IN THE MATTER OF	The Resource Management Act 1991
AND	
IN THE MATTER OF	Lake Rotorua Nutrient Management – **PROPOSED PLAN CHANGE 10** to the Bay of Plenty Regional Water and Land Plan

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STATEMENT OF EVIDENCE OF PROFESSOR GRAEME JOHN DOOLE

ON BEHALF OF THE BAY OF PLENTY REGIONAL COUNCIL

Evidence topic: Economic impacts of Plan Change 10 at the catchment level

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Contents:

Qualifications and experience

1. My full name is **Graeme John DOOLE**. I am currently the Professor of Environmental Economics at the University of Waikato. I have held this position for eighteen months. Prior to this appointment, I held various positions at the University of Western Australia (Perth, Australia) where I worked for fourteen years.

2. The focus of my research has concerned the economic assessment of policies and practices to improve environmental outcomes, such as water quality, through changing behaviour on private land, mainly in rural areas. This has principally involved the development and application of mathematical models of catchments, farm systems, contaminant loss, and populations.

3. I have the following qualifications: Bachelor of Applied Science (Natural Resource Management) from Massey University, Masters of Applied Economics (Natural Resource and Environmental Economics) from Massey University, and Doctor of Philosophy (Agricultural and Resource Economics) from the University of Western Australia. I am currently a member of the Australian Agricultural and Resource Economics Society (AARES) and the New Zealand Agricultural and Resource Economics Society (NZARES).

4. I was contracted by DairyNZ in 2015 to perform part of the economic assessment described in the report entitled, “On-farm effects of diverse allocation mechanisms in the Lake Rotorua catchment”. This report was written by Oliver Parsons (DairyNZ), myself, and Alvaro Romera (DairyNZ). This report is referred to throughout this document as Parsons et al. (2015). Overall, this study explores how diverse allocation mechanisms could be expected to influence farmer decision making—encompassing both land-use decisions and on-farm mitigation behaviour—and how trade in nutrient entitlements could be utilised to overcome distortions arising from their initial allocation.

5. I have been called as an expert witness by BOPRC, and DairyNZ are aware of this invitation. I also continue to work with both BOPRC and DairyNZ on a number of projects that do not involve Lake Rotorua. However, as an expert witness, I do not advocate for any party within this process and consider my role as one of providing impartial assistance to the Hearings Panel on matters within my area of expertise.

6. I have read the Expert Witness Code of Conduct set out in the Environment Court's Practice Note 2014 and I agree to comply with it. I confirm that the issues addressed in this statement of evidence are within my area of expertise, except where I state that I am relying on the specified evidence of another person. I have not omitted to consider material facts known to me that might alter or detract from my expressed opinion.

Scope of Evidence and Summary

7. My evidence concerns the modelling method used to integrate the farm-level data provided by Lee Matheson (Perrin Ag., Rotorua) with information regarding different allocation systems for nutrient entitlements. The chief focus of this project was to assess the economic effects of different allocation and trading scenarios for nitrogen-leaching entitlements at the **catchment** level. It is distinct from the work performed by Market Economics (ME). ME obtained the estimates of the financial impacts that were generated by Parsons et al. (2015). These were then used to identify the aggregate impacts of the policy at the **district, regional, and national levels**.

8. A wide variety of economic-modelling techniques can be applied to assess the potential financial and distributional impacts of environmental policies. I have performed and published research that has applied many of these methods to problems in water management, for more than a decade. A key lesson from these applications has been that no single method is superior to the rest, each technique has its own strengths and weaknesses. The most-suitable method for a problem therefore depends on the context, with resource availability and the nature of the questions that need to be answered being key drivers of the decision around what constitutes the most-suitable approach.

9. The economic model applied in the Parsons et al. (2015) report is consistent with an established economic-modelling formalism known as the Land Allocation and Management (LAM) catchment-modelling framework (Doole, 2012, 2015). The broad applicability and relevance of this model is demonstrated in its utilisation across a number of contexts in which agricultural and urban emitters are responsible for water-quality decline, both nationally (Doole, 2010, 2012, 2013; Howard et al., 2013; Holland and Doole, 2014; Doole, 2016a–b, Doole et al., 2016a–d) and internationally (Beverly et al., 2013, 2016; Doole et al., 2013). The model structure and its application has been reviewed in multiple contexts, demonstrated in its extensive publication in

international, peer-reviewed journals (e.g. Doole, 2010, 2012; Doole et al., 2013; Holland and Doole, 2014; Beverly et al., 2016).

10. My evidence addresses five general questions:

(a) Is the LAM assessment methodology applied in Parsons et al. (2015) appropriate for this form of analysis?

(b) What was the justification underlying the way that different scenarios were described in the model applied in Parsons et al. (2015)?

(c) Is the calibration method utilised in Parsons et al. (2015) justified for this form of analysis?

(d) What was the justification underlying the way that time was represented in the model applied in Parsons et al. (2015)?

(e) What was the justification underlying the way that variation in input data was represented in the model applied in Parsons et al. (2015)?

11. I also provide commentary on the modelling results.

12. The report Parsons et al (2015) was a joint project and the authors had responsibility for their sections. My evidence does not focus on the sections written by others. This includes sections “3.2 Input data” regarding the methodology used to generate input data; “Appendix 1–data” that contains data generated by Lee Matheson; “4.4 Implications of scenarios for land prices”; and “4.5 Implications of scenarios for debt servicing and equity”.

13. My evidence explains the research project, process, and results, and provides information/analysis on the structure of the model used. My evidence focuses specifically on the model structure given (a) it is within my area of expertise, and (b) that was the area of my expert input into the report. The chief assumptions that have been made in this element of the report are common, in line with good practice for model development and application, and are well justified, both theoretically and empirically.

Background to the report

14. I was approached by DairyNZ in mid-2013 to take part in a project in which they wished to assess the on-farm economic impacts of different nitrogen-allocation

mechanisms in the Lake Rotorua catchment. This project was undertaken jointly with the Bay of Plenty Regional Council (BOPRC) over 2014 and 2015, but I was contracted by DairyNZ solely to perform this work. The project was managed by Oliver Parsons (DairyNZ) and Sandra Barns (BOPRC). The project contained a strong participatory element, through working extensively with stakeholders throughout the assessment process to ensure that the application of the model was consistent with their understanding and objectives.

