Economic Valuation of Water Quality Improvements in New Zealand

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Abstract

Efficient decision making in environmental management requires good data on the costs and benefits of changes in environmental quality. However, full assessment of the benefits of better water quality has been a challenge because some of the component values cannot be directly measured. The advent of non-market valuation techniques has made it possible to estimate these values. In this thesis, the travel cost random utility model and fishing choice data from the National Angling Survey are used to assess the benefits of better water quality for trout anglers in the Rotorua Lakes and a choice experiment is used to assess the benefits of cleaner streams for Karapiro catchment residents. We also explore three methodological aspects which may affect non market value estimates, namely within season variability, scale heterogeneity across individuals and respondent perceptions of the status quo.

Accounting for within-season variability in site attributes that are variable across the season may reduce multicollinearity. We find that differences in welfare estimates between models accounting for within-season variability and those that do not may result from differences in attribute and collinearity levels or the combined effect of both. We assess whether benefit estimates remain stable over time using models that account for scale heterogeneity across individuals and demonstrate that ignoring scale heterogeneity across the sampled population may result in researchers erroneously concluding that estimates of marginal willingness to pay are stable over time. A choice experiment on preferences for stream water quality is used to assess the effects of respondent's perception of status quo conditions on welfare estimates. The results build on earlier findings which suggest that failure to take account of respondents' beliefs leads to biased welfare estimates.

Overall we find that lakes with better water clarity, that are larger in size, with bigger fish, more facilities and more forest cover are preferred. Similarly, streams with water quality that is suitable for swimming and where trout are found, are preferred. We estimate the aggregate annual benefit for anglers of a one metre increase in water clarity in all the Rotorua Lakes which currently have poor or average water water quality to be NZ$2.3 million. The travel cost RUM is also used to assess the overall benefit that trout anglers obtain from each lake. The annual level of these benefits totals NZ$21.7 million.

Publications from this thesis

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Marsh, Dan; Mkwara, Lena (2009), Comparing welfare estimates from fixed status quo attributes vs. people's perceived attributes of water quality, *New Zealand Agricultural and Resource Economics Society*, Nelson, New Zealand; 27-28 August, 24pgs.

Working papers

Marsh, Dan; Mkwara, Lena (2010), An assessment of the benefits of cleaner streams: A New Zealand case study, Waikato Management School, 14pgs.

Marsh, Dan; Mkwara, Lena; Scarpa, Riccardo (2010), Does respondent perception of the status quo matter in non-market valuation with choice experiments: An application to New Zealand freshwater streams, Waikato Management School, 22pgs.

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Acronyms

ASC	Alternative Specific Constants
AVC	Asymptotic Variance Covariance
BOD	Biochemical Oxygen Demand
CE	Choice Experiment
CHLA	Chlorophyll a
CL	Conditional Logit
COD	Chemical Oxygen Demand
CS	Compensating Surplus
CV	Compensating Variation
CVM	Contingent Valuation Method
DO	Oxygen Demand
DOC	Department of Conservation
EBOP	Environment Bay of Plenty
EIJVS	Economic Impact Joint Venture Studies
EPA	Environmental Protection Agency
EU	European Union
FGNZ	Fish and Game New Zealand
GMMNL	Generalized Mixed Multinomial Logit
HP	Hedonic Pricing
LakeSPI	Lake Submerged Plant Indicator
MA	Millennium Ecosystem Assessment
MFE	Ministry for the Environment
MMNL	Mixed Multinomial Logit
MNL	Multinomial Logit
MoH	Ministry of Health
MPI	Ministry for Primary Industries
MWTP	Marginal Willingness to Pay
NERMN	Natural Environment Regional Monitoring Network
NIWA	National Institute of Water and Atmospheric Research
NOAA	National Oceanic and Atmospheric Administration

NPS-FM	National Policy Statement for Freshwater Management
OLS	Ordinary Least Squares
PCE	Parliamentary Commissioner for the Environment
RDC	Rotorua District Council
REC	River Environment Classification
RMA	Resource Management Act
RUM	Random Utility Model
RWLP TLI	Regional Land and Water Plan Trophic Level Index
SD	Secchi Depth
S-MNL	Scaled Multinomial Logit
TCM	Travel Cost Method
TEV	Total Economic Value
TLI	Trophic Level Index
TN	Total Nitrogen
TOC	Total Organic Carbon
TOD	Total Oxygen Demand
TP	Total Phosphorus
TSC	Total Solid Carbon
UK	United Kingdom
UNEP	United Nations Environment Programme
US EPA	United States Environmental Protection Agency
US$	United States Dollar
USA	United States of America
USGS	United States Geological Survey
VTTS	Value of Travel Time Saving
WHO	World Health Organization
WTA	Willingness to Accept Compensation
WTP	Willingness to Pay

CHAPTER ONE

INTRODUCTION

1.0 Background

New Zealand's numerous lakes, rivers and streams have been described as the nation's "crown jewels". These freshwater resources play a vital role throughout the economy and society and provide ecological, aesthetic, scientific, recreational, tourism and educational benefits to the country. These lakes, rivers and wetlands are also integral to the cultural and spiritual well-being of Māori.

New Zealand is renowned for its abundance of high quality freshwater. The 2007 state of the environment report by the Ministry for the Environment (MFE) indicated that "freshwater is both clean and plentiful in supply" by international standards (MFE, 2007, p.261). This position was supported by international researchers such as Carr & Rickwood (2008) for the United Nations Environment Programme (UNEP) who concluded that New Zealand had the best water quality in the world, based on the UNEP water quality index. However, falling water quality in many streams, rivers and lakes especially in areas exposed to intensive agricultural production over the last two decades, is a major environmental issue facing New Zealand. The levels of nitrogen and phosphorus nutrients entering waterways have led to a progressive decline in water quality and increased incidence of algal blooms (MFE, 2007, 2008).

The level of social concern about declining water quality has grown since the mid-1990s. There have been numerous attempts to address these concerns including joint action such as the Dairying and Clean Streams Accord; campaigns by special interest groups such as the Fish and Game dirty dairying campaign[1]; and government programmes such as the 2011 Fresh Start for Fresh Water Programme

[1] http://www.nzfishing.com/Issues/DirtyDairying.htm

(Minister for the Environment and Minister of Agriculture, 2011; MPI, 2012). Policy makers in both central government and regional councils have explored the full range of regulatory and other instruments to try to attain environmental improvement, including the use of market-based tools as in the Taupo cap-and-trade scheme (Barns & Young, 2012)[2].

There is also increased interest in the use of non-market valuation methods to assist with environmental management decisions. For example, the Ministry for the Environment (MfE) initiated a project called the Economic Impact Joint Venture Studies (EIJVS)[3] which aims to provide economic analysis to support central government decision making on setting freshwater quality and quantity objectives and limits[4] (Akehurst *et al.*, 2013). A key component of the Joint Venture Project is to assess the costs and benefits of central and regional government water quality policies, including the non-market values of water.

Assessing the full value of water, including recreational and ecological values, remains a big challenge because these important components cannot be directly measured in dollar terms in the market. It has been argued that limited ability to quantify non-market environmental benefits and costs has often led to market benefits being given precedence over non-market costs, resulting in poor decision making and environmental degradation (Bennett & Blamey, 2001; Loomis, 2005; Navrud & Pruckner, 1997). However, with the development of non-market valuation methods (e.g. travel cost method and choice experiments) these values

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[2] The Taupo cap-and-trade scheme which became operative in 2011 under the Waikato Regional Plan Variation 5 is the first market for diffuse emissions of nitrogen in New Zealand. The objective of this scheme is to provide long term protection of water quality in Lake Taupo, New Zealand's largest lake (Barns & Young, 2012).

[3] This is a joint study by the Ministry for the Environment (MfE), Ministry for Primary Industries (MPI), and Department of Conservation (DOC).

[4] It also aims to work with regional councils to develop economic analysis on the economic, environmental, social and cultural trade-offs in managing water quality and quantity (Akehurst *et al.*, 2013)

can now be estimated. In New Zealand non-market valuation methods have been applied in the estimation of water-based resources since 1974. The New Zealand Valuation Database documents all the studies conducted up to 2010[5]. However, Yao and Kaval (2007), who provide an overall assessment of the New Zealand non-market valuation literature, reported a severe lack of water resource studies. This was also highlighted in a review of fresh water non-market studies by Marsh & Mkwara (2013). Availability of sufficient data on these values is an essential prerequisite to any attempt to assess the costs and benefits of water quality regulatory policies.

The importance of non-market valuation in improving environmental decision making, together with the lack of suitable valuation data (especially in New Zealand) are the main motivations for undertaking this thesis. Two case studies are used to investigate New Zealanders' preferences for better water quality. The first case makes use of the travel cost random utility model to assess the preferences and value that trout anglers place on improved water quality in the Rotorua Lakes. The second involves a choice experiment investigating the value of better stream water quality for residents in the Karapiro catchment area.

1.1 Research questions and motivation of the study

The main research question addressed in this thesis is:

How much is clean water worth?

This question is answered by investigating the preferences and willingness to pay (WTP) for better water quality in the Rotorua Lakes and in Karapiro catchment streams. The travel cost random utility model (RUM) is a state of the art technique that is generally applied in environmental valuation involving multiple recreational sites. This technique is used for the first time in New Zealand to assess the impact of water quality on trout angling in the Rotorua Lakes. Through the use of the travel

[5] http://www2.lincoln.ac.nz/nonmarketvaluation/

cost RUM, it is possible to account for substitution patterns across recreational sites induced by policy changes at one or more of the sites. This specific feature makes RUM the most popular modelling framework in recreational valuation literature (Parsons & Kealy, 1992; Phaneuf & Smith, 2004). Surprisingly, this technique has not been applied in over 30 years of New Zealand non-market valuation research.

The RUM technique is applied, for the first time in New Zealand to assess the value that anglers place on improved water quality in the Rotorua lakes. A sample of 414 anglers obtained from the 2007/08 National Angling Survey is used. The Rotorua lakes, comprising twelve major lakes, are highly treasured natural assets, located around a major tourist destination. Water quality in these lakes ranges from excellent to poor. Through the change in fishing licence sales, the decline in angler usage of some lakes due to falling water quality is documented (Pitkethley, 2008; Unwin, 2009). What remains unknown is a quantifiable measure of the value that anglers derive from fishing in clean water or their loss in welfare due to poor water quality. This motivates the first research question (Q1):

(Q1). Does water quality influence anglers' choice of lake for fishing? If so, what value do they place on water quality improvements?

The answer to this question is the main focus in Chapter Two. A conditional logit fishing site choice model is developed and used to simulate anglers' WTP for better water quality. The anglers' recreational losses due to possible lake closure are also estimated.

As is the case with any valuation method, a number of methodological issues regarding the use of the travel cost method (TCM) have been addressed by various authors. The overall objective of these authors is to improve the use of these techniques to ensure more reliable value estimates. One issue which is relatively less explored is the seasonal variability (within-season variability) in recreational site attributes across the recreational season. Within-season variability in site attributes, such as fishery regulations, catch rates and congestion is acknowledged

(Andrews, 1988; Clark, 1980; Provencher & Bishop, 2004; Swallow, 1994). However, our knowledge of this subject area remains sparse. This can partly be attributed to insufficient variation in natural conditions that characterizes most datasets of recreational site attributes. In other cases researchers might implicitly assume such variability to be too small to have any substantial effects on recreational site choice decisions and implied welfare estimates. This leads to the second research question:

(Q2) Does accounting for within-season variability in recreational site attributes that are variable across the season matter?

The Rotorua Lakes present an opportunity to explore this question for two reasons. First, the 2007/08 National Angling Survey from which this study's sample is drawn accounts for seasonality in angler demand. In addition to region and licence type, the survey was stratified by time, with the 12 month survey period divided into six two-monthly intervals (Unwin, 2009). This was done to account for the variability in angler usage of water bodies across the fishing year.

Second, water quality and fish growth tend to vary across the year and between lakes. Extensive water quality monitoring data for the Rotorua Lakes was obtained from the Environment Bay of Plenty (EBOP) regional council. This enabled computation of the bimonthly averages of water clarity corresponding to the two monthly partitions used in the National Angling Survey. Similarly, the corresponding bimonthly averages of the weight of fish were computed from comprehensive monitoring data obtained from the Eastern Region Fish and Game Council. This was to ensure that anglers' preferences are estimated using water quality and weight of fish attribute levels existing during the period they recorded a fishing trip. To answer research question two, welfare estimates from models using annual versus bimonthly averages of water clarity and weight of fish are compared. The availability of alternative data types for water clarity and weight of fish motivates the third research question:

(Q3) Can the use of less aggregated data reduce multicollinearity in revealed preference data?

The problem of multicollinearity is ubiquitous in revealed preference data relating to recreation. Multicollinearity, defined as the intercorrelation among regressors in a model and its effect on estimated parameters, is well documented (Koutsoyiannis, 1977; Maddala, 1992). Currently, the generally acceptable methodology to reduce multicollinearity is through the joint estimation of revealed and stated preference data, commonly denoted as RP-SP. The strategic design of attribute levels in stated preference surveys can reduce some of the collinearity inherent in revealed preference quality characteristics (Adamowicz *et al.*, 1994). The use of revealed preference data alone still remains more common than RP-SP due to its less extensive data requirement.

Transportation studies have explored the use of less aggregated data to reduce collinearity between travel time and travel costs (Brown & Nawas, 1973; Gum & Martin, 1975). However, there is little evidence that such an approach can reduce collinearity due to the strong association between these variables (Allen *et al.*, 1981).

The use of less aggregated data to reduce collinearity is tested further, where disaggregation is done across time and involves non-monetary site quality characteristics. Specifically, the extent to which increased variability from the use of bimonthly averages of water clarity and weight of fish can reduce multicollinearity is tested. The determinants of the asymptotic variance covariance matrix (AVC) computed from the negative of the Hessian of the log-likelihood function from models using annual and bimonthly averages of water clarity and weight of fish are compared. This investigation leads to another pertinent but yet unexplored issue:

(Q4) Does collinearity typically considered tolerable have a significant effect on welfare estimates?

The generally prescribed solution (for those not wanting to adopt the RP-SP approach) is to accept some level of collinearity.

There is no clearly defined cut-off point for the acceptable level of collinearity, but as a rule of thumb, collinearity of 0.8 or more is thought to be sufficiently high to affect estimated parameters (Hensher *et al.*, 2005). Some earlier econometric studies suggested that even moderate or low levels of collinearity can affect the precision of the parameter estimates (Koutsoyiannis, 1977; Maddala, 1992). However, it remains unclear whether these tolerable levels of collinearity can have a significant effect on welfare estimates, an issue investigated in this thesis.

