Catchment Sensitivity, Nutrient Limits, Nutrient Spiralling & Forecasting Future Landuse Impacts in Hawke's Bay

NIWA Client Report: HAM2009-001
January 2009

NIWA Project: ELF09202 & ELF09204

HBRC Publication No. 4849
Report No. SD16-21

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Prepared for

Hawke's Bay Regional Council

NIWA Client Report: HAM2009-001
January 2009

NIWA Project: ELF09202 & ELF09204

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Executive Summary

This report is in response to four Envirolink Requests from Hawke's Bay Regional Council: 625-HBRC76 Catchment Sensitivity, 624-HBRC77 Forecast Future Landuse Impacts, 627-HBRC78 Nutrient Limits and 636-HBRC79 Nutrient Spiralling. These four topics are closely related because many streams in the region are sensitive to nutrients which are affected by landuse change, and so are addressed together in this single report.

Conclusions

1. No reports were sighted which assess the actual or potential level of contamination by agrichemicals.

2. There is evidence that groundwaters approach or exceed drinking water guidelines for faecal coliforms in some places – possibly associated with septic tanks.

3. There is evidence that groundwaters approach or exceed drinking water guidelines for nitrate in some places – possibly associated with septic tanks and soil drainage from intensively grazing, cropping and horticulture.

4. The SOE monitoring wells show lower nitrate concentrations than a subset of 144 wells, and may not be truly representative.

5. Surface water quality in many Hawke's Bay rivers is sensitive to groundwater inflows during summer, and hence to groundwater and surface water abstractions.

6. Groundwater is likely to be a source of nitrate to Hawke's Bay rivers during summer low flows when rivers become N limited.

7. Surface-groundwater interactions are not fully understood although studies are underway to identify gains and losses and to model the Ruataniwha Plains aquifers/rivers.

8. There does not seem to be a need for additional work to identify catchment sensitivity to point source discharges. Existing information on point source discharges needs to be collated and included in catchment-scale analysis.

9. Council is well placed to identify catchments sensitive to over-allocation of surface water using standard methods such as the IFIM.

10. It is not possible from the reports sighted to quantify the extent to which groundwater and surface water allocation adversely affects surface water quality. This is an important question that merits investigation.

11. Biggs (2000) gives a method suitable for assessing the combined effects of nutrient concentration and the time between floods on periphyton biomass which can be used to determine the sensitivity of catchments to increasing nutrient concentration.

12. The impression gained from this 2003 study is that many small streams on the Ruataniwha Plains were not fenced to exclude stock.

13. Council has rather sparse information on the effectiveness of riparian fencing. There is evidence that the majority of large rivers and gulleys are fenced (possibly because the risk of losing stock is high) but that many small streams and ephemeral channels are unfenced.

14. Council has work underway to determine N and/or P limitation. This work will help quantify catchment sensitivity.

15. The Biggs method can be applied where monitoring data exist but it is difficult to determine nutrient sensitivity for the entire catchment.

16. It is desirable to repeat the longitudinal survey work on nutrient spiralling planned for 2007-2008.

17. It is desirable to undertake nutrient addition experiments in conjunction with the longitudinal survey, and to deploy nutrient diffusing substrates.

Recommendations

1. Council in consultation with stakeholders: (1) list the important values in each catchment; (2) identify the main pressures and responses; (3) attempt to rank the relative importance of values in each catchment; (4) compare rankings between different catchments in the Hawke’s Bay region, and adjust the rankings if necessary; and (5) hence identify the key management issues and conflicts in each catchment.

2. Collate available information on landuse, aquifer hydrogeology and nitrate groundwater concentrations. Undertake a desk study to estimate where nitrate concentrations are likely to approach drinking water limits. Review the location of monitoring wells to ensure they adequately monitor high risk areas.

3. Review the suitability of the SOE monitoring wells for detecting increases in groundwater nitrate concentration associated with landuse intensification.

4. Collate available information on (1) river flows, groundwater gain/loss rates and groundwater/river abstraction rates, and (2) landuse, aquifer hydrogeology, groundwater nitrate concentrations, surface water nitrogen concentrations. Undertake a desk study to (1) identify where groundwater gain/loss rates are high and (2) to estimate the importance of groundwater N inputs on river N concentrations, now and in the future.

5. Further work be conducted to quantify the sensitivity of water quality to cattle grazing in ephemeral channels and small streams, and the effectiveness of Council policy to exclude cattle from such areas.

6. Council initially assess catchment sensitivity to nutrient and flow issues which are known to be important in the Hawke's Bay. When the SPARROW/CLUES models for faecal microbes and sediment are available, Council consider using them to refine the assessment of catchment sensitivity.

7. Council review the provisional guidelines and the nutrient equations in Biggs (2000), using local knowledge and Hawke's Bay monitoring data, to ensure they meet management needs.

8. Make an assessment of nutrient sensitivity in catchments where monitoring data exist using the Biggs methods.

9. In a catchment where monitoring data exist adapt the CLUES and Biggs periphyton models to predict nutrient concentrations and periphyton biomass for comparison with observations. If the method proves satisfactory, use it to forecast the effects of future landuse and to assess catchment nutrient sensitivity in other catchments.

10. Calibrate and test the Chapra model using results for the proposed nutrient spiralling study.

11. Re-programme the Chapra model to include multiple sources. Collate available landuse, groundwater gain/loss, groundwater N/P, flow and streamwater N/P, and periphyton biomass data, and use it to calibrate and test the model.

12. Use the model to quantify the distance downstream from point and diffuse sources where periphyton growth and biomass is elevated as a result of nutrient inflows, and to guide the re-calibration of the CLUES model.

1. Introduction

This report is in response to four Envirolink Requests from Hawke’s Bay Regional Council:

- 625-HBRC76 Catchment Sensitivity
- 624-HBRC77 Forecast Future Landuse Impacts
- 627-HBRC78 Nutrient Limits
- 636-HBRC79 Nutrient Spiralling

In request 625-HBRC76 Council seeks a proposal that outlines the work and resourcing requirements to undertake a river catchment sensitivity analysis of the rivers of the Hawke's Bay Region. Gaining an understanding of the sensitivity of river catchments will help Council more effectively implement policies to achieve desirable environmental quality for the region. The advice will be used to develop a programme to characterise the rivers and significant areas of Hawke's Bay, to enable a prioritisation for future investigations based on the differentiation of sensitive areas which will be used for the development of policy and the scoping of regional standards.

In request 624-HBRC77 Council seeks a work programme that outlines methods and resources needed to forecast future landuse impacts on river ecosystems of the Hawke's Bay Region. Forecasting future land use impacts will greatly assist council in developing effective policies and rules on land use within our region. The advice will be used to provide guidance on future work programmes planned as part of the LTCCP process.

In request 627-HBRC78 Council seeks a proposal for a work programme that outlines methods, costs and resources needed to provide nutrient limits in river catchments to prevent excessive periphyton growths during low flow periods. The nutrient limits are to be applied in the regional plan (LTCCP) such that if they are exceeded, landuse mitigation steps must be taken. The advice will be used to programme the work into the LTCCP budget over the 2009/10, 2010/11 and 2011/12 years.

In request 636-HBRC79 Council seeks advice on how to incorporate the modelling of nutrient spiralling to predict nutrient concentrations in rivers. Council notes that nutrient spiralling is often overlooked when managing nutrients in river catchments. Understanding the dynamics of nutrient spiralling will enable Council to better set targets for nutrient management within river catchments. The advice will be used to assess the potential of using nutrient spiralling models to assist with setting nutrient management targets within river catchments.

These four topics are closely related because many streams in the region are sensitive to nutrients which are affected by landuse change, and so are addressed together in this single report.

2. Catchment sensitivity

2.1 Values, pressures and responses

In order to determine catchment sensitivity, it is necessary to identify:

- the important issues (values);
- the processes that adversely affect these values (pressures);
- how values respond to pressures (responses); and
- the parameters used to measure response (parameters).

Table 1 summarises values, pressures, responses and parameters based on the Regional Resources Management Plan (2006) (RRMP) and other literature supplied for this project. There are four important caveats concerning Table 1. First, it makes no attempt to quantify the relative importance of issues. Second, it makes no attempt to identify where in the region these issues arise. Third, it may not be complete. Fourth, for some issues (e.g., point source discharges) policies and procedures are mature while for other issues (e.g., diffuse source nutrient enrichment) the science is complex and policies and procedures are still evolving.

2.2 Phase 1: ranking values

The first phase of catchment sensitivity involves drawing up some variant of Table 1. Much of the information required to refine Table 1 is available within Council but needs to be condensed, and then agreed with stakeholders. To be useful to management Table 1 needs to help identify the key management issues so that attention can be focused on them.

Recommendation: Council in consultation with stakeholders:

1. **list the important values in each catchment;**
2. **identify the main pressures and responses;**
3. **attempt to rank the relative importance of values in each catchment;**

4. **compare rankings between different catchments in the Hawke's Bay region, and adjust the rankings if necessary; and**

5. **hence identify the key management issues and conflicts in each catchment.**

In attempting to rank the relative importance of values, bearing in mind relevant pressures and responses, Council and stakeholders will quickly encounter conflicts. For example, it is obvious that pastoral farming adversely affects water quality and river ecology, and consequently Council and stakeholders will face choices in ranking. For example, '…in this particular catchment are wild, scenic and ecological values more important than pastoral farming…?' Conflicts should not be avoided since their identification and resolution, although difficult, is at the heart of sound management.

It may not be possible to finalise rankings in an individual catchment in the first phase of analysis. For example, almost inevitably there will statements like '…we want wild, scenic and ecological values and pastoral farming…' Detailed analysis and consultation may be required before conflicts can be resolved, compromises explored and agreed and hence rankings finalised.

More detailed analysis may show that cost effective restrictions can be put in place on activities to ensure that particular values are protected. During the first phase of analysis some values may be given equal ranking but the exercise will be pointless unless Council and stakeholders are prepared to be realistic about what can and cannot be achieved.

What should be possible in the first phase of analysis is to compare different catchments. For example '…in the Mohaka river ecology has a higher ranking than does river ecology in the Tukituki River…relative to pastoral farming…' The implication is that more stringent measures will be put in place in the Mohaka than in the Tukituki. For example '…in the Mohaka River pristine water quality will be maintained and this means that no more than x% of the catchment area can be developed for pastoral agriculture…' Note that these examples are by way of illustration and are not recommendations.

2.3 Phase 2: parameter selection

Having listed values and identified conflicts, attention can be focused on more detailed analysis of pressures and responses. This invariably involves modelling – for example to predict the effects of pastoral farming on nutrient concentrations and river ecology. This requires selecting suitable parameters to measure and predict. Parameters need to:

- define the relevant response(s);
- relate quantitatively to the pressure(s); and
- enable suitable guidelines to be set.

Selecting the right parameters requires careful thought. It is not possible or sensible to predict the response of all aspects of the catchment. The standard approach is to predict indicators that are known to be robust surrogates for many aspects of ecosystem health and for which there is a sound modelling framework. Examples include: faecal microbes as indicators of public health risk, and nutrients as indicators of plant growth and associated ecosystem health.

2.4 Phase 3: pressure response analysis

It is beyond the scope of this study to list or review all the available models potentially relevant in the Hawke's Bay. However, in the following sections there is a discussion of models that can be used to help assess catchment sensitivity using nutrients.

Table 1: Summary of values, pressures, responses and parameters. Source: Regional Resources Management Plan (2006).

