of the changes in food webs and trophic cascades caused by eutrophication because these changes are not easily restored by nutrient control (Feature Box 10.2). This approach reflects the complex interactions among the physical, chemical and biological drivers of habitat degradation in lakes. Such complexity places a constraint on the manipulation of fish populations in eutrophic lakes because it requires an understanding of all the mechanisms governing their abundance and these are often multifactorial and/or unknown. This complexity also

Table 10.1 Rehabilitation approaches for consideration by managers

Step	Rationale	Suggestions to managers
(1) Identify scale of problem	Scale constrains and focuses rehabilitation options	*If spatial scale is large, concentrate efforts on:* (1) Large-scale database development (2) Capacity building for data analytics (3) Classification of ecosystems/ fishery types *If spatial scale is small, focus on:* (1) Relationship building (2) Recruitment and use of lake/ stream-keepers (3) Consider potential technology and tools
(2) Triage	Limited resources should be spent where fisheries benefits are largest given available resources	Be wary of suggestions to spend resources on systems and options that cannot succeed in face of uncontrollable change
(3) Identify drivers you can control versus those you cannot control	Management should focus on controllable factors with potential to conserve or increase the Safe-Operating Space (Fig. 10.1)	*Controllable tools* Harvest, modifications to hydrologic structures, invasive species prevention, land management, inlake habitat management, contaminant reductions *Uncontrollable context* Climate change, macroeconomics, human behaviour, cultural values, political shifts, war
(4) Check feasibility	Certain options may be impractical based on scientific information, social acceptability, cost	(1) Develop longlist of rehabilitation options (2) Conduct literature review to identify actions with desired effect (3) Engage public on social palatability of options (4) Focus groups, advisory councils and professional facilitators can be helpful
(5) Select rehabilitation strategy	Make the choices	(1) Develop proposed actions (2) Re-engage public to consider proposed actions (3) Retool strategy as needed based on feedback
(6) Conduct and track responses to rehabilitation	Complete. Long-lasting rehabilitation efforts require monitoring and openness to adaptive management	(1) Follow-through with monitoring/evaluation of rehabilitation efforts (2) Communicate findings back to stakeholders (3) Learn, adapt and tune effort

limits the restoration of fish affected by other stressors including habitat loss from water level change, pollution and invasion by exotic species.

Box 10.2 Experience from Danish Lake Restorations

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Most Danish lakes are eutrophic due to increased nutrient (nitrogen and phosphorus) loading from agriculture and human waste water. During the past 40 years, a number of management plans have been implemented to remedy this, but the reduction has not always been sufficient to achieve the desired ecological quality, and in other cases the lakes do not respond satisfactorily due to chemical or biological resilience. To overcome this resilience, restoration has been conducted in almost 100 lakes in Denmark during the past decades (see Table 10.2). The most commonly used lake restoration method has been biomanipulation by removing zooplanktivorous and benthivorous fish species, mainly roach (*Rutilus rutilus*) and bream (*Abramis brama*). If sufficient numbers (typically > ca. 200 kg/ha) were removed, strong effects were obtained due to a top-down controlled improved grazing capacity of large zooplankton filterers but also due to diminished resuspension caused by less benthic feeding activity. Stocking of predatory fish (northern pike, *Esox lucius*) has been widely used with the intention to decrease the number of zooplanktivorous species, but the effects have been weak. A recent restoration method has been the addition of aluminium to chemically fix phosphorus in the sediment and reduce the impact from internal loading. Immediate effects are usually seen, but long-term effects are still not well documented. The combination of methods, for example phosphorus fixation to decrease the bottom-up effects from internal phosphorus loading combined with biomanipulation to increase the top-down effects, may be a promising way forward.

The growing international awareness of fish decline in lakes has resulted in a number of tools and strategies being developed to restore fish populations (see Feature Box 10.1). In general, the methods used in New Zealand lakes can be classified into those requiring a manipulation of habitat (e.g. littoral zone habitat, spawning habitat, water quality), restoration of connectivity for fish migrations (e.g. barriers to upstream movements), invasive species control (e.g. reductions in piscivores and invasive macrophytes) and translocations (e.g. the use of fish stocking to spread indigenous fish populations more widely).

In this chapter, we review New Zealand examples of native fish restoration in lakes categorised by restoration method (viz. habitat improvement, connectivity restoration, invasive species control, translocation) and by species. We then note

Table 10.2 Overview of Danish lake restorations (excluding methods to reduce the external nutrient loading) and their effects

Restoration type	Cases (no)	Effects/results	Comments	Danish references
Biological methods				
Biomanipulation/ fish removal—mainly roach and bream	Ca. 50	Marked effects on nutrient and chlorophyll a concentrations and most trophic levels over the first 6–8 years after fish removal. Afterwards the lakes tend to return to more turbid conditions	Most restorations were conducted in eutrophic lakes with too high nutrient loading to expect permanent effects	Søndergaard et al. (2008) and Jeppesen et al. (2012)
Pike stocking (fingerlings)	Ca. 65	Only few cases with detectable effects	No longer recommended as a lake restoration measure	Skov (2002)
Protection of submerged macrophytes from waterfowl grazing	Exp.	Higher growth rates in large-scale exclosures protected from waterfowl grazing	Particularly important in the recovery phase when macrophytes start to recolonise	Lauridsen et al. (2003)
Physical–chemical methods				
Hypolimnetic oxygenation (O_2)	7	Reduced hypolimnetic accumulation of PO_4 and NH_4 during years of oxygenation, but no clear effects on epilimnetic quality in most cases	Accumulation of NH_4 and PO_4 increases again if oxygenation is stopped. Risk of a higher mobile P pool in the sediment. Long treatment needed	Liboriussen et al. (2009)
Hypolimnetic oxidation (NO_3) (whole-lake experiment)	Exp.	Reduced accumulation of both NH_4 and PO_4 in the hypolimnion	Only added during stratification to avoid NO_3 addition to/accumulation in the epilimnion	Søndergaard et al. (2000)
Aluminium addition	6	Strong immediate effects on phosphorus concentrations and water turbidity	Long-term effects not yet well documented. Risk of toxic effects at both high and low pH	Egemose et al. (2011)
Iron addition	Exp.	Little effect	P retention depends on redox conditions in the sediment	Jeppesen et al. (2012)
Sediment removal	1	Conducted in 150 ha Lake Brabrand, but poor effects because of still too high external loading	Sediment removal conducted in a number of small lakes (<10 ha) to deepen and restore habitats for isoetid plants	
Combined treatments	Exp.	Good effects obtained in whole-lake experiments combining iron addition and fish removal	To be studied further	Jeppesen et al. (2012)

Exp. is an in situ/whole-lake experiments only. In some lakes, both fish removal and stocking were conducted the main features of fish restoration in New Zealand lakes to date and review evidence for its effectiveness.

10.2 Littoral Habitat Restoration

In New Zealand lakes affected by eutrophication, one of the main reasons for fish decline is the degradation of littoral zone habitats. Endemic species such as common smelt, dwarf inanga (*Galaxias gracilis*) and the common bully are all dependent on littoral zone habitats (Rowe and Chisnall 1996; Rowe et al. 2003; Rowe and Taumoepeau 2004). Consequently, they are affected by reductions in these habitats, and their restoration depends on identifying and manipulating the key factors reducing the amount and quality of their littoral habitat. In New Zealand lakes, eutrophication causes increased turbidity and siltation of littoral substrates. However, factors other than eutrophication can also degrade littoral habitat for fish in lakes. Of these, manipulation of water levels and invasion by exotic macrophytes are the two most important (Rowe and Graynoth 2002; Champion et al. 2002; Rowe et al. 2003). These need to be considered alongside the effects of eutrophication in the restoration of littoral zone habitats because they often occur together as a suite of anthropogenic stresses.

10.2.1 Smelt

The common smelt is present in many North Island lakes and its abundance declines as water clarity is reduced by eutrophication (Rowe and Taumoepeau 2004). The precise mechanism(s) responsible for this decline are yet to be determined, but a reduction in littoral zone spawning habitat due to siltation is thought to be one of the major factors. This is because smelt food supply (i.e. zooplankton) is not limited in productive lakes and smelt can readily feed on planktonic prey in the turbid waters characterising eutrophic lakes (Rowe and Dean 1998; Rowe et al. 2002b). Furthermore, the decline in density of smelt larvae as lake trophic status increases suggests that spawning and/or larval recruitment is reduced. A decline in spawning habitat in the littoral zone would account for this (Rowe and Taumoepeau 2004).

The common smelt is essentially a shoaling, pelagic species in lakes, and adults deposit eggs over shallow (0.5–2.5 m deep), mainly sandy substrates in the littoral zone (Stephens 1984; Rowe et al. 2002b; Blair 2012). In eutrophic lakes, this habitat becomes increasingly covered by silt from increased turbidity. This could both inhibit spawning and increase egg mortality through smothering. Restoration of water clarity through a reduction in turbidity and siltation would, therefore, be expected to improve smelt recruitment by improving the amount and quality of spawning substrate (i.e. clean sand) in the littoral zone (Rowe and Taumoepeau 2004).

A decline in the trophic status of Lake Rotoiti (North Island) was first noted in the 1960s (Fish 1975) and, by the 1980s, de-oxygenation of the hypolimnion occurred annually over summer months. Water clarity had been steadily declining since the 1950s because of increasing nutrient supply and a consequently higher concentration of phytoplankton and turbidity. By the summer of 2003, the lake had to be closed to water contact for extended periods because of toxic cyanobacterial blooms. Lake closure sparked major efforts to restore this lake and, in 2008, a diversion wall to channel the nutrient-enriched inflow from Lake Rotorua around the edge of Rotoiti and directly to the lake's outlet was completed (Hamilton et al. 2009). Smelt monitoring in Lake Rotoiti was carried out annually from 2005 to 2015 as a condition of the resource consents for this diversion wall, with the main aim being to determine whether the seasonal smelt migrations up the Ohau Channel to Lake Rotorua would be reduced by it. However, this monitoring was also designed to determine how the smelt population in Lake Rotoiti responded to the wall and to any consequent improvement in lake trophic status (Rowe et al. 2015).

Lake trophic status is measured by the Trophic Level Index (TLI), which is based on water clarity, total nitrogen concentration, total phosphorus concentration and chlorophyll a such that a decrease in TLI indicates a decrease in trophic status. The TLI for Lake Rotoiti prior to 2008 was over four and from 2008 to 2014 it decreased steadily to less than 3.5 (Rotorua Te Arawa Lakes Company 2015). Larval smelt abundance in the water column was measured using a Wisconsin closing drop-net at 30 sites spread throughout the lake following the two main smelt spawning periods in spring and late summer. The annualised mean density of larval smelt began to increase in Lake Rotoiti shortly after the diversion wall was completed (Fig. 10.3). By 2015, it was four times higher than the 2005–2008 level. Hence, a reduction in

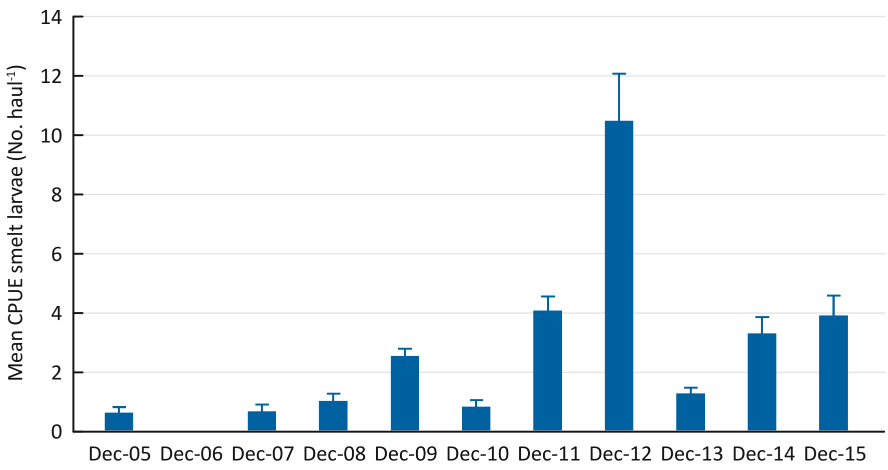

Fig. 10.3 Changes in the mean density of larval smelt in Lake Rotoiti (North Island) after nutrient diversion was started in July 2008 (*vertical bars are one standard errors*). Data from Rowe et al. (2015)

eutrophication and turbidity in this lake resulted in a sustained increase in the recruitment of smelt.

Smelt recruitment in lakes can also potentially be reduced by a low lake level during spring and summer months when the common smelt spawns in the littoral zone. For example, a drawdown of water level (e.g. for hydroelectric power generation) could reduce the amount of spawning substrate in the littoral zone. A model to determine the effect of lake level lowering on the location and total area of smelt spawning habitat in Lake Taupo was developed to inform decisions on water use for hydroelectric generation (Rowe et al. 2002a). This model was based on fine-scale acoustic mapping of the lake edge down to 10 m coupled with ground-truthing to determine substrate composition. It indicated that the total area of smelt spawning habitat in this lake was maximal at a lake level of just over 357 m (above mean sea level) and declined as the lake level dropped to its legal minimum of 355.8 m. This model indicated that maintenance of the lake level at about 357 m (during the smelt spawning season) was required to maintain the maximum area of smelt spawning habitat and that this would drop by up to a third as the lake level dropped to its legal minimum.

Common smelt are the main prey species for trout in Lake Taupo, accounting for over 90% of the total food intake for trout between 250 mm and 350 mm in length and 80% of the total food for all trout over 250 mm (Cryer 1991). Because lacustrine smelt are semelparous (spawning once before dying) and have a short lifespan (1–2 years), any large decrease in their recruitment will be rapidly reflected in a reduced abundance of adult smelt and then a decline in mean weight of adult trout. Thus, if a low lake level reduces smelt spawning habitat and hence recruitment, it could be expected to produce a decline in mean trout size within 1–2 years. Conversely, a high water level can be expected to be associated with an increase in smelt abundance and thence mean trout weight in 1–2 years.

To test this hypothesis, data on the mean annual weight of trout measured between 1910 and 2008 (DOC 2013) were compared with data on the water level of Lake Taupo during spring/summer months for the years 1935–2008 (Electricity Commission 2009). Mean weight of trout was low and/or declining during three main periods: 1935–1939, 1951–1965 and 2000–2008. Conversely, the mean weight of trout was high and/or increasing during the three periods: 1940–1950, 1966–1971 and 1993–2000. Optimal conditions for smelt spawning, defined in terms of a high (>356.5 masl) and rising lake level during spring/summer months, occurred four times more frequently during the periods when mean annual trout weight was high and increasing than when it was low and declining (Fig. 10.4). There was also a linear relationship between changes in annual mean trout weight and the proportion of time when optimal smelt spawning conditions occurred (Fig. 10.4).

This relationship suggests that low and/or declining water levels due to a drawdown for hydroelectric generation may reduce smelt recruitment in this lake, resulting in a low abundance of adult smelt and then reduced growth of trout. This relationship could indicate that reduced water inflows into Lake Taupo during drought years could reduce both lake level and nutrient supply to the limnetic zone, respectively reducing spawning habitat and zooplankton production. This

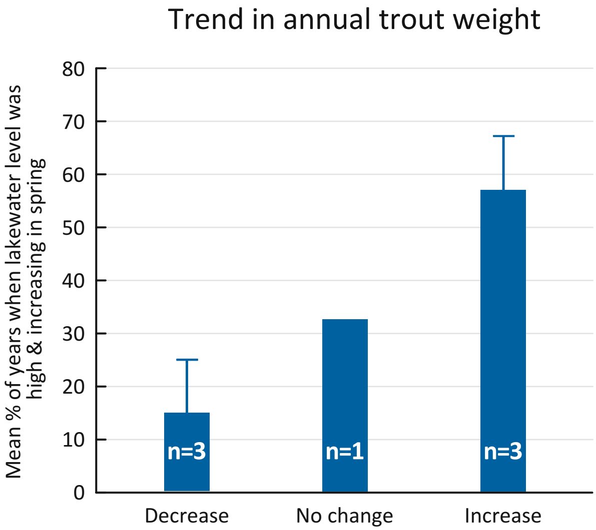

Fig. 10.4 Trends in trout weight during periods when smelt spawning habitat was relatively low, medium and high. (*vertical bars show the upper range for values*)

would then cascade down to a lower production of smelt. Both mechanisms could act separately, or in concert, to reduce smelt. This aside, such findings underscore the potential importance of research to identify key or limiting habitats for indigenous fish in lakes and they emphasise the need to maintain lake hydrology during key life-history periods for littoral-spawning fish. They also highlight the fact that complex food-web interactions may need to be considered when restoring fish communities across multiple trophic levels (see also Feature Box 10.1).

10.2.2 Dwarf Inanga

In New Zealand lakes, degradation of littoral habitats used by small, indigenous fish has also been caused by the invasion of exotic species of macrophytes. These cover the littoral zone and both reduce and simplify littoral zone habitats. In 1990, beds of the invasive weed *Elodea canadensis* extended from the bed of Lake Waingata (Northland) up to the water surface. These weed beds greatly restricted the area of open-water habitat in the littoral zone, which is frequented by schools of dwarf inanga (*Galaxias gracilis*) in other lakes (Rowe and Chisnall 1996, 1997). The large (>50%) loss of open-water habitat in the littoral zone of Lake Waingata was, therefore, thought to be a major factor responsible for the low abundance of dwarf inanga in this lake (Rowe and Chisnall 1997). As this fish is a genetically rare, lake-adapted stock derived from diadromous stocks of *Galaxias maculatus* (Ling et al. 2001), its restoration was important. Hence, the removal of *Elodea* was undertaken to increase the area of open-water littoral habitat and the abundance of dwarf inanga.

To determine the effects of *Elodea* removal and the consequent increase in open-water littoral habitat on dwarf inanga abundance, macrophytes were eradicated from

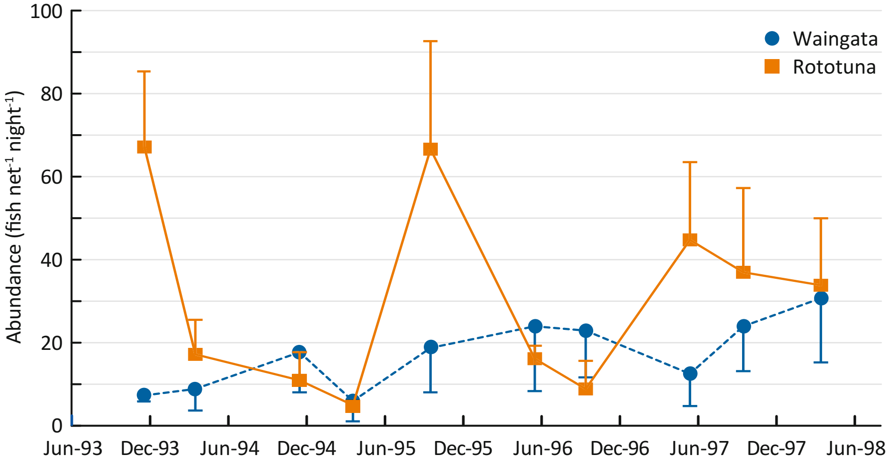

Fig. 10.5 Changes in the annual abundance (mean CPUE) of dwarf inanga in Lakes Waingata (weed-removal treatment lake) and Rototuna (control lake), Northland (*vertical bars are one standard error*)

Lake Waingata using a combination of herbicide treatment and grass carp (*Ctenopharyngodon idella*) (Rowe et al. 1999). The abundance of dwarf inanga was measured annually (using small, fine-meshed fyke nets placed around the lake edge) in both this treatment lake and in a nearby control lake (Lake Rototuna) where no weed removal occurred. The results (Fig. 10.5) indicate that, although the CPUE of adult dwarf inanga in the control lake fluctuated greatly from year to year, it showed no upward trend, whereas CPUE increased steadily in Lake Waingata from a low of 7 fish net^{-1} night^{-1} up to 28 fish net^{-1} night^{-1} 5 years later. Removal of the *Elodea* and restoration of open-water, littoral habitat was, therefore, associated with an increase in the relative abundance of dwarf inanga to four times its baseline level.

10.2.3 Common Bully

The abundance of small benthic fish, such as the common bully, is reduced by invasive macrophytes. The common bully is a small fish (max. length $<$ 100 mm), and it is most abundant in the littoral zone of lakes during summer months where it feeds on small invertebrates and lays defended nests of eggs on hard substrates. The replacement of the low-growing (1–3 m high) native plant species in the littoral zone of lakes with dense, surface-reaching stands of invasive macrophytes such as *Elodea canadensis*, *Egeria densa*, *Lagarosiphon major* and *Ceratophyllum demersum* reduces the benthic habitat for common bullies. Hence, removal or reduction of these macrophytes can be expected to increase common bully abundance in the littoral zones of lakes.

Large increases in the littoral abundance of common bullies occurred in Lake Waingata following the eradication of the *Elodea* beds (Rowe et al. 1999). Similarly, large increases in bully abundance occurred after *Egeria densa* was eradicated in Parkinson's Lake (Mitchell 1986). These results confirmed the inverse relationship between large standing beds of exotic macrophytes and bully abundance in lake littoral zones. In more recent times, large increases in bully abundance have been recorded in the littoral zones of other lakes following removal of invasive macrophytes. For example, the mean CPUE of bullies was monitored in Lake Opouahi (Hawkes Bay) using an array of fine-mesh fyke nets and it increased by 46 times after removal of *Hydrilla verticillata* (Smith and Rowe 2011). Similarly, bully abundance increased by 20 times after a 90% reduction of *Hydrilla* in Lake Tutira (Hawkes Bay) (Smith and Rowe 2011).

Such large increases in bully abundance in the littoral zones of lakes following the removal of invasive weeds are related not only to an increase in the area of benthic, littoral habitat occupied by bullies, but also to the proliferation of benthic, chironomid larvae, which are a major prey species for bullies in lakes (Hayes and Rutledge 1991). Chironomid larvae increased in the silty sediment of the littoral zone of Parkinson's Lake after the exotic macrophyte beds were removed (Mitchell 1986). Hence, removal of exotic macrophytes in lakes changes the littoral food web, with benthic production from chironomid larvae largely replacing the macrophyte-based invertebrate food web (Hofstra and Rowe 2008). This change in littoral zone habitat and benthic food web reflects the same processes involved in the increase of common bully as lake trophic status increases (Rowe 1999).

These examples of indigenous fish recovery following manipulations of lake trophic status, water level and invasive macrophytes highlight the importance of littoral zone habitats for these indigenous fish species in New Zealand lakes and they illustrate the roles that different stressors have on the various habitats. Although weed control was undertaken specifically to improve littoral habitat for indigenous fish in Lake Waingata, this is not the norm and most weed control is currently undertaken to restore other lake values (e.g. swimming, angling and boating). Hence, the effect of invasive macrophyte removal on indigenous fish habitat in lakes is incidental rather than intentional. This reflects the paucity of knowledge about fish habitats and abundance in New Zealand lakes and, in the context of lake restoration, underscores the need for greater research in this area.

10.3 Restoring Connectivity

Although loss of littoral zone habitat is a major factor responsible for the decline of small indigenous fish in a number of New Zealand lakes, loss of connectivity for fish movements and migrations can also have a large and widespread impact on fish abundance. For example, the migrations of diadromous species from the sea into inland lakes can be reduced or prevented by: (a) low water levels in outlet streams because of large water abstractions, (b) water control gates and diversions for drainage or irrigation schemes and (c) dams for hydroelectric power generation and potable water supply. Migrations of non-diadromous fish populations within lake systems may also be impeded. For example, some juvenile, indigenous fish migrate from lakes into inlet streams to reach adult habitats, whereas adults migrate from lakes into inlet streams to spawn. Both juvenile and adult lacustrine smelt undertake seasonal upstream migrations between Lake Rotoiti and Lake Rotorua, two interconnected lakes near Rotorua. Such inter-lake movements are also important for maintaining fish populations and loss of connectivity, because of barriers to these movements, could reduce their populations.

10.3.1 Longfin and Shortfin Eels

Over the past 50 years, eel stocks in many New Zealand waters have declined because of a combination of over-fishing, habitat loss and reduced elver recruitment (Jellyman 2012). In particular, widespread over-harvest of adult eels together with reduced recruitment of elvers into many lakes because of dam construction is thought to have contributed to a recent (post-2000) decline in the annual nationwide recruitment of elvers (Jellyman 2012).

Both shortfin eel (*Anguilla australis*) and longfin eel (*Anguilla dieffenbachii*) are particularly susceptible to a loss of connectivity from barriers to upstream migration. This is because, although elvers of both species are good climbers and can readily ascend the near-vertical, wetted rock faces of natural falls up to at least 20 m high, as well as dam faces up to 43 m (McDowall 1990), many dams are too high and/or steep even for elvers to negotiate. As eels cannot breed in freshwater nor develop landlocked populations in lakes (as do many other diadromous species in New Zealand), their density in otherwise pristine lacustrine and riverine habitats above such barriers decreases.

In addition to high dams, weirs, flood control structures (e.g. floodgates, pump stations) and perched culverts can also prevent elver migrations and stop the annual recruitment of elvers to smaller, lowland lakes and ponds. Furthermore, changes in channel morphology and/or flow can reduce elver recruitment, as in Lake Tutira (Hawkes Bay) where a natural fall on the outlet stream became insurmountable to elvers, presumably after the 1931 Napier earthquake (Smith and Rowe 2011). Closure of river mouths due to gravel deposition at the outlet to the sea can also create barriers to elver migration. This occurs naturally at the outlet to Te Waihora (Lake Ellesmere) and has occurred in the outlet stream for Roto o Waiwera (Lake Forsyth) in Canterbury (Jellyman and Cranwell 2007). Thus, a number of factors other than high dams also create barriers to elver migrations into lakes and not all of these are caused solely by human activity.

High dams are known to block or restrict elver migrations to a large number of the major inland lakes and reservoirs of New Zealand, including those listed in

Table 10.3 New Zealand lakes and reservoirs where a lack of eel life-cycle connectivity is being overcome by construction of artificial elver passage and/or stocking of elvers

Lake/reservoir affected	River system	Dam blocking elver access
North Island		
Wilsons Dam Reservoir	Waiwarawara	Wilsons Dam[a]
Lake Waikare	Waikato	Lake Waikare Weir[a]
Lake Karapiro, Arapuni, Waipapa, Maraetai, Ohakuri	Waikato	Karapiro Dam[a]
Lake Matahina	Rangitaiki	Matahina Dam[a]
Lake Aniwhenua	Rangitaiki	Aniwhenua Dam
Lake Whakamarino, Waikaremoana	Wairoa	Piripaua Dam[a]
Lake Ratapiko (aka Rotopiko)	Waitara	Motukawa Dam[a]
Lake Rotorangi	Patea	Patea Dam[a]
South Island		
Lake Brunner	Arnold	Arnold Dam[a]
Lakes Tekapo, Pukaki, Ohau, Aviemore, Benmore	Waitaki	Waitaki Dam[a]
Lakes Wakatipu, Hawea, Wanaka	Clutha	Roxborough Dam[a]
Lake Dunstan	Clutha	Clyde Dam
Lake Manapouri, Te Anau	Waiau	Mararoa Weir[a]

[a]Dams where elver passes exist or where they are trapped for transfer to lakes above dams

Table 10.3. The restoration of eel stocks in these lakes and reservoirs has, therefore, focused on the restoration of connectivity rather than on control of over-fishing or restoration of degraded habitats and food sources. Restoration of elver recruitment over dams to upstream lakes and reservoirs in New Zealand was initially attempted by the construction of eel ladders over high dams such as the Patea (Mitchell et al. 1984) and Matahina Dams (Mitchell and Boubée 1989). These passes were designed to exploit the climbing ability of elvers and were constructed from lengths of 100 mm diameter PVC tubing, lined with a coarse material such as gravel, or filled with bottlebrush to provide elvers with purchase. They were positioned so that a small flow of water constantly trickled down them. Although they worked well on a short-term basis, their efficiency declined over time mainly because of a lack of on-going maintenance. Hence, “trap-and-transfer” of elvers over dams is now used more generally throughout New Zealand (Martin et al. 2013). Trap-and-transfer involves the installation of an elver capture facility in the river at the base of the dam during the elver migration season. Vehicular transport of the elvers over the dam is then carried out by those responsible for the fisheries. As a consequence, elver stocking is limited to lacustrine sites accessible by road. Trap-and-transfer has proved more successful than eel ladders and is now the preferred restoration measure for elver recruitment by the South Island Eel Industry Association (Chisholm Associates 2009).

Today, elver trap-and-transfer operations occur annually at a large number of dams throughout New Zealand (Boubée and Jellyman 2009), with elvers being transferred into most of the major reservoirs or lakes behind high dams in both the

North and South Island (Table 10.3). The number of elvers transferred above dams has increased over time as the trap-and-transfer operations expand and become more efficient. For example, the number of elvers transported above the Matahina Dam into the waters of lakes Matahina and Aniwhenua, central North Island, increased from less than 500,000 per annum between 1983 and 1997 to over two million by 2001, and then to 3.5 million during the years 2007–2009 (Smith et al. 2009). The number of elvers estimated to have been transported above all dams at Ministry of Primary Industries elver monitoring sites in New Zealand increased from 20,000 in 1983 to 7.9 million in 2009 (Martin et al. 2013).

Elver stocking has undoubtedly resulted in an increase in adult eels in habitats once rendered inaccessible by the construction and operation of dams. However, the effects of long-term elver stocking on eel populations in lakes and reservoirs are still largely unknown. Reliable evidence that the trap-and-transfer operations has restored eel stocks in lakes and reservoirs is limited because few standardised surveys have been carried out to measure changes in adult eel density in these restored environments. Chisnall et al. (1998) carried out eel stock assessments in three Waikato River lakes above dams and recorded age frequency distributions for the eels showing that long-term survival of at least some of the stocked elvers had occurred, but adult abundance was not measured. Fyke-net surveys in Lake Matahina and Aniwhenua carried out in 1988, 1996, 2007 and 2008 indicated that the CPUE of adult shortfin eels more than doubled between 1988 and 2008, but this was mainly because of a very high catch rate recorded in 2008 (Smith et al. 2009). There was little difference in CPUE between 1988 and 2007; hence, the main increase in CPUE occurred between 2007 and 2008. This abrupt increase does not reflect a long-term trend of increasing eel abundance as would be expected with annual elver stocking. It is more likely to be due to some factor increasing eel catchability in 2008.

Stronger evidence that elver stocking restores adult eel abundance above dams was obtained in Lake Ohakuri in the Waikato River system. Here, a comprehensive CPUE survey with 20 baited fyke nets indicated that there were no eels above the Ohakuri Dam in 1996. In contrast, a repeat survey carried out at the same sites in 2012, after annual elver stocking had commenced, recorded adult eels in all nets with the total catch of 87 eels ranging in length from 20 cm to 75 cm TL (Rowe et al. 2013). Although the mean CPUE of adult eels in this reservoir was relatively high a decade or more after annual elver stocking had started, there is no known baseline for eel abundance in lakes or reservoirs where elver recruitment is not limited. Hence, it is currently difficult to assess the overall success of elver trap-and-transfer efforts in New Zealand lakes and reservoirs.

Trap-and-transfer is now the main method for restoring elver recruitment to New Zealand lakes but fish ladders and passes are still maintained at some dams (e.g. Martin et al. 2013; NIWA 2015). Restoration of elver recruitment to Te Roto o Waiwera (Lake Forsyth), where closure of the river mouth by wave-induced gravel deposition prevents elver entry, is also being planned. This will involve artificial opening of the gravel bar at the lake outlet during the spring elver migration season (Mitchell and Associates 2010). Similarly, a mussel spat rope (David et al. 2009;

David and Hamer 2012) has recently been placed over the natural fall below the Lake Tutira outlet stream to allow elvers to climb this barrier and hence restore elver recruitment to this lake (Department of Conservation 2014). The success of these restoration methods is yet to be documented, but the approaches offer promise for facilitating fish passage into lakes and reservoirs for a range of indigenous climbing fish species as well as elvers.

Eels are not the only indigenous species in lakes affected by barriers to migration and a consequent loss of connectivity. Vertical slot fish passes have been designed and installed to allow the passage of other fish into New Zealand lakes. For example, a fish pass on the Mararoa Weir was installed in 1999 to allow both eels and salmonids to access Lake Manapouri (Boubée et al. 2008). Similarly, a fish pass was constructed on the outlet of Lake Taharoa (Waikato) to facilitate the upstream movement of juvenile grey mullet (*Mugil cephalus*) into this lake (Spartan Construction 2010). Again, the success of these measures is yet to be determined.

These case studies show that restoration of fish connectivity to and within New Zealand lakes and reservoirs is steadily expanding and improving. The methods used are still relatively unsophisticated and, unlike the large engineered approaches used in North American and European lakes, most are small-scale and customised to the climbing or swimming abilities of the various indigenous species affected. As a consequence, there is much scope for further improvement and refinement but this will be dependent on proper, before-and-after monitoring, along with more rigorous control versus treatment study designs.

10.4 Piscivore Control

The avian fauna of New Zealand has a long history of decline related to the invasion of terrestrial environments by introduced, mammalian predators (Holdaway 1989). Similarly, in freshwater environments, endemic galaxiid fish in New Zealand lakes have been decimated by the introduction of predatory brown (*Salmo trutta*) and rainbow (*Oncorhynchus mykiss*) trout (McDowall 1990). In particular, trout have decimated the galaxiids in many New Zealand lakes (McDowall 1990; Rowe and Graynoth 2002; Rowe et al. 2003). The vulnerability of New Zealand's freshwater fish fauna in lakes to these exotic predators is thought to be a consequence of the comparatively few indigenous species present and their relative lack of specialisation, such that exotic species introduced from continental land masses can readily fill vacant niches (e.g. secondary carnivore role) or prey on and outcompete the smaller and more vulnerable indigenous species.

Self-recruiting populations of brown and/or rainbow trout are found mainly in the larger, high-altitude, inland lakes of New Zealand where indigenous fish diversity is relatively low because of limited upstream penetration by the mainly diadromous species. Conversely, in the more numerous and generally smaller, lowland lakes, indigenous fish diversity is often higher because their outlets are closer to the sea and there is annual recruitment from a number of diadromous species. Trout are less common in these smaller, lowland lakes because they often lack the inlet tributary streams required for trout spawning. Annual stocking is, therefore, required to maintain trout populations for anglers in these lowland lakes. But there are many such lakes where trout are not stocked. Hence, the impact of trout on indigenous fish diversity in New Zealand's smaller, lowland lakes is expected to be less widespread than in the larger, high-altitude, inland lakes.

Although trout are less of a concern for biodiversity loss in the smaller, lowland lakes of New Zealand, perch (*Perca fluviatilis*) is now widely distributed in such lakes. Perch cause few problems in European lakes where their natural aquatic predators, such as the piscivorous pike (*Esox lucius*), occur, but perch have reduced indigenous fish populations in some Australian and New Zealand lakes lacking indigenous piscivores (Rowe et al. 2008a). Without natural controls, large populations of perch readily develop in these lakes and reduce indigenous biodiversity. This is well illustrated by the introduction of perch to Lake Rototoa, formerly known as Lake Ototoa, near Auckland. The illegal introduction of perch to this lake between 2000 and 2002 resulted in a rapid and large increase in perch and a correspondingly large decline in the native fish populations present. The mean catch rate of dwarf inanga in ten small-meshed fyke nets placed around the perimeter of this lake dropped from 33.2 to 0.1 fish net^{-1} night^{-1} between 2003 and 2011 (i.e. a 97% reduction in CPUE). This large decline in CPUE was also reflected in the mean catch rate of dwarf inanga in 20 Gee minnow traps spread around the lake edge. This dropped from 22.8 to 0.1 fish net^{-1} night^{-1} between 2003 and 2011 (i.e. a 99% reduction in CPUE). This substantial decline in dwarf inanga abundance in Lake Rototoa between 2003 and 2011 was accompanied by an 84% decline in the mean CPUE of common bully and a 91% decline in the mean CPUE for freshwater crayfish (*Paranephrops planifrons*) (author's unpublished data). The introduction of perch, therefore, decimated the indigenous fish community in Lake Rototoa and perch can be expected to have similarly affected indigenous fish communities in many other small lowland lakes in New Zealand. The presence of perch also provides a major constraint on efforts aimed at restoring native fish communities and indigenous biodiversity in smaller, shallower, eutrophic lakes. This is because perch can reduce water quality in these lakes through top-down effects, especially when other invasive fish species are present (Rowe 2007). Perch is, therefore, considered to be a major threat to indigenous biodiversity in the lowland, coastal lakes of the North Island of New Zealand (Rowe and Wilding 2012). The true extent of damage created by the spread of perch will not become apparent until the indigenous fish populations in these smaller, coastal lakes of New Zealand are more fully explored.

Although both trout and perch have substantially reduced indigenous fish biodiversity in New Zealand lakes through predation, these fish, especially the salmonids, also provide valued recreational fisheries in many lakes. Hence, control of their impacts is not widely supported. Predator control has been rarely attempted in New Zealand lakes because of this and because cost-effective technologies are lacking and the risk of re-introduction by anglers is high. Despite these limitations, we are aware of at least three attempts to demonstrate that control of perch and trout in lakes can increase indigenous fish abundance.

10.4.1 Perch Predation on Common Bully

In 1998, several small lakes in the Sinclair Wetlands of the Waipori River catchment (Otago) contained perch populations dominated by large predatory adults and relatively few juvenile fish (David 2001). Low numbers of common bully were present in these lakes indicating that perch predation was likely suppressing the abundance of common bully. Removal of perch in three lakes (areas 0.2–0.8 ha) was carried out over 2 years (1999–2000) and the resulting fish communities were compared with those in three control lakes matched in size to determine whether the bully populations recovered (Ludgate and Closs 2003). Perch were removed by intensive fishing in spring using both gill and fyke nets.

The capture rate of large adult perch in all three treatment lakes declined to near zero indicating that these fish were all removed or greatly reduced, whereas large adult perch continued to be captured in the non-manipulated or control lakes (Ludgate and Closs 2003). In both the small and medium-sized treatment lakes, where the perch population was reduced to near zero (or eliminated), the catch rates of common bully increased, surpassing catch rates in the medium-sized control lake (Fig. 10.6). In contrast, removal of adult perch in the largest lake resulted in the production of high numbers of juvenile perch and low numbers of common bully throughout the rest of the study. Removal of the larger adult perch in the largest lake was thought to have reduced cannibalism allowing high juvenile perch survival and continued suppression of common bully. These results indicate that common bully

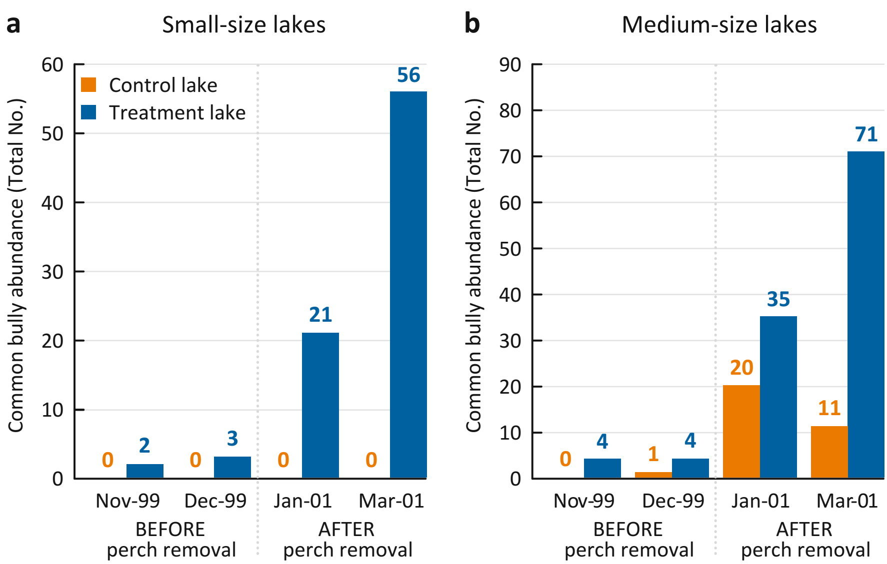

Fig. 10.6 Increases in common bully abundance following perch reduction or removal in: (**a**) small-sized lake (area 0.2 ha), (**b**) a medium-sized lake (0.5 ha), in Otago, South Island (*numbers above bars are the total fish captured*)

recovery can be achieved by perch removal in small and medium-sized lakes. However, the extent and timing of perch removal is critical. If larvae or young-of-the year perch remain, they can produce a population explosion of juvenile perch, which will prevent the recovery of common bully.

10.4.2 Trout Predation on Galaxias in Dune Lakes

Dwarf inanga is currently present in 14 small Northland dune lakes but has been reduced in four of these by exotic fish predators (Rowe et al. 1999, 2003). As the abundance of dwarf inanga had been observed to drop rapidly in several lakes within 2–3 years of trout first being stocked into them (Fish 1966; Rowe and Chisnall 1997), trout predation was thought to be responsible for the scarcity of dwarf inanga in Lakes Taharoa and Waikere (Northland). Mitochondrial DNA studies of dwarf inanga in lakes Taharoa and Waikere showed that this fish differs genetically from the other dwarf inanga populations (Ling et al. 2001), hence it is now known as the dune lakes galaxias and its restoration has a greater priority.

It was thought that restoration of the dune lakes galaxias would require the removal of trout (Rowe et al. 1999). To test this, annual trout stocking was suspended in Lake Waikere in 1993, and gill netting was used over the following 2 years to remove all remaining trout. By 1995, predation by trout on dune lakes galaxias was negligible (Rowe et al. 1999). Meanwhile, annual trout stocking continued in the adjacent Lake Taharoa to provide a control situation to the Lake Waikere treatment. Monitoring of dune lakes galaxias in both lakes, using high frequency echosounding to target schooling juveniles and small-meshed fyke nets in the littoral zone at night to target adults, provided comparable measures of mean CPUE for this fish in both lakes before and after trout removal in Lake Waikere.

Echosounder profiles taken in Lake Waikere revealed a marked increase in schools of juvenile dune lakes galaxias in the limnetic zone (3–10 m deep) from 1994 onward (Fig. 10.7). This reflects their recovery following trout removal. However, adults did not recover and this was attributed to an increased mortality related to a large increase in gambusia in Lake Waikere (Rowe 2003). Diet studies (Rowe and Chisnall 1996; Pingram 2005) had shown that adult dwarf inanga (which are closely related to dune lakes galaxias) were much more dependent on the littoral zone for feeding than juveniles. Furthermore, adult dwarf inanga are pelagic during the day but enter the littoral zone of lakes to feed on littoral prey at night (Rowe et al. 1999; Rowe 2003). Adult dune lakes galaxias were, therefore, expected to display the same behaviour and feed in the littoral zone of Lake Waikere at night. Gambusia increased markedly in the littoral zone of Lake Waikere after trout removal and caused a high mortality of adult dune lakes galaxias through fin-nipping (Rowe 2003). No such large increase of gambusia occurred in Lake Taharoa where trout had not been removed and there were no reports of dune lakes galaxias mortalities in this control lake. Thus, the removal of trout resulted in a recovery of the juvenile stage of dune lakes galaxias in Lake Waikere, but the adults failed to recover because

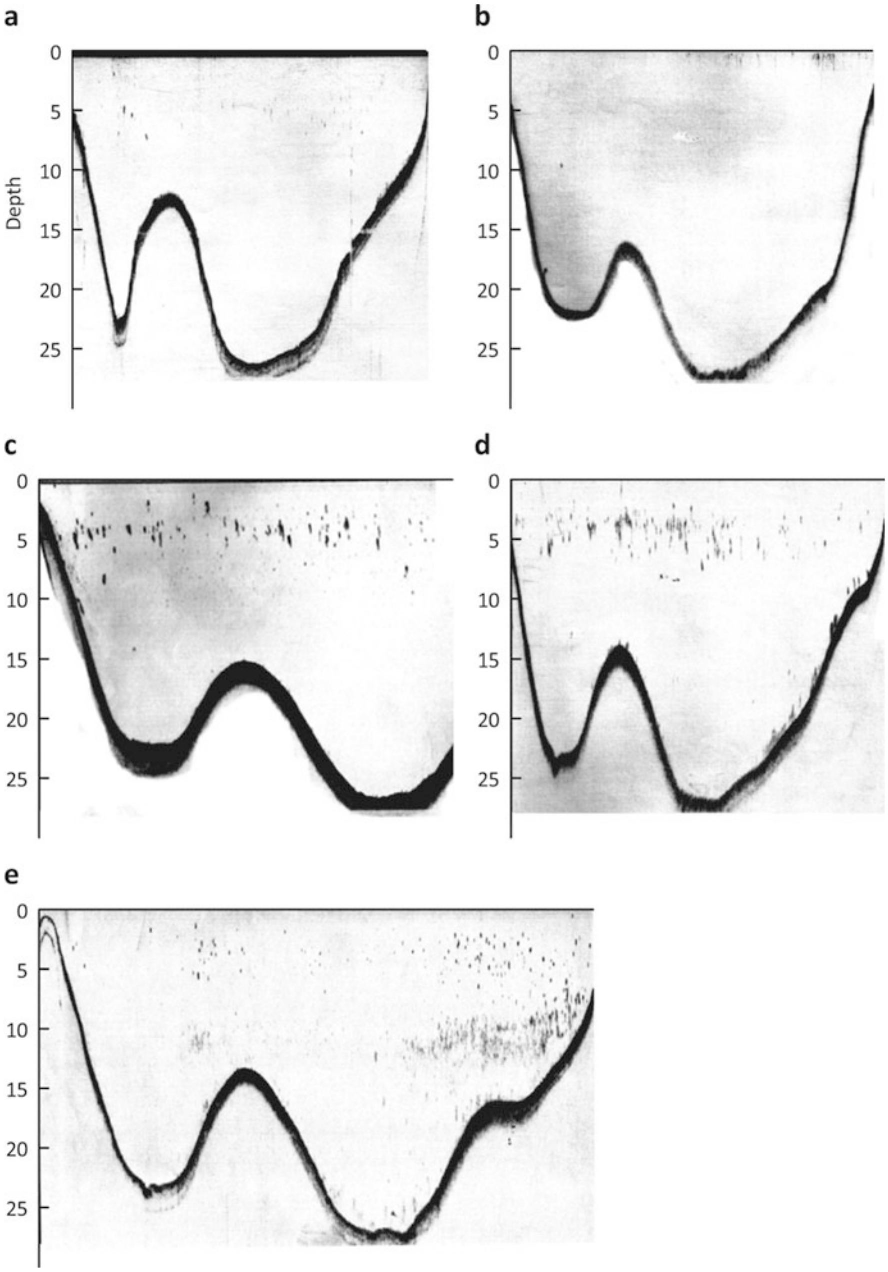

Fig. 10.7 Acoustic profiles across Lake Waikere, Northland, for: (**a**) 1993, (**b**) 1994, (**c**) 1995, (**d**) 1996 and (**e**) 1997 (*the black vertical dashes at depths of 3–7 m are schools of dwarf inanga*)

gambusia increased and provided a constraint on adult dune lakes galaxias survival here. Thus, single-species predator control of invasive species may not always achieve the desired outcome (see also Feature Box 10.1).

10.4.3 Trout Predation on Banded Kōkopu

Banded kōkopu (*Galaxias fasciatus*) abundance increased substantially after brown trout were removed from both the upper Karori Reservoir (Wellington) and its tributary streams using the piscicide rotenone (Pham et al. 2013) (Fig. 10.8). Numbers of brown trout, especially in the larger tributary stream, were initially high and 480 dead trout were retrieved post-treatment with the majority of those being juveniles. Fifty-four larger (adult) trout were recovered from the treated reservoir. Prior to rotenone treatment, repeated electric fishing and spotlighting coupled with dip-netting was used to remove 159 banded kōkopu and 7 freshwater crayfish to upstream refuge areas prior to treatment. After rotenone levels in the tributaries had dropped to non-detectable levels, these banded kōkopu and crayfish were returned to the tributary from which they were captured.

Fish sampling was carried out after the rotenone treatment to help determine the extent of trout removal. One live trout was found in post-treatment monitoring above the barrier used to set the upstream extent of the treatment, but no further trout have been found there or in the tributaries or the reservoir in annual monitoring up to 2015 (Pham et al. 2013; Duggan et al. 2015). After trout were eradicated from the reservoir and its tributaries, a large increase in banded kōkopu occurred in the inlet stream (Fig. 10.8).

A major contribution to the increase in banded kōkopu was provided by the large number of juveniles (i.e. whitebait) recruiting back to the tributary from the reservoir. This naturally diadromous species has a marine larval phase with juveniles

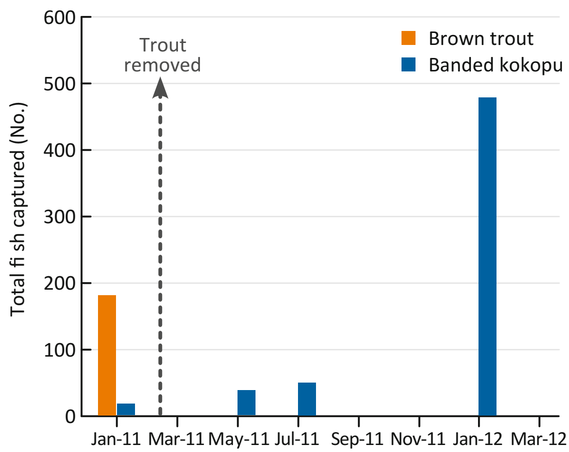

Fig. 10.8 Number of fish counted by spotlighting in a tributary of the Karori Reservoir, Wellington, before and after removal of brown trout (*dashed line indicates when trout were removed*). No trout were captured after May 2011

migrating back to riverine habitats where the adults live and breed. In lakes and reservoirs, landlocked populations can establish with the limnetic zone acting as an inland sea for the development of the larval stage. Although dams prevent the maintenance of diadromous populations in riverine habitats above the dam, reservoirs can allow the development of landlocked populations, provided that adult habitat occurs in the inlet streams and that trout are rare or absent.

10.4.4 Trout Predation on Kōaro

The introduction of smelt to the Rotorua lakes after 1925 benefitted the trout fisheries in these lakes (see Sect. 10.5.2), but it resulted in a further decline in kōaro such that this indigenous fish species is now extinct in at least three of these lakes and rare in the other ten (Rowe et al. 2008b). This is likely to be because the introduction of smelt de-stabilised the relatively simple food chain extending up from zooplankton to kōaro and thence to trout (Rowe 1993). Theoretical studies have shown that simultaneous competition with and predation on another fish species occupying the same trophic niche is not sustainable and eventually leads to the decline of the less competitive species (Pimm and Rice 1987; Vadas 1990). Smelt extinction (as against a severe reduction) occurred in two of the smaller and shallowest, Rotorua lakes (e.g. Lake Rotoehu and Rerewhakaaitu) because they lack the refugia that occur in the larger, deeper lakes where small, relict populations of kōaro still persist (Rowe 1993). Removal of all trout in an inlet stream by the installation of a trout barrier coupled with upstream electric fishing to remove trout above the barrier has been carried out to restore the relict population of kōaro in a section of the Hamurana Stream, a major inlet tributary for Lake Rotorua (Griffiths 2013). Initial results indicate that this has been successful and that both kōaro abundance and distribution has increased above the trout barrier. In time, this can be expected to increase kōaro recruitment to Lake Rotorua.

The above examples of piscivore control demonstrate that dramatic and rapid improvements can occur in indigenous fish abundance in some lakes and reservoirs and in their inlet streams after the removal or severe reduction of a single predator. Although limited in number, these predator control trials have provided experimental confirmation that introduced piscivores (rainbow trout, brown trout, perch) have reduced indigenous fish abundance in New Zealand lakes. These trials have also indicated that whereas piscivore removal can be a successful restoration method, it is not always straightforward and that other factors may limit the recovery of indigenous fish in some lakes.

10.5 Translocation

Translocation has proved to be a useful and successful tool for the conservation of indigenous species of bird in New Zealand's terrestrial environments (Griffith et al. 1989). It has been used to great effect where native birds are translocated to offshore islands or fenced refugia after the eradication of mammalian predators. Translocation (or fish stocking) has also been used to successfully re-establish and restore indigenous fish populations to a number of land-locked lakes in New Zealand. This is feasible in lakes that have few or no tributary streams and hence are ecologically analagous to islands in terms of their isolation and feasibility for predator control. In particular, fish stocking has been used extensively in North America to repair degraded fish stocks (see Feature Box 10.1) and in New Zealand, primarily to establish lacustrine populations of fish from stocks that are normally riverine and diadromous. The earliest freshwater fish translocations into New Zealand lakes were carried out by pre-European Māori (see also Chap. 15). McDowall (2011) has commented extensively on the relatively common practice of translocation of eels and possibly kōaro by Māori in pre-European times. Following the arrival of European settlers, translocation was used primarily by Acclimatisation Societies (now Fish and Game Societies) to restore small indigenous fish species (e.g. smelt and common bullies) to lakes in order to provide a forage fish for trout.

10.5.1 Common Bully

This naturally diadromous riverine species was present in many landlocked lakes in pre-European times, indicating that it was widely stocked into lakes by Māori. These translocations were successful because the naturally diadromous common bully readily establishes landlocked populations in lakes, with the lake acting as an inland sea for the rearing of its pelagic larvae. In more recent times, adult common bullies were stocked into Lake Parkinson (Auckland) following removal of all fish, including the invasive species rudd (*Scardinius erythrophthalmus*) and tench (*Tinca tinca*) by rotenone treatment (Rowe and Champion 1994). These common bullies were sourced from a nearby lake (Lake Tomorata), and the stocked fish readily established a new self-recruiting population in Lake Parkinson. This lake was later re-stocked with rainbow trout to restore the put-and-take fishery, which was dependent on small, native fish as forage species.

10.5.2 Smelt

Between 1925 and 1940, riverine smelt were stocked by the Government Tourist Department into many Central North Island (CNI) lakes near Rotorua in order to replace landlocked populations of kōaro, whose abundance had been greatly reduced by rainbow trout predation (Burstall 1980; McDowall 1990; Rowe 1993). The diadromous smelt rapidly adapted to a lacustrine existence in all of these lakes and soon formed landlocked, breeding populations. The smelt proved to be a viable and sustainable forage fish for trout (Smith 1959) and so it was subsequently stocked into many other North Island lakes, including Lake Taupo (McDowall 1990; Ward et al. 2005). Smelt now underpin trout fisheries in all of the major CNI lakes. Thus, a more specialised and resilient planktivore than kōaro was able to sustain the trout fisheries in these lakes probably because it provided a more stable trophic link between limnetic secondary production (i.e. zooplankton) and trout.

10.5.3 Dwarf Inanga

Other small, indigenous fish have also been successfully stocked into New Zealand lakes to increase native fish diversity, improve fisheries and assist species conservation. In 1988, dwarf inanga from Lake Kanono were stocked into Lake Rototoa on the South Kaipara Head (Northland) to provide a forage species for the stocked rainbow trout population in this lake (Thompson 1989). This translocation was spectacularly successful, and dwarf inanga remained abundant in Lake Rototoa until 2000 when perch were illegally introduced (see Sect. 10.4). Over this period, dwarf inanga provided a key prey species for the stocked trout but also added a new population to the 13 threatened dwarf inanga populations present in Northland, New Zealand. Hence, trout fishery maintenance and the conservation of native fish species were, in this case, aligned.

In general, translocation of diadromous fish species with marine, larval-rearing stages has been highly successful in New Zealand. This is because the lakes act as an inland sea for the rearing of larvae and allow the development of landlocked populations. As a consequence, a landlocked population of common bully is now present in most New Zealand lakes, and translocated populations of the common smelt are present in many North Island lakes as well. Conversely, lacustrine populations of endemic galaxiid species have been largely decimated by fish introductions to lakes. Translocation of other diadromous species, including galaxiids such as inanga (*Galaxias maculatus*), banded kokopu and giant kokopu (*Galaxias argenteus*), has not occurred to any great extent, even though naturally occurring, landlocked populations of these fish exist in some lakes. Hence, there is scope to increase indigenous fish diversity in New Zealand lakes. However, predator control will be required before this is attempted and the consequences of such translocation for ecosystem functioning are, as yet, largely unknown.

10.6 Future Prospects

To date, most methods used to restore native fish in New Zealand lakes have been related to the specific vulnerabilities of the various species, to the types of lacustrine environment in which they occur and to the specific stressors to which they are vulnerable. Hence, these restoration methods tend to be lake as well as species-specific, as occurs in many Northern hemisphere freshwater fish restoration programmes (see Feature Boxes 10.1 and 10.2).

Historically, stocking of small, indigenous riverine species into lakes was used to expand the populations of indigenous fish species in New Zealand lakes as also occurred in North America (see Box 10.1). However, this practice was used extensively in pre-European times by Māori to increase food resources in lakes. After 1900, when most European settlers arrived, stocking and translocation of small indigenous fish was used by Acclimatisation Societies to improve trout fisheries in lakes. More recently, translocation has been used for some conservation purposes, but most of the fish restoration efforts in New Zealand lakes and reservoirs are now applied to overcoming barriers to elver recruitment. This is because legislation, specifically the New Zealand Fish Pass Regulations (1983) and the New Zealand Resource Management Act (1991), requires the construction of fish passes over dams and other barriers. Eel fishery restoration is also mandated by the Treaty of Waitangi because eels underpin many Māori fisheries in lakes and reservoirs and the Treaty guarantees their protection both today and retrospectively.