The Rotorua Lakes fishing choice data set is used further to investigate the fifth question:

(Q5) Do WTP estimates remain constant over time?

Assessing the stability of values over time is considered vital because non-market valuation studies only provide a snapshot of values at a particular point in time. However, policy analysts are often required to extrapolate these values to some future time periods (Liebe *et al.*, 2012; Loomis, 1989). This issue has received very little attention in the recreational demand literature using revealed preference data. So far, only two studies have addressed this issue (Bhattacharjee *et al.*, 2009; Parsons & Stefanova, 2009). The availability of two independent fishing choice data sets for the Rotorua Lakes, collected six years apart, permits this investigation to be carried out.

Comparison of different data sets raises other concerns including scale factor differences. Swait & Louviere (1993) were the first to recognize that parameter estimates in MNL models from different data sets may differ in magnitude due to scale factor differences. Typically, the scale and utility weights are confounded and cannot be separately identified unless specific reparameterisations, and hence assumptions, are implemented. This problem is circumvented in logit model estimation by normalising the scale or standard deviation of the idiosyncratic error to a constant. More recently, models that allow for scale heterogeneity to be accounted for at individual level have been developed (Fiebig *et al.*, 2009; Greene & Hensher, 2010). These models are employed to investigate the sixth question:

(Q6) Can scale heterogeneity across individuals significantly contribute to differences in WTP across data sets?

To the best of the author's knowledge all environmental non-market valuation studies testing the stability of values over time have used models that assume scale homogeneity across respondents. The work presented here represents one of the first applications in environmental non-market valuation studies to investigate this issue. More recently, empirical evidence from the field of transportation appears to suggest that scale heterogeneity across sampled individuals may contribute to differences in mean estimates of the value of travel time saving across studies (Hensher *et al.*, 2011).

The Karapiro catchment choice experiment study is used to answer the last research question:

(Q7) Do respondents' perceptions of the status quo matter in non-market valuation with choice experiments?

In environmental non-market valuation studies using choice experiments, researchers often provide descriptions of status quo conditions which may differ from those perceived by respondents. Studies have shown that description of the status quo, or its mere presence in the choice context, is not neutral to the choice outcome (Adamowicz *et al.*, 1998a; Boxall *et al.*, 2009a; Brazell *et al.*, 2006; Breffle & Rowe, 2002; Dhar & Simonson, 2003; Scarpa *et al.*, 2005b). One area where our understanding is relatively poor is that of identifying the specific effect that respondents' perceptions of status quo conditions have on implied welfare estimates. This issue is explored by comparing willingness to pay between respondents using their own perceived quality of streams and those provided with descriptions of the status quo conditions. The Karapiro catchment choice experiment study carried out by Marsh (2008) is used in this investigation.

1.2 Contributions of the study

In this thesis eight original and significant contributions to the literature on non-market valuation are made: 1) the travel cost RUM is used for the first time in New Zealand; 2) this research contributes to the small pool of studies investigating the effects of within-season variability on recreational site choice decisions and welfare estimates; 3) this thesis contributes to continuing research efforts to address the problem of multicollinearity by testing whether the use of less aggregated data can reduce collinearity levels; 4) this study includes the first investigation of whether collinearity levels typically considered tolerable can affect welfare estimates; 5) this research adds to the limited number of studies testing the stability of welfare estimates over time in recreational demand literature; 6) this research is the first in environmental non-market valuation literature to investigate whether scale heterogeneity across data sets can significantly contribute to differences in welfare estimates; 7) this study contributes to the current small pool of choice experiment studies investigating the effect of respondents' perceptions of status quo conditions on welfare estimates and 8) findings from this thesis add to the limited pool of fresh water non-market valuation data in New Zealand.

1.3 Outline of the thesis

This thesis has eight chapters. In this chapter (Chapter One) the background, research questions, motivation and contributions of the thesis are outlined. In Chapter Two, an investigation of how water quality is measured is carried out. A detailed investigation of water quality in the Rotorua Lakes is also provided.

A review of freshwater values and non-market valuation methods is carried out in Chapter Three. The main objective is to gain an understanding of the different non-market values provided by freshwater bodies and the most appropriate non-market valuation techniques that can be used to assess these values. Methodological issues regarding the use of these techniques and potential gaps in literature are also investigated.

The first research question (Q1) is addressed in Chapter Four. The travel cost RUM is used to assess the effects of water quality on trout anglers. The anglers' WTP for better water quality is assessed. An outline of recreational fishing data and methods is provided.

Research questions Q2 to Q4 are addressed in Chapter Five. In this chapter the effects of accounting for within-season variability in site attributes on welfare estimates is investigated. Specifically, welfare estimates from models accounting for and those ignoring within-season variability in water clarity and weight of fish are compared. An investigation of whether the use of less aggregated data can reduce collinearity levels and whether tolerable levels of collinearity can affect welfare estimates is explored.

In Chapter Six, an investigation of whether recreational fishing values remain stable over time is carried out. This is accomplished by comparing the marginal WTP for lake attributes obtained from two independent fishing choice data sets collected six years apart. The extent to which scale heterogeneity across individuals can contribute to differences in the marginal WTP is assessed.

The last research question (Q7) is addressed in Chapter Seven. The Karapiro catchment choice experiment study is used to assess residents' preferences for better stream quality. The choice experiment survey procedures and description of the study area are outlined. The WTP between respondents using their own perceived quality of streams and those provided with descriptions of the status quo conditions are compared.

A summary of the findings and policy recommendations based on Chapters One to Seven is presented in Chapter Eight.

CHAPTER TWO

WATER QUALITY, MEASUREMENT AND MANAGEMENT

2.0 Introduction

Surface freshwater ecosystems serve a wide range of purposes including supply of potable water for drinking, recreation, habitat and commerce. Over the years these uses have come under threat both locally and internationally, mostly due to declining water quality. Consequently, initiatives to restore and protect water quality have taken central stage in many countries including New Zealand. Effective and efficient implementation of water quality management policies often requires integrated inputs from various stakeholders. Economists play a major role in assessing the costs and benefits of various water pollution control policies. Many of the benefits of good water quality, such as ecological health, cannot be directly assessed in dollar terms in the market. Non-market valuation methods have been developed and have proved to be a very useful tool for assessing the value of environmental resources for which there is no price tag. One of the earliest examples, is the use of the contingent valuation method to assess the environmental damages caused by the Exxon-Valdez oil spill in 1989 in Prince William Sound in Alaska (Carson *et al.*, 1992; Portney, 1994) [6].

The success of non-market valuation exercises requires an understanding of how water quality is measured and subsequently choosing the most appropriate measure that map directly onto individuals' perceptions of water quality. The main objective in this chapter is to explore how water quality is measured internationally and locally. Specifically, a range of water quality measures for possible use in the assessment of non-market values of water quality in the Rotorua Lakes are

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[6] Non-market valuation methods are also important for cost-benefit analysis of new regulations and projects, environmental costing and accounting (Bennett & Blamey, 2001; Navrud & Pruckner, 1997).

investigated. An in-depth outline of the status of water quality in the Rotorua Lakes is also provided.

In the following section an outline of how water quality is defined and measured internationally is provided. This is followed by an investigation of how water quality is measured in New Zealand. A review of water quality in the Rotorua Lakes is investigated in the remainder of the chapter.

2.1 Water quality: Definition and measurements

Water is a multi-attribute commodity, defined in terms of its physical, chemical and biological properties. It is also a multi-product good serving a wide range of purposes. Furthermore, the quality of surface water cannot be viewed as a distinct domain; it requires recognition of the influence of the complex interconnections with ground water and the atmosphere through the hydrological cycle. All these factors contribute to the complexities of water quality analysis. Also, the unique nature of each water body in terms of its physical, chemical and biological make-up adds to the existing complexities. The uniqueness of the physico-chemical and biological composition of water bodies can be attributed to different climatic, geomorphological and geochemical conditions prevailing in the drainage basin and the underlying aquifer. In addition, even within one water body, the quality of water may vary at different locations due to spatial and temporal variations depending upon the hydrodynamic characteristics of that particular water body (Meybeck & Helmer, 1996).

In view of all of these complexities, no single definition can sufficiently describe water quality. At best water quality can be regarded as a term used to describe the physical, chemical, and biological characteristics of water in relation to its suitability for a particular use (Meybeck *et al.*, 1996; USGS, 2008).

Due to the multi-dimensional nature of water, effective water quality measurement requires a collective assessment by experts from various fields, including physical scientists, biologists, hydrologists and social scientists (Bergstrom *et al.*, 2001).

Physical scientists measure the health of a particular water body at a specific site using specific physico-chemical parameters. Notable water quality parameters used in most of these studies include biochemical oxygen demand (BOD), oxygen demand (DO), chemical oxygen demand (COD), total organic carbon (TOC), total oxygen demand (TOD), total solid carbon (TSC), turbidity, temperature, pH, dissolved nutrients and sediments (EPA, 2010; Hayward *et al.*, 2000; Smith *et al.*, 1982; Spulber & Sabbaghi, 1994; Tietenburg, 1998).

On their part, biologists contend that the presence and number of types of fish, insects, algae, plants and other organisms can be used as an indication of the health of a specific water body. According to this group of scientists, the presence of certain types of micro- as well as macro-organisms can be used as a proxy of the physico-chemical state of a particular water body. Indicator micro-organisms are generally used to assess the quality of water for drinking and recreational purposes. Common microbiological indicators used to assess the suitability of water for recreation include cyanobacterial toxins and pathogen indicators such as enterococci (Ashbolt *et al.*, 2001; Plancherel & Cowen, 2004; Tison *et al.*, 2008). Biologists also champion the use of macro-organism indicators as a general measure of surface water quality. It is argued that macro-organisms such as benthic macro-invertebrates can be relied upon to measure the health of water bodies because of their ability to respond to a wide range of stresses, an aspect that cannot be easily discerned through the use of physico-chemical indicators (EPA, 2010; Reice & Wohlenberg, 1993; Rosenberg & Resh, 1993; Wilhm & Dorris, 1968).

Another standard measure of water quality is the use of hydrological properties. It is argued that hydrological conditions such as discharge rate, velocity of flow, turbulence and depth can have a large effect on water quality (Kuusisto, 1996). Therefore, there is a general consensus that a well-balanced assessment of water quality should be based on physico-chemical, biological and hydrological characteristics (Meybeck *et al.*, 1996). In general most countries classify the quality of water using gradations for different end uses based on these water quality indicators. For instance, the US EPA classifies water quality as good (fully supporting), good (threatened), impaired and not attainable. On the other hand, the EU Water Framework Directive defines water quality as being high, good, moderate, poor and bad (Viscusi *et al.*, 2008).

Each designated use tends to have different water quality requirements depending upon the minimum acceptable pollutants (Callan & Thomas, 2007). While the most stringent measures may be applied to drinking water as prescribed by the World Health Organization (WHO, 1993), relatively high water quality may be required for recreational purposes and relatively lower water quality may be required for irrigation and waste disposal.

2.2 Water quality: Measurement and classification in New Zealand

Water quality measurement in New Zealand follows similar international standards to those described in the preceding section. However, the use of physico-chemical and biological indicators is predominant. The trophic level index (TLI) is the main physico-chemical parameter employed to measure the eutrophication status of lakes (Scholes & McIntosh, 2009). The TLI is an aggregate measure of total nitrogen, total phosphorus, chlorophyll a and Secchi disc depth. Lakes are defined according to their eutrophication status ranging from ultra-microtrophic (pristine conditions) to hypertrophic (over-saturated with nutrients) as presented in Table 2.1 below. Other physico-chemical indicators used include dissolved oxygen, temperature, pH and turbidity (Verburg *et al.*, 2010).

Table 2.1: Lake classification based on the trophic level index

Trophic level	Lake type	Perceived lake quality
0.1 – 1.0	Ultra-microtrophic	Excellent
1.1 – 2.0	Microtrophic	
2.1 – 3.0	Oligotrophic	
3.1 – 4.0	Mesotrophic	
4.1 – 5.0	Eutrophic	
5.1 – 6.0	Supertrophic	
6.1 –7.0	Hypertrophic	Very Poor

Source: EBOP (2011)

On the other hand, the Lake Submerged Plant Indicator (LakeSPI) is the main macro-biological measure of water quality used in New Zealand[7]. The LakeSPI Index is an overall measure of the ecological condition of lakes and is constructed based upon the Native Condition and the Invasive Impact Indices defined as follows:

> The Native Condition Index captures the native character of vegetation in a lake bed based on diversity and quality of indigenous plant communities. A higher score means healthier, deeper and diverse beds. Invasive Impact Index captures the invasive character of vegetation in a lake bed on the degree of impact by invasive weed species (Edwards & Clayton, 2009 p. 13).

The ecological conditions of lakes are classified into different gradations based on the LakeSPI Index as depicted in Table 2.2 below.

[7] In addition to plant indicators, aquatic macroinvertebrates are also often used to measure changes in the ecological status of fresh water bodies (Scarsbrook *et al.*, 2000).

Table 2.2: Lake classification based on the Lake SPI Index

LakeSPI	Perceived Ecological Condition
>75%	'Excellent'
>50-75%	'High'
>20-50%	'Moderate'
>0-20%0	'Poor'
0	'Non-vegetated' (defined as having a macrophyte cover of <10%)

Source: Verburg *et al.* (2010 p.4)

Additionally, microbiological indicators including enterococci, *E. coli* and cyanobacterial toxins are used to assess the suitability of freshwaters for contact recreation following the Ministry for the Environment (MFE) and Ministry of Health (MoH) guidelines. Suitability for recreation grades ranging from "very high", "high", "moderate", "low", and "very low" are computed and health warnings are issued whenever acceptable contamination levels are exceeded (MFE, 2002; Northland Regional Council, 2009).