Value	Pressure	Response	Parameter	Comment
drinking water	sprays, drenches	human health	agrichemicals	orchards, vineyards
drinking water	food processing	taste, odour	BOD, DO	point sources
drinking water	sewage	human health	FC, pathogens	septic tanks
drinking water	land waste disposal	human health	FC, pathogens	sludge & industrial
drinking water	soil drainage	human health	nitrate	
drinking water	stock in streams	human health	FC, pathogens	defecation
ecosystem health	stock in streams	stream habitat	bank/bed erosion	disturbance
ecosystem health	stock in riparian	riparian habitat	shade, temperature	browsing
ecosystem health	contaminated g/w	species diversity	agrichemicals	toxicity
ecosystem health	soil drainage	eutrophication	nitrate	plant growth
ecosystem health	erosion	clarity, siltation	SS	diffuse sources
ecosystem health	irrigation, stock	eutrophication	N & P	diffuse sources
ecosystem health	N & P	periphyton growth	biomass, DO, pH	
ecosystem health	periphyton, DO, pH	invertebrates	MCI	
ecosystem health	periphyton, DO, pH	fish		
ecosystem health	food processing	deoxygenation	BOD, DO	point sources
ecosystem health	oxidation ponds	eutrophication	BOD, DO, nutrients	point sources
ecosystem health	oxidation ponds	public health	FC, pathogens	
ecosystem health	oxidation ponds	toxicity	heavy metals	

2.5 Issues, policies and objectives

Table 1 of the RRMP lists the objectives, and Table 2 the policies, for the region. The objectives most relevant to this report relate to groundwater quantity and quality (OBJ 21-24, 41-43) and surface water quantity and quality (OBJ 25-27, 40). Ground and surface waters are linked because in many parts of the region streams gain flow from, and lose flow to, groundwater and this affects surface water quantity and quality.

2.6 Groundwater

2.6.1 Groundwater quality

The preamble and explanatory notes in Section 3.8 of the RRMP indicate that groundwater contamination arises from several activities, as summarised in Table 1. The parameters vary depending on the activity and, although not stated in Section 3.8, some parameters are listed in Table 1.

Shallow, unconfined aquifers are more sensitive than deep, confined aquifers. Dravid and Brown (1977) identify unconfined and weakly sealed aquifer areas in the Heretaunga Plains that are vulnerable to contamination. Similar information is available for the Ruataniwha Plains.

Groundwater contamination by <u>toxins</u> is known to occur in a small number of locations – notably at tip sites. Council has monitoring in place at these sites and policies to address this problem.

Groundwater contamination by <u>agrichemicals</u> is of concern. Orchards and vineyards are prone to diffuse source contamination from washoff and over-spray while localised point sources may arise from unauthorised disposal of waste agrichemicals (sprays and drenches). The location of orchards and vineyards is well documented and Council is well placed to identify catchments sensitive to problems from agrichemicals. Baalousha (2008b) alludes to wells monitored for pesticides.

Conclusion: no reports were sighted which assess the actual or potential level of contamination by agrichemicals.

Groundwater and surface water contamination by <u>pathogens</u> is also of concern. The principal sources are septic tanks and land disposal of wastes. Baalousha (2008b) collates data from a survey of 144 shallow wells (<25 m) conducted in 2003-2004. Faecal coliforms above the Ministry of Health drinking water guideline (>1 cfu per 100 mL) were found in 17 wells (12%). All contaminated wells were shallow and landuse in the vicinity was farming, horticulture or rural residential. There are over

1700 shallow wells (<25 m) in the Hawke's Bay region and the 144 sampled are a carefully chosen subset. Mapped results indicate that small coastal communities have a high incidence of exceedance – possibly associated with septic tank drainage.

Conclusion: There is evidence that groundwaters approach or exceed drinking water guidelines for faecal coliforms in some places – possibly associated with septic tanks.

Groundwater contamination by nutrients (notably nitrogen N and phosphorus P) are important issues. The main pressures are:

- diffuse drainage and/or runoff from agriculture;
- land disposal of wastes; and
- diffuse sources including septic tanks.

Nitrate is highly mobile and finds its way into rivers from agriculture and septic tanks via groundwater. Urine patches, drainage from effluent irrigation areas and drainage from cropping land are major sources of nitrate. Phosphorus tends to be less mobile than nitrate and is transported principally by erosion. The pressure from agriculture is increasing, especially with dairy conversions.

Two values identified in the RRMP as being affected by nutrients are:

- drinking water quality (affected by nitrate); and
- surface water quality (affected by nitrogen and phosphorus).

The RRMP states that '…nitrate exceeded drinking water standard in samples collected at Bridge Pa…' (RRMP 2006 page 49). Baalousha (2008b) collates nitrate data from 144 shallow wells (<25 m). Nitrate concentrations above the Ministry of Health drinking water guideline (>11.3 mgN L^{-1}) were found in 8 wells (6%) and concentrations >50% of the guideline in another 6 wells (4%). All contaminated wells were shallow. Mapped results indicate some contaminated wells in small coastal communities – possibly associated with septic tank drainage – and some where landuse was farming or horticulture – probably associated with soil drainage. As mentioned earlier the 144 wells sampled are a carefully chosen subset of the ~1700 shallow wells (<25 m) in the Hawke's Bay region.

Conclusion: There is evidence that groundwaters approach or exceed drinking water guidelines for nitrate in some places – possibly associated with septic tanks and soil drainage from intensively grazing, cropping and horticulture.

Baalousha (2007a) reviewed data from SOE monitoring wells and found that the average nitrate concentration in the Hawke's Bay region is 1.2 mg L^{-1}. This is well below the drinking water standard (11.3 mgN L^{-1}). Baalousha (2007b) states that '…all the analysed chemical parameters are within the limit of the New Zealand drinking water standards…' These statements are at variance with Baalousha (2008b).

There appear to be 'hot spots' of higher than average nitrate concentration – possibly the Bridge Pa well is one such. It appears that the SOE wells were selected because they are somewhat remote from the localised effects of abstraction. However, this means they may also be remote from sources of contamination. If so then the SOE monitoring wells may yield lower nitrate concentrations than other parts of the aquifer and may not give early warning of the adverse effects of land-use intensification.

Conclusion: the SOE monitoring wells show lower nitrate concentrations than a subset of 144 wells, and may not be truly representative.

Recommendation: Collate available information on landuse, aquifer hydrogeology and nitrate groundwater concentrations. Undertake a desk study to estimate where nitrate concentrations are likely to approach drinking water limits. Review the location of monitoring wells to ensure they adequately monitor high risk areas.

High nitrate in groundwater poses a threat to river water quality and ecosystems where rivers gain groundwater during summer low flows. This is an emerging issue that is currently poorly understood.

Recommendation: Review the suitability of the SOE monitoring wells for detecting increases in groundwater nitrate concentration associated with landuse intensification.

Further discussion of this issue is included below.

2.6.2 Groundwater quantity

Section 3.9 of the RRMP outlines objectives and policies aimed at controlling groundwater takes so that, amongst other things, there are no significant adverse effects on surface water resources. Groundwater is relevant to this report to the extent that summer droughts are common in Hawke's Bay and low flows in many rivers are maintained by groundwater inflows. If aquifers become depleted (naturally, through over-allocation, or through climate change) then stream low flows are likely to decrease.

Hawke's Bay rivers naturally experience low flows during summer. Water quality is usually lower during summer low flows (e.g., high temperatures, high plant biomass, fluctuating dissolved oxygen and pH, loss of sensitive macroinvertebrates and fish) than at other times of the year. To a certain extent indigenous biota have adapted to summer low flow conditions, although sensitive organisms (including the introduced trout) become stressed during extreme summer low flow conditions. The potential exists for over-allocation of groundwater and/or climate change to further reduce minimum flows.

Conclusion: Surface water quality in many Hawke's Bay rivers is sensitive to groundwater inflows during summer, and hence to groundwater and surface water abstractions.

A second groundwater quality issue is that where groundwater feeds streams, high nitrate concentrations in groundwater pose a potential threat to surface water quality because it may stimulate plant growth. This is a significant issue in the central volcanic plateau where streams are nitrogen limited and nitrate concentrations in streams have increased in recent years as a result of historical and recent landuse intensification (Vant and Smith 2004).

For much of the year the ratio of SIN/DRP is high suggesting P limitation (Ausseil 2008). However, at very low flows SIN concentrations drop below detection limits indicating N limitation. Wilcock et al. (2007) recommend the management of both N and P under such circumstances.

The average groundwater nitrate concentration in the Hawke's Bay of 1.2 mg L^{-1} does not pose a human health risk but is of environmental significance. The RRMP does not set guidelines for SIN concentrations in Hawke's Bay rivers but ANZECC (2000) suggests values of 0.120 and 0.440 mg L^{-1} for the control of nuisance algal growths. Clearly the average groundwater nitrate concentration of 1.2 mg L^{-1} exceeds these guidelines.

Conclusion: Groundwater is likely to be a source of nitrate to Hawke's Bay rivers during summer low flows when rivers become N limited.

HBRC (2006) and Baalousha (2008a) make it clear that surface-groundwater interactions are not fully understood or quantified. Groundwater is known to flow into the Tukituki River in the lower parts of the Ruataniwha Plains – an area where intensive agriculture is already occurring and further intensification is forecast. Ausseil (2008) points out that SIN concentrations increase significantly in this part of the Tukituki River. It seems a reasonable inference that this SIN comes from agricultural runoff and that a significant proportion is delivered via groundwater. Brooks (2007b) recommended developing a groundwater model for the Ruataniwha Plains and this work is underway. HBRC (2006) mentions a research study that seeks to identify patterns of gain and loss along river reaches (Study 411-24) but no results have been sighted.

Conclusion: Surface-groundwater interactions are not fully understood although studies are underway to identify gains and losses and to model the Ruataniwha Plains aquifers/rivers.

It is desirable to quantify the importance of groundwater as a source of nutrients, and the effects of land use changes.

Recommendation: Collate available information on (1) river flows, groundwater gain/loss rates and groundwater/river abstraction rates, and (2) landuse, aquifer hydrogeology, groundwater nitrate concentrations, surface water nitrogen concentrations. Undertake a desk study to (1) identify where groundwater gain/loss rates are high and (2) to estimate the importance of groundwater N inputs on river N concentrations, now and in the future.

2.7 Surface water

2.7.1 Point source discharges

Section 3.10 of the RRMP states that point-source discharges affect surface water quality including sewage discharges from inland communities (e.g., Waipukurau and Waipawa) and food processing wastes. It is clear that Council has well developed policies and procedures for dealing with point source discharges. Although some stakeholders have raised concerns about poor water quality (e.g., below oxidation pond discharges into the Tukituki River) there are well established procedures in respect of point source discharges. The major point sources of contamination are known to Council and hence catchments sensitive to point source discharges are known.

Conclusion: There does not seem to be a need for additional work to identify catchment sensitivity to point source discharges. Existing information on point source discharges needs to be collated and included in catchment-scale analysis.

Point source discharges are assessed for their impact on temperature, suspended solids, BOD, DO and pathogens. In addition, point source discharges are assessed for their impact on nutrient concentrations because of their effect on periphyton and other ecosystem values. Consents issued for the Waipukurau and Waipawa sewage treatment plants limit the discharge of phosphorus.

2.7.2 Diffuse sources

Whereas proven policies and procedures exist for point source discharges, the situation is different for contamination arising from diffuse source discharges. This situation is not unique to the Hawke's Bay – elsewhere in New Zealand, and overseas, managers struggle with the complexity of the science and the social issues associated with diffuse source pollution.

Diffuse source discharges are recognized in the RRMP including diffuse-source runoff, sedimentation, and bacterial contamination (page 61). Arguably, diffuse source contamination by pathogens and nutrients are the two biggest challenges currently facing managers in the Hawke's Bay and elsewhere in New Zealand.

2.7.3 Surface water objectives

Section 3.10 of the RRMP (pages 61 and 99) state the following objectives:

'…maintenance of water quantity of the rivers and lakes in order that it is suitable for sustaining aquatic ecosystems in catchments as a whole and ensuring resource availability for a variety of purposes across the region, while recognizing climatic fluctuations…'

'…maintenance or enhancement of water quality of rivers, lakes and wetlands in order that it is suitable for sustaining or improving aquatic ecosystems in catchments as a whole, and for contact recreation where appropriate…'

'…maintenance of the water quality of specific rivers in order that the existing species and natural character are sustained while providing for resource availability for a variety of purposes, including groundwater recharge…'

These objectives establish that rivers, lakes and wetlands are to be managed on the one hand for aquatic ecosystems (and where appropriate for contact recreation) while on the other hand providing water for a variety of purposes. These purposes include consumptive uses (e.g., abstraction for stock water, irrigation, and industrial use) which reduce flow. They also include practices that increase nutrient inputs (e.g., land disposal of wastes, septic tank discharges, point source discharges of sewage and food processing wastes, and landuse practices including pastoral farming and horticulture).

