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General principles underlying the development of the Healthy Rivers Wai Ora (HRWO) economic model

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General principles underlying the development of the Healthy Rivers Wai Ora (HRWO) economic model

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1. Introduction

The Healthy Rivers Plan for Change: Waiora He Rautaki Whakapaipai (HRWO) Project (www.waikatoregion.govt.nz/healthyrivers) will establish targets and limits for nutrients (N and P), sediment, and *E. coli* in water bodies across the Waikato/Waipa catchment. Different targets and limits regarding the level of these contaminants in waterways within this catchment will have diverse impacts on economic outcomes observed throughout the greater Waikato region. Accordingly, a central contribution of the Technical Leaders Group (TLG) to the HRWO project is the development and utilisation of an economic model that will integrate diverse information such that the size and distribution of abatement costs—across farm, catchment, regional, and national levels—associated with alternative limits and targets can be predicted. The primary objective of this document is to outline the reasons why certain key decisions have been made during the design and development of this HRWO economic model.

The report is structured as follows. Section 2 describes the certain approach used to identify how land use, agricultural intensity, and point sources have to change their management in order to attain alternative limits and targets. This is the overall framework that integrates a broad amount of data from many information-gathering streams within the HRWO process. The way that this data is collected and why is explained subsequently, in Sections 3–4. A certain fraction of the output of the catchment-level model is subsequently entered into a regional-level model, to highlight the regional- and national-level impacts associated with alternative limits. This regional framework is described in Section 5. Section 6 concludes.

2. Catchment-level modelling approach

The economic-modelling approach utilised in the catchment-level analysis involves the employment of optimisation methods to identify how land use and land management within the catchment would have to change if a given set of limits was to be achieved at least cost. The forms of land use and land management described in the model are meant to be meaningful reflections of what is currently observed throughout the region. However, because of data limitations, this provides only a coarse description of the current state.

Land use and land management are described using diverse data that has been drawn from extensive data-collection processes; therefore, the insights provided by the model are solely conditional on the information contained therein. The methods utilised to collect this data are described in detail below, in a bid to improve clarity around the procedures that have been implemented. The development of such an economic model is an intensive undertaking subject to real resource constraints, especially those pertaining to time, funding, and relevant expertise. Accordingly, it is critical to recognise that while best efforts have been made to collect the most meaningful information for a model of this kind, there remain critical uncertainties given our limited capacity to address the complexity of the problem in its complete entirety.

The method of optimisation utilised within the catchment-level model is known as non-linear programming (Bazaraa et al., 2006), which involves the use of a mathematical algorithm to identify how decision variables within a problem must change to achieve a certain goal at least cost. Any change in these decision variables from their current state is bound by a set of constraints, which define the feasibility of alternative choices available within the model. Within the catchment model employed here, the key decision variables represent land use and the degree of mitigation activity performed within each land use. Key constraints placed on the level of decision variables essentially define the logic that characterises the optimisation model. For example, these can involve placing an upper bound on the amount of dairy farming on peat soils that is carried out in a region, thus reflecting real biophysical constraints, and not allowing the use of a stand-off pad to reduce nitrogen leaching on horticultural land. The most important constraints within the HRWO model define that certain pollutant concentration targets are set at alternative locations within the catchment, and the model is tasked with determining how land use and land management will have to change within different parts of the catchment to meet these at least cost, given the set of input data being employed.

This optimisation model defined at the catchment-level studies the characteristics of alternative equilibria (also known as steady-state or stationary-state outcomes); thus, the dynamics of transition across time, and other processes that are inherently temporal, are studied at a very-coarse level only. This approach is consistent with those methods used broadly for the economic assessment of alternative environmental goals (Baumol and Oates, 1988; Doole and Pannell, 2008a; Hanley et al., 2007), both in New Zealand (Daigneault et al., 2012; Doole and Pannell, 2012) and overseas (Kampas and White, 2004; Doole et al., 2013). Dynamic mathematical programming models have been applied in the past to agricultural problems (e.g. Heady and Candler, 1958; Candler, 1960), but have received little application overall, especially for catchment-level analysis of this kind. The key reasons that dynamic models are frequently not employed are:

1. It is difficult to identify how farmers will adapt across time to limits placed on contaminant loss from farmland. Indeed, there remains no work in New Zealand addressing how a population of producers can be expected to perform such adaptation. The development of such information, especially based on empirical data, therefore remains a critical research gap.
2. No work has been done within New Zealand that estimates how the relationship between abatement and farm profit changes across time given variability in output and input prices, productivity, and innovation. Critical barriers to this work are cost, the specific nature of any output, and the extended time it would take to generate.
3. It is generally difficult to work with dynamic models because such models quickly become too large to make sure they are not free of errors and solve in an appropriate amount of time. This is especially true if it is assumed that prices will vary across time in such a way that farmers cannot adequately predict them *a priori*. For this reason, it is notable that most optimisation models describing dynamic, stochastic systems (i.e. those that represent variability in model parameters) are defined across two periods only (Prekopa, 2010).
4. It is challenging to identify how long transition paths (i.e. the time period studied within such a model) should be to incorporate within such a model. This is a crucial problem, because the model becomes increasingly difficult to solve as the length of the transition period that is represented within the model increases.
5. Economic theory broadly postulates that people act according to a rule of perfect rationality: the alternative outcomes that they face are known with certainty and they select that action with the highest pay-off. In reality, humans have limited cognitive abilities (Kahnemann, 2003) and uncertainty complicates a decision maker's assessment of relative options (Pindyck, 2007). This has a significant impact on the distribution of payoffs, both spatially and temporally, during a transition period.

However, there are no broadly-accepted ways of representing this process of sub-optimisation within a catchment-level model, even if such data existed (see Point #1). Accordingly, the neoclassical assumption of perfect rationality is operationalised in the primary economic models of land-use optimisation employed throughout New Zealand (Anastasiadis et al., 2013), including that model utilised in the HRWO process.

6. Output from dynamic models is heavily biased by the initial and terminal conditions defined during model formulation. Accordingly, a number of key conceptual studies (Throsby, 1967; Klein-Haneveld and Stegeman, 2005) have highlighted that equilibrium-optimisation models, rather than dynamic-optimisation models, focused on the identification of stationary solutions are the most-useful type.
7. It is difficult to formulate consistent insight when analysing the output of dynamic models, given the large volume of results that is generated. Indeed, alternative equilibrium solutions are often superior because there are a limited number of them. The value of these isolated solutions is particularly promoted by the factor outlined in Point #6.

For these reasons, the economic model developed for the HRWO process does not represent dynamic decision-making explicitly. Instead, it studies the impacts of alternative limits through representing how they affect equilibrium decision-making under different situations.