In contrast, much less effort is applied to restoring lost fish habitat or to overcoming the effects of invasive species on native fish in lakes because the legislation mandating this is comparatively weak and the methods available are undeveloped and/or costly. Although the New Zealand Biosecurity Act 1993 provides a legislative basis for the removal or control of invasive species of fish and plant in lakes, such control is currently targeted primarily at high biosecurity risk species (because of their likely economic impact) and not at native fish recovery, ecosystem restoration or increasing aquatic biodiversity.

It is apparent that the type and extent of native fish restoration applied to New Zealand lakes today is strongly influenced by both the legislative environment and socio-economic considerations. It is not related to objective assessments of need based on conservation priorities, the maintenance of biodiversity or even ecosystem restoration. However, this is expected to change in the near future. The increasing role of Māori in the co-management of freshwater environments and the consequent exercise of Kaitiakitanga (Guardianship) can be expected to bring a more holistic approach to the restoration of New Zealand lakes and lead to greater emphasis on the recovery of indigenous fish. In turn, this is expected to reveal the paucity of information on fish habitats in these ecosystems and it will highlight the need for research to identify key fish habitats in lakes and reservoirs while stimulating the development of better restoration strategies, tools and techniques (see Feature Boxes 10.1 and 10.2).

Over the next decade, restoration of connectivity based on improved knowledge of fish passage requirements can be expected to continue and spread to more lakes and reservoirs. Meanwhile, the development of tools and methods for restoring fish habitat and reducing the impacts of pest fish will become increasingly important because of the increasing iwi and public demand for lake restoration. Because of the complexity involved and the multifactorial nature of both lake rehabilitation and fish stock restoration, better strategies and approaches will be required to guide restoration (see Feature Box 10.1). Failure to develop such tools will greatly constrain future fish restoration efforts in lakes. In addition, cost-effective management of restoration requires assessment of the success or failure of the available methods and, to date, such assessment has proved difficult. This is because few studies are able to provide useful before-and-after comparisons based on adequate pre-treatment baseline data or a comparable control lake well matched to the treatment lake. These limitations reflect the high costs involved in fish restoration in lakes, as well as the long timeframe required for monitoring, but such assessment will be essential if progress is to be made.

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Chapter 11
Indicators of Ecological Integrity

Marc Schallenberg, Mary D. de Winton, and David J. Kelly

Abstract Freshwater Ecological Integrity (EI) incorporates the concepts of ecosystem "health", unimpaired structure, composition and function and a capacity for self-renewal and, as such, it is a holistic advance over standard water quality metrics for assessing lake condition. In the New Zealand freshwater context, EI has been defined as a composite of nativeness, pristineness, diversity and resilience to perturbations. Measurable lake attributes have been proposed and calibrated against pre-impaired "reference" conditions for different lake types. Related to EI, LakeSPI (Submerged Plant Indicator) also assesses lake ecological condition and has been calibrated for a wide range of New Zealand lakes. These EI approaches are thus able to measure departures from reference conditions (or other defined endpoints) and for all these reasons, EI approaches are beneficial for setting lake restoration goals or targets and for tracking the success and progress of restoration activities. Recently, EI approaches have been making inroads with regard to environmental policy and monitoring. This chapter discusses the current development and future possibilities for using EI to help restore degraded lakes.

Keywords Anthropogenic pressures · Reference state · Attributes · Multimetric · LakeSPI

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11.1 Ecological Integrity

When restoring lakes, it is useful to have a restoration goal or endpoint. To date, both management and restoration goals for lakes in New Zealand have focused on water quality and, as a result, lakes are usually monitored for trophic state variables (i.e. chlorophyll *a*, total nitrogen, total phosphorus and, sometimes, water clarity), with little regard to other aspects of lake ecosystems, except sometimes where nuisance species are present. However, popular concepts such as ecosystem "health" and the "life-supporting capacity" of freshwaters (a management target enshrined in the Resource Management Act 1991) indicate that there is an interest in looking beyond trophic state when assessing lake condition and trends, and this is supported by a large body of academic work which has attempted to define and utilise more holistic concepts such as "ecosystem health" (Steedman 1994; Scrimgeour and Wicklum 1996; Rapport et al. 1998), "biotic integrity" (Karr and Dudley 1981; Karr 1996) and "ecological integrity" (Miller 1991; Barbour et al. 2000; Bunn and Davies 2000). Such concepts may be closely aligned to the Māori concept of *mauri*, which can be translated as the embodiment of an "essential life force" (Tipa and Teirney 2006).

Such approaches have been criticised as being subjective and normative (e.g. Peters 1991; Sagoff 2000). However, from a management and restoration point of view, normative, holistic concepts like ecological integrity (EI) may be useful precisely because they incorporate values and value judgements and, as such, can link directly to goals and objectives relating to lake health within policy tools (e.g. catchment management plans, water plans, national policy statements). Furthermore, EI may better reflect how humans perceive freshwater values and condition than trophic state indicators do. Therefore, these more holistic concepts are increasingly being employed at the policy level [e.g. Canadian National Parks Act (2000), Lee et al. (2005), Europe's Water Framework Directive (Boon and Pringle 2009)].

The status of, or change in, lake condition may not accurately track the level of anthropogenic pressures. For example, lakes may show inertia to restoration activities that have reduced or removed major anthropogenic pressures (Scheffer 2004). Such complexities of lake systems can be difficult to understand and predict without a more holistic perspective on lake ecosystem function and restoration. In their formal application of the concept of EI to lakes, Drake et al. (2011) and Özkundakci et al. (2014) reported that, in multi-lake comparisons, environmental pressure gradients (e.g. invasive species, agriculture in the catchments, etc.) only correlated weakly with many common measures of lake condition. In contrast, a gradient of EI (as determined by expert assessments of the lakes after site visits) correlated strongly with many of the pressure gradients tested (Drake et al. 2011). Use of the concept of EI encourages the development and use of multivariate, multi-gradient and multimetric assessments of lake condition which are more likely to track significant ecological responses and help identify aspects of lake ecosystems that may facilitate lake restoration.

11.1.1 New Zealand Definitions of Ecological Integrity

The concept of EI has been developed and refined for New Zealand's terrestrial (Lee et al. 2005) and freshwater environments (Schallenberg et al. 2011). Schallenberg et al. (2011) proposed that, in the freshwater context, EI could be defined as:

> The degree to which the physical, chemical and biological components (including composition, structure and process) of an ecosystem and their relationships are present, functioning and maintained close to a reference condition reflecting negligible or minimal anthropogenic impacts.

Of the four common types of EI definitions listed by Manuel-Navarrete et al. (2004), the freshwater EI definition above is a "wilderness-normative" definition that places pristineness (i.e. departure from a reference condition) at the core of EI. In addition to both functional and structural pristineness, Schallenberg et al. (2011) also proposed three other quantifiable components of freshwater EI: nativeness, diversity and resilience (Fig. 11.1).

In a global context, New Zealand has a high proportion of endemic species which are vulnerable to predation and competition from invasive species (Howard-Williams et al. 1987; McDowall 2006). Nativeness refers to the degree to which ecosystems are composed of biota indigenous to regions of interest. Thus, a high proportion of indigenous taxa in a lake will contribute to the EI of that lake. Accordingly, the assessment of nativeness requires detailed information on the taxonomic composition of biological communities of a lake. The pristineness component of EI can relate to structural (e.g. presence of macrophytes, food web structure, etc.), functional (e.g. productivity, oxygen depletion, etc.), physico-chemical (e.g. water quality, sediment characteristics, etc.) and connective (e.g. dams, diversions, etc.) aspects of lakes, regardless of whether native or exotic biota contribute to these aspects of lake EI.

Biodiversity is a key ecosystem value as indicated in the global Convention on Biological Diversity of which New Zealand is a signatory. Of all habitats on Earth, anthropogenic impacts are having the greatest negative impacts on the biodiversity of freshwaters (Strayer and Dudgeon 2010). Therefore, the maintenance and enhancement of biodiversity within lakes is of great importance. However, linking diversity to EI is not a simple matter because diversity is often unimodally related to disturbance in ecosystems (Flöder and Sommer 1999). In addition, non-native species contribute to diversity, potentially conflicting with the value of nativeness. Furthermore, diversity is measured in a variety of ways (e.g. species richness, Simpson diversity, Shannon diversity, alpha and gamma diversity, etc.) and is dependent on spatial extent or scale of the study area. Despite these operational complexities, diversity is recognised as an important aspect of EI in freshwaters.

Ecological resilience reflects the ability of an ecosystem to return to its original state after a disturbance or perturbation. Resilience is related to the long-term stability of an ecosystem within the context of varying and changing environmental conditions. As such, the inclusion of ecological resilience as a component of EI extends the concept into the temporal dimension. In the context of lake EI, resilience

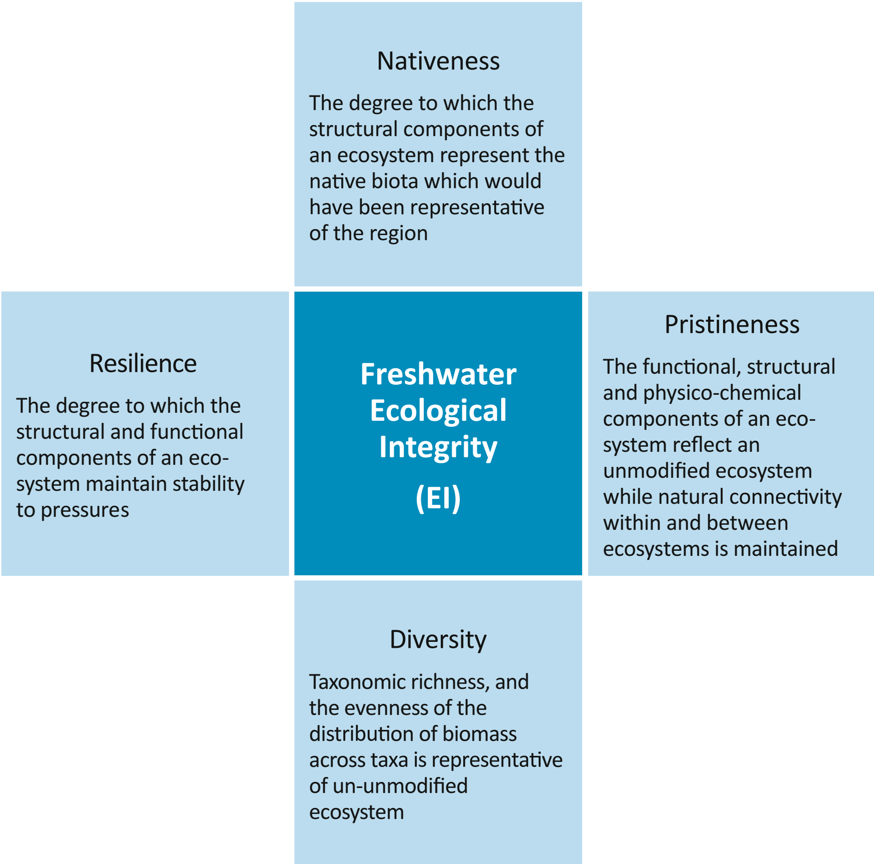

Fig. 11.1 Components of freshwater ecological integrity [after Schallenberg et al. (2011)]

reflects a lake's ability to maintain its structural and functional ecological characteristics despite exposure to environmental variability and change. In this sense, it relates to the presence of beneficial ecological feedbacks within the lake, providing a resistance to (i.e. inertia), and resilience from (i.e. ability to recover), anthropogenic pressures. There are no standard measures of ecological resilience for lakes, so this component of EI may be defined in terms of characteristics which could suggest that a lake is close to an ecological threshold or tipping point (*sensu* Scheffer 2004).

Both Lee et al. (2005) and Schallenberg et al. (2011) calibrated their definitions of EI to a reference condition for terrestrial and lake ecosystems, respectively, emphasising the importance of pristineness within the New Zealand terrestrial and freshwater contexts. In addition to discussing the issue of calibrating EI to reference conditions, Schallenberg et al. (2011) discussed a number of other issues to consider in the implementation of EI, including the scale-dependence and variability (spatial

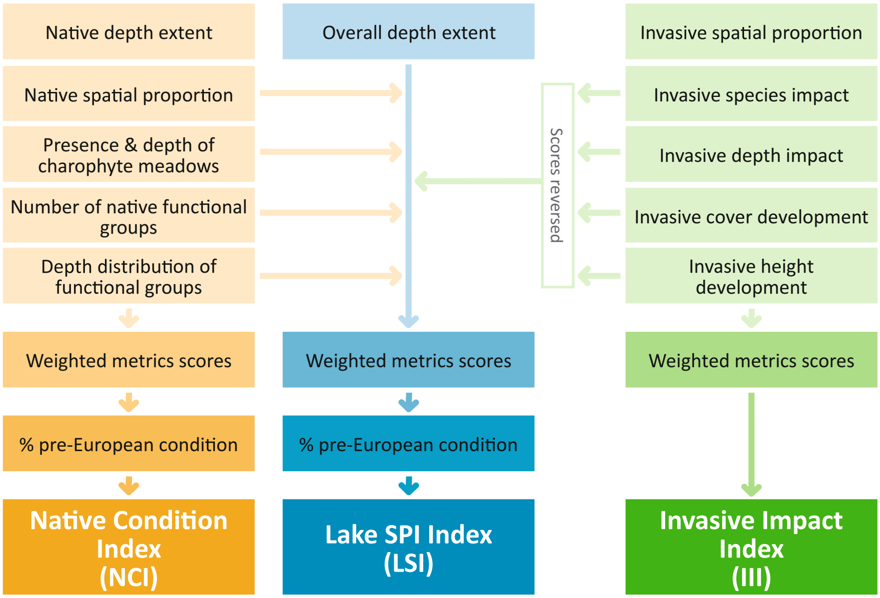

Fig. 11.2 Conceptual overview of the LakeSPI method showing vegetation elements measured and scoring procedure leading to the calculation of three condition indices [modified from de Winton et al. (2012)]

and temporal) of some components of EI and the potential to refine estimates of EI by accounting for different lake types.

Though not specifically designed to assess lake ecological integrity, LakeSPI (Submerged Plant Indicators) is a biological indicator method developed to assess ecological quality of New Zealand lakes (Clayton and Edwards 2006a, b; de Winton et al. 2012). Component metrics were selected for sensitivity to habitat degradation and invasion by alien weeds. LakeSPI metrics assess macrophyte nativeness, diversity and cover (Fig. 11.2) that are related to components of EI (Schallenberg et al. 2011) and could be integrated to a more holistic EI assessment approach for lake restoration (Table 11.1).

In LakeSPI, the emphasis for assessment is on "pristine" vegetation elements that are common to widely varying lake types and geographical locations, so that comparisons between lakes are possible. Nevertheless, LakeSPI does include normalisation for lake maximum depth as an important driver of natural macrophyte depth constraints.

Reference condition of lake vegetation was inferred from a wide range of pristine lakes (from vegetation surveys for >380 water bodies) as well as historical accounts of submerged vegetation (e.g. Kirk 1871; Cunningham et al. 1953). Elements of a pristine (pre-European) lake condition included:

Table 11.1 Examples of LakeSPI indices or metrics, their description, categorisation of metrics within the four components of EI defined by Schallenberg et al. (2011) and relevance for lake restoration

LakeSPI indices/ metric	Definition	EI component	Relevance
LakeSPI index	Integrated measure of % pristine state	Pristineness	Non-vegetated lakes (0% LakeSPI) represent most severely impacted lakes; regime shifts represent major effect on EI (Schallenberg and Sorrell 2009)
Vegetation/native maximum depth/ native distribution	Depth to which plants extend	Pristineness (functional), resilience	Depth extent reduction due to eutrophication (Schwarz et al. 2000) Fluctuations over time indicate instability/possible macrophyte collapse
Charophyte meadows	Depth to which high covers of charophytes extend	Pristineness (structural, functional)	Indicates vegetated area in deeper water Sensitivity threshold to eutrophication (Penning et al. 2008) Potential water quality benefits (Blindow et al. 2014)
Native ratio (note exotic ratio is reciprocal)	Spatial proportion of vegetated area occupied by native plants	Nativeness, resilience	Measure of native plant presence Potential for seed bank formation and maintenance (de Winton and Clayton 1996) Littoral dominance by invasive weeds destabilises system (Champion 2002; Schallenberg and Sorrell 2009)
Nature of invasive cover/invasive maximum height	Measure of invasive performance	Nativeness, pristineness (structural, functional)	Degree of native plant exclusion Change to dense canopy formers Weed impacts on biogeochemical cycles (e.g. Ribaudo et al. 2014)
Invasive species impact	Ranking of species according to invasive ability	Nativeness	Degree of native plant exclusion
Native diversity	Representation by species belonging to up to five functional groups	Diversity	Species richness proxy, structural diversity, depth niche diversity, functional diversity
Native distribution	Key functional group extend >5 m depth	Pristineness (functional)	*Isoëtes* sensitive to combined water clarity and sediment modifications (Brucet et al. 2013)

Note that metrics may span more than one EI component

- Presence of a high number of functional plant groups.
- Development of high cover "meadows" of charophytes, which may extend beyond the depth limits of vascular submerged plants.
- Vegetation development to $\geq$20 m depth or to the lake maximum depth.
- Absence of alien invasive weeds.

It should be noted that LakeSPI cannot be applied to lakes affected by salinity (e.g. coastal lakes and lagoon systems), high altitude (e.g. >c. 1300 m), geothermal water or extremes of pH.

11.2 Key Attributes of Lake Ecological Integrity

Attributes (also known as metrics or indicators) are quantitative measures of EI that are applicable to lakes. Schallenberg et al. (2011) proposed four sets of attributes (referred to as indicators) for measuring lake EI—one set for each of the four components of EI. These attributes were tested conceptually against a set of assessment criteria, and the attributes that appeared most promising in terms of monitoring are presented in Table 11.2.

Lakes and lake habitats are highly diverse, so some specific EI attributes may be better suited to certain lake types than others. Here, the use of a typology divides lakes into three classes that may be useful: (1) polymictic, (2) brackish lakes (including intermittently open and closed lakes and lagoons or ICOLLs) and (3) seasonally stratified lakes (Table 11.3).

Polymictic lakes are lakes which do not thermally stratify on a seasonal basis and are generally fully mixed, but may stratify for short periods of time (e.g. hours or days). In New Zealand, these tend to be shallow lakes (e.g. <10 m maximum depth). Brackish lakes and ICOLLs are generally shallow, coastal lakes with some saline influence through a permanent or intermittent connection to the sea. Seasonally stratified lakes are generally deeper lakes which are thermally stratified for a substantial part of the year, but undergo complete mixing during winter.

Table 11.3 shows that in general, (1) useful nativeness attributes include fish and macrophyte nativeness, (2) pristineness attributes include those typically indicating eutrophication, (3) diversity attributes include species richness of benthic invertebrates, rotifers, macrophytes and phytoplankton and (4) attributes of ecological resilience include cyanobacterial cell density, food chain length and the degree of balance in available N and P (e.g., as indicated by the ratio of dissolved inorganic nitrogen to total phosphorus concentrations).

Özkundakci et al. (2014) analysed the relationships between a range of anthropogenic pressure gradients and EI attributes for 25 seasonally stratifying New Zealand lakes. Instead of applying a typology to the lakes, these authors statistically removed the effects of non-anthropogenic differences in lakes before

Table 11.2 Suggested list of attributes for the assessment of ecological integrity in lakes [from Schallenberg et al. (2011)]

Component of EI	Indicator of attribute	Examples of related stressors
Nativeness	Catch per unit effort (CPUE) of native fish	Invasion by/introduction of exotic species
	Percentage of species native (e.g. macrophytes, fish)	Invasion by/introduction of exotic species
	Absence of invasive fish and macrophytes	Invasion by/introduction of exotic species
	Proportion of shoreline occupied by native macrophytes	Invasion by/introduction of exotic species
Pristineness		
(a) Structural	Depth of lower limit of macrophyte distribution	Eutrophication (benthic effects)
	Phytoplankton community composition	Eutrophication
(b) Functional	Intactness of hydrological regime	Connectedness, abstraction, irrigation, artificial human barriers
	Continuity of passage to sea for migratory fish (potentially indicated by diadromous fish)	Connectedness, artificial human barriers
	Water column DO fluctuation	Eutrophication
	Sediment anoxia (or rate of change of redox state with depth)	Anoxia, eutrophication (ben-thic effects)
(c) Physico-chemical	TLI (or its components)	Eutrophication
	Non-nutrient contaminants	Depends on pressures
Diversity	Macrophytes, fish, invertebrate diversity indices	Loss of biodiversity
Resilience	Number of trophic levels	Loss of top predators
	Euphotic depth compared to macrophyte depth limit	Macrophyte collapse
	Instance/frequency of macrophyte collapse or recorded regime shifts between clear water and turbid states	Macrophyte collapse
	Compensation depth compared to depth of mixed layer	Potential for light or nutrient limitation of phytoplankton growth
	N:P nutrient balance (DIN:TP)	Risk of cyanobacterial blooms
	Presence of potentially bloom-forming cyanobacteria	Risk of cyanobacterial blooms

analysing the pressure-response relationships. They found that 11 attributes were significantly related to anthropogenic pressure gradients, as shown in Table 11.4.

The information in Tables 11.2, 11.3 and 11.4 suggests that some useful EI attributes are likely to be inter-correlated, supplying redundant information to an assessment of EI. The use of the EI framework should include (if possible) attributes

Table 11.3 Attributes or indicators of components of ecological integrity by lake type

Lake type	EI component	Attribute
Polymictic	Nativeness	• % native fish species • % native macrophyte species • % macrophyte cover attributable to native macrophytes
	Pristineness	• Total nitrogen concentration • Total phosphorus concentration • Trophic level index (TLI) • Chlorophyll *a* concentration • Nitrogen loading rate per unit lake area
	Diversity	• No robust attributes were identified
	Resilience	• Cyanobacterial cell density • Food chain length • Ratio of dissolved inorganic nitrogen to total phosphorus
Brackish and ICOLLs	Nativeness	• % native macrophyte species
	Pristineness	• Chlorophyll *a* concentration • Total nitrogen concentration • Total phosphorus concentration • Trophic level index (TLI) • % of the macroinvertebrate community consisting of (Ephemeroptera + Plecoptera + Odonata) by species counts • Maximum macrophyte depth limit
	Diversity	• Benthic invertebrate species richness • Phytoplankton species richness
	Resilience	• Cyanobacteria cell density
Seasonally stratified	Nativeness	• No robust attributes were identified[a]
	Pristineness	• Maximum macrophyte depth limit • Total phosphorus concentration • Chlorophyll *a* concentration • Total nitrogen concentration
	Diversity	• Rotifer species richness • Macrophyte species richness • Phytoplankton species richness
	Resilience	• Ratio of dissolved inorganic nitrogen to total phosphorus

[a]For nativeness, reference conditions should reflect 100% native communities and, because many non-native species disrupt lake ecosystems (Champion 2002; Closs et al. 2004), a departure from 100% native species composition reflects a departure from reference condition

from all four components while reducing the redundancy (over-representation) of attributes within components (Schallenberg et al. 2011). A robust assessment of EI will contain attributes that are more-or-less evenly spread across all four EI components.

It is also apparent that for some lake types, useful EI attributes have been elusive to date. For example, with respect to ecological resilience and diversity, it has been more difficult to identify attributes that are monotonically related to EI. To identify indicators of ecological resilience, the analysis of long-term data sets covering the responses of lake condition over documented perturbation times is required.

Table 11.4 Attributes of ecological integrity (EI) for seasonally stratifying lakes that were significantly related to anthropogenic pressure gradients [from Özkundakci et al. (2014)]

EI attribute	EI component	Proportion variance explained
Trophic level index	Pristineness	0.80
Total phosphorus	Pristineness	0.68
Total nitrogen	Pristineness	0.67
Rotifer species richness	Diversity	0.55
Soluble reactive phosphorus	Pristineness	0.55
Macrophyte native species count	Nativeness	0.51
Chlorophyll *a*	Pristineness	0.49
Macrophyte total species count	Diversity	0.49
Ammonium	Pristineness	0.41
Macrophyte maximum plant depth	Pristineness	0.38
Macroinvertebrate species richness	Diversity	0.38

Therefore, further work is needed to quantify the relationships between diversity and EI and to identify lake attributes that robustly reflect the ecological resilience, resistance and vulnerability of lakes to anthropogenic pressures.

As EI is a normative concept, one could expect that adding a human interpretive element to the assessment could be advantageous. In a study on 45 shallow coastal lakes, Drake et al. (2011) compared EI assessments based on measured attributes with EI assessments made by three limnologists who had visited the lakes and had ranked the lakes independently in terms of an EI gradient. Not only were the three independent expert assessments of the lakes highly correlated, but they correlated with both measured lake attributes and with four anthropogenic pressure gradients impinging on the lakes. In contrast, the measured lake EI indicators correlated more weakly with the anthropogenic pressure gradients, indicating that expert assessments following site visits can provide useful information to EI assessments that is not easily captured by measuring EI indicators (Table 11.5).

11.3 Implementing Ecological Integrity for Lake Restoration

11.3.1 Reversing the Decline of Ecological Integrity

With the arrival of European settlers to New Zealand in the mid-1800s, the modern era of rapid environmental change was initiated, affecting many lakes (Augustinas et al. 2006; Kitto 2010; Schallenberg et al. 2012; Schallenberg and Saulnier-Talbot 2015). Along with decreases in water quality (Gluckman 2017), many native freshwater fish species have become threatened (Goodman et al. 2014), macrophyte communities have been extirpated or invaded by non-native species (Howard-

Table 11.5 Cross-validated correlations of boosted regression tree models built using WoNI (Waters of National Importance) pressure indices [native catchment vegetation removal, imperviousness, N load and P load) from Drake et al. (2011)]

Measured variable	Cross-validated correlation with four modelled WoNI pressure indices
Native fish species % in this survey	**0.30**
Native fish CPUE (common bully + shortfin eel + longfin eel)	0.24
Benthic invertebrate Pielou evenness	−0.05
Macrophyte weighted Simpson index	0.10
pH	**0.65**
Water colour	0.03
Light attenuation coefficient	0.23
TN:TP/Redfield N:P	0.20
Food web mean distance to centroid	0.12
$\delta^{15}N$ range of consumers	0.06
Chlorophyll *a*	**0.36**
TLI	**0.42**
Expert assessment	**0.77**

Values in bold type are statistically significant at $\alpha = 0.05$

Williams et al. 1987; Kelly and Hawes 2005; Schallenberg and Sorrell 2009), hydrology and hydrological connectivity has been altered (McDowall 2006) and other anthropogenic impacts on lakes have occurred (Weeks et al. 2015). Hence, there is an increasing need to restore the ecological integrity and associated ecosystem services of degraded lakes (Schallenberg et al. 2011).

Shallow lake ecosystems tend to exhibit strong biological interactions, and non-linear responses to changes in ecological drivers are common among such ecosystems because of the implicit importance of ecological feedbacks (Scheffer 2004). Such feedbacks confer inertia, resistance and resilience to perturbations, and while this may confer a surprising amount of assimilative capacity for pollutants in well-functioning lake ecosystems, feedbacks can also impart non-beneficial ecological inertia when they impart resistance to attempts to restore a lake. Accordingly, a lake may respond suddenly to gradual changes in pressures and it may appear recalcitrant toward substantial efforts to restore the lake by reducing pressures on it. For this reason, it is important to understand that ecological feedbacks and inertia may be beneficial for management (by enhancing assimilative capacity), but may also hinder restoration. Embedding EI into lake restoration planning is more likely to identify solutions for overcoming undesirable feedback mechanisms.

11.3.2 Ecological Integrity and Reference Condition

Lake restoration is facilitated by clearly defining the goals of the restoration activities undertaken. These goals can be difficult to establish, especially if relevant information for "reference lakes" (lakes similar to that being lake restored, but in pristine condition) is not available (see Feature Box 11.1 for methods to determine reference conditions). To address this, palaeolimnological studies of degraded lakes allow inferences about the specific historical conditions of the lake prior to its degradation to be made. Carefully interpreted studies of the microfossils and geochemical signatures archived in sediment deposited on the lake bed during historical and pre-historical times provide useful information on reference conditions of the lake. These results, combined with the use of statistical transfer functions relating modern distributions of taxa to environmental conditions, allows quantitative historical inferences about attributes such as phytoplankton biomass, pH, temperature and salinity to be made. Biological proxies of these environmental attributes include chironomids (Woodward and Shulmeister 2006), diatoms (Reid 2005; Schallenberg et al. 2012; Schallenberg and Saulnier-Talbot 2015), pollen (Cosgrove 2011), plant macro-fossil remains (Ayres et al. 2008) and cladoceran remains (Luoto et al. 2013).

Another method for estimating reference conditions is to employ a pressure-response model in a so-called space for time approach. This approach is based on surveys of many lakes spanning a broad gradient of EI. The data from the survey lakes can be used to construct a gradient of EI, which can be used to quantify attributes of the most pristine lakes, thereby providing estimates of the conditions of similar lakes under unimpacted or minimally impacted conditions. Similarly, the use of LakeSPI to determine reference conditions also relies on a large database of lakes sampled in recent times to establish likely macrophyte conditions in minimally impacted or unimpacted lakes. While process-based lake modelling is commonly used to test future scenarios (i.e. climate change scenarios), Hawkins et al. (2010) suggested its use for predicting the historical reference conditions at a site or lake. However, to date, this has not been a typical application of lake deterministic modelling.

Box 11.1 Techniques to Define Reference Conditions in Lakes

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By defining reference conditions, we seek to answer the question "what is the 'natural' ecological condition of a lake?" In New Zealand, reference conditions can, therefore, be considered to relate to pristine conditions present prior to human colonisation. Assessing the extent that the present-day ecological integrity of a lake has departed from a reference state provides a useful

(continued)

Box 11.1 (continued)
measure of impairment which can help to set objectives for a lake restoration programme. Reference conditions may be derived for a range of variables. These include: nutrient concentrations, trophic status, primary productivity, phytoplankton community composition, sediment deposition rates, pH and conductivity. A variety of methods has been proposed to define reference conditions (see Table 11.6), and the choice of which method(s) to use will depend on the extent of existing data, the variables of interest, the number of lakes being studied and the resources that are available.

Table 11.6 Summary of key techniques to estimate lake reference conditions

Method	Advantages	Disadvantages	Further reading
Use data for existing undisturbed sites	• Accurate • Uncontentious • Statistical models can be used to extrapolate results to lakes elsewhere based on characteristics such as lake morphology and soil type	• Undisturbed examples of most lake types no longer exist in New Zealand, e.g. lowland lakes	Cardoso et al. (2007) and Herlihy et al. (2013a)
Paleolimnology (analysis of lake sediments)	• A well-developed field of limnology that has led to the development of advanced techniques which can be used for a range of applications • Quantitative and precise	• Requires resource intensive and site-specific sampling and analysis • Diagenesis processes can alter sediment core composition • May not be applicable to variable of interest. Transfer functions that link biotic indicators (e.g. diatoms) with historic water quality can only yield information about variables that limit productivity	Reid (2005) and Herlihy et al. (2013b)
Stressor-response modelling (hindcasting using statistical models that relate current water quality to human pressures and natural factors in lake catchments)	• Quantitative and precise • Can provide information about a range of variables • Can be used to derive additional knowledge using existing monitoring data • Can account for	• Requires a large sample size • Requires detailed data about catchment characteristics • Does not explicitly reflect in-lake processes	Herlihy et al. (2013b)

(continued)

Table 11.6 (continued)

Method	Advantages	Disadvantages	Further reading
	natural variability in factors such as soils and climate • Models can be extrapolated to un-monitored lakes		
Process-based lake ecosystem modelling	• Quantitative and precise • Can account for inherent natural variability of freshwater ecosystems, e.g. due to seasonal cycles • Can simulate a wide range of variables • A tool to formulate new hypotheses and research questions	• High uncertainty with configuring reference state processes • Extensive resource and data requirements, although simple eutrophication models ("Vollenweider models") may provide a parsimonious approach to simulate dominant in-lake processes and provide static predictions of trophic state variables • Largely unproven for estimating reference conditions	See Chap. 3 of this book
Review local oral and written (non-scientific) historical records	• Can help to engage local communities with lake restoration planning • Can provide useful information about historic abundance of *mahinga kai* species	• Unlikely to provide quantitative data • Information does not pre-date start of human disturbance • Issues with "shifting baselines"	Tipa and Nelson (2008)
Calculate upper statistical distribution (e.g. 25th percentile) of monitoring data for a specific lake type	• Straightforward	• Not a true estimate of reference state • Dependent on the sample composition	Herlihy et al. (2013b)
Expert knowledge	• Essential for interpreting the results of other methods	• Subjective • Imprecise	

11.3.3 Other Ecological Integrity Restoration Endpoints

Benchmarking lake status relative to a pristine reference condition may not be relevant or achievable in all cases. Duarte et al. (2009) showed that simply reducing key human pressures did not guarantee the return of the system to its pristine state. The differing trajectories and endpoints between degradation and recovery observed in four coastal ecosystems (Duarte et al. 2009) may have been due to biological inertia apparent in systems (discussed above) or to the focus on only single pressure and response variables.

Artificial water bodies, such as hydro-electric reservoirs, present a particular challenge for identifying restoration endpoints because the prior natural state of a reservoir was usually a river and, therefore, the pristine reference state is not relevant in the context of restoring a degraded reservoir. However, EI may still be a useful concept for determining management and restoration endpoints for reservoirs even though the target of reference state is not appropriate. In the case of artificial water bodies, which are constructed predominantly for human purposes, other types of goals (e.g. recreation and sports fisheries) may be deemed more important than ecologically derived endpoints. Nevertheless, the management and restoration of such systems should benefit by holistically considering the EI of the systems and their surrounding aquatic habitats.

Societal perspectives and norms are known to shift over time and, therefore, perspectives concerning which conditions are natural and acceptable can shift (Hawkins et al. 2010). A societally acceptable or desirable lake condition (e.g. open park-like lake margins that allow public access and views) may be one that is quite different from a past state. The past acclimatisation of non-indigenous species can result in alien species being considered as accepted components of lakes, while some invasive species are societally valued (e.g. sports fish) or otherwise utilised (e.g. *Salix* sp.—willows—for erosion control). Therefore, in these cases, desired restoration targets may not reflect a maximisation of EI or a return to a reference condition, and such cases may be problematic for tracking improvements in EI or recognising ecological degradation.

Existing ecological lake values may also deviate from the lake's pristine reference state. Palaeolimnological study of a Norfolk Broad, UK, showed that dredging to restore an earlier reference state was likely to remove contemporary substrate that favoured a valued, nationally rare submerged macrophyte (Ayres et al. 2008). Moreover, Ecke et al. (2010) noted that rare and threatened freshwater species were widely represented in European lakes with lower ecological status scores, and that restoration of the lakes to a more pristine state could produce uncertain outcomes for those species.

In recognition of such issues, Stoddard et al. (2006) identified three additional benchmark conditions that should not be confused with pristine reference state: a "historic condition", the "least disturbed condition" and the "best attainable condition" (Fig. 11.3).

It has been argued that all reference conditions reflect a point in the historical trajectory of a lake because lakes naturally undergo aging and succession over time (Ayres et al. 2008; Kowalewski 2013). Thus, historically based restoration targets should specify relevant historical time frames. For example, both LakeSPI and EI as defined by Schallenberg et al. (2011) calibrate to a pre-European-colonisation (i.e. prior to c. 1850) reference state.

Minimally disturbed reference lakes (Fig. 11.3) are lakes with physico-chemical conditions indicative of low levels of anthropogenic impact, located in catchments with the lowest levels of land-use modification. Stoddard et al. (2006) defined "best

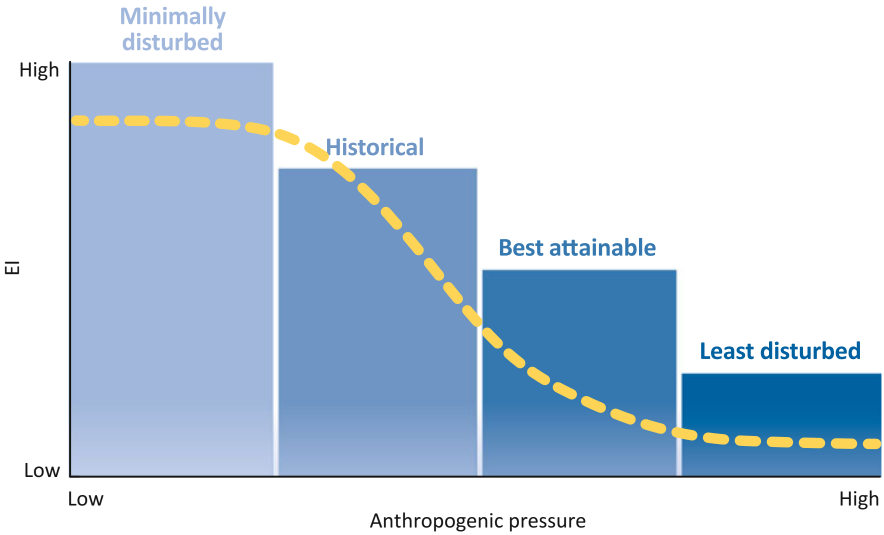

Fig. 11.3 Positions of different restoration target conditions along a gradient of anthropogenic pressure recognising that the targets may be set in relation to (1) a reference (minimally disturbed) condition, (2) a historical condition, (3) a best attainable condition associated with current management practices or (4) to a least disturbed (planned or future) condition [modified from Stoddard et al. (2006)]

attainable condition" as the expected outcome from the application of "best management practice". This is perhaps the most difficult condition to define as it represents an agreed potential, based on current practicable technologies, resourcing and management/societal will. This concept might be applicable to artificial water bodies or those where multiple, often conflicting ecological and human values preclude the attainment of a near-pristine restoration target. Finally, a restoration target set as the best attainable condition may not necessarily preclude further degradation of the water body.

Cultural perspectives can also provide important endpoints for EI. In New Zealand, recent freshwater management reforms have identified the importance of Māori perspectives in freshwater management and restoration (see Chap. 16), in particular for identifying critical components of species composition [harvestable species (*kai*), sacred species (*taonga*)] or function (navigation) (Harmsworth et al. 2013). While the concept of EI is likely to have broad linkages with cultural values and perspectives, so far these have not been specifically considered within an EI framework (e.g. Schallenberg et al. 2011).

Although the European Water Framework Directive (WFD) requires lake assessment to be expressed relative to a minimally disturbed state (see Feature Box 11.2), the endpoint for any restoration is the achievement of "good" ecological status (Brucet et al. 2013). The definition of a good status is contentious, but can best be defined where ecological thresholds in the lake can be quantified (Brucet et al. 2013).

An ecological threshold may occur at a discontinuity in the response relationship of an ecosystem component to a pressure gradient, at a cross-over in the responses of tolerant vs sensitive taxa, or a threshold may result from the breach of a threshold of an attribute having an indirect effect on the response attribute of interest (e.g. a chlorophyll *a* boundary associated with significant retraction in macrophyte spatial extent).

It is important to establish clear and relevant timelines for restoration actions to achieve EI endpoints. For example, where grass carp have been stocked to lakes to eradicate alien weed species for biosecurity reasons, there can be a temporary, unfavourable reduction in water clarity and EI (see Chap. 8). The primary aim of the stocking is to restore native vegetation values by regeneration from seedbanks once invasive species have been eradicated, and the grass carp have been removed. Grass carp browsing during this process could cause some EI attributes to deteriorate (e.g. reduction in plant depth limit, loss of vegetation influence on water quality), while others could indicate improvement (e.g. reduced invasive weed presence and development).

All types of restoration endpoints are likely to benefit from the use of an EI framework, even if the attainment of reference conditions is not the agreed upon goal of a restoration. For a restoration project to be successful, it is helpful for all involved to clearly articulate and agree on the endpoint sought and the timeline for achieving it. In this way, restoration progress can be tracked and the success or otherwise of restoration efforts can be robustly assessed, whatever the goals may be.

Box 11.2 Assessing Ecological Conditions of Lakes Across Europe

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Water is not a commercial product like any other but, rather, a heritage which must be protected, defended and treated as such. This acknowledgement is the cornerstone of the EU's water policy.

The Water Framework Directive (2000/60/EC) is the most substantial piece of water legislation from the European Commission (EC) to date. A core concept of the EU Water Framework Directive is that the structure and functioning of aquatic ecosystems is used to assess the ecological status of surface waters.

Biological communities, such as phytoplankton, aquatic flora, benthic invertebrates and fish fauna, are used to assess the health of aquatic ecosystems. Assessment of quality is based on the extent of deviation from the reference conditions, defined as the biological, chemical and morphological conditions associated with no or very low human pressure. Good status means "slight" deviation from reference conditions, providing sustainable ecosystem

(continued)

Box 11.2 (continued)
and acceptable conditions for human use. The general objective of the WFD is to achieve “good status” for all surface waters by 2015.

Since the adoption of the European Water Framework Directive in 2000, huge progress has been made in the ecological assessment of European waters. Over 90 lake ecological assessment methods are currently in use across Europe. These assessment methods are composed of several metrics (e.g. chlorophyll-*a*, total phytoplankton biovolume and abundance of cyanobacteria), and combination rules are applied to calculate the ecological assessment results for the whole system. The final assessment is expressed as an Ecological Quality Ratio (EQR)—the ratio of the observed assessment value to the expected value under reference conditions. To ensure methods comparability, intercalibration has been carried out by Member States—62 lake assessment methods were intercalibrated and published in an EC Decision (see Table 11.7).

Most of the assessment methods are based on phytoplankton and macrophyte communities, while fewer methods are developed using fish fauna, benthic invertebrates and phytobenthos. Most of the methods focus on the assessment of eutrophication pressure, which is one of the major and the best understood human impacts in Europe, while only a few methods address hydromorpohological alterations and multiple pressures.

For the whole lake assessment, assessment of biological methods are combined using a “one-out-all-out principle”, i.e. the worst status of the elements used in the assessment determines the final status of the water body. Still, the validity of this principle has been strongly debated.

The ecological status of more than 19,000 lake water bodies was assessed using the WFD classification tools (European Environment Agency 2012). Of those, 44% are reported to be in less than good ecological status and will need restoration measures to meet the “good status” objective.

11.3.4 Suggested Ecological Integrity Attribute Guidelines

In Sects. 11.1 and 11.2, some suggested attributes for measuring EI in lakes were presented. However, other attributes may also be relevant, especially for specific lakes and lake types. Ecological Integrity attributes useful for lake restoration should meet a number of criteria. Ideally some consideration should be given to the time frame of measurement because some attributes may respond more quickly than others to changes in environmental pressures/drivers. For example, community structure can be sensitive to anthropogenic stressors, whereas changes to ecosystem metabolism (i.e. community productivity, respiration) may be slower to respond (Schindler 1987). In deep New Zealand lakes, physico-chemical attributes of EI

Table 11.7 Overview of lake assessment methods (only intercalibrated methods)

Biological community	Most typical metrics included	Member states	Pressures addressed	Selected references
Phytoplankton	Chlorophyll-*a* Total biovolume Biovolume of cyanobacteria Sensitivity indices	AT, BE, CY, DE, DK, EE, ES, FI, IE, IT, NL, NO, PL, PT, SE, SI, UK	Eutrophication	Carvalho et al. (2013) and Poikane et al. (2010, 2014)
Macrophytes	Colonisation depth Sensitivity indices	AT, BE, DE, DK, EE, FI, FR, IE, IT, LT, LV, NL, NO, PL, SE, SI, UK	Eutrophication	Pall and Moser (2009) and Schaumburg et al. (2004)
Phytobenthos	Diatom trophic indices	BE, DE, FI, HU, IE, PL, SE, SI, UK	Eutrophication	Kelly et al. (2014) and Schaumburg et al. (2004)
Benthic invertebrates	Total taxa richness Shannon diversity Sensitivity indices	BE, EE, DE, FI, LT, NL, NO, SE, SI, UK	Hydromorphological alterations Acidification Eutrophication	McFarland et al. (2010) and Sidagyte et al. (2013)
Fish fauna	Total biomass Biomass of cyprinids Functional indices	AT, DE, FI, IE, IT	Eutrophication Multiple pressures	Kelly et al. (2012) and Olin et al. (2013)

AT Austria; *BE* Belgium; *CY* Cyprus; *DE* Germany; *DK* Denmark; *EE* Estonia; *ES* Spain; *FI* Finland; *IE* Ireland; *IT* Italy; *LT* Lithuania; *LV* Latvia; *NL*, the Netherlands; *NO* Norway; *PL* Poland; *PT* Portugal; *SE* Sweden; *SI* Slovenia; *UK* United Kingdom

were more strongly related to anthropogenic pressures than biological indicators of EI (Özkundakci et al. 2014). Useful attributes have large “signal-to-noise” ratios, are highly sensitive to anthropogenic pressures and are less sensitive to natural environmental variations.

We have defined EI as a multimetric concept, which should consider attributes reflecting the state of a lake with respect to nativeness, pristineness, diversity and resilience. The inclusion of metrics covering four separate components of EI encourages the assessment of EI to be broad in scope, covering multiple ecological gradients. To date, there has been no attempt to combine attributes from the four EI components into an overall EI score for lakes, although this has been conducted for New Zealand rivers (Clapcott et al. 2011). Currently, nativeness, pristineness and diversity can be measured more confidently than resilience, and more work is needed to develop resilience indicators. Similarly, there continues to be debate about how diversity correlates to ecosystem stressors, functioning and integrity. As work progresses on these questions and more robust attributes for EI components are developed, consideration should be given to how each of the four EI components should be weighted within an overall EI index. While the future development of an overall EI index could be useful, examining the four components separately provides a clear picture of the status and trends of the individual EI components—components which may respond differently across anthropogenic pressure gradients. For this reason, it may be advantageous to examine the EI components separately when undertaking lake restoration.

11.4 Ecological Integrity in Lake Management Policy and Restoration Practice

11.4.1 New Zealand

In New Zealand, regional councils generally focus on water quality in lakes, while the Department of Conservation tends to focus on conserving indigenous biodiversity and habitats (Park 2000). The concept of EI encourages a unification of these two approaches, as well as the consideration of lake ecological resistance (inertia) and resilience (recovery) to anthropogenic pressures. The recent development of a New Zealand freshwater definition of EI (Schallenberg et al. 2011) and its subsequent use in assessments of lake EI and reference condition (Drake et al. 2011; Schallenberg and Kelly 2013; Özkundakci et al. 2014; Schallenberg and Schallenberg 2014) encourage more holistic approaches to lake monitoring, management and restoration.

Among regional and national government departments and ministries, a substantial amount of information is being collected on New Zealand lakes that can already contribute to assessments of lake EI. For example, water quality information collected by regional councils, together with information on indigenous biodiversity and non-indigenous species distributions collected by the Department of Conservation and the National Institute of Water and Atmospheric Research (NIWA), and information on cyanobacterial cell densities collected by public health offices and the Ministry of Health, together represent attributes that cover all four of the EI components. Combining of this information would allow assessment of EI at a national scale and could facilitate the development of policies focused on the maintenance and enhancement of lake EI. The Department of Conservation has further considered an amalgamated monitoring programme as part of its biodiversity monitoring strategy which would combine regional councils' monitoring of water quality with focused monitoring on nativeness and biodiversity to inform a broader assessment of EI (Kelly et al. 2013).

LakeSPI bio-assessments are frequently undertaken to complement traditional water quality monitoring. Individual LakeSPI metrics (see Fig. 11.2), indicating departure from an expected or "pristine" reference state, could contribute to a multimetric EI approach for monitoring water bodies. To date LakeSPI assessments are available for >260 lakes (http://lakespi.niwa.co.nz/).

Loosely related to the concept of EI are the concepts of ecological condition and ecological value that have been used by regional councils in prioritisation schemes to identify high quality lakes. For example, ecological condition was one line of evidence used to prioritise lakes for biodiversity management in the Waikato region, central North Island (Reeves et al. 2009), leading to the identification of significant natural areas for protection. Ecological value was used to identify lakes (i.e. moderate-high to outstanding value) for priority management in Northland (Champion and de Winton 2012) and has guided initiatives to protect and restore lakes (Champion and Wells 2014). While restoration potential was also considered for Waikato lakes by Reeves et al. (2009), this was based more on feasibility and time-frame required rather than EI criteria. None of these schemes identified reference conditions or indicated the degree of departure of individual lakes from an expected pristine state, but they have identified some minimally disturbed conditions for lakes in several of New Zealand's regions.

With the development of the New Zealand lake EI framework (Schallenberg et al. 2011), the opportunity has arisen for regional council lake managers to place their lakes within an EI context. For example, studies commissioned by the Tasman District Council (Schallenberg 2011), Environment Southland (Schallenberg and Kelly 2013) and Environment Canterbury (Schallenberg and Schallenberg 2014) have used the EI concept to assess the ecological condition of lakes in those regions to help define reference conditions for the lakes and to calculate current departures of the lakes from their EI reference conditions (Fig. 11.4).

While the concept of lake EI hasn't yet been explicitly used to direct lake restoration, the identification of a poor state of an EI attribute could encourage the adoption of specific lake management and restoration actions. The type of information summarised in Fig. 11.4 can help set restoration goals and targets, even if achieving them is aspirational rather than realistic in the short term.

11.5 Future Prospects

Recent government guidance has set some national freshwater guidelines for maintaining or improving ecosystem health, defined as,

> supporting a healthy ecosystem appropriate to that freshwater body type, where ecological processes are maintained, there is a range and diversity of indigenous flora and fauna, and there is resilience to change. (Ministry for the Environment 2014)

This goal appears to be compatible with maintaining and enhancing EI, although current lake attributes are limited to trophic state variables such as nitrogen, phosphorus and chlorophyll *a* concentrations. Thus, the EI framework provides useful guidance for selecting ecological attributes (beyond water quality attributes) which could contribute to more holistic assessments of lake health.

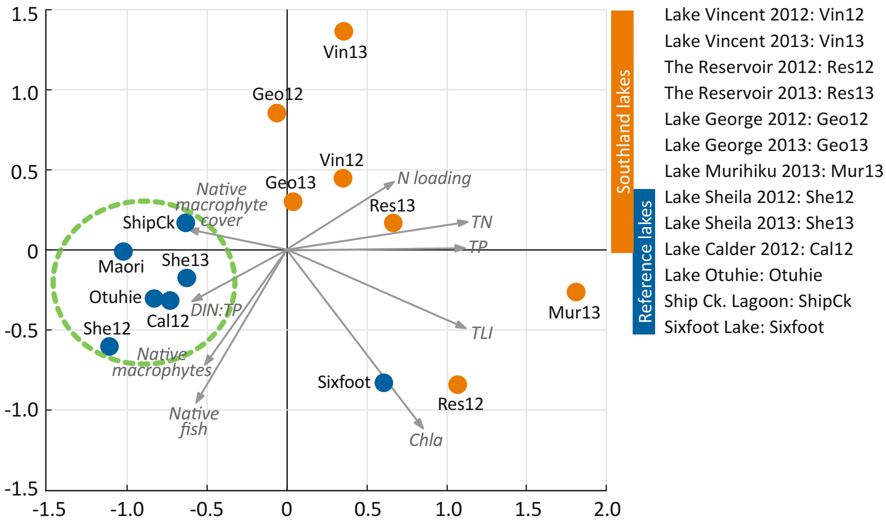

Fig. 11.4 Ordination plot showing relationships between Southland reference lakes (blue circles) and other Southland lakes (orange circles), based on nativeness, pristineness and resilience indicators of EI. Six-foot Lake (Campbell Island) was considered a reference lake although it was eutrophic. The green, dashed circle encloses the other reference lakes. The *x*-axis explains 47% of the variation and can be interpreted as a gradient of pristineness and nativeness. The *y*-axis explains 20% of the variation [from Schallenberg and Kelly (2013)].

Ecological Integrity encourages the adoption of a holistic perspective on lake policy, monitoring, management and restoration. This chapter highlights how EI can be useful for establishing lake restoration goals and targets, whether they be set for the purpose of restoring to reference conditions or to another endpoint along the lake EI gradient. Either way, the EI concept encourages the setting of multiparameter restoration endpoints, resulting in more holistic monitoring of lake health status and recovery. Ecological Integrity encourages progression away from a common restoration perspective which argues that if the physico-chemical environment is restored, then the rest of the ecosystem will necessarily restore itself. It has been shown that restoration strategies that focus only on reductions in nutrient loading (for example) often fail to achieve predicted outcomes (e.g. Duarte et al. 2009). Holistic approaches to restoration that are aimed at restoring diverse ecosystem components are more likely to achieve desired restoration outcomes because some key synergistic interactions among ecosystem components are likely to increase the rate of recovery and the probability of restoration success, while others are likely to hinder these outcomes. In addition, the adoption of an EI approach to restoration encourages the safeguarding and restoration of lake ecological resilience, which should produce more reliable long-term restoration outcomes.

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Chapter 12
Biodiversity Genomics: Monitoring Restoration Efforts Using DNA Barcoding and Environmental DNA

Ian D. Hogg, Jonathan C. Banks, and Steve M. Woods

Abstract We review recent advances in the use of molecular techniques as they apply to monitoring restoration efforts in lakes. Using DNA sequence data, biodiversity can now be assessed to levels previously unattainable using traditional, morphological assessments. In particular, DNA barcoding, the use of small standardised fragments of DNA, has become an increasingly widespread and common approach to identify species. Global initiatives such as the International Barcode of Life (iBOL) have coordinated these efforts and facilitated publically accessible reference databases such as the Barcode of Life Datasystems (BOLD). Such databases can be used for routine identification of specimens as well as for the assessment of community composition and monitoring of changes over time. Through the application of Next Generation Sequencing techniques, multiple samples can be run simultaneously (metabarcoding), greatly automating and streamlining the monitoring process. Reference databases can also be applied to environmental DNA (DNA that is shed into the environment by plants and animals). Here, species can be identified "sight unseen" through analyses of environmental samples (e.g. water, sediment). This latter method has proven useful for the monitoring of exotic fish species, particularly following eradication efforts. Ongoing developments in sequencing technology are likely to further enhance the utility of molecular techniques for assessing and monitoring restoration efforts in New Zealand.

Keywords DNA barcoding · Barcode of life datasystems · Genetic characterisation of fauna · eDNA · International barcode of life

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12.1 Introduction

The accurate assessment of species-level diversity within and among natural habitats is fundamental to restoration-based monitoring studies. To date, our ability to accurately identify species has been primarily based on morphological characteristics—the basis of traditional taxonomic approaches—and relies on the availability of appropriate taxonomic guides and taxonomic expertise to ensure accurate identifications. Monitoring programmes based on community-level analyses such as those based on macroinvertebrates (e.g. Stark 1993) or the Rotifer Trophic Level Index (Duggan 2007) also require the time-consuming task of sorting samples followed by the identification of individual taxa by skilled taxonomists. For some groups, access to taxonomic guides and appropriate experts is limited. Further, numerous morphologically delineated species are now known to consist of complexes of morphologically similar, yet genetically distinct populations (e.g. Hogg et al. 2006). As an alternative, molecular techniques such as DNA barcoding (Hebert et al. 2003a)—the use of short, standardised fragments of gene sequences—have shown promise to simplify and streamline the process of routine identification and monitoring of freshwater systems.

This chapter provides an overview of the progress that has been made in the development of protocols for the identification of lake fauna and community-level analyses using DNA barcoding. The chapter provides background on the development of reference libraries, the use of reference libraries for species and community-level analyses, and an example of a related field study. The use of "environmental" DNA to detect invasive species such as brown trout or koi carp is also covered. The chapter concludes with a section on future directions in the field of biodiversity genomics as they relate to restoration of lake habitats.

12.2 Genetic Identification of Lake Inhabitants

The premise of genetic identification is to use underlying genomic features of an organism to categorise individuals as belonging to a particular population or species. This is most often achieved through a direct analysis of an individual's DNA sequences, although indirect genetic techniques such as allozyme electrophoresis (e.g. Hogg et al. 1998) have previously been used to assess differences among individuals or populations. DNA sequences are a "string" of nucleotides consisting of four bases: adenine, cytosine, guanine and thymine (A, C, G, T), the order of which varies among species. It carries the "blueprint" for every living organism, determining physical and behavioural attributes. By identifying appropriate gene regions, individual species can be identified using short (<700 nucleotides) fragments of appropriate target genes. In this way, each species has a unique DNA "signature" or "DNA barcode".

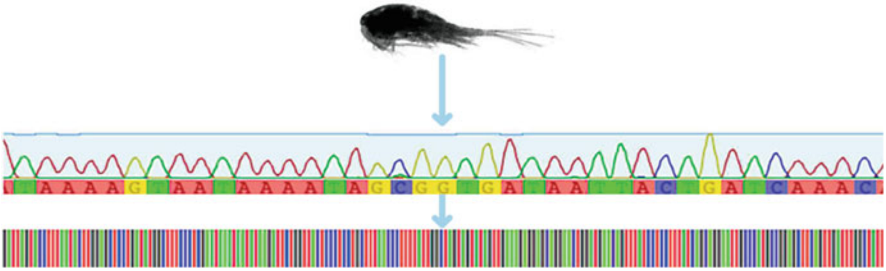

Fig. 12.1 A freshwater copepod (upper frame) is processed and a sequence obtained (middle frame) for a 658 nucleotide fragment of the cytochrome c oxidase subunit 1 (COI) gene—only 41 of the 658 nucleotides are shown for simplicity. This sequence then provides the "DNA barcode" (shown in illustrated form, lower frame) which can be added to the database and used to identify future individuals with similar sequences

To facilitate genetic identifications of species, it is first necessary to obtain reference specimens for the taxon of interest. Reference specimens can be obtained through field sampling or from museum collections. Ideally, reference specimens are then identified or identifications are confirmed by a recognised expert for the particular taxon. A small tissue sample (e.g. leg for invertebrates, or section of fin for fish) is removed from the specimen and processed for DNA sequencing for an appropriate, standardised gene region (Fig. 12.1). DNA sequences from such standardised gene regions are termed DNA barcodes. For the majority of eukaryotic animals, the most widely used genetic marker is the cytochrome c oxidase subunit 1 (CO1) gene (Hebert et al. 2003a, b). However, other markers are also used, occasionally in conjunction with CO1, and include the 16S, 18S and 28S ribosomal RNA gene or the cytochrome b (CytB) gene loci (e.g. Sevilla et al. 2007).