2.7.4 Surface water quantity

Articles in the press indicate that conflict exists between stakeholders and council on water allocation. On the one hand, primary producers want more water to increase production (e.g., orchards, vineyards, cropping land and dairy pasture) while on the other hand, recreationalists and ecologists want adequate residual flows to maintain water quality and ecological integrity in streams, lakes and wetlands. It is clear that Council recognises this conflict and has been handling it for many years. The RRMP sets out objectives and policies for setting minimum flows and controlling abstractions when river flows become critically low. River flows in the region are monitored at key locations and licensed water allocations are documented.

Conclusion: Council is well placed to identify catchments sensitive to over-allocation of surface water using standard methods such as the IFIM.

It is beyond the terms of reference of this study to review the methods used to set minimum flows.

2.7.5 Surface water quality

Groundwater/surface water interactions

One emerging issue that needs to be addressed is the combined effect of decreasing river flows and increasing nutrient inputs on water quality. The potential exists for decreasing flows to interact with increasing nutrient inputs to degrade water quality and adversely affect river ecosystems. Two alternative hypotheses are advanced by stakeholders. The first is that over-allocation has reduced summer low flows and that this has lead to worsening water quality. The second is that flows are naturally low in summer and that poor summer water quality occurred prior to recent high allocation.

Conclusion: It is not possible from the reports sighted to quantify the extent to which groundwater and surface water allocation adversely affects surface water quality. This is an important question that merits investigation.

Multiple stressors

The effects of multiple stressors are not easy to quantify and are the subject of ongoing research. Council is not alone in having policies and procedures that address single stressors (viz., separate guidelines for flow and nutrients). Currently robust methods are not available which quantify the combined effects of multiple stressors (e.g., flow, nutrient, sediments, temperature and light – which together determine periphyton biomass) although these are being developed (Rutherford et al. 1999). The Periphyton Guideline (Biggs 2000) outlines a method for assessing the combined effect of nutrient concentrations and the time between floods, and this is detailed later in this report.

Conclusion: Biggs (2000) gives a method suitable for assessing the combined effects of nutrient concentration and the time between floods on periphyton biomass which can be used to determine the sensitivity of catchments to increasing nutrient concentration.

While the Biggs approach has limitations, more sophisticated models are still at the development phase.

2.8 Riparian protection

Riparian margins are seen by Council as one way of addressing problems with diffuse source pollution including nutrient and sediment runoff (RRMP page 67).

Where cattle have access to streams they increase pathogen and sediment inputs. Cattle probably have a second-order effect on nutrient inputs. Policy 45 sets out non-regulatory methods to encourage landowners to fence riparian zones, exclude cattle and provide a buffer against the adverse effects of landuse. Council makes some money available for fencing and re-planting.

Sarazin and Zimmerman (2003) surveyed 320 km of riparian habitat along 5 rivers and 18 streams on the Ruataniwha Plains. The main stems of the Waipawa and Tukituki Rivers were assessed as having ‘good’ overall riparian buffer scores. Many of the tributaries, however, had ‘poor’ or ‘very poor’ buffer scores. The authors concluded that ‘…the best way to improve scores is to fence off stream margins from stock...and…introduce plants that will provide adequate shade…’

Conclusion: the impression gained from this 2003 study is that many small streams on the Ruataniwha Plains were not fenced to exclude stock.

In 2006-2007 a student surveyed fencing around spring-fed streams in the Ruataniwha – about 40-50% were fenced on both sides (Andrew Curtis, *pers. comm.*). It was noted that stock were sometimes deliberately being grazed inside well fenced buffers.

Spring-fed streams are easy to fence but in other streams flood damage is a problem. In one flood-prone stream, riparian fences have been rebuilt 3 times in 5 years.

It is estimated by Council staff that:

- about 75% of all streams in the Ruataniwha are unfenced;
- the majority of ephemeral channels (dry in summer) are unfenced;
- about 40% of main channels that flow all year in the eastern part are unfenced;
- the majority of steeper gulleys in the western part are fenced; and
- even where fenced stock are often present in the riparian zone.

It must be stressed that this is an assessment and is not based on detailed site surveys.

Many riparian buffers are periodically grazed to 'clean them up'. While there are merits in short, controlled grazing of grass filter strips (to remove the nutrient they have trapped) there is anecdotal evidence of prolonged and uncontrolled grazing by cattle within riparian buffers which negates the benefits of fencing.

No reports were sighted that summarise Council expenditure on riparian fencing although staff offered to 'dig out' the data. Staff also pointed out that some farmers do their own riparian fencing without Council subsidies.

Conclusion: Council has rather sparse information on the effectiveness of riparian fencing. There is evidence that the majority of large rivers and gulleys are fenced (possibly because the risk of losing stock is high) but that many small streams and ephemeral channels are unfenced.

Recommendation: further work be conducted to quantify the sensitivity of water quality to cattle grazing in ephemeral channels and small streams, and the effectiveness of Council policy to exclude cattle from such areas.

2.9 Nutrient modelling

An emerging issue in the Hawke's Bay and elsewhere in New Zealand is the effect of landuse intensification on water quality and river ecosystems. As discussed in Section 2.8.5 this is technically complex. Nevertheless, models do exist for predicting the effects of landuse on nutrient concentration, and for nutrient concentration and flow on periphyton biomass. These models can be used by Council to help identify the sensitivity of catchments to landuse intensification and changes in flow. The models suggested for this task are described in Sections 3 and 4.

2.10 Faecal microbes and sediments

Council may wish to assess the sensitivity of catchments to faecal microbes and sediments. Currently the SPARROW model is being modified to address both of these issues. SPARROW is a component of the CLUES model. However, this work is not yet complete.

Recommendation: Council initially assess catchment sensitivity to nutrient and flow issues which are known to be important in the Hawke's Bay. When the SPARROW/CLUES models for faecal microbes and sediment are available, Council consider using them to refine the assessment of catchment sensitivity.

3. Nutrient limits

This report is not intended to quantify the effects of eutrophication in any particular catchment or to recommend nutrient limits. Rather it is intended to outline a framework (viz., the information required, and the steps to be followed) for determining the sensitivity of catchments to flow and nutrients.

A catchment is sensitive to nutrient enrichment if:

- its water quality and ecosystems respond to nutrient inputs;

  and

- there are controllable point or diffuse nutrient sources in the catchment.

Uncontrollable nutrient sources must not be so large that they cause unsatisfactory water quality and/or adversely affect ecosystems – clearly in this situation action to reduce controllable sources would be ineffective. A catchment is sensitive to flow reductions if its water quality and ecosystems respond to decreasing flow and flows in the catchment are controllable – natural low flows may be so large that they adversely affect water quality and ecosystems.

A methodology is outlined below that enables an assessment to be made of the sensitivity to nutrients and flow of periphyton biomass in the rivers of a catchment. The methodology should help review and if necessary revise nutrient limits for Hawke's Bay rivers.

3.1 Periphyton

Rivers in the Hawke's Bay are sensitive to the proliferation of periphyton. Periphyton is the slime and algae found on the bed of streams and rivers. This group of organisms is essential for ecosystem functioning but under certain circumstances periphyton can proliferate, causing management problems including degrading aesthetic, recreational and biodiversity values; fluctuating dissolved oxygen concentration and pH; and the loss of fish and sensitive macroinvertebrates. Proliferations can also taint water, make it toxic to stock, and clog abstraction intakes. Hawke's Bay rivers are particularly prone to periphyton proliferations because of the shallow, gravel/cobble nature of river beds, high sunlight, warm waters and nutrient enrichment from natural and anthropogenic causes.

3.2 Nutrient limitation and trophic state

Periphyton growth rate and biomass respond to nutrient inputs where one or more nutrients (typically nitrogen N or phosphorus P) is in short supply. Periphyton biomass alone is not always a reliable indication of trophic state. If flows are stable for long periods of time, high biomass can accumulate in rivers with low nutrient inputs through slow growth over a long period of time. However, consistently high periphyton biomass in rivers with frequent high flows (that mobilise the bed and scour plants) indicates a eutrophic river.

An important step in determining the sensitivity of streams in a catchment to nutrient input is to determine whether periphyton growth rate is limited by the availability of N or P, both N and P, or neither N or P. Where nutrient concentrations are high, plant growth rate and biomass may not be sensitive to further nutrient input. At high nutrient concentration, plants become 'saturated' and growth rate reaches a plateau. Plant biomass is then controlled by factors other than nutrient (e.g., shade, flow disturbance, temperature, grazing and mobile substratum). On the other hand where stream nutrient concentrations are low, plant growth rates increase where nutrient inputs occur. This does not necessarily lead to higher plant biomass in situations where other factors are limiting. However, when and where these other factors cease to be limiting (e.g., during summer low flows) then high biomass is more likely in streams where nutrient inputs are high.

In lakes trophic status is commonly defined based on annual mean concentrations of total phosphorus and total nitrogen (Vant 1987). Thus oligotrophic lakes (low nutrient) are those in which annual mean TP < 30 mg m^{-3} and/or TN < 300 mg m^{-3} where TP and TN = total phosphorus and total nitrogen respectively.

The situation is different in rivers because nutrient residence times are low. In winter nutrients rarely cause water quality problems because plant growth rates are limited by light and temperature, and high flows prevent the accumulation of biomass through scour. Nutrient inputs may increase periphyton growth rates, leading to increased biomass accrual rates. However, periphyton biomass depends not only on growth rate but also the interval between high flows that 'reset' periphyton biomass (Biggs 2000). Most problems with periphyton occur during stable, low flow periods in summer when high light and water temperature promote high growth rates, and scour losses are low.

During low flow periods when periphyton are actively growing, dissolved inorganic nitrogen (DIN) and dissolved reactive phosphorus (DRP) concentrations are poor indicators of trophic state. This is because when plants are actively growing, they consume DIN and DRP thereby lowering their concentration. As a result correlations are often poor between periphyton biomass and DIN and/or DRP concentration (Biggs

2000). Correlations are also poor between periphyton biomass and TN and/or TP concentration – in marked contrast to lakes. The Periphyton Guideline (Biggs 2000) overcomes this problem by using the annual average of monthly mean DIN and/or DRP concentration to define trophic state. This avoids the bias likely when using just growing season concentrations which may be depleted by plant uptake. On the other hand it includes high winter values that have little effect on summer periphyton. Nevertheless, the guideline is robust because it is based on correlations of measured maximum and summer mean periphyton biomass with the annual average of measured monthly mean DIN and/or DRP concentrations. Biggs (2000) points out that the equations used in the Guideline were derived from a subset of South Island streams and may not be applicable everywhere in New Zealand. Nevertheless, the method is valuable and can be adapted to the Hawke's Bay.

3.3 Which nutrient is limiting?

Groundwater in the central volcanic plateau are naturally high in P and there is evidence that lakes and streams are N limited. Some streams in Hawke's Bay drain volcanic soils and are similar to those around Taupo and Rotorua. Other Hawke's Bay streams (e.g., those draining greywacke) do not have naturally high P concentrations. Plant growth in the latter streams may not respond to increased N inputs if P is in short supply. However, if P inputs also occur (e.g., agricultural runoff, septic tank drainage, land disposal, sewage discharges) then the combined effects of elevated N and P concentrations may be to stimulate plant growth, with resulting water quality problems especially during summer low flows.

The most direct way to determine which nutrient, if any, is limiting is to conduct bioassays. Methods include laboratory incubations, enzyme measurements and diffusing substrates. Some work has been done in the Mohaka River using diffusing substrates (Stansfield 2008). It is not clear whether a sufficiently large number of streams in the region have been studied to allow reliable mapping of nutrient limitation. If nutrient limitation has not already been quantified in other catchments, then it should be a priority as part of a nutrient sensitivity assessment.