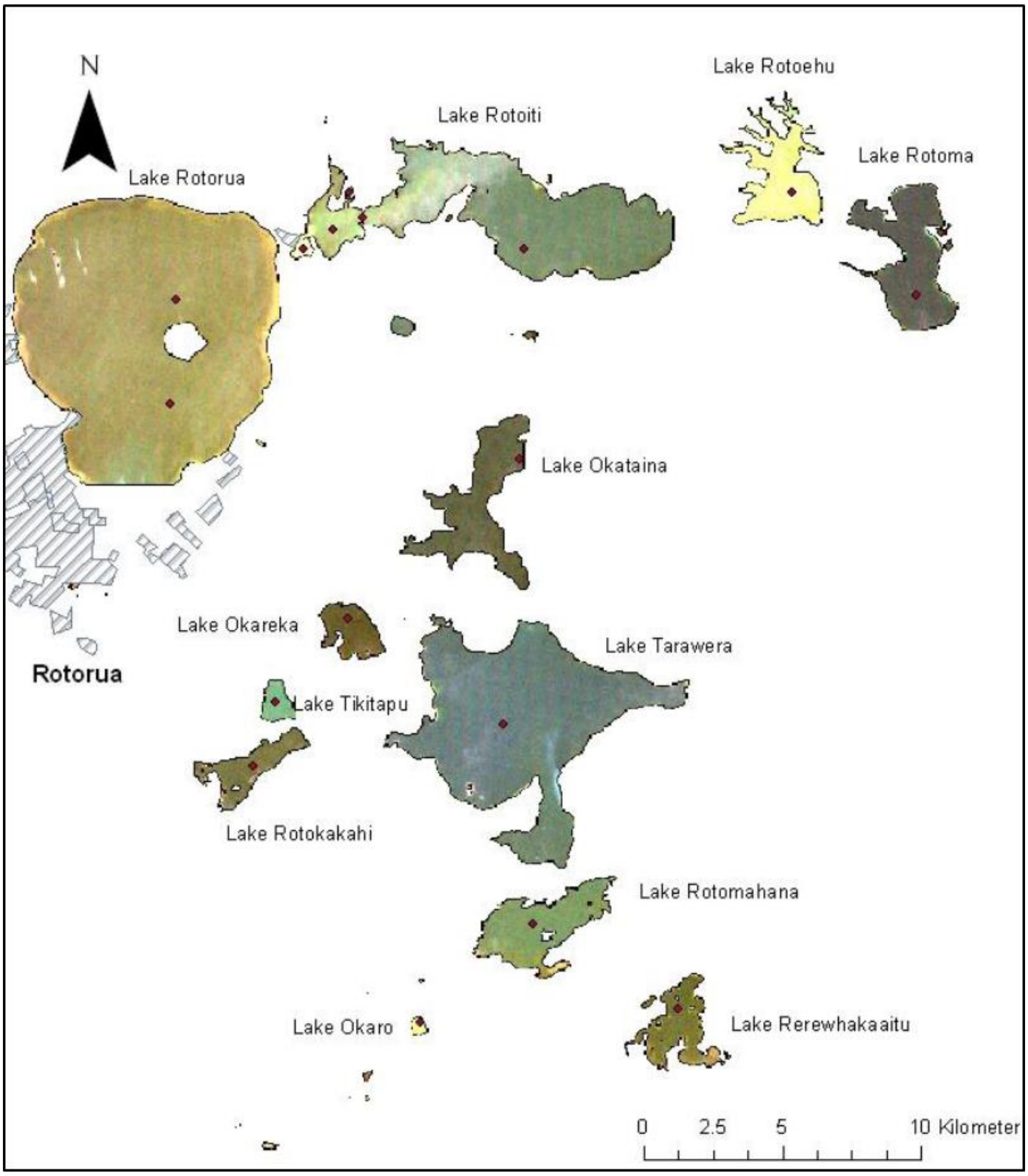
2.3 Water quality in the Rotorua Lakes

The name Rotorua Lakes refers to twelve main lakes all located in the Rotorua District (Figure 2.1)[8]. These lakes, coupled with other attractions in the region including geothermal activity, parks, reserves and Maori culture and history, have made the Rotorua Region one of the most popular tourist destinations in New Zealand for both domestic and international visitors[9].

[8] Lake Rotokakahi is not open to the public, therefore the focus is on the remaining eleven lakes.

[9] The popularity of the Rotorua region as a major tourist destination stems back to the 19th Century and the arrival of early European missionaries, travellers and traders. Describing the stunning beauty of the Rotorua Lake, Colenso (1841, p.34), wrote[9]: "[…] upon gaining the summit of a high hill [...] had a fine prospect of the principal Lake of Rotorua - a fine sheet of water, about six miles in diameter, with a very picturesque island nearly in the midst."

Figure 2.1: The Rotorua Lakes



Source: Allan (2008)

Trout fishing has been one of the main attractions in the region since they were released in 1888 and following their successful acclimatisation to the Rotorua Lakes[10] (Shaw, 1992a). Anglers make a major contribution to the region. Shaw

[10] Currently, management of fish in the region is undertaken by Eastern Region Fish and Game Council. This is one of the twelve Fish and Game Regional Councils within Fish and Game New Zealand (FGNZ) mandated with the responsibility of managing sports fish and game resources in New Zealand under the Conservation Act 1990. Through licence sales the change in fishing

(1992a) estimated that anglers spent a total of $13 million on fishing related expenditures in the Rotorua Lakes during the 1986/87 season. Horgan (2001), estimated the Rotorua Lakes trout fishery value to be in the range of $50 to $70 million representing 30% of the national trout fisheries which was estimated to be between $160 million to $300 million.

While the lakes continue to be vital for recreational fishing and other purposes, the declining water quality in some of the lakes poses a major threat to the preservation of these values[11]. Several of the Rotorua Lakes have experienced a marked decline in water quality over the past 30 years. This is largely attributed to increased levels of nitrogen and phosphorus nutrients which have led to the eutrophication of a number of lakes (Burger *et al.*, 2007; Hamilton, 2003; PCE, 2006). Currently, the trophic status of the lakes ranges from supertrophic to oligotrophic. Lake Okaro is supertrophic. Lakes Rotorua and Rotoehu are eutrophic while Rotoiti, Rotomahana, Rerewhakaaitu and Okareka are mesotrophic. Lakes Tikitapu, Okataina, Tarawera and Rotoma are oligotrophic.

demand in the Rotorua Region over the years is documented. For instance, Shaw (1992a) reported an increase in licence sales from 6,251 in the 1948/1949 to 43,998 in the 1983/1984 fishing season representing an increase of over 600%. By 2001 about 37,000 trout fishing licences were estimated to be sold every year in the district of which 40% were sold to international visitors[10] (Horgan, 2001).

[11] The impacts of falling water quality are also being documented. Recently, Pitkethley (2008), the manager of the Eastern Region Fish and Game Council reported a decline in short term fishing licence sales by $100,000 in the summer of 2003. At the same time angler usage was reported to have dropped by 65% in Lake Rotoiti and algal blooms were cited as the major contributor. The decline in angler usage for Lakes Rotoiti and Rotorua over the past decade is also reported in the National Angling Survey by Unwin (2009).

The status of water quality in these lakes is explored in more detail in Table 2.3 to Table 2.5 for the period from the 1990s up to 2009[12]. This period covers the study years for the fishing choice destination data used in this thesis. Water quality is explored in terms of the trophic level index (TLI), chlorophyll a (CHLA), Secchi depth (SD), total nitrogen (TN) and total phosphorus concentrations (TP).

Table 2.3: Water quality in the Rotorua Lakes during the period 1990 to 2009 for supertrophic and eutrophic Lakes

	Lake Name and Trophic Status														
	Okaro (Supertrophic)					Rotoehu (Eutrophic)					Rotorua (Eutrophic)				
Period	**CHLA (mg/m3)**	**SD (m)**	**TP (mgP/m3)**	**TN (mg/m3)**	**TLI**	**CHLA (mg/m3)**	**SD (m)**	**TP (mgP/m3)**	**TN (mg/m3)**	**TLI**	**CHLA (mg/m3)**	**SD (m)**	**TP (mgP/m3)**	**TN (mg/m3)**	**TLI**
1990-1991						2.6	4.52	31.28	285.44	3.84	20.6	2.35	37.62	379.08	4.76
1991-1992	29.55	1.38	84.33	1177.83	5.65	1.59	5.28	28.53	324.4	3.67	7.47	2.28	42.55	423.48	4.57
1992-1993	27.75	2.08	91.67	1015.67	5.49	35.45	2.67	48.19	801.75	5.2	15.82	2.6	52.57	371.95	4.76
1993-1994	12.35	1.93	101.1	1259.33	5.39	10.28	2.63	46.44	502.5	4.7	7	2.14	92.94	457.1	4.84
1994-1995	12.15	1.87	138	1193.71	5.47	11.67	1.76	47.06	443.76	4.81	8.73	2.93	54	421.76	4.61
1995-1996	22.85	1.86	165.83	1271.83	5.73	7.9	2.81	44.82	405.45	4.52	7.77	3.18	28.31	344.23	4.28
1996-1997	42.74	1.85	146.78	1492.44	5.92	11.9	3.45	35.63	434.97	4.52	14.55	3.16	30.78	421.9	4.55
1997-1998	81.68	1.72	119.33	1246.33	5.99	13.54	2.81	33.85	529.85	4.67	21.69	2.9	39.68	490.95	4.82
1998-1999	55.9	1.94	126	1754	5.98	10.33	2.57	32.5	458.5	4.56	5.77	3.21	30.13	401.37	4.27
1999-2000						13.55	2.45	30.1	461.1	4.63	10.81	2.48	31.43	536	4.63
2000-2001	17.1	2.02	99.17	1013.54	5.39	12.65	2.25	37.09	486.43	4.72	29.17	2.56	47.51	459.24	4.97
2001-2002						14.53	2.12	30.27	459.9	4.69	14.51	2.71	40.35	386.26	4.65
2002-2003	26.54	1.85	107.94	936.54	5.53	13.38	2.52	26.5	438.12	4.56	28.13	2.03	41.96	447.46	4.98
2003-2004	19.73	2.38	103.66	984.78	5.38	10.07	2.78	32.63	382.76	4.47	26.36	2.17	46.57	531.98	5.03
2004-2005	77.18	1.48	92.76	1266.94	5.95	13.27	2.99	39.83	426.34	4.63	19.93	2.56	35.7	452.92	4.77
2005-2006	17.05	2.74	84.67	986.73	5.24	7.97	2.8	33	433.43	4.45	21.82	2.6	45.35	464.23	4.87
2006-2007	19.97	2.42	75.33	975.78	5.28	11.05	2.78	54.68	418.16	4.69	23.24	2.52	32.55	481.36	4.81
2007-2008	27.28	2.37	62.15	1034.78	5.33	11.75	2.56	30.25	349.35	4.49	16.06	2.61	31.11	483.03	4.68
2008-2009	24.46	2.6	54.61	1256.07	5.29	10.21	3.6	45.67	275.87	4.4	19	2.57	34.52	407.74	4.71
Average	**32.14**	**2.03**	**103.33**	**1179.14**	**5.56**	**11.77**	**2.91**	**37.28**	**437.79**	**4.54**	**16.76**	**2.61**	**41.87**	**440.11**	**4.71**

Source: Scholes (2009 p.71-81)

[12] Blanks imply that water quality monitoring was not done in that year or for a particular water quality indicator.

Table 2.4: Water quality in the Rotorua Lakes during the period 1990 to 2009 for mesotrophic lakes

	Lake Name																			
	Rotoiti					Okareka					Rotomahana					Rerewhakaaitu				
Period	**CHLA (mg/m3)**	**SD (m)**	**TP (mgP/m3)**	**TN (mg/m3)**	**TLI**	**CHLA (mg/m3)**	**SD (m)**	**TP (mgP/m3)**	**TN (mg/m3)**	**TLI**	**CHLA (mg/m3)**	**SD (m)**	**TP (mgP/m3)**	**TN (mg/m3)**	**TLI**	**CHLA (mg/m3)**	**SD (m)**	**TP (mgP/m3)**	**TN (mg/m3)**	**TLI**
1990-1991																3.9	4.95	9.83	321.86	3.6
1991-1992											7.95	4.36	44.17		4.43	3.14	7.34	8	356.96	3.37
1992-1993	7.58	5.5	20.43	267.1	3.91	3.16	10.56	5.86	231.75	3	3.8	4.95	34.75	251.33	3.91	2.74	7.41	5.47	310.22	3.17
1993-1994	4.15	5.07	21.33	273.7	3.8	2.44	9.97	4.91	226.15	2.89	2.9	4.46	40.56	228.42	3.88	2.2	6.92	5.4	334.14	3.15
1994-1995	4.48	5.52	19.97	253.27	3.74	3.88	7.79	6.23	220.63	3.17	3.72	4.35	29.43	206	3.83	2.42	6.37	6.5	329.85	3.26
1995-1996	5.28	5.74	23.4	265.23	3.84	3.1	8.04	5.92	229.5	3.1	4.63	5.07	16	237	3.69	7.8	3.78	8.86	398.14	3.91
1996-1997	5.13	6.33	17.7	244.67	3.69	9.7	7.36	7.79	260.14	3.57	5.94	4.28	31	240	4.03	8.43	4.34	10.53	446.27	3.98
1997-1998	6.09	5.8	22.28	288.33	3.89	5.4	9.06	6.25	245.25	3.25	5.63	4.84	13	239.33	3.7	17.18	4.45	9.67	547.67	4.21
1998-1999	6.49	4.64	27	285.07	4.04	5.23	8.55	5.75	236.4	3.22	7.38	3.1	6	195	3.6	7.38	2.78	6	499.5	3.94
1999-2000	8.01	5.53	22.56	252.56	3.94	5.12	7.92	7	209.45	3.26										
2000-2001						3.62	8.7	6.2	216.86	3.11	4.21	4.36	20.15	219.48	3.76	2.97	5.62	5.74	359.46	3.34
2001-2002	7.3	4.44	23.06	249.15	3.99	4.03	8.31	5.41	210.41	3.1						3.33	4.88	5.65	348.07	3.41
2002-2003	17.63	3.32	31.05	354.14	4.53	3.9	8.03	5.75	183.45	3.08	4.65	5.49	13.69	181.38	3.53	2.93	6.68	6.93	333.33	3.32
2003-2004	12.03	4.3	39.83	447.77	4.5	5.21	6.45	6.72	229.03	3.36	4.62	5.29	37.12	226.29	3.93	2.35	8.25	9.61	376.59	3.33
2004-2005	13.39	5.05	34.51	374.08	4.38	2.93	7.42	7.75	197.55	3.15	4.38	5	30.75	198.43	3.83	3.42	5.88	7.5	338.63	3.43
2005-2006	7.15	5.21	33.12	307.06	4.12	4.51	7.59	10.7	215.21	3.39	3.84	4.94	37.09	202.18	3.86	2.82	7.01	8.78	389.92	3.42
2006-2007	5.7	5.84	24.67	289.37	3.9	3.12	7.72	9	225.3	3.24	3.76	5.7	47.64	235.69	3.94	2.87	5.77	10.98	469.68	3.62
2007-2008	7.35	5.36	20.41	277.27	3.93	4.32	7.88	6.61	219.43	3.22	4.31	5.28	48.33	249.23	4.02	3.87	4.93	8.54	483.54	3.68
2008-2009	7.67	5.5	21.77	209.82	3.86	4.62	8.41	10.14	199.32	3.32	5.11	5.12	43.05	237.13	4.03	5.15	4.29	12.17	429.7	3.88
Average	**7.84**	**5.2**	**25.19**	**289.91**	**4**	**4.37**	**8.22**	**6.94**	**220.93**	**3.2**	4.8	4.79	**30.79**	**223.13**	**3.86**	**4.72**	**5.65**	**8.12**	**392.97**	**3.56**