In order to maximise faunal coverage and characterise aquatic biodiversity worldwide, a coordinated, multinational approach has been required. As an overall coordinating body, the International Barcode of Life (iBOL; www.ibol.org) has facilitated the development of a global network and database. The initiative was focused initially on major biomes including freshwater habitats, and specific working groups were used to target selected taxa. For example, Working Group 1.7 (Freshwater Biosurveillance) targeted "environmental indicator" taxa including Odonata (damselflies and dragonflies), Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies). Other lake inhabitants such as fish and birds were covered by Working Group 1.1 (Vertebrate Life). International participation as part of the iBOL initiative has also been a critical element for the recognition of potentially invasive species moving among geographic areas (e.g. Armstrong and Ball 2005; Duggan et al. 2012).

The main repository for the DNA barcode data is the publically accessible Barcode of Life Datasystems (BOLD; Ratnasingham and Hebert 2007; www.boldsystems.org). Here, detailed information is provided for each specimen including photographs, collection data, identification to lowest taxonomic level possible, and location of the voucher specimen. For the DNA sequence (barcode) data, this

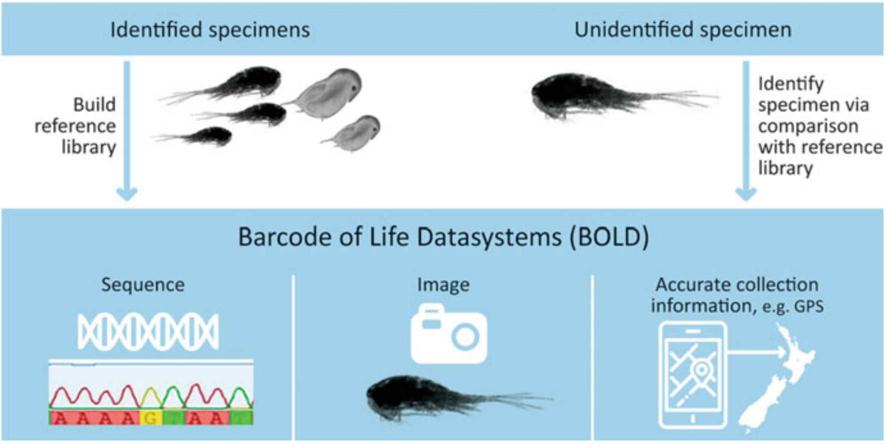

Fig. 12.2 Flow chart for building the Barcode of Life Datasystems (BOLD) reference database and for identifying an unknown specimen using the identification function on the BOLD database (www.boldsystems.org)

also includes copies of the "trace" files (an example of a trace file is shown in the middle frame of Fig. 12.1), which represents the raw data used to generate the actual DNA barcodes. Trace files are also used to verify the quality of the DNA sequence used to generate the barcode. A workflow diagram for using and adding data to BOLD database is provided in Fig. 12.2: BOLD also provides a range of bioinformatics tools that allow users to query, analyse and present their data. Data can be kept "private" (viewable only by specified individuals) such as during data collection and/or verification for publication. Following publication, all data are normally made fully open-access. There is currently no charge to access or add data to the BOLD database. Sequence data are also cross-referenced and available on the National Center for Biotechnology Information GenBank database (www.ncbi.nlm.nih.gov/genbank/). Full online support and instructions for using BOLD and/or obtaining a dedicated user account are provided on the website.

All records which are added to BOLD, and fulfil quality control standards (e.g. sequence length >500 nucleotides, compliance with metadata requirements), are assigned a Barcode Index Number (BIN) based on a clustering algorithm that connects it with individuals with similar sequences already on BOLD (Ratnasingham and Hebert 2013). This is particularly useful for taxa that are not yet attached to a morphological identification in the BOLD database or for specimens that correspond to currently undescribed or cryptic species. They can also be helpful for querying and resolving misidentifications within the database in cases where individuals having different taxonomic names/designations are found within the same BIN. The BIN designations can also provide an estimate of Operational Taxonomic Units (OTUs) and serve as a surrogate for "species" richness in the absence of more detailed morphological taxonomy.

For species not yet barcoded, the reference database can be continually updated as new specimens are obtained. When DNA sequences are analysed from species that are not yet in the BOLD database, an exact species-level identification will not be possible. However, comparison against international records will provide a match to the highest taxonomic level possible such as order or family. For any currently undescribed or cryptic species, the Barcode Index Number (BIN) will be assigned by BOLD and allow for similar sequences to be associated with each other. Any future specimens or formal identifications can then be attributed back to these initial specimens.

As of 2018, over five million sequences from approximately 600,000 species are currently available on the BOLD database. For New Zealand water bodies, over 5000 specimens and 4100 DNA barcodes have been added to the database representing over 500 species. This total now includes native and exotic fish, molluscs, rotifers, cladocerans, copepods, and amphipod crustaceans, as well as an extensive range of aquatic insects including mayflies, stoneflies, caddisflies, and chironomid midges. With the rapid growth of the BOLD database, these data can now be used for routine monitoring purposes as well as to enhance ecological studies through the rapid identification of taxa. They can also be used for a range of other applications including environmental monitoring of community assemblages ("metabarcoding") or for detecting species on the basis of their DNA being shed into the environment (environmental DNA or eDNA). These aspects are covered further in the following sections of the chapter. Figure 12.3 provides an outline of potential applications for the DNA reference library.

12.3 Community-level Analyses

12.3.1 Assessing Invertebrate Communities Using DNA Analyses

The application of Next Generation Sequencing (NGS) platforms to whole samples such as zooplankton tows or benthic cores has the potential to revolutionise the efficiency of molecular-based approaches. NGS platforms, such as the MiSeq 2000 (Illumina) and the Ion Torrent (Life Technologies), allow for the metabarcoding of DNA directly from environmental samples (Baird and Hajibabaei 2012; Lodge et al. 2012; Metzker 2010; Quail et al. 2012). It is, therefore, possible that an entire freshwater invertebrate community can be characterised directly from environmental samples. NGS techniques have already been applied to marine zooplankton community samples with some success (Lindeque et al. 2013; Machida et al. 2009).

NGS approaches allow multiple samples to be run simultaneously. This can include multiple individuals from a single sample/lake or, with appropriate "labels", multiple individuals from several samples or lakes. This approach removes the need for sorting individual animals from a sample. In essence, an entire sample can be

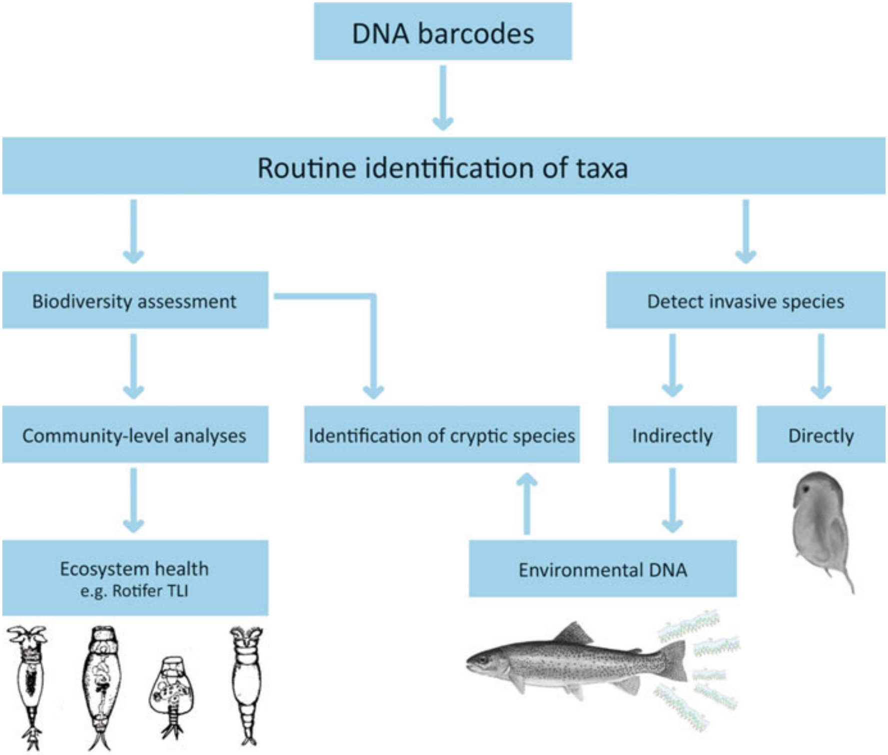

Fig. 12.3 Flow chart of possible applications based on a reference library of DNA barcodes such as that found on the Barcode of Life Datasystems (BOLD)

collected, homogenised, and then the collective DNA analysed to determine which species are present. For example, zooplankton or benthic samples can be collected using standard methods such as those outlined in Chapman et al. (2011). For zooplankton, collection would involve casting a fine mesh conical net from the shore and dragging it through the water using a rope. For benthos, D-nets could be used in shallow areas or airlift samplers for deeper areas. Excess water should be carefully drained from samples and replaced with 100% ethanol and refrigerated at 4 °C or below for best preservation. The use of formaldehyde or other preserving fluids (e.g. Kahles) will degrade DNA and should be avoided. Samples should also be kept out of direct sunlight as UV light will degrade DNA. Work to date has been undertaken on a range of taxa including river benthos (Hajibabaei et al. 2011; Dowle et al. 2016), nematodes (Porazinska et al. 2010), protists (Nolte et al. 2010), terrestrial insects (Yu et al. 2012), and freshwater zooplankton (S.M. Woods et al. unpubl. data).

At present, NGS approaches are limited by the length of sequence fragment they are able to analyse—currently around 400 nucleotides which is shorter than full-length (658 nucleotide) CO1 barcode sequences. This necessitates finding conserved (non-variable) regions within the barcode sequence that can be used to “trim” the sequences to a length <400 nucleotides for NGS sequencing. For aquatic insects or fish, this is relatively straightforward. However, for groups such as zooplankton, which include the rotifer phylum as well as the crustacean orders Copepoda, Cladocera, and Ostracoda, this is not possible. For samples containing such divergent taxa, the 28S rRNA gene has proven to be more effective in the interim. However, as 28S rRNA is a relatively conserved gene (has less variability than COI), it can also target other taxa such as phytoplankton which greatly outnumber zooplankton.

Another strategy is to first isolate the zooplankton DNA using commercially available products such as MYbaits (www.mycroarray.com). Mybaits consist of synthetic DNA probes, designed from reference sequences that are complementary to regions in the genomes of the target taxa. The complementary sequences enable the capture of targeted DNA regions. The captured DNA can then be sequenced using next generation sequencing without the need for universal primers as a PCR step is not required (Dowle et al. 2016). Further, many of the biases inherent to PCR-based approaches such as "swamping" of rare taxa by more abundant taxa are avoided (Dowle et al. 2016). However, designing effective capture probes is dependent on an adequate database of reference sequences for the organisms of interest (Dowle et al. 2016). With the rapid advances in NGS technologies, longer sequence reads will inevitably enable use of other more specific gene markers such as COI.

12.3.2 A Molecular Version of the Rotifer Trophic Level Index

New Zealand lake water quality is often assessed using physical characteristics such as dissolved nitrogen, pH and conductivity to produce a Trophic Level Index (TLI) of the lake (Burns et al. 1999). However, water quality indices based on physical parameters reflect the conditions at the time the sample was collected and may miss important, albeit brief, changes in water quality (e.g. during intense rainfall). Thus, it has been proposed that using key bioindicator species will give an integrated measure of water quality over time rather than the "snap-shot" obtained from physical measurements (Bongers 1990; Bianchi et al. 2003; Auckland Regional Council 2005). The composition of rotifer communities in a lake has been shown to vary with trophic status (Duggan 2007; Duggan et al. 2001a, b). One disadvantage of using indices based on bioindicator species is that characterising the community is often too slow and expensive for routine use and requires expert taxonomic skills to identify the species in a community. By integrating NGS techniques into routine monitoring programmes, the process of characterising freshwater zooplankton communities could become considerably more automated. Here, rotifer samples are collected as a bulk sample (e.g. plankton net), stored in ethanol at 4 °C or lower, and processed with almost the same effort as individual samples. DNA is extracted from samples, quantified, amplified, and cleaned using similar techniques. The

Fig. 12.4 Aerial view of the two experimental (*Impact* and *Control*) ponds at Hamilton Zoo. Pond margins have been outlined for clarity. The downstream (*Impact*) pond on the left-hand side of the figure contained the added carp

major difference is the addition of identification tags being applied to the samples for identification, along with primers specified for the NGS platform (e.g. MiSeq). The samples are then pooled and sent to an appropriate laboratory facility for NGS. The resulting sequences are then compared against a reference database to gain species-level identification.

12.3.3 An Example of a NGS Study Using Zooplankton

In 2012, a field experiment was undertaken to examine the effects common carp (*Cyprinus carpio*) on zooplankton community composition. For this study, two connected ponds (Control and Impact) located at the Hamilton Zoo (Fig. 12.4) were sampled monthly for 16 months prior to fish addition (Before) and for 14 months post-treatment (After) using a Before After Control and Impact (BACI) design (*sensu* Stewart-Oaten et al. 1986). A 28S rRNA barcode reference library for all zooplankton species recorded in the ponds had previously been assembled using at least three individuals from each morphologically recognised species to help assess any intra-species variation. All sequences were added to the BOLD database. Following the initial 16 months of monitoring, carp were added to the Impact (downstream) pond at a density of 300 kg ha^{-1} while the other pond served as the control. Whole zooplankton samples were taken monthly from both ponds using a

PVC pipe placed vertically in the water column. The contents were filtered through a 37 μm mesh net. Samples were then placed in plastic conical centrifuge tubes with 95% ethanol and kept on ice in the field. Environmental measurements taken in conjunction with zooplankton samples included nutrients, physical and chemical parameters, and suspended sediments. Following completion of the study, the pond containing carp was fully drained and all carp removed.

On return to the laboratory, samples were counted using traditional morphological techniques, until a minimum of 300 individuals had been counted. Ethanol was replaced with fresh 100% ethanol and samples were stored at −20 °C until genetic analyses. DNA was extracted by filtering the enumerated samples through nitrocellulose filters and using a PowerSoil (MOBIO, Carlsbad, USA) DNA Extraction kit. Extracted DNA was stored at −20 °C until needed for further processing. A polymerase chain reaction (PCR) was used to amplify a region of the 28S rRNA gene, for use in the MiSeq (Illumina) sequencer. The 28S rRNA gene was selected for genetic analyses owing to a high success rate with zooplankton (particularly rotifers) and the existence of an extensive reference library. Preliminary results revealed that phytoplankton taxa were also amplified, a disadvantage of using more conserved genes such as 28S rRNA. However, if required, there are further options to specifically target the zooplankton component and omit the phytoplankton data. For example, a series of small commercially prepared probes (MYbaits) can be used to select only the zooplankton DNA. The MYbaits use magnetic beads to attach to specific sequences (i.e. zooplankton found in the ponds), allowing exclusion of non-target DNA (i.e. phytoplankton). The samples are then prepared as normal for analyses using the Illumina MiSeq platform.

12.3.4 Analyses of Community-level Sequence Data

There are several computer software programs appropriate for metabarcoding analyses including: *QIIME* (Quantitative Insights Into Microbial Ecology; Caporaso et al. 2010) which combines many of the programs into a single user-friendly interface; Mothur (Schloss et al. 2009) which combines necessary algorithms into a script-based program; and *U-PARSE* which provides a more conservative approach to OTU clustering (Edgar 2013). A comprehensive list of programs, along with a detailed description of analytical steps required is provided in Bik et al. (2012). Ongoing comparative work focussing on the advantages and disadvantages of the various programs can provide further guidance (e.g. Cristescu 2015). Programs such as *Mothur* and *QIIME* have extensive literature associated with them, providing users with standard operating procedures as well as online forums. This allows users with only a basic knowledge of genetics (and bioinformatics), the ability to adequately analyse and extract relevant information from their data. Here, we provide a brief summary of the process required for such analyses.

Raw sequences first go through a general clean-up to remove primers, sample identifying markers and other secondary products (e.g. primer dimers, low quality and short sequences), and trimmed to provide high-quality sequences. The identities of OTUs are assigned through comparison with the reference library (e.g. BOLD). If the library is incomplete or limited reference material is available, sequences can be first matched against the reference library, and those failing to match are sorted using de novo algorithms. If similar sequences exist, databases such as BOLD or GenBank can be used to enhance the dataset. It is helpful to have some estimate of the intra- and inter-specific variation for target specimens and selected gene regions when defining species (or OTU) boundaries. For example, similarity levels >98% may be an appropriate level for designating OTUs for the COI gene, although 98% similarity is likely to merge distinct species (or OTUs) when applied to more conserved genes such as 28S rRNA. It is also essential to understand any limitations/assumptions of the algorithm being used and particularly how the OTUs are assigned. In the absence of a reference library, sequences may still be sorted entirely de novo. However, in these instances, it is possible that an overestimation of the number of taxa (or OTUs) will occur from unclassified sequences and chimeras (multiple genetic types resulting from a single individual).

Errors occurring during amplification include substitutions, insertions/deletions, and polymerase slippage. Use of Pfu DNA polymerase that has a "proof reading" capability can aid in reducing replication errors (Cline et al. 1996). If there is adequate coverage of targeted specimens in the reference library, these erroneous sequences can be detected and removed. However, even with a large reference dataset, the potential problem of chimeras will not be completely eliminated. PCR chimeras (*sensu* Haas et al. 2011) are sequences that appear valid, but are derived from incomplete amplification products acting as primers for other gene regions and thus will not match those occurring in the reference library. This means there is a potential to overestimate the species richness in such samples (Bik et al. 2012). Fortunately, new software to identify chimeras has emerged (e.g. Edgar et al. 2011; Quince et al. 2011) and should be used after the assigning OTUs. Relevant taxonomic designations can then be assigned to the OTUs and phylogenetic or taxon identification trees produced.

12.4 Assessing the Presence of Fish Using Environmental DNA

Traditionally, determining the presence of a species in a given habitat is achieved either through observation (e.g. spotlighting for fish) or through direct collection. This can be problematic for species which are difficult to observe or are found in low densities (e.g. recent introductions) and provide significant challenges for restoration efforts. However, species can also be detected within waterbodies by detecting DNA that they have shed into the environment, so-called eDNA. The term eDNA is now generally accepted to represent DNA obtained from environmental samples such as soil or sediment rather than directly from the target species (Ficetola et al. 2008;

Dejean et al. 2012; Thomsen and Willerslev 2015). The source of the DNA in samples is usually material such as sloughed skin or gut cells shed into an organism's environment. To date, the technique has shown promise for detecting and confirming the presence of invasive species (e.g. Jerde et al. 2011; Mahon et al. 2013), rare species, or endangered species (Biggs et al. 2015; Sigsgaard et al. 2015; Spear et al. 2015) and even the source of faecal contamination in waterways (Wood et al. 2013).

Environmental DNA has been used to detect species in a diverse range of environments and sample types such as soils (Hofreiter et al. 2003), ice cores (Willerslev et al. 2007), and faeces (Banks et al. 2009). Environmental DNA has also been used to infer the range of animals such as introduced silver carp (*Hypophthalmichthys molitrix*), bighead carp (*H. nobilis*) (Jerde et al. 2011) and koi carp (Takahara et al. 2015); six species of fish in the Yura River, Japan (Minamoto et al. 2012); weather loach in northern Europe (Sigsgaard et al. 2015); and the distribution of endangered salmonids in North America (Laramie et al. 2015). Other species that have been monitored using eDNA include frogs and salamanders (Goldberg et al. 2011; Spear et al. 2015), invasive amphibians (Ficetola et al. 2008) and aquatic plants (Scriver et al. 2015). More recent studies have characterised entire fish communities, rather than a single species, using the 12S rRNA gene (Civade et al. 2016; Evans et al. 2016; Hänfling et al. 2016; Olds et al. 2016; Shaw et al. 2016; Valentini et al. 2016). In addition, primers targeting the mitochondrial 16S rRNA gene of fish have been used either alone (Deagle et al. 2013), in combination with 12S (Shaw et al. 2016), or with 12S and cytb (Evans et al. 2016; Olds et al. 2016).

In New Zealand, studies have been conducted on the feasibility of using eDNA to detect brown trout, *Salmo trutta*, and koi carp, *Cyprinus carpio*. An eDNA assay for brown trout was developed to test feasibility for monitoring the success of an eradication programme. A second eDNA assay was developed for koi carp to assess the eradication of koi carp from lakes. These studies are outlined in more detail below.

12.4.1 Monitoring Brown Trout Eradication

Brown trout has been linked with the decline of many New Zealand native fish species either directly by predation or indirectly through competition for food (Townsend 1996; McDowall 2003; McIntosh et al. 2010). In the Zealandia Wildlife Sanctuary which is located in the North Island of New Zealand, near the city of Wellington (S41.2907°, E174.7531°), the aim is to restore the local prehuman flora and fauna to the regenerating shrub land and hardwood forest. One aspect of Zealandia's restoration is the removal of introduced fish such as brown trout from the sanctuary's waterways. The sanctuary consists of a 252 ha retired municipal water supply catchment with a series of small streams draining into an upper and lower reservoir formed by two dams. Measures to eradicate brown trout are dependent on detecting the presence of fish. However, brown trout can be difficult to detect visually as they tend to remain motionless and under cover (e.g. bankside vegetation) in rivers (Hicks and Watson 1985).

To assess the suitability of eDNA for detecting brown trout, water samples (2 L) were collected from stretches of streams known to contain trout and stretches where trout was known to be absent owing to physical barriers or ephemeral flow. Water samples were taken 1 day before and 85 and 164 days after an application of rotenone (a piscicide) to selected stream reaches known to contain trout. Presence of trout was established based on repeated electric fishing and/or spotlighting of each stream reach where stream flows were high enough to support trout. All water samples were filtered through glass fibre filters (pore size 1.6 μm). DNA was extracted from the filters using PowerWater kits (MoBio, Carlsbad, USA). Polymerase chain reactions were used to amplify a 148-nucleotide fragment of the mitochondrial d-loop region from brown trout with the primers STF and STR (Banks et al. 2016). Test results were then assessed by electrophoresis of the PCR products—the presence of visible bands on the electrophoresis gel was used as a positive indication of trout DNA within the sample.

Environmental DNA assays were positive for all samples below the trout barriers, and no amplicons were obtained from samples taken above the putative trout barriers for the pre-rotenone samples. Samples that produced a PCR product were sequenced using traditional (Sanger) sequencing and then subjected to a BLASTn search on the NCBI reference database (www.ncbi.nlm.nih.gov/BLAST/). All sequences from the positive assays matched to brown trout sequences in the reference database.

Water samples collected on 18 May 2011, 85 days after the rotenone application, did not produce any positive PCR amplicons and were, therefore, not sequenced. However, a positive test result was observed from water collected at one site in the true right tributary 164 days after the rotenone application. Negative and positive PCRs were then sequenced from the 164 days post-rotenone application. Brown trout sequences were obtained from three of the ten sample sites, one site in the true right tributary, and two sites in the true left tributary (one of which was treated with rotenone). Both of the other sites (i.e. from the true left and true right tributaries) had not received rotenone treatment. It was then determined that a trout had been captured in a non-treated area of the true left tributary 80 days after the rotenone application. However, no other trout was captured from the stretches of the streams that received rotenone.

The positive result at three sites, 164 days after the application of rotenone, suggests that trout DNA may have persisted either from trout that were not killed by rotenone or from DNA shed into the environment (e.g. from carcasses). Within the Zealandia Sanctuary, it was assumed that, because of the small size of streams, trout or their carcasses were thought to be unlikely to evade surveillance post-rotenone when all trout should have been removed. An alternative explanation is the possibility of a “false positive” resulting either from contamination during laboratory or field procedures or detection of incidental eDNA (e.g. from an animal or bird that had eaten brown trout from elsewhere).

12.4.2 Koi Carp

Removal of koi carp from lakes within the Waikato Region of New Zealand has been a key target for eradication efforts (e.g. Hicks et al. 2015). Accordingly, tests of an eDNA assay for koi carp, *Cyprinus carpio*, presence in lakes have also been undertaken. One litre water samples were collected from each of two sites at Lake Koromatua (S37.83721°, E175.2259°), Lake Puketirini (S37.5690°, E175.3470°), and Lake Kimihia (S37.5240°, E175.1920°). Extensive electrofishing surveys have reported the koi carp densities as Lake Kimihia 189.4 kg/ha in September 2012 and Lake Puketirini 55.1 kg/ha in February 2014 (Hicks et al. 2015).

Lake water (800 mL) was filtered through 2-mm^2 and 47-μm mesh filters with the aid of a peristaltic pump. The filtrate was centrifuged in a Sorvall RC26 Plus centrifuge (with Fiberlite rotor-FT0-6x500Y; Thermo Scientific) at 12,785 × *g* for 20 min at 4 °C. The supernatant was poured off until 5 mL remained. The concentrated material was re-suspended in the 5 mL and transferred to a 50-mL conical centrifuge tube. Material was pelleted by centrifugation at 4000 × *g* for 10 min at 4 °C and frozen in 1 mL of supernatant at −20 °C until the DNA could be extracted. To extract the DNA, 500 μL of the frozen material was thawed and then centrifuged at 10,000 × *g* for 4 min, and the supernatant was removed. DNA was extracted from the pellet using the MOBIO Power Soil DNA Isolation kit.

Polymerase chain reactions repeatedly copy target DNA template using a DNA polymerase enzyme to double the concentration of target DNA (dsDNA) at each amplification round. Traditionally, DNA amplification is verified by visualising DNA products on an agarose gel using dyes that bind DNA and fluoresce under various wavelengths of light (e.g. ultra violet), or in the case of quantitative PCR, visualised in real time by incorporating fluorescent dyes or hydrolysis probes that fluoresce in proportion to the amount of DNA amplified. If the target is not present, then there is no amplification of DNA and the test is negative.

A hydrolysis probe-based quantitative PCR (qPCR) assay was developed to target a portion of the cytochrome b oxidase gene (*cytb*) of koi carp using the primers JCB F TCCTATCTGCCGTACCATACA, JCB R GTAGGAAGTGGAATGCGAAGAA, and the probe 56 FAM/TGTTGCATT/ZEN/GTCTACTGAGAACCCACC/3IABkFQ. Test reactions comprised 12.5 μL of buffer, 0.2 mM dNTPs (Custom Science), 1.5 mM $MgCl_2$, 1X buffer, and 1.12 μg non-acetylated bovine serum (BSA; Sigma New Zealand), 0.5 U Platinum Taq DNA polymerase (Invitrogen, New Zealand), 50 nM of hydrolysis probe (Integrated DNA Technologies, USA), and 900 nM of each primer. Thermocycling conditions were 95 °C for 2 min followed by 50 cycles of 95 °C for 10 s and 68 °C for 40 s. Signal acquisition occurred at 510 nm after excitation at 470 nm at 68 °C with the gain adjusted to produce an initial signal of 10 fluorescence units. Standards for the qPCRs were generated from PCR product obtained from DNA extracted from muscle tissue of a koi carp. The primers JCB F and JCB R were used to amplify PCR product. Standards were quantified using a Qubit V.2 fluorometer (Invitrogen, Carlsbad, CA) and converted to gene copies per reaction from sequence data.

Koi carp were detected in both lakes known to contain koi carp. Koi carp DNA was detected in all 12 samples from Lake Kimihia (koi density 189.4 kg/ha) at 10–1000 gene copies per reaction. Signal was detected in 6 of 11 samples from Lake Puketrini (koi density = 55.1 kg/ha), at 110–1000 gene copies per reaction. The koi qPCR assay was also able to support the suggestion that no koi carp occur in Lake Koromatua as none of the 12 samples produced a signal using the qPCR assay.

12.4.3 Challenges of Using eDNA with Fish

The results from the eDNA assays illustrate some of the difficulties encountered in detecting species "sight unseen" (*sensu* Jerde et al. 2011). In particular, there are challenges in reconciling genetic results that conflict with physical results such as false positive results (a species is detected but in reality is absent) and false negative results (a species is present but not detected). Persistence of DNA in the environment from carcasses is thought to be minimal, as DNA has been shown to be detectable for only relatively short periods of time in the environment. For example, Dejean et al. (2011) found that DNA was detectable for less than 1 month after tadpoles were removed from aquaria, and sturgeon were removed from artificial ponds. Potentially, there is a risk of the transfer of trout DNA by birds such as shags (*Phalacrocorax* spp.) defecating in the water as prey DNA has been shown to be amplifiable from faeces (Murray et al. 2011) or by human interference. Specifically, trout eradication was opposed by some members of the community who had threatened to restock streams following rotenone treatment. An added complication is that, as the ratio of target DNA to other DNA decreases, false positives are more likely to be generated (Wilcox et al. 2013). Minimising false positive results requires good laboratory practice including running "blank" controls to detect systematic contamination of reagents with target DNA (Feature Box 12.1).

False negatives are similarly undesirable. False negatives are likely to be affected by a number of factors including sampling density, minimum detectable amounts of DNA and flow rates of streams. Detectability has been shown to be dependent on the method used to capture eDNA and the method used to extract the DNA from the captured material (Deiner et al. 2015). For example, Deiner et al. (2015) found that a bead beating method similar to the method we used for the Zealandia samples had a lower yield of amplifiable DNA compared with the Qiagen DNeasy that uses proteinase K and surfactants to break up cells and release the DNA. The false negative rate can be reduced by taking multiple samples from the same site especially when target DNA concentrations are low. Ficetola et al. (2015) suggest that at least six replicate samples from each site should be analysed routinely and if detection probability is very low, eight samples should be analysed.

Environmental DNA has also been used to study other communities in restored aquatic systems or those undergoing restoration (Feature Box 12.2). For example, McInerney et al. (2016) used eDNA obtained from Surber samples of Australian streams to assess the effect of invasive willows on a range of communities including algae, fungi, and flowering plants. In New Zealand, bacterial communities have been studied, and a Bacterial Community Index has been developed to assess changes in stream water quality (Lear and Lewis 2009).

The presence of faecal-indicator bacteria (FIB) have also been used as proxies for the presence of human pathogens in contaminated water (Griffin et al. 2001). Additionally, microbial source tracking (Scott et al. 2002) can be used to identify contaminant sources contributing to water quality degradation. Microbial source tracking uses DNA extracted from environmental samples such as water as the template for either endpoint PCR or QPCR (Scott et al. 2002). Through the use of primers that only amplify DNA from host specific bacteria, the source of the faecal contamination can be identified (Field and Samadpour 2007) and mitigation measures prioritised. Further details are provided in Wood et al. (2013).

Box 12.1 Sensitivity and Reliability of DNA Based Detection for Aquatic Restoration Applications

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Environmental DNA is increasingly being used as a method for detection of aquatic species, particularly those that are rare and sparsely distributed (e.g. Dejean et al. 2012; Goldberg et al. 2013; Moyer et al. 2014). Within relatively contained water bodies such as lakes, this method can provide a full community analysis as well as informing managers on the presence of recent incursions by invasive species (Rees et al. 2014). As with all survey methods, sensitivity and reliability are paramount if eDNA methods are to be successfully implemented. Sensitivity in this context is the probability of detecting the target DNA when the species of interest is known to be present. The trace nature of eDNA can lead to false positives (incorrectly detecting the target species when it is absent) or false negatives (failing to detect the target species when it is present), resulting in misinformed and misguided management (Yoccoz et al. 2001). Because the false negative rate is inversely proportional to the target species' abundance (Gu and Swihart 2004), reducing the false negative rate in low-density eDNA surveys is critically important, and this has been previously been acknowledged (Darling and Mahon 2011) but to date is rarely implemented as standard practice.

Environmental DNA detection is a multistep process involving the sample collection, isolation, extraction, and PCR amplification of DNA from an environmental sample. Two main sources of error can lead to a false negative in eDNA species-specific detection surveys: method error, or insufficient detection sensitivity. Method error may result from poor sampling, isolation, extraction or amplification procedures, or poor handling or preservation

(continued)

Box 12.1 (continued)

practices throughout these stages. Insufficient detection sensitivity can result from a failure to sample the target species' DNA despite it being present in the environment or failure to amplify the target species' DNA despite it being present in a PCR aliquot. Key factors contributing to the sensitivity of eDNA include the differences in concentration and dispersion of target DNA molecules among sites.

Methods are now available that address detection sensitivities and enable the estimation of the concentration and dispersion of target DNA at survey sites (Schmidt et al. 2013; Furlan et al. 2015). However, these methods won't overcome the issue of reliability, which is a direct result of method errors that have the potential to spuriously affect these sensitivity estimates. As such, amplification of a species-specific assay and/or an endogenous control assay will indicate successful methods throughout the entire eDNA process. Failure to amplify either assay means the success of laboratory procedures cannot be confirmed and could indicate errors resulting from poor sample handling, inappropriate laboratory methods and/or inhibition. These samples should be then be re-assayed with a fresh extraction and/or re-amplified with dilution to minimise the concentration of potential inhibitors (Wilson 1997). If both assays still fail to amplify, the sample should be discounted as a probable false negative. In order for eDNA to be implemented as a routine monitoring tool, it is essential that robust protocols are used and that these are quantifiable. Using the approach outlined here, there is now a species-specific European carp (*Cyprinus carpio*) assay available for eDNA surveys of Australian freshwater systems (Furlan and Gleeson 2017). This has been particularly important for the implementation of this survey method in the eradication program of carp from Lake Sorrell in Tasmania (Diggle et al. 2012).

Box 12.2 Environmental DNA Applications in the Laurentian Great Lakes: From Targeted Species Surveillance to Biodiversity Monitoring

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(continued)

Box 12.2 (continued)
Measuring and monitoring bio-diversity of freshwater fish species is challenging in any aquatic system, but made that much more difficult when the system represents a large expanse (>245,000 km^2) containing approximately 21% of the world's freshwater (EPA 2014). The Laurentian Great Lakes hold a diversity of fishes (e.g. Fig. 12.5), with complex food webs and life histories with ecosystem pressures from recreational and commercial fishing, interactions with established and new invasive species, climate change, and pollution. Since 2009, environmental DNA (eDNA) survey methods have been used in the Great Lakes for surveillance of Bighead (*Hypophthalmichthys nobilis*) and Silver (*H. molitrix*) Carp (Jerde et al. 2011, 2013), two invasive fishes poised to establish and potentially cause disruptions to ecosystem services (Wittmann et al. 2014). From one water sample, genetic assays can screen for each species and provide managers with greater insight into presence (Darling and Mahon 2011) and ultimately inform managers about ongoing incursion response efforts to protect a freshwater fishery valued at >4 billion US$ annually (NOAA 2014). But we can do so much more than targeted detection of a species from a water sample. Instead of asking, is the DNA of Bighead Carp present, high-throughput sequencing and bioinformatics enables the assessment of DNA from entire communities of species (Thomsen et al. 2012; Lodge et al. 2012). This approach has a number of practical advantages of which two aspects are critical to the Laurentian Great Lakes. A distinct advantage of eDNA is the ease of field implementation. It uses a water sample, avoiding such needs as specialised nets or fishing boats, trained biologists in taxonomic identification, complicated permits, and significant labour costs. An eDNA surveillance program can, as demonstrated in surveys of Great Lakes bait shops (Nathan et al. 2014), rapidly achieve broad geographical coverage with greater sensitivity than traditional approaches used by resource managers. Another advantage of the genomic approach to eDNA surveillance is detection of unexpected invaders (Mahon et al. 2014). Many of the invaders, now residing in the Great Lakes, were discovered only after establishment, when management options were limited and initial impacts were less noticeable (Mills et al. 1993). Launched in 2010, the Great Lakes Restoration Initiative (GLRI) has supported much of the eDNA research through funding to State and Federal agencies, non-government organisations, and academic institutions. This investment in eDNA studies and research has had the outcome of preventing the spread of invasive species and protecting native species of the Laurentian Great Lakes.

Fig. 12.5 Asian carp emerging from the Illinois River, which is connected to the Laurentian Great Lakes via the Chicago Sanitary and Ship Canal. Photo by W. L. Chadderton (The Nature Conservancy)

12.5 Future Prospects

Technology is changing rapidly, particularly within the NGS realm. Examples include the length of sequences that can be processed by the various sequencing platforms. For example, the Ion Torrent platform was originally able to read sequences of only 200 nucleotides in length. However, with minor changes in chemistry, sequence reads of 400 nucleotides are now possible and sequences exceeding 600 nucleotides will be possible in the very near future. Alternative technologies and approaches will also inevitably appear and replace existing methods. As with any technology, some of the current platforms may no longer be supported by the manufacturers and will become obsolete. However, in all cases, the fundamentals will remain the same. In particular, having access to properly archived sample material in national collections as well as comprehensive DNA sequence repositories such as BOLD or GenBank will underpin all future efforts employing molecular techniques in the monitoring of restoration efforts.

By integrating NGS techniques with molecular identification, the ability to detect or monitor native and exotic species in a range of habitats will become much more automated. For example, the use of NGS approaches will also shift the analyses of community-derived assessments of water quality from morphologically-based to DNA-based. This, in turn, will reduce the reliance on specialised taxonomists reducing costs and speeding up the routine assessment of water quality. These NGS developments and further refinements of eDNA methods will see a revolutionary change occurring in biological monitoring of freshwaters.

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Chapter 13
Automated High-frequency Monitoring and Research

Chris G. McBride and Kevin C. Rose

Abstract High-frequency sensor measurements provide new opportunities to better understand and manage water resources. Recent advances in sensor and information technologies have enabled autonomous measurement and analysis of the aquatic environment at ever-increasing spatial and temporal resolution. Here, we describe the fundamentals of automated high-frequency lake monitoring, including hardware and telemetry design, sensor types and measurement principles, maintenance requirements, and quality assurance/quality control of datasets. These aspects require careful consideration to collect data that are suitably robust for monitoring and research needs. Examples are provided of the value of high-frequency measurements and derived data products for analysing short- and long-term lake processes. When applied with rigor, automated sensor measurements can improve programmes of management, monitoring, and research by providing baseline data, enabling rapid response to disturbance events, reducing some long-term costs, and opening new windows of opportunity to better understand the present era of declining water quality and environmental change.

Keywords High-frequency monitoring · Water quality sensors · Fluorometry · Data management

13.1 Introduction

Advances in technology can be leveraged in numerous ways to generate new insights into processes and characteristics of ecosystems. Whereas the foundations of modern limnology were laid with manual labour—field observations and laboratory analyses—more recently, sensor and information technologies have enabled

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autonomous measurement and analysis of the aquatic environment at ever-increasing spatial and temporal resolution. These technologies can be used to 'fill the gaps' inherent in traditional sampling designs both in time and space, detect rapid environmental change, and analyse short-term processes that can have cascading impacts through time and space. Perhaps most importantly, these technologies open the door to investigations and analyses previously unthinkable, or at least impractical, using more traditional methods.

High-frequency in situ monitoring of lakes entails the application of water quality and meteorological sensors with autonomous data collection platforms for continuous measurement of sub-daily physical, chemical, and biological dynamics. Sensors and instrumentation are typically mounted to a floating platform with on-board power and data acquisition systems (Fig. 13.1), and data are often transmitted in near-real-time to a database with a user interface enabling querying and visualisation. Platforms are usually solar-powered, providing capability for deployments in remote locales and enabling rapid collection of vast datasets with relatively little labour. Hence, it may be tempting to consider this approach as a short cut to data collection or as a substitute for routine monitoring and research. However, high-frequency lake measurements are often complementary to many aspects of traditional monitoring and research and are most valuable within the context of a broader programme of data collection.

Water quality research and monitoring require highly dependable measurements. Myriad factors must be considered in order to collect robust, accurate, and continuous high-frequency data, as well as for its useful interpretation. This chapter seeks

Fig. 13.1 Monitoring buoy on Lake Waikaremoana (Image: M. Osborne)

to provide an overview of many important considerations when undertaking automated measurements of lake water quality using sensor technologies. Continuous, high-quality data can be critical to understanding lake dynamics and have proven a very valuable asset to programmes of lake management and restoration globally.

13.2 Why Monitor at High Frequency?

Autonomous monitoring and in situ sensor technologies enable more comprehensive understanding of lake processes through space and time and are a growth area in aquatic monitoring and research (Meinson et al. 2015). Continuous, near-real-time ('streaming') data provides a range of benefits over and above pure research. High-frequency data can facilitate adaptive and/or reactive management as in early warning detection of harmful algal blooms at recreational sites in lakes, selective off-take withdrawal from reservoirs, or prediction of impending regime shifts. For example, dissolved oxygen depletion in the hypolimnion of lakes during periods of thermal stratification is a key regulator and important indicator of lake 'health'. However, oxygen depletion in some conditions (e.g. polymixis) can be near-impossible to accurately characterise using a sampling protocol of infrequent (e.g. weekly to monthly) measurement. High-frequency monitoring enables a more thorough understanding of important processes by enabling accurate estimation of hypolimnetic oxygen depletion during stratification events often lasting a matter of just days.

Buoy data can provide a basis for hypothesis generation and context for more detailed or targeted experiments and investigations. A continuous record of environmental conditions and lake water quality is often very useful when conducting in-lake field experiments, for example, mesocosm studies or bioassays. Furthermore, the continuous nature of such monitoring records can ensure that 'ground-truth' data are always available, for example, during major storm events that can rapidly change water quality conditions or periods spanning discrete satellite observations of lake water quality. Importantly, long-term datasets of continuous measurements can be used in the calibration, validation, and refinement of water quality models. A continuous record of key state variables enables much better assessment of model performance than might be achieved with irregular and/or infrequent manual monitoring. Additionally, because some short-term processes have temporal legacies, high-frequency data can produce more accurate model forecasts by better representing transient conditions.

Streaming data can find important utility in education and public outreach if made available over the web. For example, anglers might utilise buoy data feeds to assess on-lake conditions for safety or to find the depth of optimum temperature for a temperature-selective target species (e.g. rainbow trout), and sailors may use wind speed and direction data to better chart their course. Publicly available data feeds can also be an effective tool for improving awareness of water quality issues as well as a valuable teaching resource. There are a range of approaches to sharing high-frequency sensor data. Increasingly, many organisations have policies whereby data are made publicly available to the maximum extent possible. It is generally encouraged to make data freely available to all those interested, notwithstanding some important limitations. Users often need to be made aware that near-real-time data is provided raw, without post-processing as necessary and often without important contextual metadata. This limits the immediate utility of the data, but provides for rapid dissemination. Frequently, multiple data releases are issued, with later versions providing calibrated, contextualised data.

In summary, buoy systems are flexible and adaptable to meet a broad range of monitoring and research needs. When data are collected and processed carefully, they can provide a comprehensive baseline water quality monitoring record and support a wide range of applications.

13.3 Types of Monitoring Platform

A typical high-frequency lake research and monitoring station consists of hardware including a buoy platform, data logger, solar power (batteries, panels and controller), communications, moorings, sensors, and/or multi-probes. By optimising the form and features of a buoy platform, an ideal system can be created to match specific needs, applications, and budget. Common buoy designs include both fixed depth and 'profiling' (variable depth) sensors. The specific design and configuration will depend on multiple factors, many of which are described herein. While we make an attempt to provide a discussion on many of the most critical design and configuration elements, there are several other resources available that complement this chapter and provide additional information on high frequency aquatic research and monitoring practices. Examples are the European Cooperation in Science and Technology (COST) Action NETLAKE (Networking Lake Observatories in Europe) series of 'toolbox' technical reports, which are available online (netlake.org).

13.3.1 Fixed Depth Sensor Buoys

Water quality sensors are suspended in the water column at fixed depths. Most commonly, sensors are hung from a surface-mounted buoy and thus sensor depth is relative to surface. Figure 13.1 shows such a fixed-sensor buoy deployed in Lake Waikaremoana, New Zealand. Occasionally, sensors are suspended from a sediment-mounted platform by a cable connected to a buoyant object and, thus, sensor depth is relative to the lake bottom (Fig. 13.2). For either configuration, the most common sensors include a temperature 'string' of multiple water temperature sensors located at varying depths on a single cable and arrays of additional sensors near-surface, near-bottom, and sometimes at specific depths in between. Although this is the most common type of sensor system, the cost of sensor replication and

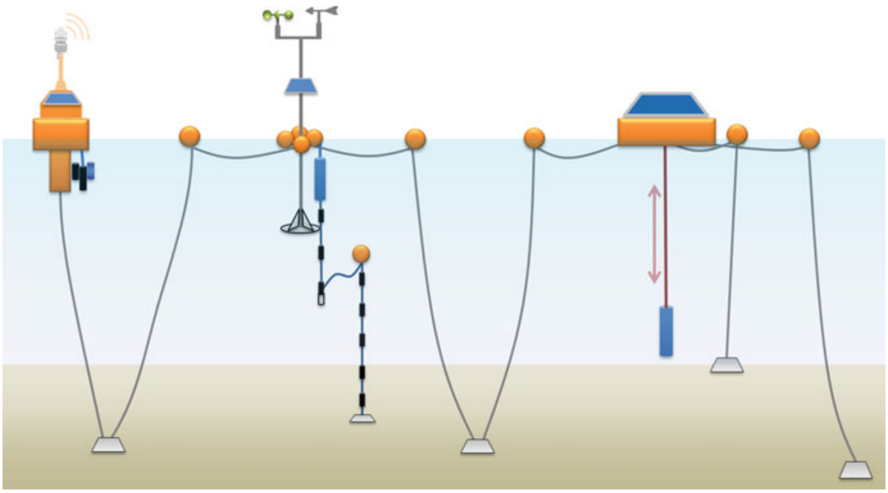

Fig. 13.2 Schematic diagram of three alternative buoy and mooring configurations. (Left) Single point mooring with surface buoy, water quality sensors, and a compact weather station. (Middle) Vertical 'spar' buoy with two-point mooring, multi-probe (sonde), dual bottom- and surface-referenced temperature sensor string, and component weather station. (Right) 'Profiling' system on three-point mooring with arrows indicating vertical control of a sonde

burden of sensor maintenance (including cross-calibration of replicated sensors) mean that measuring multiple variables at even moderate vertical resolution is often impractical. Accordingly, most variables are measured at a single near-surface depth (commonly about 1 m), assumed to be representative of the surface mixed layer. This design is sufficient to characterise overall water quality in well-mixed systems where temperature stratification is of lesser importance, but is often inadequate for capturing vertical gradients in chemical and biological processes. For example, in lakes that exhibit strong seasonal stratification, sensors deployed near the surface do not readily respond to processes occurring nearby or below the thermocline (e.g. deep chlorophyll maxima). However, because temperature readings are usually collected throughout the water column and simultaneously, these systems are ideal for measuring physical processes in lakes, for example, heat flux, internal waves, and seiches.

13.3.2 Profiling Sensor Buoys

A water quality sensor package is attached to an automated winch and is raised and lowered through the water column on a pre-determined (user-adjustable) sampling routine (e.g. 12 profiles per day measuring at 1 m vertical resolution through a 20-m water column). These systems enable detailed vertical measurement resolution whilst eliminating the need for replication and cross-calibration of sensors at multiple depths. Relative to fixed depth sensor systems, control equipment is more complicated and has moving parts and must, therefore, be well-designed and robust, with error-handling capabilities and a programme of regular maintenance. Depending on the depth of the water column and the necessary sensor equilibration time at each sample depth, profiling sensor buoys may only return to a specific depth every hour or longer, whereas single depth sensors can record at frequencies in the order of seconds to minutes. However, because profilers can sample all variables throughout the water column, they are much more powerful than fixed depth sensor systems for understanding chemical and biological processes both vertically and temporally, for example, bottom water oxygen consumption, pH variability, and estimated phytoplankton distribution.

13.3.3 Towed or Autonomous Probes

Although outside the scope of this chapter, many of the sensor technologies discussed within are also used on towed or autonomous underwater vehicles. While data buoys are generally used to provide a continuous record of water quality at a single geo-coordinate, towed and autonomous probes are most often used to investigate vertical and horizontal variation in water quality during short-term surveys (see Feature Box 13.1). Towed platforms complement buoys and can provide better characterisation of spatial variability in water quality, at the expense of temporal coverage. In contrast, fixed and profiling platforms characterise water quality through time at a point or depth gradient. Surveys using towed probes can be useful to ground-truth data for remote sensing studies.

Box 13.1 Remote Sensing Techniques for Lake Assessments: The Lake Biwa Example

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Phytoplankton blooms associated with eutrophication can be detected at the lake water surface with satellite or airborne sensors, and spatial distributions can, therefore, be monitored remotely. High concentrations of suspended sediments can also be readily detected from space (Fig. 13.3). On the other hand, lake water constituents can sometimes change dramatically as a result of interactions between the water column and the bottom sediments. These deep-water processes are essentially invisible to airborne detection systems. In order to overcome this difficulty, we have applied advanced in situ sensing

(continued)

Box 13.1 (continued)
technologies including autonomous underwater vehicles (AUV), remotely operated vehicles (ROV), side scan sonar (SSS), and multi-beam sonar (MBS). These optical or acoustic sensing technologies enable observations of changes in deep-water conditions. Such approaches were first developed for ocean monitoring, but they are highly applicable to large deep lakes and gaining popularity. One example is the deep water exploration and monitoring of Lake Biwa, the largest and oldest lake in Japan. Hydrothermal vents were first discovered in 2009 using a high definition video camera mounted on the AUV 'Tantan'. Since then, vent activities have been observed continuously by AUV and it has been found that the number of vents is increasing. Gas bubbles from bottom sediments were also detected by ROV, SSS, and MBS.

Figure 13.3 shows the stereoscopic view of gas bubbles taken by MBS around the submerged mountain in Lake Biwa. The dark red colour indicates the shallower area (around 20 m depth) and the dark blue colour corresponds to the deepest area (around 100 m depth). The horizontal and vertical scales are about 1.5 km. Several series of gas bubbles can be clearly seen around the mountain, and the positions of those gas bubbles were able to be compared with the vent positions observed by AUV. In large deep lakes like Lake Biwa, it is not easy to take samples at a specific position within a small area. The target area has to be narrowed down and defined as accurately as possible. Acoustic remote sensing provides a powerful approach to identify key locations. On the other hand, optical remote sensing in water is sensitive to sediment disturbance and turbidity, which impairs the ability to obtain clear images. The AUV 'Tantan' is, therefore, maintained at a depth that is always 1 m above the bottom sediments in Lake Biwa, and transects are run at that precise depth over distances of 1–2 km. The best approach to investigate the bottom waters and sediments of lakes is to apply a combination of methods based on optical and acoustic sensing techniques. Each of these techniques has strengths and limitations, but the array of information allows much greater understanding that complements the highly resolved surface-water information obtained from satellites and airborne sensors.

13.4 System Design

For in situ buoy applications, a large disc or square float typically serves as the physical platform, providing buoyancy and stability for a waterproof housing of data logging, solar power production, and communications equipment. Platforms must be of appropriate size and buoyancy to support the necessary sensor, control, and power equipment and stable enough to collect accurate meteorological data (if required). Deep-set ballast weight can help to improve platform stability. Vertical 'spar' buoys are an excellent design to cope with strong winds and large waves, because buoys

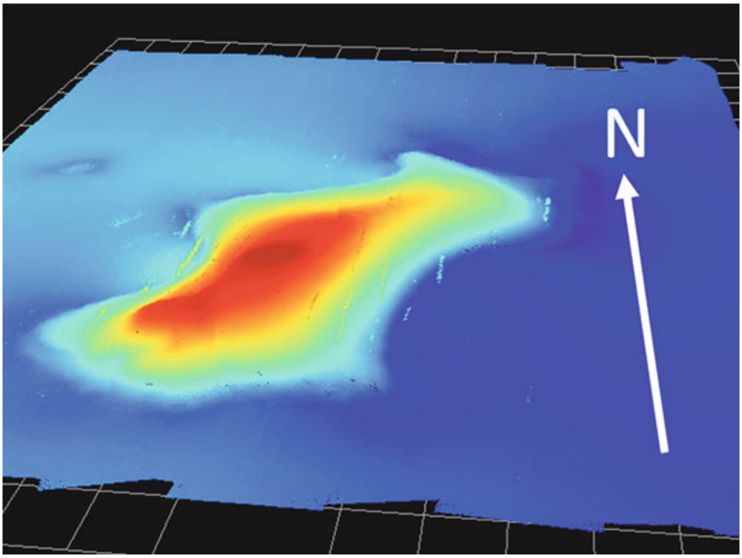

Fig. 13.3 The stereoscopic image of bottom and gas bubbles in Lake Biwa measured by MBS. Dark red colour indicates shallow area (20 m depth) and dark blue colour indicates deep area (100 m depth)

with larger bases will tip to angles concurrent with wave motion unless the platform is bigger than the wave period.

Floatation platforms should be solid (e.g. ionomer foam) or foam-filled to prevent sinking if damaged (e.g. boat strike), able to be easily accessed and maintained using available vessels, and resilient to all environmental conditions. For example, if vandalism of equipment is a concern, then exposed wires and visible sensors might be undesirable; however, sensors should also be easily retrievable for necessary calibrations and maintenance. Care should be taken to ensure that sensitive electronics are effectively sealed against water ingress, particularly around points of cable entry. Multiple barriers to ingress (i.e. double enclosures) can increase reliability. There are a number of manufacturers of buoys, and platforms can often be purchased without any sensors or other hardware, providing users with the ability to construct fully customised packages.

13.4.1 Mooring

Buoys must be properly moored to ensure they remain securely located and sensors remain undamaged. Buoys are usually deployed at or near the deepest location of the lake to ensure that measurements can be made throughout the water column and are representative of whole lake characteristics. Moorings should be resistant to abrasion and corrosion (mating of dissimilar metals in aquatic environments is generally to be avoided due to the potential for galvanic corrosion) and should help absorb the impact of wave and wind action whilst also allowing enough 'travel' to cope with water level variation. Where surface level changes dramatically, it can be challenging to achieve a mooring design that can adjust between levels without leaving too much slack at low levels such that potential damage to hanging sensors could be caused by drifting to undesirable locations or entanglement of sensor and mooring lines. In shallow lakes with variable water level, a fixed pole or platform can be an effective alternative to a moored floating platform.

Floating buoys can be moored with single, two-, or three-point mooring depending on the location, depth, and types of sensors used (Fig. 13.2). Single point moorings are used when buoy-based sensors are deployed on or immediately below the buoy flotation and do not hang far into the water column. Surface sensors are protected and less vulnerable to damage caused by subsurface debris, high currents, and entanglement from anchor lines. Sensors hung deeper (e.g. temperature strings) can become damaged as the buoy spins around the central axis of a single mooring line. Single point moorings are uncommon because many buoy systems include sensors deep in the water column, and because a single mooring may result in loss of the whole system should the mooring line become damaged.

Two-point moorings are commonly used when sensors are deployed in the water column below the buoy. The mooring lines are often held taut and away from the buoy by two smaller surface or subsurface floats which are connected to parent mooring lines that run to the lakebed and connect via a bottom chain to anchors. These surface-mounted buoys pull double duty by reducing the likelihood that passing boaters will pull up directly alongside, potentially damaging the main buoy. This design allows sensors to hang freely in the water column beneath the buoy and be retrieved independently of mooring lines.

Three-point moorings are commonly used in deep systems which can experience high winds and large waves. Reinforced connection points and routine checks can be necessary in exposed locations. For example, a single season of large waves and high winds is sufficient to cause wear through a 10-mm diameter stainless steel D-shackle connection. Meteorological sensors can benefit from three-point mooring to reduce listing and turning of the buoy, which can affect wind speed and direction measurements (although some wind sensors have an internal compass to correct for buoy rotation). Three-point moorings also provide additional security, reducing the risk of excessive drifting and sensor entanglement should one mooring line become damaged.

13.4.2 Ice Cover

Many lakes experience seasonal ice cover. Ice presents serious challenges and threats to deployed buoys. Frequently, buoys are removed just before predicted ice-in and re-deployed after the last ice melts. This avoids potential damage to the buoy and sensors, but limits the water quality monitoring duration. There are several other solutions to maintaining high-frequency sampling in lakes that form ice. One option is to redeploy sensors below ice cover using a subsurface mooring buoy below the maximum predicted ice cover depth. In this approach, sensors are deployed relative to the bottom rather than relative to the surface. Another option is to leave the buoy deployed but remove mooring lines, which can get twisted and broken if left in place. This approach risks damage to the buoy structure. Furthermore, the buoy can drift with the ice and in the spring and may move great distances. If the monitoring buoy is removed from the lake during the ice cover periods, management of the existing moorings must be considered. It can be very difficult to pull up mooring lines that have buried deeply into sediment. An approach is to again use subsurface buoys below the depth of maximum ice cover and retrieve the mooring lines after ice-off using divers.