In the absence of bioassays, insights can be gained from nutrient measurements by examining N/P ratios. Care must be taken when using this method. If plant biomass is low, plants are not growing or nutrient concentrations are high then nutrient ratios are not reliable. Aquatic plants utilise dissolved inorganic forms of nitrogen and phosphorus. When plants biomass is high and plants are actively growing (e.g., during summer low flows) the absence of dissolved inorganic nitrogen (DIN) and the presence of dissolved reactive phosphorus (DRP) is an indication of nitrogen limitation, and *vice versa*. Low biomass accrual rates at such times confirm nutrient limitation.

The material sighted indicates that Council is in the process of determining whether N or P limitation occurs in Hawke's Bay rivers. Results to date suggest that the Tukituki River is P limited at moderate flows but becomes N limited at very low, summer flows (Ausseil 2008). In the Mohaka nutrient limitation varies moving from the headwaters to the lower reaches (Stansfield 2008).

Conclusion: Council has work underway to determine N and/or P limitation. This work will help quantify catchment sensitivity.

3.4 Sensitivity to nutrients

The Periphyton Guideline (Biggs 2000) sets out a methodology for estimating periphyton biomass based on nutrient concentration and the time between floods. This can be used to determine the sensitivity of a catchment to nutrient enrichment and to forecast the effects of future landuse. The Guideline also gives provisional values for periphyton biomass that maintains aesthetic, recreational, biodiversity and angling values, as summarised below.

Table 2: Provisional biomass and cover guidelines. Source Biggs (2000).

Instream value/variable	Diatoms/cyanobacteria	Filamentous algae
Aesthetics/recreation (1 Nov-30 April)		
Maximum cover of visible stream bed	60% >0.3 cm thick	30% >2 cm long
Maximum AFDM (g/m^2)	N/A	35
Maximum chlorophyll *a* (mg m^{-2})	N/A	120
Benthic biodiversity	15	15
Mean monthly chlorophyll *a* (mg m^{-2})	50	50
Trout habitat and angling		
Maximum cover of whole stream bed	N/A	30 % >2 cm long
Maximum AFDM (g/m^2)	35	35
Maximum chlorophyll *a* (mg m^{-2})	200	120

The Guideline contains relationships between peak periphyton biomass and the primary controlling variables of time between flood events (accrual time) and nutrient concentration (monthly mean concentrations measured over at least one year).

$$\log_{10}(C_{max}) = 4.29 \log_{10}(T_a) - 0.929 \log_{10}(T_a)^2 + 0.504 \log_{10}(DIN) - 2.95 \qquad 1$$

$$\log_{10}(C_{max}) = 4.72 \log_{10}(T_a) - 1.08 \log_{10}(T_a)^2 + 0.494 \log_{10}(DRP) - 2.74 \qquad 2$$

where DIN = soluble inorganic nitrogen (mg m^{-3}), DRP = soluble reactive phosphorus (mg m^{-3}), C_{max} = maximum chlorophyll concentration (mg m^{-2}) and T_a = accrual period (days). These equations explained about 75% of the variance in the observations used in their development.

It may be desirable to refine Equations 1 and 2 from the Periphyton Guideline (Biggs 2000) to better suit Hawke's Bay conditions. This would involve using local monitoring data to re-calibrate the equations, or to develop similar equations with slightly different independent variables.

A refinement that merits consideration is to calculate the average DIN and/or DRP concentrations in spring or summer when periphyton biomass is low (viz., while light and temperature limit growth, or after floods have reduced biomass to low levels). This approach was developed in lakes where maximum summer phytoplankton biomass was found to be correlated with spring phosphorus concentration and maximum summer diatom biomass with spring silica concentration. Before this method could be used, periphyton biomass and nutrient concentrations would need to be monitored in Hawke's Bay streams and the results used to develop correlations between nutrient concentrations and periphyton biomass. Some data already exist but SOE monitoring only measures periphyton biomass once per year which may not be often enough to obtain a reliable re-calibration.

Equations 1 and 2 were used by Biggs (2000) to develop nutrient guidelines for various growth periods to ensure that peak biomass does not exceed the biomass guidelines for the various instream values as summarised below (Table 3).

The RRMP gives a guideline DRP of 15 mgP m^{-3} but gives no guideline for DIN. Calculations summarised in Table 3 suggest that during summer low flows >30 days a DRP concentration of 15 mgP m^{-3} would not prevent periphyton from exceeding guideline values.

Recommendation: Council review the provisional guidelines and the nutrient equations in Biggs (2000), using local knowledge and Hawke's Bay monitoring data, to ensure they meet management needs.

Table 3: Concentrations of dissolved inorganic nitrogen (DIN) and dissolved reactive phosphorus (DRP) that prevent periphyton biomass from exceeding guideline values, as a function of time between floods (accrual time). Source Biggs (2000).

Study	Chlorophyll *a* = 50 mg m^{-2}		AFDM=35 g m^{-2} Chlorophyll *a* = 120 mg m^{-2} filamentous Chlorophyll *a* = 200 mg m^{-2} mat forming	
Accrual time days	DIN mg m^{-3}	DRP mg m^{-3}	DIN mg m^{-3}	DRP mg m^{-3}
20	<20	<1	<295	<26
30	<10	<1	<75	<6
40	<10	<1	<34	<3
50	<10	<1	<19	<2
75	<10	<1	<10	<1
100	<10	<1	<10	<1

3.5 Assessing sensitivity in monitored catchments

The steps required to assess nutrient sensitivity in a catchment where there are nutrient and periphyton monitoring data are:

1. determine the limiting nutrient (N or P);
2. analyse flow records to estimate accrual times;
3. analyse water quality monitoring results to estimate annual average DIN and/or DRP concentrations;
4. use Equation 1 or 2 to estimate periphyton biomass; and
5. vary DIN or DRP concentrations and accrual times to determine the sensitivity of periphyton biomass.

Step 2. Accrual times vary from year to year, and a long flow record is required to reliably estimate accrual times (e.g., mean annual, 5-year maximum etc.). This is straightforward in rivers with flow recorders but poses challenges in ungauged rivers. Step 3. Biggs (2000) recommends using annual average DIN or DRP concentrations based on water quality monitoring for at least one year, preferably longer. Again this is straightforward in rivers with monitoring data but poses challenges in ungauged rivers. Equations 1 and 2 could be re-calibrated using local monitoring data, in order to better quantify Hawke’s Bay rivers. Council monitor periphyton biomass once a year after a prolonged period of summer low flow. Data from several years would enable testing of the equations, and may be sufficient for their re-calibration if necessary. Step 5. Care must be taken selecting realistic combinations of flow and concentration. Flow and concentration may be correlated. At low flows periphyton may remove DIN and/or DRP thereby reducing (biasing) concentrations. At high flows DIN and/or DRP may be elevated above the concentrations during growth periods.

This methodology relies on monitoring to provide input data on DRP or DIN concentration. During summer low flows DRP and DIN concentrations vary along a river in response to inputs, periphyton uptake and recycling. Nutrient data collected at one point may not accurately describe concentrations at another point.

Conclusion: The Biggs method can be applied where monitoring data exist but it is difficult to determine nutrient sensitivity for the entire catchment.

It is beyond the scope of this study to review the available monitoring data, and comment on which rivers can, which cannot, be analysed by this method. Where reliable and extensive flow and nutrient data exist the sensitivity analysis should be robust. Where there are few flow and nutrient data, useful insights into potential sensitivity can still be gained. It may be necessary to infer low flow statistics from another catchment with similar catchment area, rainfall, topography and geology. It may be necessary to infer nutrient concentrations from another catchment with similar geology, soils and land use. Biggs (2000) shows that conductivity is a useful surrogate for nutrient concentration and this could be exploited in catchments where nutrient data are sparse but conductivity is known. Conductivity is correlated with geology and in the absence of both nutrient and conductivity data, this may enable an estimate to be made of nutrient concentrations that is sufficiently reliable for a preliminary assessment of sensitivity.

Recommendation: Make an assessment of nutrient sensitivity in catchments where monitoring data exist using the Biggs methods.

4. Forecasting the effects of future landuse

Models such as OVERSEER (farm scale) and CLUES (catchment scale) can be used to estimate nutrient loads based on information on landuse, soils and climate. Both OVERSEER and CLUES (which includes OVERSEER) predict annual average nutrient loads which is consistent with the input requirements of Equations 1 and 2 for predicting periphyton biomass. However, the models predict TN and TP rather than DIN and DRP concentrations, and this currently limits their usefulness.

Currently CLUES is the most suitable model for examining the effects of landuse change on nutrient loads at catchment scale. Desirable features of CLUES are:

- it incorporates the nutrient model SPARROW;
- SPARROW has been calibrated against monitoring data from the national Water Quality Monitoring Network;
- SPARROW quantifies attenuation;
- CLUES is easy to use;
- CLUES can be used to predict the effects of landuse change on annual average TN and/or TP loads.

For the case where P is the limiting nutrient this would involve:

1. using CLUES to predict annual average TP load knowing landuse;
2. converting annual TP load to annual average DRP concentration;
3. using Equation 1 or 2 from Biggs (2000).

Steps 1 and 3 are straightforward. Step 2 requires data on river flows, the correlation between flow and TP concentration, and the correlation between TP and DRP concentration. It would be necessary to estimate this using available monitoring data. Step 2 would require careful analysis and is not without risk of high uncertainty.

CLUES does not do everything Council require when assessing catchment nutrient sensitivity. Notably CLUES does not:

- simulate nutrient spiralling although it does predict nett attenuation;
- predict summer low flow DIN or DRP concentrations (as required by the Periphyton Guideline – see earlier discussion).

Work is underway to refine CLUES so that it predicts summer low flow concentrations as well as annual means. This will probably be done by correlating summer low flow and annual mean concentrations at a national scale. The success of these refinements will not be known until next year. It would be preferable to do a similar exercise at a local level and not rely on the national re-calibration of CLUES. The steps involved would be to

- use CLUES to model the effects of land use change on annual mean TN and TP concentration;
- correlate DIN and DRP concentrations to TN and TP concentrations using local Hawke's Bay data;
- hence estimate annual average DIN and DRP concentrations;
- use the Periphyton Guideline equations (or refinements) to estimate periphyton biomass, knowing accrual time.

Alternatively it would be preferable (if the data allow) to:

- recalibrate the Biggs equations using Hawke's Bay data to predict periphyton biomass from annual average TN and TP concentration or annual load, and accrual time;
- use CLUES to model the effects of land use change on annual average TN and TP concentration or annual load;
- hence estimate periphyton biomass, knowing accrual time.

Underpinning this approach is the assumption that periphyton biomass is a good indicator of ecosystem health. This is a reasonable assumption provided the guideline values for periphyton are appropriate. An essential step in this approach is therefore to review the available monitoring data and determine the desired maximum summer low flow periphyton biomass to ensure ecosystem health.

Recommendation: In a catchment where monitoring data exist adapt the CLUES and Biggs periphyton models to predict nutrient concentrations and periphyton biomass for comparison with observations. If the method proves satisfactory, use it to forecast the effects of future landuse and to assess catchment nutrient sensitivity in other catchments.

5. Nutrient spiralling

A major difficulty applying the Periphyton Guideline (Biggs 2000) is to estimate DIN or DRP concentrations for input into Equations 1 or 2. Similarly if local monitoring data is used to refine the Periphyton Guideline then the difficulty is to decide what nutrient concentrations to use as independent variables to correlate to periphyton biomass. These difficulties arise because the two related process termed ‘attenuation’ and ‘nutrient spiraling’.

5.1 Attenuation

Nutrient attenuation is the permanent loss or temporary storage of nutrient between where it is generated (e.g., in the paddock) and where it affects water quality (e.g., in the lake). Natural loss mechanisms include: denitrification, conversion to refractory dissolved organics and burial of particulates. Storage mechanisms include: uptake of dissolved inorganic forms by attached plants, uptake of dissolved organic forms by heterotrophic biofilms, adsorption to minerals and settling of particulates. Release mechanisms from storage include: remineralisation, erosion and scour.