15. The stakeholder group did not participate in the selection of the modelling framework, to my knowledge, given that this decision was influenced by many specialised technical and project-management factors. Mainly, these involved the assessment of the advantages and disadvantages of different modelling formalisms in terms of the (a) quality and quantity of available input data; (b) the resolution of output that was required by BOPRC to fulfil legislative requirements; (c) the availability of funding; (d) the availability of the requisite skills to develop and apply the model(s); (e) the capacity to link models, this was important given the flow of modelled information between Lee Matheson, Parsons et al. (2015), and Market Economics; (f) how much time was available to develop the model; and (g) how much time was available to update the model to reflect different scenarios during interaction with the stakeholder group. Involving the stakeholder group in the decisions regarding the type of model utilised in the Parsons et al. (2015) report may have increased their ownership of the modelling study and its outputs. However, this would also have posed a risk in that objectives (a)–(g) listed above may not have been considered to the requisite degree. –

16. The objective of the report was to evaluate the broad effects of several proposed nitrogen (N) allocation systems for producers in the Lake Rotorua catchment. These allocation systems were:

- Sector Averaging – Each sector is allocated a given amount of nitrogen. This corresponds to allocations to dairy of 45.2 kg N/ha/yr, to drystock of 20.78 kg N/ha/yr, and to forestry of 3 kg N/yr.

- Sector averaging with consideration of biophysical characteristics – The dairy and drystock sectors experience a uniform proportional reduction to achieve the sector averages identified in S1.

- Single range – A single percentage clawback applied to all commercial grazing properties, with final allocations in the range of 16–52kgN/ha.

- Natural capital allocation – Allocation is based on the inherent productivity of each spatial zone.

- Equal allocation – All land uses are allocated the same amount, with this amount varying between land parcels that have a slope less/greater than 26 degrees.

- Range OA – Final drystock allocation range 15.5–31 kg N/ha/yr, with an average of 20.4 kg N/ha/yr. Final dairy allocation within a range of 43.5–58 kg N/ha/yr, with an average of 46.6 kg N/ha/yr.

- Range 1 – Final drystock allocation range 15.5–43.5 kg N/ha/yr, with an average of 20.4 kg N/ha/yr. Final dairy allocation within a range of 43.5-58 kg N/ha/yr, with an average of 46.6 kg N/ha/yr.

- Range 2 – Final drystock allocation range 15.5–31.5 kg N/ha/yr, with an average of 20.4 kg N/ha/yr. Final dairy allocation within a range of 40–53 kg N/ha/yr, with an average of 46.6 kg N/ha/yr

The results of the Parsons et al. (2015) report chiefly provided information about the distribution of private benefits and costs—measured primarily in terms of farm profit (as represented by Earnings Before Interest and Tax)—when different amounts of nitrogen were allocated, and could subsequently be traded, across diverse spatial zones and land uses. A focus of the study was the Range 2 scenario that had been selected by the stakeholder group as their preferred management option.

17. The project brief was developed collaboratively between BOPRC, DairyNZ, and Beef + Lamb New Zealand—with key input from the Rotorua Stakeholder Advisory Group (StAG) members—during the latter part of 2014. Draft modelling results were presented to the StAG committee in March, April, May, June, and July 2015, and feedback was incorporated in the report up to and including August 2015[1]. The report provided direct information for the StAG and BOPRC, as well as providing data to support wider district-level economic modelling undertaken by Market Economics. These discrete pieces of work support the section 32 report associated with new nitrogen rules for the Lake Rotorua catchment.

[1] A copy of the project brief can be provided at the hearing if required.

18. A broad number of people contributed to the report. Key contributors were Sandra Barns, Alastair MacCormick, and Michele Hosking (Bay of Plenty Regional Council); Carly Sluys, Andrew Burtt and Ben O'Brien (Beef + Lamb New Zealand); Sharon Morrell (DairyNZ); Lee Matheson (PerrinAg); Phil Journeaux and Mark MacIntosh (AgFirst); and Simon Park (Landconnect). The Rotorua STAG played a central role in the project, especially regarding: the design of the allocation scenarios put forward for assessment, the determination of what results were important to present and understand, and contributing to the iterative updating of the model.

Results of the report

19. Results show a modest overall impact on total catchment profit. However, the impacts on profit are distributed unevenly across sectors, land-uses, and biophysical zones. Different allocation regimes create further variation in this distribution of cost. In general, drystock farm profits benefit from the ability to sell nitrogen (to businesses with higher profit per kilogram of nitrogen and the incentives fund). Dairy farm profits fall due to the need to acquire nitrogen in order to continue operating. Under allocations with more redistribution (such as equal-allocation and natural-capital systems), dairy farm profits fall further, but drystock profits are not correspondingly improved. This is due to a large number of allowances being transferred from dairy farms to foresters under these regimes, rather than other pastoral uses. Allocation regimes that require a large amount of redistribution also result in increases in the nitrogen price, due to greater dependence on trading and increased market demand. Pastoral farming profit within the catchment is reduced by around 5% in both land-use scenarios—that is, when a limit of 5000 ha of land-use change is set and when no upper bound is placed on land-use change—when a 50% trading friction is introduced to the model. Trading rigidities in the market have significant implications for the price of N, increasing the price for perpetual allowances from around \$118 and \$60 kg N^{-1} in the 5,000 ha limited and unlimited land-use change scenarios, to around \$444 kg N^{-1} (up to \$551 kg N^{-1} for natural-capital allocation). This higher price reflects an increased scarcity of nutrient entitlements in the market and is consistent with economic theory that stipulates that a reduced supply will increase prices in a market, under typical conditions. These results highlight that practices to pragmatically address rigidities in the market for nitrogen-leaching entitlements in the Lake Rotorua catchment will have direct benefits for increasing the amount of nitrogen that could be

purchased by the incentive fund, while also reducing on-farm costs through promoting more cost-effective nutrient mitigation.

20. In summary, the main conclusions from the report of Parsons et al. (2015) outline that limits to nitrogen loss in the Lake Rotorua catchment are likely to:

(a) Promote greater areas of plantation forest to be grown in the catchment, with the predicted level of increase being around 61% (from 7,095 ha to 11,403 ha) when the level of optimal land-use change was constrained at 5000 ha and around 85% (from 7,095 ha to 13,098 ha) when no upper bound was placed on land-use change. The value of afforestation is that it provides a cost-effective means to reduce nitrogen losses to water, although returns from forest are obviously different from pastoral agriculture in that they do not occur annually but at the end of an extended period (around 28–30 years).

(b) Reduce incentives for retaining extensive dairy farming in the catchment. The area allocated to dairy farming decreases by around 45% (from 5,024 ha to 2,754 ha) when the level of optimal land-use change was constrained at 5000 ha and around 39% (from 5,024 ha to 3,046 ha) when no upper bound was placed on land-use change. Dairy farming leads to high nitrogen losses to the environment, relative to the other land uses in the catchment. A reduction in the amount of nitrogen that is allowed to be leached in the catchment therefore means that it becomes costlier to maintain this form of agriculture. Some types of dairy farm earn comparatively low profits per hectare, relative to the amount of nitrogen that they leach, which makes land-use change and the sale of nitrogen a more-profitable option than mitigation for these particular farms.

(c) The decrease in dairy-farming area in the catchment leads to a decline in cow numbers, nitrogen-fertiliser application, supplement use, and farm labour in the dairy sector. There was a 43%, 61%, 35%, and 19% decrease in cow numbers, nitrogen-fertiliser application, supplement use, and farm labour when the level of optimal land-use change was constrained at 5000 ha. There was a 37%, 56%, 27%, and 16% decrease in cow numbers, nitrogen-fertiliser application, supplement use, and farm labour when no upper bound was placed on the level of optimal land-use change.