13.4.3 Visibility

Local regulations often require that sensor platforms be clearly marked and visible both night and day, particularly where powered craft operate. Relevant authorities should be consulted to gain consent for the installation, decide on location, and determine the correct colouring of the platform as well as requirements for lighting and flash patterns. Solar-powered marine LED navigation beacons are a convenient way to mark buoy installations. It is often worthwhile (or required) to install placards at nearby boat ramps explaining the design of the platform and its purpose, as well as marking the buoy itself with contact information. These can be great tools to engage with the public about what is being measured, how data are being used, and how those interested might be able to access the data.

13.4.4 Power Requirements

Before designing or choosing a system, it is important to consider the cumulative power consumption of chosen sensors and communications, as well as seasonal patterns in power production. A detailed power audit may be needed to determine necessary power generation and reserve supply, including an understanding of solar collection efficiency and local climate variability (sunlight hours). Too much power is usually better than not enough; however, multiple high-capacity sealed lead-acid batteries and large solar panels can quickly make for a cumbersome platform. Because of their heavy weight, batteries are often stored deep within a buoy to act as extra ballast. Most commonly used batteries (typically sealed lead-acid) ‘off-gas’ hydrogen and should not be charged within hermetically sealed enclosures.

13.4.5 Circuit Protection

Proper grounding of sensors and equipment is essential to ensure proper operation and to prevent damage from lightning-induced voltage surges or electrostatic discharge. Grounding and circuit protection are complex subjects, and a detailed description is beyond scope here. Nevertheless, the general principal of lightning and surge protection is to provide a lowest-resistance path to ground which bypasses sensitive components of the system. Many pieces of equipment commonly used in data buoys (data loggers, solar regulators, etc.,) have some degree of in-built circuit protection, be it resettable fuses or gas discharge tubes. Proper grounding of all equipment is necessary to ensure the functionality of in-built protection, usually requiring connection to a 'ground plane' which in terrestrial systems is commonly provided by a copper rod driven into the earth. In the aquatic environment, achieving an adequate connection to ground can be more challenging, particularly where conductivity is low (i.e. in most lakes). Additional measures are usually required, and this is an area of system design where careful consultation of equipment manufacturer's recommendations and advice or assistance from qualified personnel is strongly advised.

13.4.6 Data Collection (Logging and Telemetry)

Cellular, radio, or satellite telemetry enable near-real-time transmission of water quality data over the lifetime of a monitoring buoy. Cellular modems require a network subscription and adequate wireless signal at the lake's surface, bearing in mind that data can usually be somewhat reliably transmitted well below the threshold of reception needed for a tolerable voice call. Radio systems require a radio base established within range of the transmitter and, where feasible, can be an attractive alternative to the ongoing cost of cellular communications subscriptions. In the rare instance that neither cellular nor radio signals are sufficient, a satellite modem can be used to remotely access data, although this can be an expensive option. Alternatively, buoys can log data internally to then be retrieved at user-specified intervals. Due to the low cost of large-capacity memory chips, buoy visits can often be in the order of months without running out of available memory.

Data loggers are commonly used to store and transmit data from sensors as well as control sampling frequency and power supply. Data loggers are generally highly robust and programmed with relative ease. Important considerations include compatibility with preferred options for telemetry and sensor communications. For example, some data loggers will support a limited number of serial devices (e.g. RS232, RS485) and so may not be an ideal choice where multiple sensors are required and use such communications. Many systems allow the data logger to be controlled remotely, for example, allowing for adaptive sampling by modulation of sensor sampling frequency, from anywhere in the world.

13.4.7 Open-Source, Low Cost Hardware

Commercial data loggers are perhaps the most common ‘brain’ for automated water quality monitoring. However, open-source microcontrollers and miniature computers are low-cost and flexible alternatives that are gaining increasing use with sensor technologies. Platforms such as Raspberry Pi (www.raspberrypi.org), Arduino (www.arduino.cc), and BeagleBoard (www.beagleboard.org) provide the opportunity to replace data loggers with low-cost open technology that can be in many ways more analytically powerful than traditional data loggers. For example, these technologies can be used to push analytics onto the sensor platform itself thereby enabling a buoy to perform its own diagnostics and some degree of quality assurance and control. This approach can be used to assess sensor stability and the likelihood of unwanted sensor drift, connect sensors to cue one another, or otherwise develop an ‘internet of things’ approach to environmental sampling. Nevertheless, in many cases, care and expertise is required in order to collect research quality data using certain technologies. For example, collecting precise and accurate readings from analog sensors can require additional hardware if the microcontroller does not natively include an appropriate precision reference voltage against which to measure the signal from the sensor.

13.4.8 Software and Data Access

During set up, users will generally define intervals for sensor measurement and data transmission. A typical system might log data from sensors every 15 min with an integration time of 30 s and transmit data to a central server every hour or every day. Limitations on logging and communications frequency are often at the discretion of the operator; however, power consumption during logging and communications may sometimes limit very high frequencies of logging and transmittance. Some sensors require a relatively long equilibration period to adjust to water quality gradients. For example, some optical dissolved oxygen sensors can take over 5 min to equilibrate to accurately measure a drastic change in dissolved oxygen concentration. It should be noted that many dissolved oxygen sensor manufacturers claim a short equilibration period (e.g. <30 s), but often quote accuracy to 60 or 90% over this period. Further accuracy (e.g. to 99%) can take several additional minutes.

13.4.9 Alarms and Notifications

A key component of buoy-based monitoring and research is the ability for end users to respond rapidly to sensor readings that exceed pre-defined thresholds. For example, dredgers using turbidity sensors may have to respond quickly to high suspension levels to reduce sediment loads and deterioration of downstream water quality. Most databases can be configured to send automated alarm notifications via SMS or e-mail that provide immediate alerts when sensor readings exceed pre-defined thresholds. This technology can also facilitate adaptive sampling of water quality. Abnormally, high or low values may signal sensor failure, calibration drift, or biofouling; thus, buoy operators can be automatically notified of the need to check sensor function and fidelity. As described above, microcontrollers and miniature computers also increase the capability of sensors to cue other measurements. For example, stream and lake sensors can be set to sample more frequently based on meteorological sensors detecting passing storm fronts, or sensors can be cued in order to sample based on a running-averaged-mean or deviation from a particular distribution.

13.5 Sensors

There are literally thousands of possible sensors for buoy-based applications. Sensors are usually classified either by parameter type (e.g. physical, chemical, or biological sensors) or by the technology or measurement principle that they use (e.g. electrochemical, optical). The measurement principle employed has implications for sensor cost, measurement stability, susceptibility to fouling, calibration, and maintenance requirements. Sensor technologies are constantly advancing. One such example is optical dissolved oxygen sensors, which have largely replaced electrochemical sensors in common use despite a somewhat higher cost, due to increased long-term stability and lower maintenance and calibration requirements.

13.5.1 Sensor Communications

Sensors use a variety of methods to transmit measurements to data collection platforms (e.g. data loggers). It is worthwhile remembering that most sensors are measuring some sort of electrical signal, which must then be amplified and/or interpreted into a meaningful value. Such interpretation can be performed within the sensor, by the data logger, at the database, or during post-processing. Accurate record keeping of data transformations, scaling factors, and unit conversions is an essential component of metadata collection.

Analog sensors output a raw electrical signal, which may be a single-ended voltage (e.g. 0–5 V), differential voltage (e.g. ±250 mV), or current loop (e.g. 4–20 mA).

Digital sensors generally contain an on-board analog to digital converter (ADC) and may or may not also have on-board data logging. These sensors output digital information using a variety of protocols such as RS232 or RS485. Another popular serial data standard, SDI-12, allows connection of multiple sensors using three wires (power supply, ground, and data) and communicates with sensors individually via allocated sensor addresses.

13.5.2 Meteorological Sensors

Understanding the response of lake processes in response meteorological changes is an essential component of lake research and monitoring. Therefore, many lake buoys include meteorological sensors. Typical meteorology measurements include wind speed, wind direction, precipitation (multiple forms, e.g. rain, hail and snow), barometric pressure, air temperature, and relative humidity. Compact weather stations such as the Vaisala WXT520 or Lufft WS-series instruments are popular choices. These multi-sensors provide several key variables in a single package and may contain no moving parts due to the use of ultrasonic anemometers (wind speed sensors). For higher demands with regard to sensor accuracy, component meteorological sensors can also be used. Regardless of the technology employed, platform stability, overall height, and deterrence of birds (usually by installation of spikes) are important considerations.

13.5.3 Temperature Sensors

Water temperature is one of the most commonly reported lake measurements because it is extremely important to a number of aquatic processes and is relatively inexpensive and easy to measure. Fisheries management, hydroelectric plants, selective withdrawal dams, and numerous aspects of aquatic and sediment research often depend on having water temperature data from throughout the water column. Temperature in many lakes is measured continuously using strings of temperature sensors that span the water column, with sensors located at fixed depths. Commercially available temperature sensors range greatly in accuracy (c. $\pm 0.5\,^{\circ}\mathrm{C}$ to $\pm 0.001\,^{\circ}\mathrm{C}$) and corresponding cost (a few cents to hundreds of dollars per sensor). Temperature sensors are commonly put close to one another near the surface or epilimnion of the lake (often up to every 0.25 m), so that high-spatial-resolution temperature data is collected in this zone, where temperatures can vary over short spatial and temporal scales. Below the metalimnion, sensors may be relatively sparsely located because temperature gradients are smaller and more predictable deeper in the water column.

13.5.4 Light Sensors

Light and water transparency are two fundamental properties of aquatic ecosystems. Transparency regulates a number of in-lake processes, including the maximum depth of photosynthesis and oxygen production as well as changes in temperature throughout the water column. Above surface atmospheric sensors can measure incident (downward) light or reflected light (albedo) by pointing a sensor at the lake surface. Below water, light sensors can be used to measure transparency. Underwater measurements typically include two or more sensors deployed at different depths. The difference in light between the sensors is used to estimate the diffuse attenuation coefficient for light. Underwater light sensors are typically located on an arm away from the buoy to reduce or eliminate potential shading effects from the buoy above, and regular or automated cleaning of light sensors is essential. Many different sensors exist depending on the wavelengths of interest. Examples include visible light (400–750 nm), PAR (photosynthetically active radiation, 400–700 nm), ultraviolet light UVA and UVB (280–400 nm), infrared (700–3000 nm), total short-wave radiation, total long-wave radiation, and total global radiation. PAR sensors are the most commonly used light sensors underwater because PAR wavelengths are used by photosynthesising algae.

13.5.5 Other Sensors

Table 13.1 summarises a range of common sensors and measurement technologies. When designing a system and choosing sensors, it is worth spending time carefully evaluating research and monitoring questions, and how this relates to requirements for sensor accuracy, available resources for maintenance, and the need for spatial sensor replication. These aspects strongly influence the total cost of the system, and cost may ultimately determine the final configuration. All sensor-based measurements must be collected and interpreted carefully in order to draw robust conclusions from data (see case study on fluorescence, below).

13.5.6 Multi-probes (Sondes)

A sonde (French for probe) is used to house, protect, and connect many underwater sensors within a single, integrated package. Sondes provide common power and communications for a range of sensors and usually store sensor calibration information. Typically, sondes have on-board power and data logging but can also be easily configured for communications with external power and data loggers. Sondes can be expensive compared to component sensors, but the convenience of a common interface saves much time and effort in cabling, calibration, and system

Table 13.1 Overview of common measurement variables, sensor technologies, advantages/disadvantages and example manufacturers and instruments

Variable	Units	Sensor type	Description	Pros	Cons	E.g. *manufacturer*: model
Physical						
Water temperature	°C	NTC thermistor	Temperature-dependent resistor with negative temperature coefficient	• Fast response time with calibration • Long-term stability • Accurate and precise	• Non-linear calibration • Commercial options can be relatively expensive	*PME*: T-Chain
		IC temperature sensor	Integrated circuit with analog or digital output	• Pre-processed output (e.g. linear analog, digital) • Inexpensive	• Lower accuracy from factory calibration	*Dallas Instruments*: DS18B20
Photosynthetically active radiation (PAR)	μE m^{-2} s^{-1}	Silicon photodetector	Silicon photodetector with optical bandpass filter (400–700 nm) and diffuser	• Inexpensive • Multiple sensors can be used to estimate light attenuation (K_d)	• Requires careful mounting underwater. Sensitive to shading and rotation	*Li-Cor*: LI192 quantum sensor
Turbidity	FTU, FNU	Nephelometric, non-ratio	Light source and photodetector at 90° to one another	• Accurate at low turbidity	• Non-linearity at high turbidity	*Turner Designs*: cyclops C7 'T'
	FNRU	Nephelometric, ratio	Light source and 90° primary photodetector with additional detectors at other angles	• Improved accuracy at very low turbidity • Reduced interference from colour	• Non-linearity at high turbidity	*Hydrolab*: 4-beam GLI-2
	NTU, BU	Backscatter	Light source and photodetector at <90° to one another	• Wide measurement range • Improved accuracy at high turbidity	• Less accurate at low turbidity	*Campbell Scientific*: OBS-3+

Transmission	%, m^{-1}	Beam transmission and attenuation	Reduction in light intensity over a fixed path length	• Accounts for both scattering and absorption	• Can be limited in dynamic range	*Wet Labs*: C-Star
Water level	m	Echo sounder	Time delay between emission and return of sound pulse determines water column depth	• Buoy mounted	• Interference (e.g. sensors, moorings)	*Airmar*: DT800
		Pressure sensor (absolute)	Hydrostatic pressure on sensing diaphragm	• No vent tube required	• Must be bottom-mounted	*INW*: PT12
		Pressure sensor (differential)	Hydrostatic and atmospheric pressure on sensing diaphragms	• Inherent atmospheric compensation	• Must be bottom mounted • Vent tube to atmosphere required	In-situ: level TROLL 500
Chemical						
Dissolved oxygen	% sat, mg L^{-1}	Polarographic	Silver anode and noble metal cathode in a potassium chloride solution	• Lower cost	• Consumptive, requires stirring • Warm up period required • Measurement drift	*Sea-Bird Electronics*: SBE43
		Polarographic pulsed	Polarographic but with third (silver) electrode to enable pulsing of measurement	• Reduces need for solution stirring	• Warm up period required • Measurement drift	*YSI*: 6562
		Galvanic	Anode and cathode of dissimilar metals (zinc or lead anode, silver or nickel cathode)	• Possible to achieve fast response time	• Shorter lifetime • Sensitive, e.g. to H_2S	*AMT*: galvanic micro-sensor
		Optical: intensity	Quenching of fluorescent response by dye to blue light excitation, dependent on dissolved oxygen concentration	• More stable than electrode sensors	• Slow response time • Susceptible to photobleaching • Moderate cost	*InSite IG*: Model 10
		Optical: lifetime luminescence	Lifetime (phase shift) of quenching response, dependent	• Better long-term stability than fluorescence intensity	• Slow response time • Moderate cost	*ONSET*: HOBO DO data logger

(continued)

Table 13.1 (continued)

Variable	Units	Sensor type	Description	Pros	Cons	E.g. *manufacturer*: model
			on dissolved oxygen concentration	• Insensitive to interference by ambient light		
		Optical: fast response	Lifetime luminescence but without permeable light barrier	• Very fast response time	• Higher cost	*JFE Advantek*: RINKO series
Dissolved carbon dioxide		Non-dispersive infrared	Diffusion of gas through an oil resistant matrix interface, concentration-dependent absorption of IR electromagnetic radiation	• Relatively low cost and maintenance	• Slow response time (minutes)	*Pro-Oceanus*: Mini-Pro CO2
Methane	g m^{-3}	Spectroscopy	E.g. cavity enhanced absorption	• Measure all GHGs simultaneously • Accuracy, sensitivity	• Cost, size • Requires surface/pumped chamber	*LGR*: GHG Analyzer
		Tunable diode laser absorption spectroscopy	Membrane diffusion with concentration-dependent laser light intensity	• Accuracy, sensitivity • Long-term stability	• Cost, size	*Contros*: HydroC CH_4
		Non-dispersive infrared	Diffusion of gas from liquid through an oil-resistant matrix interface	• Relatively low cost and maintenance	• Slow response time (minutes)	*Pro-Oceanus*: Mini-Pro Methane
Conductivity	μS cm^{-1}	Contacting: 2, 3 or 4 electrode	Conductivity-dependent current produced when voltage is applied to immersed electrodes	• Low-cost • Accurate	• Cell constant must be appropriate to intended range of measurement • Sensitive to fouling and corrosion	*Ponsel*: digisens-4-electrode

		Inductive	AC drive coil voltage induces current in a receive coil, proportional to the conductivity of the solution	• More robust than electrode sensors • Insensitive to fouling and interference by suspended solids	• Less accurate at low conductivity • Relatively higher cost	*Stevens Greenspan*: EC300
pH	Unitless	Combination electrode	Electric potential created between a glass electrode and reference electrode is a function of pH	• Lower cost	• Can't be stored dry • Require regular cleaning and calibration	*Ponsel*: Digisens pH/ORP
		ISFET (solid state)	Silicon chip in contact with solution measures voltage potential between its surface and underlying semiconductor	• Robust (no glass), can be stored dry • Easily cleaned • On-board temperature compensation	• Often greater sensor drift than for glass bulb sensors • Higher cost	*Campbell Scientific*: CS526L
Oxidation/reduction potential	mV	Combination electrode	Voltage between inert metal electrode and reference electrode	• Often shares reference electrode with pH sensor	• Drift	*INW*: TempHion pH/ORP
Nitrate	g m^{-3}	Ultraviolet spectrometry (optical)	Spectral absorption c. 200–400 nm. Stable UV light source, fibre optic sensing probe, and precision spectrometer. Nitrate determined by least squares fitting algorithm	• Long-term stability • Real-time nitrate measurements	• Very high cost	*Satlantic*: SUNA v2
Phosphate, ammonium, nitrate	g m^{-3}	Ion-selective electrode	As for pH, but with membrane and reference electrode specific to other ions	• Low cost	• Poor sensitivity • Drift	*INW*: TempHion ISE
		Automated wet chemistry	Multi-beam fibre optic colorimeter with silicon detector and fluorimetric measurement. Automated wet chemistry with reagent reservoirs	• Laboratory-style analysis, in-situ • Sensitive and accurate	• Labour intensive • High cost • Consumables (reagents)	*Systea*: WIZ probe

(continued)

Table 13.1 (continued)

Variable	Units	Sensor type	Description	Pros	Cons	E.g. *manufacturer*: model
Dissolved organic matter	fDOM	Fluorometry	Fluorescence using LED excitation and optical detection at wavelengths targeted for dissolved organic matter	• Long-term stability • Moderate cost	• Temperature-dependence	*Wet Labs*: ECO FL CDOM
Biological						
Pigment fluorescence	RFU	Fluorometry	Fluorescence using LED excitation and optical detection at wavelengths targeted for chlorophyll and or accessory pigments including phycocyanin	• Long-term stability • Moderate cost	See case study	See Table 13.2

configuration. Popular manufacturers include YSI, Eureka Environmental, Hydrolab, and In-Situ. Sensors typically integrated by sondes include pressure (depth), temperature, conductivity (salinity), pH/ORP, dissolved oxygen, turbidity, chlorophyll, and cyanobacteria (phycocyanin/phycoerythrin).

13.5.7 Acoustic Sensors

Although infrequently deployed directly on buoys, acoustic sensors are an important class of water quality sensors. Acoustic sensors are occasionally deployed concurrently with buoys to understand a wide range of lake characteristics. Acoustic Doppler Current Profilers (ADCPs) are most commonly deployed on lake bottoms with upward facing sensors, but can also be surface mounted. The sensors can identify the location and movement of particles by using the Doppler effect of sound waves scattered back from particles in the water column. The frequency of the emitting sensor dictates in part the particle size measured. The sensors can be used to study characteristics such as flow and circulation, waves, gas flux, bottom tracking (depth), zooplankton movement (e.g. diel vertical migration), and fish movement.

13.6 Photosynthetic Pigment Fluorometers: Measurement Principle, Pitfalls, and Interpretation

Phytoplankton biomass is usually quantified by either manual counting with biovolume measurements or estimated by determining sample concentration of extracted photosynthetic pigment (chlorophyll *a*). All phytoplankton contain chlorophyll *a*, and many also contain accessory pigments which harvest light energy for transfer to the photosynthetic reaction centre. Photosynthetic pigments are ‘fluorescent’ compounds. Fluorescence is the absorption of energy as light with instantaneous emission of light at longer wavelengths (lower energy), and each fluorescent compound has signature peak absorbance and emission wavelengths. Laboratory measurement of chlorophyll *a* requires extraction of pigment from the cells by acetone/ethanol/methanol digestion, followed by in vitro measurement of the fluorescence intensity of the extracted pigment using a benchtop fluorometer. By contrast, in vivo fluorometers measure the fluorescent response of living phytoplankton cells and as such can be used for high-frequency unattended monitoring and integration with real-time data collection systems. In vivo fluorescence is typically less precise and more error-and interference-prone than extracted in vitro fluorescence; therefore, instrumentation and data need to be managed meticulously in order to maximise the utility of these measurements.

13.6.1 Measurement Principle

The primary pigments analysed for freshwater phytoplankton are chlorophyll and phycocyanin. Chlorophyll pigments can be measured separately (e.g. *a*, *b*, *c*) or as total chlorophyll. Each pigment has slightly different excitation and emission spectra. Chlorophyll *a* absorbs light predominantly from the blue wavelengths (maximum excitation at 440 nm) and has a fluorescence peak at 675–685 nm (Lee et al. 1995; Gregor and Maršálek 2005). Phycocyanin is a dominant accessory pigment in freshwater cyanobacteria (the analogue in marine cyanobacteria is phycoerythrin) and absorbs light in the orange and red wavelengths (550–630 nm) with maximum excitation at 620 nm and emission at around 650–660 nm (Fig. 13.4) (Lee et al. 1995; Gregor and Maršálek 2005; Bastien et al. 2011).

In vivo fluorometers typically use an LED light source to excite a 'sensing volume' of water in front of the optical window and measure emitted fluorescence via detection filter located at a 90° angle to the excitation source. Single pigment instruments utilise excitation and detection wavelengths specific to the pigment being measured. Different manufacturers use slightly varying excitation and emission wavelengths, and this should be considered when comparing instruments or using different sensors at different sites (Table 13.2). The different excitation and emission wavelengths of the various manufacturers are in part controlled by the optical quality of the glass and filters used in the sensors. High quality materials

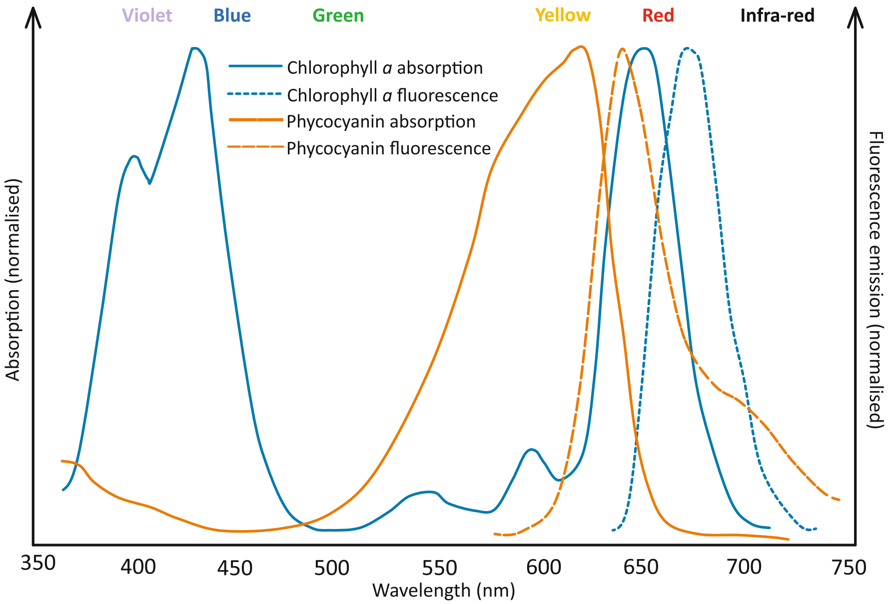

Fig. 13.4 Normalised absorption and emission (fluorescence) spectra of chlorophyll *a* and phycocyanin, the photosynthetic pigments most often measured in vivo for fluorometric sensing of phytoplankton. Spectra adapted from Taiz and Zeiger (2010)

Table 13.2 Excitation and detection wavelengths for several commonly used chlorophyll and phycocyanin fluorometers

		Chlorophyll		Phycocyanin	
Manufacturer	Instrument	Excitation (nm)	Detection (nm)	Excitation (nm)	Detection (nm)
Turner designs	Cyclops-7	460	696/44	595	≥630
Yellow Springs Instruments (YSI)	6600 series	470	650–700	595	650
	EXO series[a]	470 ± 15	685 ± 20	590 ± 15	685 ± 20
Chelsea technologies	TriLux[a]	470	685	610	685
Trios GmBH	Micro-Flu	470	685	620	655
Seapoint Ltd	Chlorophyll	470 ± 15	685 ± 15	Na	Na
Wet Labs	ECO series	470	695	630	680
BBE	Fluoroprobe[a]	450	680	610	680

[a]Combination sensor with multiple excitation/single detection. BBE probe has three additional excitation wavelengths not given above

typically have a smaller wavelength range of excitation and emission but are also more expensive. Narrow bandwidths are required to separate different types of chlorophyll, hence why many sensors measure ‘total chlorophyll’. ‘Combination’ fluorometers have multiple excitation LEDs but use a single detection filter. This method operates on the principle that light energy captured by accessory pigments is rapidly transferred to chlorophyll *a* where it is used for photosynthesis. As such these instruments aim to measure the proportion of pigments in the phytoplankton sample. An example of this approach is the BBE FluoroProbe which utilises six excitation wavelengths (370, 470, 525, 570, 590, and 610 nm) and detects the fluorescent response of chlorophyll *a* at 680 nm.

Fluorometers amplify detected fluorescent light energy and convert to output either as analog (typically 0–5 V) or digital (serial) signals. Uncalibrated sensor output is typically reported as ‘relative fluorescence units’ (RFU), or if the zero value for the instrument has been set using deionised water, measurements may be reported as ‘raw fluorescence units blanked’ (RFUB). Raw measurements can provide insight into short- to medium-term phytoplankton dynamics. However, the relationship between phytoplankton biomass and fluorescence is complex and can be influenced variously by species assemblage, sample concentration, ambient light interference, non-photochemical quenching, bio-fouling, and temperature. Non-photochemical quenching in particular can have large effects on the accuracy of chlorophyll fluorescence sensors and, therefore, many sensors provide poor characterisations of chlorophyll under bright light conditions. Interpretation of fluorescence data should carefully consider these multiple influences.

13.6.2 Gain, Signal Linearity, and Quenching

Most in vivo fluorometers have adjustable gain in order to provide appropriate measurement resolution across a broad range of phytoplankton concentrations. If fixed, the sensor range needs to be sufficient to cover all conditions through time and, if it is to be controlled dynamically, it may be desirable to perform calibrations across multiple ranges or record range for each measurement for analysis during post-processing because measurement ranges provided by manufacturers for different gain settings on their instruments are often nominal only.

High absorbance in the water column can 'quench' fluorescence and result in erroneously low values. At high chlorophyll concentrations, fluorescence can be quenched via absorption of emitted light by the phytoplankton themselves. This means that detected fluorescence at very high concentrations may actually be *lower* than at moderate concentrations (Fig. 13.5). High absorbance by dissolved organic matter can also quench fluorescence. To account for this, some pigment sensors have fluorescence dissolved organic matter sensors (e.g. BBE FluoroProbe). Quenching can be avoided by dilution of highly concentrated samples, although this approach may be impractical for continuous and/or remote measurement. Thresholds for quenching may be determined by analysing a dilution series; however, the relationship between quenching and phytoplankton concentration may differ with phytoplankton physiology (e.g. filamentous vs. colonial). High turbidity from non-algal sources can also adversely affect signal linearity, by either increasing detected fluorescence at low phytoplankton abundances due to increased scattering or decreasing detected fluorescence at high phytoplankton abundance due to increased absorption.

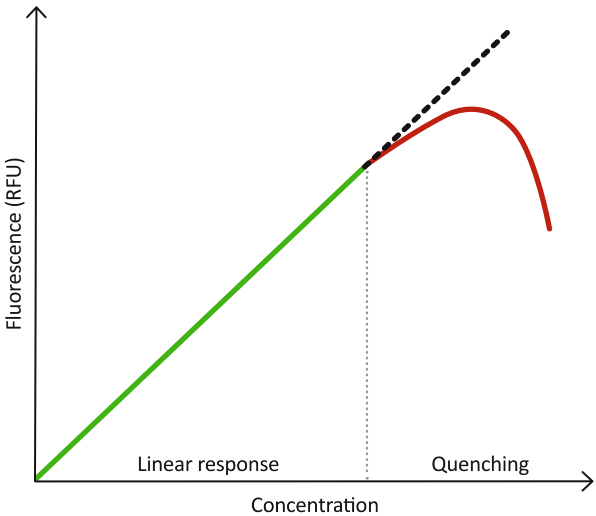

Fig. 13.5 Linear and 'quenching' regions of a generalised fluorometer response to increasing sample concentration

13.6.3 Ambient Light Interference

In some cases, ambient light can interfere with the detection optics of fluorometers, compromising data quality during brightly lit daylight hours. Advanced fluorescence sensors minimise this source of error by modulation of the LED light source, and measuring continuously whereby differences in fluorescence when the excitation LED is on and off are used to remove ambient light effects. Sensitivity to ambient light varies between manufacturers and models; therefore, it can be important to understand the instrument used, the light climate of the water, and any protection or shading offered by the buoy platform itself, in order to determine if any additional measures are needed to combat the effect of ambient light, e.g. shade caps or flow-through cells.

13.6.4 Temperature Effects

Fluorescence measurements are inherently temperature dependent. Algal fluorescence decreases as temperature increases, and a commonly used equation for temperature correction of chlorophyll fluorescence is

$$Fr = Fs^{[n(Ts-Tr)]}$$

where n is a temperature coefficient, Tr is the reference temperature, Fr is the calculated fluorescence reading at the reference temperature, and Fs is the observed fluorescence reading of the sample at the time of reading the sample temperature, Ts.

A common temperature coefficient for in vivo chlorophyll fluorescence is 1.4 (as recommended by Turner Designs Ltd).

13.6.5 Non-photochemical Quenching

Non-photochemical quenching (NPQ) is a mechanism used by phytoplankton to prevent damage to photochemical reaction centres during exposure to high intensity light. Absorption of excess light stimulates NPQ whereby excess energy is emitted mostly as heat, thus reducing emission of energy as photons (fluorescence). In practice, NPQ results in lower observed chlorophyll fluorescence than would be observed from identical biomass in low-light conditions. This photo-protective response can occur rapidly and is substantial; it can reduce estimated chlorophyll concentrations by nearly an order of magnitude (Fig. 13.6). Therefore, care must be taken when interpreting daytime fluorescence measurements particularly where sensor readings are being related to extractive chlorophyll measurements from grab samples collected during daytime for the purpose of calibration (Fig. 13.7).

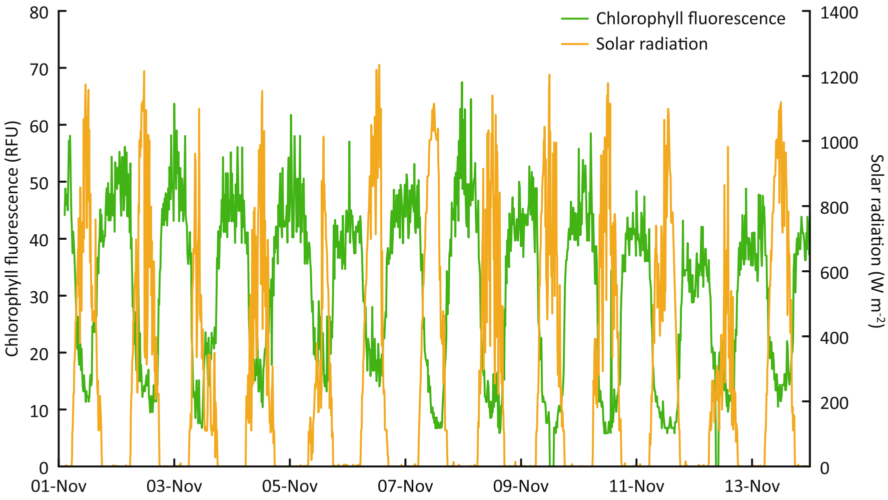

Fig. 13.6 Chlorophyll fluorescence (1 m depth) and short-wave radiation at Lake Rotorua. Fluorescence readings (green) demonstrate the effect on measured fluorescence of non-photochemical quenching (NPQ) in response to solar radiation (yellow)

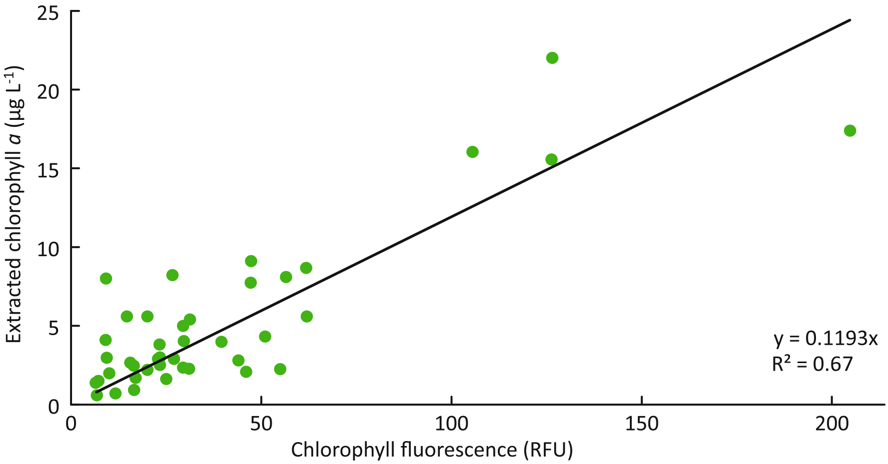

Fig. 13.7 Relationship between extracted chlorophyll and corresponding daily average pre-dawn-post-dusk chlorophyll fluorescence measurements, from Lake Tutira New Zealand, 2009–2012

Modelling and adjusting for the effects of NPQ can be difficult because it can vary day-to-day due to drivers including mixing depth, mixing rate, surface irradiance, light attenuation coefficient, as well as algal species, cell geometry, and cell nutrient status. A common approach is to disregard chlorophyll fluorescence measurements from dawn to slightly after dusk and focus on night-time measurements. While quick and easy, this approach has the disadvantage of removing a substantial portion of the data and preventing analyses of sub-daily dynamics (e.g. effects of diurnal stratification).

13.6.6 Fluorometer Calibration and Fluorescence Versus Biomass

When deploying in vivo fluorometers for long-term continuous measurement, a constant relationship between raw measurements and chlorophyll *a* concentration should not be assumed, due to multiple influences including temporal and spatial variation in species composition, light exposure, and sample temperature, as described above. Therefore, it is good practice to record measurements in raw units in the first instance, whereby raw fluorescence can be converted to chlorophyll or biomass equivalents during post-processing, if required. In situ sensors may also be sensitive to integration time and particle size effects. For example, if a sensor only operates for a 1-s integration time every occasion it turns on, and during its ‘on’ period a large colonial cell is in front of the optics, it may register erroneously high fluorescence. One way to minimise this effect is to use as large an integration time as the sensor and power system can enable, in order to average-out sources of variability.

Chlorophyll fluorescence often relates well to measurements of extracted chlorophyll *a* (Fig. 13.7); however, where species assemblages are complex and dynamic, and particularly where cyanobacteria are frequently dominant, fluorescence for multiple pigments is desirable, along with regular extractive measurements and species enumeration with biovolume estimates spanning seasonal and inter-annual variation. This represents considerable effort, and, as such, fluorescence can be a quick and easy method of measuring relative change in phytoplankton communities. However, due to the limitations described above, chlorophyll fluorescence sensors are often best considered as a tool to complement rather than replace traditional, lower-frequency monitoring strategies. In some cases, such as when buoyant cyanobacteria are present, there can be a substantial vertical gradient in chlorophyll and cell counts. Even profiling buoys, which better represent dynamics through the water column than fixed depth sensors, often cannot characterise cyanobacterial surface scums which can vary over spatial scales of millimetres.

13.6.7 Methods of Fluorometer Calibration

13.6.7.1 Pigment Laboratory Standard

Fluorometers typically come with a factory calibration and default dynamic range(s), often based on a primary standard of extracted pigment (e.g. phycocyanin pigment from prozyme diluted in deionised water, Turner Designs Cyclops C-7 phycocyanin). User-performed calibrations with primary standards are not always practical. For example, extracted chlorophyll must be suspended in acetone and, therefore, requires a specialised vessel or coupling to avoid damaging the optical window of the instrument, and phycocyanin standards can be expensive to procure and degrade quickly. It should be noted that chlorophyll in vivo may fluoresce differently to an equivalent concentration of extracted chlorophyll (e.g. due to variation within a colony of algal cells), and so calibrations based on extracted suspensions may not provide a precise conversion of in situ fluorescence to chlorophyll extracted from grab samples.

13.6.7.2 Secondary Synthetic Standard

Secondary synthetic standards are used to quantify long-term stability of field instrumentation, rather than to calibrate sensors in order to accurately estimate chlorophyll *a* concentrations. Standards can be a solid or liquid fluorescent material used to provide repeatable measurements in order to evaluate the consistency of instrument output. Some manufacturers supply standards made from a disc of solid fluorescent material mounted in a sleeve to ensure consistent orientation (e.g. Turner Designs' solid secondary standard range), whereas others recommend the use of a solution of fluorescent dye (frequently Rhodamine). Dye solutions are able to be easily reproduced, but are also subject to dilution error and batch variability. Solid materials are less prone to user error but are difficult to replicate exactly if the original is damaged or lost. Earp et al. (2011) provide an excellent review of multiple dyes and solid materials for chlorophyll fluorometers, recommending Plexiglass Satinice® 'Plum' for a solid standard and fluorescein dye for a liquid standard. When using a liquid standard, it is advisable to purchase a single batch of dye sufficient to perform multiple standard measurements, and all secondary standards should be kept in the dark to reduce degradation.

13.6.7.3 Cell Culture Dilution Series

The linear range of a sensor, as well as the relationship between biovolume and fluorescence, can be determined by lab experiments measuring a dilution series of cultured cells. The complexity of the relationship between phytoplankton biomass and fluorescence is highlighted by variability in the fluorescent responses of similar biovolumes among different species (Fig. 13.8). Therefore, a site-representative mixed assemblage may be the most appropriate method to relate fluorescence to biovolume.

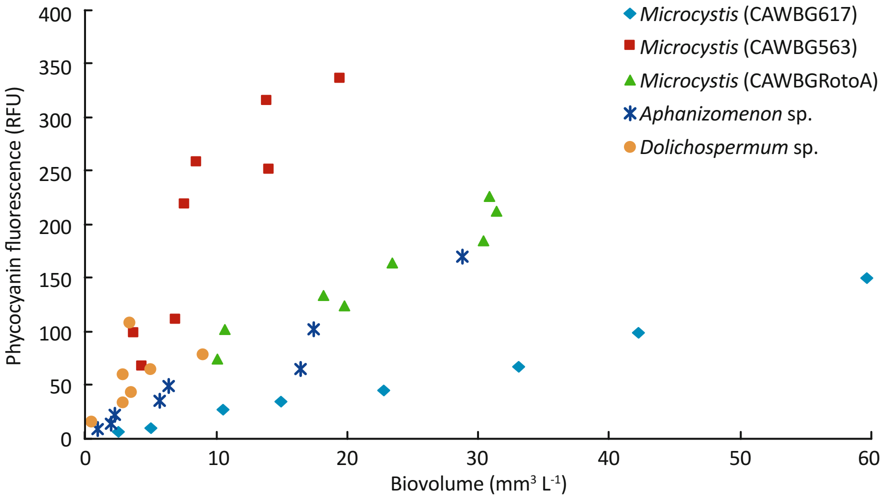

Fig. 13.8 Relationship between raw phycocyanin fluorescence and biovolume for five phytoplankton species, using a Chelsea Instruments 'TriLux' fluorometer. Data from Hodges (2016)

13.6.7.4 Grab Samples

A useful means of calibration is to relate sensor measurements through time with corresponding quantitative measurements of water samples collected at lower frequencies (Fig. 13.7). Relationships must be carefully constructed and not assumed to be universal across lakes or highly consistent through time due to the combined influences of species succession, quenching, temperature-dependence, ambient light interference, and NPQ described above.

13.7 System Maintenance and Calibration

Regardless of the platform and sensors used, periodic maintenance and calibration are required to ensure quality data collection. Maintenance intervals are largely dependent on sensor specifications, site location, water quality conditions, and other variables. Common maintenance intervals depend on water conditions, for example, waterways that are highly productive with high phytoplankton biomass are more susceptible to fouling and degradation of measurement quality (Fig. 13.9). Each sensor usually has a recommended calibration frequency; consult instrument-specific literature for more details.

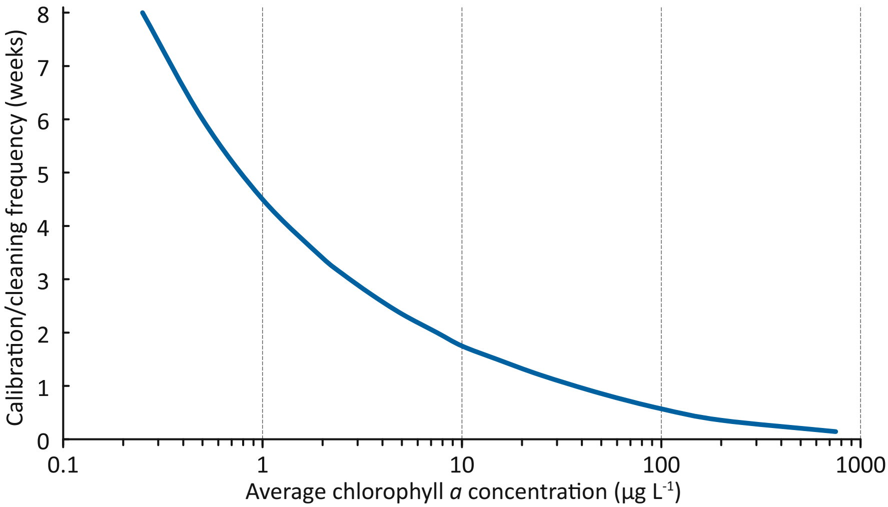

Fig. 13.9 Conceptual diagram of recommended cleaning frequency for near-surface-mounted sensors

13.7.1 Anti-fouling Technology

A common problem for in situ measurements of any kind, especially during extended sensor deployments, is fouling. Either biological (active) or non-biological (passive) fouling can occur. Active fouling refers to the growth of plants and animals over an instrument's measurement surfaces, whereas passive fouling results from substances such as silt, clay, and organic residue accumulation. Fouling can have substantial effects on sensor readings. For example, for many optical sensors such as dissolved oxygen, chlorophyll, CDOM, or turbidity sensors, fouling can block the passage of light from the source beam and to the detector. Biological fouling can increase measurements of dissolved oxygen and fluorescence as attached plants grow near the sensor window. In the interest of long-term deployments, particularly those in high-fouling waters, many in situ sensors now possess anti-fouling equipment to increase the accuracy and minimise the frequency of sensor maintenance. Many sensors have the option of in-built mechanical wiping devices that sweep fouling agents from the sensor surfaces prior to measurement (Fig. 13.10). For sensors that do not have the option of a mechanical wiper, an after-market third party wiper can sometimes be installed on the sensor. Other anti-fouling approaches include copper surfaces such as a ring around the sensor face, blasting of compressed air, ultrasonic disruption, automated pumping of biocides, and anti-fouling paint.

Fig. 13.10 An Apogee Instruments Quantum (PAR) sensor mounted in a Zebra-Tech 'Hydro-Wiper' automatic sensor wiper after 2 years deployed in oligotrophic Lake Waikaremoana, New Zealand. Although the wiper body and arm (pointing to 5 o'clock) were heavily grown over, the sensor's optical surface (centre) was clean and reading accurately (Image: M. Osborne)

13.8 Data: Quality Assurance and Quality Control

Automated monitoring platforms are capable of collecting vast quantities of data in short time. Quality assurance and quality control of sensor data are vital and can be challenging.

13.8.1 Data Collection: Quality Assurance

Assuring data quality begins before sensors are deployed. Sensors should ideally be calibrated according to manufacturer's recommendations and in consideration of the environment in which they will be submersed. Calibration consistency is important. For example, measurement temperature and/or atmospheric pressure (elevation) might affect calibration values for a host of sensors, including pH, dissolved oxygen, and fluorescence. Sensor performance should be carefully evaluated, being mindful of the age and lifespan of sensors, for instance, a three-point calibration may be more useful than two-point calibration, if sensor linearity is a concern. Sometimes, it is preferable to log raw values and apply calibration equations during post-processing of data. Regardless of the approach, accurate record keeping is always vital, including sensor calibrations, observed bio-fouling and timing of manual sensor cleaning, sensor gain changes or other wiring modifications, and any other factors which may later help explain sudden changes or unexpected values when processing data.

13.8.2 Data Processing: Quality Control

A single platform measuring 20 variables every 5 min will produce more than half a million individual observations per year. Because of the nature of sensor measurements, it is unlikely that such a dataset will proceed to analysis or publication without the need for error detection, correction, interpolation, and/or data transformations. Even meticulously collected (good quality assurance) datasets require thorough examination before graduating from raw streaming data to research, outreach, or other disseminated products. Erroneous data are sometimes glaringly obvious within the parent dataset. Often, automated protocols can be implemented to identify and remove these data. However, many sensor errors are more subtle and less easily detected by coarse 'filters'. There are various methods to quality control high-frequency datasets and most often a combined approach of automated and manual quality control is adopted.

Many data collection systems have capability for a range of automatic data processing, where observations must meet a set of criteria in order to avoid being quarantined or flagged for later investigation. Commercial time-series management platforms, including Kisters' 'WISKI' and Aquatic Informatics 'Aquarius', provide a range of native and customisable tools for detection of outliers or suspicious measurements. Organisations such as the USA's National Ecological Observatory Network (NEON) have invested extensively in establishing a range of tools and protocols for automated quality control of sensor network data (Taylor and Loescher 2013). Criteria by which data may be subjected to automated filters include unit checks, upper and lower limits, maximum expected rate of change, and missing or repeated (stalled) measurements. Once identified, erroneous measurements are not always straightforward to correct. For example, suppose one identifies negative light measurement values in a dataset. Clearly, these are erroneous. However, should they be removed or adjusted, and if they are adjusted, should all light measurements be similarly adjusted (i.e. are the negative values isolated incidents, or do they indicate a consistent negative bias for all measurements)? These questions are not trivial to answer as they can impact data patterns and results.

Ultimately, while automated quality control tools are a useful or necessary first step, many datasets benefit greatly from visual interrogation and expert user input. There is no substitute for knowing your waterbody, your sensors, and your data. Subtle sources of error may not be detected by pre-determined data editing limits, for example, where a slight measurement drift has affected the saturation reading of a dissolved oxygen sensor. Additional tools may be used to assist with post-processing of data following the 'coarse filter' of automated quality control. These tools may assist with retrospectively applying scaling factors to raw data (sensor calibration), back-correcting electronic sensor drift, or editing outliers not detected by pre-set thresholds. The user communities of software platforms such as MATLAB and R are useful resources, as is community-driven software (e.g. see Case Study 13.9).

13.8.3 Metadata

Metadata is information that describes important aspects of data. The creation and maintenance of metadata is an important component of quality data stewardship. In general, it is advised to develop or adopt a ‘controlled vocabulary’ both for data itself and associated metadata. Several organisations have developed controlled vocabulary applicable to high-frequency water quality measurements, including the Global Lake Ecological Observatory Network (GLEON) and the Consortium of Universities for the Advancement of Hydrologic Science, Inc. (CUAHSI). Common metadata that are generated for high-frequency sensor measurements include:

What are the characteristics of the system in which the measurement was made? When comparing across sites, it is frequently beneficial to have summary information on the lake, for example, mean and maximum depth, surface area, and long-term average water quality conditions (e.g. nutrient concentrations, chlorophyll *a* concentration, DOC concentration, and water clarity).

Where was the observation made? Depth, latitude, longitude, as well as more local information such as if the sensor was hung on a shaded side of a buoy and if the sensor depth is relative to the surface or bottom. This also includes information on the location relative to its position in the lake. For example, most buoys are deployed at or near the deepest location in a lake.

What generated the observation? Make and model of the sensor as well as an instrument-specific identifier.

How was the observation generated? Aspects such as the frequency of sampling, sensor integration time, and sensor dwell time (for profiling applications) can impact measurements and their interpretation.

When was the data created? Time stamps are nearly always associated with all high-frequency data, but given the diversity of date-time formats around the world as well as time changes (e.g. ‘daylight savings time’), care should be taken to ensure proper interpretation of data-associated time stamps.

To what level of quality assurance and control were the sensor and observation subjected? This includes information such as the most recent date of calibration (and observed drift between calibrations), how the sensor was calibrated (including the calibration standard(s) used), whether the sensor had accessory components to minimise instrument error (e.g. shade caps, anti-fouling paints or materials, wipers, etc.,), and the degree of fouling observed during routine maintenance and calibration. Information on replaceable sensor components should also be maintained. For example, many optical dissolved oxygen sensors have membranes that are recommended to be replaced regularly (e.g. yearly). The date the sensor cap was replaced and the calibration specifications of the membrane should be maintained.

What post-collection processing was used on the data? Information such as automated or manual data quality control protocols and data versioning is important to record. In general, it is advised that raw data are always maintained alongside any processed data derivatives. Often, it is advisable that users conduct quality assurance and quality control within a scientific workflow environment such as Kepler (https://kepler-project.org/). Scientific workflows provide a process for designing, executing, archiving, and sharing data, processing steps, and provenance information.

Metadata specific to multiple measurements (e.g. site and sensor information) may need to be stored differently to metadata relating to each individual measurement (e.g. post-processing data transformations). Storage of metadata can take various forms, from informal field notes, standalone software tools (see case studies below), or within a formalised database or time-series data management software. The objective is to make the metadata standardised, repeatable, and accessible.

13.9 Case Study: Quality Control of High-Frequency Monitoring Data Using Community-Driven Freeware

High-frequency environmental datasets often utilise a range of sensing technologies, each with myriad factors that may compromise measurement accuracy and precision. Comprehensive quality assurance and quality control are vital for effective and appropriate use of these data, yet rigorous error detection, data editing, and metadata management can be challenging, particularly where the user does not have access to established cyberinfrastructure tailored to these tasks.

'B3' is a freeware, standalone application designed specifically to assist with quality assurance and quality control of environmental sensor datasets. Its aim is to provide an intuitive working environment based on a visual interface for semi-automated editing of outlying data and erroneous measurements (Fig. 13.11). A range of sensor-specific thresholds can be specified, and a variety of methods is provided for rapid error detection. Subsets of data can be selected from an interactive plot and subsequently mathematically transformed or deleted. B3 encourages collection of a comprehensive set of site and sensor metadata, including sensor calibration records which can be used to quickly and easily post-correct electronic drift common to many sensors. Furthermore, B3 records a detailed log of data modifications, including provision for a user to input reasons for any changes. Raw data are then stored as a separate layer so that the original data (or earlier modified data) can be re-created at any time.

The development of B3 was inspired by a need within the Global Lake Ecological Observatory Network (GLEON) research community for an accessible tool for data management (for information about GLEON see Feature Box 13.2 and for similar emerging networks see Feature Box 13.3). The software was developed at the University of Waikato, New Zealand, and has attained broad uptake by lake researchers and managers globally, with user feedback driving ongoing development of its capabilities.

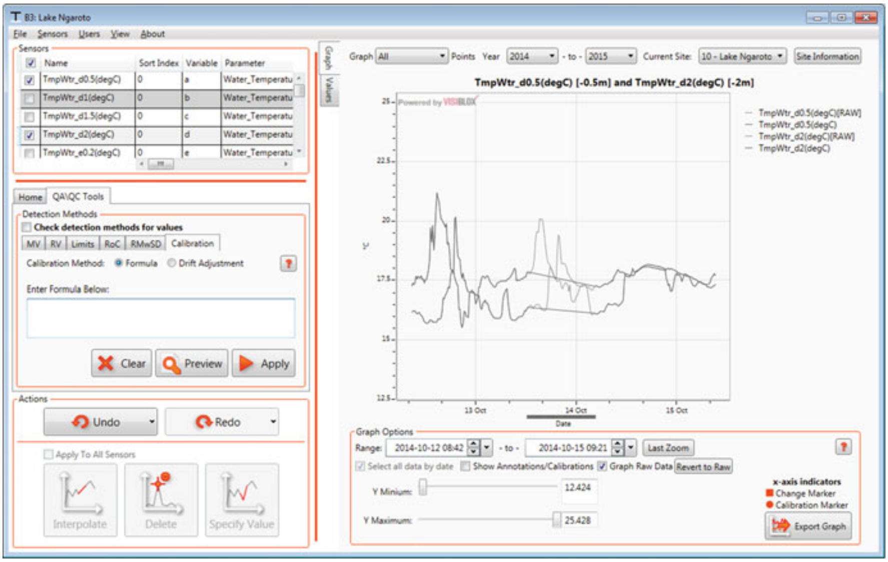

Fig. 13.11 Screenshot from the assisted data quality control freeware 'B3' (https:/www.lernz.co.nz/tools-and-resources/b3)

Box 13.2 Global Lake Ecological Observatory Network

Kathleen C. Weathers[1] and Paul C. Hanson[2]
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[2]Center for Limnology, University of Wisconsin, Madison, WI, USA

The Global Lake Ecological Observatory Network (GLEON), a grassroots network founded in 2005, conducts innovative science by sharing and interpreting high-resolution sensor data to understand, predict, and communicate how lakes respond to a changing global environment. GLEON is primarily a network of people, with more than 500 members (150 students) in 50 countries and growing rapidly. GLEON is also a network of lakes, with 60 lake observatories across 6 continents and in 34 countries (see examples in Fig. 13.12). Finally, GLEON is a network of data, including high-frequency data gathered from sensors mounted on buoys (see figure below). While the lakes and data aspects of the network underpin the scientific products and analyses that GLEON members produce, explicit attention to the people network has served both GLEON science and its members exceedingly well (Weathers et al. 2013). Formed in 2005, GLEON serves as a model of bottom-up international research collaboration. GLEON members are committed to

(continued)

Box 13.2 (continued)

using a 'network science' approach that brings together the networks of people, ecosystems, and data to study lakes worldwide. This approach enables researchers to analyse data from a broad spectrum of lakes and conduct cross-site comparisons and experiments, advancing research on topics such as metabolism and carbon cycling in lakes, the role of wind and advection in lake physics, the development of lake models, and response and recovery of lakes to extreme events. GLEON advances lake science through in-person meetings, including at annual meetings of the network held around the world, and by using a variety of cyber-enabled technologies and working group formats. GLEON engages with lake management organisations, stakeholders and citizen scientists through co-developed research and monitoring programmes. For more information, visit www.gleon.org.

Fig. 13.12 A selection of GLEON lake monitoring platforms, clockwise from top left: L. Vanajanselkä, Finland (Image: L. Arvola), L. Sunapee, USA (M. Eliassen), L. Waikaremoana (M. Osborne), Laguna La Salada, Argentina (L. Borre), L. Mendota, USA (T. Bier), Yang Yuan L., Taiwan (D. Liao), L. Winnebago, USA (M. Felix), and Trout Bog, USA (T. Meinke)

Box 13.3 Networking Lake Observations in Europe (NETLAKE)

Eleanor Jennings
Centre for Freshwater and Environmental Studies, Dundalk Institute of Technology, Dundalk, Ireland

The NETLAKE (Networking Lake Observatories in Europe) COST Action (www.netlake.org) (2012–2016) was set up by European GLEON members, together with other scientists who had been active in automated high frequency monitoring (AHFM) in European lakes. Its members came from 26 - European countries, with additional participants in Australia, New Zealand, and the USA. The outputs included practical guidance, as downloadable factsheets, on station deployment and on data analysis, collaborative research papers, a citizen science project, and a review of the use of AHFM across Europe. The information on AHFM in Europe was captured in both a critical review (Marcé et al. 2016) and in a metadatabase of sites. The latter included data on where lakes were being monitored, details on the frequency and duration of monitoring, and the sensors used. Taken together, these two sources provided a unique snapshot of lake AHFM in Europe in 2015–2016. In total, metadata for stations on 67 European lakes were captured. Twenty-nine of these were Swiss lakes, many at higher altitudes, where only water temperature was measured. However, all other sites had stations measuring multiple parameters. It was of note that only ten sites had data archives that spanned over a decade, seven of which (Muggelsee (Germany), Feeagh (Ireland), Windermere (UK), Estwaithe (UK), Balaton (Hungary), Erken (Sweden)) were stations that had originally been deployed in the EU-funded REFLECT and CLIME projects of the late 1990s and early 2000s. In general, monitoring stations for the metadatabase sites were being used for research purposes: only seven lakes were drinking water sources, while one was a very large Czech fish pond (2.38 km^2, and 6 m in depth). However, Marcé et al. (2016) gave case study examples where AHFM was also being used to provide data on water quality at recreational beaches at lakes, inform salmonid fish management. Of particular note was a complex network of AHFM systems deployed in 18 water supply reservoirs in Sardinia (Italy) by the Ente Acque della Sardegna (ENAS), not included in the NETLAKE database. NETLAKE has provided a foundation for installation, maintenance, and networking of AHFM systems in lakes across Europe and will be a rich resource for researchers as AHFM becomes increasingly mainstream for monitoring of, and research on, lakes globally.