The structure of the economic model is based on the Land Allocation and Management catchment framework (Doole, 2015). Its flexibility is evident in that it has now been broadly applied across both Australia and New Zealand, within contexts encompassing a diverse range of agricultural activities, biophysical resources, and hydrological situations (Doole and Paragahawewa, 2011; Doole, 2012; Roberts et al., 2012; Howard et al., 2013; Beverly et al., 2013; Doole, 2013; Doole et al., 2013; Holland and Doole, 2014). Multiple benefits are associated with the use of the LAM framework:

1. Its structure is basic enough to use within a participatory-modelling context. Indeed, the model has been applied several times in this context (e.g. Roberts et al., 2012; Beverly et al., 2013; Doole, 2013).
2. The calibration of the model is straightforward, thereby improving the clarity and interpretation of model output. This contrasts calibration methods that employ

nonlinear calibration functions estimated using positive mathematical programming (Doole and Marsh, 2014a, b).

3. Its structure is sufficiently flexible to promote its application in diverse circumstances (cf. Roberts et al., 2012; Holland and Doole, 2014).
4. Different components of the model can be developed independently, before their integration in the catchment model. This is observable in the HRWO process, where output from multiple work streams has been integrated within the catchment-level model.
5. Modern optimisation codes and computers are so well developed that any relevant number of land uses, mitigation options, and pollutants can be incorporated (Doole, 2010).
6. The LAM framework is sufficiently flexible that it can utilise information from a complex hydrological model in a clear and structured way (Doole, 2013, 2015). This is a key benefit since the utilisation of the economic model developed to inform the HRWO process has required the explicit consideration of attenuation, groundwater lags, and the linkages between sub-catchments present within the catchment network (Section 4).
7. The model does not possess a complex structure. This improves the ability to communicate about what is required among different members of a modelling team.
8. The model focuses on agricultural land uses, but point sources (e.g. urban areas) and natural sources (e.g. geothermal activity) are easily incorporated.
9. The complexity of the model can be altered, depending on the quantity and quality of resources (e.g. time, budget, expertise) available. This is a key requirement of a modelling framework that is to be applied in multiple contexts (Doole and Pannell, 2013).
10. The model is efficiently coded and solved in spreadsheet (Roberts et al., 2012) or optimisation (Beverly et al., 2013; Doole, 2013) software. It is particularly straightforward to code in nonlinear optimisation software, such as the General Algebraic Modelling System (GAMS) (Brooke et al., 2014), that allows matrix generation (e.g. Doole, 2013; Doole et al., 2013).

Nevertheless, alternative frameworks for catchment-level analysis could be employed in the HRWO process.

One alternative approach would involve the utilisation of a much-simpler framework. This commonly involves the simulation of a (usually small) number of pre-defined scenarios—generated by technical experts and/or stakeholders—within a spreadsheet model containing a low number of equations (frequently less than 20) to highlight the potential costs of these alternative outcomes (Harris and Snelder, 2014). Effective participatory modelling seeks to balance the benefits of a streamlined modelling framework with its ability to provide a powerful description of the problem at hand. In the HRWO process, in line with previous applications in the New Zealand context (Daigneault et al., 2012), the goal of the modelling exercise is to provide a detailed description of the problem at hand. This supported by the inherent size and intricacy of the water-quality challenges facing the Waikato and Waipa River catchments. Indeed, this large catchment contains: (a) significant heterogeneity in terms of farm systems, (b) numerous point sources, (c) broad-scale diversity in attenuation, (d) non-trivial surface water linkages, (e) complex groundwater legacies (Doole, 2013; Elliott et al., 2014), and (f) a large number of industries that wish that a meaningful description is provided of their sector. Moreover, when seeking to evaluate alternative limits and targets using scenario generation, it is difficult to decide between what effects that different policies will have on land use and agricultural management within the catchment of interest. Indeed, the evaluation of a given set of limits will be constrained to the assumed land-use and land-management combination that is defined to achieve these goals. Accordingly, the relative value of different outcomes is strongly biased by the assumptions made regarding what is the perceived outcome of these instruments (Doole and Pannell, 2008b).

A core difficulty here is the lack of a precise objective that acts as a standard that can be used to compare different scenarios while they are being generated. Indeed, it is common practice to set scenarios in an informal way characterised by trial-and-error and the perceptions of the stakeholder group. In contrast, an optimisation model provides a more-formal assessment, drawing together relevant economic and biophysical data, and then utilising a measure of relative cost-effectiveness to compare alternative outcomes during a search for a solution that achieves a goal at least cost. This can provide a more-structured means of providing concrete insight into what direction land-use and agricultural management should take, if water quality is to be improved. It is particularly relevant when the catchment is large and contains a significant number of stakeholders, as the model provides an appropriate vehicle to describe this rich context, relative to a smaller, more-streamlined framework.

In contrast, an alternative approach is to utilise the NZFARM (New Zealand Forestry and Agriculture Regional Model) (Daigneault et al., 2012, 2014) framework. The structure of this model is very similar to that utilised within the LAM model, with one critical difference. In contrast to the LAM framework, the NZFARM model employs a series of nonlinear functions—within a broad approach known as positive mathematical programming (PMP) (Howitt, 1995)—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). LAM model is developed according to the philosophy that appropriate calibration functions are non-trivial to develop, while they also introduce inherent bias in scenarios away from the reported baseline (de Frahan et al., 2007; Doole and Marsh, 2014a, b). Thus, the economic model used within the HRWO process takes a different approach, to provide a rich assessment of how land use may be expected to change under different scenarios. First, the division of each sub-catchment into separate partitions—each containing individual representative farming systems—dampens the susceptibility of the model to wide divergences from the baseline land allocation and provides structure on which to base any such deviation. Second, appropriate transition costs are defined, such that the model must consider the cost of land-use change when there are departures to the baseline land allocation. Third, land-use change is constrained according to historical land-use patterns (see next paragraph for more information), to reflect that land-use change is relatively inflexible in the short-term and governed by biophysical factors that vary spatially (Kerr et al., 2012). Last, both constrained and unconstrained land-use change scenarios are explored, to reflect how model output changes when constraints on land-use change are omitted.

The economic model used within the HRWO process utilises historical land-use patterns to constrain land-use changes to realistic levels. This approach was deemed appropriate in this application because it is straightforward to code, much easier to formulate and less prone to error than forcing calibration through the use of arbitrary calibration functions (Doole and Marsh, 2014a, b), draws on regionally-specific data, and is the only land-use calibration method that has a rich theoretical justification (Onal and McCarl, 1991; Chen and Onal, 2012). Historic land-use patterns observed for a distinct region (i.e. sub-catchment) provide specific insight into the type of land-use change that can occur there. Indeed, these patterns provide spatial information regarding the implicit aggregate and biophysical factors that guide land-use change within this area. Using this historical information within the catchment model applied here allows the specification of a well-behaved aggregate model, despite lacking data for individual farms (Onal and McCarl, 1991; Chen and Onal, 2012). To use this approach, historic land use for each sub-catchment across 1972–2012 was drawn from the work of Hudson et al. (2015). The optimisation procedure then identified the best weighted average of these land-use patterns that attained the environmental limits set out by each scenario at least cost.