Source: Scholes (2009 p.71-81)

Table 2.5: Water quality in the Rotorua Lakes during the period 1992 to 2009 for oligotrophic lakes

	Lake Name																			
	Okaitana					Tikitapu					Tarawera					Rotoma				
Period	**CHLA (mg/m3)**	**SD (m)**	**TP (mgP/m3)**	**TN (mg/m3)**	**TLI**	**CHLA (mg/m3)**	**SD (m)**	**TP (mgP/m3)**	**TN (mg/m3)**	**TLI**	**CHLA (mg/m3)**	**SD (m)**	**TP (mgP/m3)**	**TN (mg/m3)**	**TLI**	**CHLA (mg/m3)**	**SD (m)**	**TP (mgP/m3)**	**TN (mg/m3)**	**TLI**
1992-1993	1.92	11.1	9.92	139.44	2.85	2.93	7.37	4.21	203.74	2.97						2.52	13.63	3.45	176.89	2.58
1993-1994	1.56	9.57	5.56	117.91	2.61	1.46	6.56	3.07	180.89	2.67						0.98	11.99	3	117.5	2.2
1994-1995	1.23	11.37	6	124	2.52	1.56	6.7	2.42	185.27	2.62	1.23	8.05	7	116.05	2.67	0.93	12.71	3.44	139.67	2.26
1995-1996	1.71	10.44	5	112.17	2.55	1.23	7.27	3.75	179.69	2.65	1.17	7.93	5.6	92.7	2.52	0.85	15.61	2.83	139.83	2.08
1996-1997												8.93			2.8		14			2.09
1997-1998							7.86			2.98										
1998-1999																				
1999-2000	2.16	11.21	5.14	121.5	2.62	1.52	6.78	4.25	190.71	2.79	1.89	7.98	7.11	113.79	2.79					
2000-2001																1.96	9.81	3.27	137.91	2.54
2001-2002	3.34	8.11	6.4	143.62	2.99	2.89	4.44	4.59	209.9	3.17	1.81	7.91	7.5	112.32	2.8					
2002-2003	3.11	8.2	8.12	111.28	2.95	2.69	5.04	4.47	221.64	3.12	1.81	7.08	8.13	130.12	2.91	1.47	9.36	4	129.84	2.53
2003-2004	2.76	11.08	10.7	142.13	2.98	1.78	7.24	8.74	281.31	3.17	1.42	7.52	9.45	108.21	2.81	1.37	14.16	4.91	147.07	2.45
2004-2005	2.12	10.47	8.7	144.95	2.87	2.12	6.43	7.3	210.79	3.11	1.21	9.4	10.59	112.69	2.73	1.33	13.01	4.85	150.24	2.48
2005-2006	1.89	11.53	9.16	150.48	2.83	1.54	6.85	6.29	214.47	2.96	1.76	9	12.36	109.2	2.89	1.27	14.3	6.33	164.55	2.54
2006-2007	1.65	11.42	7	134.53	2.67	1.94	6.51	4.5	235.24	2.96	1.57	9.47	14.05	142.98	2.97	1.23	11.23	4.1	162.93	2.49
2007-2008	2.35	10.2	8.75	163.9	2.95	1.74	6.84	5.5	223.05	2.96	1.61	9.18	10.59	154.23	2.92	1.21	13.05	5.72	195.75	2.59
2008-2009	2.46	10.84	8.81	119.77	2.84	2.04	6.33	8.25	196.78	3.12	1.56	8.94	14.19	102.22	2.88	1.33	14.2	5.55	122.67	2.42
Average	**2.17**	**10.43**	**7.64**	**132.74**	**2.79**	**1.96**	**6.59**	**5.18**	**210.27**	**2.94**	**1.55**	**8.45**	**9.69**	**117.68**	**2.81**	**1.37**	**12.85**	**4.29**	**148.74**	**2.42**

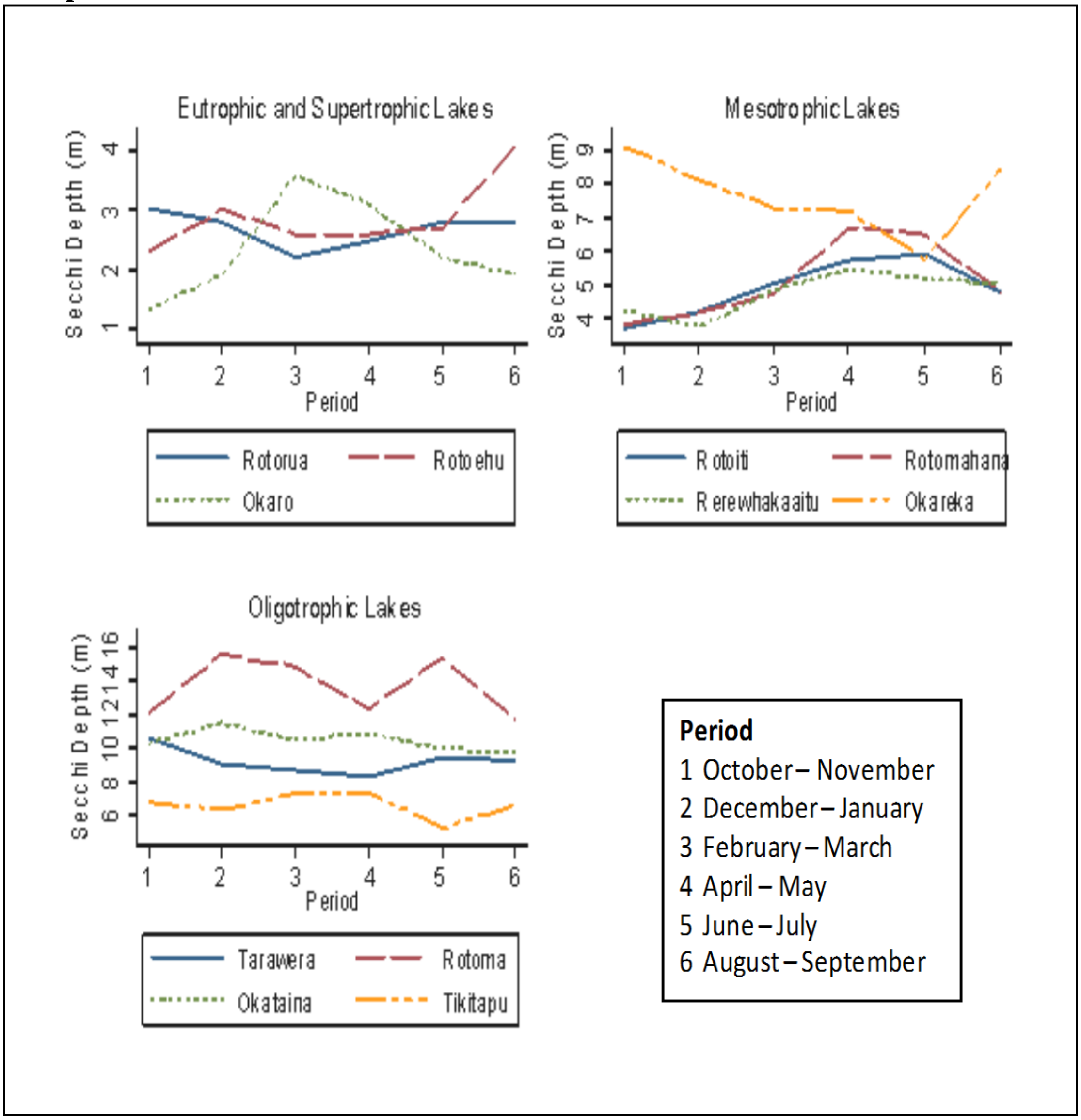
Source: Scholes (2009 p.71-81)

The average TLI during the entire period ranged from 2.42 (Lake Rotoma) to 5.56 (Lake Okaro). Notably, the lakes displayed a wide variability in SD ranging from an average of 2.03 m (Lake Okaro) to 12.85 m (Lake Rotoma) during this period. Lake Okaro also recorded the highest average total nutrient loads in terms of TP, TN, and CHLA concentrations in the range of 103.33 mgP/m^3, 1179.14 mg/m^3 and 32.14 mg/m^3 respectively. In contrast, Lake Rotoma registered the lowest average nutrient loads for TP and CHLA of 4.29 mgP/m^3 and 1.37 mg/m^3 respectively. On the other hand, Lake Tarawera had the lowest average TN nutrient loads of 117.68 mg/m^3. In general, the lakes displayed a wide range of variability in these water quality indicators during this period.

In addition, the Rotorua Lakes are characterized by variations in water quality across the year, with warmer months showing poorer water quality, and some lakes experiencing more algal blooms in warmer months[13] (Allan *et al.*, 2007). The seasonal variability in water quality is explored in Figure 2.2 to Figure 2.4 using the SD, TN, TP and CHLA concentrations for the period October 2007 to September 2008. This period is chosen because it corresponds to the anglers' fishing choice data used in this thesis. The seasonal variability in water quality might impact upon anglers' timing of when to go fishing as well as their choice of fishing destinations and hence is worth exploring in more detail.

[13] The lakes are also characterized by spatial variability in water quality within the same lake, an issue not investigated in this thesis.

Figure 2.2: Seasonal variations in Secchi depth over the period October 2007 to September 2008



Source: EBOP (unpublished data)

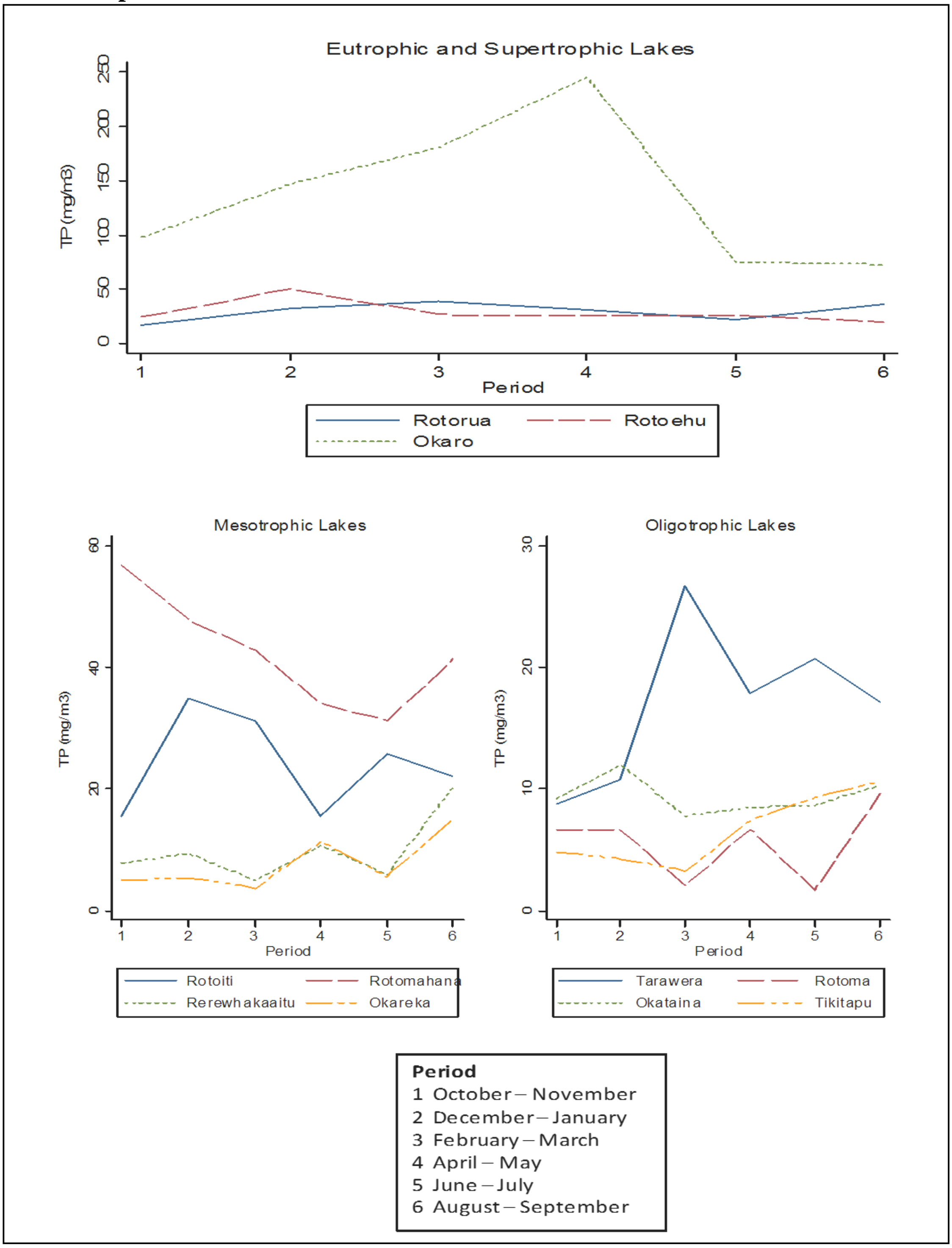
Figure 2.2 presents the seasonal variability in SD measured in metres (m). The SD is a measure of water clarity. Generally, the lakes displayed different patterns of variability in water clarity during this period. For lakes in the eutrophic category, the largest variability in water clarity is observed for Lake Okaro with a minimum of about 1.2 m in period 1 and a maximum of around 3.6 m in period 3, representing a change in water clarity of about 2.4 m. This is followed by Lake Rotoehu with a minimum SD reading of about 2 m in period 1 and a maximum of around 4 m in period 6, representing a change in water clarity of about 2 m between the two periods. For Lake Rotorua, a maximum change in SD of about 1 m is observed between period 1 and 3.