(d) Reduce the need for extensive dairy-support activity. Two types of dairy-support activity are represented in the model: dairy support, and sheep and dairy support (a mixed activity). Dairy support stays the same across each scenario; this was fixed in the model because most dairy farmers will prefer to support their own stock where they have the resources available already, none of the principal model runs suggested that decreasing this amount of land was necessary to achieve the mitigation goals that were set, and the mixed activity (sheep and dairy support) is a more-flexible form of land use historically, whose incidence fluctuates according to demand from the dairy sector. In comparison, limits to nitrogen loss lead to a reduction in the area utilised for the mixed activity involving both sheep and dairy-support. The scale of decrease was around 37% when the level of optimal land-use change was constrained at 5000 ha, and around 67% when no upper bound was placed on land-use change. A reduction in the amount of nitrogen that is allowed to be leached in the catchment means that it becomes costlier to maintain this land use. Additionally, there is less demand for this type of agriculture because of the reduced incentives for intensive dairy farming that occur under the evaluated scenarios.

(e) Promote the need for dairy farms to purchase nitrogen-leaching entitlements to remain economically viable. Pastoral dairy farming leads to high losses of nitrogen because 75–90% of nitrogen ingested by dairy cattle is excreted into concentrated urine spots; these are associated with high losses of nitrogen to groundwater when water supply exceeds plant demand, commonly in autumn and winter. However, dairy farming is also a profitable form of pastoral agriculture. The economic value of dairy farms in this catchment means that it is not excluded entirely when a policy scenario is implemented; rather, dairy-farm systems with higher nutrient efficiency continue to be used or are adopted, though they may need to buy leaching entitlements for their higher environmental footprint to be sustained.

(f) Incentivise dairy-support and sheep-and-beef enterprises to de-intensify at reasonable cost and sell nitrogen-leaching entitlements to dairy farms and the incentives fund. There is a reduction in the area of land used for both sheep and dairy-support activities, with the scale of decrease around 37% when the level of optimal land-use change was constrained at 5000 ha and around 67% when no upper bound was placed on land-use change. Additionally, there is a reduction in the area utilised for sheep-and-beef production, with the decrease being around 14% when the level of optimal land-use change was constrained at 5000 ha and around 30% when no upper bound was placed on land-use change. These sectors earn lower profits and have a lower environmental footprint, relative to intensive dairy farming, under typical management. Thus, they play a key role in the trading scenarios in which they sell their entitlements for leaching to dairy farms where more profit can be gained per unit of nitrogen leached.

(g) The total farm-profitability impacts differ for individual farmers, depending on their allocation relative to their current discharges, the type of land use they operate, and the characteristics of that representative farm, such as soil type, rainfall, and slope. For all farmers, per-hectare income is likely to increase. The biggest increases are for forestry and dairy support, mostly because of pastoral farmers shifting to these sectors and selling existing saleable assets, such as Fonterra shares and livestock. The income from the sale of these assets is annualised in the model. The increased profitability per hectare for dairy and sheep-and-beef farmers is relatively small.

(h) There may be frictions in the market for nutrient entitlements. These are associated with producers receiving a given amount of nutrient entitlements, but wanting to retain an amount above their economically-efficient level to partially insure themselves against risk associated with volatility in product markets, climate, environmental targets, or the political environment. This is explored in the model through limiting the amount of trade that occurs to 50% of the optimal level. Profit within the catchment is reduced when frictions are present within the market for nitrogen permits. Indeed, catchment profit decreases by around 5% when land-use change is capped at 5,000 ha and when it has no upper bound. This highlights that hoarding within a permit market reduces efficiency at the catchment level, and the cost of this inefficiency falls on the producer population.

(i) A key source of this inefficiency is barriers to land-use change. For example, the presence of frictions means that the area in dairy and sheep-and-beef farming is higher than optimal, while the area of sheep and dairy support and forestry is lower than optimal when an upper bound of 5,000 ha land-use change is investigated in the model. In contrast, the presence of frictions means that the amount of dairy and forested land is lower than optimal in the unlimited land-use change scenario due to an inability to acquire allowances, while land allocation to sheep-and-beef and sheep and dairy support activity is too high.

(j) These rigidities have a significant impact on the price of nitrogen within the market for entitlements. The price for nitrogen increases from around $118 and $60 kg N^{-1} in the 5,000 ha and unlimited land-use change scenarios, respectively, to around $444 kg N^{-1} within both 50% trading friction scenarios. This higher-price outcome is intuitive because it reflects an increased scarcity of nutrient entitlements in the market arising from a reduced supply. It has a significant practical implication in that rigidities in the entitlements market will compromise the capacity of the incentives fund to purchase nitrogen. Moreover, these rigidities promote on-farm costs associated with nutrient mitigation, as farmers have less flexibility with respect to how they mitigate contaminant loss.

(k) Different allocation regimes have a significant impact on the distribution of cost across spatial locations and sectors. In general, dairy farms must purchase nitrogen to continue operating. In contrast, sheep-and-beef farms and plantation forestry benefit from the ability to sell nitrogen to businesses that earn more profit per kilogram of nitrogen (e.g. dairy farms) and the incentives fund. For the Range 2 allocation scenario and a maximum level of land-use change of 5,000 ha, the cost of buying entitlements for dairy is around 4% of profit, while profit increases by 2% and 20% for sheep-and-beef and forestry land, respectively, with the sale of entitlements. For the Range 2 allocation scenario and unconstrained land-use change, the cost of buying entitlements for dairy is around 3% of profit, while profit increases by 4% and 10% for sheep-and-beef and forestry land, respectively, with the sale of entitlements.

(l) Under allocation systems that involve more re-distribution (such as natural-capital and equal-allocation systems), dairy farm profits fall further. This is due to a significant reallocation of nutrient entitlements from dairy farms to forested land, rather than other pastoral uses, under these regimes. Indeed, dairy farms purchase entitlements for 39 and 43 kg nitrogen ha^{-1}, on average, within the natural-capital and equal-allocation scenarios, respectively (for an upper bound for land-use change of 5,000 ha). This compares with forested land that sells entitlements for 17 and 18 kg nitrogen ha^{-1}, on average, within the natural-capital and equal allocation scenarios, respectively (for an upper bound for land-use change of 5,000 ha).

(m) Allocation systems that involve more re-distribution (such as natural-capital and equal-allocation systems) hence have major ramifications for sector profit. For the natural-capital scenario and a maximum level of land-use change of 5,000 ha, profit for dairy falls by 11% but increases by 5% and 19% for sheep-and-beef and forestry land, respectively. For the natural-capital scenario and unconstrained land-use change, profit for dairy falls by 7% but increases by 16% and 10% for sheep-and-beef and forestry land, respectively. The key driver for these disparate results are the purchase and sale of nutrient-leaching entitlements by these sectors. The trading of nutrient entitlements plays a particularly important role in these allocation systems, as producers who receive an allocation that is substantially different from what they leach currently can buy/sell the right to emit, to minimise disruption to their current management strategy. Nevertheless, though trading allows such activity, they can have a marked impact on sector profit.