Another alternative catchment-level modelling framework that could be applied within the HRWO process 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), Hawkes Bay (Schilling et al., 2012), Southland (NZIER, 2014), Taupo (Anastasiadis et al., 2013), and Waikato (Doole, 2010; Doole et al., 2011; Doole and Pannell, 2012; Doole et al., 2012). 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. An agent-based framework is not utilised here because of a lack of suitable 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 (Doole et al., 2011), and the significant cost and time associated with collecting suitable data from producer populations to inform model development.

An additional model that could be utilised within the HRWO process is the Waikato Integrated Scenario Explorer (WISE) framework (Hart et al., 2013). This is an integrated model that deals with a broad range of factors relevant to the Waikato Regional Council, including biodiversity, climate, demography, economics, hydrology, and land use. The land-use model within WISE determines how transition between alternative enterprises occurs under different circumstances. The supply of land is dependent on land suitability, proximity of other enterprises, and zoning restrictions, while the demand for land is driven by economic forces. The WISE framework has been developed to inform long-run planning for the

Waikato Regional Council. However, its broad focus over a wide range of issues of interest to the Waikato Regional Council reduces its particular value in the context of estimating farm-level costs arising from limit setting. Evaluating the potential implications of alternative limits within the HRWO process requires a rich description of many complex relationships, both economic and biophysical, which are not presently defined within the WISE model and, based on preliminary discussions with the people involved in the development of WISE, would be expensive and time-consuming to incorporate. As outlined above, improving water quality within the Waikato and Waipa River catchments is a large and complex problem, containing a lot of land, significant heterogeneity in terms of farm systems, numerous point sources, broad-scale diversity in attenuation, nontrivial surface water linkages, and complex groundwater legacies. These complexities are more likely to be easily integrated within a land-use optimisation model, such as the LAM framework, which is specifically designed for this purpose rather than a system-dynamics model primarily built to study other issues. Furthermore, the WISE model focuses on dynamic decision-making. The equilibrium approach utilised within the LAM model is favoured relative to this approach, given the limitations to dynamic modelling identified above, particularly those related to a scarcity of pertinent data relating to how important economic and biophysical parameters change across time.

A key motivation to employ the LAM model is also based on one of general cost-effectiveness. The HRWO process has been allowed access to a comprehensive land-use optimisation model, consistent with the LAM approach, developed by the Waikato Economic Impact Joint Venture process (Romera et al., 2014). This model received significant investment from partners within the Economic Impact Joint Venture, particularly related to the accurate depiction of the relationship between farm profit and nutrient mitigation on dairy farms and sheep and beef enterprises. The development of this model framework represented a significant investment from partners within the Economic Impact Joint Venture, in terms of expertise, resources, and time. Accordingly, it is a better use of resources to extend this current model than seek to build an alternative framework, a decision that is reinforced when the insufficiency of alternative approaches is considered.

3. Estimation of abatement-cost relationships for nitrogen

There is an inherent relationship between farm-system management and nitrogen loss within the agricultural enterprises in the Waikato region (Doole, 2013). Therefore, a key data input into the LAM framework utilised in the HRWO process is the relationship between farm profit and nitrogen mitigation (i.e. abatement cost curves, Hanley et al., 2007) for representative farms on dairy, dairy support, sheep and beef, horticulture, and cropping land. The primary data collection to derive these abatement-cost curves occurred during Stage 2 of the Economic Impact Joint Venture process in the Waikato region (Romera et al., 2014). Various key decisions were made during this process to ensure that the quality of the data was adequate. The following paragraphs discuss these factors and justify the key decisions that were made.

The goal of collecting farm-level data for a catchment-level model is to provide an adequate description of the average or expected cost of abating a given level of contaminant(s) arising from a farm population in a given spatial zone. These spatial zones are usually defined in terms of rainfall, soil type, and slope. 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 across an entire population (Roberts et al., 2012; Doole et al., 2013), and not just for individual landholders (Doole, 2010). The precision with which individual farms are represented within a catchment model varies most importantly according to the resources, time, and quality of data available. Catchment models typically vary across a continuum, from those that incorporate:

1. Individual farms (Doole, 2010, 2012).
2. Representative farms where average, individual enterprises are each used to represent a broad part of the catchment (Ramilan et al., 2011; Daigneault et al., 2012; Doole, 2013).
3. A single farm that is scaled up and used to depict an entire catchment (Doole and Pannell, 2011).

Most catchment models utilise approach #2, as employed in the HRWO model, because it allows for: (a) key differences arising from spatial and sectoral diversity, (b) the depiction of heterogeneity in abatement-cost relationships, (c) the number of representative farms considered to be altered depending on the availability of information and resources, and (d) it does not introduce substantial error through relying on data of a poor quality to generate models that purport to provide a rich description of the behaviour of diverse, individual producers (a key problem with approach #1 listed above).

There were different streams of work that sought to identify these abatement-cost relationships for dairy, dairy support, sheep and beef, horticulture, and cropping enterprises within the HRWO process. Work streams involved representative sampling or case-study analysis of farming activities to identify and represent different systems of each land-use type. The dairy farm work stream identified 26 representative dairy-farming systems, with up to 18 relevant mitigation scenarios modelled for each one, and ten dairy-support systems across the region (DairyNZ Economics Group, 2014). For drystock farming, five representative systems were described, over which five levels of mitigation were modelled (see details in Olubode et al., 2014). Horticulture was represented by three representative rotations and seven mitigation options (see details in AgriBusiness Group, 2014).

DairyNZ Economics Group (2014) described the method used to generate the abatement-cost curves for the dairy enterprises. The subsequent discussion of how this data is generated is loosely based on this material.

Biophysical and financial data were collected for 500 dairy farms for the 2012/13 season and OVERSEER files were created for each farm using a DairyNZ protocol. From this large sample, 26 farms from the Waikato and Waipa River catchments were selected, based on the range of farm types present within this broad geographical area. These 26 farms were chosen because they covered a range of locations with different bio-physical characteristics and they also represented a range of systems. Accordingly, the representative farms possessed heterogeneous levels of financial performance and nitrogen loss per ha. More specifically, this range of farm types included diversity in farm production system (as determined by external feed input), level of nitrogen fertiliser application, milk production per hectare, infrastructure, soil types, rainfall levels, and nitrogen leaching per hectare.

A baseline FARMAX file was created utilising the physical and financial data collected for each farm. OVERSEER (Version 6.1.2) and FARMAX were used simultaneously, as FARMAX allows the user to ensure that energy requirements are met for cows and the impact of mitigation options on farm financial records is clear, while OVERSEER allows the impact of mitigation options on nitrogen loss to be modelled. Mitigation options were generated through consultation among DairyNZ staff and farmers and a generalised mitigation protocol was developed and documented. This mitigation protocol described what, when, and to what degree different mitigation options were enacted on each farm, so that all farms generally followed the same overall process. Nonetheless, there were subtle differences in mitigation use between farms, due to differences in their individual characteristics. Mitigations were applied to two farm scenarios for each of the 26 farms. These can be broadly described as management changes within the current farm system first (Scenario 1), followed by an infrastructure change (Scenario 2). Scenario 1 involved the baseline farm scenario, where cows were wintered on a support block. Scenario 2 involved the incorporation of a stand-off pad.