For mesotrophic lakes, the largest variability in SD is observed for Lakes Okareka and Rotomahana with a decline from 9.0 m in period 1 to 6.0 m in period 5 and an increase in SD from about 4 m in period 1 to around 7 m in period 4, respectively. Lakes Rotoiti and Rerewhakaaitu registered a maximum change in SD of about 2 m between periods 1 and 4.

In the case of the oligotrophic lakes, Lake Rotoma registered a minimum of about 12.0 m in periods 1 and 4 and a maximum of around 15.6 m in periods 2 and 5. Lake Tarawera showed a decline in SD from about 11.0 m in period 1 to about 8.0 m in period 4. A general decline in water clarity is observed for Lakes Rotoma and Tarawera between periods 2 and 4. Lake Tikitapu registered its highest SD reading of about 7 m in periods 3 and 4 and the lowest reading of about 5 m in period 5. On the other hand, Lake Okataina showed an SD reading of about 12 m in period 2 and the lowest reading of about 10 m in periods 1, 5 and 6.

The seasonal variability in TP measured in mg/m^3 for this period is presented in Figure 2.3. Lake Okaro had the highest concentrations of TP nutrients ranging from as high as 250 mg/m^3 in period 4 to as low as 75 mg/m^3 in periods 5 and 6. For mesotrophic lakes, the largest concentrations of TP nutrients were observed for Lake Rotomahana, followed by Lake Rotoiti. These lakes also displayed a marked variability in nutrient concentrations during this period, with the former experiencing a decline from around 60 mg/m^3 in period 1 to about 30 mg/m^3 in period 5. In the case of oligotrophic lakes, Lake Tarawera had the largest TP nutrient level and also displayed the largest variability in these nutrients ranging from as low as 9 mg/m^3 in period 1 to as high as 27 mg/m^3 in period 3. In general, all the lakes in this category displayed a wide range of variability in TP loads across the periods.

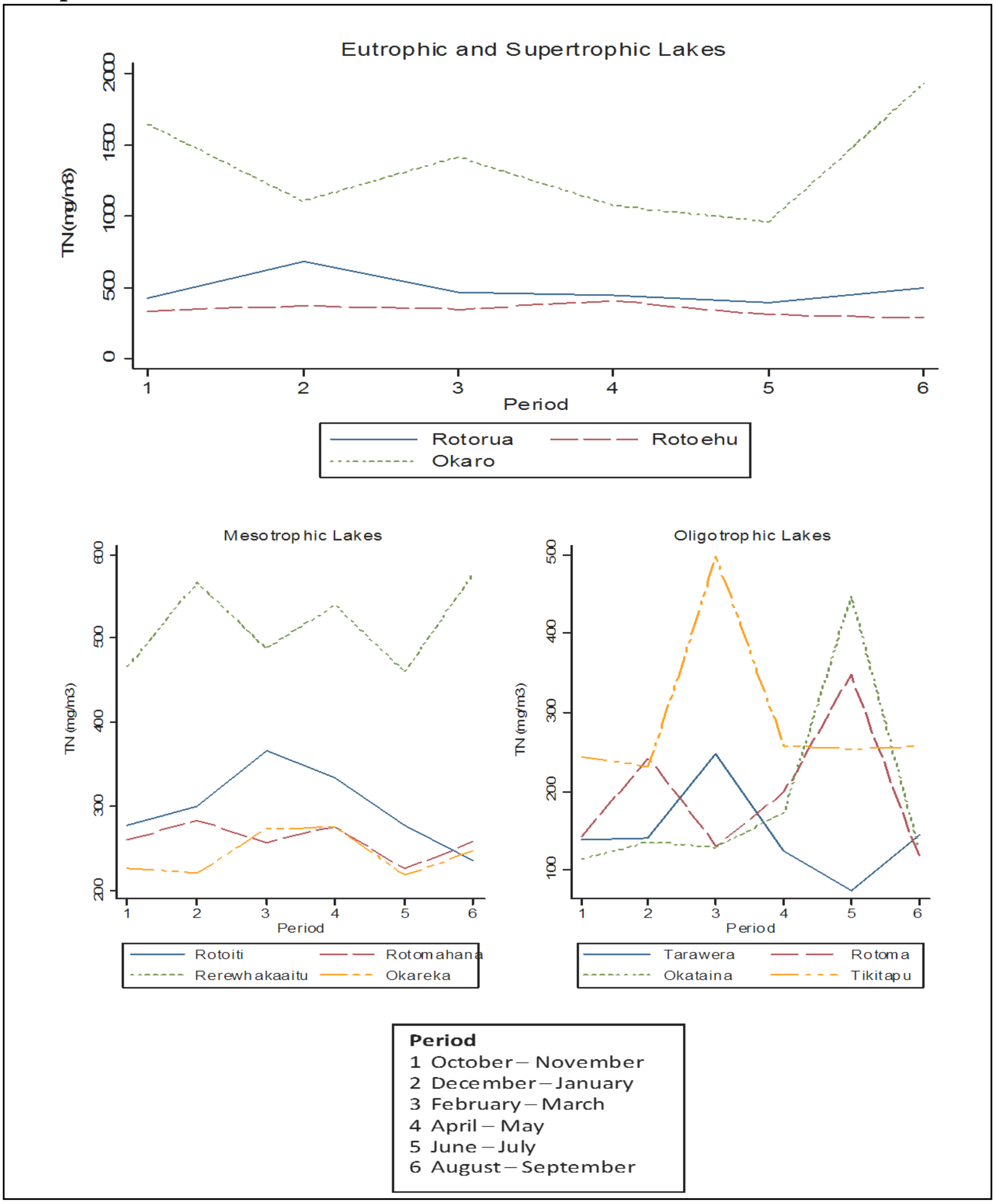
Figure 2.3: Seasonal variations in total phosphorus over the period October 2007 to September 2008.



Source: EBOP (unpublished data)

Figure 2.4 shows the distribution of total nitrogen loads during this period. For lakes in the eutrophic and supertrophic categories, Lake Okaro had the highest nitrogen nutrient loads with a minimum of about 1000 mg/m^3 in periods 4 and 5 and a maximum of about 2000 mg/m^3 in period 6.

Figure 2.4: Seasonal variations in total nitrogen over the period October 2007 to September 2008.

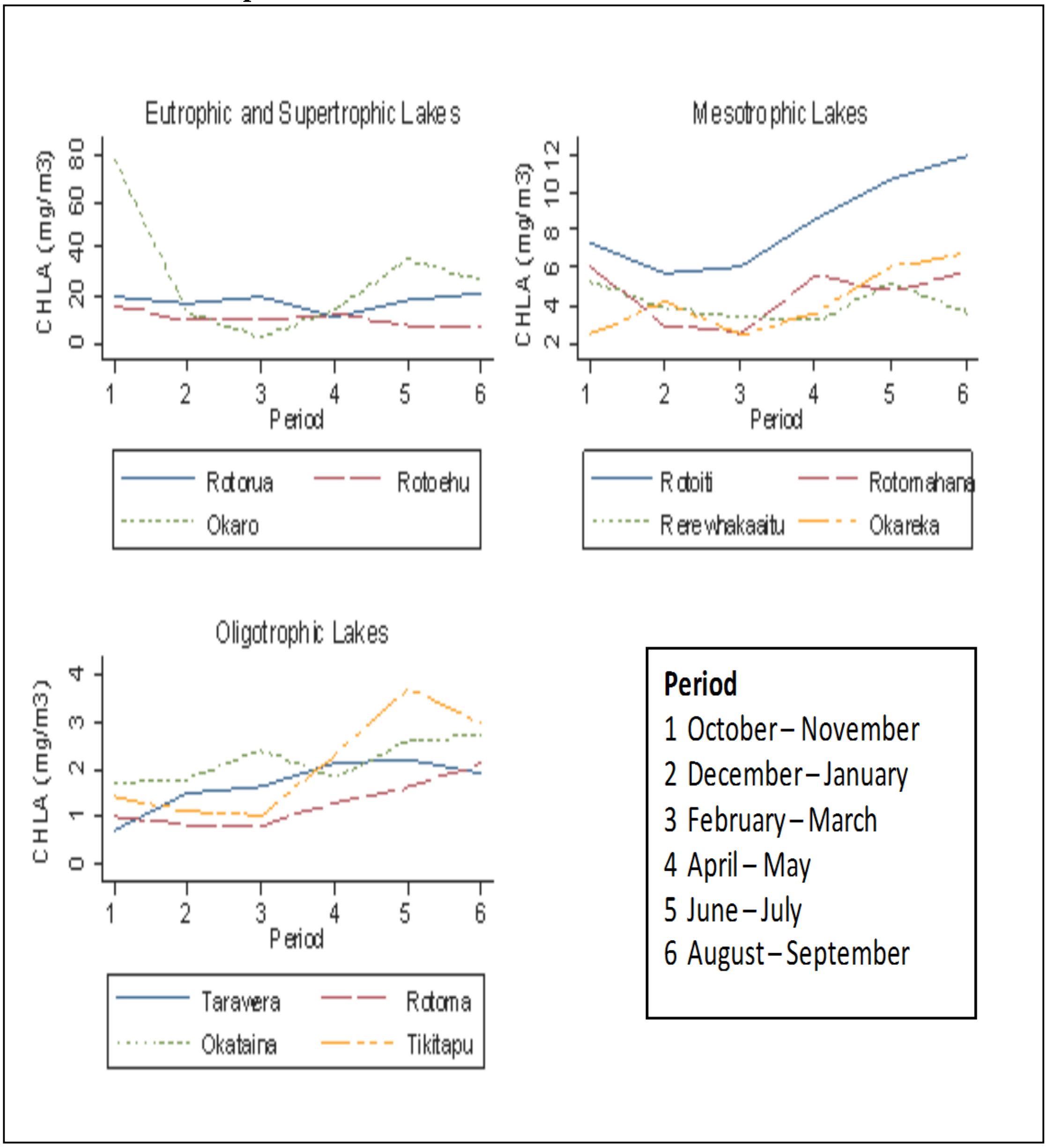


Source: EBOP (unpublished data)

For mesotrophic lakes, Rerewhakaaitu had the highest nitrogen nutrient loads ranging from as low as around 470 mg/m^3 in periods 1 and 5, to as high as about 580 mg/m^3 in period 6. However, Lake Rotoiti displayed the largest variability in nitrogen loads with a maximum of about 370 mg/m^3 in period 3 and a minimum of around 240 mg/m^3 in period 6, representing a change of about 130 mg/m^3. Generally, Lakes Rotomahana and Okareka experienced the least variability in nitrogen loads during this period.

In the case of oligotrophic lakes, Lake Okataina recorded the highest variability in nitrogen nutrient loads, ranging from about 110 mg/m^3 to around 450 mg/m^3 in periods 1 and 5 respectively. High variability in nitrogen nutrient loads was also observed for Lake Tikitapu with a minimum of about 230 mg/m^3 in period 2, rising to a maximum of about 500 mg/m^3 in period 3. This was followed by Lake Rotoma with a minimum of about 130 mg/m^3 in periods 1 and 3 and a maximum of around 350 mg/m^3 in period 5. Overall, Lakes Tikitapu and Tarawera experienced the highest nitrogen loads in period 3 while Lakes Okataina and Rotoma registered the highest nutrient loads in period 5.

Figure 2.5: Seasonal variations in chlorophyll a concentrations over the period October 2007 to September 2008.



Source: EBOP (unpublished data)

The distribution of CHLA concentrations during this period is presented in Figure 2.5. For lakes in the eutrophic and supertrophic category, Lake Okaro registered the largest variability in CHLA concentrations, ranging from as high as 80 mg/m^3 in period 1 to about 2 mg/m^3 in period 3. On the other hand, Lakes Rotorua and Rotoehu recorded relatively smaller fluctuations in CHLA, oscillating within the 0-20 mg/m^3 band across the entire period. For mesotrophic lakes, Lake Rotoiti registered the highest CHLA loads and also the highest variability with a minimum of about 6 mg/m^3 in period 2 and a maximum of about 12 mg/m^3 in period 6. In the oligotrophic category, Lake Tikitapu registered the highest variability in CHLA with a minimum of around 1 mg/m^3 and a maximum of about 4 mg/m^3 in periods 3 and 5, respectively.

CHLA is used as an indicator of the amount of algae in a lake. Blooms are said to occur whenever the nitrogen to phosphorus ratio (N:P ratio) falls below the 22:1 threshold (MFE, 2011). Consequently, some lakes such as Okaro, Rotoehu and Rotoiti usually experience multiple blooms in a year once the N:P ratio falls under this threshold. Furthermore, because of the spatial variability in water quality within the same lake, algal blooms may affect the whole or just part of the lake. Consequently, the lakes have been monitored for the presence of cyanobacterial blooms since 1997 when the problem of algal blooms became apparent (Scholes, 2009). In the past decade health warnings have been issued with respect to cyano-bacterial blooms in some lakes or just part of some lakes. A health warning is issued when total cyanobacterial counts reach or exceed a threshold for recreational contact of 15,000 cell/ml at a site (Scholes & Bloxham, 2005). For instance, in the summers of 2003 and 2004, Lake Rotoiti was closed due to toxic algal blooms. Other lakes that have been seriously affected by algal blooms leading to health warnings include Okaro, Rotoehu, Rotorua and, to a lesser extent, Tarawera (MFE, 2011).

In addition to the declining water quality, the general ecological health of the lakes has deteriorated since the 1960s. This is attributed to the introduction of a wide range of invasive plant species since the 1930s (Coffey & Clayton, 1988). In recent decades, the introduction of egeria and hornwort in most of the lakes has posed a major challenge (Edwards & Clayton, 2009). Regular monitoring of the ecological health of lakes is carried out using the LakeSPI Index[14]. Presently, the ecological health of the Rotorua Lakes ranges from poor to high. Lake Rotomahana is ranked

[14] Additional ecological indicators used include Kakahi (freshwater mussels) and Koura (freshwater cryfish) (Edwards & Clayton, 2009).

the best while Lake Rotoehu is ranked the poorest with LakeSPI indices of 63% and 18% respectively, as shown in Table 2.6.