13.9.1 Data Analysis: Derived Variables

Quality controlled high-frequency data can be used to estimate higher level derived products. These derived products characterise aspects of lakes in ways that their source measurements do not provide directly. Some of the most common derived products relate to lake physics/thermodynamics and carbon processing (lake metabolism). Repeatable calculation of derived products depends on using consistent algorithms. Accurate calculation of derived products can be ensured by accessible tools with transparent code. Fortunately, many developers subscribe to a culture of open data, code, and standards. Here, we highlight three software tools used to calculate a number of derived characteristics (Table 13.3). All of those packages are freely available with documented code and use methods and algorithms consistent with established literature, thereby facilitating comparisons of derived products among sites around the world.

13.9.2 Lake Analyzer

Lake Analyzer (Read et al. 2011) is a set of Matlab and R scripts that enable rapid calculation of indices of water column mixing and stratification, using high-frequency data collected from instrumented lake buoys. Derived products (Table 13.3) depend on inputs of bathymetry (hypsographic curve), water temperature, wind speed, water level, and salinity, but not all inputs are required for every product. Lake physical stability indices, surface mixing depth, and thermocline depth are calculated according to established literature definitions and output as a time series. Lake Analyzer also enables the visualisation of derived products and quality checking of data through multiple selectable mechanical screening procedures. The software provides a means to compare mixing and stratification indices in lakes across gradients of climate, hydrophysiography, and time, providing context for resulting biogeochemical processes at different spatial and temporal scales. LakeAnalyzer was produced through collaboration between Lake Ecosystem Restoration New Zealand (LERNZ), GLEON, and other research institutes.

13.9.3 Lake Heat Flux Analyzer

Lake Heat Flux Analyzer (Woolway et al. 2015) is a MATLAB package for ingestion of high-frequency buoy data to calculate energy fluxes. Lake surface energy fluxes regulate many physical processes between lakes and the atmosphere. The strength of energy fluxes regulates the degree of coupling between lake and atmosphere and can have major impacts on lake ecology. However, direct measurement of many fluxes at the water surface can be expensive and difficult. The Lake

Table 13.3 Community-developed software tools for calculating derived products from high-frequency lake water quality datasets

Programme	Derived products	Minimum high-frequency data inputs	Programme source	References
Lake Analyzer	Metalimnion extent Thermocline depth Friction velocity Lake number Wedderburn Number Schmidt stability Mode-1 vertical seiche period Brunt–Väisälä buoyancy frequency	Bathymetry Water temperature Wind speed	http://lakeanalyzer.gleon.org/ or https://cran.r-project.org/web/packages/rLakeAnalyzer/rLakeAnalyzer.pdf	Read et al. (2011)
Lake Heat Flux Analyzer	Surface fluxes of momentum Sensible heat Latent heat Heat transfer coefficients Incoming long-wave radiation Outgoing long-wave radiation	Air temperature Water temperature Relative humidity Wind speed Short-wave radiation	http://heatfluxanalyzer.gleon.org/ or https://github.com/GLEON/HeatFluxAnalyzer	Woolway et al. (2015)
Lake Metabolizer	Gross primary production Ecosystem respiration Net ecosystem production	Dissolved oxygen Water temperature Light Wind speed	https://github.com/GLEON/LakeMetabolizer or https://cran.r-project.org/web/packages/LakeMetabolizer/index.html	Winslow et al. (2016)

Heat Flux Analyzer programme requires time-series inputs of air and water temperature, relative humidity, wind speed, and short-wave radiation. Available outputs include surface fluxes of momentum, sensible and latent heat and their transfer coefficients, and the flux of long-wave radiation.

13.9.4 Lake Metabolizer

Lake Metabolizer (Winslow et al. 2016) is an R package for estimating gross primary production, ecosystem respiration, and the balance between the two (net ecosystem production). Together, these derived products characterise lake metabolism, which is a measure of the biologically mediated flux of carbon dioxide. Lakes are globally important hotspots of carbon cycling. In recent years, research and monitoring of lake gas fluxes has become increasingly common, especially with the increasing use of high-frequency optical dissolved oxygen sensors. Lake Metabolizer addresses this research and management priority by providing tools for consistent comparison of metabolism across sites and through time. Derived products depend on time-series inputs of dissolved oxygen, water temperature, light (commonly PAR), and wind speed.

Because there are multiple methods to calculate metabolism parameters, Lake Metabolizer implements up to five different metabolism models that can be linked with one of several gas flux models in order to provide a broad choice of conceptual assumptions and statistical techniques. However, some models require additional data inputs and parameterisation. Lake Metabolizer also contains an array of functions to manipulate data for related outputs, such as calculations of oxygen saturation and optical conversion.

13.10 Case Study: Monitoring Long-Term Changes in Hypolimnetic Dissolved Oxygen Depletion Rate Using High-frequency Monitoring

Lake Rotorua was the first lake monitoring buoy installation by the University of Waikato, New Zealand, in 2007. Rotorua is a lake of national significance and cultural importance that has suffered eutrophication since the 1960s with increasing pressure (nutrient loads) from pastoral intensification and urbanisation. An important aspect of high-frequency monitoring in the lake has been the tracking of bottom water oxygen depletion during periods of stratification in the polymictic lake. Oxygen depletion can range from a few days to several weeks in duration (Fig. 13.13). Several management strategies have been employed for Lake Rotorua, including land management, improved wastewater treatment, and alum dosing of two major inflows to the lake commencing in 2007. Water quality and the incidence of algae blooms improved substantially between 2003 and 2015.

High-frequency dissolved oxygen data were interrogated to investigate the rate of bottom water oxygen consumption (hypolimnetic oxygen demand) over the duration of the buoy installation. Tens of periods of stratification were captured, for which a selection of daily averaged dissolved oxygen concentrations are presented in Fig. 13.14, where day 1 is defined as the day most recent to the oxygen minimum where oxygen saturation was >85%. It can be seen that whereas in the mid- to late-2000s time to anoxia was generally 15 days, it took up to 27 days to reach anoxia

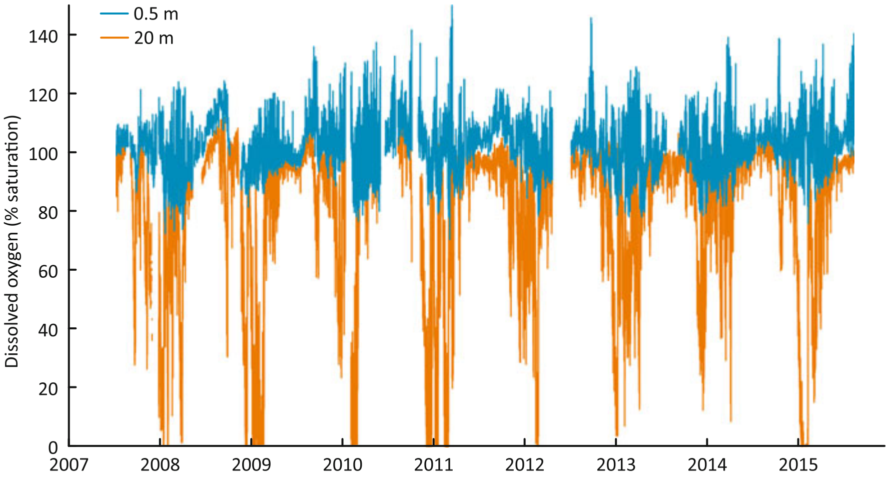

Fig. 13.13 Dissolved oxygen in near-surface (0.5 m) and near-bottom (20 m) waters of Lake Rotorua New Zealand. Data collected at 15 min intervals nearly continuously from mid-2007 to mid-2015

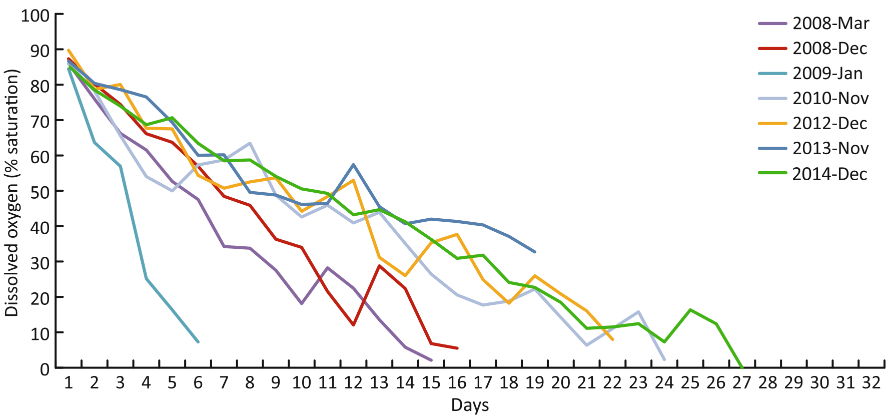

Fig. 13.14 Dissolved oxygen in bottom waters of Lake Rotorua, New Zealand, demonstrating long-term change in the duration taken to reach hypoxia in bottom waters during seven distinct periods of thermal stratification

post-2010. Because nutrients (particularly phosphorus) are released to the water column during hypoxic conditions in many lakes, stratification events have been an important driver of poor water quality in Rotorua (Burger et al. 2007). Understanding change in transient stratification and hypoxia/anoxia is vital for lake management in order to understand and attempt to disrupt the self-reinforcing feedback loop of internal nutrient loading.

13.11 Conclusions and Future Prospects

High-frequency water quality measurements represent an important and growing component of lake monitoring, research, and overall water resource management. High-frequency sensing leverage advances in sensor technologies, cyber-infrastructure, and associated hardware and software. However, sensors are not a panacea to water quality monitoring and research challenges. There are limitations to many types of sensors that should guide the application and interpretation of generated data. Furthermore, high-frequency sensor measurements do not always directly correspond to traditionally measured variables, meaning that users should not expect similar patterns or characteristics. These challenges have slowed the evolution and uptake of high-frequency sensors in some domains. However, despite these limitations, high-frequency sensor measurements open new windows of opportunity to understand phenomena that operate across multiple temporal and spatial scales. Use of high-frequency sensors can improve monitoring and management programmes by enabling rapid responses, reducing some long-term costs, and better characterising transient phenomena that can have legacies across space and time.

High-frequency buoy-based sensor measurements are poised to change in several important ways in the years ahead. First, the increasing development and application of data analysis capabilities has the potential to move buoy-based technologies beyond simply logging data to an environment where buoys and associated cyber-infrastructure can natively conduct many aspects of analytics, including quality control, adaptive sampling, and hypothesis generation, based on automated detection of thresholds, trends, and impending change. Secondly, sensor technologies are continually evolving and changing. While it is difficult to predict what new sensors will come to market in coming years, new technologies and methods for analysing existing data will no doubt provide opportunity to directly or indirectly infer additional water quality characteristics. Furthermore, miniaturisation, open hardware and software, and economies of scale will likely improve the accessibility of sensor technologies, while Moore's Law and software development should ensure the feasibility of measurement, storage, and processing of data at ever-finer spatial and temporal scales.

Sub-daily measurements are immensely useful. Nevertheless, the value of such datasets increases dramatically when their total duration extends to years or even decades. Therefore, future research and monitoring will inevitably benefit greatly from the existence of long-term high-frequency sensor data. Presently available technologies are mature to the point where they are applicable to a wide range of uses, and in an era of declining water quality and rapid environmental change, high-frequency sensors provide new opportunities to better understand and manage our global water resources.

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Chapter 14
Remote Sensing of Water Quality

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Abstract There is a need for increased monitoring of freshwater resources to effectively manage water quality. Remote sensing has the potential to substantially improve the spatiotemporal resolution of monitoring. Satellites sensors vary in temporal, spatial, radiometric, and spectral resolution. For remote sensing of optically active water constituents, there is no one definitive remote sensing solution for any lake or group of lakes, and the method needs to be tailored to the lake size and optical complexity of the system and the spectral and spatial resolution of the sensor. There are three general categories of sensors for spaceborne remote sensing, including hyperspectral sensors, broadband medium spatial resolution sensors, and narrow band low spatial resolution satellite sensors. The satellite sensor's spatial resolution will determine the minimum lake size that can be monitored via remote sensing, while the sensor's spectral resolution will determine the ability of the sensor to differentiate optically active constituents. Algorithms for remote sensing of water constituents can be divided into empirical, semi-analytical, or analytical methods. Empirical methods are applicable where there is a simple relationship between the constituent of interest (e.g., chlorophyll *a*) and reflectance and are therefore usually limited to lakes where only one water quality constituent dominates reflectance. Semi-analytical algorithms can often be applied in place of empirical algorithms and have a number of advantages. Semi-analytical algorithms can be developed independently of in situ samples, are applicable to multiple satellite sensors, have greater spatiotemporal applicability, and are designed to determine more than one water quality parameter simultaneously. Analytical methods are based on radiative transfer modelling or simplifications thereof.

Keywords Landsat · Sentinel · Bio-optical · Satellite

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14.1 Introduction

Remote sensing is the science of attaining information about an object without direct contact with that object. According to this definition, remote sensing can be postulated as being pioneered by early astronomers such as Galileo and Copernicus. Modern environmental remote sensing was developed through military reconnaissance carried out during World War I and enabled by technological advances accelerated during early space exploration programs (Bukata et al. 1995).

Remote sensing (RS) sensors measure electromagnetic (EM) radiation reflected or emitted from Earth and may be "active" or "passive." Active remote sensing measures the reflectance of a signal that is first emitted from an aerial platform, for example, Light Detection and Ranging (LIDAR) applied to topographic surveys. When applied to surface waters, RS is generally "passive," analyzing reflected solar energy to characterise physical, chemical, and biological characteristics of water including temperature, turbidity and suspended particles (SP), water clarity (e.g., Secchi disk depth), coloured dissolved organic matter (CDOM), and chlorophyll *a* (chl *a*) concentration. Aquatic RS is most often associated with measurements from satellite-mounted spectral sensors; however, alternative aerial platforms including small aircraft and drones are becoming more widespread in use. Temporal resolution of RS is highly variable, depending on the orbital period or flight frequency of the aerial platform, and spatial resolution (the pixel size of a remotely sensed image) can vary from <1 to >1000 m depending on the sensor used. Spectral sensors measure reflectance within discrete "bands" (ranges of wavelength), and analyses may involve as little as a single band, for example, from a satellite-mounted sensor, to many hundreds of bands, for example, from an airplane survey using a hyperspectral imager (see Feature Box 14.1). Algorithms are then used to convert the spectral signature of reflectance across these bands to estimates of water and land properties. Such algorithms may be either empirical (by statistical relationship with in situ measurements) or analytical (using bio-optical models).

Box 14.1 Hyperspectral Remote Sensing to Assess Water Quality Parameters in Chilean Lakes

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In 2014, the Chilean water authority (http://www.dga.cl) and Centro de Ecologia Aplicada (http://www.cea.cl) carried out an innovative project to evaluate the use of airplane-mounted hyperspectral remote sensing as a monitoring tool for the Chilean Lake Network. Corresponding measurements from remote sensing and field sampling were used to assess empirical algorithms for deriving chlorophyll *a* and suspended solids from spectral reflectance

(continued)

Box 14.1 (continued)
signatures. The objective was to provide a robust method for describing spatial variation in surface water quality in order to improve understanding of diverse processes within Chilean lakes. Aerial hyperspectral surveys included 21 lakes and reservoirs of varying trophic status (oligotrophic to hypertrophic), latitudes (33.1°S–41.5°S), and altitudes (50–2700 masl). Datasets were acquired using a hyperspectral sensor, Specim AISA Eagle, with 128 spectral bands of 5 nm width in the visible and near-infrared (VNIR) range between 380 nm and 960 nm. The aerial platform was a Piper Navajo airplane and the flight was planned at about 3000 m altitude above water surface, producing a 2.5-m pixel. The flight plan considered a solar angle range between 30° and 60° to avoid specular reflection on water. During the flights, corresponding "ground level" spectral measurements were taken at five lakes, along with water samples for laboratory analysis. Spectral measurements were acquired using a spectroradiometer ASD HandHeld 2, also in the VNIR range from 325 to 1075 nm, and water samples were analyzed for total chlorophyll *a*, suspended solids, and coloured dissolved organic matter (CDOM). Through comparison of spectral data with field observations, diverse algorithms were tested to evaluate the performance of specific wavelengths as well as spectral indices. Constituents present in water diversely affect the electromagnetic spectrum, due principally to absorption and reflection by pigments (e.g., chlorophyll *a*), dissolved matter (e.g., CDOM), and SP. Distinct spectral signatures were observed among the surveyed Chilean lakes (Fig. 14.1).

Calibrated hyperspectral data enabled the production of detailed maps of spatial patterns in chlorophyll *a* and sediments coming from tributary rivers. In some lakes and reservoirs, substantial spatial heterogeneity was observed, prompting redesign of the sampling protocols for official water quality monitoring in order to better capture sub-basin-scale processes affecting water quality (Fig. 14.2).

An important application of remote sensing is assessment of spatial heterogeneity in surface water quality. Horizontal variation is an important feature of many lakes and reservoirs, particularly where bathymetry is elongated, dendritic, or disproportionately influenced by surface inflow(s) at certain shores. Further, spatial distribution of harmful algal blooms may be highly variable depending on chemical (nutrient supply) and physical (wind, advection) forcing. Quantifying this variation using traditional sampling is laborious at best, entirely impractical at worst, yet can be quickly and routinely assessed using established remote sensing methods if the water body is sufficiently large. Additionally, the existence of a vast historical archive of remote sensing images from a variety of satellites and sensors enables the "hindcasting" of water quality at places and times where no manual samples have been collected. From the perspective of lake restoration, hindcasting estimation of water quality in unmonitored lakes may provide information on historical water quality, aiding in target setting for restoration scenarios.

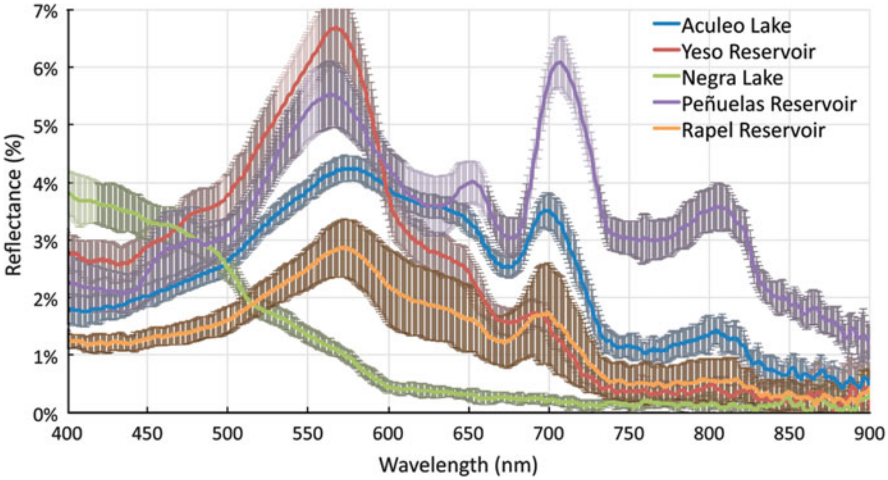

Fig. 14.1 Spectral signature of water in five lakes and reservoirs in Central Chile

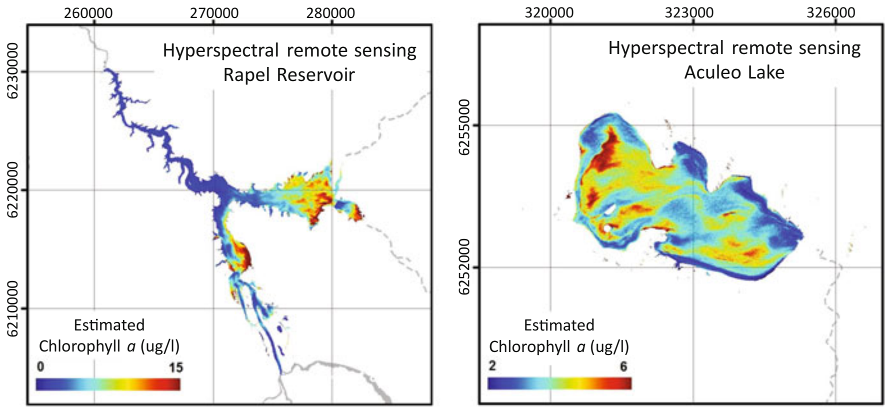

Fig. 14.2 Estimated chlorophyll *a* concentrations (μg L^{-1}) in Rapel Reservoir and Aculeo Lake, derived from empirical algorithms applied to hyperspectral data

14.2 Remote Sensing Theory

Passive remote sensing involves sensors that record EM energy reflected or emitted by the earth. The energy can be modeled either as a wave or as a particle which travels at the speed of light. The most important characteristic of EM for the purpose of remote sensing is the wavelength (λ). Longer wavelengths are associated with lower frequency and energy according to Eq. (14.1) which defines Q, the amount of energy held by a photon (measured in joules), as:

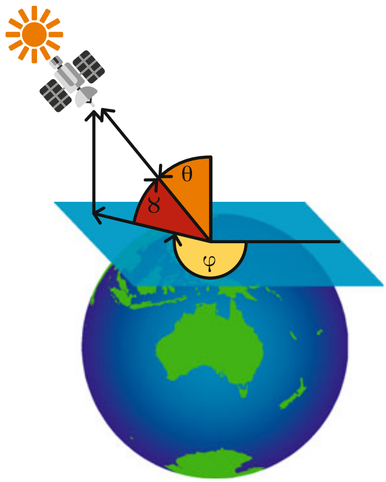

Fig. 14.3 Satellite and solar geometry. θ is azimuth angle, ϑ is elevation angle, and φ is zenith angle, which can apply to either solar or sensor (view) geometry

$$Q = hv = h\frac{c}{\lambda} \tag{14.1}$$

where h is Planck's constant, v is frequency, and $c = 2.998 \times 10^8$ m s^{-1}.

Remote sensors record EM in the form of radiance, $L(\theta, \varphi, \lambda)$, which is defined as the radiant flux per unit solid angle per unit area at right angles to the direction of EM propagation (W sr m^{-2}), with a specified azimuth (θ) and zenith (φ) angle. Radiance is a subset of irradiance $E(\lambda)$, which is defined as the total radiative flux incident upon a point on a surface from all directions above the surface. The geometry associated with a remote sensing instrument is shown in Fig. 14.3.

The simplified radiance recorded by a remote sensor can be expressed (assuming no adjacency effects or bottom reflection) as:

$$\begin{aligned} L_t(\theta_v, \varphi_v, \theta_s, \lambda) = {} & L_r(\theta_v, \varphi_v, \theta_s, \lambda) + L_a(\theta_v, \varphi_v, \theta_s, \lambda) \\ & + T_{atm}(\theta_v, \theta_s, \lambda) L_s(\theta_v, \varphi_v, \theta_s, \lambda) \end{aligned} \tag{14.2}$$

where:

$L_r(\lambda)$ = radiance due to multiple scattering by air molecules (Rayleigh scattering)
$L_a(\lambda)$ = radiance from multiple scattering of aerosols
$T_{atm}(\lambda)$ = atmospheric transmittance from water to sensor
$L_s(\lambda)$ = surface-leaving radiance
λ = wavelength
θ_v = view or observation zenith angle
φ_v = view azimuth angle
θ_s = solar zenith angle

Adjacency effects are caused by reflection from a nontarget area (e.g., land) being scattered into the field of view by the atmosphere. If the nontarget area is brighter than the target, there will be an increase in the apparent brightness of the target area and a reduction of image contrast (Santer and Schmechtig 2000).

Surface-leaving radiance, $L_s(\theta_v, \varphi_v, \theta_s, \lambda)$, is estimated through applying an atmospheric correction, described below. Water-leaving radiance is estimated from Kallio (2012):

$$L_w(\theta_v, \varphi_v, \theta_s, \lambda) = L_s(\theta_v, \varphi_v, \theta_s, \lambda) - \sigma_L L_{sky}(\theta_v, \varphi_v, \theta_s, \lambda) \tag{14.3}$$

where:

$L_{sky}(\theta_v, \varphi_v, \theta_s, \lambda)$ = sky specular reflectance at sensor and σ_L = Fresnel reflectance

Water surface remote sensing reflectance (sr^{-1}) is given as:

$$R_{rs}(\theta_v, \phi_v, \theta_s, \lambda) \equiv \frac{L_w(\theta_v, \phi_v, \theta_s, \lambda)}{E_d(\lambda)} \tag{14.4}$$

where E_d is downwelling irradiance immediately above the water surface.

Subsurface remotely sensed reflectance $r_{rs}(\lambda)$ can be estimated from $R_{rs}(\lambda)$ by correcting for air–water interface effects, assuming a nadir viewing sensor and optically deep waters where bottom reflection does not occur (Lee et al. 2002):

$$r_{rs}(\lambda) \approx \frac{R_{rs}(\lambda)}{0.52 + 1.7 R_{rs}(\lambda)} \tag{14.5}$$

The general term reflectance used in this chapter refers to $r_{rs}(\lambda)$. Subsurface irradiance reflectance $R(0-)(\lambda)$ is given as (depth = 0, spectrally dependent):

$$R(0-) = \frac{E_d}{E_u} \tag{14.6}$$

where E_u is upwelling radiance.

14.2.1 Water Colour Theory

The colour and clarity of water depend on the interaction of the optical properties of water and the concentrations and properties of optically active substances (substances which scatter and absorb light) and relate to the bulk optical processes of absorption and scattering (Davies-Colley et al. 1993). Absorption and scattering characteristics of a water body are determined by inherent optical properties (IOPs). Absorption refers to the transfer of light energy into another form of energy (e.g., heat) and is quantified by the spectral absorption coefficient, $a(\lambda)$ (m^{-1}), which is the fraction of incident light absorbed divided by the thickness of the layer. Scattering is defined as deflection of light photons from their original path (Davies-Colley et al. 1993), and can be quantified by the spectral scattering coefficient, $b(\lambda)$ (m^{-1}), which is the fraction of the incident light which is scattered divided by the thickness of the layer (Kirk 2010). Light scattering is further quantified by the volume scattering function ($\beta(\theta)$), where θ is the forward scattering angle (Mobley 1994). The integral of $\beta(\theta)$ for angles from 0 to π yields b:

$$b = 2\pi \int_{0}^{\pi} \beta(\theta) \sin(\theta) \mathrm{d}\theta \quad (14.7)$$

The backscattering coefficient is a subset of the angle used to define b:

$$b_{\mathrm{b}} = 2\pi \int_{\pi/2}^{\pi} \beta(\theta) \sin(\theta) \mathrm{d}\theta \quad (14.8)$$

The backscattering ratio B_{b} is defined as $b_{\mathrm{b}}(\lambda)/b(\lambda)$.

Apparent optical properties (AOPs) are determined by the combined effects of the geometric structure of the light field and water constituents (Kirk 2010). They are therefore partly determined by the solar zenith angle and local atmospheric conditions as well as the IOPs of the water body (Bukata et al. 1995). The underwater vertical diffuse attenuation coefficient (K_{d}), Secchi depth, and irradiance reflectance are considered to be AOPs (Preisendorfer 1976; Mobley 1994). In contrast, IOPs of a water body are completely quantified by $\beta(\theta)$ and $a(\lambda)$ and are affected only by the medium. Radiative transfer theory describes the connection of IOPs and AOPs (Mobley 1994).

In natural ecosystems, clear water contains low concentrations of optically active constituents (OACs). Consequently, the spectral reflectance is low and the spectral shape is similar to that for pure water molecules (Pope and Fry 1997). It shows an exponential increase in absorption toward longer wavelengths and an increase in scattering at shorter wavelengths of the visible range of the electromagnetic spectrum (Rudorff 2006).

Algae-laden water exhibits a reflectance peak in the green region, which represents an aggregate pigment absorption minimum and another reflectance peak at 700 nm. Absorption troughs occur in the blue and red/ IR wavelength range (Han 1997), with the exact location and width of these troughs dependent on phytoplankton species' assemblages and their physiological state (Kirk 2010). Fluorescence is an optical process which involves the absorption of part of the energy of a photon and re-radiation of a lower energy photon in an arbitrary direction, which can sometimes influence the colour of natural waters (Davies-Colley et al. 1993). Phytoplankton display a fluorescence peak centered at 685 nm, meaning concentrations can be measured using fluorometers when photoreaction centers are not saturated (Yentsch and Yentsch 1979), and reflectance measured by satellites at this wavelength can also be used in algorithm development to estimate chl *a* concentration.

Suspended particles (SP) include sand, silt, clay, and other inorganic material such as atmospheric dust, as well as organic matter. The optical properties of SP are affected by the shape and size distribution of particles and their chemical makeup, which have a major effect on their absorption and scattering properties (Bukata et al. 1995). Increasing concentrations of SP result in a linear increase in reflectance in the IR, with a coefficient of variation near to one. In algae-laden water, additive effects of SP on reflectance occur at all wavelengths (Bukata et al. 1995).

The absorption of coloured dissolved organic matter (CDOM) increases exponentially at shorter wavelengths of the visible spectrum, and there is little absorption above 700 nm (Bricaud et al. 1981). The effect of CDOM on light scattering is minimal and is usually ignored.

14.2.2 Atmospheric Correction of Remote Sensing Data

Atmospheric correction is a crucial step in any time series analysis of satellite imagery as a satellite only captures a fraction of radiation coming from a target. Radiance recorded by remote sensing can comprise greater than 90% path radiance, which originates from scattering of solar radiation by air molecules and aerosols (suspended liquid and particles such as salt, dust, ash, pollen, and sulfuric acid) (Vidot and Santer 2005). These molecules and particles can also attenuate radiation through absorption. Scattering can occur multiple times, and there is variability of aerosol particulate distributions (Bukata et al. 1995), resulting in complex photon paths.

There are three classes of methods of atmospheric correction: (1) vicarious calibration correction, (2) image-based correction, and (3) radiative transfer modelling correction. Vicarious calibration involves measuring solar irradiance and reflectance concurrently with satellite overpass, enabling the estimation of path radiance. The most common image-based atmospheric correction to estimate L_s involves dark object subtraction, in which the darkest pixel in each band is used as an estimate of path radiance and subtracted from radiance at the top of the atmosphere (Chávez 1996).

The application of radiative transfer models offers the flexibility to address the complexities of atmospheric correction over inland waters (e.g., Campbell et al. 2011). Such complexities include adjacency effects and variations in elevation, which affect molecular calculations due to changes in air pressure. In addition, heterogeneous concentrations of aerosols and aerosol content at coastal and inland locations need to be considered, although it is often assumed that aerosol optical depth (AOD) is homogeneous at spatial scales of 50–100 km (Vidot and Santer 2005). There is also interaction of absorption and scattering, and the radiative transfer model must provide a complete description to provide accurate simulation. Gaseous absorption is therefore calculated for each scattering path while absorption is computed as a simple multiplicative factor. The principal absorbing gases are oxygen, ozone, water vapor, carbon dioxide, methane, and nitrous oxide, of which ozone and water vapor are assumed to vary most in time and space (Kotchenova et al. 2008). The calculation of scattering is much more complex and includes contributions from Rayleigh and aerosols. Second Simulation of a Satellite Signal in the Solar Spectrum (6sv) (Kotchenova et al. 2008) is a radiative transfer atmospheric correction code accounting for radiation polarisation, which enables accurate simulations of satellite and airborne observations, accounting for a molecular and aerosol mixed atmosphere.

An alternative approach to atmospheric correction of ocean colour data is the use of the SeaDAS image analysis software functions for atmospheric correction (Ruddick et al. 2000; Wang et al. 2009; Bailey et al. 2010), which preserve the near-infrared (NIR) reflectance peak (Odermatt et al. 2012a). However, this procedure can return negative reflectance at blue wavelengths over turbid or inland waters (Goyens et al. 2013). A promising alternative, which obviates negative reflectance, is a neural network (NN)-based inversion approach (Schroeder et al. 2007), which has been shown to outperform SeaDAS atmospheric correction algorithms over turbid waters (Goyens et al. 2013).

14.3 Methods of Remote Sensing of Water Quality

Case 1 waters occur where variation in $r_{rs}(\lambda)$ is primarily due to phytoplankton, and simple empirical band ratio models can be used to determine chl *a* (Gordon and Morel 1983). Case 2 waters are classified as optically complex, and $r_{rs}(\lambda)$ varies with differing IOPs and concentrations of optically active constituents, namely, chl *a*, SP, and CDOM. Most inland waters can be considered Case 2; however, there are inland waters where concentrations of IOPs are low enough to enable application of algorithms designed for Case 1 conditions.

There are three general approaches to retrieving water quality parameters via remote sensing: empirical and analytical methods, with the term semi-empirical/semi-analytical applied to approaches somewhere between the two (Carder et al. 1999). Empirical methods use statistical techniques to directly relate in situ samples to concurrently acquired satellite data. These models offer a simple approach but have limited spatiotemporal applicability. Analytical methods are based on bio-optical models which use radiative transfer modelling or simplifications (semi-analytical) of radiative transfer models. Bio-optical models can be developed independently of in situ data; however, they require knowledge of, or assumptions about, inherent optical properties. The spatiotemporal applicability of empirical and analytical models depends on variations in IOPs of the target water and how these affect the water quality retrieval of the parameter of interest.

14.3.1 Bio-Optical Models and IOPs

Bio-optical models are generally used in two forms: complex radiative transfer models such as HYDROLIGHT (Mobley 1994) and simplifications such as the analytical model of Gordon et al. (1988). In the latter model, $r_{rs}(\lambda)$ is given by:

$$r_{rs}(\lambda) = g_0 u(\lambda) + g_1 [u(\lambda)]^2 \tag{14.9}$$

where:

$$u(\lambda) = \frac{b_b(\lambda)}{a(\lambda) + b_b(\lambda)} \tag{14.10}$$

$b_b(\mathrm{m}^{-1})$ is the total backscattering coefficient, $a(\mathrm{m}^{-1})$ is the total absorption coefficient, and g_0 and g_1 are empirical constants that depend on the anisotropy of the downwelling light field and scattering processes within the water.

The model of Gordon et al. (1988) was successfully applied to eutrophic inland waters (Dekker et al. 1997) (spectral dependence not shown) as:

$$R(0-) = r_1 \left[\frac{b_b}{b_b + a}\right] \tag{14.11}$$

where $R(0-)$ is subsurface irradiance reflectance and r_1 depends on the anisotropy of the downwelling light field and scattering processes within the water.

The absorption and backscattering coefficients comprise the sum of individual optically active components:

$$b_b(\lambda) = b_{bw}(\lambda) + B_{bSP} b^*_{SP}(\lambda)\, C_{SP} \tag{14.12}$$

$$a(\lambda) = a_w(\lambda) + a^*_{\phi}(\lambda)\, C_{\phi} + a^*_{SP}(\lambda)\, C_{SP} + a_{CDOM}(\lambda) \tag{14.13}$$

$$a_{CDOM}(\lambda) = a_{CDOM}(\lambda_0) \exp[-S(\lambda - \lambda_0)] \tag{14.14}$$

$$b^*_{SP}(\lambda) = b^*_{SP}(\lambda_0) \left(\frac{\lambda}{\lambda_0}\right)^n \tag{14.15}$$

where:

(λ) = wavelength
(λ_0) = reference wavelength
$b_{bw}(\lambda)$ = backscattering coefficient of water
B_{bSP} = backscattering ratio of SP
$b^*_{SP}(\lambda)$ = specific scattering coefficient of SP
C_{SP} = concentration of SP
$B_{p\phi}$ = backscattering ratio of phytoplankton
$b^*_{\phi}(\lambda)$ = specific scattering coefficient of phytoplankton
$a_w(\lambda)$ = absorption coefficient of pure water
C_{ϕ} = concentration of chl *a*

$a_{\phi}^{*}(\lambda)$ = specific absorption coefficient of phytoplankton
a_{SP}^{*} = specific absorption coefficient of SP
$a_{\mathrm{CDOM}}(\lambda)$ = absorption by coloured dissolved organic matter (CDOM)
S = the spectral slope coefficient for $a_{\mathrm{CDOM}}(\lambda)$
n = the exponent for SP scattering

Inherent optical properties vary temporally and spatially (Kostadinov et al. 2010; Devred et al. 2011; Moisan et al. 2011), which presents a potential source of error when inverting bio-optical models in the estimation of optically active constituent concentrations. Variance in the phytoplankton pigment-specific absorption coefficient ($a_{\phi *}(\lambda)$) is a result of several factors such as phytoplankton pigment composition, cell size, packaging effect, light accumulation, and nutrient limitation (Babin et al. 1993; Babin 2003; Bricaud 2004; Blondeau-Patissier et al. 2009). Variation of $a_{\phi}^{*}(\lambda)$ causes non-linearity between light absorption and chl *a* concentration (Bricaud 2004).

The backscattering ratio $B_{p\phi}$ and the specific scattering coefficient $b_{\phi}^{*}(\lambda)$ of phytoplankton are determined by their size, physical structure, and outer coating (Stramski et al. 2004). With increasing phytoplankton biomass, there is greater cell wall surface area, which causes increased scattering (Yacobi et al. 1995). The presence of gas vacuoles in some cyanobacteria has been found to substantially increase the value of $b_{\phi}^{*}(\lambda)$ (Dubelaar et al. 1987; Volten et al. 1998).

The specific scattering coefficient of SP, $b_{\mathrm{SP}}^{*}(\lambda)$, varies in time and space, mostly due to variations in particle size distribution and the bulk refractive index of the particles (Tzortziou et al. 2006). The refractive index of inorganic SP is generally higher than that of organic SP (including phytoplankton) due to the high water content of organic material (Twardowski et al. 2001). The slope of the SP backscattering coefficient is related to the particle size distribution and, in combination with angular scattering, can be used to estimate the refractive index of particles (Twardowski et al. 2001).

Concentrations of CDOM are often substantially higher in inland, estuarine, and coastal ecosystems compared with open oceans. The spectral slope coefficient of CDOM, S, is known to vary with the proportion of fulvic and humic acids (Carder et al. 1989) and molecular weight of the DOM pool (Helms et al. 2008). Autochthonous sources of CDOM include decay products from phytoplankton, for example, which can lead to covariance of CDOM with chl *a* in open ocean waters (Carder et al. 1989). Allochthonous sources include decaying organic matter such as that derived from leaf litter and are often associated with river inputs.

Bio-optical models have generally been applied to satellite imagery with high spectral resolution and to hyperspectral data, both of which can be used to derive precise measurement of spectral slopes of reflectance. Various methods of solving (or inverting) these models are used, including Monte Carlo simulation (Gordon and Brown 1973; Morel and Prieur 1977; Kirk 1981), invariant embedding (Preisendorfer 1976; Mobley 1994), matrix operator method (Fischer and Grassl 1984; Fell and Fischer 2001), and finite-element method (Kisselev et al. 1995; Bulgarelli et al. 1999). Bio-optical models have also been inverted efficiently using the Levenberg–

Marquardt nonlinear least-squares procedure (Maritorena et al. 2002; Vanderwoerd and Pasterkamp 2008) and neural networks (Doerffer and Schiller 2007).

14.3.2 Spaceborne Earth Observation Sensors

There are three general categories of sensors for spaceborne remote sensing: hyperspectral sensors such as Hyperion, broadband medium spatial resolution terrestrial remote sensing sensors such as Landsat's Enhanced Thematic Mapper (ETM+), and narrow band low spatial resolution satellite sensors such as MOderate Resolution Imaging Spectroradiometer (MODIS) and MEdium Resolution Imaging Spectrometer (MERIS). Satellite sensors vary in temporal, spatial, radiometric, and spectral resolution (Table 14.1). The spatial resolution will determine the minimum lake size that can be monitored via remote sensing, while the radiometric and spectral resolution will determine the complexity of any derived algorithms and the ability of the sensor to differentiate optically active constituents (Matthews 2011).

The MODIS sensor developed by NASA has been fitted to the Terra (data available from February 2000) and Aqua (data available from July 2002) satellite platforms and enables twice-daily global coverage. NASA now has a policy of full and open sharing of imagery for Earth observing satellites spanning more than 40 years. This policy has contributed to increased use of remote sensing for many applications including monitoring quality of inland waters. The high spatial resolution of NASA Landsat series of sensors and free availability of the data archive have made Landsat the sensor of choice for monitoring inland water quality in small lakes. The Landsat satellite sensor has relatively high spatial resolution (30 m) compared with ocean colour sensors such as MODIS (250–1000 m resolution) and is therefore able to resolve finer-scale features within lakes, as well as to record data from small lakes where there would otherwise be inadequate resolution. Landsat Multispectral Scanner (MSS) imagery is available from 1972 to 1981, Landsat 5 Thematic Mapper (TM) was launched in 1984 and is still operating, Landsat 7 ETM+ was launched in 1999, and Landsat 8 in 2013. The repeat cycle of image capture is 16 days, and each scene is 185 km wide and 120 km high.

14.3.3 Low-Resolution Applications of Remote Sensing for Optically Active Constituent Retrieval of Case 2 Waters

Remote sensing of Case 2 waters using ocean colour sensors has been applied using various algorithms to estimate optically active constituents, including traditional ocean colour band ratios, and empirical, semi-analytical, and analytical algorithms

Table 14.1 A list of satellite platforms with a demonstrated history or potential to monitor water quality operationally

Satellite	Spatial resolution (m)	Repeat cycle (days)	Spectral bands (400–1100 nm)	Life span (years)	Minimum water body size (ha)	Ideally suited to simultaneous retrieval of Chl, CDOM, and SP
Landsat 5	30	16	5	1999–2004+	1.5	No
Landsat 8	30	16	6	2013–2023+	1.5	No
Sentinel-2 A&B	10	5	9	2015–2023+	0.5	No
Sentinel-3 A&B	300	2	21	A: 2016–2025+	100	Yes
MODIS TERRA and AQUA	250–1000	1	16	1999–2017+	100	Yes
MERIS	300	3	15	2002–2012	100	Yes
VIIRS	375–750	1	10	2011–2016+	100	Yes

Note: At time of writing, Sentinel-3B not yet launched

(Odermatt et al. 2012a). There has been a large body of research on remote sensing of the Laurentian Great Lakes (e.g., Budd and Warrington 2004; Witter et al. 2009; Binding et al. 2010; Mouw et al. 2013), and comparisons can be drawn with remote sensing of other environments. The trophic status of the Great Lakes ranges from oligotrophic in Lake Superior to eutrophic (and higher) in the western basin of Lake Erie. In a recent review of satellite ocean colour algorithms applied to the Great Lakes (Lesht et al. 2012), it was shown that no single retrieval method produced consistent results due to the unpredictable nature of the IOPs. The review by Lesht et al. (2012) highlighted the complexities of remote sensing of Case 2 waters and demonstrated the complicated and often conflicting information that has been presented in the literature. As with the Great Lakes, universally applicable algorithms for the remote sensing of optically active constituents have not yet been established (Odermatt et al. 2012a). Band ratio ocean colour algorithms have been applied with various degrees of success. In some instances, there are strong linear relationships between ratios and in situ chl *a*; however, the slope of the relationships often deviates significantly from one, requiring some regional parameterisation. Budd and Warrington (2004) applied the OC2-V2 algorithm in oligotrophic Lake Superior and found strong covariance between estimated chl *a* with in situ samples, but there was a threefold overestimate of chl *a* concentration. Witter et al. (2009) compared 12 empirical ocean colour algorithms for assessment of chl *a* concentration in Lake Erie and found reasonable relationships between in situ and satellite-derived chl *a*, but with significant bias at high and low chl *a* concentrations. These authors then developed a regional empirical algorithm specific to the western, eastern, and central basins of Lake Erie and concluded that the eastern and central basin algorithms produced promising results; however, regional algorithms were not applicable to the western basin.

Empirical ocean colour algorithms have been applied assuming Case 1 conditions in inland waters (e.g., Heim 2005; Horion et al. 2010); however, in a review paper of remote sensing of Case 2 waters, inaccuracies in chl *a* retrieval were considered to result from independently varying CDOM concentrations, especially at short wavelengths (Odermatt et al. 2012a). Using a bio-optical model based on that of Bukata et al. (1985), Lesht et al. (2012) demonstrated that traditional empirical and band ratio algorithms for chl *a* retrieval are susceptible to error due to varying concentrations of CDOM and SP. Modifications have been applied to ocean colour algorithms in order to minimise the influence of CDOM using MERIS data, with some success in oligotrophic waters of Lake Superior (Gons et al. 2008). However, their retrieval approached an asymptote at chl *a* concentrations above 2.7 μg L^{-1} and is therefore not applicable to meso- and eutrophic waters.

Semi-analytical algorithms have been applied to remote sensing of optically active constituents in the Great Lakes. They can be used to estimate SP by inversion of field-measured subsurface irradiance reflectance (Bukata et al. 1985) and inversion of MODIS volume reflectance (Binding et al. 2010) and CDOM (Bukata et al. 1985; Mouw et al. 2013). The success of bio-optical models for the estimation of chl *a* ranged from complete failure (e.g., Bukata et al. 1985; Mouw et al. 2013) to algorithms with poor accuracy (e.g., Li et al. 2004; Shuchman et al. 2006). Mouw et al. (2013) attributed the failure of the bio-optical algorithm in retrieving chl *a* in Lake Superior to the dominant influence of CDOM on the absorption budget. Bukata et al. (1985) attributed the chl *a* retrieval failure to varying IOPs and the dominance of non-algal particles and CDOM on retrieved reflectance via absorption and scattering processes. Shuchman et al. (2006) applied a version of the Bukata et al. (1985) algorithm using updated IOPs from Bukata et al. (1991); however, chl *a* retrievals were underestimated by an order of magnitude (Lesht et al. 2012).

In addition to the complexities of remote sensing of water quality caused by independently varying concentrations of optically active constituents in Case 2 waters, there may be atmospheric correction errors resulting from incorrect assumptions such as the use of fixed aerosol models (Odermatt et al. 2008). Accurate atmospheric correction is critical for remote sensing of oligotrophic lakes, as the reflectance of surface water is small compared to the atmospheric path radiance (Guanter et al. 2010). Adjacency effects also tend to be more pronounced for oligotrophic waters with low reflectivity (Odermatt et al. 2008).

For the estimation of chl *a* concentrations for oligotrophic inland waters, it has been suggested that more complex, physically based inversion models are necessary in order to differentiate optically active substances (Odermatt et al. 2010). The Modular Inversion and Processing System (MIP) is a set of processing tools for the retrieval of water and atmosphere (look-up table approach) constituents from hyperspectral and multispectral satellite data (Heege and Fischer 2004). Odermatt et al. (2008) used MIP for the retrieval of chl *a* from MERIS imagery in oligotrophic to mesotrophic Lake Constance, Germany. The authors stated that the correlation between retrieved and in situ chl *a* was sufficient considering that the time differences between in situ and satellite image acquisition were up to 3 days (and in some instances longer). Odermatt et al. (2008) considered that atmospheric correction was the greatest potential source of error (including adjacency effects) but did not state the level of error; however, the method may not provide sufficient resolution for determining small changes in chl *a* (c. 1 μg L^{-1}) in oligotrophic waters.

While remote sensing of water quality by NASA has focused primarily on coastal and oceanic waters, the European Space Agency (ESA) has also funded research on development of Case 2 coastal water quality algorithms (including the MERIS algal_2 product), and MERIS lake water quality algorithms including Case-2 Regional (C2R), Boreal Lakes, and Eutrophic Lakes water quality retrieval algorithms (Doerffer and Schiller 2007). The MERIS Case 2 water quality algorithm uses a neural network (NN) inversion technique to retrieve chl *a*, SP, and $a_{\mathrm{CDOM}}(\lambda)$. The NN construction is based on 550,000 simulated entries from a forward model built using HYDROLIGHT radiative transfer code and a bio-optical model which relates IOPs to optically active constituent concentrations (Doerffer and Schiller 2007). The Case 2 Boreal processor NN was trained using IOPs from Finnish lakes, and the Eutrophic Lakes was trained using IOPs from Spanish lakes.

Odermatt et al. (2010) used the MERIS C2R processor for the retrieval of chl *a* in six perialpine lakes (including Lake Constance). It was shown that the C2R processer was applicable to oligotrophic and mesotrophic lakes; however, the root mean squared error of derived chl *a* in oligotrophic waters was similar in magnitude to the in situ chl *a* during oligotrophic periods. The authors concluded that the MERIS C2R chl *a* product did not possess the level of accuracy needed to replace traditional in situ monitoring but rather was a complement to it. Of particular note was that the authors found that remotely sensed estimates of chl *a* compared better to depth resolved in situ chl *a* (0–5 m) than to vertically integrated chl *a* (0–20 m) in Lake Constance.

While the MERIS C2R algorithms are applicable to chl *a* estimation in oligotrophic (with limitations discussed above) and mesotrophic lakes (Odermatt et al. 2010), application to lakes with very high chl *a* concentrations associated with cyanobacteria blooms has been less successful (e.g., Matthews et al. 2010; Binding et al. 2011). For example, Matthews et al. (2010) found that MERIS Eutrophic Lake and the standard Level 2 Case 2 coastal water quality retrieval algorithm estimations of chl *a* did not compare well with in situ chl *a* in a small hypertrophic lake. This failure was attributed to errors in atmospheric correction and the fact that the MERIS NN was not trained with comparable IOPs and concentration ranges to those of the lake in question. However, Matthews et al. (2010) showed strong empirical relationships between satellite data and in situ chl *a* and SP (particularly the 708/664-nm reflectance ratio), although these relationships had limited ability to separate reflectance contributions from covarying constituents.

Odermatt et al. (2012a) evaluated MERIS C2R, Eutrophic Lakes, and the WeW (Schroeder et al. 2007) algorithms for the observation of phytoplankton blooms in a stratified eutrophic lake. Estimated chl *a* showed better comparison to in situ chl *a* than those in the study of Matthews et al. (2010); however, in situ chl *a* was much lower in their study. The authors suggested that with some regional calibration for the specific absorption coefficient of phytoplankton, the MERIS algorithms had good potential for monitoring chl *a* in eutrophic lakes. Using vertical measurements of chlorophyll fluorescence, however, Odermatt et al. (2012a) demonstrated that remote sensing estimates of chl *a* are constrained by inability to resolve vertical variations in chl *a*. For monitoring both vertical and horizontal distributions of phytoplankton, the authors recommended complementing remote sensing of chl *a* with vertical measurements of chlorophyll fluorescence.

For remote sensing of chl *a* in turbid eutrophic waters, red and NIR bands have proved to be effective. Gons et al. (2008) demonstrated that MERIS data combined with semi-analytical algorithm using red to NIR ratios and an apparent backscatter coefficient worked well in the estimation of chl *a* in eutrophic Green Bay, Lake Michigan. Gitelson et al. (2008) developed a simple semi-analytical model for estimation of chl *a* in turbid waters, applicable to three bands of MERIS and two bands of MODIS sensors, using red and NIR bands. Their algorithm has application in waters with chl *a* ranging from 10 to 200 μg L^{-1}; however, in oligotrophic lakes the NIR reflectance peak required by the algorithm is not present (Giardino et al. 2007; Odermatt et al. 2008). More recently, Gurlin et al. (2011) showed that simple two-band red-NIR models outperform more complex red-NIR algorithms for retrieving chl *a* using MERIS and MODIS applications to turbid productive waters. While these red-NIR-based algorithms were applicable for estimation of chl *a* in turbid waters, they could not differentiate signals from covariant optically active constituents.

Fluorescence line height (FLH) algorithms use a relative measure of reflectance at chlorophyll fluorescence emission wavelengths (which is not necessarily derived solely from fluorescence). A major advantage of the FLH method is that it can be applied to the top of the atmosphere data, negating the need for atmospheric correction. Gons et al. (2008) demonstrated that FLH algorithms were promising at estimating chl *a* in oligotrophic waters but in practice produced noisy retrievals of chl *a*, likely due to the low chl *a* (0.4–0.8 μg L^{-1}). FLH algorithms have been shown to be applicable to the estimation of phytoplankton phenology in Lake Balaton (Palmer et al. 2015). A 10-year archive of MERIS satellite images was used to estimate chl *a* concentrations. At chl *a* concentrations less than 10 μg L^{-1}, FLH was less accurate locally tuned FUB/WeW processor. Notably the study found that cyanobacterial dominance could be identified by negative FLH values. Using a modified FLH algorithm, Matthews et al. (2012) and Matthews and Odermatt (2015) have also been able to differentiate cyanobacteria dominant waters from those dominated by eukaryote algal species (dinoflagellates/diatoms). These authors use the maximum peak height algorithm (MPH) which calculates the height of a dominant reflection peak within red and near-infrared MERIS bands. As opposed to the FLH algorithm, the MPH algorithm searches for the position and magnitude of the maximum peak in the red and near-infrared mirror spends instead of using a fixed peak. Their study demonstrated that vacuolate cyanobacteria (*Microcystis* *aeruginosa*) displays enhanced chl *a*-specific backscatter. The enhanced chl *a*-specific backscatter displayed by cyanobacteria allows differentiation of cyanobacteria dominant or eukaryote dominant algal biomass, and application of a separate algorithm to each case enables estimation of chl *a* concentration.

In order to obviate or minimise errors in optically active constituent retrieval which occur where multiple constituents contribute significantly to $r_{rs}(\lambda)$, there has been an increased focus on analytical spectral inversion algorithms to estimate concentrations of chl *a*, CDOM, and SP simultaneously (Matthews 2011; Odermatt et al. 2012b). For example, Campbell et al. (2011) applied a bio-optical model to retrieve chl *a*, CDOM, and SP from a freshwater impoundment in Australia using the MERIS satellite platform. They concluded that significant improvements in model accuracy could be achieved using differentially weighted and overdetermined equation systems.

As demonstrated from the literature above, universally applicable algorithms for the remote sensing of optically active constituents have not yet been established, primarily due to spatiotemporal variation in OAC-specific IOPs. Algorithms that return IOPs (as opposed to retrieving OACs) in Case 2 waters have greater flexibility (e.g., Pinkerton et al. 2006; Vanderwoerd and Pasterkamp 2008), allowing for the total absorption and scattering to be decomposed into OACs in separate calculations. Ideally algorithms account for optically complex waters where temporal and spatial variation of OAC IOPs occurs (e.g., Doerffer and Schiller 2007; Brando et al. 2012). Of particular note is the adaptive implementation of the linear matrix inversion method developed by Brando et al. (2012) which was demonstrated on a simulated MODIS dataset for the retrieval of IOPs, OAC concentrations, and CDOM absorption. The authors concluded that the adaptive implementation of linear matrix inversion resulted in greater accuracy of OAC and IOP retrievals, particularly for phytoplankton IOPs and chl *a* concentrations.

In addition to spatiotemporal variation OAC-specific IOPs found within inland waters, the empirically based factors g_0 and g_1 (relating to anisotropy of the downwelling light field and scattering processes within the water) have been shown to possess greater variability in Case 2 waters than in case waters and therefore should be considered as variables (Aurin and Dierssen 2012; Li et al. 2013).

In addition to the remote sensing of chl *a*, there has been an increasing focus on the remote sensing of phycocyanin concentrations as a proxy for cyanobacteria concentration (e.g., Hunter et al. 2010; Li et al. 2012; Mishra and Mishra 2014). Remote sensing investigation of phycocyanin concentration exploits the diagnostic phycocyanin absorption at 620–630 nm and the shoulder at 650 (Dekker et al. 2002a; Metsamaa et al. 2006). However, the spectral specific absorption coefficients of phycocyanin are variable and not consistent within the literature (Li et al. 2015; Mishra and Mishra 2014); therefore, the accuracy of semi-analytical algorithms for the estimation of phycocyanin concentration depends on accurate IOP parameterisation. Additionally phycocyanin diagnostic absorption troughs and shoulders may not appear at chl *a* concentrations below 8–10 μg L^{-1}, meaning that remote sensing may not be suitable as an early warning system for harmful algal blooms (Metsamaa et al. 2006).

14.3.4 Remote Sensing of Chl a in Medium to Large New Zealand Lakes Using MERIS Satellite Imagery

MERIS full resolution imagery (300 m) was accessed and downloaded from NASA's Level 1 and Atmosphere Archive and Distribution System (https://ladsweb.nascom.nasa.gov/). Image processing used the BEAM toolbox (http://www.brockmann-consult.de/cms/web/beam/) which is an open source platform for visualisation processing and analysis of satellite imagery (now superseded with Sentinel Application Platform (SNAP). Processing used the default atmospheric correction and SMILE correction (correction of observed charge-coupled device line tilting with respect to iso-wavelength line of the spectrum). Chlorophyll *a* was estimated with the MERIS regional Case 2 water algorithms.

Due to the low spatial resolution of MERIS, water quality estimation is limited to medium (greater than 100 ha) and large lakes. However, the low spatial resolution enables higher revisit capability and image capture over large areas. For example, most of New Zealand can be captured in two MERIS scenes, every 3 days. The application of remote sensing for the estimation of water quality enables a broad spatial coverage (Fig. 14.4) and allows water quality estimation of remote lakes which are not frequently monitored. As has been emphasised previously in the chapter, water quality estimations are not generally globally applicable and usually need some form of regionalisation. This is most likely the case in New Zealand glacial fed lakes, such as Lake Pukaki, where optical properties are dominated by glacial flour and display extreme reflectance (Gallegos et al. 2008). However, algorithms and software used to create these chl *a* estimations present a significant step toward national and global application or remotely estimated water quality in Case 2 inland and coastal waters.