The adoption of diverse mitigation options within the mitigation protocol followed a standardised sequence:

1. If the farm has an existing feed pad or standoff pad, this is utilised.
2. Autumn nitrogen fertiliser applications are reduced and then removed.
3. Spring nitrogen fertiliser applications are reduced and then removed.
4. Imported supplement use is reduced (up to a 20% reduction from the base).

If the farm has an existing standoff pad, the time that cows spend on the pad is increased up to 3 months per year (for up to 12 hours per day) to augment the proportion of nitrogen excretion that can be captured. The extent of utilisation of this mitigation option depends on the characteristics of the existing facilities. Where nitrogen-fertiliser application is reduced, autumn applications are targeted first, followed by spring-fertiliser applications. This is done in steps of 25% or the removal of whole dressings. Up to here, the use of purchased feed is kept constant as a proportion of the total DM intake, though feeds with a high-nitrogen content are replaced by low-nitrogen content alternatives. The proportion of purchased feed in the diet is then reduced, by up to 20% relative to the baseline situation. If a farm has a large crop area used to winter cows, crops with a lower nitrogen-leaching risk are also used as an initial mitigation option. This was applied to some case-study farms.

Each of these steps reduces feed supply further and further, and is therefore accompanied by a reduction in feed demand to achieve appropriate pasture cover throughout the year. This is done either by reducing stocking rate or the amount of feed eaten per cow. The process stops when stocking rate, milk per cow, and supplement use as a proportion of total feed offered reach 80% of their original level. Further reductions are not considered because drastic changes in either of these variables are unlikely to be desirable to the farmer, for a given level of management ability and a farm's biophysical assets. The relationship between the level of nitrogen leaching, measured using OVERSEER, and the level of financial performance, measured utilising FARMAX, are then related through abatement-cost relationships for each enterprise.

The relationship between nitrogen leaching and profit for all enterprises is a critical component of the economic model used within the HRWO process. The protocol generated and applied by DairyNZ Economics Group (2014) results in abatement-cost curves that are directly upward-sloping; that is, mitigation cost increases as the amount of abatement performed increases. This is in line with existing theory from environmental economics that stipulates that abatement effort usually imposes an explicit cost on individual enterprises (Hanley et al., 2007). Nonetheless, this finding contrasts a number of New Zealand case studies, in which profit is increased and contaminant loss is reduced simultaneously on NZ pastoral farms (AgFirst, 2009; Doole, 2012; Doole and Kingwell, 2014; Ridler et al., 2014). (Such outcomes are commonly referred to colloquially as "win-win" options.) Doole and Kingwell (2014) outline how such outcomes can arise in the context of grazed dairy systems within New Zealand. Their chief insight relates to an inherent need for producers to improve the efficiency with which feed (pasture and/or supplement) is utilised, such that more profit is earned for a fixed level of nitrogen input, especially that related to the use of nitrogen fertiliser and imported supplement. Jiang (2011) highlights that there is substantial scope for New Zealand dairy farmers to increase their efficiency. However, Doole and Kingwell (2014) suggest that win-win solutions are the exception, rather than the rule, for grazed dairy farms within New Zealand. This is consistent with empirical data presented within Doole (2012), whom identified that a 10% reduction in nitrogen across a population of 410 actual dairy farms allowed a number of them to experience win-win outcomes. However, profits unequivocally fell for all farms when greater reductions in nitrogen-leaching were required. Accordingly, this author, "highlights the absence of significant win-win abatement technologies for nitrogen in New Zealand dairy systems" (Doole, 2012, p. 17). These results are in line with observations made during the Upper Waikato Sustainable Milk Project, an intensive farm-planning exercise that involved over 700 farms in the Upper Waikato catchment. In this project, it was observed that profit would generally increase or stay the same when 10–15% reductions in nitrogen leaching were needed, given the scope for improving efficiency of nutrient use, but it was also found that profit was likely to fall when greater decreases in nutrient loss were required (Mike Scarsbrook, DairyNZ, pers. comm.).

The input data utilised in the catchment-level model for the HRWO process generally represents abatement-cost curves in which mitigation cost generally increases as nitrogen leaching falls within a given farming system. There are several reasons that this is most judicious:

1. There is both anecdotal (AgFirst, 2009; Ridler et al., 2014) and empirical evidence (Doole, 2012; Doole and Kingwell, 2014) that win-win outcomes can be achieved in grazed dairy systems in New Zealand. However, there remains a pertinent lack of data relating to whether these can be achieved across an entire population and, if so, to what degree. A key example is that farmlet research (Glassey, 2013) and modelling work (Vogeler et al., 2014) has shown that dairy systems in the Waikato can increase production, increase profit, and markedly reduce leaching, even at a low stocking rate. Nevertheless, the extent to which this set of mitigation practices is able to effectively diffuse through a population of farmers of different management skill remains unknown, particularly as it requires a high degree of knowledge such that feed demand and supply is cost-effectively balanced throughout the year.
2. It is costly to identify win-win outcomes for individual farms (Doole and Kingwell, 2014), particularly because these will be different across a farm population within a given catchment due to broad diversity in management skill, current organisation, and biophysical resources (Doole, 2012).
3. Even if win-win outcomes are *identified* for individual farms, the extent of *their actual adoption* by producers remains unclear due to barriers to uptake that are not considered during standard financial evaluations. Indeed, it is clear that while profit is important to farmers, it is not the only objective of concern (Bewsell and Brown, 2011). Such barriers can be related to risk, uncertainty, adjustment costs, system

impacts, incompatibility with lifestyle and values, and complexity (Pannell et al., 2006, 2014). Additionally, some managers are unwilling to deviate from established management plans, given a strong drive to repeat learned actions, even in the presence of new opportunities or constraints (Gonzalez and Dutt, 2011). This is identified in the case of water-quality improvement in New Zealand by AgFirst (2010), who found that the adoption of win-win solutions identified in AgFirst (2009) was marred in several circumstances because of risk aversion and perceived limitations in the economic assessment of these practices (AgFirst, 2010).