Several large catchment studies have found that nutrient transported at the outlet is significantly less than the sum of the inputs. Behrendt & Opitz (2000) studied 100 large river basins in Europe (121-194,000 km^2) and found that at low hydraulic load (<10 m yr^{-1}, where hydraulic load = mean depth/residence time) attenuation averaged ~20-30% (range 10-40%) for TP and DIN. Attenuation decreased with increasing hydraulic load and was ~0 at >200 m yr^{-1}. A similar relationship exists between hydraulic load and nutrient retention in lakes. Williams et al. (2004) found that the Ipswich River basin, northeastern Massachusetts attenuates (retains) about 50% of gross N inputs, mostly in terrestrial components of the landscape. Loss and retention of total nitrogen (TN) in the aquatic environment was about 9% of stream loading. Basin-wide losses due to aquatic denitrification were considerably lower than estimates from several recent studies and range from 4 to 16% (average 9%) of TDN in stream loading. These conclusions are very much dependant on how accurately the *inputs* were estimated – clearly a lot tougher than measuring the outputs.

In four sub-basins of the Waikato (2,700-4,610 km^2) Alexander et al. (2002) calibrated the SPARROW model and used it to estimate that TN & TP retention ranged from 39-89%. The largest retention occurred in Lake Taupo. Studies in small, intensively farmed catchments have found *quite low attenuation*. Monaghan et al. (2007) found that the measured yield at the outlet of the 2,480 ha Bog Burn catchment, Southland, for 2001-05 was 8.2 kgN ha^{-1} yr^{-1}. Using OVERSEER the predicted N-leaching at farm scale totalled 10.1 kgN ha^{-1} yr^{-1}. The implied attenuation is only ~20%.

Phosphorus attenuation estimated in the same manner was ~30%. However, in a recent email discussion Ross Monaghan believes that OVERSEER may over-estimate N-leaching, in which case attenuation in Bog Burn may be negligible. Wilcock et al. (2006) found that in 1995-97 the TN load at the outlet of the Toenepi catchment averaged 35 kgN ha^{-1} yr^{-1} and that in 2002-04 it was 13 kgN ha^{-1} yr^{-1}. 2002-04 (mean flow 176 L s^{-1}) was 40% drier than 1995-97 (289 L s^{-1}) and if yield is proportional to mean flow then the 1995-97 yield 'adjusted' to 2002-04 flows is 21 kgN ha^{-1} yr^{-1}. In 1995-97 95% of dairy sheds discharged to streams via treatment ponds whereas in 2002-04 this had dropped to 78%. OVERSEER runs indicate that a shift to effluent land application reduces farm scale N losses by 3-7 kgN ha^{-1} yr^{-1} (Ross Monaghan, pers. comm.). The 'adjusted' yields for Toenepi (~20 kgN ha^{-1} yr^{-1} (1995-97) and 13 kgN ha^{-1} yr^{-1} (2002-04)) are at the lower end of the published range for nitrate leaching rates from dairy pasture (typically 20-50 kgN ha^{-1} yr^{-1}). However, the significant difference between periods cannot be explained from the available hydrological and agricultural statistics. This illustrates the difficulties of estimating nutrient losses from agricultural land even when landuse and farming practice are well documented, and hence the difficulty in estimating nutrient attenuation from the difference between estimated nutrient inputs and yields measured at the catchment outlet. At Tutaeuaua near Taupo, Macaskill and Broekhuizen noted that nitrate concentrations in the main stem of the river channel were spatially uniform – suggesting that instream removal rates in the main channel were low. This is consistent with the main channel having very few aquatic plants and a mobile sand bed. However, a detailed nutrient budget has not yet been developed and so it is not clear whether input is small, or input and uptake are equal.

A 'rule of thumb' is that about 50% of nutrient inputs fail to reach the catchment outlet. This 50% attenuation figures comes from work in several North Island catchments including Lake Taupo, the Waikato River and the Manawatu River. There is a large uncertainty in this estimate but it may be a useful guide when making a preliminary assessment of catchment sensitivity. For example if nutrient inputs are estimated using a model such as OVERSEER, and/or from measurements of point sources, then it would be reasonable (in the absence of any better information) to assume that only 50% of the total input is 'available' to periphyton in the river. The CLUES model includes an assessment of attenuation through its use of the SPARROW model (Alexander et al. 2002).

5.2 Spiralling

In streams, nutrient in the water column is carried downstream while nutrient associated with the streambed is immobile. Exchange between these pools results in alternating downstream transport and periods in the bed. The important components of nutrient spiralling are:

- uptake of dissolved inorganic nutrient (DIN) by plants, bacteria & fungi; or
- adsorption onto particulate inorganic nutrient (PIN);
- conversion of solutes to particulate organic nutrient in biomass (PON); or
- transformation (e.g., denitrification, nitrification, etc.) into other forms of DIN;
- re-release of DIN (e.g., respiration, nitrification etc.);
- senescence of biomass PON to detrital PON;
- leaching of DIN and dissolved organic nutrient DON from fresh detrital PON;
- remineralisation of detrital PON by bacteria & fungi and release of DIN and DON;
- photo-oxidation of DON to DIN;
- settling & burial of PON;
- sloughing, abrasion, scour of PON;
- transport of DIN and PON.

Collectively (1, 2) represent ‘uptake’, (5, 7, 8, 9) ‘re-cycling’ and (11, 12) ‘transport’.

5.3 Nutrient addition and tracer studies

A lot of effort has gone into measuring processes (1) and (2) using ‘addition’ or ‘tracer’ experiments. ‘Addition’ experiments involve adding soluble nutrient (e.g., DRP, NH_4N or NO_3N) plus a conservative solute (e.g., Cl) and measuring the distance downstream at which nutrient concentrations return to ‘background’ concentrations. Addition experiments quantify *nett* uptake (viz., plant uptake, adsorption etc. minus excretion, respiration, mineralisation, desorption etc.). ‘Tracer’ experiments involve adding (small) amounts of ‘labelled’ nutrient (e.g., $^{15}N\text{-}NO_3$) and measuring the distance downstream at which it is no longer detectable. ‘Tracer’ experiments measure *gross* uptake, and can be used to determine where nutrient goes in the food chain.

The majority of addition and tracer experiments are short-term (hours-days), small-scale (100-1000s of metres) and are conducted during stable flows. This limits their ability to quantify long-term processes that affect uptake and recycling including biomass growth & senescence, accumulation of nutrient in the benthos, flushing of particulates during floods, etc.

5.4 Oligotrophic and eutrophic streams

Several studies of streams with low background nutrient concentrations (e.g., forest headwater streams) have found very high rates of nutrient uptake. Cooper and Thomsen (1988) measured high removal rates of DRP and TIN below springs at Purokohukohu – attributable to uptake by aquatic plants and terrestrial grasses. Howard-Williams et al. (1986) measured high NO_3 removal rates below springs in the Whangamata Stream during summer – attributable to uptake by watercress. Newbold et al. (1981) in a much quoted paper measured a spiralling length of 193 m for SRP using ^{32}P in a small woodland stream. They concluded that the availability of regenerated P was not very different from the added SRP. Ensign & Doyle (2006) review 52 published studies of which 69% were in 1st and 2nd order streams. Uptake lengths for NH_4 (median = 86 m) and PO_4 (96 m) were significantly different than NO_3 (236 m).

In contrast, several 'addition' studies in eutrophic streams have found very low net uptake rates. Haggard et al. (2001, 2005) measured net uptake of DIN, NO_3, NH_4 and SRP below a sewage treatment plant. For DRP and NO_3 uptake lengths were S_{net} = 9-31 km and 3-12 km respectively compared with 0.2-0.9 km in nearby streams draining forest and agricultural land. They tabulate long net uptake lengths below several other STPs. Gucker & Pusch (2006) measured net uptake rates in 2 eutrophic streams in Germany using 'addition' methods. Retention mechanisms were similar in pristine and eutrophic streams. However, eutrophic streams exhibited nutrient uptake lengths of several km and nutrient uptake was unable to reduce nutrient exports from the study catchments. Bernot et al. (2006) studied 6 agriculturally influenced streams in Indiana and Michigan using nutrient addition and isotopic tracer studies. Nitrate uptake was saturated in these streams whereas ammonium and phosphorus uptake increased with concentrations, although phosphorus uptake was likely approaching saturation. Biological activity (GPP, algal biomass) was higher than in pristine streams and the authors postulate this influences nutrient retention and transport to downstream ecosystems but do not discuss the processes. Marti et al. (2004) studied uptake below 15 point nutrient sources in Spain. There was no significant net uptake in dilution-corrected concentration of DIN and SRP in 40% and 45% of streams. In the remaining streams, net uptake lengths were 0.14-29 km (DIN) and 0.14-14 km (SRP) - longer (i.e., lower retention) than from non-polluted streams of similar size. Marti states '…this study demonstrates that the efficiency of stream ecosystems to remove nutrients has limitations and supports the hypothesis that large nutrient loadings saturate stream communities...' This is an incorrect use of '…saturation…' as commonly understood and as described by Bernot et al. (2006). In Monod kinetics gross uptake rate increases with concentration and becomes 'saturated' at high concentrations at the *maximum* gross uptake rate. The very low *net* uptake rates of DIN in the Marti study suggest that *gross* uptake rate almost equals mineralisation rate.

It is important to realise that 'addition' experiments measure short-term, *net* removal from the water (uptake minus re-release, remineralisation). 'Tracer' experiments measure short-term, *gross* uptake. Neither measures *long-term, permanent removal* from the ecosystem although this is implied by some authors (e.g., Ensign & Doyle 2006). To quantify permanent removal it is also necessary to quantify *recycling*. This is rarely done.

5.5 Recycling

Recycling involves detritus, bacteria and fungi whose structure and function are not well understood. '…Internal nutrient regeneration and exchange between biofilms and the overlying water have not been satisfactorily addressed...' (Paul et al. 1991).

When nutrients are in short supply, Mulholland et al. (1995) found that nutrients were 'tightly held' within the epilithon and re-cycled very efficiently. McColl (1974) found that in a nutrient poor stream the biota rapidly assimilated DRP whereas in a nutrient rich stream it was not significantly removed. Eutrophic systems are more 'leaky'.

This is consistent with findings outlined in the previous section. In oligotrophic streams, gross uptakes are below 'saturation' (viz., increase with increasing concentration) but release rates are low so that net uptake is high (net uptake = gross uptake – release). In eutrophic streams, gross uptake are high and may be 'saturated' (viz., independent of concentration) but release rates are also high so that net uptake is low. These processes may be understood qualitatively but the question remains whether they can be quantified with sufficient accuracy to assist management.

One major complication is that recycling often occurs at different times and locations from uptake. For example, aquatic plants remove soluble nutrients from the water column during summer but during floods biomass is transported downstream where it may subsequently settle and decompose. Small-scale, short-term tracer studies struggle with such issues. It might be possible to quantify stream attenuation using nutrient spiraling models (e.g., the new Chapra model) but they would require calibration and testing.

5.6 Bioavailability

An important issue when considering re-cycling is the fractions of DON and PON in freshwater systems that are bioavailable (viz., continue to spiral) and the fractions that are refractory (viz., are 'lost' from each spiral).

In many aquatic systems there is a large pool of dissolved organic nitrogen (DON). About 70% of nitrogen transported by rivers globally is dissolved organic nitrogen (DON). The traditional view is that: (1) DON is largely refractory and unimportant to phytoplankton nutrition, and (2) DON fuels bacterial production with relatively long turnover times. There have only been a handful of investigations of the availability of land-derived DON to aquatic biota which report 2-80% bioavailability. Stepanauskas et al. (1999) review the biochemistry of DON and confirm that bacteria are major users of DON. On the other hand Bronk et al. (2006) challenge the traditional view and illustrate that refractory compounds can be a source of bioavailable N to plankton in coastal ocean ecosystems, and that DON fuels a significant amount of autotrophic production.