Table 2.6: Ecological health of the Rotorua Lakes based on LakeSPI

LAKE	LakeSPI Index (%)	Native Condition Index (%)	Invasive Impact Index (%)	Overall Condition
Rotomahana	63	61	30	High
Rotoma	47	53	56	
Okataina	45	47	60	
Rerewhakaaitu	41	52	64	
Okareka	34	39	76	Moderate
Tikitapu	32	28	63	
Rotorua	27	31	78	
Tarawera	22	27	92	
Okaro	21	13	77	
Rotoiti	21	29	89	Poor
Rotoehu	18	26	85	

Source: Edwards & Clayton (2009 p. 10)

2.4 Sources of water pollution

Studies indicate that point and non-point sources and internal loads from bottom sediments are the major sources of phosphorus and nitrogen nutrients. Excessive nutrients from septic tanks are considered to be the main point source of phosphorus and nitrogen nutrients to the lakes. For instance, the decline in water quality in Lake Rotorua is largely attributed to the direct input of effluents from the waste water treatment plant in the 1980s prior to the diversion of sewage inflows to land-based treatment in 1991 (Rutherford *et al.*, 1996).

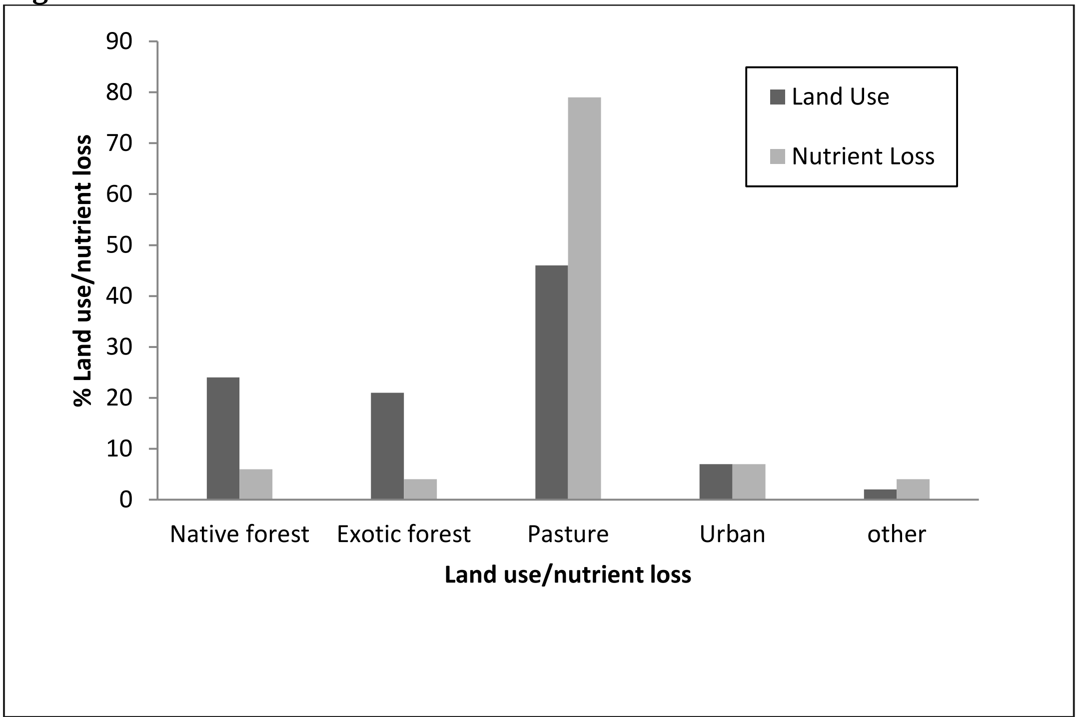
Non-point sources of nutrients in the lakes are mainly attributed to agricultural production in the catchments and to a lesser extent storm water, geothermal inputs, rainfall and erosion (PCE, 2006). According to Chapman (1970), up until the 1900s most of the Rotorua Lakes catchment was covered in dense forests consisting of native trees and manuka scrub. Large-scale sheep and dairy farming around the lakes began in the late 1940s and 1950s (Edwards & Clayton, 2009). Currently, a considerable proportion of land in the catchments for a number of lakes constitutes agricultural developments. For instance, 90% and 69% of the catchments for Lakes Okaro and Rerewhakaaitu respectively are devoted to agriculture. Lakes Tikitapu and Okataina have the least agricultural development, with only 2% and 8% respectively of the land in the catchments devoted to agriculture (LakesWater Quality Society, 2011). Appendix 1 shows the land use in the catchments for the eleven lakes under study.

The link between agricultural development and lake eutrophication has been demonstrated both internationally and locally. For instance, Moss (1998) highlighted a number of channels through which agricultural developments may enhance the export of nutrients from farms into water bodies including increased fertilizer usage. This is also confirmed in the recently released study by the Millennium Ecosystem Assessment (2005) as outlined in the introductory chapter.

Locally, a study by Mitchell (1988), found that out of 17 lakes in which agricultural land constituted more than 40% of the catchment area, 14 were eutrophic. Current water quality conditions in Lakes Okaro and Rotorua offer some evidence in support of the above studies. Lake Okaro is supertrophic while Lake Rotorua is eutrophic with 90% and 46% of the land in the catchment devoted to agriculture respectively. However, Hamilton (2003) points out that the link between nutrient loads and lake eutrophication is also dependent on lake depth. Shallow lakes with high nutrient loads are likely to be more eutrophic than deeper ones. This may explain why lakes such as Rerewhakaaitu, with 69% of its catchment area devoted to agriculture, have relatively better water quality compared to that of Lake Rotorua with pasture land covering only 46 % of the catchment.

Furthermore, the contributions of agriculture and other land uses in the catchments to nutrient losses for the Rotorua Lakes have been documented. The link between land use and nutrient losses to the lakes is demonstrated for Lake Rotorua in Figure 2.6. The highest nutrient losses, in the region of about 79%, emanate from pasture land, while the other land uses contribute minimal amounts of nutrients, which are all below 10%.

Figure 2.6: Land use and nutrient loss in Lake Rotorua catchment area



Source: LakesWater Quality Society (2011)

Internal nutrient loads are another contributor to declining water quality in the Rotorua Lakes. The mechanism of possible lake eutrophication due to internally regenerated nutrients is explained by Environment Bay of Plenty (EBOP), (2004 p.16) as follows:

> When the water is well oxygenated there is a net loss of nutrients to the [lakebed] sediment. When lakes stratify, dead algal cells and other organic material falling into the bottom waters depletes the oxygen due to the decomposition process. No replenishment of oxygen is possible from the atmosphere. As the bottom waters run out of oxygen the chemistry of the sediment surface is changed and nitrogen and

phosphorus are released from the sediments into the water. The nutrients are trapped in the bottom water until the lake mixes vertically. With a flush of nutrients algal production is enhanced if other environmental or climatic factors favour this after mixing.

2.4 Water quality management and restoration policies

Mandated under the Resource Management Act 1991, regional councils are charged with the responsibility of managing water and other natural resources with the Ministry for the Environment (MFE) as an overseer. The management of the Rotorua Lakes is jointly undertaken by the EBOP Regional Council and the Rotorua District Council (RDC), with the former providing overall leadership and co-ordination[15]. Water quality monitoring in the Rotorua Lakes began in the 1960s and EBOP assumed this responsibility in the 1990s. This is carried out under a programme for monitoring the general state of the environment called the Natural Environment Regional Monitoring Network (NERMN) (Scholes, 2010).

In the case of the Rotorua Lakes, initiatives to protect and restore water quality are being implemented under a project called the Rotorua Lakes Protection and Restoration Action Programme. This is coordinated by representatives from EBOP, Rotorua District Council, and Te Arawa Lakes Trust (PCE, 2006). Each lake has an objective TLI determined based upon past water quality[16] as required under objective 10 of the Regional Land and Water Plan (RWLP). The objective TLI and the corresponding yearly TLI for each lake from the 1990s up to 2011 are shown in Figure 2.7 to Figure 2.9 below.

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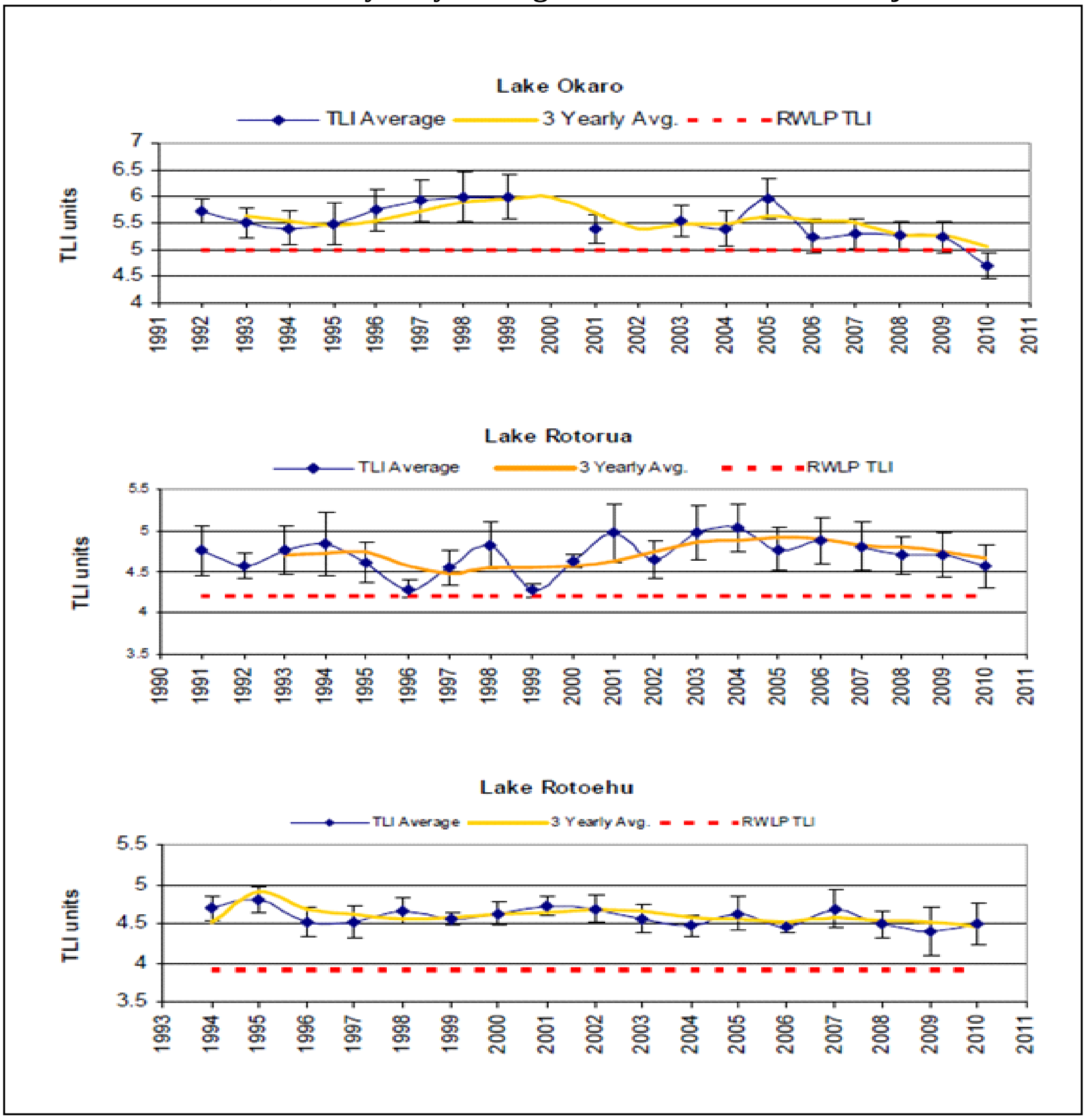
[15] Other organizations involved in the management of different aspects of the lakes include the Ministry for the Environment, Te Arawa Lakes Trust, Fish and Game (Eastern Region) New Zealand, Department of Conservation, Lake Okareka Ratepayers and Residents Association, University of Waikato Centre for Biodiversity and Ecology Research: Rotorua Lakes Database and Lake Ecosystem Restoration New Zealand (EBOP, 2011).

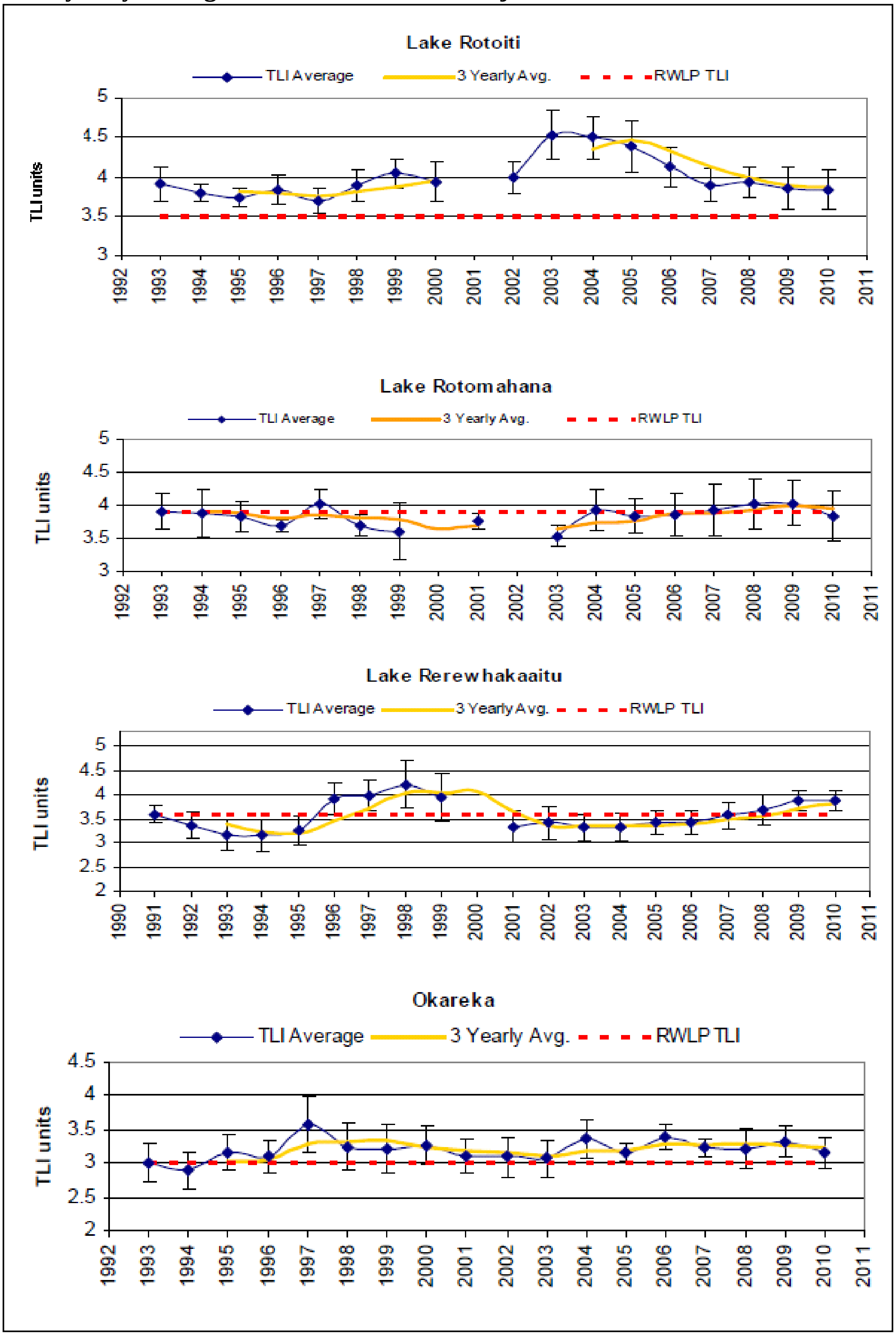
[16] Nutrient targets are also calculated for each lake, as well as policies and timelines for reaching those targets.