4. Win-win solutions are likely to exacerbate pollution through the Jevons paradox, which states that improvements in efficiency are likely to stimulate further intensification, as these improvements open up new opportunities (Alcott, 2005; Doole and Kingwell, 2014).
5. Broad evidence of win-win solutions in grazed dairy systems arises from linear-programming models of these enterprises (e.g. Doole, 2010; Ridler et al., 2014). These linear-programming models provide a very coarse and restrictive description of grazed dairy systems due to their high level of linearity. For example, the linear-programming frameworks utilised by these authors assume fixed pasture growth and quality, constant cow intake, and represent no endogenous feedback between stocking rate, herbage allowance, and pasture utilisation. These simplified assumptions greatly reduce the complexity of the model, allowing it to be developed and solved much more easily. Nevertheless, linear-programming frameworks of grazed dairy systems have been shown to provide inaccurate predictions of how these systems behave in reality, given these simplifying assumptions (Doole and Romera, 2013; Doole et al., 2013).
6. Substantial transformations have been observed in some nations, due to the broad adoption of win-win solutions. For example, erosion rates on cultivated land have been greatly reduced throughout Australia and South America, due to the wide-scale diffusion of reduced-cultivation practices that stimulate crop yield through water conservation (Berger et al., 2010; McRoberts and Rickard, 2010). These examples are indicative of disruptive technologies that can revolutionise an industry through offering extensive private benefits above those accruing to standard practice, such as the tractor replacing draught power in the North American cotton industry in the early 1900s (Day, 1967). There is extensive evidence that such disruptive technologies do

not exist in the context of nitrogen loss from New Zealand farming systems, as current mitigations do not achieve substantial private benefits with high certainty, relative to current management options (Doole and Romera, 2014; Doole, 2015; Doole and Kingwell, 2015). Accordingly, an upward-sloping abatement-cost curve is deemed to be the most-applicable representation of mitigation costs facing an average producer. Additionally, even if such practices were beneficial to the environment, their rate of use across space and time may vary markedly in response to changes in the physical and economic environment (Pannell et al., 2014)—especially in response to fluctuations in price and seasonal climate (Llewellyn et al., 2012)—thus reducing their general value for achieving conservation outcomes.

7. The introduction of limits for nitrogen leaching in a given region could provide an evolutionary force that will lead to the exit of producers that are inefficient from a nutrient-loss perspective, and their replacement with producers that are efficient users of nutrients. While this is possible, it is likely that if it indeed occurred, it would take an extended time period for it to take place, probably constituting decades. This reduces the possibility that evolutionary forces introduced by limits will be the source of win-win outcomes at the population level, especially once discount rates are applied that act to greatly reduce the scale of benefits experienced over periods constituting decades. The application of a standard discounting procedure augments the relative value of benefits and costs incurred closest to the present time. Industry transformation in the near term will impose a cost as less-efficient farmers withdraw. These costs will be significant, relative to the potential gains, because discounting has much less effect on net benefits arising in the near future, relative to prospective benefits arising decades from now.

Abatement-cost relationships were also determined for drystock farms. Based on a recent WRC survey of 450 drystock farms in the region (Kaine, 2013), 20 farms were selected for case-study analysis, to provide an adequate representation of farm system and spatial diversity within the catchment. Biophysical and financial information were collected for each farm during the case-study survey. These data were extrapolated to different spatial regions within the catchment also, using regional climate and financial data, for the purpose of generalisation. Utilising this information, FARMAX and OVERSEER were employed to identify the relationship between nitrogen leaching and farm profit for different scenarios on five representative farms. The farm-level data were validated through comparison with previous research and review by farmers, industry representatives, rural consultants, and scientists. This work revealed the presence of one “win-win” set of outcomes, associated with increasing the sheep: cattle ratio on hill country with pasture-based dairy support. Here, farm profit increases by 91% and nitrogen leaching falls by around 20% when the largest reduction in N is achieved. A subsequent review of this work identified that these gains are possible, but rely on existing price relativities and a willingness for producers to expand sheep operations, which is converse to the preference of some drystock farmers (Olubode et al., 2014). It is also inconsistent with recent rises in the beef price received by New Zealand graziers. Accordingly, this outcome was reviewed and other data derived for sheep and beef farms, upon the request of the Collaborative Stakeholder Group within the HRWO process.

Three horticulture farm types were represented in the model. On behalf of the Ministry of Primary Industries and Horticulture NZ, the AgriBusiness Group (2014) collected physical, financial, and environmental data from horticulture farms throughout the Pukekohe region. This was used to establish three representative types of horticultural system, each characterised by a different rotation of vegetables, for the Lower Waikato region. Three mitigation techniques were also generated based on information from industry experts, prior research, and growers. The rotation options were an extensive rotation, intensive rotation, and traditional market-garden arrangement. The mitigations were limiting nitrogen fertiliser application, reducing nitrogen fertiliser application, and improved water management through altering standard irrigation practices. OVERSEER was used to estimate the nitrogen losses associated with these enterprises, while gross-margin analysis was used to assess the financial implications of adopting these alternative mitigation practices across the different farm systems.

The profitability of alternative pastoral management systems was evaluated utilising FARMAX software throughout the generation of farm data for the economic-modelling effort in the HRWO process (Bryant et al., 2010; White et al., 2010). FARMAX is the leading software product in New Zealand utilised for evaluating alternative management systems in pastoral farming. It has been extensively applied and validated under New Zealand conditions (e.g. Bryant et al., 2010), and is broadly-used for extension (e.g. AgFirst, 2009; PAC, 2014) and research (e.g. Li et al., 2012) purposes. FARMAX provides a consistent benchmark for estimating profitability across the dairy, dairy support, and drystock enterprises represented in the HRWO economic model. As a simulation model, it does not endogenously identify the management system that maximises profit within a given scenario, such as one involving a certain level of use of a given mitigation or requiring a given level of reduction in leaching to be achieved (Doole, 2015). Accordingly, the personal preferences and experience of the user are likely to have a significant impact on the quality of model output (Doole and Pannell, 2008b). For this reason, the FARMAX simulations that were undertaken across all enterprises were guided by the application of mitigation protocols, developed before the modelling took place through discussion among producers, extension officers, and scientists. This was deemed to be a more rigourous process than employing optimisation models to identify these relationships, despite the capacity of optimisation models to more efficiently identify those management plans that maximise farm profit in a given set of circumstances (Doole, 2014, 2015). This decision is appropriate because the protocols can be used to ensure that the simulated producer response is in line with expectations regarding the response of real farmers to the imposition of limits. This is in contrast to optimisation models, in which management plans can change drastically and be inconsistent with expected responses given the primacy of managing a farm to minimise the cost of limits when the problem is defined in this way. Moreover, it is consistent with significant doubts raised with the ability of commercial linear-programming models to adequately describe New Zealand grazing systems (Doole et al., 2013b) and their much lower use in industry, compared with FARMAX. In comparison, the financial implications of alternative management systems within the horticulture industry were determined using standard farm-budgeting techniques, outside of a commercial package. This approach is justified given that it is less pertinent to evaluate system-level effects for a pre-determined crop rotation on a horticultural farm typical of the Lower Waikato, given the explicit lack of integration with other parts of the farming system in such a context.

The OVERSEER model was employed to estimate the nitrogen-leaching loads associated with different enterprises. It is the leading software used to identify the implications of alternative management strategies for nitrate-leaching loads in New Zealand farming systems (Doole and Paragahawewa, 2011). Hence, it is extensively used for this purpose. In particular, it is widely applied for extension (AgFirst, 2009; PAC, 2014), research (Ledgard et al., 1999; Doole and Pannell, 2011), nutrient management (Monaghan et al., 2007), and limit-setting (WRC, 2011). Cichota and Snow (2009, p. 243) highlight that, "it is well suited for handling management practices and environmental conditions particular to New Zealand". Extensive validation of OVERSEER has occurred (Wheeler et al., 2006, 2010; Shepherd and Wheeler, 2012). For example, Wheeler et al. (2006) reported that OVERSEER had a 99 percent accuracy rate when predicting N leaching loads. Doubts have been raised about variability in OVERSEER estimates, especially as versions of the software change (Howard et al., 2013; AgriBusiness Group, 2014). Nevertheless, Shepherd et al. (2013, p. 7) stated that, "Precision in the context of OVERSEER is about precision of inputs. Better precision and reduction of uncertainty could be attained by developing comprehensive guidelines for entering input data into the model." Accordingly, it is ensured that the same practitioner is used to perform the OVERSEER analysis for each farming system in the data-collection process. Precision is also aided through the utilisation of a mitigation protocol in each enterprise.