Wiegner et al. (2006) measured the bioavailability of DON and DOC to bacteria. 23% ± 4% of the DON (2 ± 1 μM) was bioavailable in 7 rivers but none bioavailable in 2 rivers. Of the TDN consumed by bacteria, DON comprised 43% ± 6%, demonstrating that DON is an important nitrogen source for bacteria. In contrast, only 4 ± 1% of DOC (12 ± 3 μM), was bioavailable in the 9 rivers. Stepanauskas et al. (2000) investigated DON bioavailability to bacteria during a spring flood During the flood, DON bioavailability increased from 19–28% at baseflow to 55–45%. Only 5–18% of DON was identified as urea or free and combined amino acids, suggesting that bacteria also utilized other DON compounds. A major portion of the annual export of labile nitrogen occurred during a few weeks of spring flood.

Howard-Williams et al. (1983) studied *in vitro* N losses over 73 days under aerobic conditions from decaying watercress and found that 44% of the original plant N is refractory either as PON (23%) or DON (21%).

Seitzinger et al. (2002) found the proportion of DON utilized by estuarine plankton communities ranged from 0-73%. Urban stormwater was more bioavailable DON (59% ± 11%) than runoff from pastures (30% ± 14%) and forests (23% ± 19%). ~80% of the total dissolved N (TDN) from urban/suburban runoff is bioavailable, whereas a lower proportion (20–60%) is bioavailable from forests and pastures. N budgets for aquatic ecosystems based on only DIN loading underestimate bioavailable N loading, whereas total N or TDN budgets overestimate bioavailable N inputs. Brookshire et al. (2005) found that added labile DON (urea & glutamic acid) was rapidly taken up from the water in a N-limited headwater forest stream, mineralised and nitrified. Vahatalo

& Zepp (2005) showed that photochemical oxidation can increase DON bioavailability.

5.7 Streambed denitrification

Several studies have demonstrated that denitrification occurs in the sediments of many streams. For example Mulholland et al. (2004) measured denitrification rates in a small, forested headwater stream (Walker Branch <0.1 mgN L^{-1}). Denitrification comprised 16 ± 10% of total NO_3N uptake under ambient conditions, and averaged 12 ± 8 μmol m^{-2} h^{-1} (3.9 ± 2.6 mgN m^{-2} d^{-1}) – within the published range for other streams with low NO_3N concentrations. Hill & Sanmugadas (1985) measured rates of nitrate removal in the sediments of Ontario rivers using cores incubated in the laboratory at 21°C for 48 h. Rates of NO_3N loss varied from 37 to 412 mg m^{-2} d^{-1}. Denitrification accounted for 80-100% of the nitrate loss. Rates of nitrate exhibited a highly significant positive correlation with the water-soluble carbon content of the sediments. Hill (1983) studied nitrogen transport during summer low flows in a 20-km reach of the Nottawasaga River which drains an intensively cropped sand plain underlain by a shallow aquifer. About 38% of the daily nitrate input entered the river through ground water contaminated by nitrogen fertilizer (>10 mgNO_3N L^{-1}). The average daily nitrate loss represented ~40% of the ground water input. Laboratory experiments suggested that the bulk of the nitrate loss during river transport was caused by denitrification in bottom sediments. Findlay (2004) states that '…many hyporheic systems are at least periodically anoxic, and this has led to considerable work on denitrification in these sediments (Triska et al. 1989, Duff & Triska 1990). In some cases, denitrification has been shown to be a major component of stream nitrogen budgets despite the apparent oxic nature of the system. Denitrification allows hyporheic sediments to serve a nitrogen removal function (analogous to riparian buffers) that may ameliorate the downstream effects of high N loads to stream systems (Triska et al. 1993)…'.

However, at catchment scale several authors have shown that although denitrification rates in lowland streams are high, stream channel denitrification removes only a small fraction of the total nitrate flux. Fellows et al. (2006) state that '…most studies assume denitrification is a relatively small component of NO_3N uptake in the oxic environment of streams, especially streams with low NO_3N concentrations (Hall and Tank 2003). A recent study by Mulholland and others (2004) using an addition of ^{15}N–NO_3 has confirmed that this is the case…'. Hall & Tank (2003) state that '…given the low NO_3 concentrations, oxic conditions, and little accumulation of organic-rich sediment, we assume that denitrification rates are extremely low and limited by nitrate and carbon availability, as well as anoxic conditions. Lab denitrification assays on stream sediments in anoxic conditions show that when nitrate concentrations fall below ~750 μgN L^{-1}, denitrification rates are nitrate limited...'. Inwood et al. (2005) studied the effects of land use on the relationships among denitrification, NO_3N, DOC, and other environmental parameters in 9 headwater streams (3 forested, 3 agricultural, and 3 urban) in the Kalamazoo, Michigan. Sediment denitrification rates were determined using the chloramphenicol-amended acetylene inhibition technique. Sediment denitrification rates were highest in agricultural streams and lowest in forested streams. Availability of NO_3N was the dominant predictor of sediment denitrification rate, limiting denitrification when NO_3N concentrations were below a calculated threshold of 0.4 $mgNO_3N/L$. The amount of NO_3N removed by sediment denitrification relative to NO_3N load was highest in forested streams (k = 141) and significantly lower in both agricultural (k = 31) and urban (k = 18) streams. (k = estimated proportion of the annual average load removed by sediment denitrification neglecting remineralisation). Sediment denitrification rates were high in headwater streams…but agricultural and urban streams were unable to significantly reduce N export. Williams et al. (2004) calculated N budgets and conducted nutrient uptake experiments to evaluate the fate of N in the aquatic environment of the Ipswich River basin, northeastern Massachusetts. A mass balance indicates that the basin retains ~50% of gross N inputs, mostly in terrestrial components of the landscape, and the loss and retention of total nitrogen (TN) in the aquatic environment was about 9% of stream loading. Retention or loss of NO_3 was observed in a main stem reach bordered by wetland habitat. Nitrate removal in urban headwater tributaries was because of denitrification in wetlands. A mass balance using an entire river network indicates that basin-wide losses due to aquatic denitrification are considerably lower than estimates from several recent studies and range from 4-16% of TDN load.

Streambed denitrification can significantly reduce NO_3N flux in oligotrophic systems (e.g., forest streams) where both denitrification rates (<5 mg m^{-2} d^{-1}) and NO_3N concentrations (<0.1 gN m^{-3}) are low. Rates are higher (50-500 mg m^{-2} d^{-1}) in eutrophic streams (1-10 gN m^{-3}) than in oligotrophic streams. However, this is outweighed by higher NO_3N flux and consequently denitrification does *not* significantly reduce N flux in eutrophic streams. Streambed denitrification can, however, be discounted as a significant attenuation mechanism in *eutrophic* streams.

5.8 Deposition in the hyporheos and floodplain

Trapping in the hyporheos is a potential *temporary* store for POM during low flows. However, this POM is mobilised during scour events and transported to downstream lakes/estuaries. There is very little information about how much POM is denitrified or converted to refractory DON while trapped in the streambed. Overall we *suggest* that trapping in the hyporheos does *not* represent a significant, permanent sink in New Zealand rivers.

Floodplain deposition is an important sink for POM and sediment in long, arid rivers (e.g., Australia). We postulate that it is *not* a major sink in New Zealand. Floodplains are important sites for denitrification and retention of nitrogen. Anoxic sediments of lakes, rivers and wetlands are all active sites of denitrification (Howarth et al. 1996). Fluvial lakes (Hillbricht-Ilkowska, 1999) and wetlands (Saunders & Kalff, 2001) retain more NO_3N than rivers. Work in tropical (Kern et al. 1996; Esteves et al. 2001) and temperate systems (Tockner et al. 1999) suggest that floodplains also retain N. Diversion of a portion of Mississippi River through the Bonnet Carre Spillway resulted in a 28–42% decrease in nitrate concentrations (Lane et al. 2001).

Fluctuating water levels resulting from floods create the aerobic and anaerobic conditions particularly effective for enhancing nitrification and denitrification (Reddy & Patrick, 1975; Groffman & Tiedje, 1988) in alluvial soils (Ponnamperuma, 1972; Keeney, 1973). More denitrification occurs in forested wetlands when the hydrologic regime is maintained than in restored wetlands where the hydrologic regime has not been re-established (Hunter & Faulkner, 2001).

Flooding patterns on most major river floodplains have been altered for flood control and navigation. Dams have lowered the height of flood peaks in many rivers, reducing the frequency and amount of overbank inundation. Flood-control levees restrict the lateral flow of water and accompanying nutrients from rivers to their floodplains. The effect of water level changes on biogeochemistry has been established at local scales (Hill, 2000), but predicting the impact of changing hydrology on biogeochemical fluxes over broad scales remains an enormous challenge (Pinay et al. 2002). Gergel et al. (2005) combined a statistical model of floodplain topography with a model of hydrology and nitrogen biogeochemistry to simulate floods of different magnitude. Model results suggest that dams reduce nitrate processing. Levees increase areal floodplain denitrification rates, but this effect was offset by a reduction in the area inundated. The cumulative N processed by frequent small floods was estimated to be large relative to that processed by large, less frequent floods. Floodplain denitrification may be greatly reduced by flood-control measures.

5.9 Lakes & reservoirs

The Vollenweider model and its applications allow reasonable estimates to be made of nutrient retention. Typically the ‘retention coefficient’ is ~ 10 m yr-1 meaning that lakes with long residence times ‘retain’ a greater fraction of their N & P input than lakes with short residence times. Taupo and Rotorua ‘retain’ ~85% and ~55% of their inputs respectively. Methods are well described by Vant (1987).

Internal loads are an issue for some eutrophic lakes. These are the build up of nutrient that occur in the hypolimnion during summer stratification if/when the bottom waters become seriously depleted in dissolved oxygen. NIWA has done work on internal loads in the past (Vant 1987, Rutherford et al. 1996). Key overseas studies on internal loads include Nurnberg (1984). Professor David Hamilton at the University of Waikato has an active research programme underway on internal loads in the Rotorua lakes (Burger et al. in press). Questions remain about nutrient retention in: small, rapidly flushed, highly eutrophic lakes (e.g., Okaro, Horowhenua); macrophyte infested lakes (e.g., Lower Waikato lakes, Omapere); and in farm dams.

5.10 Nutrient spiralling in Hawke's Bay rivers

During the summer of 2007-2008 it was proposed to investigate nutrient spiralling in the Tukituki River below the Waipukurau and Waipawa sewage treatment plant discharges. The issue was the length of river below the discharges where periphyton biomass is high because of the nutrients from those point sources.

NIWA has been working to develop computer models for nutrient spiralling and periphyton growth in collaboration with Professor Steven Chapra of Tufts University, USA. Chapra developed a simple model (currently unpublished) and it was intended to test and refine this model based on 2007-2008 fieldwork. The aim of that fieldwork is to:

- monitor periphyton biomass accrual visually over a summer low flow period;
- undertake 2-3 longitudinal surveys when biomass is high to measure nutrient concentrations (DIN, TN, DRP, TP) and periphyton biomass at several places below the discharges;
- estimate nutrient inputs from groundwater and tributaries along the study reach; and
- calibrate and refine the Chapra model.

Floods prevented Step 2, and so Step 3 and 4 were not done.

Conclusion: It is desirable to repeat the longitudinal survey work on nutrient spiralling planned for 2007-2008.

Fieldwork has recently been completed in streams at Taupo to measure uptake rates following nutrient additions (Dr Fleur Matheson *pers. comm.*). The nutrient addition protocols are discussed earlier. Results provide valuable insights into uptake mechanisms. It is also desirable to deploy nutrient diffusing substrates as part of the survey to determine which nutrient limits periphyton growth.

Conclusion: It is desirable to undertake nutrient addition experiments in conjunction with the longitudinal survey, and to deploy nutrient diffusing substrates.