21. These conclusions remain correct for the set of input data that was used to generate them.

22. My most-significant concern relating to the current relevance of the conclusions pertains to (a) how the nitrogen-loss estimates provided by Lee Matheson (Perrin Ag, Rotorua) have changed with subsequent updates of the Overseer model used to generate them, and (b) how the profit-loss estimates provided by Lee Matheson (Perrin Ag, Rotorua) have changed based on volatility in the economic environment in which pastoral farming takes place. The former is a problem that is common in the economic assessment of water-quality policies in New Zealand (e.g. Howard et al., 2013). An example of the latter is the high variation observed in the milk price over the last few years[2]. I believe that the impact of variation in Overseer estimates is likely to be of minor concern in practice, given that mitigation efficacy appears to not have greatly changed between Overseer versions 6.1.3 and 6.2.3 (Lee Matheson, pers. comm., 11/1/2017). Moreover, BOPRC data indicates that the relative scale of required reductions remains very much the same across each sector, when computed in these disparate versions (Alistair McDonald, pers. comm., 11/1/2017). Likewise, the

[2] A price of $6/kg of milk solids was used for milk revenue. This was the nominal average Fonterra price ($6.07/kgMS) for the period 2006/07 through 2014/15. The real (CPI adjusted) milk price since 1975 is just under $6/kgMS (Parsons et al, Appendix 5).

use of long-term average prices mitigates any concern associated with price volatility. Only a structural shift in the prices in one industry versus another would cause the price set that has been utilised to no longer be relevant. This has not occurred in my opinion since the initial data was generated by Lee Matheson.

Basis of my opinion

Is the assessment methodology applied in Parsons et al. (2015) appropriate for this form of analysis?

23. The model applied in the Parsons et al. (2015) report (hereafter referred to as, “the model”) describes the catchment of Lake Rotorua as a landscape divided into many different partitions. Each partition is described in terms of its rainfall, its soil type, a typical farm type (i.e. land use) found on that combination of rainfall and soil type (based on what is observed in reality), and the size of the area that fits this description (in hectares). Using this classification method, it was observed that the same land use is typically found over diverse rainfall and soil-type bands. Similarly, it was observed that different land uses are generally found within the same rainfall and soil-type bands. These factors are generally evident in Table 1 of Parsons et al. (2015).

24. In the context of the Parsons et al. (2015) report, the rainfall, soil type, and size of these partitions were generated by BOPRC. The farm types present in each partition were determined by Lee Matheson (PerrinAg), Oliver Parsons (DairyNZ), and Alvaro Romera (DairyNZ). (These farm types are commonly known as “representative farm types” or “average farms” in applied work.) Within each partition, the performance of an average farm is represented on a per-hectare basis and not in terms of the total farm area. This allows farms to be defined across partitions of different sizes. For example, see Table 1 in the report of Parsons et al. (2015), where partitions range in size from 1 ha to 1,852 ha. In the 1,852 ha partition, total profit for this parcel of land would be generated through multiplying profit per ha with parcel size (1,852 ha), for instance.

25. The goal of the partitioning was to provide an adequate description of the average or expected cost of reducing a given level of nitrogen loss from a farm population in a given spatial zone. Thus, in contrast to the analysis of individual farms where exceptional results may occur (e.g. Doole and Romera, 2014), the focus is placed on the identification of general relationships that are expected to hold true on average across farms with a high degree of commonality (Roberts et al., 2012; Doole et al.,

2013). That is, rather than trying to predict how individual farmers would adapt (Doole, 2010), the intention is to identify how an average producer in a given rainfall, soil, and land-use partition would respond to the need to reduce their environmental footprint.

26. The precision with which individual farms are represented within a catchment model—such as that applied by Parsons et al. (2015)—varies along a continuum, which includes:

(a) The depiction of individual farms (e.g. Doole, 2010, 2012).

(b) The definition of representative farms where average enterprises are defined to represent a broad part of the catchment (e.g. Parsons et al., 2015).

(c) A single farm is used to depict an entire catchment (e.g. Doole and Pannell, 2011).

27. Most catchment models utilise the approach identified in clause 26 (b), as employed in the Daigneault et al. (2012) study and the Parsons et al. (2015) report. This is because it allows for:

(a) Key differences arising from spatial and sectoral diversity.

(b) Depiction of variation in the cost of mitigation across diverse farming systems.

(c) The representation of adequate levels of heterogeneity between economic agents. This is needed because a realistic depiction of trading in nutrient entitlements requires a rich description of the differences that exist between enterprises in terms of their relative cost of mitigating nitrogen (Hanley et al., 2007).

(d) The number of representative farms to be altered, depending on the availability of information and resources. The lower resource demands of approach #2 are more consistent with the quantity and quality of data that is typically available to perform assessments of policy instruments for water-quality improvement in New Zealand.

(e) Less reliance on data of a poor quality. Models that purport to provide a rich description of the behaviour of diverse, individual producers require rich data for their calibration (Windrum et al., 2007; Doole, 2010; Doole et al., 2011). This data is often missing in practical studies of water-quality improvement.

28. An alternative approach is one that involves the representation of individual producers and explicit behavioural rules guiding their decisions and interaction with one another. These frameworks come under the general title of “agent-based models” and have been extensively applied throughout New Zealand, such as in Canterbury (Daigneault and Morgan, 2012) and the Waikato (Doole, 2010; Doole et al., 2011) regions. Such frameworks provide a very rich description of individual agents, with diversity represented in risk aversion, personal networks, management objectives, and production-system intensity, among other factors. However, they also require a rich set of appropriate empirical data that can be used to generate a realistic description of the personal characteristics of diverse individual producers within a given catchment and/or allow a validation of model predictions outside of the baseline situation. These are common constraints accruing to the application of agent-based models (Windrum et al., 2007), but are particularly relevant in New Zealand because of privacy restrictions, integral data being held across diverse organisations, and the significant cost and time required to collect suitable data from producer populations to inform model development.

29. For each representative-farm type, Lee Matheson determined the profit and nitrogen-loss levels for a broad range of different management strategies for each representative farm type. One strategy was their current management plan (commonly known as the “baseline strategy”), while the others represented how these farms were likely to reduce nitrogen losses if required to (commonly known as “mitigation strategies” or “abatement strategies”). (Detailed data for each mitigation strategy are detailed in the appendices of the Parsons et al. (2015) report.) Each of these mitigation strategies contained one or more abatement activities that an average farm in each partition would be expected to perform, if required to cost effectively reduce their nitrogen loss.