There are available substitutes for OVERSEER, though their relative quality is highly variable. The Soil Plant Atmosphere System Model (SPASMO) (Rosen et al., 2004) has been shown to provide closely-equivalent output to that generated by OVERSEER (Green et al., 2004; McKay et al., 2012). However, it was not utilised here given that the farm-systems experts who generated the leaching estimates were experienced with OVERSEER; SPASMO has received most use in horticulture (Cichota and Snow, 2009); SPASMO assumes a uniform return of excreta to the soil, reducing its relevance to pastoral systems (Cichota and Snow, 2009); OVERSEER is freely available and widely used; and the use of SPASMO would have added little given their close equivalence (see McKay et al. (2012) and citations therein). The Nitrogen Leaching Estimator (NLE) (Di and Cameron, 2000) provides an estimate of nitrogen leaching based on an empirical relationship between the amount of nitrogen available for leaching and that which is typically lost under a given land use. The NLE procedure has not been calibrated for areas outside of Canterbury and overall does not deal with diversity in leaching risk across the year according to differences in drainage and plant uptake across months/seasons. ECOMOD (Johnson et al., 2008) and the Agricultural Production Systems sIMulator (APSIM) (Keating et al., 2003) also provide some ability to estimate nitrogen leaching, but should mainly be considered research tools given that a large amount of input data and specialist knowledge is required to apply them (Cichota and Snow, 2009). In this process, it was determined that the richer output provided by this set of process-oriented models was not justified. Rather, there was an explicit need for different industry experts to be able to interact with the framework and represent a high number of scenarios across a broadly-disparate set of farm operations and systems. To this end, the broad application of OVERSEER in previous projects, its applicability to New Zealand conditions, and its user-friendly interface were key drivers of its selection to use in the HRWO process.

4. Hydrological modelling

The HRWO process focuses on four contaminants arising from non-point and point sources in the Waikato River catchment. OVERSEER is used to estimate the nitrogen-leaching loads from agricultural land (Section 3). This section describes the estimation of the baseline loads of the remaining contaminants (phosphorus, sediment, and *E. coli*), how they are distributed through the flow network, attenuation rates, and links to attribute levels at each monitoring site. It also outlines why the adopted means of formulating these estimates are appropriate, in comparison to available alternatives.

The HRWO economic model incorporates diverse hydrological models that relate contaminant losses within and across sub-catchments to pollutant concentrations at the various monitoring sites represented within the catchment. These models concern *E. coli* (Semadeni-Davies et al., 2015a), sediment (Yalden and Elliott, 2015), nitrogen (Semadeni-Davies et al., 2015b), and phosphorus (Semadeni-Davies et al., 2015b). These models are broadly based on the Catchment Land Use for Environmental Sustainability (CLUES) model (Semadini-Davies et al., 2007), which has been broadly applied throughout New Zealand to represent the flows of a broad range of contaminants through a catchment network, both within environmental (e.g. Monaghan et al., 2010; Palliser and Elliott, 2013; Elliott et al., 2014) and economic (e.g. Doole, 2013) studies. Key reasons for using this unifying framework are that it is flexible enough to deal with variation in data quality, quantity, and form among diverse contaminants (cf. Yalden and Elliott (2015) and Semadini-Davies et al. (2015b)); it has been broadly applied across New Zealand in a variety of contexts; its modular structure is amenable to reconstruction within an optimisation model of a typical form (Doole, 2013); and it has been applied successfully on previous occasions in the Waikato context (e.g. Semadini-Davies and Elliott, 2012; Elliott et al., 2014). The integration of these models into the economic model allows the depiction of an explicit relationship between land management, point-source management, and concentrations of chlorophyll *a*, Total Nitrogen, Total Phosphorus, nitrate, *E. coli*, and black disc measurements at different sites across the catchment.

A key feature of these hydrological models are estimated fate-transport matrices, which specify the flow and attenuation of contaminants between linked sites in the monitoring network. Various limits are evaluated in the scenarios through specifying the attribute concentrations that meet the scenario's desired band for median concentrations of chlorophyll-a, maximum concentrations of chlorophyll-a, Total Nitrogen concentration, Total Phosphorus concentration, median nitrate concentration, 95^{th} percentile nitrate concentration, median *E. coli* concentration, 95^{th} percentile *E. coli* concentration, and water clarity. More detail about the models utilised for each contaminant are outlined in the associated reports listed above, with an overview provided below.

It is possible to develop very comprehensive models of watersheds, based on existing frameworks and associated methods of parameterisation (Beven and Binley, 1992; Beverly et al., 2005; Abbaspour et al., 2007). However, these require a rich data sets if they are to be adequately applied (Craig Beverly, personal communication, 22/1/2015), and therefore are not appropriate to apply in this instance. Indeed, good modelling practice requires that the complexity of a modelling framework matches the quantity and quality of available data (Doole and Pannell, 2013). In line with this principle, a generalised high-level framework broadly based on the structure of the routing framework in CLUES (PCE, 2013; Elliott et al., 2014) is employed in this analysis. This follows the successful application of a similar hydrological-network model in the Upper Waikato in Doole (2013).

Baseline estimates of nitrogen and phosphorus losses formulated for different enterprises in the Waikato River Independent Scoping Study (WRISS) approach (NIWA, 2010) did not account for slope and rainfall. Accordingly, nitrogen losses for each representative farm were estimated utilising OVERSEER in the HRWO process (Section 3). Additionally, phosphorus losses were estimated with this software, during Stage 2 of the Economic Impact Joint Venture. The movement of these nutrients throughout the catchment within the flow network, attenuation rates across space, and the link to concentrations observed at each monitoring site are estimated using CLUES (Semadini-Davies et al., 2015b).

There are available substitutes for OVERSEER for estimating phosphorus losses; for example, APSIM and SPASMO are both suited to this purpose. However, OVERSEER is applied here for estimating phosphorus loads, based on the reasons outlined in the last paragraph of Section 3, which pertain to why this framework was also preferred for estimating nitrogen loss. These are mainly related to the suitability of OVERSEER for studying nutrient loss in New Zealand conditions, the experience of industry experts with this framework, and to maintain consistency with the way that nitrogen-loss estimates were formulated. This is in line with typical industry practice. Indeed, in a recent analysis of nitrogen and phosphorous loss in Southland, OVERSEER was the preferred tool for estimating the leaching levels of both nutrients (Journeaux and Wilson, 2014).