5.11 Modelling nutrient spiralling and attenuation

The CLUES model is well suited to predicting the effects of future landuse on annual average TN and TP loads. However, it does not simulate nutrient spiralling which limits its usefulness for predicting periphyton biomass and other aspects of ecosystem health.

A sound conceptual modelling framework exists for nutrient spiralling. Three versions of a spiralling model exist and have been used for research purposes.

The simplest of these models, originally developed by Professor Steven Chapra while on sabbatical at NIWA in 2001, is ideally suited to modelling nutrient spiralling below the oxidation ponds in the Tukituki River. Currently the Chapra model has not been calibrated or tested using reliable field data. However, if the nutrient spiralling study outlined above is carried out, it will provide the necessary data. The Chapra model can be adapted to model multiple and diffuse sources although this will require further computer programming.

The Chapra model will allow changes in DRP and/or DIN concentration along the river channel to be predicted. The model requires knowledge of N and P inputs – supplied by a catchment model such as OVERSEER or CLUES. These changes result from the combination of diffuse inputs, uptake by periphyton and recycling. Currently it is very difficult to interpret monitoring results and apply the nutrient guidelines because there is no convenient way to quantify these changes.

Recommendation: Calibrate and test the Chapra model using results for the proposed nutrient spiralling study.

Recommendation: Re-programme the Chapra model to include multiple sources. Collate available landuse, groundwater gain/loss, groundwater N/P, flow and streamwater N/P, and periphyton biomass data, and use it to calibrate and test the model.

Recommendation: Use the model to quantify the distance downstream from point and diffuse sources where periphyton growth and biomass is elevated as a result of nutrient inflows, and to guide the re-calibration of the CLUES model.

6. Conclusions and recommendations

Council sought a proposal that outlines the work and resourcing requirements for rivers of the Hawke's Bay Region: (1) to undertake a catchment sensitivity analysis, (2) to forecast future landuse impacts on river ecosystems, (3) to provide nutrient limits in river catchments to prevent excessive periphyton growths during low flow periods, and (4) for advice on how to incorporate the modelling of nutrient spiralling to predict nutrient concentrations in rivers. The following work is recommended.

Task	Comment	Resources (estimated person-days)
In each catchment, rank the relative importance of values in Table 1, identify the main pressure(s) and pressure(s).	Council staff in preparation for the next task	5-10 Council staff
In consultation with stakeholders: rank relative importance of values in each catchment, compare different catchments in the region, and adjust rankings. Hence identify key management issues and conflicts in each catchment.	Phase 1 of catchment sensitivity analysis. In consultation with stakeholders.	5-10 Council staff. May involve consultants.
Select parameters for the key value-pressure-response issues identified during Phase 1.	Phase 2 of catchment sensitivity analysis.	5-10 May involve consultants.

Task	Comment	Resources (estimated person-days)
Pressure-response analysis	Phase 3 of catchment sensitivity analysis	Components detailed separately below
Collate available information on landuse, aquifer hydrogeology and groundwater nitrate concentrations.	There is evidence groundwater nitrate concentrations approach or exceed drinking water guidelines in some places. However, the SOE monitoring wells show consistently low nitrate concentrations. It is not clear from the reports sighted where groundwater nitrate concentrations approach or exceed drinking water guidelines.	10-20 Council staff
Undertake a desk study to estimate where nitrate concentrations are likely to approach drinking water limits.		10 May involve consultants.
Review the location of monitoring wells to ensure they adequately monitor high risk areas.		5 Council.
Review and document available information on actual and potential problems arising from agrichemicals in orchards and vineyards, septic tanks, and land disposal sites.		10 Council.

Task	Comment	Resources (estimated person-days)
Collate available information on river flows, groundwater gains/losses, abstractions, landuse, aquifer hydrogeology, and aquifer/river nitrate concentrations.	Surface water quality in many Hawke's Bay rivers is sensitive to groundwater levels and groundwater inflows during summer. Surface-groundwater interactions are not fully understood. This data will help identify catchments sensitive to groundwater inputs of nitrogen, and will support modelling.	Work is already underway to identify gains and losses and to model the Ruataniwha Plains aquifers/rivers. Council.
Undertake a desk study to estimate the importance of groundwater N inputs on river N concentrations, now and in the future.	Groundwater is likely to be a source of nitrate to Hawke's Bay rivers during summer low flows when rivers become N limited.	10-15 Council and consultant.
Better quantify the effects of groundwater and surface water abstractions on river flows.	It is not possible from the reports sighted to quantify the extent to which groundwater and surface water allocation adversely affects surface water quality.	Work is already underway to (1) model groundwater/surface water in the Ruataniwha Plains, (2) meter large abstractions, and (3) better monitor compliance. Council.
Better quantify the extent and effectiveness of riparian fencing.	Council has rather sparse information on the effectiveness of riparian fencing. There is evidence that the majority of large rivers and gulleys are fenced (possibly because the risk of losing stock is high) but that many small streams and ephemeral channels are unfenced.	20-30 Council.

Task	Comment	Resources (estimated person-days)
Continue work to determine limiting nutrients in each catchment.	Council has work underway to determine N and/or P limitation. This work will help quantify catchment sensitivity.	Work is already underway. Council.
Review the provisional guidelines and the nutrient equations in Biggs (2000), using local knowledge and Hawke's Bay monitoring data, to ensure they meet management needs.		15-20 Council and consultant.
Make a preliminary assessment of nutrient sensitivity in catchments where monitoring data exist using the methods of Biggs (2000).	Biggs (2000) gives a method suitable for assessing the combined effects of nutrient concentration and the time between floods on periphyton biomass which can be used to determine the sensitivity of catchments to increasing nutrient concentration.	15-20 Council and consultant.
In a catchment where monitoring data exist adapt the CLUES and Biggs periphyton models to predict nutrient concentrations and periphyton biomass for comparison with observations		15-20 Council and consultant.
If the method above proves satisfactory, use it to forecast the effects of future landuse and to assess catchment nutrient sensitivity.		20-30 Council.

Task	Comment	Resources (estimated person-days)
Undertake the nutrient spiralling study planned for 2007-2008, including nutrient addition and nutrient diffusing substrate experiments.	Consider fortnightly monitoring of periphyton biomass and nutrients in the lead up to the longitudinal survey.	20 Council and consultant
Calibrate and test the Chapra model using results of the spiralling study.		10-20 Consultant.
Re-programme the Chapra model to include multiple sources. Collate available landuse, groundwater gain/loss, groundwater N/P, flow and streamwater N/P, and periphyton biomass data, and use it to calibrate and test the model.		15-20 Consultant.
Use the model to quantify the distance downstream from point and diffuse sources where periphyton growth and biomass is elevated as a result of nutrient inflows, and to guide the re-calibration of the CLUES model.		15-20 Council and consultant.

7. References

Alexander, R.B.; Elliott, A.H.; Shankar, U.; McBride, G.B. (2002). Estimating the sources and transport of nutrients in the Waikato River Basin, New Zealand, *Water Resources Research 38(12)*: 1268, doi:10.1029/2001WR000878, 2002.

ANZECC (2000). Guidelines for Fresh and Marine Water Quality. Australian and New Zealand Environment and Conservation Council. Agriculture and Resource Management Council of Australia and New Zealand.

Ausseil, O. (2008). Water quality in the Tukituki catchment – state, trends and contaminant loads. Aquanet Consulting Limited. Draft – 29th September 2008.

Baalousha, H. (2007a). Review and Assessment of the Groundwater Monitoring Network. EMI 0704. HBRC Plan 3966. June 2007.

Baalousha, H. (2007b). Groundwater Monitoring Strategy 2007-2012. EMI 0727. HBRC Plan 3984. September 2007.

Baalousha, H. (2008a). Water Balance Model for Ruataniwha Basin. EMI 0805. HBRC Plan 4017. March 2008.

Baalousha, H. (2008b). Nitrate in groundwater in Hawke's Bay region. EMI 0727. HBRC Plan 3984. April 2008.

Behrendt, H.; Optitz, D. (2000). Retention of nutrients in river systems: dependence on specific runoff and hydraulic load. *Hydrobiologia 410*: 111-122.

Bernot, M.J.; Tank, J.L.; Royer, T.V.; David, M.B. (2006). Nutrient uptake in streams draining agricultural catchments of the midwestern United States. *Freshwater Biology 51*: 499–509.

Biggs, B.J.F. (2000). New Zealand Periphyton Guideline. Ministry for the Environment, Wellington.

Bronk, D.A.; See, J.H.; Bradley, P.; Killberg, L. (2006). DON as a source of bioavailable nitrogen for phytoplankton. *Biogeosciences Discussion 3:* 1247–1277.

Brooks, T. (2007a). Ruataniwha Basin Water-Resource Data and Allocation. EMI 0731. HBRC Plan Number 3396. December 2007.

Brooks, T. (2007b). Groundwater Quantity Monitoring Strategy 2007-2012. EMI 0728. HBRC Plan Number 3985. August 2007.

Brookshire, E.N.J.; Valett, H.M.; Thomas, S.A.; Webster, J.R. (2005). Coupled cycling of dissolved organic nitrogen and carbon in a forest stream. *Ecology 86(9)*: 2487-2496.

Cooper, A.B.; Thomsen, C.E. (1988). Nitrogen and phosphorus in streamwaters from adjacent pasture, pine and native forest catchments. *New Zealand J. Mar. Freshwater Res*. *22*: 279-291.

Dravid, P.N.; Brown, L.J. (1977). Heretaunga Plains groundwater study. Hawke's Bay Regional Council and the Institute of Geological and Nuclear Sciences. HBRC, Napier.

Duff, J.H.; Triska, F.J. (1990). Denitrification in sediments from the hyporheic zone adjacent to a small forested stream. *Canadian Journal of Fisheries & Aquatic Science 47:* 1140–1147.

Ensign, S.H.; Doyle, M.W. (2006). Nutrient spiraling in streams and river networks. *Journal of Geophysical Research 111*.

Esteves, F.A.; Enrich-Prast, A.; Biesboer, D.D. (2001). Potential denitrification in submerged natural and impacted sediments of Lake Batata, an Amazonian lake. *Hydrobiologia 444*: 111–117.

Fellows, C.S.; Valett, M.H.; Dahm, C.N.; Mulholland, P.J.; Thomas, S.A. (2006). Coupling nutrient uptake and energy flow in headwater streams. *Ecosystems 9*: 788-804.

Findlay, S. (2004). Importance of surface-subsurface exchange in stream ecosystems. *Limnology & Oceanography 40(1):* 159-164.

Gergel, S.E.; Carpenter, S.R.; Stanley, E.H. (2005). Do dams and levees impact nitrogen cycling? Simulating the effects of flood alterations on floodplain denitrification. *Global Change Biology 11*: 1352–1367.

Groffman, P.M.; Tiedje, J.M. (1988). Denitrification Hysteresis During Wetting and Drying Cycles in Soil. *Soil Science Society of America Journal 52*: 1626-1629.

Gucker, B.; Pusch, M.T. (2006). Regulation of nutrient uptake in eutrophic lowland streams. *Limnology & Oceanography 51(3)*: 1443–1453.

Hawke's Bay Regional Council (2006). Water Resource Strategy 2006-2011. EMI 20/1.

Haggard, B.E.; Storm, D.E.; Tejral, D.E.; Popova, Y.A.; Keyworth, V.G.; Stanley, E.H. (2001). Stream nutrient retention in three Northeastern Oklahoma agricultural catchments. *Trans. ASAE. 44(3)*: 597–605.

Haggard, B.E.; Stanley, E.H.; Storm, D.E. (2005). Nutrient retention in a point-source enriched stream. *Journal of the North American Benthological Society 24(1)*: 29-47.

Hall, R.O.; Tank, J.L. (2003). Ecosystem metabolism controls nitrogen uptake in streams in Grand Teton National Park, Wyoming. *Limnology & Oceanography 48(3):* 1120-1128.

Hill, A.R. (1983). Denitrification: its importance in a river draining an intensively cropped watershed. *Agricultural Ecosystems & Environment 10*: 47-62.