Figure 2.7: Supertrophic and eutrophic lakes annual average TLI with standard error bars, three yearly average TLI and RWLP TLI objectives.



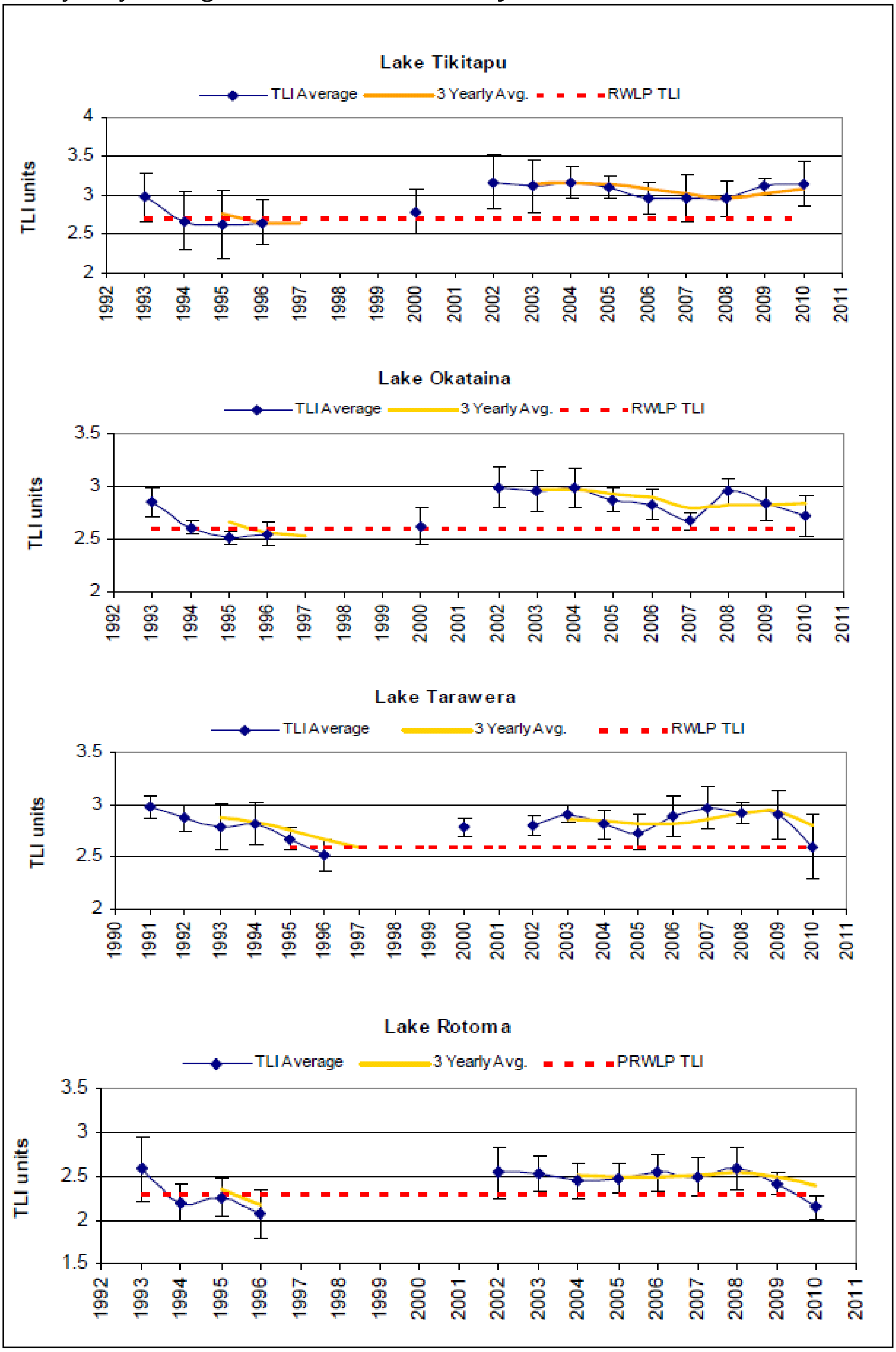
Source: Scholes (2010 p. 4-6)

Figure 2.8: Mesotrophic lakes annual average TLI with standard error bars, three yearly average TLI and RWLP TLI objectives



Source: Scholes (2010 p. 4-6)

Figure 2.9: Oligotrophic lakes annual average TLI with standard error bars, three yearly average TLI and RWLP TLI objectives



Source: Scholes (2010 p. 4-6)

For mesotrophic lakes, Lake Rotoiti registered the largest disparity between the TLI and the objective RWLP TLI. The highest disparities are observed between 2002 and 2005 when the lake experienced occasional algal blooms, but since then there has been some improvement in water quality. Lakes Rotomahana and Rerewhakaaitu remained relatively stable, with the TLI almost at par with the objective RWLP TLI and in some years falling below it, while the TLI for Lake Okareka remained slightly higher than the objective RWLP TLI.

In oligotrophic lakes the TLIs for all the lakes have generally remained higher than the objective RWLP TLI from 2002 to 2010 with the exception of Lakes Tarawera and Rotoma, which experienced a decline in the yearly TLI in the year 2010, falling to at least the objective RWLP TLI threshold. A more detailed analysis is provided by Scholes (2010).

The foregoing investigation demonstrates the need for policy measures to restore water quality for almost all Rotorua Lakes. Action plans are in place for Lakes Rotorua, Rotoiti[17], Okareka, Rotoehu, Okaro and Rotoma and action plans for the other lakes are in progress, except for Lake Rotomahana which is considered to be less urgent since its TLI does not exceed the 0.2 unit trigger (Scholes, 2010). The programme includes measures to address different sources of nutrients entering the lakes. Some of the mitigation measures being considered and in some cases already in place include sewerage works, treatment or diversion of nutrient-rich streams, capping lake sediments to lock up nutrients, construction of wetlands, and land management changes. The total cost of restoration is estimated at $144.2 million, of which the central government has committed $72.1 million over ten years starting from March 2008. So far, some of the restoration projects already implemented include the diversion of treated sewage from Lake Rotorua to the Rotorua Land Treatment Site in 1991, construction of a diversion wall to prevent

[17] Action plans for Lakes Rotorua and Rotoiti are implemented jointly since the two lakes are joined through the Ohau Channel.

high nutrient from flowing directly into Lake Rotoiti in July 2008, and construction of wetlands for Lakes Okareka and Okaro (MFE, 2011).

Although some of the lakes, such as Rotoiti and Rotorua, have experienced some improvements in water quality over the past few years, the extent to which such improvements can be attributed to the restoration programmes is unclear. For instance, PCE (2006 p.38), based on a conversation with Professor David Hamilton on the subject, reports:

> Note that a direct relationship between the diversion of sewage from Lake Rotorua and an improvement in water quality is difficult to discern, and is thus not necessarily the causal factor. It is likely that the changes in water quality after the sewage diversion were part of a natural cycle in phytoplankton biomass, the causes of which have still to be fully clarified.

Furthermore, some studies, including that of Rutherford *et al.* (1996), have shown that even with remedial measures in place, the recovery of water quality in the water column may take up to 20 years while for the bottom sediments it may take up to 200 years.

2.5 Chapter summary

With a focus on the Rotorua Lakes, the aims of this chapter were fourfold. Firstly, the chapter explored how water quality is measured, both locally and internationally. Secondly, an in-depth investigation of the status of water quality in the Rotorua Lakes was carried out. Thirdly, this chapter explored the different sources of water pollution in the Rotorua Lakes. Fourthly, the chapter investigated different mitigation measures that are being employed to protect and restore the water quality in the Rotorua Lakes.

Physico-chemical and biological indicators are the main measures employed by scientists to monitor the health of water bodies both locally and internationally. In the case of the Rotorua Lakes, the TLI is the main physico-chemical indicator of the eutrophication status of the lakes. The TLI is an aggregate measure of water quality based upon nitrogen, phosphorus and chlorophyll a concentrations and the Secchi depth. LakeSPI is the main biological measure employed to measure the ecological status of lakes. This measure is based on the Native Condition and Invasive Impact Indices. Additionally, microbiological assessment for toxic algal blooms is carried out to ensure that water quality is suitable for contact recreation.

This investigation has also shown that water quality in the Rotorua Lakes is quite variable, ranging from poor to good. Lake Okaro has the poorest water quality while Lake Rotoma has the best water quality based on the trophic level index. In terms of the LakeSPI Index, Lake Rotomahana has the best ecological condition while Lake Rotoehu has the poorest ecological health. Apart from the variability in water quality across lakes, seasonal variability of water quality within the same lake is not uncommon among the Rotorua Lakes. For most of the lakes, water quality tends to be poorer in warmer months and is often characterized by algal blooms. Over the past decade, Lakes Okaro, Rotoiti, Rotoehu and Rotorua have been the most affected by algal blooms.

Point and non-point sources coupled with internally regenerated nutrients are considered to be the main source of nitrogen and phosphorus loads, which, if excessive, lead to the eutrophication of lakes. Studies conducted so far indicate that sewage inflows and agriculture intensification around the lake catchments are the main sources of water pollution. In the case of water quality restoration and protection, relevant policies are in place, most of which are specifically designed to address the main sources of pollution. However, the extent to which pollution mitigation policies currently implemented have managed to reduce water pollution levels is still under investigation, although over the last few years some lakes have experienced some improvements in water quality. The work in this thesis intends to compliment these efforts by assessing the non-market benefits of improved water quality to trout anglers. A review of freshwater values and non-market valuation approaches is outlined in the subsequent chapter.

CHAPTER THREE

REVIEW OF FRESHWATER VALUES AND NON-MARKET VALUATION METHODS

3.0 Introduction

Since ancient times, market prices have been used as surrogate measures of value for most goods and services. However, many of the values or benefits of naturally endowed resources such as freshwater bodies cannot be directly assessed in dollar terms, and are referred to as "non-market values". Non-market valuation methods have been developed and have proved to be a very useful tool for assessing the value of environmental resources for which there is no price tag. These methods enable policy makers to take account of the costs and benefits of alternative policies, including both market and non-market values. Four main questions are addressed in this chapter. First, what values do freshwater ecosystems provide? Second, how are these values measured? Third, what are some of the methodological issues facing different non-market valuation techniques? Fourth, what are some potential gaps in the New Zealand freshwater non-market valuation context?

This chapter begins with a review of the benefits provided by freshwater bodies. This is followed by an outline of how non-market values are measured. The different non-market valuation approaches are explored including their potential strengths and limitations. In conclusion, a brief review of freshwater non-market valuation in New Zealand is provided. Some potential areas requiring further research are also highlighted.

3.1 The economic value of freshwater bodies

The concept of value has been a bone of contention throughout human history and continues to attract diverse and often conflicting notions from different schools of thought (Costanza, 1980; Farber *et al.*, 2002; Goulder & Donald, 1997;

Schumpeter, 1978). Philosophically, resources are conceptualized as possessing both intrinsic and extrinsic values. Intrinsic values refer to the value of a resource independent of its various benefits to humankind. On the other hand, resources may be valued extrinsically because of their relative contribution to the satisfaction of human needs (Bockstael & Freeman, 2005; Costanza, 2000; Farber *et al.*, 2002; Goulder & Donald, 1997). The economic concept of value refers to the latter and is mainly anthropocentric in nature. However, as noted by (Bockstael & Freeman, 2005 p. 521).

> The anthropocentric focus of economic valuation does not preclude a concern for the survival and well-being of other species. Individuals can value the survival of other species not only because of the uses people make of them (for food and recreation, for example), but also because of ethical concerns. The latter can be the source of existence or non-use values. Furthermore, this anthropocentric focus does not preclude the valuation of ecosystem services, properties and processes such as nutrient recycling, decomposition and biodiversity. To the extent that ecosystems enhance human-wellbeing through these services and processes, they have value.

In contrast, proponents of the economic concept of value contend that "ecosystems or species have intrinsic rights to a healthful, sustaining condition that is on a par with human rights to satisfaction" (Farber *et al.*, 2002 p. 376). Consequently, no amount of money can measure the value of natural resources because doing so would be to undermine the worth of such resources.