However, a workshop that brought together contaminant-loss experts from across New Zealand to review the catchment-level framework developed by the Economic Impact Joint Venture highlighted that some important sources of phosphorus loss are not dealt with well in OVERSEER software (Ross Monaghan, personal communication, 22/9/2014). Also, this model assumes good management practice, so will not capture improvements in farm management. Accordingly, the OVERSEER estimates formulated by contractors to the Economic Impact Joint Venture were reviewed twice by a team of experts (Ross Monaghan and Richard McDowell at AgResearch). Additionally, the level of phosphorus loss associated with sediment loss from agricultural land and stream banks is considered. This is identified through determining the amount of erosion that is occurring (see below) and using this to compute the total level of phosphorus present in this material, which subsequently enters waterways within the catchment. The phosphorus content of sediment was estimated through calibration of this parameter within the catchment model built to estimate phosphorus attenuation across the Waikato and Waipa River catchments. The phosphorus content of sediment was constrained to lie within a lower and upper bound, drawn from the literature, and the value of this parameter was then identified such that the difference between observed and predicted TP levels at each monitoring site was minimised, after dissolved phosphorus loads (estimated through OVERSEER) were accounted for.

The baseline levels of sediment loss across the catchment are estimated using the New Zealand Empirical Erosion Model (NZEEM) (Palmer et al., 2013; Dymond, 2014). NZEEM is an empirical model that estimates the erosion rate present on a given piece of land, as determined by rainfall, slope, parent material, and vegetation cover (Dymond et al., 2010). NZEEM is calibrated to about 200 sediment-yield data sets from across New Zealand; thus, it represents an appropriate tool for the identification of loss rates in this catchment. The most appropriate frameworks that could be utilised to represent the routing of sediment throughout the catchment are SedNetNZ and the Catchment Land Use for Environmental Sustainability (CLUES) model (Semadini-Davies et al., 2007). SedNetNZ is a spatially-distributed model that routes sediment through a stream and river network, accounting for losses in water bodies and deposition (Dymond, 2014). In comparison, CLUES has been broadly applied throughout New Zealand to represent the flows of a broad range of contaminants through a catchment network, both within environmental (e.g. Monaghan et al., 2010; Elliott et al., 2014) and economic (e.g. Doole, 2013) studies. The CLUES model is utilised in this study to account for the attenuation and flow of sediments across the Waikato and Waipa River catchments. This is justified by the utilisation of CLUES to estimate the attenuation and flow of the other contaminants (nitrogen, phosphorus, and *E. coli*) studied in the HRWO process and reservations around the estimates of streambank erosion arising from the SedNet model (Hughes, 2015).

The baseline median loads of the microbial indicator *E. coli* arising from agricultural land and point sources in each part of the catchment were estimated using the CLUES model (Semadeni-Davies et al., 2015a). The estimation of microbial loads utilising the CLUES model relies mainly on an estimation procedure based on that employed within the SPAtially Referenced Regressions on Watershed attributes (SPARROW) model (Smith et al., 1997). The estimation performed within this part of the project predicts the annual average loadings of *E. coli* arising from agricultural land, while accounting for attenuation, drainage, monitoring samples observed at given measurement sites, rainfall, routing, and storage (Semadeni-Davies et al., 2015a). The SPARROW model within CLUES has been broadly applied throughout New Zealand (Elliott et al., 2005), particularly for *E. coli* estimation (Palliser and Elliott, 2013; Elliott et al., 2014), and its development involved extensive communication with the developers of the original estimation procedure in the U. S. A. (Elliott et al., 2014). The suitability of this approach for estimating baseline microbial loads and its previous application, both in the study region and throughout New Zealand, are key reasons why this method has been selected. Another key justification for using the CLUES model is that there appears to be no alternative estimation procedures that are readily available for predicting *E. coli* concentrations at the catchment scale in the New Zealand water-management context. In addition, alternative means of estimating *E. coli* loads—such as generating mean loads per land use type and then aggregating these up to the catchment level utilising data regarding the amount of area present in each land use (NIWA, 2010)—are less structured and hence are necessarily more heuristic. They also create the potential for values observed on-farm that are inconsistent with those concentrations observed at monitoring stations throughout the catchment.

The Economic Impact Joint Venture project also provided an estimate of the costs of reducing critical point-source loads within the Waikato River catchment. These are outlined in the report provided by Opus International Consultants (2013), but have also been updated based on consultation between the Waikato Regional Council and each individual point source.

5. Regional-level economic modelling

The regional- and national-level impacts of alternative water-quality targets are important to assess, given that changes at the catchment level often have flow-on effects associated with changes in spending behaviour by producers and the amount of product flowing from farms along subsequent steps in the value chain. A number of alternative strategies are available. This section provides an overall justification of why an input-output model has been utilised for this purpose within the HRWO project. The following draws from the comprehensive advice provided to Regional Councils regarding the advantages and disadvantages of alternative means of assessing the regional economic impacts of alternative water-quality improvement mechanisms in McDonald (2014).

There are three general forms of models typically employed within New Zealand to estimate the regional economic impacts of alternative water-quality policies. These are input-output models, computable general equilibrium models, and spatial decision support systems. Economic-multiplier methods are also available and broadly applied (Daley, 1997), but receive no attention here given that they are closely related to input-output models and because their quality is typically too low to justify their use for ascertaining the regional impacts of environmental policy (McDonald, 2014).

Input-output models are the most widely-applied method for estimating the regional impacts of environmental policy, both in New Zealand and overseas (McDonald, 2014). Moreover, they are the one of the most popular economic methods applied globally (Miller and Blair, 2009), based on their clarity and descriptive capacity. These models study the flow of products, inputs, and sales between households and industries. Their primary advantage is that they describe the complex interdependency between different sectors within an economy, allowing the consideration of numerous flow-on relationships arising from a change in current economic activity. Accordingly, input-output models provide a means to estimate the regional impacts of a given policy mechanism, based on the idea that an initial change in net revenue entering into a regional economy—for example, in response to a change in milk production arising from reduced dairy production intensity—will lead to new, subsequent spending in other industries within this economy, but the effect of these contributions will dissipate over time due to the leakage of funds from the local economy (e.g. through expenditure outside of the region or through saving) (Mills, 1993). Such models have many benefits; namely, their ability to capture interrelationships between different sectors, low cost, and apparent simplicity, which helps to promote the clarity of their output and the capacity to communicate key assumptions to stakeholders. Moreover, the equilibrium structure of input-output models is consistent with the steady-state approach employed in the catchment-level model utilised in the HRWO process (Section 2). The input-output model applied in the HRWO process allows the impacts of different limits to be assessed across 106 different industries across a variety of spatial scales (including the individual Freshwater Management Units identified by the CSG), a detailed degree of insight that would be missing if alternative procedures (e.g. a computable general-equilibrium model) were utilised (Bess and Ambargis, 2013).