14.3.5 Medium Resolution Applications of Remote Sensing for Water Quality Retrieval

In a recent review of empirical algorithms (including some bio-optical algorithms) for the remote sensing of inland and coastal water quality (Matthews 2011), 27 studies were identified which used broadband Landsat ETM or TM satellite data to retrieve chl *a*, turbidity, SP, or Secchi depth. The review showed that these water quality parameters can in some instances be obtained with high coefficients of determination with satellite reflectance (e.g., $r^2 = 0.99$ for the retrieval of SP (Dekker et al. 2002b) and $r^2 = 0.99$ for the retrieval of chl *a* (Giardino et al. 2001)). Despite the potential of the Landsat satellite to estimate water quality, there are only a small number of examples of remote sensing of inland water quality over landscape scales (e.g., Lillesand et al. 1983; Dekker et al. 2001; Koponen 2004; Allan et al. 2015; Rose et al. 2017), and even fewer applications over a time series of images (e.g., Dekker et al. 2001; Olmanson et al. 2008; Lobo et al. 2015; Allan et al.

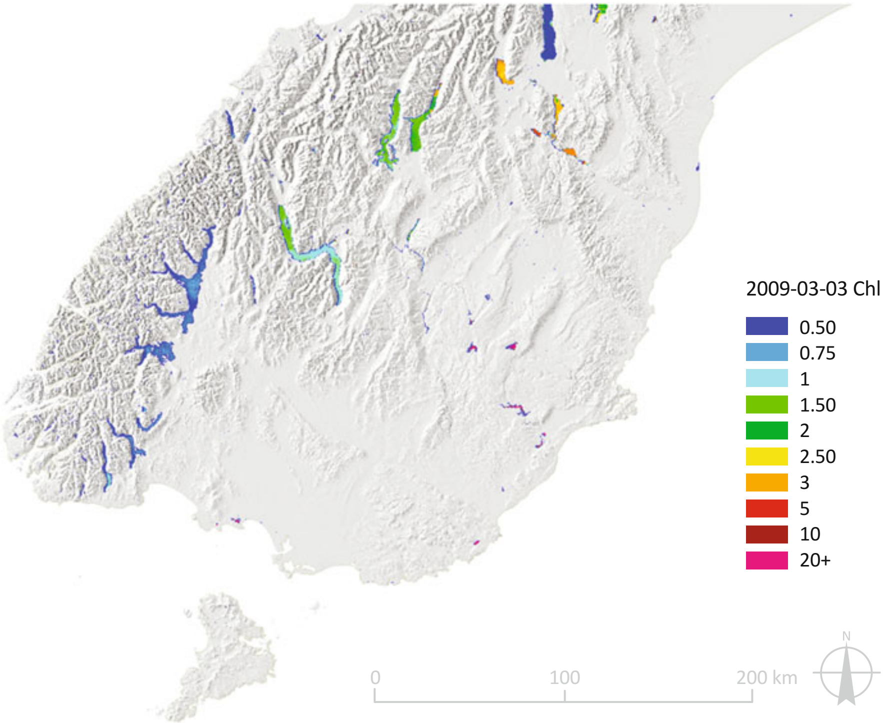

Fig. 14.4 Chlorophyll *a* (μg L^{-1}) concentrations in southern South Island, New Zealand, lakes, estimated using Case-2 regional processor

2015; Rose et al. 2017). For an application of Landsat satellite data for remote sensing of an algal bloom, see Feature Box 14.2.

Box 14.2 The Prospect For Algal Bloom Early Warning Systems Using Remote Sensing

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Cyanobacterial (blue-green algal) blooms are a particular increasing concern in inland waters across the globe. Some species produce potent toxins that pose a major hazard to human health, livestock, wildlife, and the aquatic environment. Traditional field monitoring for algal bloom detection involves identification and cell counting following WHO guidelines (Chorus and Bartram 1999). However, while reliable, use of these methods imparts a lag

(continued)

Box 14.2 (continued)
time and can be limited in spatial extent. Countries in both the developed and developing world need solutions to overcome the vast areas and number of water bodies required to be monitored; satellites and other forms of remote sensing may provide a complement to current approaches to the problem. At the heart of remote sensing approaches to the problem of bloom detection lies the estimation of turbidity or pigment concentrations. For Australian waters, we have proposed a chlorophyll-based alerting system based on Australian and WHO guidelines on cell counts and volumes (Table 14.2). Chlorophyll is a useful "starting index" for determining potential bloom formation using remote sensing; it may inform managers of the need to investigate in the field whether or not the bloom is associated with potentially toxic cyanobacterial species. More sophisticated approaches will determine the presence of phycocyanin in the remotely sensed signal but that requires sensors with sufficient spectral resolution. Recently launched satellite sensors (Sentinel 2, Landsat 8) offer the potential of high-resolution, wide scale, and frequent monitoring of water quality in inland water bodies of a range of sizes in support of early algal bloom alerts for water managers (e.g., Lymburner et al. 2016). But sophisticated systems are required to handle the frequent image acquisition and rapid processing required to deliver a warning system with low latency; for many agencies this can be a significant barrier to the adoption of remotely sensed methods. The recent emergence of "data cube" concepts (Purss et al. 2015) offers a solution. For example, the Australian Geoscience Data Cube (AGDC) serves standardised and calibrated satellite Landsat image data via high-performance data and computing structures. For the state of New South Wales, we have built an AEWS where in near real time, Landsat images are processed to inland water quality outputs (turbidity in the first instance). Data are presented to water managers through a visualisation interface where the turbidity data are translated to relevant bloom alert levels. The interface allows for visualisation at the regional scale to provide a rapid NSW state-wide overview of algal alert status, or at the scale of the individual water body, to allow the determination of spatial bloom dynamics (see Fig. 14.5 as an example with Landsat 7 and 8 turbidity images). Current data can be displayed within the context of historical data. The generic approach developed here for turbidity will be suitable for all optical water quality products into the future. As new satellite sensor data (e.g., Sentinel 2) become available in "analysis ready" form, the power of algal bloom alerting will significantly increase. Current approaches are based on semi-empirical algorithms but future work will adopt the adaptive linear matrix algorithm (aLMI), developed by CSIRO for simultaneous estimation of optically active water quality parameters from remotely sensed data in water bodies with varying water types.

Table 14.2 Proposed algal bloom alert levels based on chlorophyll concentrations and comparison to current Australian Government National Guidelines on cell counts and volumes

Alert level	Chlorophyll (μg Chl L^{-1})	Counts (cells/mL)	Volume (mm^3 L^{-1})
Green	<20	≥500 to <5000	>0.04 to <0.4
Amber	>20–50	≥5000 to <50,000	≥0.4 to <10
Red	>50	≥50,000	≥10

Of the Landsat studies cited above, only a few examine the underlying physical basis for the development of statistical algorithms (e.g., Dekker and Peters 1993; Gitelson et al. 1995; Brivio et al. 1997; Dekker et al. 2002b; Allan et al. 2015). The study of Allan et al. (2011) is typical of many empirically based Landsat studies of remote sensing of water quality, and while useful for determining spatial distributions of chl *a* at landscape scale, any derived algorithms have limited applicability in time beyond the spatial domain used in calibration.

More complex empirical algorithms such as linear mixture modelling (LMM) (Tyler et al. 2006) attempt to differentiate colour producing constituents (CPCs). Linear mixture modelling (Tyler et al. 2006) and spectral decomposition algorithms (Oyama et al. 2007, 2009) applied to Landsat have shown considerable promise in differentiating SP and chl *a*, and it is surprising that there have been few other applications of this method, possibly due to difficulty in reproducing the methodology.

Broadband remote sensing instruments such as Landsat can simultaneously record average spectral slopes of both positive and negative radiance values, as opposed to hyperspectral data (Bukata et al. 1995), but there have also been successful applications of bio-optical models for single water quality parameter retrieval using broadband sensors where the optically active constituent of interest dominates reflectance. For example, bio-optical models have been used to map horizontal distributions of SP and chl *a* concentrations in lakes using Landsat and Spot satellite data (Dekker et al. 2002b; Allan et al. 2015). The advantage of bio-optical models is that in situ data are not needed at the time of image capture, allowing for multi-site and multi-sensor comparisons over time. Dekker et al. (2002b), for example, found that semi-analytical algorithms for SP were more reliable and temporally robust than empirical algorithms.

14.3.6 High-Resolution Applications of Remote Sensing for Water Quality Retrieval

The potential of high spatial resolution remote sensing imagery for monitoring water quality has been subject to less research than low- and medium-resolution applications, mainly due to the cost-prohibitive commercial status of high-resolution imagery. However, it has been shown that empirical algorithms applicable to Landsat data (Kloiber et al. 2002) for remote sensing of Secchi depth are applicable to high spatial

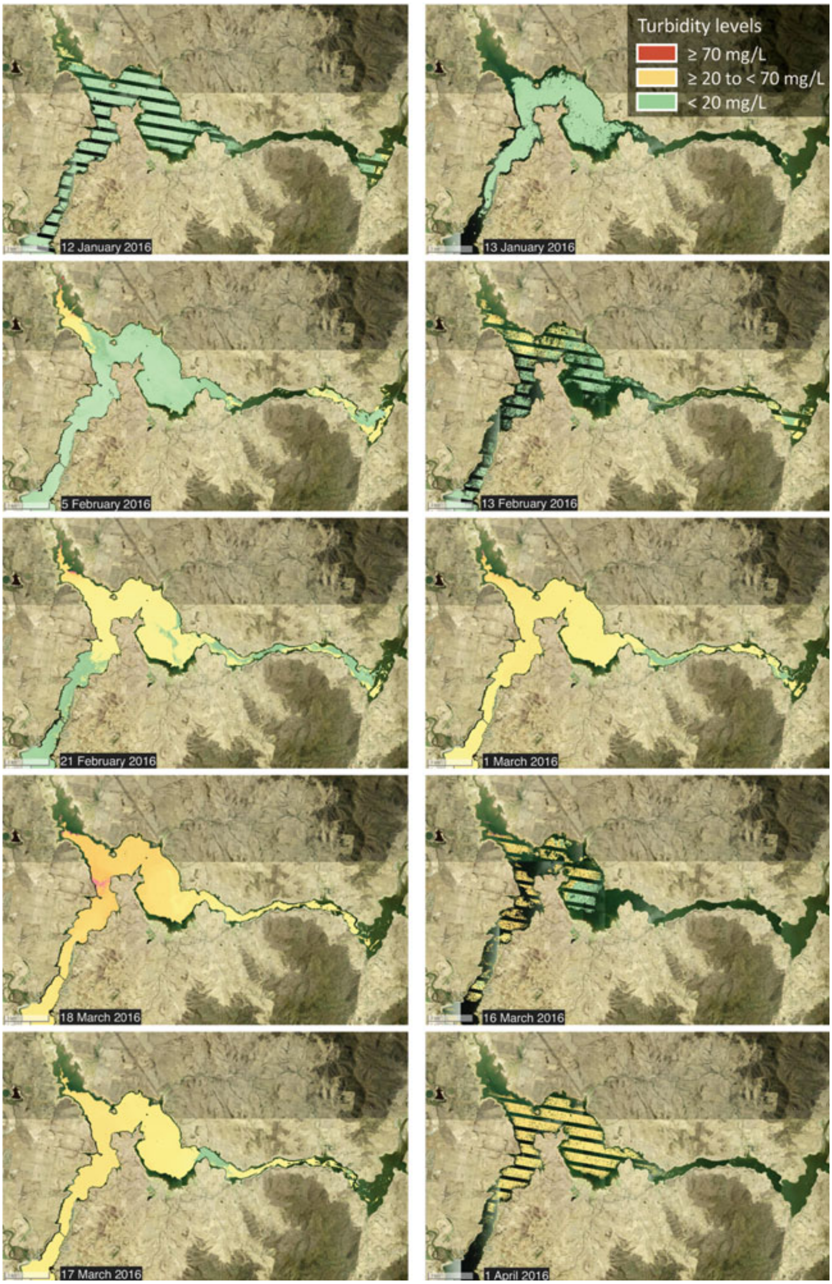

Fig. 14.5 Time series of Landsat 7 and 8 turbidity images for Lake Hume using the algal bloom visualisation system, covering the summer to autumn period mid-January to early July 2016. Turbidity levels are coloured on the basis of green, amber, and red algal alert status, depicted bottom right. Turbidity in this deep reservoir is primarily driven by changes in phytoplankton concentrations. The time series shows the development of an algal bloom from late February through March and April. This bloom provided the "seed" to stimulate a bloom in the River Murray downstream of the reservoir which affected some ~1600 km of river for 3 months to June 2016. A full time series of

resolution satellite data (4 m resolution) from IKONOS (Sawaya et al. 2003). The authors estimated Secchi depth in Minnesota lakes with a standard error of 0.22 m. The higher spatial resolution of IKONOS imagery allowed estimation of water bodies greater than 0.08 ha, as opposed to 1.5 ha using Landsat imagery.

Hyperspectral applications of remote sensing have benefited from the use of in situ (Matthews et al. 2012; Li et al. 2014; Mishra and Mishra 2014), airborne (Olmanson et al. 2013; Li et al. 2014), or satellite-based imaging spectrometry and hyperspectral data (Brezonik et al. 2015; Ryan et al. 2014). Hyperspectral data can be used to identify diagnostic absorption or scattering signatures of algal pigments, CDOM or SP. Hyperspectral data can also be resampled to match bandwidths of multispectral satellite data, which facilitates algorithm development. The disadvantage of acquiring spectral data is that a large volume of data needs to be stored.

Advanced airborne-based imagers possess both high spatial resolution and high spectral resolution allowing application of more semi-analytical and complex empirical algorithms such as partial-least-squares (PLS) regression to enable water quality estimation in optically complex aquatic environments. For example, the Portable Remote Imaging SpectroMeter (PRISM) has a spatial resolution of 2.6 m and has been applied to estimate dissolved organic carbon, chl *a*, and turbidity in the San Francisco Bay-Delta Estuary (Fichot et al. 2016).

14.4 Conclusions and Future Prospects

Remote sensing can aid lake restoration in a number of ways. Remote sensing enables potentially more precise water quality estimation in spatially heterogeneous lakes. Remote sensing can increase the spatial coverage of monitoring, enabling water quality estimation in previously unmonitored lakes, and can be used to estimate temporal trends, potentially providing a sustainable reference condition for creation of restoration scenarios.

For remote sensing of optically active water constituents, there is no single definitive remote sensing solution for any lake or group of lakes and the method needs to be tailored to the lake size, optical complexity of the system, the spectral and spatial resolution of the remote sensor, and the desired outputs. Algorithms for remote sensing of optically active water constituents can be divided into analytical, semi-analytical, or empirical methods. Empirical methods are applicable where there is a simple relationship between the optically active constituent of interest (e.g., chl

Fig. 14.5 (continued) turbidities from January 2015 is also depicted; the solid line represents the median concentration (as total suspended solids in mg/L) and the range the full range of turbidities measured spatially. The reservoir sits in an overlap region of Landsat orbits which means images may be acquired within 24 h. Striping is evident in the Landsat 7 data owing to the SLC issue associated with this sensor. Other gaps in the data are due to the presence of clouds and varying water extent associated with changes in water level.

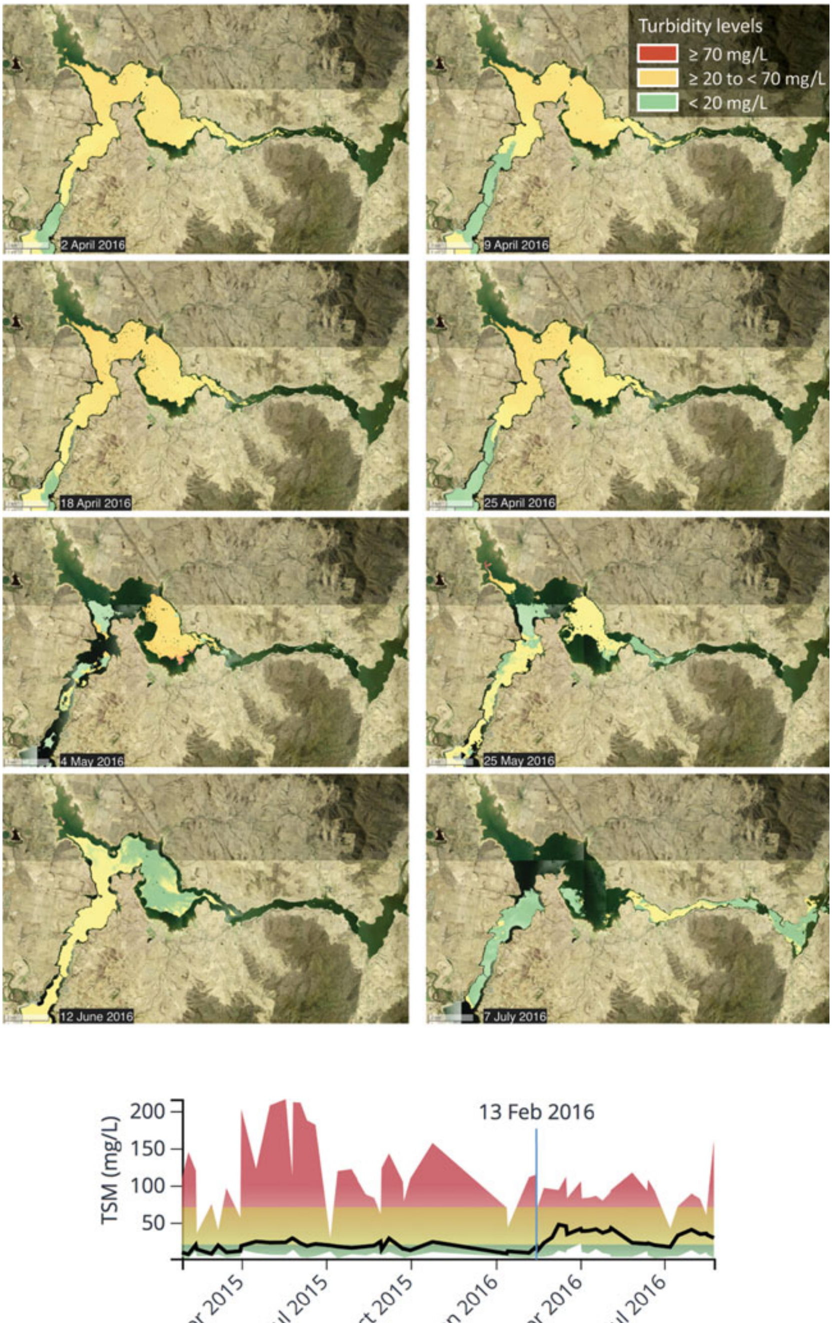

Fig. 14.5 (continued)

a or SP). Empirical methods targeting specific water quality parameters are therefore usually limited to lakes where only one water quality variable dominates reflectance. This empirical approach requires in situ samples to be taken near the time of image capture for a few representative lakes (e.g., 10–15) within the image. However, with atmospheric corrections applied to images, these estimations can be extended to other images without in situ samples, in a semi-quantitative fashion. Semi-analytical algorithms can often be applied in place of empirical algorithms and have a number of advantages. Semi-analytical algorithms can be developed independently of in situ samples, are applicable to multiple satellite sensors, and have greater spatiotemporal applicability. Semi-analytical algorithms can be designed to determine more than one water quality parameter simultaneously; however, a suitable satellite sensor is required which has the spectral resolution to separate fine reflectance features. Such satellites include the low spatial resolution ocean colour satellites such as MODIS and MERIS. The disadvantages of low spatial resolution are offset, however, by the high temporal resolution (daily imagery in the case of MODIS) and spectral resolution. Most analytical methods do not require in situ data (except for validation), but do require parameterisation with inherent optical properties.

Remote sensing remains the only method to quantitatively assess water quality over large areas simultaneously and has the potential to allow monitoring of water quality globally. However, remote sensing should not be considered as a replacement for traditional monitoring techniques, but is complementary, as it can only derive information on optically active water quality constituents. Some studies have attempted to use remote sensing to predict other water quality characteristics, such as cholera outbreaks, which cannot be directly remotely sensed. Studies like this depend on correlations between optically active substances and the target end points. Thus, information regarding concentration of variables such as nutrients or pollutants cannot be directly inferred, but can occasionally be estimated if correlated with optically active substances. In addition, remote sensing does not capture variations of optically active components deeper in the water column, which means that relevant phenomena such as deep chlorophyll maxima are not included in assessments of biomass and productivity of aquatic systems.

Recent technological developments and remote sensing technology have allowed remote sensing to focus from deriving chl *a* in Case 1 oceanic waters to deriving chl *a*, CDOM, and SP concentration in optically complex Case 2 inland and coastal waters. Along with sensor technology, water quality retrieval algorithms have also become more sophisticated. The development of sensors with moderate spatial resolution and high spectral resolution, such as MODIS, has facilitated the continued development of bio-optical algorithms for remote sensing of Case 2 waters. The Visible Infrared Imaging Radiometer Suite (VIIRS) on-board the National Preparatory Project (NPP) satellite (launched 28 October 2011) is designed to improve upon the MODIS data record, with increased spatial resolution (750 m) while maintaining similar spectral resolution.

The Landsat satellite offers the longest continuous optical record of the Earth's surface, spanning more than 40 years. While it has limited spectral resolution and limited signal-to-noise ratio (which limits temporal stability), the sensor has proved exceedingly useful for many applications, including remote sensing of large and small lakes. The launch of Landsat 8 on February 11, 2013, which supports the Operational Land Imager (OLI), has improved quantisation and spectral resolution over past Landsat satellites. The combination of data from OLI and VIIRS sensors will improve water quality and remote sensing solutions for small and large lakes, ensuring further expansion of NASA-based remote sensing of water quality. The recent launch of the ESA's Sentinel 2A (June 2015) and 2B (March 2017) provides multispectral data at 10 m resolution, which will allow the application of algorithms previously applied to the Landsat series at a much higher spatial resolution and shorter return time of 5 days. The transition of remote sensing imagery from commercial to open source (e.g., Landsat and MERIS) continues with the Sentinel series of satellites.

As with imagery, the development of open source GIS and Remote Sensing software and programming languages will allow for greater application of remote sensing for water quality. Of particular note is the growth of the use of Python for GIS and remote sensing. Python is an open source, high-level object-oriented programming language. Python is now the main scripting language for commercial remote sensing applications including ArcMap, and python scripts can now be called from Interactive Data Language (IDL), which is one of the leading commercial scripting languages used for remote sensing. In addition to open source software languages such as Python, space agencies are often providing open source image processing software along with open access imagery. An excellent example of this is the BEAM project and its successor SNAP which support image processing of numerous satellites using a graphical user interface or batch processing scripting languages. The graphical user interface allows for a logical and easy-to-use software platform and provides an excellent introduction to remote sensing for new users. The software also includes a Python API allowing users to modify or extend SNAP's image processing functions. The open source GIS and remote sensing platform, QGIS/GRASS, also supports the use of Python scripting. QGIS is arguably the world's leading open source GIS platform and now integrates many remote sensing tools originally available in GRASS, in addition to remote sensing plug-ins developed by a growing user group.

An exciting new advancement in remote sensing is the development of Unmanned Aerial Vehicles (UAV) or drone technology coupled with compact imaging sensors, reviewed by Colomina and Molina (2014). With continued technological development, the cost of the technology is decreasing and, along with the trend of miniaturisation, sensor and drone capability is set to increase in the twenty-first century. Unmanned Aerial Vehicle-based remote sensing has now reached a point to support generation of geo-information products applicable to remote sensing water quality. However, current UAV-based remote sensing require more financial and scientific input, specifically due to the requirement of specific image processing routines, which are often not open source. However, in the near future financial and scientific effort will be reduced with continued technological development of hardware and software and development of open source code and software.

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Chapter 15
Empowering Indigenous Community Engagement and Approaches in Lake Restoration: An Āotearoa-New Zealand Perspective

Erica K. Williams, Erina M. Watene-Rawiri, and Gail T. Tipa

Abstract The Treaty of Waitangi forms the underlying foundation of the Crown–Māori relationship with regard to freshwater resources in Āotearoa-New Zealand. While there is no "one" Māori world view, there are principles and values that establish and reinforce whānau, hapū, rūnanga and iwi identity, and their responsibilities and rights to manage and use natural resources, including lakes. Lake restoration approaches that are grounded in tikanga Māori and Māori values and perspectives, and are co-designed to be responsive to the needs and aspirations of Māori, will ensure that outcomes are useful and of benefit to the participating indigenous community. The resulting outcomes are more likely to strengthen and add value to existing community initiatives, thus increasing efficiencies when capacity and capability across different expertise is in demand. This requires a commitment (by agencies and funders) to move beyond conventional understandings of who is "qualified" to engage in lake research and restoration initiatives. While hapū, rūnanga and iwi undoubtedly benefit from having their members qualify by being active participants in lake research and restoration efforts, in this chapter we emphasise the need for a more holistic approach that recognises and empowers whānau to engage as co-governors, co-leaders, researchers, as knowledge holders and as teachers. A truly collaborative lake restoration programme will provide multiple roles for Māori, including the development and implementation of monitoring and evaluation approaches.

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15.1 About This Chapter

Many whānau (family groupings) have, and continue to work tirelessly to protect, restore and rehabilitate our taonga and the cultural, social, ecological and economic services provided by Āotearoa-New Zealand's lake ecosystems to their local communities, the nation and our future generations. This chapter aims to provide an insight into Te Ao Māori (Māori world view) and the cultural values, uses, and opportunities which underpin the aspirations of whānau, hapū (tribal grouping that consists of whānau), and iwi (extended tribal grouping that consists of whānau and hapū), in lake restoration, and the processes through which they should be engaged. We encourage readers to source the cited references for a more in-depth understanding of Māori cosmology, principles, concepts and the tribal and catchment histories touched on in this chapter.

Māori have particular interests in fresh water, having traditional and cultural connections with freshwater resources, as well as economic interests. Water is a taonga of paramount importance with attendant rights, interests and responsibilities. The Treaty of Waitangi (Te Tiriti o Waitangi) forms the underlying foundation of the Crown–Māori relationship with regard to freshwater resources (Iwi Advisory Group (Freshwater) 2015). In June 2009 the New Zealand government announced the New Start for Fresh Water strategy, setting out the new direction for the management of fresh water and the intended process for progressing policy development. A national hui (gathering) on freshwater was held in December 2009 to bring iwi and hapū representatives together to discuss the rights and expectations of iwi and hapū concerning freshwater. A keynote presentation delivered by the Honourable Dr Pita Sharples at this hui is referenced throughout this chapter to illustrate the desire of Māori for tenable and long-term solutions in respect of the management of our freshwater resources.

A map is provided to show the locations of the waterways discussed in this chapter (Fig. 15.1). In Te Reo Māori (the Māori language), either a macron or a double vowel is used to indicate long vowels. As Waikato-Tainui use a double vowel we have followed this protocol when we discuss lakes in the Waikato-Tainui rohe (tribal area). Macrons have been used elsewhere. A glossary is provided at the end of this chapter.

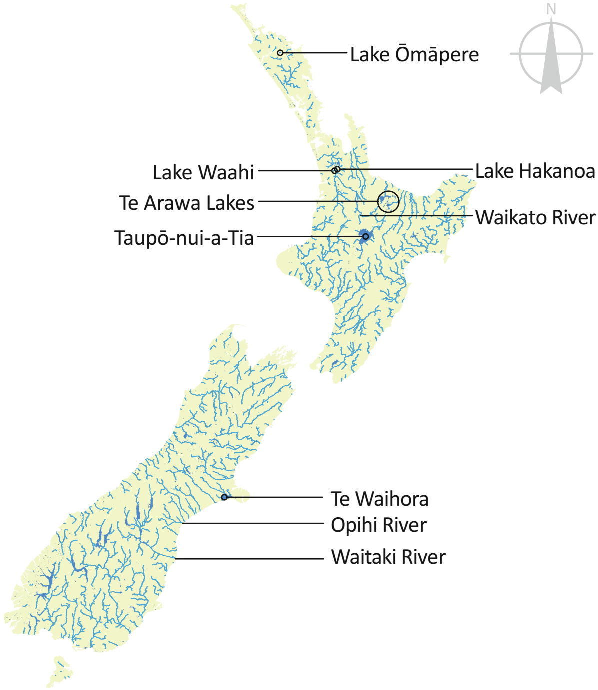

Fig. 15.1 Locations of the Āotearoa-New Zealand lake catchments and waterways featured in this chapter (Source: Erika Mackay)

15.2 Introduction

> Kei te ora te wai, kei te ora te whenua, kei te ora te tangata
>
> (The water is healthy, the land and the people are nourished)
>
> This is a truth which speaks across the universe, reflecting that the health of life on and within Papatūānuku is directly related to the wellbeing of our waters. (Pita Sharples, December 2009)

Āotearoa-New Zealand's lakes are core national assets providing significant cultural, economic, social, and environmental benefits. Competition for the use of these freshwater resources is intensifying, and many of our lake and river ecosystems are now degraded, reducing their inherent values and services they provide for society. The health, integrity and sustainability of ancestral lands, waters, forests, mahinga kai (food gathering place), wāhi tapu (sacred place) and nohoanga (dwelling place) are critical social and biophysical determinants of the health and wellbeing of ecosystems, Māori people and Māori communities (Ngā Pae o Te Māramatanga undated) as illustrated by the whakataukī (proverb) above used by the Honorable Pita Sharples to open his keynote speech at the Iwi Māori National Summit on Freshwater Management in 2009.

The linkages between the natural world and Māori, the indigenous peoples of Āotearoa-New Zealand, are explained through genealogy, tribal narratives and mythology. Collectively, these relationships are known as whakapapa (connections, genealogies), and they represent how Māori place themselves within the world, and how they understand and interact with everything around them (Environment Foundation 2018). Rivers, lakes and streams are intimately bound to people through whakapapa and are a fundamental tenant of Māori personal and tribal identity. Māori have a special relationship with Papatūānuku (the Earth Mother) and her resources—as an integral part of the natural order and as recipients of her bounty, rather than as controllers and exploiters of their environment. Papatūānuku is to be treated with reverence, love and responsibility rather than abuse and misuse (Marsden 2003). Therefore, for Māori the environment and associated natural resources are taonga (significant treasure), and how they engage with it is crucial to their integrity, sense of unique culture and ongoing ability to keep tikanga (correct procedure, custom, lore) and practices alive (e.g. Te Rūnanga o Ngāi Tahu 2013) (for example, see Case Study 15.1).

15.3 Māori Values and Practices

> We all know that providing for a stronger Māori voice in decision-making over water is an excellent way to make progress. Progress will be rapid if the ethic of kaitiakitanga is welcomed to the centre of the debate around freshwater management. And in turn, the contribution that tāngata whenua can make towards sustainably managing our water resources will be of benefit to all New Zealanders. (Pita Sharples, December 2009)

Māori have an intricate, holistic and interconnected relationship with the natural world and its resources. While there is no "one" Māori world view, there are principles and values that establish and reinforce whānau, hapū, rūnanga (tribal assembly) and iwi identity, their responsibilities and rights to manage and use natural resources, including lakes (Table 15.1). To sustain their mana (prestige, authority, status), kaitiaki (guardians) are bound to do everything they can to preserve and restore the mauri (life force) of their environment. Mauri, an internal energy or life force derived from whakapapa is an essential essence sustaining all forms of life. Thus it provides life, vitality and energy to all living things and is the binding force that links the physical to the spiritual worlds. It denotes a health and spirit that permeates all living and non-living things and damage or contamination to the environment is therefore damage to or loss of mauri (Awatere and Harmsworth 2014).

Te Wai Māori (2008) established as part of the Māori Fisheries Settlement to advance Māori interests in freshwater fisheries, explains that Māori regulated fresh waters and freshwater fisheries through a series of adaptive and considered actions by tohunga (expert, specialist), kaitiaki, whānau and hapū, as a vital resource for pā (traditional settlement) and kāinga (home, village). These actions were developed in response to intergenerational observational learnings, and practices to maintain sustainability of use for current and future generations. Complex relationships, tohu and understandings of ecosystems were utilised to inform management options, including the use of rāhui (a kind of prohibition) for temporary restrictions and tapu (restriction) for more permanent bans on the use of resources. An example of a rāhui

Table 15.1 Examples of how Māori values and important Māori environmental concepts form the basis for Māori perspectives when seeking to assess and understand ecosystems, and how a degraded lake ecosystem can impact Māori practices (adapted from Stewart et al. 2014)

Te Waihora whānau value	Brief description	Example of how this value is expressed by Te Waihora whānau (Stewart et al. 2014)	Examples of how a degraded lake environment can impact Te Waihora whānau values (Stewart et al. 2014)
Whakapapa	Connection, lineage, or genealogy between humans and ecosystems and all flora and fauna. Māori seek to understand the total environment or whole system and its connections through whakapapa, not just part of these systems, and their perspective is holistic and integrated (Harmsworth and Awatere 2013)	Whānau members who participated in the Te Waihora Contaminants in Mahinga Kai study (Stewart et al. 2014) describe whakapapa ties to Te Waihora which are maintained through continued use of the area. Such links confirm and encourage the relationship between individuals, the hapū and the iwi, and the lands and waters of Te Waihora. Whakapapa connections with Te Waihora are evident in the following ways: through collective ownership of the lakebed of Te Waihora, and a role in collaborative management, as well as an evolving role of co-governance; through the names of tūpuna (ancestors) or important iwi/hapū events that have been given to geographical locations and landmarks; through the restoration/ revitalisation of traditional practices, such as mahinga kai, within whānau	Environmental degradation and contamination issues could potentially limit the success of cultural revitalisation initiatives
Whanaungatanga	Whanaungatanga refers to the reciprocal support relationship between members of the same whānau, hapū and iwi	As well as a place to work whānau talk about Te Waihora as a place to play, explore, socialise and bond (Stewart et al. 2014). Wild kai	Changing practices, driven in part by perceived adverse effects of contamination, are having a detrimental effect on whānau. For

(continued)

Table 15.1 (continued)

Te Waihora whānau value	Brief description	Example of how this value is expressed by Te Waihora whānau (Stewart et al. 2014)	Examples of how a degraded lake environment can impact Te Waihora whānau values (Stewart et al. 2014)
		(food) gathering activities, (along with most other activities on Te Waihora) provided a forum for whakawhanaungatanga amongst different generations—where events, people, flora and fauna, atua (deities), kawa (protocols, rituals) and tikanga associated with Te Waihora were recounted and woven together	example, one interviewee advised that they no longer eat wild kai from the lake, and would no longer share it with others because of the perceived health risk
Mātauranga Māori	Mātauranga Māori is a holistic perspective encompassing all aspects of knowledge and seeks to understand the relationships between all component parts and their interconnections to gain an understanding of the whole system. It is based on its own principles, frameworks, classification systems, explanations and terminology. Similarly to western knowledge, mātauranga Māori is a dynamic and evolving knowledge system and has both qualitative and quantitative aspects (Tipa et al. in press)	Through the interaction of different generations and wider whānau networks, each of which has experienced different periods and conditions of environmental health, Ngāi Tahu whānau have a broad and often detailed knowledge of the social, cultural, environmental, and economic history of the Te Waihora area, and of their own family's connection to the lake. These teachings embody the knowledge of how to gather, prepare, preserve, distribute, and employ mahinga kai in gatherings and ceremonies—learning from kaumātua (elders). For many, knowledge transmission is of the utmost value to sustaining their community, their culture	With fewer areas open for harvest, the number of healthy mahinga kai populations declining, the amount of wild kai being introduced into the sharing network is dwindling. Whānau members, particularly kaumātua, are no longer able to obtain the types and amounts of kai that they desire, and many confirmed that some species are now only eaten on special occasions. Knowledge may be lost over the generations

(continued)

Table 15.1 (continued)

Te Waihora whānau value	Brief description	Example of how this value is expressed by Te Waihora whānau (Stewart et al. 2014)	Examples of how a degraded lake environment can impact Te Waihora whānau values (Stewart et al. 2014)
Manaakitanga	Hospitality, show kindness and respect to, look after. By nurturing and caring for our natural environment we ensure that it is able to provide us with the resources we require to provide hospitality to our manuhiri, kaumātua and whānau members (Awatere and Harmsworth 2014)	The fundamental components of manaakitanga (the process of showing respect, generosity and caring for others) are to be able to manaaki (to take care of) those in the takiwā (area, district) through the provision of safe resources, such as wild kai being safe to eat, and water being of a quality that is safe for use (e.g. drinking water) and interaction (e.g. swimming or setting nets). Another aspect centres on use of natural resources for which the lake is renowned at gatherings (i.e. culturally important ceremonies). Such gatherings are the best way to maintain and reinforce ties with other hapū members and members of the wider community. Gatherings may be especially important to kaumātua, many of whom obtain much of their wild kai at these events throughout the year	Lower availability (abundance) and loss of access directly influences mahinga kai gathering practices, and therefore manaakitanga. When wild kai is not available or accessible, and/or are contaminated, other foods are substituted, usually processed foods. These foods do not hold the same significance as traditional foods which may lead to whānau feeling a sense of loss. Further some whānau may not be able to afford it financially. In a culture where the importance of sharing food is fundamental, the insult of not being able to provide kai or potentially sharing contaminated kai is a heavy burden to bear
Kaitiakitanga	The exercise of customary custodianship, in a manner that incorporates spiritual matters, by those who hold mana whenua (mana held by local people) status for a particular area or resource	In Stewart et al. (2014), interviewees expressed extremely strong concern for the state of Te Waihora—they believe it is degraded and in a state that is unacceptable. This concern comes from a number of	Kaitiakitanga is an intergenerational responsibility. This is achieved through those who are mandated as kaitiaki utilising rules, management techniques and strategies developed from a

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Table 15.1 (continued)

Te Waihora whānau value	Brief description	Example of how this value is expressed by Te Waihora whānau (Stewart et al. 2014)	Examples of how a degraded lake environment can impact Te Waihora whānau values (Stewart et al. 2014)
		perspectives emanating from the concept and practice of kaitiakitanga (exercising customary custodianship), which encompasses the management, use, and nurturing of environments in ways that protect mauri while supporting the cultural health of mana whenua. Whānau want to be actively involved in authoritative and legitimate management of Te Waihora to sustain both its intrinsic and instrumental values	long-term and widespread tacit association with the environment which are consistent with mana whenua values, beliefs and cultural norms. If kaitiakitanga is not effective this may lead to whānau feeling a sense of loss

to preserve the tuna (eel) resource has been recorded from the Waikato region in the 1850s. It is said that the tuna availability in two main riverine lakes (Lakes Waahi and Hakanoa) was declining due to over-harvesting. In response to this, the local rangatira (chief, leader) Pōtatau Te Wherowhero placed a rāhui over the whole area for a period of time to prevent over-fishing and enable the stocks to recover. When the rāhui was lifted there was a mass celebration and a haka (ceremonial dance) to lift the tapu and make the lake noa (free from tapu) again, which is how Lake Hakanoa was named. Additionally, Raahui Pookeka is the original Māori place name for the Huntly area where these lakes are located as a reference to this time.

Water formed part of an undivided entity and holistic view of the rohe they controlled, whereby Māori did not sharply distinguish between land and water (foreshore, lakes, lagoons, rivers, swamps) (Waitangi Tribunal 1997). Since colonisation, this Māori concept of water as an undivided entity has been systematically fragmented (Te Wai Māori 2008). The practices of early Pākehā settlers more often than not conflicted with Māori values and practices. A number of activities such as the draining of wetlands, lowering of lakes, transportation of logs down rivers which destroyed pā tuna (eel weirs), building hydro-dams, and the introduction of trout, which decimated indigenous species of fish, destroyed Māori rights, interests and traditional practices (e.g. White 1998; Bargh 2006; Coombes 2007; McDowall 2011). For example, as native fish species have declined, out of necessity Ngāti Tūwharetoa have turned to trout as mahinga kai. Trout have therefore become a valued supplement to whānau and marae dining tables as well as a means to provide for manuhiri (visitors) and to carry out traditional fishing practices. Iwi and Māori still maintain tino rangatiratanga (the power to act with ultimate authority) over water and seek to manage as an undivided and indivisible entity. However, the impact of colonisation and the subsequent imposition of a statutory regime have meant that Māori have had to adapt and develop a completely different way of managing freshwater resources (Te Wai Māori 2008).

The interconnected Māori model of river, lakes, swamps, lagoons, river/lake beds and the adjoining land has been fragmented and replaced with a Pākehā (European New Zealanders) model that has strong rules about land ownership with water being a common commodity. For example, early statutes such as the Water Power Act 1903 vested in the government the "sole right to use water in lakes, falls, rivers or streams" for the purposes of generating electricity. Māori claims to ownership of rivers were disregarded by the Coal Mines Amendment Act 1903 when the government vested the beds of navigable rivers in the Crown despite an assurance earlier in the year that such a provision would be subject to "other rights lawfully held" (Durie 2004). Also the Water and Soil Conservation Act 1967 vested in the Crown the "sole right to dam any river or stream, or to divert or take natural water, or discharge natural water or waste in to any natural water, or use natural water" (Te Wai Māori 2008). The conflicting positions regarding the holistic and equitable management of New Zealand's natural resources continues to create tensions between these two world views (e.g. Prystupa 1998; Coombes 2007).

15.4 Recognising the Rights, Interests and Perspectives of Māori

> The Treaty of Waitangi forms the underlying foundation of the Crown–Māori relationship with regard to freshwater resources. There exists a shared interest and desire for tenable and long-term solutions in respect of the management of freshwater resources. The ability of Māori to act as tangata tiaki, to ensure that water is protected and used wisely, is to a large degree dependent on there being a legal framework that provides a meaningful role for Māori in the governance of freshwater. It is critical that arrangements for the governance and management of freshwater recognise the rights, interests and perspectives of Māori. (Pita Sharples, December 2009)

Treaty of Waitangi settlements between iwi and the Crown have resulted in some assets of strategic importance to the nation being placed under Māori ownership and/or management (Māori Economic Development Panel 2012), many of which are in a degraded state. Many settlements now include cultural redress which is concerned with the protection, restoration or rehabilitation of values, uses and services (and at scales) that have not previously been a strategic priority for research, restoration and monitoring by agencies. Treaty settlements which have cultural redress mechanisms within them with respect to lakes include Ngāi Tahu (Wai 27, Waitangi Tribunal 1991), Ngāti Tūwharetoa (Wai 1200, Waitangi Tribunal

Table 15.2 Provisions specific to lakes extracted from the Ngāi Tahu Claims Settlement Act 1998

Mechanism in Ngāi Tahu Claims Settlement Act 1998	Lakes affected by mechanism
Placename changes	Lakes Grassmere/Kapara Te Hau, Alabaster/Wāwāhi Waka, McKerrow/Whakatipu Waitai, Alpine Lake/Ata Puai, Browning/Whakarewa, Lanthe/Matahi
Vesting of land (under Reserves Act 1977)	Wairewa, Moeraki lake
Statutory acknowledgements	For 26 lakes, excluding coastal lagoons, and Sinclair wetlands
Vesting of lakebed in Te Rūnanga o Ngāi Tahu	Te Waihora, Muriwai (Coopers Lagoon), Mahinapua
Joint management provisions, including joint management planning	Te Waihora, Muriwai (Coopers Lagoon)
Nohoanga	Waitaki—Lake Alexandrina, Lake Benmore, Lake McGregor, Lake Tekapō, Lake Ruataniwha, Lake Ohau; Canterbury—Lake Sumner; Otago—Lake Hawea, Lake Wakatipu, Lake Wanaka; Southland—Lake Manapōuri, Lake Te Anau, Mavora Lakes; West Coast— Lady Lake, Lake Haupiri, Lake Kaniere, Lake Brunner Moana
Vesting of land—fee simple	Sinclair wetlands, leases around Te Waihora
Power to make bylaws	Te Waihora, Mahinapua, Muriwai
Statutory appointments to Boards	Guardians of Lake Wanaka, Guardians of Lake Manapōuri, Monowai, Te Anau

These are provisions that are specific to lakes. We acknowledge that there are other generic provisions that relate to mahinga kai, customary fisheries, taonga species that will also benefit lake management, including restoration

2008) and Te Arawa (Wai 240, Office of Treaty Settlements 2006). Models of co-management such as the Waikato Tainui Deed of Settlement and Te Ture Whaimana, the Vision and Strategy for the Waikato River, set a shared and agreed vision for the restoration of the catchment, while other settlements such as the Te Arawa Lakes Settlement Act provides for ownership of 13 Te Arawa lakebeds and a role in management (see Case Study 15.2).

A range of provisions that recognise the rich multilayered association of whānau with lake environs are found in the Treaty Settlements and range from comprehensive co-governance requirements to reinstatement of traditional place names. In the case of Ngāi Tahu, which has the majority of its papatipu marae on the coast, nohoanga are vitally important for reconnecting with inland waterways and reinstating patterns of use. Reinstatement of place names is another tangible means of re-establishing cultural presence. As an example, in Table 15.2 we list some of the provisions from the Ngāi Tahu Claims Settlement Act 1998 that are specific to South Island lakes.

In many cases Treaty Settlements have been the key drivers in the provision of new innovative approaches and bringing together multiple knowledge systems to inform co-management regarding values, species, catchments, and/or at scales that have not previously been prioritised by agencies. As a result of the Waikato-Tainui Deed of Settlement in relation to the Waikato River, Te Ture Whaimana o te Awa o Waikato (the Vision & Strategy) is the primary direction setting document for the Waikato River catchment (including all lakes), it states:

> Our vision is for a future where a healthy Waikato River sustains abundant life and prosperous communities who, in turn, are all responsible for restoring and protecting the health and wellbeing of the Waikato River, and all it embraces, for generations to come.

In order to realise the Vision the following objectives are being pursued by Waikato-Tainui:

(a) the restoration and protection of the health and well-being of the Waikato River;
(b) the restoration and protection of the relationship of Waikato-Tainui with the Waikato River, including their economic, social, cultural and spiritual relationship;
(c) the integrated, holistic and co-ordinated approach towards decisions that may result in significant adverse effects on the Waikato River;
(d) the adoption of a precautionary approach towards decisions that may result in significant adverse effects on the Waikato River, and in particular those effect that threaten serious or irreversible damage to the Waikato River;
(e) the recognition and avoidance of adverse and potential cumulative effects of activities undertaken both on the Waikato River and within its catchment on the health and well-being of the Waikato River;
(f) the recognition that the Waikato River is degraded and should not be required to absorb further degradation as a result of human activities;
(g) the protection and enhancement of significant sites, fisheries, flora and fauna; and
(h) the application to the above of both mātauranga Māori (knowledge system) and the latest available scientific methods.

Te Ture Whaimana will prevail over any inconsistencies in other policies, plans or processes affecting the Waikato River catchment. Relevant policies, plans and processes (e.g. National Policy Statements issued under the RMA 1991; MfE 2014), Waikato Regional Policy Statement, district plans) cannot be amended so that they are inconsistent with Te Ture Whaimana and must be reviewed and amended, if required, to address any inconsistencies. As a result of the Waikato-Tainui Deed of Settlement the Waikato River Authority and Waikato Clean-up Trust were formed to allocate a $220 million budget over a 30-year timeframe to achieve the Vision and Strategy and embrace a new era of co-management in respect of the Waikato River catchment as an appropriate way to secure the longer-term sustainability, health and well-being of the Waikato River for present and future generations.

Major changes have been seen in governance arrangements for freshwater management in Āotearoa-New Zealand as a result of Treaty Settlements (Tipa et al. 2016). The institutions through which Māori interact with the Crown, organisations and communities are multidimensional and varied across the country. These diverse governance arrangements reflect the imperative for local communities (agencies and tāngata whenua) to dictate the arrangements that are made within their rohe, taking into account differences in capacity, how whānau and hapū choose to organise themselves, and how whānau prefer to interact with local government (Tipa et al. 2016). Crawford and Ostrom (1995) define "institutions" as a rule, norm, or strategy that incentivises certain behaviours. Institutions may be prescribed in legislation, policy or procedure, or they emerge informally as norms, operating practices, or habits. In the past, there was difficulty analysing these arrangements because they are frequently the invisible elements of the policy environment. Over the last 10–15 years Māori have become more familiar with their understanding of institutions, both formal and informal arrangements, and are affecting major changes in the Āotearoa-New Zealand freshwater landscape. For iwi, hapū and rūnanga there is also the challenge of having to manage multiple portfolios (Fig. 15.2), navigate multiple regulatory resource management frameworks (Fig. 15.3), and interact with multiple agencies; therefore, the capacity of parties to collaborate in freshwater decision-making must always be a consideration of environmental restoration, management and governance frameworks.

15.5 Empowering Māori Engagement and Approaches in Lake Restoration

Treaty of Waitangi settlements have proved a key driver for the active implementation of lake rehabilitation actions in recent years (e.g. Te Waihora, Te Arawa Lakes, Taupō-nui-a-Tia). Co-management structures are intended to provide a strong legislative obligation for the government to empower Māori in the management and restoration of Āotearoa-New Zealand's lakes. Iwi and hapū environmental and fisheries management plans, where they exist, should be the first point of call for external parties wishing to engage in research and restoration of a lake catchment.

While as mentioned previously there is no "one" Māori world view, indigenous research frameworks, such as that proposed by Kovach (2009) (Fig. 15.4) can provide guidance to those who are unfamiliar with approaches that seek to empower communities to engage in environmental restoration approaches long term. Important components of Kovach's framework that are applicable to the Āotearoa-New Zealand context include:

1. Researcher preparation (e.g. evidence submitted as part of Treaty Settlements, iwi/hapū environmental management plans, Deeds of Settlement, lake management/restoration strategies, catchment histories and familiarisation hīkoi (field trips) and kanohi-ki-te-kanohi (face-to-face) discussions with mana whenua);
2. Research preparation (e.g. indigenous partners collating and strategising lake management and restoration context to be addressed, co-development of actions plans, hypotheses and questions of mutual benefit);

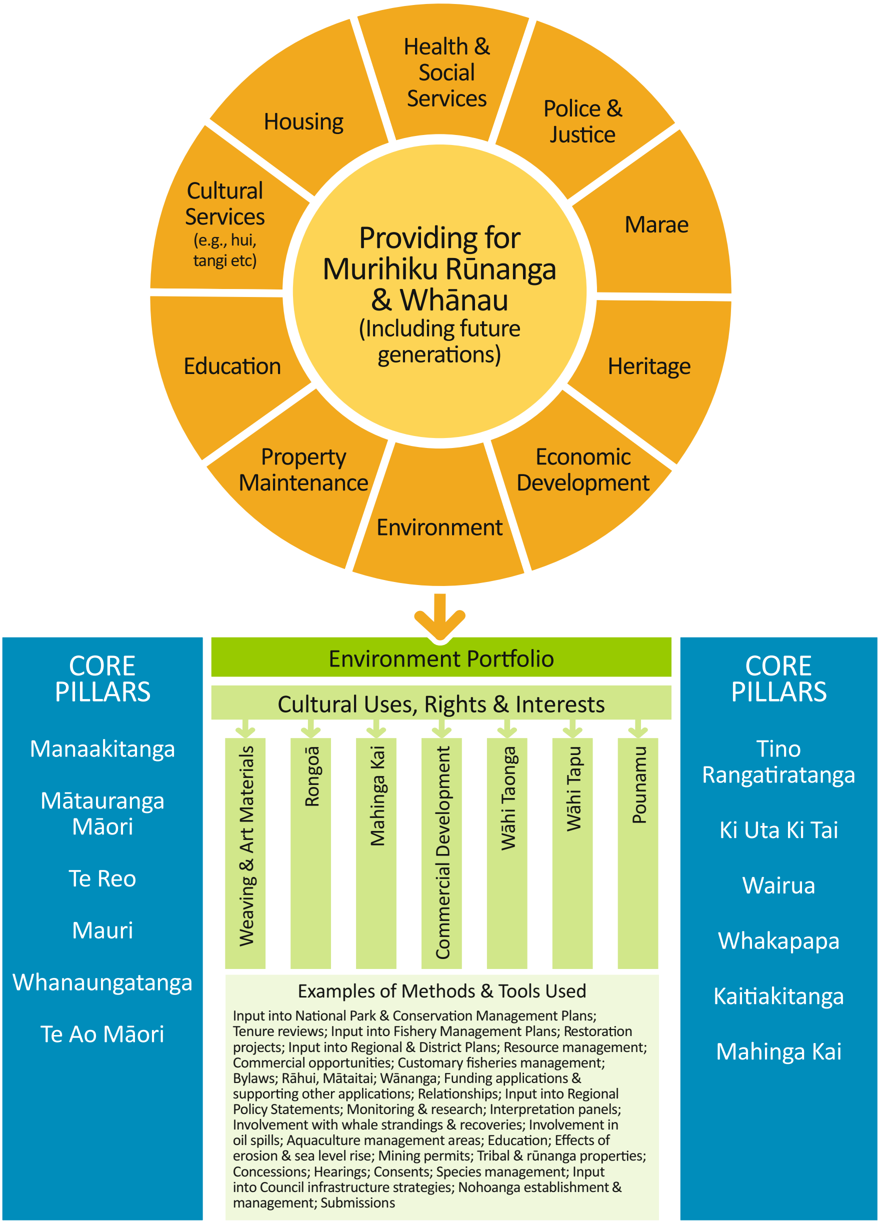

Fig. 15.2 (Left) overview of the multiple portfolios operated by Murihiku papatipu rūnanga; and (Right) the core pillars guiding the implementation of the environment portfolio and some examples of the services facilitated by Papatipu Rūnanga on behalf of, and in collaboration with, Murihuku whānau. The four Murihiku Papatipu Rūnanga are Te Rūnaka o Waihōpai Rūnaka, Te Rūnanga o Awarua, Te Rūnanga o Ōraka-Aparima and Te Rūnanga o Hokonui (Source: Kitson et al. 2014)

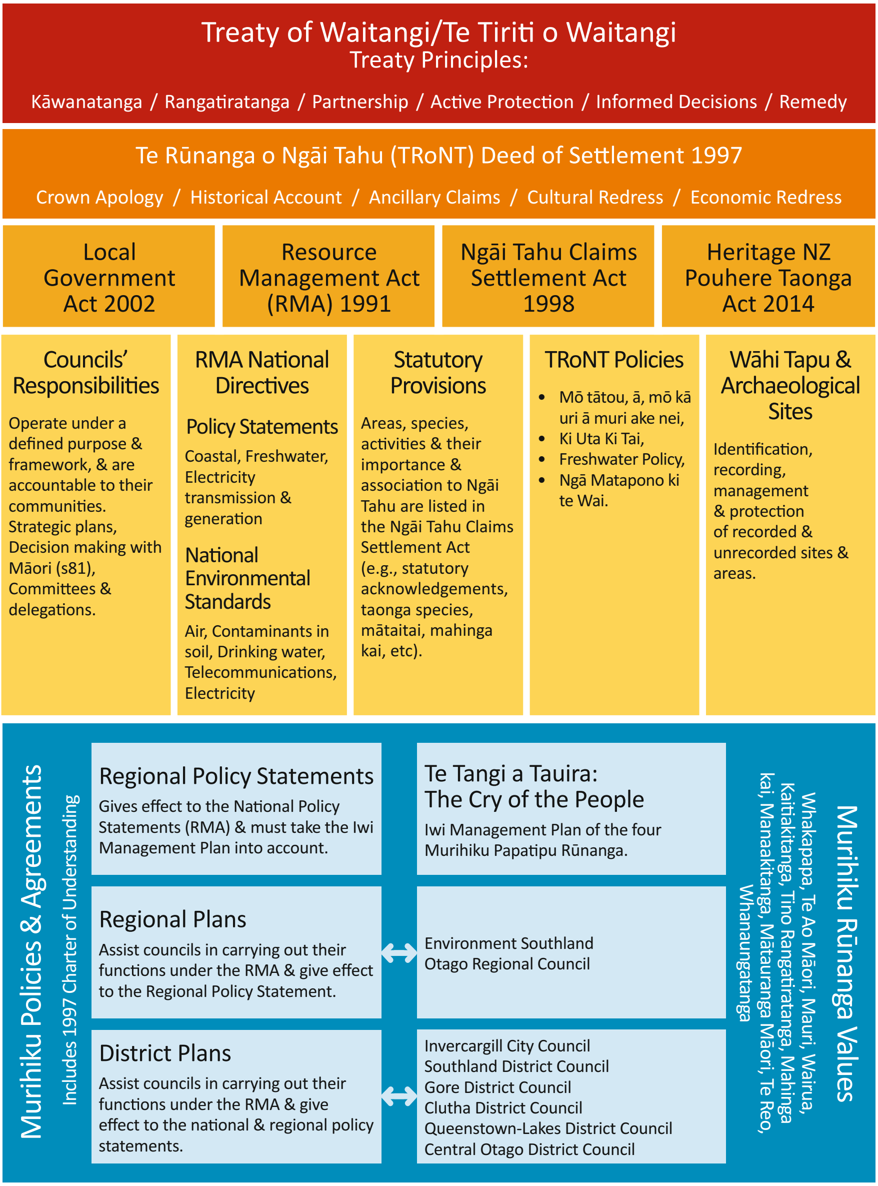

Fig. 15.3 The hierarchy of agreements, acts, policies and plans that informs Te Ao Mārama Incorporated policy development and their expectations for resource management in Murihiku/ Southland. Te Ao Mārama Incorporated has been mandated by the four Murihiku papatipu rūnanga (Te Rūnaka o Waihōpai, Te Rūnanga o Awarua, Te Rūnanga o Ōraka-Aparima, Te Rūnanga o Hokonui) (Source: Te Ao Mārama Inc)

Fig. 15.4 (Centre) A conceptual indigenous research framework adapted from Kovach (2009). This approach has several characteristics, including: tribal epistemology, decolonising and ethical aims, researcher and research preparations, making meaning of the knowledge systems gathered, and giving back. Kovach (2009) describes the epistemology (in her case Nêhiýaw knowledge) as the nest that holds within it the potential properties for the research. (Outer circle) Some of the actions completed in the research programme Ngā Kete o Te Wānanga's progress (as of December 2015) to adapt and implement this indigenous research framework in the Murihiku and Waitaki takiwā

3. Decolonising and ethics (e.g. project principles developed by indigenous partnerships, co-development of methods/approaches, all parties resourced appropriately);
4. Gathering knowledge (e.g. respecting the contributions of multiple knowledge systems, including mātauranga Māori);
5. Making meaning (e.g. co-production of new knowledge relevant to the lake and indigenous community context); and

6. Giving back (e.g. co-delivery of presentations and publications, building joint track records and recognition in all research outputs/communications).