30. The relationships between profit and nitrogen loss were determined through the application of modelling protocols, which specified the sequence of mitigation use for each farm type. These were developed in workshops involving BOPRC staff, DairyNZ, Beef + Lamb New Zealand, scientists, local extension agents, and agricultural consultants. The goal of this approach was to provide structure to what has historically been a process that has been unstructured and highly influenced by personal bias. Through developing and applying the protocols in a documented and open way, this helped to ensure that the farmer responses that were simulated were based on sound judgement from a number of domain experts. It also allowed them to

be peer reviewed across the diverse modelling team and stakeholders involved in the economic assessment, many of whom possessed different levels of knowledge regarding farm systems and options to reduce nitrogen loss from them. Overall, the selected approach was applied to increase the amount of rigour, transparency, and repeatability associated with this exercise, relative to previous applications where such protocols were not utilised.

31. SCION determined the profitability of plantation forestry within each partition under a variety of scenarios. The tool they used to identify these levels of profit was the Forest Investment Finder. Carbon liability was incorporated in these computations.

32. Lee Matheson also determined the benefits and costs associated with land-use change. Land-use transition could impose a net benefit or a net cost on producers, depending on the change that occurred. An example of a net benefit is that conversion from dairy to dairy support yielded a net annual gain of around $1,212 per ha because of the sale of livestock and Fonterra shares. An example of a net cost is that conversion from forestry to dairy yielded a net annual loss of around $3,517 per ha because of the need to build fences, construct buildings, purchase livestock, buy Fonterra shares, and so on.

33. The model applied by Parsons et al. (2015) is an optimisation model–that is, it employs an iterative search process to identify how different mitigations could be implemented to maximise the profit earned across a catchment within the defined set of circumstances (Gill et al., 1981). This optimisation model uses a method known as non-linear programming to identify the best solution from among the set of potential solutions that is available (Bazaraa et al., 2006). A computational economic model is essentially a collection of equations and decision variables that seek to describe some part of a complex reality. A decision variable in an optimisation model is a term that is identified by a model during its solution, while equations are pre-defined and outline the logic that the decision variables must obey.

34. A key type of equation utilised in the form of mathematical modelling that is utilised in this study (non-linear programming) is a constraint. These constraints can define key relationships (i.e. a relational constraint), or can be used to restrict the level of certain decision variables (i.e. a limit). A key relationship used in the economic model applied by Parsons et al. (2015) is the limit constraint placed on the level on land-use change that can occur (a maximum bound of 5000 ha is placed on land-use change, in one scenario).

35. The total amount of nitrogen lost from a parcel of land in the catchment was computed through multiplying the size of that partition (measured in ha) by the level of nitrogen loss for that partition (measured in kg of nitrogen loss per ha). Adding the total nitrogen loads for each partition defined in the catchment allowed the total nitrogen loss across all of the farm land represented in a single run of the model to be computed. Likewise, the total amount of profit earned from a partition was computed through multiplying the size of that partition (measured in ha) by the level of profit earned for that partition (measured in dollars per ha). Adding the total profit levels for each partition allowed the total profit across all of the farm land represented in a single run of the model to be computed. In the baseline strategy, the level of profit earned and nitrogen lost over the catchment represented the best estimate of the current state that could be reached within the resources available for the project.

36. In other runs of the model aside from the baseline strategy, alternative scenarios were represented in which diverse farm types were allocated different levels of nitrogen-leaching entitlements. Eight different allocation systems were evaluated (see above in clause 16), subject to varying levels of land-use change and efficiency in the market for nutrient entitlements. The context of each scenario was defined in the model through a set of mathematical constraints. Predictions of how producers could be expected to behave were then identified using the automated, iterative-search procedure applied within non-linear programming. The objective used to compare the suitability of alternative solutions was total profit earned across the catchment. Accordingly, once the model was manipulated such that a certain allocation system was represented, the iterative search procedure focused on identifying what management practices (both land use and land management) and trading patterns should be employed within each partition to maximise total profit across the catchment.

37. The use of a standard objective (profit maximisation) allowed different sets of management practices to be compared at the catchment level. This measure of economic performance is given primacy here for two reasons. First, profitability is a key driver of the uptake of technologies by farmers, even if it is not the only one (Rogers, 2003; Pannell et al., 2006, 2014; Bewsell and Brown, 2011). Second, this approach is common in economic-optimisation models, especially those applied for water-quality improvement. This is evident both in New Zealand (Daigneault et al., 2012; Doole, 2015) and international (Merel and Howitt, 2014) applications.

38. An alternative approach to optimisation is known as “simulation” and involves the search of superior solutions through the use of “trial-and-error” by a modeller to assess different scenarios within a model. This is very difficult to perform in a model of the size and complexity used in the Parsons et al. (2015) report, due to the large number of alternative situations that must be explored if superior solutions are to be identified (Doole and Pannell, 2008). This catchment contains significant heterogeneity in terms of the different farm systems and the physical environments where these diverse activities take place. Accordingly, the relative value of different outcomes identified in a “trial-and-error” strategy would be strongly biased by the assumptions made to simplify the assessment of different alternatives. This is particularly problematic in this work because trading patterns depend on the relative value of leaching entitlements across a wide variety of land parcels depicted in the model, but is hard to compute in an efficient manner without the use of optimisation methods. Indeed, this complexity is so significant that it is difficult to provide a reasonable description of trading for nutrient-leaching entitlements without the use of optimisation methods. Simplification of the model to allow its analysis with simulation would, by definition, complicate the portrayal of rich dynamics in the leaching-entitlement market. This would lead to significant underestimation of the value of a trading mechanism, given that broad heterogeneity in the cost of mitigation across a population is required to drive the operation of an efficient trading mechanism for nutrient entitlements (Hanley et al., 2007; Doole, 2010).

What was the justification underlying the way that different scenarios were described in the model applied in Parsons et al. (2015)?

39. The model incorporates the trading of nitrogen-leaching rights both among farmers, and with an incentives fund that buys out nitrogen. Nitrogen prices are generated inside the catchment model, based on the balance of the benefits and costs of trading among each of the individual land parcels represented in the model. In the absence of frictions in the market, the equilibrium or steady-state outcome of the model represents the situation whereby there are no further gains to trade; catchment profit cannot be increased through allowing any further trade in nitrogen-leaching entitlements to occur.