Nevertheless, these frameworks have some limitations, particularly associated with the inclusion of price impacts, budget constraints, and technical change. The application of input-output models is based on an explicit assumption that prices remain fixed; consequently, increased competition for scarce factors of production does not flow through to affect prices (Hughes, 2003). Additionally, it is assumed that the additional output provided by marginal changes in individual input use remains fixed across simulations (i.e. there are constant returns to scale) (Miller and Blair, 2009). These assumptions are highly stylised, but are justified in applied work based on their clarity, ease to deal with during computation, the inherent focus of these models on regional markets, and the complications associated with utilising more-detailed frameworks that do consider price feedbacks and varying returns to scale (see discussion below). Indeed, in relation to the last point, it is common that seeking to include price impacts through extending a model to become a computable general equilibrium framework or spatial decision support system will often lead to a downgrade in the amount of industry-level information that is included (Bess and Ambargis, 2013). The decision to utilise an input-output analysis within the HRWO process is also partially justified by the existence of the Waikato Region Multi-Regional Input-Output Table, which was initially developed for the Waikato Regional Council Economic Futures Model. The extension of a previous framework is more cost-effective than developing a framework from nothing, especially given that the existing framework has been applied previously and extension can take into account practices and principles that were learnt during its prior employment. This decision is also consistent with the time and budget constraints that face many limit-setting processes for water quality improvement in New Zealand, including the HRWO process. The use of an input-output model also allows the linkage of the regional economic model with the farm- and catchment-level models, such that the farm-, catchment-, regional-, and national-level implications of alternative limits are able to be ascertained in an integrated way.

An alternative approach involves the application of computable general equilibrium models. The key reason that this type of model is not utilised is that the goal of the economic analysis in the HRWO process is to utilise a multi-scale approach—constituting farm-level, catchment-level, regional-level, and national-level modelling—and it is extremely challenging to meaningfully link computable general equilibrium models with those defined at the farm- and catchment-scales (Garry McDonald, personal communication, 22/01/2015). Additionally, there is a distinct lack of up-to-date regional computable general equilibrium models in New Zealand, including for the Waikato region. This is a primary constraint given that the development of such models is typically expensive (McDonald, 2014). Indeed, for the same amount of money expended on the application of the input-output model described above, it would only be possible to construct a very coarse general-equilibrium framework for the Waikato region (Garry McDonald, personal communication, 22/01/2015). Additionally, the creation of a multi-regional CGE model that reports down to the level of Freshwater Management Units would necessitate the construction of a Social Accounting

Matrix (SAM) for each of these local areas. There is a lack of information pertaining to inter-regional investment flows, particular for transfers between economic agents (e.g. from government to households), that prevented the successful completion of this task within the time frames of the project. In comparison with input-output models, computable general equilibrium frameworks provide an adequate representation of price impacts and budget constraints (Crockett, 2013), although criticism has been broadly targeted at the way in which prices are typically assumed to adjust in these models (Kirman, 1989). A primary limitation here is that prices that change at the market level must also feed into the abatement-cost relationships defined for each farm. This necessitates detailed farm-level modelling that derives abatement-cost relationships for a broad array of market prices for a broad range of different farm inputs and outputs. This extensive exercise has yet to be performed for any analysis within New Zealand that concerns water-quality policy and the application of a CGE model. Furthermore, as with input-output models, it remains difficult to describe technical change within computable general equilibrium models, especially given concerns related to how dynamic processes are typically represented within these frameworks (Solow, 2008; McDonald, 2010). Nevertheless, the application of any economic analysis is challenged by a need to incorporate innovation, especially over the time scales of interest, given that it is non-trivial to predict how technology will evolve, diffuse, and impact any particular sector (Verspagen, 2009). This further complicates the estimation of abatement-cost curves, since not only do these need to be extended to deal with broad variation in input and output prices in a CGE context, but also in dynamic CGE models there is an explicit need to incorporate adaptation and technical change, which are difficult to predict.

The main alternative available for input-output and computable general equilibrium models is a spatial decision-support system. This provides a comprehensive, holistic description of multiple elements of a regional economy, incorporating a dynamic integration of sophisticated models, often describing climate, demographic, economic, and hydrological processes. These models also typically provide for the description of output utilising Geographic Information Systems (GIS) and attempt to provide a fully-integrated description of the important processes that drive the regional economic outcomes associated with different policy instruments, including those for water quality.

The Waikato Integrated Scenario Explorer (WISE) framework is a spatial decision-support model that has received substantial investment from the Waikato Regional Council and other partners (Hart et al., 2013). It is a possible substitute for an input-output approach, being comprehensive, allowing for the study of dynamic land use, and its reception of much previous investment. Nevertheless, several factors make its integration with the catchment-level economic model (Section 3) less appropriate than the employment of an input-output framework. First, it is conceptually inappropriate to try to integrate the catchment-level model and the WISE model, given the major differences evident in their structure. The catchment-level economic model is a steady-state framework that, in contrast to the WISE model, ignores dynamic processes. This assumption sounds limiting, but is consistent with the majority of prior research in this area (e.g. Daigneault et al., 2012; Doole, 2013) and a number of key conceptual issues facing the development of dynamic economic models for policy evaluation (Section 3). Second, the size and complexity of the WISE model complicate its use within a participatory process, such as that involving the Collaborative Stakeholder Group (CSG) in the HRWO process. Indeed, large models pose significant challenges in terms of trying to interpret what drives model output (causality) (Doole and Pannell, 2013), necessitating in some instances the use of econometric techniques to describe relationships within the results of any given simulation (Doole and Pannell, 2012). Third, despite spatial decision support systems providing a comprehensive description of intertemporal processes, there is often an explicit lack of validation regarding the rules that drive these transitions and also whether they predict sensible behaviour (Parker et al., 2003). Fourth, the structure of the model requires specialist computer programming that is outside of the direct experience of the modelling team, which complicates the team: (a) understanding how the regional model is working, (b) identifying why certain results are being observed, and (c) running the model under tight time frames during an iterative process of evaluating alternative limits within a participatory process involving the CSG. Last, the cost of extending such models to fit this particular context is prohibitive, especially as it involves having to update other parts of the framework that have not received recent revision.

6. Conclusions

The development of an economic model to predict the farm-, catchment-, regional-, and national-level economic outcomes associated with alternative environmental limits is a broadranging task. In the context of the HRWO process, it has sought to integrate diverse information into a consistent framework so that it can be considered altogether during evaluation. The primary objective of this document is to outline the reasons why certain key decisions have been made during the design and development of the HRWO economic model designed for this purpose.

The development of such a framework requires the balancing of the provision of meaningful insight with the availability of scarce resources (mainly time, expertise, and funding). Accordingly, while there are often alternate, more-comprehensive means to study a given aspect of the water-quality problems studied within the HRWO process, these are not necessarily superior to those utilised here, given that they are not justified by the available data, can be more difficult to understand for stakeholders (which is important within the participatory context of the HRWO process), and inconsistent with the resources available to conduct the economic evaluation. Indeed, while detailed simulation models have been built to represent many varying aspects of water quality, concern remains over whether their development is justified based on the quality and quantity of available data (Scheffer, 2004).

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