This is not a linear process—it requires periods of reflection and movement backwards and forwards through the process/conceptual framework. An example of how this framework is being applied in the Ministry for Business Innovation and Employment-funded Ngā Kete o te Wānanga: Mātauranga, Science and Freshwater Management research programme which is currently in progress with mana whenua in the Waitaki River, Opihi River and Murihiku/Southland takiwā (tribal area, district) is shown in Fig. 15.4.

15.5.1 Mātauranga Māori

> We have an opportunity as a nation to set the standard high; to work with mātauranga Māori and western science in parallel, investing both with equal importance. (Pita Sharples, December 2009)

The diverse characteristics of Āotearoa-New Zealand's lakes (such as origin, geology, climate, water quality, and composition of flora and fauna) produced very different ecosystems that shaped and influenced the way whānau and hapū engaged with their cultural landscapes around the country. The social and economic well-being of Māori has long relied on the sustainable utilisation (including trade), and management of their local natural resources, where lake fisheries like tuna (freshwater eels), kōura (freshwater crayfish), kōaro and kākahi (freshwater mussels) were once key components of local and regional economies and helped to sustain communities with food and resources (e.g. Hiroa 1921; White 1998; Coombes 2007; McDowall 2011). Utilisation meant whānau engaged with their waterways in all weathers, in different seasons, and observed the environment in its many different states. Changing behaviours in response to changing conditions meant that whānau were well versed in the principles and practice of adaptive management. Through intergenerational interaction a wealth of knowledge and experiences that should be used to inform management and restoration has been generated.

There is an enormous potential for the use of mātauranga Māori to enhance our understanding of lake ecosystems, underpin culturally appropriate restoration approaches, and provide a more holistic and integrated perspective for monitoring, planning and policy. Māori have accumulated qualitative and semi-quantitative information about environmental changes in their lakes and customary resources such as freshwater fisheries, birds, plants and invertebrates, for generations. This information was collected using traditional and adaptive methods which are still relevant today. Although hapū and iwi hold various levels of mātauranga Māori it has rarely been used alongside more quantitative science-based approaches to better understand our lake ecosystems. In the few examples where this has occurred, the partnership between the two knowledge systems has been able to provide innovative, effective and practical tools for monitoring and to formulate powerful hypotheses and prioritisation of future research, restoration and management (for example, see Case Study 15.3).

15.5.2 Well-Being and Culturally Relevant Spatial Scales

> This mindset, this world view, which is essentially the notion that all of our natural resources are inter-related, and intricately woven across our economic, cultural and spiritual aspirations, are part of the unique signature tāngata whenua have brought to this debate. (Pita Sharples, December 2009)

The Māori world view acknowledges a natural order to the universe, a balance or equilibrium, and that when part of this system shifts, the entire system is put out of balance. The diversity of life is embellished in this world view through the interrelationship of all living things as dependent on each other, and Māori seek to understand the total system and not just parts of it (Harmsworth and Awatere 2013). For generations Māori have emphasised the necessity of considering the environment in its entirety, a concept referred to as Ki Uta Ki Tai (Te Rūnanga o Ngāi Tahu 2003). Ki Uta Ki Tai refers to the passage of waters from their source, through a network of tributaries onto lower floodplains, to the interface with the coast and out to the sea. This Māori resource management framework reflects that resources are connected, from the mountains to the sea, and must be managed as such. Within this framework, the overall health of a waterway can be affected by the deterioration at one point of its length, that is, what happens at one point can affect all parts of the catchment (Kitson et al. 2014). This concept is used by Ngāi Tahu to describe their holistic understanding of aquatic ecosystems and how the health and well-being of the people is intrinsically linked to that of the natural environment. The significance of that relationship, and therefore the implications of lake degradation, has been articulated as (Henwood and Henwood 2011):

> [the] use of the lake is determined by the health of the lake, the health of the lake and the health and of the people are intertwined.

Furthermore, this concept reflects that we belong to the environment and are only borrowing the resources from our generations that are yet to come. It is considered our duty to leave the environment in as good as or even better condition than received from our tūpuna (Ngāi Tahu ki Murihiku 2008).

Te Roopu Taiao o Utakura was established in response to the declining state of their local environment. Their rohe includes the Utakura River and its tributaries, from the outflow of Lake Ōmāpere to Paremata near the entrance of the Hokianga Harbour. They are motivated by the kaupapa (theme, philosophy) to restore, protect and enhance their freshwater environment and fisheries—Ma uta ki tai. Implicit in their approach is the holistic, connectedness of people and places through the flow of water, the health of water systems and their resources, and the health and well-being of people as a consequence (Henwood and Henwood 2011; Henwood and McCreanor 2013) (see Case Study 15.4).

Māori well-being has long relied on the sustainable use, conservation and management of lake ecosystems. Taonga species like tuna, kōura and kākahi are integral to Māori socio-ecological systems and once sustained local and regional economies with food and resources (e.g. Hiroa 1921; McDowall 2011). These species support well-being through on--going creation and maintenance of mātauranga Māori, intergenerational knowledge transfer, strengthening connections between whānau, marae, hapū and iwi, and with valued features of cultural landscapes (e.g. Panelli and Tipa 2007, 2008; Stewart et al. 2014). As these species are also vital for maintaining ecosystem integrity and function, due to their cultural and ecological importance these species fit characteristics described by Garibaldi and Turner (2004) as Cultural Keystone Species (CKS) (Noble et al. 2016). Cultural Keystone (Taonga) Species are fundamental in the customary practices and identities of Indigenous communities and as such they must be actively involved in their restoration and management to balance competing values, needs and opportunities. Indigenous peoples around the world are concerned about the decline of many CKS which has affected their subsistence economies, and their ability to sustain cultural fishing practices and knowledge. There are many examples of Indigenous peoples driving the improved management, restoration and conservation of lake fisheries, including kōura (see Case Study 15.3, Kusabs et al. 2015a), freshwater eels (e.g. McKinnon 2007; Williams et al. 2013; Waikato Raupatu River Trust 2014) and salmon (e.g. see Feature Box 15.1; Fraser et al. 2006; Galbreath et al. 2014). Noble et al. (2016) contend that with the recognition of Indigenous cultural keystone species, we take the first step towards improving freshwater access rights, and our management of freshwaters as resilient social-ecological systems for the benefit of all stakeholders.

Box 15.1 Fisheries Management in the Duck Valley Indian Reservation, USA

Jinwon Seo
Department of Fish, Wildlife, and Parks, Shoshone-Paiute Tribes, Owyhee, NV, USA

There are approximately 310 Indian reservations and 566 recognised tribes in the United States. By area (1172.02 km^2), the Duck Valley Indian Reservation of the Shoshone-Paiute Tribes (SPT) is one of the 50 largest Indian reservations in size. It was established on April 16, 1877, and is almost evenly divided between two states, Idaho (50.2%) and Nevada (49.8%). It was deemed sufficient to provide ample fish and wildlife resources for subsistence. The East Fork Owyhee River, located in northeastern Nevada, is a tributary of the Snake River and is the southernmost drainage to the Columbia River. The river runs through the reservation and produced native populations of anadromous fish including Chinook salmon and steelhead which historically migrated hundreds of miles from the Pacific Ocean, making these fishes a significant part of historical tribal food resources and culture. However, the Northwest Power and Conservation Council (NPCC) identified the reservation as a "blocked area" for anadromous fish in 1995 due to numerous dams and diversion structures which had been constructed across the Columbia, Snake

(continued)

Box 15.1 (continued)
and Owyhee Rivers. Currently, the tribes believe that potential capacity exists today for their spawning and rearing success in the river. It is critically important to restore salmon and steelhead for the tribes' fishery resources and culture. The tribes assessed habitat conditions of the East Fork Owyhee River in 2013 through field surveys and confirmed the existence of significant carrying capacity. The tribes believe as a first step towards restoring salmon to the reservation, that it would be beneficial to develop a trap-and-haul program to transport adult salmon to the reservation from dams on the Lower Snake River. This would provide an initial test of habitat capability and also provide a tribal fishery that was lost over 80 years ago. Three reservoirs have been constructed and managed as a put-and-take trout fishery on the reservation as mitigation for federal dams: Mountain View Reservoir (~633 acres, 256 ha) located in Idaho, Sheep Creek Reservoir (~788 acres) and Lake Billy Shaw (~430 acres) located in Nevada. The "Duck Valley Reservoir Fish Stocking and Operation & Maintenance" program, funded by the Bonneville Power Administration has been managed by the Fish, Wildlife, & Parks Department and is an important program for the SPT as it partially mitigates for the loss of anadromous fishes on the reservation. The program provides tribal subsistence fishery, and generates precious economic revenue from fishing/camping permits to the general public (Fig. 15.5).

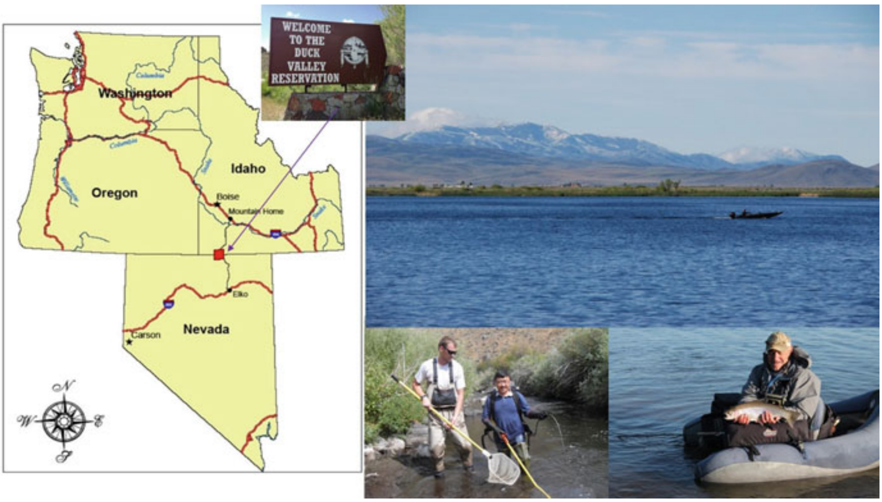

Fig. 15.5 Location of the Duck Valley Indian Reservation (left) and photos (right) showing its use as a fishery

15.5.3 Beyond Liaison and Consultation: Enduring Partnerships and Communities of Learning

> We need an environment which encourages iwi to participate; and I'm not just talking about with post-settlement tribes, or sticking rigidly to a narrow interpretation of the statute. We want to see a far more open relationship from regional councils and local authorities, to actively working with iwi in water management. The aspiration of co-management will only come about if iwi have built the capacity to fully participate in these processes. This may require further investment in education to spread understanding. Not everyone has the awareness and specialist knowledge that can help to shape our management of our natural resources. We need to build capability. (Pita Sharples, December 2009)

A commitment to the establishment and management of relationships is paramount to the facilitation of successful collaborations between Māori, researchers and government agencies (e.g. Allen et al. 2009; Lowe et al. 2009). Partners who commit to relationships that are established with the view to extend beyond the life of a particular contract are more likely to be asked back to help access ongoing opportunities to build on projects and interests within the community. In time these collaborative partnerships will foster the networks of people and communities of learning (Robson et al. 2009) that are required to realise the long term goals and objectives of large lake restoration programmes. This is an unavoidably time intensive process requiring critical input by knowledge- and cultural-bridgers. Mutual needs (shared questions) and mutual benefits are potent drivers for collaboration and knowledge co-production (for example, see Case Study 15.5). The identification of questions and goals of common interest is an important precondition for the establishment of a dialogue between different knowledge-holders (e.g. Wilcox et al. 2008; Lowe et al. 2009; Gyopari 2015).

This can be challenging to achieve in reality when the expectations (some of which are time bound) of the various parties involved are different (funders, partners, project team members, interested parties, etc.). For example, Satterfield et al. (2005) express that:

> Western business is driven by deadlines and the need to progress and make decision by certain times. Whereas Māori thought is not driven by any sort of time deadlines which can be very frustrating and sometimes unrealistic, but their focus is entirely on the process—doing it right.

Approaches that are grounded in tikanga Māori, Māori values and perspectives, and are co-designed to be responsive to their needs and aspirations, will ensure that the outcomes will be useful and of benefit to the participating indigenous community. The resulting outcomes are also more likely to strengthen and add value to existing community initiatives, thus increasing efficiencies when capacity and capability across different expertise is in demand (see Case Study 15.6).

It also requires a reassessment of who the knowledge holders are and a commitment (by agencies and funders) to move beyond the conventional understandings of who is "qualified" to engage in research and restoration initiatives. While whānau, hapū, rūnanga and iwi undoubtedly benefit from having their members qualify by being active participants in research programmes and restoration efforts, in this chapter we have emphasised the need for a more holistic approach that recognises and empowers whānau to engage as co-governors, co-leaders, researchers, as knowledge holders and as teachers. A truly collaborative programme will provide for multiple roles. Lake restoration programmes should also include a strong focus on the intergenerational capacity and training needs of Māori. The Lake Waahi Restoration project (see Case Study 15.5) is a good example of where this approach was adopted. All of the physical restoration works (fencing, clearing, planting, pest control, etc.) was undertaken by local marae members who were specifically trained for the tasks. The collective buy-in to the project was so much greater as the marae whānau already have an inherent obligation as kaitiaki to look after the lake. The project provided an opportunity for "kaitiakitanga in practice" and opportunities for whānau and rangatahi (youth) to gain qualifications while undertaking restoration activities were actively facilitated.

15.6 National Policy Statement for Freshwater Management: Potential Challenges for Lake Restoration Programmes

New collaborative approaches are being used to engage Māori under the National Policy Statement for Freshwater Management (NPS-FM) (MfE 2014) and new Resource Management Act 2014 reforms. The NPS-FM seeks to safeguard the life-supporting capacity, ecosystem processes and indigenous species including their associated ecosystems of fresh water; and the health of people and communities, at least as affected by secondary contact with fresh water. Sustainable management of the use and development of land, and of discharges of contaminants to maintain or improve freshwater quality are some of the ways being sought to achieve the objectives of the NPS-FM. The NPS-FM requires that councils set objectives for freshwater bodies that reflect a range of aspirations and values, and that limits for flow, water quantity and water quality are in turn set to ensure that the objectives are achieved (limit setting). These processes are currently grappling with the challenge of how Māori principles, values, aspirations and objectives are reflected in freshwater management approaches in Āotearoa-New Zealand.

As regional councils undertake their limit setting processes in response to the NPS-FM, lake restoration initiatives will be captured within these processes. What needs to be agreed between Māori and agencies is what is to be factored into the NPS-FM process and how. For example, in 2015 Environment Canterbury completed the limit setting process for the Selwyn-Te Waihora catchment via a plan change (McCallum-Clark et al. 2014). This limit setting process was impacted by Whakaora Te Waihora, an extensive cultural and ecological restoration programme between Ngāi Tahu and Environment Canterbury for Te Waihora, the largest lake in Canterbury and an important link in the chain of coastal lagoons and estuaries along the east coast of the South Island. Ngāi Tahu and Environment Canterbury have a shared commitment to the restoration and rejuvenation of the mauri and ecosystem health of Te Waihora so that it continues to provide current and future generations with the sustenance, identity and enjoyment that it has in the past. Case Study 15.7 suggests that the planning framework for one lake was developed based on the future state which in turn was dependent upon a number of independent (and potentially unfunded) variables occurring. Unless there is a robust monitoring programme and a commitment to regular reviews of the modelling that underpins limit setting processes, developing frameworks based on the potential future state represents a risk, especially where the funding needed to realise the future state has not been secured. As these processes are rolled out throughout Āotearoa-New Zealand whānau, hapū, rūnanga and iwi will need to be cognisant of how agencies accommodate lake restoration programmes within limit setting processes. While restoration has to be a consideration when setting limits, it needs to be clear to Māori whether regional councils are developing statutory planning frameworks based on:

1. the current state and the restoration that has been completed; or
2. the potential state after restoration for which funding is guaranteed is completed; or
3. the potential state after longer-term restoration actions are completed (whether or not funding is secured).

Challenges are also encountered by mana whenua when fit for purpose methodologies are not being used to inform measures that are suitable for assessing the state of their values. For example, some regional councils are using the trophic level index (TLI) (Burns et al. 2000) in their limit setting processes. In the limit setting processes for Te Waihora and for the Waitaki River catchment, mana whenua had to understand and assess the impacts of different contaminant loads on the TLI and in turn assess the effects on their mahinga kai values. The TLI has been used in the limit setting for both Waitaki and Te Waihora but in two different contexts. In the case of Te Waihora, a scenario was based upon delivery of a specific TLI. The impacts of this TLI-led scenario were then assessed against a number of outcomes. The second example, the case of the Waitaki, was to set a specific TLI as an outcome. The different scenarios were evaluated based on their ability to deliver this outcome. In circumstances like these it is a challenge for some whānau to understand the impact of marginal differences, for example, a change from 2.6–2.8 to 3.0, on their rights and interests, especially their mahinga kai. The components of the TLI (e.g. total nitrogen, total phosphorus, water clarity and chlorophyll-*a*) are not well linked or communicated in the terms of mahinga kai interests and aspirations. Developing alternative and complementary assessment tools that are meaningful to Māori and enable mātauranga Māori to inform freshwater management, building on approaches such as the tau kōura (Kusabs and Quinn 2009; Kusabs et al. 2015a) and Cultural Health Index (Tipa and Tierney 2003, 2006), are fundamental if whānau, hapū and iwi are to meaningfully engage in limit setting processes.

15.7 Conclusions and Future Prospects

Whether post- or pre-settlement, mana whenua have continually lobbied for protection of Āotearoa-New Zealand's lakes and surrounding environments, and are committed to seeking a reversal in the state of degradation. Collaboration with Māori is essential for the success of lake research and restoration projects in Āotearoa-New Zealand and Māori partners are entitled to participate in formulating the strategy, undertaking the research, and working towards achieving outcomes. Engagement with whānau, marae, hapū, rūnanga and iwi at multiple levels should conform to Māori ethical guidelines recognising the principles, tikanga, and the rights, roles and responsibilities of Māori. Collaboration will build consensus, foster communities of learning, aid decision-making and will contribute to social and environmental change by fostering long-term commitment and understanding of all the parties involved.

Case Study 15.1 Understanding and Acknowledging Whakapapa—The Interrelationships Between People and a Tribal Taonga

The story of Te Waihora (also now known as Lake Ellesmere) begins with the arrival of Waitaha. Ngāi Tahu oral traditions tell of Rākaihautū, who beached his waka at Whakatū (Nelson) and divided the new arrivals in two, with his son Te Rakihouia taking one party to explore the east coastline and Rākaihautū taking another southward by an inland route over the Southern Alps/Kā Tiritiri o te Moana. Te Rakihouia discovered the coastal lake (now known as Te Waihora) on his coastal journey south claiming the abundant resources of the area for his father. Hence the lake being named Te Kete Ika a Rākaihautū—the fish basket of Rākaihautū. Generations later, when Ngāti Mamoe arrived from Te Ika a Maui (the North Island) settling among the Waitaha, a prominent man of this tribe, Tutekawa, established his home at Waikākahi (Birdlings Flat), and pronounced Te Waihora as his own hence the lake's second name, Te Kete Ika a Tutekawa—the fish basket of Tutekawa. When Ngāi Tahu came south they proclaimed Orariki, Taumutu, their home and thus the resources of the lake as their own.

These historic travels symbolise and continue to reinforce tribal identity, solidarity and continuity between generations, providing the context underpinning Te Waihora as a tribal taonga. For Ngāi Tahu, Te Waihora has always been a highly valued food source. Historically, the lake and surrounding areas were renowned for its abundance of fish, waterfowl, plants (including medicinal plants), and special muds used for dyeing (Palmer and Goodall 1989; Tau et al. 1990; James 1991; Waitangi Tribunal 1991; Anderson 1998).

Case Study 15.2 Cultural Redress Mechanisms in the Te Arawa Lakes Settlement Act 2006

The Te Arawa Lakes Settlement Act 2006 is the settlement of historical claims relating to Te Arawa lakes, returning ownership of 13 lakebeds to Te Arawa Lakes Trust on behalf of Te Arawa hapū and iwi. The lakes include Rotoehu, Rotomā, Rotoiti/Te Roto kite a Ihenga Ariki ai a Kahumatamomoe, Rotorua/Te Rotorua nui a Kahumatamomoe, Ōkataina/Te Moana i kataina a te Rangitakaroro, Ōkāreka, Rerewhakaaitu, Tarawera, Rotomāhana, Tikitapu, Ngāhewa, Tutaeīnanga, Ngāpouri/Opouri and Ōkaro/Ngakaro. The Deed establishes protocols to enhance good working relationships between Te Arawa and agencies such as Department of Conservation, the Ministry of Primary Industries, the Ministry for Culture and Heritage, and the Ministry for the Environment, on cultural matters of importance to Te Arawa.

The Settlement also establishes in law, the enduring joint management collaboration—the Rotorua Te Arawa Lakes Strategy Group—which is made up of representatives from Te Arawa Lakes Trust, Bay of Plenty Regional Council and Rotorua District Council. The purpose of this collaboration is to coordinate policy and actions to:

> Contribute to promoting the sustainable management of the Te Arawa lakes and their catchments for the use and enjoyment of present and future generations, while recognising and providing for the traditional relationship of Te Arawa with their ancestral lakes.

As part of the Act, the Crown has made regulations that empower the Trustees of the Te Arawa Lakes Trust to manage the customary and recreational harvest of selected fisheries. The Te Arawa Lakes (Fisheries) Regulations (2006) cover non-commercial customary fishing within the Te Arawa fisheries area and do not provide for commercial fishing. The Act provides for the establishment of Komiti Whakahaere (a fisheries management committee established under the regulations) to manage the customary fisheries in accordance with Te Arawa tikanga and kawa. The Mahire Whakahaere or the Te Arawa Lakes Fisheries Plan is required under the regulations to provide for the sustainable management of customary fisheries. The customary fisheries include kōaro (*Galaxias brevipinnis*), tuna (both shortfin *Anguilla australis* and longfin eel *A. dieffenbachii*), common smelt (known locally as “īnanga”, *Retropinna retropinna*) and toitoi or common bully (*Gobiomorphus cotidianus*). The kōura (*Paranephrops planifrons*) and the freshwater mussel, known locally as kākahi (*Echyridella menziesi*) are also important customary species for Te Arawa.

Case Study 15.3 Use of the Tau Kōura in Te Arawa Lakes to Provide Quantitative Information for Use in Customary Fisheries Management

Kōura are a valued kai species and considered a delicacy by Te Arawa. In the past they were a staple food item, and prized for their use as a bartering item (Kusabs and Quinn 2009). Although found in many other freshwater streams and waterways, the Te Arawa lakes and Taupō-nui-a-Tia were considered among the most productive kōura fisheries in New Zealand (Mair 1923; Hiroa 1921; Best 1929). An example of this productivity was at the opening of Tamatekapua Marae at Ohinemutu in 1873, where a reputed 500 rohe (where a rohe was roughly the equivalent of a modern sack) of dried kōura and īnanga were consumed (Hiroa 1921).

A variety of methods are still used to capture kōura in the Te Arawa lakes, including the tau kōura (bundles of packed bracken tied to a rope and placed along the lake bed), the paepae or hao (a dredge net), collecting kōura by hand, rama kōura (night-time spotlighting using small dip nets) and hi kōura. The tau kōura was the favourite traditional fishing method for harvesting kōura in Te Arawa lakes (Hiroa 1921). This method involves the placement of bracken fern bundles (known as whakaweku) on the lake bed for kōura to take refuge in and then retrieving the bundles into a canoe to harvest the kōura. Tumu made out of rewarewa and ponga ferns (*Cyathea dealbata*) marked the fishing grounds and helped to delineate the boundaries of the various hapū and whānau (Hiroa 1921).

Until recently, there was a lack of quantitative information on kōura abundance and ecology which made it difficult for Te Arawa Lakes Trust to manage kōura populations in the Te Arawa lakes. The recent adaption and use of the tau kōura for monitoring (Kusabs 2006; Kusabs and Quinn 2009; Kusabs and Butterworth 2013) and research purposes (Kusabs and Butterworth 2011; Kusabs et al. 2015b) has significantly increased our understanding of kōura populations in the Te Arawa lakes. For example, Kusabs et al. (2015b) found that kōura abundance and distribution in seven Te Arawa lakes was influenced by the combined effects of lake-bed sediments, lake morphology, and hypolimnetic conditions related to trophic state. Sediment particle size was identified as the strongest driver of kōura abundance and biomass, with kōura populations increasing with increasing sediment particle size. Kōura abundance was highest in lakes Rotomā, Rotorua and Rotoiti which had a high proportion of coarse lake bed substrates and low in lakes Ōkāreka, Rotokākahi, Tarawera and Ōkaro where lake bed substrates were composed mainly of mud.

A number of customary management changes arising from the information gathered using the tau kōura are suggested in Kusabs et al. (2015a) to protect and enhance the kōura fishery. Tau kōura have now also been successfully deployed in rivers and streams in the Waikato River catchment and there are also aspirations to utilise adaptations of the tau kōura method to help reintroduce and rehabilitate kōura populations in other lakes around Āotearoa-New Zealand.

Case Study 15.4 Ma Uta Ki Tai Catchment-Wide Approaches to the Restoration of Lake Ōmāpere

Lake Ōmāpere (1197 ha, maximum depth about 2.6 m) is the largest lake in Northland. This lake is of huge significance to Ngāpuhi who regard it as a "food basket" that historically supplied tuna (eel), torewai (freshwater mussels), raupō (bulrush, *Typha orientalis*), kāpūngāwhā (clubrush, *Schoenoplectus tabernaemontani*) and harakeke (flax, *Phormium tenax*) (White 1998). Thousands of katua (migrating tuna) and tautoke (lake resident tuna) were caught each season (White 1998). The lake was an early source of timber and flax fibre, which was exported via the waterways, and also a site for kauri gum digging (White 1998). Ngāpuhi sought to formalise ownership of the lake prior to 1929, but it was not until 1955 that the Māori Land Court vested the lake in the Lake Ōmāpere Trustees on behalf of all Ngāpuhi (i.e. the lake bed, water and resources in the water are owned by Māori) (White 1998). Ngāpuhi have always regarded Lake Ōmāpere as a significant source of food which should be respected. The lake continues to be acknowledged as a highly significant taonga and mahinga kai site for the marae, hapū and iwi which surround it (Lake Ōmāpere Project Management Group 2006). With the main outflow of the lake being the Utakura River which flows out to the Hokianga harbour, the lake catchment also supports many hapū communities who reside downstream along the river and within the Hokianga harbour.

The declining water quality of Lake Ōmāpere has long been of concern for the community. Mana whenua have continually lobbied for protection of Lake Ōmāpere and environs, and are determined to reverse the state of degradation. Activities such as deforestation, establishment of pasture, drainage of wetland areas, lowering of lake water levels and intensification of pastoral farming have all contributed to a steady decline in lake water quality, and in turn, the water quality of the only outlet, the Utakura River. In 1984 dense growths of the invasive weed *Egeria densa* were first documented (Tanner et al. 1986). In 1985 the Northland Area Health Board prohibited water takes from the lake for domestic supply after Kaikohe residents complained of stomach upsets and the lake source was found to have a severe cyanobacterial bloom (Northland Regional Council 2001). Downstream, in upper reaches of the Hokianga Harbour, fish and shellfish became unsafe to eat (Grey 2012). It was apparent that the *Egeria* weed beds had collapsed, leaving Lake Ōmāpere in a turbid,

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Case Study 15.4 (continued)
de-vegetated algal-dominated state for years (Champion and Burns 2001). Grass carp were introduced between in 2000 and 2002 which has contributed to the interruption of the boom–bust cycle of *Egeria* invasion alternating with an algal-dominated state (Henwood and McCreanor 2013).

In 2004, the Lake Ōmāpere Project Management Group (LOPMG) bought together mana whenua, stakeholders, and agencies with responsibilities in the area to identify a restoration and management strategy for the lake. This strategy, based on ma uta ki tai (catchment-wide) approaches to restoration, emphasises interconnectedness and inclusion, both spatially and socially (LOPMG 2006) and aims to improve the health of Lake Ōmāpere by strengthening kaitiakitanga. It outlines a series of practical actions and outcomes which are required to achieve the overarching vision of waiora, where waiora describes the life giving element of the lake which can only be fully realised when it is restored to a healthy condition where the people and the wider environment can benefit. Four central elements are identified as important to achieving waiora:

- *Ma uta ki tai*: Emphasising the total connectedness of the natural environment and acknowledging that Lake Ōmāpere will not be restored through isolated initiatives or a narrow focus.
- *Mātauranga*: Has been viewed broadly as knowledge or teachings which underpin kaitiakitanga. There is an urgent need to focus on knowledge regarding Lake Ōmāpere and its context environmentally, historically and spiritually to enhance the cultural indicators that are appropriate for long term monitoring of the health of the lake and its environment.
- *Rangatiratanga*: Conveys the sense of authority, control and responsibility inherent in the exercising of kaitiakitanga. There is a need to ensure appropriate structures and practices are in place to support the long term restoration of Lake Ōmāpere.
- *Kotahitanga*: Refers to a unity of purpose in relation to Lake Ōmāpere and the importance of the development and maintenance of relationships to support such a purpose.

Case Study 15.5 Capacity Building for Mana whenua Embedded in Lake Restoration Objectives
Lake Waahi is located near the township of Huntly, on the western bank of the Waikato River. The area around Huntly is referred to as Raahui Pookeka and is steeped in Waikato-Tainui history. There are a number of marae in and around Huntly including Waahi Paa (which is a principal paa of Ngaati Mahuta and is

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Case Study 15.5 (continued)
the family residence of the Kaahui Ariki, the paramount family of the Kiingitanga, led by Kiingi Tuheitia), Kaitumutumu Marae, Te Kauri Marae, Te Ohaaki Marae, Hukanui Amuri Paa, Taupiri Marae and Matahuru Marae. The majority of these marae are located within a 2-km radius of Lake Waahi and the Waikato River. Lake Waahi is very significant to mana whenua. The Waahi Stream which connects the lake to the Waikato River has been fished for migrant eels (puhi) for generations. Lake outlets were strategically chosen to build rauwiri/paa tuna (weirs) and capture puhi, as they were much easier to block off. During special marae occasions it is an expectation by those who attended that puhi will be served at meal times. This expectation has built up over generations of tradition and experience during which time the people of Raahui Pookeka have fished the annual migration.

Lake Waahi has poor water clarity due to high concentrations of algae (resulting from high nutrient loads) and high sediment inflows which smother the lake bed and the submerged native plants. This is exacerbated by the presence of koi carp, which enter the lake via Waahi Stream and have led to the decline and collapse of submerged plant communities. The poor water quality has degraded ecological values and the mauri of the lake.

In conjunction with mana whenua and Waikato River stakeholders, in 2011 Genesis Energy commissioned a survey that showed a range of pest fish are present in the lake, including koi carp, goldfish, perch, tench and catfish. This is of concern to tāngata whenua as koi carp are now estimated to make up 80% of the total biomass in the lower Waikato River catchment where they compete with native fish for habitat and destroy submerged plant communities. Encouragingly, the survey also identified a range of native fish species in the lake, including shortfin and longfin eels, mullet, smelt and iinanga. Giant kookopu, banded kookopu and rare mudfish are known to be present within the lake catchment, particularly the tributary streams. These results demonstrate that the lake has good connectivity to the Waikato River via Waahi Stream, despite the presence of flood control gates near its confluence with the Waikato River. It also demonstrates that restorative actions which increase habitat potential and water quality in the lake will be successfully utilised by existing native fish species present. The 2011 survey built on decades of research on Lake Waahi, and the issues impacting the lake are now very well defined, as are the priority restoration actions.

In 2012, Waahi Whaanui Trust, Waikato Raupatu River Trust and Genesis Energy secured WRA funding to complete restoration works around Lake Waahi. The high-level objectives of the Lake Waahi project included the following: (a) A multifaceted and holistic work programme to improve the health and well-being of Lake Waahi through improved riparian and wetland habitat, and pest fish control; (b) To allow Waikato-Tainui, Waahi Whaanui

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Case Study 15.5 (continued)
Trust, and the people of its marae and hapuu, to build capacity in native plant nursery establishment and operation, riparian/wetland enhancement and protection, and ecological and kaitiaki monitoring, through the integration of maatauranga and ecological values and (c) To monitor the habitat enhancement works so that improvements can be quantified, from both Western scientific and kaitiaki perspectives. The project aimed to act as a catalyst for a Waikato-wide approach for enhancing the overall environmental quality of shallow Waikato lakes, and building capacity for taangata whenua.

Case Study 15.6 Mana whenua Driving Collaborative Relationships and Networks
Mana whenua have been the drivers of the collaborative relationships and networks required in the Lake Ōmāpere and Utakura River catchment to extensively fence and plant the riparian lake edges, conduct new research to inform restoration actions, and have adapted culturally specific assessment and monitoring tools (Henwood and Henwood 2011) which will compliment continuing water quality monitoring in assessing the direction and magnitude of response to restoration initiatives. Planting the riparian strip has not only helped absorb nutrient runoff that has played a major part in the degradation and eutrophication of the lake, but also the restoration of native plants is creating a habitat for native birds and beautifying the area.

The partnership between Te Roopu Taiao o Utakura, Ngāpuhi Fisheries Limited and NIWA began in 2008 with a lake fishery assessment that was required by mana whenua to provide a reference point for future tuna monitoring and research. As a result of this study the Lake Ōmāpere Trust placed a rāhui on the commercial harvest of lake tuna populations. Te Roopū Taiao o Utakura then commissioned a nutrient budget for Lake Ōmāpere (Verburg et al. 2012) which showed during the algal-dominated phase a nutrient legacy in the sediments fueled internal regeneration that greatly exceeded loads from the catchment, with eventual export from the lake via the Utakura River and denitrification. In contrast, during the weed bed-dominated state nutrients accumulated in the sediments (Verburg et al. 2012).

Underpinned by the four principles of ma uta ki tai, mātauranga, rangatiratanga and kotahitanga, some of the key factors underpinning the relatively rapid success of mana whenua in progressing lake rehabilitation are: strong leadership, focus on developing intergenerational capacity and capability (of themselves and other Māori and non-Māori practitioners), formation of long term collaborative relationships across multiple organisations,

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Case Study 15.6 (continued)
and development of a publication track record that assists this roopu to secure funding to develop and deliver their own research initiatives.

Henwood and Henwood (2011) report social spill-over benefits for people involved in restoration of Lake Ōmāpere:

> Other positive project outcomes to date relate to relationships, collaborations, capacity and capability. Those involved in the lake project are now part of a large network of people with expertise and an interest in environmental restoration.

Henwood and Henwood (2011) also highlight the cultural benefit of involving Te Roopū Taiao o Utakura with scientific water quality monitoring:

> Recording and dissemination of local environmental information is expected to increase understanding of the role of kaitiakitanga as a social practice, and its relationship to waiora and hauora in sustainable environmental restoration.

Case Study 15.7 Limiting Setting and Lake Restoration
In the Officers Section 42 Report to the panel hearing the Selwyn-Waihora plan change, a considerable portion of the evidence focused on what was planned in the future under the next stage of Whakaora Te Waihora. What is still missing is any commitment to fund Stage 2. In other words, the solutions package that set limits are relying on the interventions of Stage 2 which, depending on funding, may or may not be implemented.

The officer's report to Plan Change 1 stated that:

> Funding will come from public, private and community sources.

The proposed work programme for Stage 2 involves:

- Maintaining the current initiatives and supporting new initiatives to restore habitat and achieve biodiversity enhancement from the initial focus catchments into 40 key catchments including significant drains and waipuna (spring heads), primarily through 800 km of riparian planting, predator pest and weed control and instream remediation and enhancement.
- Continuing to improve current land practices through extended farm management planning, catchment strategies and broadening support for landowners and catchment communities to implement on-farm improvement measures across the whole lake catchment.
- Enhancing lake resilience to absorb legacy impacts by creating significant buffer areas through mid-catchment wetland re-establishment; lake margin

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Case Study 15.7 (continued)

protection; and retirement and enhancement of inlets and embayments for fish refugia, macrophyte establishment and mahinga kai.

- Implementing lake interventions investigated in Phase 1 for lake level control, nutrient processing and in-lake habitat creation—e.g. phytoplankton and nutrient interaction and salinity tolerance—by moving into the next stages of the design and development of lake opening and closing infrastructure, enhanced channelling, floating wetlands, macrophyte beds and in-lake islands.
- Implementing sediment traps, sand wand, and creation of riffle and refugia in streams.

Glossary of Te Reo Māori Used in this Chapter

Haka Ceremonial dance

Hapū Is a tribal grouping that consists of whānau who typically share descent from a common ancestor

Harakeke Flax, *Phormium tenax*

Hauora Healthy

Hīkoi Walk, field trip

Hui Assembly, meeting, gathering

Īnanga Typically refers to *Galaxias maculatus* with the exception of the Te Arawa Lakes where common smelt (*Retropinna retropinna*) is known locally as īnanga

Iwi Is an extended tribal grouping that consists of hapū or whānau who typically share descent from a common ancestor and associate with a distinct territory

Kaahui Ariki The paramount family of the Kiingitanga

Kai Food

Kāinga Home, village

Kaitiaki Guardian

Kaitiakitanga The exercise of customary custodianship, in a manner that incorporates spiritual matters, by those who hold mana whenua status for a particular area or resource. Kaitiakitanga is inextricably linked with concepts such as tino rangatiratanga (without which the practical implementation of kaitiakitanga is constrained or impossible) and taonga (Satterfield et al. 2005)

Kākahi Freshwater mussels (typically *Echyridella menziesi*)

Kanohi-ki-te-kanohi Face to face

Kāpūngāwhā Clubrush, *Schoenoplectus tabernaemontani*

Kaupapa Theme, philosophy, topic

Kawa Ceremonial protocols, rituals

Ki uta ki tai From the mountains to the sea

Kiingi King

Kiingitanga The King movement—a movement which developed in the 1850s, established top stop the loss of land and promote Māori authority, to maintain law and order, and to promote traditional values and culture

Kōaro *Galaxias brevipinnis*

Komiti Whakahaere Is the fisheries management committee established under the Te Arawa Lakes (Fisheries) Regulations (2006) to manage the customary fisheries in accordance with Te Arawa tikanga and kawa

Kōura Freshwater crayfish (used in this chapter to refer to the North Island species, *Paranephrops planifrons*)

Mahire Whakahaere Is the Te Arawa Lakes Fisheries Plan required under the Te Arawa Lakes (Fisheries) Regulations (2006)

Mahi Work, to make, undertaking

Mahinga Kai 1. Is referred to in the National Policy Statement for Freshwater Management 2014 as indigenous freshwater species that have traditionally been used as food, tools, or other resources2. To Ngāi Tahu mahinga kai is used to refer to their interests in traditional food and other natural resources and the places where those resources are obtained, i.e. food-gathering place

Mana Prestige, authority, status

Mana whenua Refers to the mana held by local people who have "demonstrated authority" over land or territory in a particular area, authority which is derived through whakapapa links to that area

Manaakitanga The process of showing respect, generosity and care for others

Manuhiri Guest, visitor

Māori Indigenous people of Āotearoa-New Zealand

Marae Is a place typically in front of a wharenui (meeting house) where the members of whānau, hapū, or iwi meet and engage in pōwhiri (the ceremony of greeting and encounter), and includes associated buildings, such as the wharenui (meeting house) and wharekai (dining room), and surrounding land

Mātauranga Māori Is a holistic perspective encompassing all aspects of knowledge and seeks to understand the relationships between all component parts and their interconnections to gain an understanding of the whole system. It is based on its own principles, frameworks, classification systems, explanations and terminology. Mātauranga Māori is a dynamic and evolving knowledge system and has both qualitative and quantitative aspects

Mauri Essential life force or principle, a quality inherent in all things both animate and inanimate

Murihiku A region of the South Island in Āotearoa-New Zealand. Traditionally it was used to describe the portion of the South Island below the Waitaki River, but now is mostly used to describe the province of Southland

Noa Free from tapu

Nohoanga 1. Dwelling place, abode, encampment2. Are temporary campsites for Ngāi Tahu whānau to use

Ora Alive, well

Pā Traditional settlement

Pā tuna Eel weirs
Pākehā European New Zealanders
Papatipu marae Original Māori land, a meeting place for tāngata whenua, and a focal point for rūnanga
Papatūānuku Mother Earth
Puhi Migrant eel (Lake Waahi)
Rāhui A kind of prohibition. Rāhui were imposed variously to ensure the sustainability of a resource, after waters had been polluted (usually as a consequence of a death) or to reserve a resource for one's use (White 1998)
Rangatahi Youth
Rangatira Chief, leader
Rangatiratanga Self-determination, chieftainship, decision making rights
Raupō Bulrush, *Typha orientalis*
Rohe Tribal area, district, region
Roopu Group
Rūnanga/Rūnaka Tribal assembly, council
Taiao Environment
Takiwā Area, district, region
Tangata tiaki Individuals involved in the guardianship of a resource/area
Tāngata whenua People of the land
Taniwha In Māori mythology, taniwha are beings that live in rivers, caves, or in the sea. They may be considered highly respected kaitiaki (protective guardians) of people and places
Taonga Significant treasure, possessions, something prized
Tapu Restriction. Tapu was used as a way to control how people behaved towards each other and the environment
Tau kōura A Te Arawa-Tūwharetoa traditional method of harvesting kōura
Te Ao Māori Māori world view
Te Reo Māori The Māori language
Te Ture Whaimana o te Awa o Waikato The Vision and Strategy for the Waikato River
Tiaki To protect, guard and look after
Tino Rangatiratanga As absolute power and authority refers to the person or group who has the power to act with ultimate authority when necessary (Satterfield et al. 2005)
Tikanga Māori Correct procedure, custom, lore, method and practice
Te Tiriti o Waitangi The Treaty of Waitangi. An agreement made between the British Crown and about 540 Māori chiefs first signed on 6 February, 1840
Tohu Sign, mark, symptom
Tohunga Expert, specialist
Torewai Freshwater mussel
Tumu Surface-reaching pole (tau kōura) used to mark fishing grounds and delineate boundaries between hapū (Hiroa 1921)
Tuna Freshwater eels

Tūpuna Ancestors
Wāhi tapu Sacred area/place
Waiora Health
Waipuna Spring heads
Waka Canoe
Wairua Spirit
Wānanga Learning
Whakapapa Connection, lineage, genealogy between humans and ecosystems and all flora and fauna. Māori seek to understand the total environment or whole system and its connections through whakapapa, not just part of these systems, and their perspective is holistic and integrated (Harmsworth and Awatere 2013)
Whakataukī Proverb
Whakaweku Bracken fern bundles (as a component of the tau kōura)
Whānau A family group that consists of individuals who typically share a common whakapapa and identify with a common living or recent ancestor
Whanaungatanga Whanaungatanga refers to the reciprocal support relationship between members of the same whānau, hapū and iwi
Whenua Land

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Chapter 16
Applying Citizen Science to Freshwater Ecosystem Restoration

Monica A. Peters, David P. Hamilton, and Chris Eames

Abstract Interest in citizen science is growing globally as environmental degradation continues, information needs increase and value of stronger relationships between the science community and public is recognised. How community volunteers participate in citizen science ranges from solely collecting environmental data to being fully engaged in project design and delivery. In New Zealand, community groups lead diverse environmental restoration projects. Responding to an online questionnaire, 137 groups (from a total of 296) reported carrying out their own monitoring to measure environmental change. While 98 of 239 groups reported an interest in monitoring water quality in the future, current freshwater monitoring activities were reported as limited (33 of 143 groups). Current monitoring centred mostly on stream macroinvertebrate counts. Three case studies are presented that outline how community groups have engaged in collecting water quality data. In contrast, a strong culture of volunteer water quality monitoring exists where programmes are designed to educate participants while also providing data for fundamental research, e.g. in the USA, and for government agency-led environmental decision-making. To encourage wider participation of communities, professional scientists and government agencies in citizen science, principles underpinning the development and implementation of long-term volunteer monitoring programmes are outlined. Stronger community participation in monitoring has the potential to improve both scientific and environmental literacy while building more complete data sets describing trends in freshwater resources. Furthermore, in New Zealand an informed and engaged public is in line with goals of local, regional and national

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government to increase public involvement in freshwater through participatory decision-making.

Keywords Volunteer monitoring · Water quality · Participatory decision-making

16.1 Introduction

A major issue facing humanity in the twenty-first century is the global decline in the quality of freshwater resources (United Nations Educational 2012). Recognising the need to protect and maintain water body health, volunteer water quality monitoring programmes have proliferated across the USA and Canada over the past 40 years. Hundreds of thousands of community volunteers now monitor streams, rivers and lakes in order to understand trends and identify issues (United States Environmental Protection Agency 2012; Kebo and Bunch 2013). Many of these volunteer initiatives began from environmental watchdog programmes where members of the community became advocates for their local freshwater resources (Firehock and West 1995).

These volunteer water quality monitoring programmes now form part of a wider movement known as citizen science. Broadly defined as public participation in scientific investigations (Bonney et al. 2009), monitoring carried out by community volunteers underpins an increasingly diverse range of studies across terrestrial and aquatic ecosystems (Silvertown 2009). Data generated by volunteers may be used to raise public awareness of environmental issues (Savan et al. 2003), contribute to scientific research (Hoyer et al. 2014) or inform environmental management and policy (Haklay 2015). As participants carry out field-based activities in citizen science projects, stewardship of the environment may be encouraged, and participants may also increase their knowledge of ecology (Dickinson et al. 2012). In addition, the potential also exists for increasing participants' knowledge of scientific processes, i.e. scientific literacy (Trumbull et al. 2000; Brossard et al. 2005; Jordan et al. 2011; Crall et al. 2012).

The increasing popularity of citizen science projects and volunteer monitoring programmes can be attributed to a range of factors. With government agency capacity to carry out environmental monitoring reduced due to declining budgets (Kebo and Bunch 2013), trained volunteers have proved cost effective, particularly for collecting long-term data over large areas (Hoyer et al. 2014). As a result, key scientific questions can be addressed that would otherwise lie beyond the resourcing capability of professional organisations and their staff (Carr 2004). Over time, access to simplified methods and training materials developed for volunteers has encouraged wide participation, notably for water quality monitoring (Firehock and West 1995). The growth of Web 2.0-based tools has driven the expansion of citizen science, with smartphone applications (apps) and sensors enabling rapid data collection and dissemination (Teacher et al. 2013). Haklay (2015) also identifies increasing levels of higher education, leisure time, and numbers of able-bodied retirees as factors, though also points out that few project participants comprise minority groups unless specifically targeted.

16.1.1 Participating in Citizen Science

Citizen science projects and volunteer monitoring programmes can be categorised according to the level of participation by volunteers. In contributory or scientist-led projects, for example, the primary role of volunteers is to collect data (Bonney et al. 2009). This approach typically underpins studies where long-term data are required across large distances, such as 'The Great Secchi Dip-In' (http://www.secchidipin.org/). In this programme, more than 2000 water bodies (mostly across the USA and Canada) have been monitored for five or more years by volunteers for trends in transparency. A feature of the contributory approach is that it builds on what community members may already enjoy doing; for example, regular recreational users of water bodies may also collect samples for water quality monitoring or amateur bird watchers may join in on an annual bird count (Silvertown 2009).

Co-created or collaborative approaches enable greater volunteer participation in key activities such as study design, data collection, analysis and reporting (Bonney et al. 2009). An example is researchers working with local communities neighbouring a contaminated site to co-create a project that responds to community information needs (Ramirez-Andreotta et al. 2015).

Full participation by community volunteers in developing and leading projects/programmes, as well as collecting and managing the data, may be labelled as community-led. In this approach, scientists and land managers provide technical input, but the monitoring agenda is primarily directed by the volunteers. These types of projects are widespread in New Zealand, among community groups carrying out environmental restoration-centred projects across a range of ecosystems including freshwater (Peters et al. 2015).

University professionals coordinating graduate students (teaching) and thousands of dedicated citizen scientists have resulted in the collection of reliable long-term water quality data for over 1100 lakes, 175 coastal sites, 120 rivers and 5 springs. LAKEWATCH (a Florida-based community water quality monitoring programme) has used the results to create information circulars (extension) to address lake management issues in an understandable format while maintaining scientific credibility. At the same time, committed researchers maintained a steady stream of publications in peer-reviewed journals (research) showcasing data collected by volunteers in order to achieve the three original LAKEWATCH objectives (Terrell et al. 2000; Bachmann et al. 2012; Bigham 2012; Hoyer et al. 2015).

Two major (of the many) hurdles to the early and continued success of LAKEWATCH are (1) demonstrating to professional groups that trained volunteers are capable of collecting credible data, and (2) maintaining consistent long-term funding. Studies comparing data collected by volunteers with those collected by professionals have ensured LAKEWATCH's accreditation to meet various national standards. Funding is especially critical because trained and committed core staff is needed to work alongside volunteers. Continued successes of the programme make it easier for funding agencies to move money into the programme. There is a vast army of citizen scientists waiting to get involved, and the hope is that the Florida LAKEWATCH experience assists other groups develop successful monitoring programmes (Hoyer et al. 2014).

16.1.2 Data Quality

Despite the widespread use of volunteer data and the longevity of water quality monitoring programmes such as Florida LAKEWATCH and the Lake Sunapee Protective Association (see Feature Boxes 16.1 and 16.2) the quality of data collected by volunteers is frequently questioned. Sources of error commonly include poor study design, inconsistent use of methods, or not using uniform/standardised monitoring equipment. Further errors may stem from volunteers' lack of experience, which can lead to over- or underestimation of size and/or abundance, or inadvertent bias stemming from volunteers' preconceptions (Danielsen et al. 2005; Wiggins et al. 2011).

Box 16.1 Lake Sunapee Protective Association

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Cary Institute of Ecosystem Studies, Millbrook, NY, USA

The Lake Sunapee Protective Association (LSPA) is one of the oldest lake associations in the USA, with a more than 110-year history of preserving and enhancing the environmental quality of the Lake Sunapee watershed and beyond. Located in southern New Hampshire, Lake Sunapee is an oligotrophic lake whose watershed is 80% forested and whose shoreline is rimmed with private cottages. LSPA relies on volunteers supported by 4.5 staff and an annual budget derived from membership and donations to conduct watershed restoration activities and deliver education and outreach programmes to 4000–5000 people per year. Citizen science volunteers (see Fig. 16.1) have gathered over 20 years of water quality monitoring data, and their critical observations have contributed to a better understanding of the ecosystem. One example is when cyanobacterial blooms began to appear in the lake in 2004. Lakeshore residents and monitoring volunteers were the first to notice the appearance of an unusual bloom and different kind of algae in Lake Sunapee. They went looking for help to understand changes underway and found serendipity on their side. The author (Weathers) was on sabbatical with LSPA in 2004 and a student passionate about algae was looking for an internship in the area. It was the start of what has become a major, multi-year, regional study involving many citizens and students working with

(continued)

Box 16.1 (continued)
researchers from area universities and colleges (e.g. Carey et al. 2014). The LSPA research programme also involves the use of high-frequency data from sensors mounted on a buoy in the lake and collaborations with researchers as part of the Global Lake Ecological Observatory Network (GLEON). The research has contributed not only to an improved understanding of the surprising algal blooms in Lake Sunapee—the kind of sound science that underpins management decisions—but also to an understanding of how these blooms may affect nutrient-poor lakes worldwide (Cottingham et al. 2015).

Fig. 16.1 Volunteer measuring water clarity with a Secchi disk. Photo credit: T.R. Eliassen

Box 16.2 Florida LAKEWATCH

Mark V. Hoyer
Florida LAKEWATCH, School of Forest Resources and Conservation, University of Florida, Gainesville, FL, USA

Florida LAKEWATCH is a successful long-term volunteer water quality monitoring programme. LAKEWATCH (est. 1986) began when citizens from Lake Santa Fe and Lake Broward (Alachua County, FL) contacted the University of Florida seeking answers on how to best manage their lakes. After training these volunteers to collect water from their lakes, they began monthly surface water sampling and measuring water clarity (Secchi depth), delivering samples back to the University to be analysed for nutrients (phosphorus and nitrogen, key limiting factors for biological production), and chlorophyll (an indicator of algal biomass). These first citizens gave university professionals an idea and an opportunity to use volunteers to address these three original LAKEWATCH objectives: (1) How is the limnology of Florida lakes impacted by changing geologic gradients everywhere apparent in Florida? (2) What variance is exhibited among and within Florida lakes? (3) Are there trends in the water quality of Florida lakes? Professional and public interest with citizen involvement flourished to such a degree that continued requests for state funding were heard, and the Florida Legislature officially established Florida LAKEWATCH in 1991 (Chapter 1004.49 F.S.). The evolution of the programme led to the organisational model of a Land Grant University with aspects of teaching, extension and research (Fig. 16.2).

Fig. 16.2 Left: Volunteers processing samples collected at Lake Henderson Citrus County, FL. Photo credit: Florida LAKEWATCH. Right: Volunteers from Highlands County, Florida showcasing two awards that LAKEWATCH received in 2014. The Distinguished Service Award from the School of Forest Resources and Conservation, University of Florida, Institute of Food and Agricultural Services (UF/IFAS), and the National Vision Award from the National Water Quality Monitoring Council in recognition of extraordinary vision, collaboration and leadership in water quality monitoring and environmental protection. Photo credit: Florida LAKEWATCH

Quality assurance and quality control (QA/QC) planning procedures specifically designed to assist volunteer water quality monitors were developed to address these issues (United States Environmental Protection Agency 1996), while an increasing number of studies have been carried out to determine the concordance between volunteer and professionally collected data. Both the Florida LAKEWATCH and Lakes of Missouri Volunteer programmes, for example, showed that results for variables measured—total phosphorus (TP), total nitrogen (TN), chlorophyll *a* and Secchi disk depth—were strongly correlated between professionals and volunteers (Obrecht et al. 1998; Hoyer et al. 2012). Further studies have shown similar levels of concordance for measurements of the average cyanobacterial toxin microcystin in samples (Sarnelle et al. 2010), and for determining macroinvertebrate taxon richness (Fore et al. 2001).

Wiggins et al. (2011) pinpoint three stages at which QA/QC interventions can take place, namely before, during and after participation by volunteers. Common solutions include pre-monitoring participant training and certification (Hoyer et al. 2012), inclusion of tests during online data entry to validate data, replication by multiple participants and expert review of data to highlight anomalies (Wiggins et al. 2011; Worthington et al. 2012). Effective monitoring programmes also acknowledge volunteers' limitations by using protocols that match their skill and equipment, with authorities then alerted when more detailed monitoring using specialist equipment and expertise is required (Savan et al. 2003).

16.2 New Zealand: The Freshwater Policy Context

Examples discussed thus far have included volunteer water quality monitoring programmes primarily from the USA. This restricted geographic focus raises the question of which types of programmes exist in other parts of the world. New Zealand provides a useful case study, as water quality monitoring is likely to evolve into a collaborative activity in the future. The drivers for this change are legislated changes to government agency-led management of freshwater and a greater emphasis on community participation in natural resource management.

At present, there is a pressing need for more effective freshwater management. Although point source pollution has largely been controlled through tighter regulation and enforcement, non-point source pollution (stemming mostly from agriculture) continues to have a major impact on freshwater quality (Parliamentary Commissioner for the Environment 2013). Water quality declines have been recorded across all major rivers (Ballantine and Davies-Colley 2010), and 44% of currently monitored lakes are classed as eutrophic or of higher (degraded) trophic state (Verburg et al. 2010). Public awareness of these declines has increased such that water pollution is regarded as a critical environmental issue (Hughey et al. 2013), especially for rivers and lakes (Horizon Research 2013).

Water quality programmes vary widely between local authorities due in part to programmes inherited from management structures (e.g. catchment boards) that predated the formation of regional councils (in 1989), as territorial management authorities (Office of the Auditor-General 2011; Derby 2012). The need for guidance on freshwater management from central government has resulted in the new National Policy Statement for Freshwater Management (NPS-FM), the primary function of which is to maintain or improve overall freshwater quality within a region (New Zealand Government 2014a). The NPS-FM National Objectives Framework will enable local authorities and communities to plan for freshwater objectives (i.e. the desired state of freshwater relative to the current state), through a participatory process involving members of the wider community (Ministry for the Environment 2013). The NPS-FM recognises the environmental, social, economic and cultural values of freshwater resources in line with the range of iwi [Māori tribe] and community interests. In this respect, the policy enlarges upon existing, world-leading approaches centring on the specific inclusion of Māori in negotiated arrangements for resource management.