40. Eight different allocation options (see clause 16 for a list) were evaluated for two levels of market efficiency for nutrient trading. These scenarios were based on the needs articulated by the stakeholder group (StAG) for the Lake Rotorua catchment. The impacts of changes in the efficiency of the trading market were explored through

allowing free trade in entitlements and then only 50% of that optimal level; in the latter case, the remainder of the entitlements being retained by producers following allocation. It is pragmatic to test the implications of producers retaining allocated entitlements to pollute, even though it is not done that often (see Howard et al. (2013) for a rare exception). Farmers are aware that retaining leaching rights provides some protection against risks posed by further changes in environmental, market, and political conditions. Furthermore, when faced with the possibility of adopting a given practice, risk aversion is a key factor that prevents most people using that option (Rogers, 2003). This is especially an issue with agricultural populations, given that in applied research the majority of farmers are generally found to be risk averse (Pannell et al., 2006, 2014). In alignment with these factors, trading within New Zealand water-quantity markets occurs well beneath efficient levels, given a lack of information, small markets, risk aversion, infrastructure constraints, and high uncertainty (Robb et al., 2001). These observations are consistent with recent experimental evidence that highlights that human subjects in a simulated market for pollution entitlements broadly failed to achieve the predicted equilibrium outcomes, especially due to people's aversion to potentially losing entitlements that they already have in hand (Marsh et al., 2014).

41. Eight different nitrogen-allocation options were evaluated in Parsons et al. (2015), for two levels of land-use change: an upper bound of 5,000 ha placed on land-use change, and no upper bound placed on land-use change. Land-use change is a temporal process influenced by many factors, such as input and output price trends, innovation, expectations, productivity, management preferences, farmer skills, and environmental policy. The extent of land-use change that occurred in the model was influenced through the consideration of transition costs (see above) and—in the case of the 5,000 ha upper bound—the addition of a constraint limiting the amount of land-use change that may be observed in a given situation. This upper bound placed on land-use change was introduced to reflect the fact that it is unlikely that the full, unconstrained amount of land-use change predicted by the optimisation model would occur. This approach was motivated by several benefits:

(a) The level of land-use change that was permitted to occur in the constrained scenario was determined by the stakeholder group, based on their expectations of what could be expected to reasonably occur. This is a pragmatic way of constraining land-use change in line with the expectations of stakeholders, who possess an in-depth implicit knowledge of the catchment that is difficult to gather from other sources.

(b) Such a constraint is important because land-use change—for example, from pasture to forestry—is influenced by key factors that are difficult to describe in the model. Examples are the: lack of a significant annual return in alternative land uses (e.g. forestry), complexity of learning new management strategies for other land uses, negative impacts on land prices, climate volatility, market variability, risk aversion, lifestyle impacts, current skills and knowledge, compatibility with other parts of the farming system where transition is partial, and compatibility with existing beliefs and values (Pannell et al., 2006, 2014; Rogers, 2006).

(c) Using this approach, the impacts of altering the maximum level of land-use change that was permissible in any run of the model could be assessed efficiently using sensitivity analysis. Sensitivity analysis provides established techniques to assess how the output of an optimisation model changes as key input parameters are varied (Pannell, 1997). Indeed, a broad range of other levels were evaluated, though the data was not represented in the Parsons et al. (2015) report.

(d) This approach is straightforward to program within the appropriate software and is easy for other practitioners to understand. This is important because it improves the accessibility of the framework to other modellers, stakeholders, and the public.

(e) This approach is much easier to formulate and less prone to error than forcing a model to provide a current estimate of existing land use through the inclusion of arbitrary calibration functions (Doole and Marsh, 2014). (See below for further discussion.)

(f) This approach avoids the bias associated with the use of such calibration functions when the model is used to investigate various scenarios that are different from the established baseline situation (Doole and Marsh, 2014).

(g) This approach does not require a specification of historical land-use patterns, as needed with the approach of Chen and Onal (2012). This is significant given the dearth of necessary data pertaining to the Lake Rotorua catchment.

(h) This approach does not require a specification of historical land-use patterns, as needed within the approach of Chen and Onal (2012). This is significant because the introduction of nitrogen-leaching limits will introduce a new evolutionary force that will likely affect the trajectory of land-use change in this catchment, thus forcing it to diverge from historical patterns.

Is the calibration method utilised in Parsons et al. (2015) justified for this form of analysis?

42. Model output highlights that different nitrogen-allocation mechanisms could have a significant impact on land-use patterns in the catchment. This is to be expected, given the diversity in biophysical and economic conditions present in farming systems across the catchment and the significant change to current practice that is proposed by Plan Change 10. Calibration functions help to restrain the total level of land-use change that is observed in such models, through seeking to describe many implicit factors that impact land-use change but are difficult to describe within a given model (see clause 41 (b) for examples) (Daigneault et al., 2012). Hence, they have been the subject of much debate in the development and application of optimisation models to evaluate water-quality policies (Heckelei, 2002; Heckelei and Wolff, 2003; Doole and Marsh, 2014).

43. The calibration of an economic-optimisation model involves the estimation and incorporation of non-linear functions that direct a model to return an observed baseline land-use allocation, by manipulating the relative profitability of each individual land-use (Daigneault et al., 2012). This method is known generally as positive mathematical programming (Howitt, 1995).

44. Calibration of the model using the particular method of positive mathematical programming was difficult to justify in this model for several reasons:

(a) There is an infinite number of sets of calibration function parameters that can generate the observed baseline land-use (Heckelei and Wolff, 2003).

(b) Calibration does not use any information on how the relative value of land-uses changes as land-use allocation moves away from the observed baseline (Heckelei and Britz, 2000, 2005; Heckelei, 2002). Each one of the infinite sets of calibration-function parameters—from which one is arbitrarily selected to calibrate the model to baseline data—yields a different policy response from the calibrated model. Thus, the way in which the model performs outside of the calibrated scenario is completely unpredictable (Heckelei, 2002) and prone to bias (de Frahan et al., 2007).

(c) The theoretical basis of PMP is, “weak or at least not apparent” (Heckelei and Wolff, 2003, p. 28).

(d) The relative value of alternative land-use activities is altered through the introduction of calibration functions.

(e) Functional forms used for calibration functions in PMP implementations are generally ad-hoc and difficult to justify (Heckelei and Wolff, 2003; Heckelei et al., 2012).

(f) There is a lack of data pertaining to the historical incidence of different land uses in the study region.

(g) The form and structure of the calibration functions is difficult to explain to stakeholders that played a key role in interpreting and operationalising model output.

45. These complexities are avoided in this application through the division of each sub-catchment of Lake Rotorua into separate parcels—each containing individual representative-farming systems—and the definition of appropriate transition benefits and costs that represent how a change in land use impacts economic outcomes. Additionally, the assessment also involved some rationalisation of land-use change based on stakeholder guidance, using an accessible and replicable procedure (see clause 41 above). In this way, the issue of land-use change was considered a key part of scenario generation. The alternative land-uses that were possible for each parcel and the specification of bounds dictating the permissible extent of these changes were indeed important components of an iterative discussion regarding land-use change dynamics. In this way, the specification of land-use change provided an opportunity for collaborative learning focused on the implications of changes in this important aspect of the overall mitigation approach. This approach did not explicitly consider historical trends regarding land-use change—as utilised by Chen and Onal (2012). Nevertheless, given that the introduction of nitrogen-leaching limits will alter the trajectory of land-use change relative to historical trends—in a way that is difficult to predict *a priori* (Lamblin et al., 2000)—the detrimental impact of this omission is greatly reduced.