Hill, A.R. (1988). Factors influencing nitrate depletion in a rural stream. *Hydrobiologia 160*: 111-112.

Hill, A.R. (2000). Stream Chemistry and Riparian Zones. In: Streams and Ground Waters (J. Jones, and P. Mulholland, eds.) Academic Press.

Hill, A.R.; Sanmugadas, K. (1985). Denitrification rates in relation to stream sediment characteristics. *Water Research 19(12):* 1579-1586.

Hillbricht-Ilkowska, A. (1999). Shallow lakes in lowland river systems: Role in transport and transformations of nutrients and in biological diversity. *Hydrobiologia 408/409*: 349–358.

Howard-Williams, C.W.; Pickmere, S.; Davies, J. (1983). Decay rates and nitrogen dynamics of decomposing watercress (Nasturtium officinale R. Br). *Hydrobiologia 99*: 207-214.

Howard-Williams, C.W.; Pickmere, S.; Davies, J. (1986). Nutrient retention and processing in New Zealand streams: the influence of riparian vegetation. *New Zealand Journal of Agricultural Science 20(2)*: 110-114.

Howarth, R.W.; Marino, R.; Lane, J. (1988). Nitrogen fixation in freshwater, estuarine, and marine ecosystems, 1, Rates and importance, *Limnology & Oceanography 33*: 669– 687.

Howarth, R.W.; Billen, G.; Swaney, D.; Townsend, A.; Jaworski, N.; Lajtha, K.; Downing, J.A.; Elmgren, R.; Caraco, N.; Jordan, T.; Berendse, F.; Freney, J.; Kudeyarov, V.; Murdoch, P.; Zhao-Liang, Z. (1996). Regional nitrogen budgets and riverine N and P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences, *Biogeochemistry 35*: 75–139.

Hunter, R.G.; Faulkner, S.P. (2001). Denitrification Potentials in Restored and Natural Bottomland Hardwood Wetlands. *Soil Science Society of America Journal 65*: 1865-1872.

Inwood, S.E.; Tank, J.L.; Bernot, M.J. (2005). Influence of landuse on sediment denitrification in 9 midwestern headwater streams. *Journal of the North American Benthological Society 24(2)*: 227–245.

Keeney, D.R. (1973). The Nitrogen Cycle in Sediment-Water Systems. *Journal of Environmental Quality 2*: 15-29.

Kern, J.; Darwich, A.; Furch, K.; Junk, W.J. (1996). Seasonal denitrification in flooded and exposed sediments from the Amazon floodplain at Lago Camaleão. *Microbial Ecology 32(1)*: 47-57.

Lane, R.R.; Day, J.W.; Kemp, P.G.; Demcheck, D.K. (2001). The 1994 experimental opening of the Bonnet Carre Spillway to divert Mississippi River water into Lake Pontchartrain, Louisiana. *Ecological Engineering 17(4)*: 411-422.

Marti, E.; Aumatell, J.; Gode, L.; Poch, M.; Sabater, F. (2004). Nutrient Retention Efficiency in Streams Receiving Inputs from Wastewater Treatment Plants. *Journal of Environmental Quality 33*: 285-293.

McColl, R.H.S. (1974). Self-purification of small freshwater streams: phosphate, nitrate and ammonia removal. *New Zealand Journal of Marine & Freshwater Research 8*: 375-388.

Monaghan, R.M.; Wilcock, R.J.; Smith, L.C.; Tikkisetty, B.; Thorrold, B.S.; Costall, D. (2007). Linkages between farm management activities and water quality in an intensively farmed catchment in southern New Zealand. *Agriculture, Ecosystems & Environment 118*: 211-222.

Mulholland, P.J.; Marzolf, E.R.; Hendricks, S.P.; Wilkerson, R.V.; Baybayan, A.K. (1995). Longitudinal patterns of nutrient cycling and periphyton characteristics in streams: a test of upstream-downstream linkages. *Journal of the North American Benthological Society 14(3)*: 357-370.

Mulholland, P.J.; Tank, J.L.; Webster, J.R.; Bowden, W.B.; Dodds, W.K.; Gregory, S.V.; Grimm, N.B.; Hamilton, S.K.; Johnson, S.L.; Martí, E.; McDowell, W.H.; Merriam, J.L.; Meyer, J.L.; Peterson, B.J.; Valett, H.M.; Wollheim, W.M. (2002). Can uptake length in streams be determined by nutrient addition experiments? Results from an interbiome comparison study. *Journal of the North American Benthological Society 21(4)*: 544-560.

Mulholland, P.J.; Valett, H.M.; Webster, J.R.; Thomas, S.A.; Cooper, L.W.; Hamilton, S.K.; Peterson, B.J. (2004). Stream denitrification and total nitrate uptake rates measured using a field 15N tracer addition approach. *Limnology & Oceanography 49(3):* 809-820.

Newbold, J.D.; Elwood, J.W.; O'Neill, R.V.; van Winkle, W. (1981). Measuring nutrient spiralling in streams. *Canadian Journal of Fisheries & Aquatic Sciences 38*: 860-863.

Nurnberg, G.K. (1984). The prediction of internal phosphorus load in lakes with anoxic hypolimnia. *Limnology & Oceanography 29*: 111-124.

Paul, B.J.; Duthie, H.C. (1989). Nutrient cycling in the epilithon of running waters. *Canadian Journal of Botany 67*: 2302-2309.

Peterson, B.J.; Wollheim, W.M.; Mulholland, P.J.; Webster, J.R.; Meyer, J.L.; Tank, J.L.; Martí, E.; Bowden, W.B.; Valett, H.M.; Hershey, A.E.; McDowell, W.H.; Dodds, W.K.; Hamilton, S.K.; Gregory, S.V.; Morrall, D.D. (2001). Control of Nitrogen Export from Watersheds by Headwater Streams. *Science 292*: 86-90.

Ponnamperuma, F.N. (1972). The chemistry of submerged soils. *Advances in Agronomy 24*:29-96.

Pinay, G.; Clement, J.C.; Naiman, R.J. (2002). Basic Principles and Ecological Consequences of Changing Water Regimes on Nitrogen Cycling in Fluvial Systems. *Environmental Management 30(4)*: 481-491.

Regional Resource Management Plan (2006). Hawke’s Bay Regional Council.

Reddy, K.R.; Kadlec, R.H.; Flaig, E.; Gale, P.M. (1999). Phosphorus retention in streams and wetlands: a review. *Critical Reviews in Environmental Science and Technology 29(1)*: 83-146.

Reddy, K.R.; Patrick, W.H. (1975). Effect of alternate aerobic and anaerobic conditions on redox potential, organic matter decomposition and nitrogen loss in a flooded soil. *Soil Biology & Biochemistry 7(2)*: 87-94.

Rutherford, J.C.; Dumnov, S.M.; Ross, A.H. (1996). Predictions of phosphorus in Lake Rotorua following load reductions. *New Zealand Journal of Marine & Freshwater Research 30(3)*: 383-396.

Rutherford, J.C.; Scarsbrook, M.R.; Broekhuizen, N. (1999). Grazer control of stream algae: modelling temperature and flood effects. *Journal of Environmental Engineering 126(4)*: 331-339.

Sarazin, U.; Zimmerman, E. (2003). Riparian Habitat Classification of the Ruataniwha Plains. Hawke’s Bay Regional Council Internal Report EMI 0303, HBRC Plan 3217.

Saunders, D.L.; Kalff, J. (2001). Nitrogen retention in wetlands, lakes and rivers. *Hydrobiologia 443*: 205-212.

Seitzinger, S.P. (1988). Denitrification in freshwater and coastal marine ecosystems: Ecological and geochemical significance. *Limnology and Oceanography 33*: 702-724.

Seitzinger, S.P.; Harrison, J.A.; Bohlke, J.K.; Bouman, A.F.; Lowrance, R.; Petersen, B.; Tobias, C.; van Drecht, G. (2006). Denitrification across landscapes and waterscapes: a synthesis. *Ecological Applications 16*: 2064-2090.

Seitzinger, S.P.; Sanders, R.W.; Styles, R. (2002). Bioavailability of DON from natural and anthropogenic sources to estuarine plankton. *Limnology and Oceanography 47(2)*: 353–366.

Seitzinger, S.P.; Renee, V.S.; Boyer, E.W.; Alexander, R.B.; Billen, G.; Howarth, R.W.; Mayer, B.; van Breemen, N. (2002). Nitrogen retention in rivers: Model development and application to watersheds in the eastern United States, *Biogeochemistry 57/58*: 199– 237.

Stansfield, B. (2006). Surface Water Quality Monitoring Strategy. EMI 0621, HBRC Plan 3894, June 2006.

Stansfield, B. (2008). Upper Mohaka River Catchment. Draft HBRC report.

Stepanauskas, R.; Edling, H.; Tranvik, L.J. (1999). Differential Dissolved Organic Nitrogen Availability and Bacterial Aminopeptidase Activity in Limnic and Marine Waters. *Microbial Ecology 38*: 264–272.

Stepanauskas, R.; Laudon, H.; Jørgensen, N.O.G. (2000). High DON bioavailability in boreal streams during a spring flood. *Limnology and Oceanography 45(6)*: 1298–1307.

Tockner, K.; Pennetzdorfer, D.; Reiner, N.; Schiemer, F.; Ward, J.V. (1999). Hydrological connectivity, and the exchange of organic matter and nutrients in a dynamic river–floodplain system (Danube, Austria). *Freshwater Biology 41(3):* 521-535.

Triska, F.J.; Duff, J.H.; Avanzino, R.J. (1993). Patterns of hydrological exchange and nutrient transformation in the hyporheic zone of a gravel-bottom stream: examining terrestrial-aquatic linkages. *Freshwater Biology 29*: 259-274.

Triska, F.J.; Kennedy, V.C.; Avanzino, R.J.; Zell'weger, G.W.; Bencala, K.E. (1989). Retention and transport of nutrients in a third-order stream in northwestern California: hyporheic processes. *Ecology 70*: 1893-1905.

Vahatalo, A.V.; Zepp, R.G. (2005). Photochemical mineralisation of dissolved organic nitrogen to ammonium in the Baltic Sea. *Environmental Science & Technology 39*: 6985-6992.

Vant, W.N. (1987). Lake Managers' Handbook. NWASCA, Wellington.

Vant, W.N. & Smith, P. (2004). Nutrient concentrations and water ages in 11 streams flowing into Lake Taupo (revised). Environment Waikato Technical Report 202/18R, revised April 2004.

Wiegner, T.N.; Seitzinger, S.P.; Glibert, P.M.; Bronk, D.A. (2006). Riverine DOM bioavailability. *Aquatic Microbial Ecology 43:* 277–287.

Wilcock, R.J.; Monaghan, R.M.; Quinn, J.M.; Campbell, A.M.; Thorrold, B.S.; Duncan, M.J.; McGowan, A.W.; Betteridge, K. (2006). Land-use impacts and water quality targets in the intensive dairying catchment of the Toenepi Stream, New Zealand. *New Zealand Journal of Marine & Freshwater Research 40*: 123-140.

Wilcock, R.J.; Biggs, B.J.F.; Death, R.G.; Hickey, C.W.; Larned, S.T.; Quinn, J.M. (2007). Limiting nutrients for controlling undesirable periphyton growth. NIWA Client Report HAM 2007-006, 38p.

Williams, M.; Hopkinson, C.; Rastetter, E.; Vallino, J. (2004). N budgets and aquatic uptake in the Ipswich River basin, northeastern Massachusetts. *Water Resources Research 40*: W11201, doi:10.1029/2004WR003172.

8. Acknowledgements

I am grateful to staff at Hawke's Bay Regional Council for making available reports and responding to emails and questions, notably Graham Sevicke-Jones, John Phillips, Brett Stansfield and Husam Baalousha. This study is funded by the Foundation for Research, Science and Technology through the Envirolink small grants programme. Colleagues Graham McBride and John Quinn provided useful comments on the report.