Although the date for NPS-FM implementation by local authorities was established as December 2015, full implementation may be achieved through a staged process to meet a final deadline of December 2030 (New Zealand Government 2014a). This will allow time to define the manner in which community members, including iwi, will be involved in setting water quality objectives. The NPS-FM includes a suite of variables that community groups may be encouraged to monitor in the future. These variables are similar to those commonly measured in freshwater volunteer programmes in the USA, namely water temperature, periphyton abundance, sediment, discharges, connectivity, total nitrogen and total phosphorus, fish, invertebrates and riparian margin condition.

16.2.1 Lake Water Quality Monitoring

Standardised protocols used by local authorities and science organisations for measuring lake water quality (e.g. TN, TP, chlorophyll *a* and Secchi depth) have not been widely adopted by New Zealand community environmental groups and non-government organisations. Contributing factors are diverse and are likely to include a low population density, for example, relative to North America (World Bank 2014), lake ownership and a lack of accessible monitoring toolkits that bring together basic protocols, educational support material, equipment and necessary training. Limited funding is available for setting up and sustaining monitoring programmes, let alone for meeting laboratory costs for specialist analysis of TP, TN and chlorophyll *a*.

A further explanation may lie with who ultimately takes responsibility for water quality monitoring. In New Zealand this activity is carried out by local authorities who are obliged to do so under the 1991 Resource Management Act (New Zealand Government 2014b). These authorities are also required to provide data to the Ministry for the Environment for national State of the Environment reporting (Hilliard and Breese 2010). Although communities may not perceive a need for developing their own water quality monitoring programmes, equally, local authorities have not widely considered the potential of community volunteers to greatly enhance their capacity to collect useful data. The lack of water quality monitoring programmes that are reliant on volunteers to collect data for educational, scientific and environmental management purposes contrasts greatly with the two case studies from the USA presented in Feature Boxes 16.1 and 16.2. These examples form part of a lengthy history of volunteer water quality monitoring in the USA. Programmes have been established by forward-thinking volunteer organisations, for example to raise public awareness of water quality issues (Firehock and West 1995). Throughout the 1970s and 1980s, government agencies and other private organisations developed programmes for volunteers, recognising the cost savings associated with data collection, and the added benefit of strengthening local community engagement with their natural resources (Canfield Jr et al. 2002).

16.3 Community Freshwater Ecosystem Restoration

More than 600 community environmental groups largely made up of incorporated societies and trusts are estimated to exist in New Zealand (Ross 2009). These groups represent a sizeable labour force of up to 45,000 volunteers (Handford 2011). An online questionnaire was sent to 540 of these groups and results showed that almost two-thirds (63%; $n = 286$) were actively engaged in restoring, enhancing and protecting native habitat associated with freshwater ecosystems (Peters et al. 2015). Of this number, 32 groups (18%) carried out lake and catchment restoration (see Fig. 16.3). An overview of groups' freshwater restoration projects reveals a broad suite of activities within water bodies, in the riparian zone and extending into the catchment. Common activities undertaken by volunteers include pest animal and weed control, and revegetating with native species. Public education programmes and advocacy in the form of preparing submissions on environmental matters are also common, underscoring the diverse ways in which groups address issues around environmental degradation (Peters et al. 2015). Groups' efforts are generally poorly documented in the peer-reviewed literature; however sources such as group newsletters and reports highlight various freshwater ecosystem-related outcomes. Stream-based groups have controlled invasive aquatic weeds (Friends of Waiwhetu Stream 2013), improved habitat for native fish by manipulating the instream environment, restored riparian vegetation (Collier et al. 2008) and improved waterway health after best practices were adopted on adjacent farms (Allen et al. 2014). Internationally recognised protection for wetlands has been secured (Ravine 2007), and partnerships with researchers have been developed to investigate sediment traps and floating wetland effectiveness (New Zealand Landcare Trust 2013). Large-scale lake catchment projects have resulted in predator-proof fencing, intensive pest animal control and reintroduction of native fauna (Campbell-Hunt and Campbell-Hunt 2013; Rotokare Scenic Reserve Trust 2014).

Fig. 16.3 Community freshwater ecosystem restoration: riparian planting at Lake Cameron/ Kariotahi, Waikato, New Zealand. Photo credit: NZ Landcare Trust

Despite increased public awareness of water quality declines, groups' environmental monitoring activities were predominantly focused on terrestrial ecosystems in support of objectives to control/manage pest animals and weed species (Peters et al. 2015). Although 41% of community groups ($n = 239$) reported an interest in monitoring water quality in the future, most groups currently only carried out stream invertebrate counts (22%). The use of freshwater monitoring toolkits such as the Stream Health Monitoring and Assessment Kit/SHMAK (Biggs et al. 2002) and Cultural Health Index/CHI (Tipa and Teirney 2003) has therefore been very limited (8%; $n = 157$), despite these tools being developed specifically to assist community groups and iwi (Māori tribes) to quantify water body health.

16.3.1 Lake 'Care' Groups

The following case studies provide insights into the diverse ways that volunteers have mobilised around their local lakes. Despite the absence of volunteer toolkits and wider programmes to support the collection, analysis and management of volunteer data for lakes, public concern over water quality declines has motivated some community groups and individuals to collect their own data. Other group efforts have centred on building a better knowledge base about water quality declines in order to improve local authority lake management, as well as inform policy.

16.3.2 Developing Long-Term Citizen Science Monitoring Programmes

A common thread running through each of New Zealand case studies is the desire for knowledge, environmental stewardship of the local area and engagement in activities that lead to positive environmental as well as social outcomes. There is a big step between these individual community group initiatives and the large-scale, long-term water quality monitoring programmes commonly encountered in the USA such as Florida LAKEWATCH, and the Lake Sunapee Protective Association initiatives. However, the shifting focus of water management in New Zealand from agency-only to multi-stakeholder collaborations has created a political environment amenable to an expanded role for community volunteers in freshwater science and monitoring.

With an abundance of well-established programmes to reflect on, in particular across the USA, Canada and Europe, a suite of interdependent management, science and social priorities can be brought together to guide the development of new, long-term citizen science programmes centring on volunteer monitoring. The priorities listed here are neither intended to serve as a complete checklist nor are they presented in any hierarchical order; instead, they contribute to a growing dialogue on citizen science best practice (e.g. Shirk et al. 2012). Comprehensive frameworks that outline the practical steps for developing, implementing and evaluating citizen science projects (that can also be applied to volunteer monitoring programmes) have been produced for the UK Environmental Observation Framework (Tweddle et al. 2012) as well as for Canadian community monitoring networks (Conrad and Daoust 2008) (Table 16.1).

Table 16.1 Key components for long-term citizen science monitoring programmes

Key component	Questions
Defined project objectives and outcomes	• Who will participate and what role(s) will each play? • What is the scope, nature and purpose (e.g. educational, research or both) of data collected? • How will project progress and results be communicated?
Fit-for-purpose programme structure	• Which form of coordination will be most successful—scientist-led, community-led or a hybrid approach combining both?
Robust monitoring design and appropriate protocols	• What design and protocols will meet project objectives and be functional for community users?
Ethical data use and ownership	• Who owns and has access to the data?
Adequate resourcing	• What is the most appropriate funding model for the project?
Engaged participants	• What will motivate volunteers to join and continue participating in the project?
Effective communication	• Which channels of communication will be used and how?
Ongoing evaluation	• How will evaluative procedures be integrated into the project from the outset?

16.3.3 Defined Objectives and Outcomes

Citizen science objectives commonly include education and public engagement, contributing to scientific research on ecosystems and phenomena, enhancing community capacity for decision-making, providing data for environmental management or supporting policy development (Shirk et al. 2012). Defining volunteer monitoring programme objectives as well as outcomes from the outset has major implications for who will participate, and how they will participate. The scope and nature of data collected will also be affected, an example being that an educational or a public engagement monitoring programme may not prioritise data quality to the same level as a programme designed to collect data for research or management purposes (Savan et al. 2003). Additionally, the way in which information is communicated and results are shared will also reflect the project purpose, as the target audience will vary, and how data are used will be programme specific. Further detail on these key priorities is outlined in the following sections.

16.3.4 Fit-for-Purpose Programme Structure

Three primary approaches that characterise the structure of citizen science programmes are described in this chapter as consultative/scientist-led, co-created and community-led. Each approach has evolved in response to factors such as differing information needs, level of resourcing available and type of participants targeted for the programme (Tweddle et al. 2012). The increased complexity of data management in programmes spanning wide temporal and spatial scales typically requires a consultative approach (i.e. led by scientists and resource-managers) (Ely 2008), Florida LAKEWATCH being an example (Feature Box 16.1). In contrast, programmes with a more restricted geographic focus may rely on community volunteers to provide leadership and management roles as demonstrated by the LSPA (Feature Box 16.2) and by New Zealand community environmental group initiatives. The three approaches outlined above may be combined in order to achieve programme outcomes. The main body of volunteers participating in the programme may collect primary data, while a core group of more engaged volunteers may participate more fully in the research process, e.g. developing new research questions and analysing data (Tweddle et al. 2012).

16.3.5 Robust Monitoring Design and Appropriate Protocols

Citizen science monitoring programmes are commonly designed to test hypotheses or contribute observations to environmental databases (Miller-Rushing et al. 2012). In both instances, long-term data sets can provide unique insights into ecosystem function particularly if programmes are guided by research questions and not by 'mindless data collection' (Lindenmayer and Likens 2010). Posing good research questions is undoubtedly challenging; however adapting the programme to suit emerging questions can contribute to programme longevity (Lindenmayer and Gibbons 2012). Firehock and West (1995), for example, highlight how stream monitoring programmes have evolved over decades owing to increased understanding of freshwater ecosystems, changing management needs and availability of new, more user-friendly monitoring equipment. Pilot studies can determine the suitability of protocols for volunteer use while also determining whether the quality of volunteer-generated data meets programme objectives (Lindenmayer and Likens 2010; Tweddle et al. 2012).

16.3.6 Ethical Data Use and Ownership

The question of who owns and has access to the data produced has implications both for disseminating the research and how the research outputs are used (Scassa and Chung 2015). Intellectual property rights come to the fore when considering the ways in which volunteers can participate in programmes and agreements (contractual or otherwise), made between volunteers and programme coordinators. Copyright may apply to volunteers' photographs, videos and text-based contributions; however, programme designers that seek intellectual, innovative or creative input from participants, such as app design or improved methods, may need to consider the possibility of copyright infringements as well as patents (Scassa and Chung 2015).

16.3.7 Adequate Resourcing

With an abundance of citizen science monitoring programmes to learn from, the first step is identifying successful models of funding from other projects (including those from outside the environmental sector), most likely to suit the objectives and structure of the new programme (Lindenmayer and Gibbons 2012). Sourcing long-term funding for monitoring has proved challenging due to the lack of importance attributed to the sustained collection of environmental data (Conrad and Hilchey 2011; Lindenmayer and Gibbons 2012; Hoyer et al. 2014).

The cost-saving nature of citizen science compared with professional monitoring is often emphasised (Levrel et al. 2010), although this risks not adequately funding other aspects of the programme. Set-up costs, for example, can be substantial when considering the monitoring equipment required, along with programme infrastructure development (e.g. smartphone apps, website and database design). Volunteers also require ongoing support following initial monitoring training, and specialist analyses necessitating skilled scientific personnel (often using expensive analytical equipment and specialist labs), which also contribute to costs (Tweddle et al. 2012).

Many successful programmes rely strongly on institutional support (Tweddle et al. 2012), as well as funding from diverse sources including private donations, memberships and government funding (Maine Volunteer Lake Monitoring Programme 2012). Other means toward enhancing the scope of programmes while contributing to their longevity include leveraging off existing monitoring programmes and monitoring-related events such as 'BioBlitzes' (Lundmark 2003). Lindenmayer and Gibbons (2012) stress the importance of informing programme partners and funders of programme outputs and outcomes while also underscoring the importance of monitoring, as well as the costs of *not* monitoring.

16.3.8 Engaged Participants

Citizen science would not exist were it not for the willingness of volunteers to donate their time, knowledge, skills and, at times, personal resources. Volunteers may comprise members of the general public, school children, indigenous groups or special interest groups such as recreational anglers (Ansell and Koenig 2010; Goffredo et al. 2010; Open Air Laboratories 2013).

Although an ethic of inclusivity underpins much of citizen science discourse, Haklay (2015) points out that middle-class white men continue to be overrepresented in projects. Encouraging the participation of disadvantaged communities, for example, as has occurred through then Open Laboratories Programme (Open Air Laboratories 2013), requires a purposeful approach so that volunteers may be meaningfully engaged in the programme (Haklay 2015). Meaningful engagement relies on understanding volunteers' motivations for participating in the monitoring programme (e.g. widening social networks; contributing to science or society; educational), and therefore understanding their expectations (e.g., social engagement; purposeful collection of data; quality communication) (Clary and Snyder 1999). Powell and Colin (2008), however, warn that few professionals attempting to 'engage in engagement' are clear about specific programme objectives and outcomes relating to engagement, and whether the means used to facilitate engagement will achieve these ends. This underscores the need for considering the complex nature of programme social dynamics from the beginning.

16.3.9 Effective Communication

The skills required for developing and implementing volunteer monitoring programmes are considerable and include scientific expertise, project management and volunteer facilitation. The ability to communicate effectively, as well as promote the programme, is becoming increasingly important as communication channels diversify and information is sought across different media. Communication in the development phase of the programme is typically targeted at establishing the programme team and at volunteer recruitment (Tweddle et al. 2012). During programme implementation, regular communication that acknowledges volunteers' input, staff and partner roles as well as funders' support helps to build a sense of community and shared purpose (Dickinson et al. 2012). Communication outputs may include informal updates or points of interest for the wider community using tweets, blog posts, newsletters, radio, television or other media. Formal channels include peer-reviewed journals or reports mainly designed for partners and funders. An alternative example of the latter are citizen State of the Environment reports that share programme outputs and outcomes with volunteers, programme staff, resource managers, researchers and other stakeholders (Maine Volunteer Lake Monitoring Programme 2012; Open Air Laboratories 2013).

16.3.10 Ongoing Evaluation

Although citizen science monitoring programmes have been conceptualised as a series of steps with feedback loops at specific junctures (e.g. Bone et al. 2012; Tweddle et al. 2012), the actual process of implementing these programmes is most likely to be repetitious and discursive (Pollock and Whitelaw 2005). The dynamic and multifaceted nature of citizen science monitoring programmes heightens the need for a considered approach to evaluation that captures the range of social educational, scientific or environmental management-related outcomes sought. With a defined set of outcomes to guide the evaluation of programme effectiveness, iterative testing by users should also be viewed as part of a wider, ongoing process of evaluation that can be adapted in line with programme evolution (Tweddle et al. 2012).

Case Study 16.1 Lakes Water Quality Society
The Rotorua lakes comprise 11 volcanic lakes ranging in size from 79.8 km^2 (Lake Rotorua, Fig. 16.4) to 1.4 km^2 (Tikitapu/Blue Lake), and are situated in the central North Island. From the late 1990s, lakeside inhabitants had begun witnessing cyanobacterial blooms occurring annually in several of the Rotorua lakes. The Lakes Water Quality Society (http://lakeswaterquality.co.nz/) evolved in response to these ongoing water quality declines and the lack of remedial action taken by local authorities. Over 12 years, the LWQS ran a series of eight symposia, both triggering interest in the science and disseminating scientific knowledge to decision-makers and the wider community. The LWQS has also been extremely effective at strategic policy change. In 2000, a Regional Water and Land Plan (RWLP) was proposed by the regional resource management agency, Environment Bay of Plenty. The Plan included Rule 11 which prohibited any additional land-use intensification that could increase nutrient discharge to five lakes (including Lake Rotorua) with the overall objective of maintaining water quality above, or at, the level it was in 1960.

(continued)

Case Study 16.1 (continued)
Action Plans were developed to support remedial works for these five lakes. The LWQS successfully fought for further Action Plans to be developed for the seven remaining lakes, with works to be initiated when their water quality declined below a threshold level. The LWQS has also played a key role in securing major funding from central government (a total of $79.3 million), and encouraged action by local authorities resulting in improved sewerage reticulation, in-lake engineering works, land-use change from farming to forestry, more effective on-farm nutrient management and precipitation of phosphorus from some lakes and streams (summarised from McLean 2014).

Fig. 16.4 Lake Rotorua. Photo credit: Craig Putt, Bay of Plenty Regional Council

Case Study 16.2 Lake Tarawera Ratepayers Association

At 51.0 km^2, Lake Tarawera (Fig. 16.5) forms the second largest of the Rotorua lakes group in the central North Island. In 2005/2006 a summer student from the University of Waikato was employed by the Lake Tarawera Ratepayers Association (LTRA) to determine nutrient concentrations of the lake and tributary inflows as well as to measure tributary discharges. Local resident Terry Beckett, a LTRA member and avid fly-fisherman, recognised the value of continuing the monitoring. He has continued the monitoring quarterly (following protocols used by the University), to now provide a long-term (10-year), continuous data set. The samples, collected on behalf of LTRA are supplied to the University of Waikato for analysis of ammonium, soluble reactive phosphorus, nitrate, TP and TN, complementing data from the regional council state of the environment monitoring which is restricted solely to the lake itself. Surface water discharges also continue to be measured by the LTRA. This comprehensive data set has revealed the linkage of Lake Tarawera to other lakes by outflows, and the marked differences in composition between geothermal and cold-water spring inflows. This citizen science work has led to a realisation that a wider view than just the immediate lake catchment needs to be taken into consideration when investigating ways in which anthropogenic nutrient inputs can be reduced. The dissemination of data via newsletters has resulted in an informed and supportive community whose annual donations support costs associated with data collection and analysis (Terry Beckett, Pers. Comm.). Donations also help support a monitoring buoy that forms an important source of 'live' information for local residents (Environment Bay of Plenty, undated).

Case Study 16.3 South Wairarapa Biodiversity Group

The Oko(u)rewa/Onoke Lagoon (Fig. 16.6) covers 6.3 km^2 and is part of a heavily modified system linked to Lake Wairarapa (78 km^2) at the southern end of New Zealand's North Island. The vision of the South Wairarapa Biodiversity Group is '. . . a healthy coastal lagoon within the Wairarapa Moana Complex supporting indigenous flora and fauna, for the education and enjoyment of the N.Z. public' (http://swbg.weebly.com/okorewaonoke-lagoon.html). To achieve this vision, the group is revegetating the lagoon margins and has conducted baseline monitoring to develop a report of current lagoon health. In 2013 funding was received for the group to sample invertebrates and collect water for laboratory analysis. A group member with a science background conducted her own research into suitable methods to use—a challenge given the lagoon's variable flushing regime, fluctuating salinity, and soft, sticky substrate. As a well-established group, strong

(continued)

Case Study 16.3 (continued)
partnerships have evolved with government and other agencies. For example, university staff and students provided technical assistance while sampling and carried out analyses of water quality samples. The group is considering whether to continue monitoring, though will need to source additional funding as well as verify the best protocols to use for this complex system (Jane Lenting and Heather Atkinson pers. comm).

16.4 Future Prospects

Citizen science is continuously evolving. Advances in Web 2.0-based technology have enabled large-scale initiatives to flourish (e.g. Worthington et al. 2012), achieved by simplifying data collection and management, automating quality control and facilitating communication between stakeholders (Newman et al. 2012). Wireless sensor networks and smartphones are rapidly emerging as powerful tools for data collection. The latter function as mobile, Internet-enabled computers that provide access to geographical information systems (GIS), and contain global positioning systems (GPS), scanners, microphones, camera and video (Teacher et al. 2013). Over half (54%) of New Zealanders now own a smartphone, a similar figure to the USA (56%) (Ipsos MediaCT 2013a, b). Although only 20 out of 205 community environmental groups in New Zealand currently download smartphone software apps, future interest in using apps is high (117 groups out of

Fig. 16.5 Lake Tarawera (Lake Okareka in foreground). Photo credit: Phillip Capper, Flickr CC BY 2.0

Fig. 16.6 Oko(u)rewa/Onoke Lagoon. Photo credit: Joe Hanson, Department of Conservation

205), indicating a willingness to trial new and potentially more efficient methods of data capture.

Smartphone apps related to water quality monitoring are becoming increasingly diverse. Volunteers can now log sightings of harmful algal blooms (Xiao et al. 2011), or determine the likelihood of algal bloom occurrence in shallow lake waters (University of Wisconsin Parkside 2014). Apps such as Creekwatch allow users to enter qualitative data on water quantity, rate of flow and level of pollution, as well as upload images of the waterway evaluated (California State Water Resources Control Board 2012). More comprehensive tools currently in development include a Global Lake Ecological Observatory Network app that will allow users to enter diverse geo-referenced data, including quantitative values of dissolved oxygen, water temperature, pH and Secchi depth, as well as aquatic vegetation (http://gleon.org/node/4370). In the future, inexpensive sensors attached to smartphones may enable measurements to be made of nitrate and phosphate concentrations. Such analyses are mostly conducted by highly trained personnel utilising specialist equipment and laboratories.

The greater volumes of data generated from wireless sensor networks and mobile technology will necessitate better data management. Newman et al. (2012) create a vision of future database interoperability, where computer-to-computer interaction will occur automatically, generating metadata and tracking changes to data sets, thus broadening the scope of data use and function.

16.5 Summary

Contributory citizen science projects such as those led by resource management agencies and scientists are well underway in New Zealand (e.g. New Zealand Marine Studies Centre 2012; Spurr 2012; Brumby et al. 2015). At the same time, community-led initiatives include efforts by community environmental groups monitoring environmental change within their restoration projects (Peters et al. 2015). However, freshwater quality monitoring by volunteers is infrequently carried out. In contrast, diverse state-wide programmes exist in the USA where volunteer data contribute to research and management planning and simultaneously achieve educational outcomes. Many useful lessons can be learned from these large-scale, long-term volunteer monitoring programmes which can be adapted to suit New Zealand and other countries where similar programmes are yet to be developed. Formulating a coordinated approach between all stakeholders and designing the infrastructure required to support citizen science programmes—particularly if data are to be used by agencies and researchers—are undoubtedly challenging. Yet, as a steadily increasing number of citizen science initiatives demonstrate, programmes with a long-term vision, effective data collection, storage, sharing and retrieval mechanisms, a focus on participation and collaboration, effective leadership and sufficient resourcing can result in mutually beneficial outcomes for volunteers and their wider communities, resource managers and researchers.

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Chapter 17
Reflections on Lake Restoration

Clive Howard-Williams, Kevin J. Collier, David P. Hamilton, and John M. Quinn

Abstract Widespread public concern in the last few decades over the state of freshwater ecosystems has driven a strong commitment to their rehabilitation in many countries. In New Zealand, this commitment is being expressed at levels from individual communities through to the government. Lake ecosystems are at the forefront of public discussions and policy development. Māori are integrally involved as the indigenous people of New Zealand who have rights and interests in water as well as environmental stewardship roles. New Zealand has acquired a substantial knowledge base on restoration actions, many of which integrate across environmental, social, cultural and policy domains, and which in some cases allow for a comprehensive evaluation of their effectiveness. Principal pathways to lake restoration identified in the chapters of this volume are through:

1. Reduction in contaminant loads and concentrations, particularly sediments and nutrients, where these have caused a decline in lake health
2. Reduction of internal nutrient loads in lakes
3. Alteration of lake stratification patterns where these exacerbate degradation
4. Control of invasive plant and animal pest species
5. Remediation of migration barriers to enhance connectivity for native species
6. Biodiversity enhancement
7. Lake monitoring and modelling to prioritise actions
8. Improved community participation processes

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Here we review the depth and breadth of restoration actions broadly delineated into five sections: (A) management and modelling; (B) water quality restoration; (C) biodiversity restoration; (D) monitoring and indicators; and (E) social and cultural contexts. Our reflections here include both general concepts and future directions in lake restoration, with a call for a unified national approach to this issue.

Keywords Nutrients · Biosecurity · Engineering · Data management · Modelling · Social involvement

17.1 Introduction

The world's large lakes have been disproportionately altered by human activity (Carpenter et al. 2011) and lakes in general have suffered, as catchment receiving waters have changed due to human-induced alterations to landscapes. Nutrient-enriched runoff from agriculture and urban land has been a major cause of eutrophication that has resulted in harmful blooms of algae, deoxygenation, loss of economically valuable species and human health problems. Carpenter et al. (2011) and Moss (Box 1.1) remind us that freshwater and land restoration are mutually inseparable. New Zealand lakes have been subjected to the same changes as a result of changing land use (Gluckman 2017; MfE 2017). More recently, New Zealand has contributed significantly to knowledge on restoration of degraded lake ecosystems. This has been driven by intense public concern, particularly in the last decade, over the state of the country's freshwater ecosystems (Hughey et al. 2013), and more recently by a requirement for management and planning authorities (e.g. regional councils) to prevent further degradation through the National Policy Statement for Freshwater Management (NPS-FM) (MfE 2014). If water bodies are in breach of national objectives, then Regional Councils are mandated to improve them. Background to some of the costs associated with such improvements for lakes is provided in Chap. 1 of this volume. There have also been large cumulative investments by farmers and industry in catchment restoration, such as the Sustainable Dairying Water Accord (DairyNZ 2015), supported by the Department of Conservation, regional councils, Māori authorities and non-profit organisations such as the New Zealand Landcare Trust, while community groups have participated strongly in restoration projects on individual lakes (Peters et al. 2015).

The change in perspectives from management-only to both management and restoration of lakes in New Zealand began in the 1990s. It is of interest that the Lake Manager's Handbook, which synthesised knowledge of lake management in New Zealand up to the 1980s, did not include restoration (Vant 1987) but focussed on knowledge of lake processes (e.g. nutrients and eutrophication) and management (e.g. water levels, nutrient loads). A move towards aquatic restoration science and practices was signalled subsequently by general publications on aquatic habitats (Collier 1994) and on lakes in particular (Howard-Williams and Kelly 2003; Rowe 2004). More recently Hamilton et al. (2016) and Hamilton and Dada (2016) have focussed on general perspectives of lake restoration in New Zealand. Contemporary issues involved in lake restoration have also been elaborated in works relating to ecosystem services (Schallenberg et al. 2013) and climate change (Hamilton et al. 2013). In addition to the effects of pollution, some lakes have been affected by hydromorphological modifications within their catchments, such as dam construction and increased abstraction (Box 4.2). Restoration targets will need to consider such 'irreversible' changes.

This volume has taken our knowledge further by providing practical learnings on lake restoration approaches and techniques that have been used in New Zealand, embedding these in the context of current international restoration initiatives and best practices. Chapter 1 refers to the New Zealand policy framework that has been further strengthened by a government announcement in 2017 of a 10-year $100M 'Freshwater Improvement Fund'. The fund aims at supporting projects designed to contribute to improvement of the management of New Zealand's freshwater bodies. Criteria include improvements in water quality and increased biodiversity in lakes, rivers and wetlands.

Future directions for lake restoration in general will require that attention be given, not only to the technical details discussed in this handbook, but also to collaborative input from communities living in lake catchments. These communities are central to the tenet of the NPS-FM achieving a collaborative agreement on the limits for water quality and quantity, and therefore to determining the long-term goals for restoration. Restoration is also a key plank in indigenous Māori aspirations for the future state of New Zealand lakes, as clearly described in Chap. 15, and aspirations are expressed at a high level in the goals of the NPS-FM.

This book has five sections with chapters covering New Zealand initiatives: (A) management and modelling (Chaps. 2 and 3); (B) water quality restoration (Chaps. 4–7); (C) biodiversity restoration (Chaps. 8–10); (D) monitoring and indicators (Chaps. 11–14); and (E) social and cultural context (Chaps. 15–16). Here we summarise key concepts and future directions in lake restoration that are identified in the five sections taking into account information in feature boxes by international contributors.

The recovery of systems, including lakes, following rehabilitation or mitigation actions may follow one of several pathways, ranging from near-complete recovery of values, services and resilience, through to an altered state with different ecosystem properties to the one that preceded degradation. Full restoration of community values and attainment of pre-development levels are often not possible (Fig. 17.1) because of extensive land-use changes and conflicts between values and services such as urban and industrial uses, pastoral and arable farming, hydropower and flood or drainage management (Naiman 2013).

17.2 Management and Modelling

Modelling for lake restoration needs to be considered across a wide range of scales from catchment to ecosystems. Modelling helps to provide an understanding of the causes of degraded water quality and can guide restoration efforts to improve water quality and biodiversity. Modelling results can also help to identify knowledge gaps and prioritise future monitoring, as well as identify monitoring that might be unnecessary.

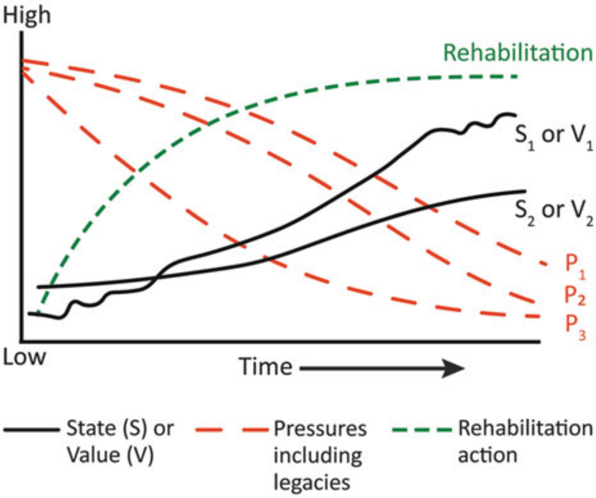

Fig. 17.1 Responses to rehabilitation showing different reductions in pressures (P_1, P_2, P_3) including legacies and corresponding ecosystem state (or value) improvements (S_1, V_1 or S_2, V_2)

Catchment models can quantify existing conditions and are used in restoration efforts to define hotspots of contaminant contribution where actions would be most beneficial as well as those types of actions that are likely to be most beneficial. Chapter 2 describes the continuum of catchment models that exists and provides information for choosing specific models for various management applications. In New Zealand these models have been used to address a wide range of scientific and policy-driven issues including those related to developing nutrient-reduction and lake-protection strategies across different geographic regions. Catchment models have predicted the likely consequences of alternative land management strategies, which may substantially reduce the costs of restoring and managing water resources. In this regard, catchment models can be greatly assisted by on-farm nutrient budget models that project losses from land under different farming systems. The use of models for improving farm practices to reduce nutrient losses has increased considerably in the last decade (e.g. www.agresearch.co.nz/news/online-tool-helps-farmers-land-and-waterways/). The Overseer nutrient budget model (OVERSEER®) is now a widely used farm-scale nutrient management tool and is one of a growing number of online tools that have benefited farmers in cost-effective nutrient management. Overseer has also become an important tool for regional councils in their environmental stewardship role, which includes managing water quality to benefit downstream aquatic ecosystems.

At the next scale down from catchments, the renaissance in lake modelling research (Chap. 3) over the last decade has seen improved representation of eco-physiological processes, improved practices in model evaluation and an increased awareness and communication of model uncertainty (Box 3.2). In-lake modelling has been coupled with catchment modelling to understand and predict management and restoration options, as highlighted in several case studies. Models have been broadly categorised as theoretical, heuristic or predictive according to the types of questions they are designed to answer. Chapter 3 also comments on the ability of models to simulate aquatic biogeochemical variables and it describes the importance of data assimilation in combining observations with models to improve model predictive capability. There is no doubt that the success of the use of in-lake models in New Zealand and internationally (see Box 3.1) will result in their further use and development, particularly when these are linked to catchment models (see Box 2.1) that provide quantitative results of scenarios for lake management and restoration.

17.3 Water Quality Restoration

A good understanding of the sources of contaminants and the processes that facilitate and modify their transport within freshwaters is the first step in the restoration of good water quality in agricultural catchments. In-depth knowledge of how contaminant sources are influenced by different agricultural systems and practices is a prerequisite to the development of effective tools that enable desired water quality objectives to be met. The application of processes and practices to reduce contaminant losses from land to water is currently a significant development area in New Zealand (McDowell et al. 2013). Such tools can be used to optimise farming systems to identify where agricultural production and desired water quality outcomes can be accommodated. While lakes can respond to changes in external pressures relatively quickly (see Box 4.2) in many cases the speed of recovery is affected by time lags in both contamination pathways and in ecosystem responses. Thus, legacy effects due to groundwater lag times, and effects due to the different patterns of attenuation of nutrient species in soils need to be taken into account when making decisions on land management options to enhance lake water quality.

The reduction of nutrients available to lake phytoplankton involves two major components: (1) reducing the inputs from land and (2) reducing the cycling of nutrients within a lake. While Chap. 4 discussed the sources of land-based nutrients, Chap. 5 emphasised that for restoration interventions, information is needed on processes, including microbial denitrification, and internal nutrient loading derived from both in situ measurements and from a mass balance approach. Various in-lake restoration techniques exist to target different fluxes within nutrient cycles but this can only be achieved through adequate quantification of those parts of the cycle that are most likely to successfully reduce nutrient availability and hence phytoplankton blooms. Treatment of the source of the problems rather than the in-lake symptoms needs to be a major consideration in restoration projects (see Box 5.1).

Input loads to lakes from catchments are used in nutrient budgets and are often modelled (see Chaps. 2 and 3), but this may not provide sufficient accuracy at appropriate temporal scales. Similarly, measured loads often suffer from inadequate sampling frequency. Basic mass balance models for lakes usually focus on N and/or P, but in some lakes other factors may drive or limit phytoplankton productivity and biomass, including light availability in the mixed layer, zooplankton grazing, water temperature and micronutrient availability (Schallenberg 2004;

Downs et al. 2008). Such cases require dynamic and perhaps even three-dimensional (3-D) in-lake models that are calibrated and validated with intensive sampling (Chap. 3). Simple mass balance models such as those described in Chap. 5 are attractive as lake management and restoration tools, partly because their simplicity enables rapid and widespread estimation, although uncertainty is not quantified and may be considerable. However, they could be improved by explicitly accounting for different mixing regimes as well as the influence of climate in general on lake nutrient budgets (Hamilton et al. 2013). Wider use of simple, steady-state mass balance models for lake restoration and management will need to more explicitly represent errors and uncertainties in the model predictions and outputs (Loucks and van Beek 2005).

Once contaminants reach lakes, restoration requires application of techniques to reduce their effects. Mitigation may include a combination of physical (mechanical), chemical and, on occasions, biological applications. Chapter 6 examines the possible options for removing or preventing thermal stratification in degraded lakes by accentuation of lake mixing as a strategy for rehabilitation (see Boxes 6.1 and 6.2). Technical detail is provided on aeration to induce vertical movement of the water column as well as several water column mixing systems, and the option of bottom water reoxygenation without destratification. The rehabilitation potential of traditional and new engineering devices needs to be tied to the requirements of the end users of the water, be it for ecosystem recovery and/or external uses (e.g. drinking, irrigation or recreation). Chapter 6 refers to the key points to consider in any mechanical process for enhancing water quality as part of lake restoration and/or management, including the lake size, depth and morphometry, engineering design and availability of in-lake models to predict outcomes. Sherman (Box 6.1) points out that the Australian experience in artificial destratification shows that this technique is more effective at managing water quality (dissolved oxygen, nutrient or heavy metal release from sediments) than it is at preventing the occurrence of cyanobacterial blooms. The few case studies in New Zealand appear to support this although time will tell as to the success of reductions in frequency and intensity of cyanobacterial blooms.

In addition to the physical or mechanical approaches described in Chap. 6, the European experience has included biomanipulation as a restoration technique to alter food webs. In Danish lakes this has involved removal of zooplanktivorous fish to enhance zooplankton grazing and thereby reduce excess phytoplankton biomass (see Box 10.1).

Chemically based mitigations using both flocculant and sediment-capping materials are also available (Chap. 7) for nutrient control. Such mitigations are covered within the concept ‘geoengineering’ (see Box 7.1). Effective use of these materials requires an understanding of the in-lake processes that control the recycling of nutrients stored in the sediments, their speciation (i.e. dissolved reactive phosphorus vs. organic-bound phosphorus) and their interactions with microbial communities and aquatic plants (see Box 7.2). For example, the chemical control of phosphorus (P) with P-inactivation agents has been shown to be effective in DRP removal but not for particulate or organic forms of P. In contrast, flocculants can effectively remove particulate P forms, including detritus, fine sediment and algal cells, from the water column to the sediments. A range of commercially available flocculation agents, as well as passive and active sediment-capping agents targeting P, are available and have been used in New Zealand. Experiences have provided key learnings about the efficacy of such methods at the lake scale. Future prospects for chemical lake restoration methods are likely to include using improved flocculation and sediment capping techniques to settle and seal phosphorus.

The argument is put forward that both nitrogen and phosphorus need to be managed to ensure effective restoration of lakes (Abell et al. 2010) and Chap. 7 covers the use of flocculants which can remove both particulate N and P from the water column. Recently developed products in New Zealand, such as Aqual-P®, can target both nutrients. The use of zeolite, which has a natural sorption affinity for ammonium, appears to be a promising tool to target ammonia released from the sediment as does the concept of incorporation of a zeolite biofilm ‘biozeolite’ that can also remove nitrate from the overlying water column (Huang et al. 2012, 2013).

Without in-lake intervention, nutrients in the lake sediments provide a legacy of human activity that will potentially remain to fuel internal loads (Chap. 5) long after catchment remediation strategies have reduced external loads. Appropriate consideration of where to place these geoengineering agents and their long-term effects needs to be part of any lake restoration strategy.

17.4 Biodiversity Restoration

Restoring biodiversity in lakes involves consideration of the impacts and invasion pathways of aquatic weeds, pest fish and alien invertebrates, and how removal of barriers to migrating native organisms affects the ability of invasive species to reach lakes. The proliferation of invasive plant species (discussed in Chap. 8) invariably has a detrimental effect on native plant biodiversity. Once invasive aquatic weeds have been effectively managed, restoration of native plants can occur via passive regeneration from adjacent sites, seed banks and dispersal by waterfowl, or via active planting. The future of invasive aquatic plants and the threats they pose to New Zealand lakes are linked to the continuing arrival of new plants, underscoring the importance of biosecurity as a key priority for lake protection. Climate change also needs to be considered in the ability of pest species to establish, thrive and expand existing ranges (Hamilton et al. 2013).

Methods available for weed control include habitat manipulation (e.g. water levels), and biological, chemical, mechanical, manual and integrated control approaches. The selection of these is dictated by the target weed species, characteristics of the lake, and the restoration goals. The toolbox for weed management must continually be developed to meet the challenges of new invasive species in an effective and sustainable way, through the smarter use of existing products and new techniques validated by scientific research. One such novel approach is being trialled in Ireland (see Box 8.1) where regeneration of native plants following removal of *Lagarosiphon major* has been aided by laying geotextile across the sediments on treated areas. Many of the gains made in the management of aquatic weeds in New Zealand have been due to the availability of selective herbicides, permitting their control without damage to native species, and these have changed little over the last few decades. The use of herbicides to control aquatic weeds has always been controversial with perceived risks posed by their use, including toxicity through bioaccumulation and environmental persistence to humans, animals, non-target plants and algae, and effects on ecosystem processes (e.g. as plants decompose). As a result of these perceived risks there will be a need to evaluate other techniques such as mechanical harvesting or even cost-effective biological controls.

Lake restoration in many New Zealand lakes will also be hampered by the numerous invasive animal species that have established populations (Chap. 9). The lessons from the Laurentian Great Lakes (see Box 9.3) are salutary, where management of invasive species has been hampered by the arrival of other invasive species with different ecosystem effects. This has resulted in unpredictable outcomes from restoration efforts. However, some success has been reported in the case of prevention of transport of invasive species via ballast water in lake shipping (see Box 9.1).

Focus for management must be on preventing initial establishment in preference to expensive post-establishment control. While control is possible with ongoing effort for vertebrate pests, complete eradication is typically difficult even within a single water body. Wisniewski (Box 9.2) points to the considerable efforts in removing European carp (*Cyprinus carpio*) from Lakes Crescent and Sorrel in Tasmania that have involved a variety of controls in an integrated approach. Piscicides have been used in some small lakes and ponds, as has drainage in some parts of New Zealand (West 2015). Examples exist overseas of successful eradication of pest fish through integrated management. In New Zealand, netting, trapping, electro-fishing, and cages, baits and one-way barriers have been used for control purposes but in most instances the ecological outcomes have not been monitored.

Emerging technologies that may assist in the management of non-indigenous fish include the use of pheromones to enhance capture rates, the introduction of taxon-specific pathogens and the genetic modification of fish to produce single-sex (male-only) progeny. In addition, genetic methods for detecting species (Chap. 12) are rapidly improving and will assist in identifying an incursion prior to a subsequent establishment phase (see Boxes 12.1 and 12.2). The usefulness of eDNA was evaluated during the eradication of brown trout in streams flowing into a Wellington reservoir with a PCR product specific to brown trout, which was able to be amplified from samples before, but not after, eradication (Wood et al. 2013). Continued development of new detection and control technologies will be essential to keep pace with the increasing threat of invasive aquatic animals establishing in New Zealand lakes. We note that warming due to climate change may increase the number of species able to establish populations in New Zealand lakes.

In addition to invasive plant and animal species, a range of anthropogenic stressors has resulted in the steady depletion of New Zealand's indigenous freshwater fish fauna over the last century. Translocations to replace depleted populations were common in the past but are less common today. Most restoration effort has been applied to the trap and transfer of elvers over high dams to support commercial and customary eel fisheries. Habitat restoration of small benthic fish, through removal of invasive macrophytes (Chap. 8) and control of pest fish (Chap. 9) and of pelagic species, through water quality improvement, has occurred recently. There have also been significant moves to restore large invertebrates such as benthic mussels (*Echyridella menziesii* in Lakes Rotoroa and Omapere) and plans are underway for restoration of freshwater crayfish (*Paranephrops* spp.) using approaches based on traditional Māori enhancement and harvesting methods (Kusabs and Quinn 2009). While in-lake habitats for native fish have improved in some cases, the loss of connectivity to the sea caused by barriers to migration, such as low water levels in outlet rivers, perched culverts, water control gates, pump stations and dams, has had serious repercussions in preventing upstream access to lakes by New Zealand's many diadromous fish species. However, unrestricted access in some parts of the country may also open opportunities for invasion by pest fish, and passage restoration needs to be carefully managed to prevent this. International experience shows a need for 'realistic and reconciled' restoration goals (see Box 10.2). In many cases the irreversible nature of the changes to lakes requires an integrated approach to restoration that may involve a mix of geoengineering, biomanipulation, watershed management and in-lake engineering.

17.5 Monitoring and Indicators of Restoration

Successful restoration of degraded lakes will require holistic consideration of ecosystem health. Chapter 11 discusses the use of the ecological integrity (EI) concept that incorporates ecosystem structure, composition and function, and resilience as a basis for assessing restoration success. This approach is considered to be an advance over standard water quality metrics for assessing lake condition (Duarte et al. 2009). Ecological integrity encourages a move away from a common perspective that if the physicochemical environment is restored, then the rest of the ecosystem will necessarily restore itself. In the New Zealand freshwater context EI has been defined as a composite of nativeness, pristineness, diversity and resilience, and practitioners are thus able to measure departures from defined conditions such as reference lakes. Chapter 11 discusses how EI approaches have been applied for setting lake restoration goals or targets, and for tracking the success and progress of restoration activities. Holistic approaches aimed at restoring diverse ecosystem components are more likely to achieve desired restoration outcomes because some key synergistic interactions among ecosystem components are likely to increase resilience and probability of restoration success. A key consideration in defining restoration targets is that of using reference conditions as indicators of restoration success. A variety of methods have been proposed to define reference conditions (see Abell—Box 11.1). The European Union Water Framework Directive bases assessments of water quality on deviation from reference conditions (see Poikane—Box 11.2) but equally reductions in such deviations may be used as indicators of restoration success.

Monitoring condition and hence recovery trends requires assessments of biodiversity change. Recent advances in the use of molecular techniques as they apply to monitoring restoration efforts in lakes, discussed in Chap. 12, have allowed the assessment of biodiversity to levels previously unattainable using traditional, morphological assessments. Global initiatives such as the International Barcode of Life have facilitated publicly accessible reference databases which can be used for routine identification of specimens, early warning of invasive species incursions and assessment of community composition changes over time. Reference databases can also be applied to environmental DNA where species can be identified through analyses of environmental samples (e.g. water and sediment). This latter method has proven useful for monitoring the success of invasive fish eradication efforts (see Chap. 10). An eDNA surveillance programme has several advantages (see Box 12.2) in addition to rapid and cost-effective detection of invasive species. These include broad taxonomic assessments and presence of rare species that will aid in interpreting restoration trajectories.

Alternative technologies and approaches will inevitably appear and replace existing methods. In particular, having access to properly archived sample material in national collections as well as comprehensive DNA sequence repositories such as BOLD (Barcode of Life Data System) or GenBank will underpin future efforts employing molecular techniques in the monitoring of restoration efforts and the ability to detect or monitor native and exotic species will become much more automated. Chapter 12 notes that the use of Next Generation Sequencing (NGS) approaches will shift the analyses of community-derived water quality indicators from morphologically based methods to DNA-based approaches. This, in turn, will reduce costs and speed up the routine assessment of water quality indicators. A revolutionary change can be expected in biological monitoring of freshwaters as these NGS developments and further refinements of them improve and become used routinely.

Restoration actions, particularly on larger lakes, require considerable commitment in terms of financial, scientific and social inputs. Hence, continued monitoring of restoration effectiveness will be essential for justifying those commitments, checking progress towards goals and enabling adaptive decision-making as restoration progresses. Chapters 13 and 14 focus on remote monitoring techniques as effective methods of resolving spatial and temporal changes in water quality. Chapter 13 reviews recent advances in high-frequency monitoring, which is the topic of considerable current international research and application. Recent advances in sensor and information technologies have enabled greatly increased spatial and temporal resolution in autonomous measurement and analysis of aquatic environments that allow interpretations of change in ways that traditional field sampling could not accomplish. The application of data analysis capabilities may allow buoy-based technologies to move from simply logging data to more analytical aspects such as quality control, adaptive sampling and automated detection of thresholds and trends.

Case studies provide practical information on the use of these techniques for both long-term and short-term monitoring. New Zealand is a significant participant in GLEON, the Global Lake Observatory Network, where high-resolution sensor data are shared to assist understanding of how lakes respond to a changing global environment (see Weathers and Hanson—Box 13.1, Jennings—Box 13.2). Such methods and refinements will be of inestimable value in more specific monitoring of lake restoration actions. It is clear that presently available technologies are applicable to a wide range of current uses and the use of high-frequency sensors and measurement systems has been shown to improve lake water quality monitoring and management programmes 'enabling rapid responses, reducing some long-term costs, and better characterising transient trends that can have legacies across space and time'. Rapid technological changes, already foreseen, will only improve this situation.

Chapter 14 emphasises that remote sensing has the potential to greatly increase the temporal and spatial resolution of current monitoring methods. Remote sensing techniques need to be applied to lakes and their catchments to provide assessments of land cover change that intersect with in-lake restoration (see Box 4.1). With order-of-magnitude differences in lake sizes and their optical complexity, there is no one definitive remote sensing solution for any lake or group of lakes. The approach needs to be tailored to lake size and water quality attributes of concern that are measurable from spectra. These may now include constituents such as algal blooms that would greatly assist lake managers (see Box 14.1) and aid in documenting restoration processes.

Advanced in situ sensing technologies including autonomous underwater vehicles (AUVs), remotely operated vehicles (ROVs) and sophisticated sonar technologies (originally designed for oceanographic studies) have also been used in large lakes. While aerial based remote sensing can only detect lake surface conditions, underwater remote sensing applies to deepwater conditions (see Box 14.2) and should be considered as highly complementary to aerial sensing.

In the case of aerial remote sensing, spectral and spatial resolution of the sensor can be satellite or aircraft based or, more recently, UAV (drone) based. Chapter 14 describes categories of space-borne remote sensing that vary in spatial and spectral resolution. A satellite sensor's spatial resolution will determine the minimum lake size that can be monitored via remote sensing, while the sensor's radiometric and spectral resolution will determine the ability of the sensor to differentiate optically active water quality constituents (see Box 14.3). While remote sensing remains the only method to quantitatively assess water quality over large areas instantaneously, it should be considered complementary to traditional monitoring techniques as it can only derive information on optically active water quality constituents from surface waters of lakes. Properties of deeper layers are not amenable to remote sensing.

17.6 Social and Cultural Context

The Treaty of Waitangi forms the underlying foundation of the relationship between Māori and the Government of New Zealand with regard to freshwater resources. While there is no 'one Māori' world view, there are principles and values that establish and reinforce the responsibilities and rights of Māori to manage and use natural resources, including lakes. Māori have continually lobbied for protection of New Zealand's lakes and surrounding environments and are committed to seeking a reversal in the state of degradation. Lake restoration approaches that are grounded in Māori values and perspectives, and are co-designed to be responsive to the needs and aspirations of Māori, will ensure benefit to the indigenous community participating in the restoration. This requires a commitment to move beyond conventional understandings of who is 'qualified' to engage in lake research and restoration initiatives. Chapter 15 emphasises the need for a more holistic approach that recognises and empowers Māori to engage as co-governors, co-leaders, researchers, knowledge holders and teachers. A truly collaborative lake restoration programme will provide multiple roles for Māori, including the development and implementation of monitoring and evaluation approaches.

Traditional Māori fisheries have been lost in many, particularly North Island, lakes. Collaboration between tribal authorities and management agencies will be increasingly required as restoration goals are defined and recovery of lost fisheries gains momentum, not only in New Zealand but also across many other parts of the world (see Box 15.1).

There is clearly potential for the input of Māori knowledge to enhance understanding of lake ecosystems, underpin culturally appropriate restoration approaches and provide more holistic and integrated indicators for monitoring. Māori have accumulated qualitative and semi-quantitative information about environmental changes in their lakes and customary resources such as freshwater fisheries, birds, plants and invertebrates harvested for food. Traditional knowledge has rarely been used alongside more quantitative science-based approaches to understand and manage lakes in New Zealand. Chapter 15 argues that, where this has occurred, the partnership between the two knowledge systems has been able to provide innovative and enduring outcomes.

Interest in citizen science is growing globally as environmental degradation continues, information needs increase and value of stronger relationships between the science community and public is recognised (Chap. 16). Community volunteers may participate in citizen science in a range of activities from the collection of environmental data to being fully engaged in project design and delivery. In New Zealand, community groups lead diverse environmental restoration projects. An issue that has arisen early in the citizen science process is that of ongoing monitoring following restoration actions. Only half the groups that expressed interest in monitoring were active in this regard. This contrasts with a strong culture of volunteer water quality monitoring that exists in the USA, where programmes are designed to educate participants while also providing data for fundamental research and for government agency-led environmental decision-making. For instance, the Lake Sunapee Protective Association that relies on volunteers has a history of more than 110 years in preserving and enhancing the environmental quality of the lake (see Weathers—Box 16.1), while on a larger scale the 30-year volunteer project 'Florida LAKEWATCH' has involved university professionals and thousands of dedicated citizen scientists in the collection of long-term water quality data from over 1100 lakes, 175 coastal sites, 120 rivers and 5 springs (see Hoyer—Box 16.2).

Chapter 16 outlines principles underpinning the development and implementation of long-term volunteer monitoring programmes to encourage wider participation of communities, scientists and government agencies in citizen science. Strong relationships between the science community and the wider public are essential for effective intergenerational lake restoration. Community groups in New Zealand lead diverse environmental restoration projects and volunteers participate in citizen science projects that range from collecting environmental data to being engaged in project design and delivery. The 600 groups in New Zealand represent a sizeable labour force of up to 45,000 volunteers (Handford 2011) whose efforts offer great potential to improve lake restoration outcomes. Citizen science is continuously evolving and it is evident that community participation in lake monitoring that builds on state and trend identification in lake resources has the potential to greatly improve scientific and environmental knowledge. In New Zealand an informed and engaged public is increasingly being called on to align with goals of local, regional and national government that require increased public involvement in freshwater management through participatory decision-making.

Chapter 16 discusses how advances in Web-based technology (e.g. Worthington et al. 2012) have enabled restoration initiatives with simplified data collection and management to automate quality control and enhance communication between stakeholders (Newman et al. 2012). Wireless sensor networks and smartphones are now powerful tools for data collection and provide access to geographical information systems and global positioning systems coupled to scanners, microphones and cameras (Teacher et al. 2013). Although few community environmental groups in New Zealand currently download smartphone software applications (apps), future interest in using this software and the universality of hardware (i.e. smartphones) indicates willingness to trial new methods of data capture. Smartphone apps related to water quality monitoring are also becoming increasingly diverse. Volunteers can now log sightings of harmful algal blooms (Xiao et al. 2011) or determine the likelihood of algal bloom occurrence in shallow lake waters. In the future, inexpensive sensors attached to smartphones may enable measurements to be made of nutrient concentrations—analyses that are currently only conducted by highly trained specialists and laboratories.

There are considerable challenges in formulating a coordinated approach to lake restoration between all stakeholders, including indigenous communities and the wider public. Designing the infrastructure required to engage and support these communities is challenging, particularly if data are to be accessible to public agencies and researchers. However, the increasing number of citizen science initiatives demonstrate that well-led and -resourced programmes with a long-term vision, effective data collection, storage, sharing and retrieval mechanisms, and a focus on participation and collaboration can result in beneficial outcomes for all concerned.

17.7 Conclusions

There have been dramatic changes in lake management in the last 25 years in New Zealand with an increasing emphasis on restoration rather than ecosystem maintenance. This has occurred particularly in the last decade when lake management has been focussed not only on prevention of deterioration but also on active restoration to predefined states. More recently, these changes have been accelerated through the collaborative approaches to water management. In 1992 the National Research Council, USA, called for a National Restoration Strategy (National Research Council 1992) that would include areas where large restoration projects require catchment or sub-catchment planning and suggested that, because of the interdependence of aquatic ecosystems, restoration should be undertaken at landscape (catchment) scale and should include hydrology, water quality and flora and fauna. The Council pointed out that unfortunately the vast majority of restoration projects are small in scale and uncoordinated on a regional or national basis. Therefore, principles for priority setting would include:

1. Adopting a landscape perspective
2. Using principles of adaptive management and planning
3. Evaluation and ranking based on opportunity cost (not cost/benefit analyses)

The key to restoration is the setting of goals and restoration objectives with realistic time frames. Such objective setting is often an intensely political and community-driven process. Howard-Williams and Kelly (2003) argued that it is the local human community that has to decide on what constitutes a restored ecosystem. This decision depends on the cultural background of the community, its ability to afford restoration and its plans for water body use in the future. The science of aquatic restoration requires an acknowledgement that the restoration process is often not linear and that hysteresis plays an important role in the process. For instance, Lake et al. (2007) provide examples where a restoration trajectory may follow:

1. The degradation trajectory closely as stress is reduced—the ‘rubber band’ model
2. A non-linear pathway requiring much greater stress reduction than caused by the original degradation trajectory—the ‘hysteresis’ model
3. Various trajectories but the end point is a distinctly lower condition than prior to degradation—the ‘humpty-dumpty’ model

Several chapters and international examples (Feature Boxes) in this book have referred to the need for a holistic or an integrated approach to lake restoration. For instance, managing lakes to prevent algal bloom formation may require a combination of nutrient reduction, destratification, engineering (algal harvesting, diversions) and/or biomanipulation. It should be recognised that most lakes may not recover to a state desired by the community and that timescales of change may also be longer than anticipated by communities.

There is a difference in approaches to lake management versus lake restoration. Lake management generally considers means of prevention (or slowing) deterioration through mitigations in the lake catchment while restoration relies on not only managing deterioration but also reversing it and setting the developmental trajectory on a path to an agreed state.

To assist in accelerating the implementation of lake restoration actions across New Zealand we suggest active consideration of an agreed process across agencies in which restoration initiatives can be compared, optimised and evaluated. Based on the New Zealand and international examples and concepts presented in this book, such a process may follow a series of logical steps including:

1. Regional identification of degraded lakes requiring restoration (identified through monitoring, community consultative processes including iwi settlements)
2. Agreements on restoration goals (i.e. agreed lake-ecosystem state for restoration success given that this may not be the natural state), and agreement on pathways to meet final goals (i.e. intermediate goals that would be applicable for long time frames for the restoration process, see also point 8 below)
3. Identification of priority issues (i.e. water quality degradation, invasive species, cultural uses, biodiversity decline) to be addressed by restoration
4. Identification of actions to address issues and meet restoration goals (catchment improvements, in-lake engineering, chemical interventions, pest control)
5. Modelling of restoration scenarios to assess potential success of actions
6. Analysis and assessment of economic and social implications of most favoured modelled scenarios
7. Identification of timelines for final goal achievement
8. Agreed monitoring programmes and associated funding
9. Contributions to assist adaptive management if pathways to restoration goals are not met

The information summarised in this book, alongside the acknowledgements of current and future changes to methods, technologies and social processes, demonstrates that lake restoration is practical, is needed and is socially supported. We contend that following the suggested nine-point process above will increase success in applying the accumulated information captured in this book and advance future lake restoration efforts.

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