What was the justification underlying the way that time was represented in the model applied in Parsons et al. (2015)?

46. The model identifies the equilibrium or steady-state outcome associated with a given model scenario. This approach is adopted for a range of reasons:

(a) When the model was developed, there was little empirical work available that characterised how the farming population in the Lake Rotorua catchment would be expected to adapt to the different allocation scenarios. This deficiency exists throughout New Zealand, with little information focused on the economic and environmental impacts of different transition pathways for the land uses and municipal centres found within catchments.

(b) Variation over time in key drivers of management behaviour (e.g. output price, input price, productivity, climate, innovation) is high and difficult to predict.

(c) When the model was developed, there was little empirical work available that characterised how the farming population in the Lake Rotorua catchment would be expected to adapt to variation in these key drivers of management behaviour.

(d) Dynamic models are difficult to develop and utilise (Doole and Pannell, 2008). This assessment is no different, in that the catchment contains such significant diversity that trying to predict how it would change annually for a series of years is exceedingly difficult.

(e) Output from dynamic models is heavily biased by the beginning and end points defined during model formulation (Klein-Haneveld and Stegeman, 2005). This made it particularly difficult to study transition, because at the inception of the project there was little understanding of what the end point for the catchment under each allocation was likely to be, let alone what the best way to get there was.

(f) This approach is standard in economic-optimisation models, especially those applied for the assessment of policies targeted at water-quality improvement (Daigneault et al., 2012; Doole, 2015). This reflects the complexities listed above in elements (a)–(e).

What was the justification underlying the way that variation in input data was represented in the model applied in Parsons et al. (2015)?

47. The model contains economic and biophysical data that is consistent with long-term averages and identifies how, given this data, that land use and land management could be altered to reach an environmental goal while maximising economic returns. It assumes that agents have perfect knowledge of the implications of the alternative mitigation strategies available; as such, input data is represented by a single, point estimate and not a statistical range or distribution. Such a model is commonly known as a "deterministic model".

48. Various techniques have been developed to accommodate uncertainty present in input data. Stochastic programming permits the explicit inclusion of probability distributions that describe the variation believed to exist for key data (Kingwell, 1996), whereas chance-constrained programming allows some constraints in the model to be infeasible for a proportion of the time (Kall and Mayer, 2005). In comparison, sensitivity analysis allows the impact of uncertain data on model output to be tested (Pannell, 1997), while robust optimisation involves defining uncertain input data in terms of their lower and upper bounds, if appropriate distributional information or point estimates are unavailable (Doole and Kingwell, 2010).

49. A deterministic framework is used in the model applied in the Parsons et al. (2015) report for a variety of reasons, although the alternatives listed in clause 48 exist. These reasons are:

(a) This approach is standard in economic-optimisation models, especially those applied for water-quality improvement. This is evident both in New Zealand (Daigneault et al., 2012; Doole, 2015) and international (Merel and Howitt, 2014) applications.

(b) Deterministic models are easier to develop and apply than stochastic-programming and robust-optimisation models, particularly within the resources available within the project described by the Parsons et al. (2015) report. This is evident in that neither alternative approach has yet to be used in New Zealand to assess water-quality policy within a collaborative decision-making process.

(c) Detailed information describing the variation evident in data can be difficult and/or expensive to obtain, more so than averages.

(d) Stochastic models can provide misleading insight into the relative value of different opportunities, because of the simplistic way that uncertainty is typically considered in large models (Pannell et al., 2000). Indeed, parameter variation is most meaningful to include when studies focus on the best response of managers to this uncertainty as it unfolds across time (Kingwell, 1996; Pannell et al., 2000). However, as discussed above in clause 46, there are numerous complexities that hamper the development of models that assess the performance of environmental policies across time.

Reports/Update

50. I am aware of two peer reviews of the Parsons et al. (2015) report that have been made available to me by the Bay Of Plenty Regional Council. These are:

Journeaux, P. (2015a), *Review of economic modelling papers (6 August 2015)*, AgFirst, Hamilton.

Journeaux, P. (2015b) *Review of economic modelling papers (14 August 2015)*, AgFirst, Hamilton.

Conclusion

51. A broad number of assessment methods can be applied to assess the economic impacts of alternative policies to achieve water-quality improvement, each having their own advantages and disadvantages. In my opinion, the economic model applied in the Parsons et al. (2015) report to assess the biophysical and economic impacts of different nitrogen-allocation mechanisms in the Lake Rotorua catchment is consistent with good practice. The chief assumptions that have been made are common practice and are well justified, both theoretically and empirically.

Appendices

52. Parsons, O.J., Doole, G.J., and Romera, A.J. (2015), *On-farm effects of diverse allocation mechanisms in the Lake Rotorua catchment*, BOPRC/DairyNZ, Hamilton.

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Date: 11 January 2017

References

Bazaraa, M.S., Sherali, H.D., and Shetty, C.M. (2006), *Nonlinear programming; theory and application*, Wiley, New York.

Beverly, C., Roberts, A.M., Stott, K., Vigiak, O.V., and Doole, G.J. (2013), 'Optimising economic and environmental outcomes: water quality challenges in Corner Inlet, Victoria', *Proceedings of the 20th International Congress on Modelling and Simulation*, Adelaide, Australia, 1–6 December 2013, pp. 2117–2123.

Beverly, C., Roberts, A., Doole, G.J., and Bennett, F.R. (2016), 'Assessing the net benefits of achieving water quality targets using a bio-economic model', *Environmental Modelling and Software* 85, pp. 229–245.

Bewsell, D., and Brown, M. (2011), 'Involvement: a novel approach for understanding responses to nutrient budgeting', *New Zealand Journal of Agricultural Research* 54, pp. 45–52.

Chen, X., and Onal, H. (2012), 'Modelling agricultural supply response using mathematical programming and crop mixes', *American Journal of Agricultural Economics* 94, pp. 674–686.

Daigneault, A., and Morgan, F. (2012), 'Estimating impacts of climate change policy on land use: an agent-based approach', *American Agricultural and Applied Economics Association 2012 Annual Meeting*, Seattle.

Daigneault, A., McDonald, H., Elliott, S., Howard-Williams, C., Greenhalgh, S., Guysev, M., Kerr, S., Lennox, J., Lilburne, L., Morgenstern, U., Norton, N., Quinn, J., Rutherford, K., Snelder, T., and Wilcock, B. (2012), *Evaluation of the impact of different policy options for managing to water quality limits*, MPI Technical Paper No 2012/46, Wellington.

Doole, G.J., and Pannell, D.J. (2008), 'On the economic analysis of crop rotations', in Berklian, Y.U. (ed.), *Crop rotation: economics, impact, and management*, Nova Science Publishers, Hauppauge, pp. 71–106.

Doole, G.J. (2010), 'Indirect instruments for nonpoint pollution control with multiple, dissimilar agents', *Journal of Agricultural Economics* 61, pp. 680–696.

Doole, G.J., and Kingwell, R.S. (2010), 'Robust mathematical programming for natural resource modelling under parametric uncertainty', *Natural Resource Modelling* 23, pp. 283–302.

Doole, G.J., Ramilan, T., and Pannell, D.J. (2011), 'Framework for evaluating management interventions for water quality improvement across multiple agents', *Environmental Modelling and Software* 26, pp. 860–872.

Doole, G.J. (2012), 'Cost-effective policies for improving water quality by reducing nitrate emissions from diverse dairy farms: an abatement-cost perspective', *Agricultural Water Management* 104, pp. 10–20.

Doole, G.J. (2013), *Evaluation of policies for water quality improvement in the Upper Waikato catchment*, University of Waikato client report, Hamilton. URL: http://www.mfe.govt.nz/issues/water/freshwater/supportingpapers/evaluation-water-quality-upper-waikato.pdf.

Doole, G.J., Vigiak, O.V., Pannell, D.J., and Roberts, A.M. (2013), 'Cost-effective strategies to mitigate multiple pollutants in an agricultural catchment in North-Central Victoria, Australia', *Australian Journal of Agricultural and Resource Economics* 57, pp. 441–460.

Doole, G.J., and Marsh, D.K. (2014a), 'Methodological limitations in the evaluation of policies to reduce nitrate leaching from New Zealand agriculture', *Australian Journal of Agricultural and Resource Economics* 58, pp. 78–89.

Doole, G.J., and Marsh, D.K. (2014b), 'Use of positive mathematical programming invalidates the application of the *NZFARM* model', *Australian Journal of Agricultural and Resource Economics* 58, pp. 291–294.

Doole, G.J. (2015), 'A flexible framework for environmental policy assessment at the catchment level', *Computers and Electronics in Agriculture* 114, pp. 221–230.

Doole, G.J. (2016a), *Model structure for the economic model utilised within the Healthy Rivers Wai Ora process*, Healthy Rivers Wai Ora report HR/TLG/2015-2016/4.8, Hamilton.

Doole, G.J. (2016b), *Evaluation of scenarios for water-quality improvement in the Waikato and Waipa River catchments: Business-as-usual assessment*, Healthy Rivers Wai Ora report HR/TLG/, Hamilton.

Doole, G.J., Elliott, A.H., and McDonald, G. (2016a), *General principles underlying the development of the Healthy Rivers Wai Ora economic model*, Healthy Rivers Wai Ora report HR/TLG/2015-2016/4.7, Hamilton.

Doole, G.J., Elliott, A.H., and McDonald, G. (2016b), *Economic evaluation of scenarios for water-quality improvement in the Waikato and Waipa River catchments: Assessment of first set of scenarios*, Healthy Rivers Wai Ora report HR/TLG/2015-2016/4.1, Hamilton.

Doole, G.J., Elliott, A.H., and McDonald, G. (2016c*), Evaluation of scenarios for water-quality improvement in the Waikato and Waipa River catchments: Assessment of second set of scenarios*, Healthy Rivers Wai Ora report HR/TLG/2015-2016/4.2, Hamilton.

Doole, G.J., Quinn, J.M., Wilcock, B.J., and Hudson, N. (2016d), *Simulation of the proposed policy mix for the Healthy Rivers Wai Ora process*, Healthy Rivers Wai Ora report HR/TLG/2016-2017/4.5, Hamilton.

Gill, P.E., Murray, W., and Saunders, M.A. (2005), 'SNOPT: an SQP algorithm for large-scale constrained optimization', *SIAM Review* 47, pp. 99–131.

Hanley, N., Shogren, J., and White, B. (2007), *Environmental economics: in theory and practice*, Palgrave Macmillan, Basingstoke.

Harris, S., and Snelder, T. (2014), 'The role of scenarios in limit setting', *Presentation at the Water Directorate Water Economics Workshop (27 August 2014)*, Nelson.

Heckelei, T., Britz, W., and Zhang, Y. (2012), 'Positive mathematical programming approaches—recent developments in literature and applied modelling', *Bio-based and Applied Economics* 1, pp. 109–124.

Holland, L.M., and Doole, G.J. (2014), 'Implications of fairness for the design of nitrate leaching policy for heterogeneous New Zealand dairy farms', *Agricultural Water Management* 132, pp. 79–88.

Howard, S., Romera, A.J., and Doole, G.J. (2013), *Selwyn-Waihora nitrogen loss reductions and allocation systems*, DairyNZ, Hamilton.

Kall, P., and Mayer, J. (2005), *Stochastic linear programming: models, theory and computation*, Springer Verlag, Heidelberg.

Kingwell, R.S. (1996), ‘Programming models of farm supply response: the impact of specification errors’, *Agricultural Systems* 50, pp. 307–24.

Klein-Haneveld, W.K., and Stegeman, A.W. (2005), ‘Crop succession requirements in agricultural production planning’, *European Journal of Operational Research* 166, pp. 406–429.

Lamblin, E.F., Rounsevell, M.D.A., and Geist, H.J. (2000), ‘Are agricultural land-use models able to predict land-use intensity?’, *Agriculture, Ecosystems and Environment* 82, pp. 321–331.

Marsh, D., Tucker, S., and Doole, G.J. (2014), ‘An experimental approach to assessment of trading and allocation mechanisms for nutrient trading’, Paper presented at the *58th Australian Agricultural and Resource Economics Annual Conference*, February 4–7 2014, Port Macquarie, Australia. Available at: http://purl.umn.edu/167195.

Pannell, D.J. (1997), ‘Sensitivity analysis of normative economic models: Theoretical framework and practical strategies’, *Agricultural Economics* 16, pp. 139–152.

Pannell, D.J., Malcolm, B., and Kingwell, R.S. (2000), ‘Are we risking too much? Perspectives on risk in farm modelling’, *Agricultural Economics* 23, pp. 69–78.

Pannell, D.J., Marshall, G.R., Barr, N., Curtis, A., Vanclay, F. and Wilkinson, R. (2006), ‘Understanding and promoting adoption of conservation practices by rural landholders’, *Australian Journal of Experimental Agriculture* 46, pp. 1407–1424.

Pannell, D.J., Llewellyn, R.S., and Corbeels, M. (2014), ‘The farm-level economics of conservation agriculture for resource-poor farmers’, *Agriculture, Ecosystems, and Environment* 187, pp. 52–64.

Robb, C., Morgan, M., and Harris, S. (2001), *Attitudes and barriers to water transfer*, Lincoln Environmental Report 4464/1, Lincoln.

Rogers, E.M. (2003). *Diffusion of innovations*, Free Press, New York.

Windrum, P., Fagiolo, G., and Alessio, M. (2007), ‘Empirical validation of agent-based models: alternatives and prospects', *Journal of Artificial Societies and Social Simulation* 10, URL: http://jasss.soc.surrey.ac.uk/10/2/8.html.