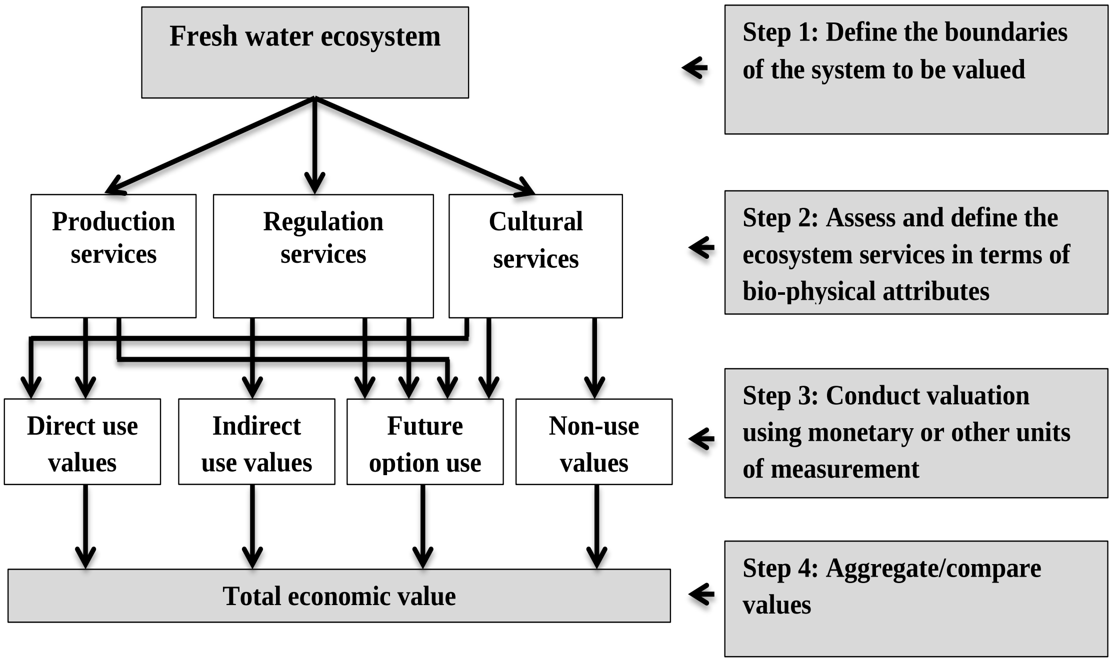
Nevertheless, the economic concept of value is widely accepted as a defensible means of measuring the worth of environmental resources. Therefore, in this thesis the word "value" refers to the economic concept of value. The term non-market value refers to the fact that many environmental attributes are not valued in the market. For example, people may gain enjoyment and satisfaction from visiting a river with clean waters without paying any entry fee. In this case, the market may not provide any indication of the benefit of clean waters to society. In contrast the prices of cars and other market goods provide the value of these goods and services.

In practice the situation is more complicated, since many goods have market and non-market components. For example, we may obtain some indication of the minimum value that anglers place on fishing by observing behaviour in the market through expenditure on fishing licences and fishing gear. However, many anglers may state that the value that they obtain from fishing far exceeds their direct expenditure in the market. The measurement of non-market values is the main focus of this thesis.

3.2 Applied framework for valuing freshwater ecosystem benefits

The total economic value (TEV) framework provides a widely-accepted means of aggregating the value of services provided by ecosystems. In this application the term ecosystem services refers to those contributions of freshwater bodies which generate goods and services which people value. Goods refer to physical products, for instance provision of fish, as well as less tangible goods such as flood control (Bateman *et al.*, 2011). The Millennium Ecosystem Assessment (2005) classifies services offered by freshwater services into four main categories: provisioning; regulating; cultural; and supporting services. These classifications and related examples are illustrated in Appendix 2. Hein *et al.* (2006) give an outline of how the different classes of ecosystems services relate to the TEV framework, as shown in Figure 3.1 below. The necessary steps required for ecosystem valuation are also illustrated.

Figure 3.1: TEV framework for valuing freshwater ecosystem services



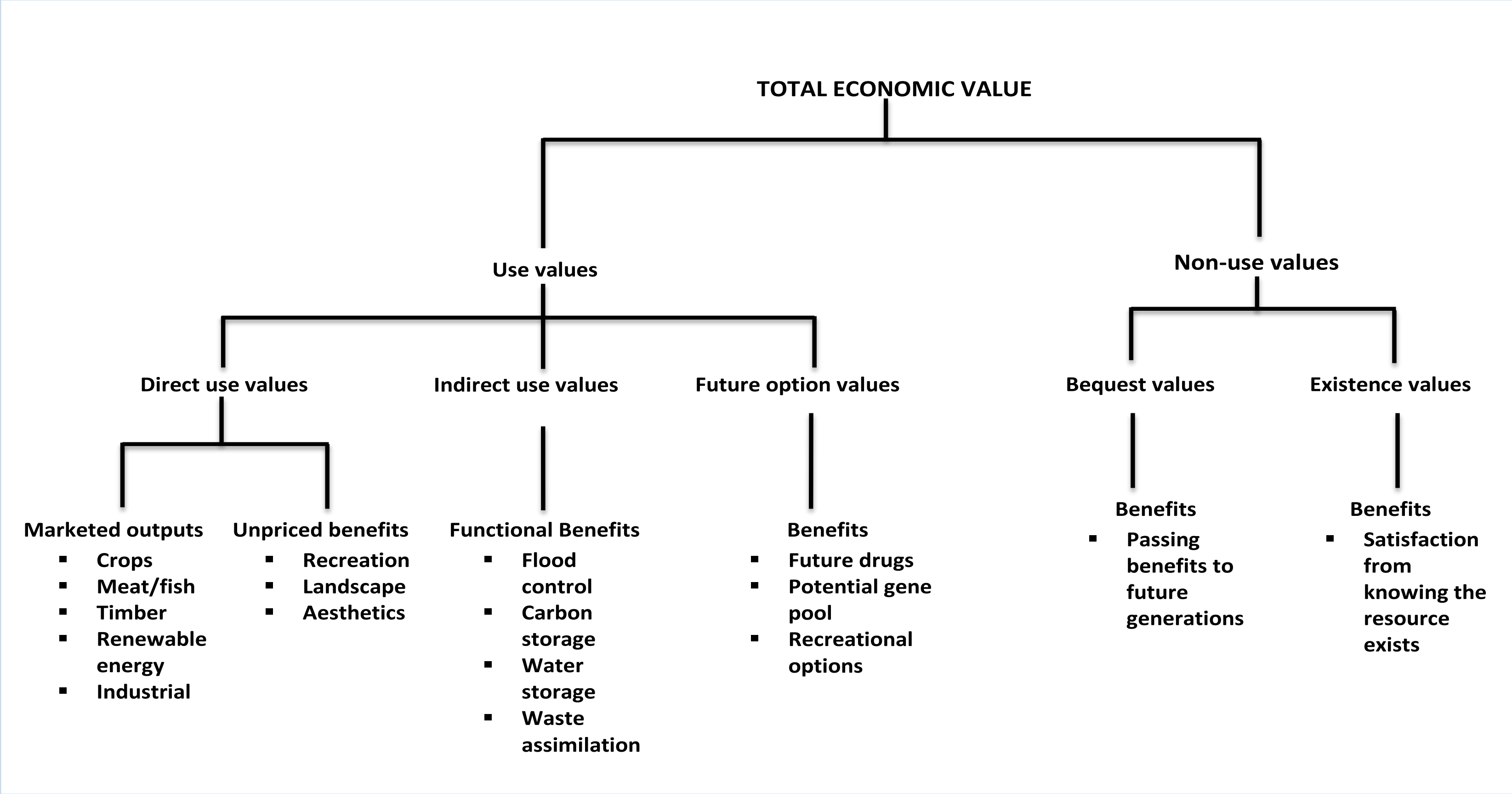
Adapted from Hein *et al.* (2006)

The TEV in Figure 3.2 provides a similar classification including examples of the various category values. The values or benefits provided by freshwater bodies are classified into use and non-use values.

Use values

Use value refers to the benefits that individuals derive from actual use of freshwater resources. It can be subdivided into direct, indirect and option values.

Figure 3.2: Total economic value (TEV) with examples of different categories of values



Source: Adopted from Anderson *et al.* (1999 p. 3)

Direct use values are separated into marketed outputs (e.g. crops, timber, renewable energy) and non-market use values (e.g. recreation, landscape and aesthetics). Marketed services are those that can be bought or sold through market transactions. For example, water for irrigation is combined with other factors of production to produce crops which can be sold in a market. This also includes commercial fisheries, electricity generation and other industrial uses of water. Many of the freshwater services identified do not have price tags associated with them and are classified as non-market services.

Indirect use values are those that are gained from freshwater through support and protection of other economic activities: for example, freshwater use in diluting, storing and detoxifying waste products and pollutants ensures a healthy environment for human well-being.

Option values recognize that people may wish to secure a resource for their own future direct or indirect use (Desvouges *et al.*, 1987; Hartman & Plummer, 1987; Shafran, 2014; Smith, 1987). For example, individuals can attach value to the continued availability of trout in Lake Rotorua for their own future trout angling use.

Non-use values

The notion of non-use (passive use) values was introduced by Krutilla (1967). Non-use values are independent of people's own use of resources and are classified into bequest and existence values. Under bequest value people place a value on a healthy freshwater ecosystem to ensure its preservation for future generations. Alternatively, people might derive satisfaction from the mere knowledge that a resource exists or that its quality is preserved independent of their own use and that of future generations (Bockstael & Freeman, 2005). This aspect of non-use value is called existence value.

3.3 The theoretical framework for measuring value

The basic premise for measuring value is founded on neoclassical welfare economics. Welfare economic theory assumes that people have preferences for goods and that preferences for bundles of goods are ordered based on the utility that is derived from the consumption of each bundle[18](Flores, 2003; Freeman, 2003). The bundles of goods may consist of both market and non-market goods. The standard neoclassical price theory assumes that the quantity of market goods bought depends on individual preferences, the relative prices of the market goods, and income. For non-market goods, the demand curve is not directly observable. To accommodate the demand for non-market goods, the quantity of market goods is assumed to be a function of the level of income, prices of other market goods and some rationed level of non-market goods (Flores, 2003). The value that people place on non-market goods can be measured by how much income an individual is willing to pay or to accept that would leave them as well off as before a change[19].

The concept of willingness to pay dates back to the work of Dupuit (1844) who proposed a monetary measure of value as the price associated with a given amount of goods along the consumer's demand curve. Based upon this concept of value Marshall (1890) proposed a measure of the benefits associated with different levels of utility known as consumer surplus. Marshall defined consumer surplus as the difference between what consumers are willing to pay for the product and what they actually pay for it.

This concept of consumer surplus was further developed by Hicks (1939b) and Kaldor (1939). Since then it has been generally regarded as the accepted measure of the benefits or losses arising from quantity, price, income and policy changes. In environmental valuation consumer surplus can be viewed in terms of willingness

[18] Goods as used in this context refer to both goods and services.

[19] Willingness to pay or accept will depend on whether or not an individual has the property right to the initial endowment, however, for most environmental goods property rights are not clearly defined.

to pay (WTP) for specific improvements or preservation of environmental values, or willingness to accept compensation (WTA) in case of a loss or degradation. Since the demand for environmental resources is latent, non-market valuation techniques are used to identify the value that society places on these resources.

3.4 Non-market valuation methods

The origin of non-market valuation techniques dates back to the late 1940s. Since then, two pathways have emerged: revealed preference and stated preference techniques. Under revealed preference techniques the value of an environmental good for which a market does not exist is inferred indirectly from actual market transactions. The most prominent techniques include the travel cost method and hedonic pricing. Stated preference techniques use hypothetical markets to infer the value of non-market goods and the main valuation approaches are the contingent valuation method and choice experiments.

Travel cost method

Since its proposal by Hotelling (1947) the travel cost method (TCM) has become the main non-market valuation technique for valuing the recreational use of natural resources. The TCM is a revealed preference technique which relies on weak complementarity between marketed and non-market goods. In particular this technique postulates that the latent recreational use value of environmental resources can be inferred indirectly from what people are willing to pay to access the site. The cost of accessing the recreational site, which mainly includes travel costs and the opportunity cost of time, is used as a proxy for the price of the recreational enjoyment.

Clawson (1959) and Clawson & Knetsch (1966) were among the first to apply the TCM in recreational literature. These earliest studies were applied to single sites using aggregated visitation rates (zonal data) for individuals living within different concentric zones around the recreational site. Application of the TCM to micro-level data was initiated by Burt & Brewer (1971). To date the use of individual level data has dominated the recreational literature. Count data models are the predominant approach used in the analysis of single-site travel cost data to account for the non-negative integer characteristics of recreational trips.

However, the use of the single-site TCM is found to be of limited use in measuring the value of changes in the quality of recreational sites where site attributes do not vary over the sampled recreational users (Parsons, 2003; Phaneuf & Smith, 2003)[20]. Furthermore, while it is possible to include prices and other site attributes of substitute sites within a single-site travel cost model, it becomes rather complicated to estimate as the number of substitute sites increase (Freeman, 2003). Additionally, they can only be used to estimate recreational demand for the sites visited by an individual and are unable to predict recreational demand for unvisited sites. The prevalence of unvisited sites is referred to as the extreme corner solution problem (Bockstael *et al.*, 1987b).

The above cited limitations of the single-site travel cost approach led to the development of multiple-site travel cost recreational demand models. These models are classified into site choice, and site choice and participation models (Thiene & Signorello, 2008). Travel cost random utility models (RUMs) are the most popular site choice models. RUMs were introduced by McFadden, (1974) and were first applied to recreation by Hanemann (1978) and were further developed by Bockstael *et al.* (1986). Since then numerous studies have utilized these models.

In addition to overcoming the weaknesses of the single-site travel cost models, RUMs are particularly attractive due to their ability to link statistical models with well-founded behavioural theory in describing individual choice decision processes (Hunt, 2010). Despite their popularity, RUMs are of limited use for estimating seasonal welfare estimates that account for changes in recreational participation levels induced by changes at one or more sites. To overcome this

[20] To overcome this limitation, some researchers opt for the use of perceived measures of environmental quality as opposed to objective measures, but the former measures are said to be of limited use in evaluating management policy options (Adamowicz *et al.*, 1997a).

limitation, a number of approaches for linking RUMs to trip frequency or participation models have been developed.

Unlike the RUM, trip frequency models regard trip choice occasions by an individual over a recreational season to be interdependent and hence more suited for predicting recreational demand due to environmental/policy changes at a site (Creel & Loomis, 1992). Linking the site choice and the participation models enables analysts to predict seasonal welfare gains/losses that take account of both the substitution effect from the site choice model and changes in the number of recreational trips through the participation model (Parsons *et al.*, 1999a).

Bockstael *et al.* (1987a) were among the first to propose the need to link the site choice model and participation models. They proposed a methodology in which the participation equation is regressed against the inclusive value index calculated from the site choice model. According to Loomis (1995 p.60) the inclusive value index "represents the net utility (benefits of site visit-directly related to site quality-minus the travel costs) from the site being available on any choice occasion[...]".

The product of the per trip benefit welfare measure from the RUM and estimated number of trips from the participation model is considered to be a proxy for the aggregate seasonal welfare measures following a policy change at a recreational site (Loomis, 1995; Parsons *et al.*, 1999a).

Morey *et al.* (1993) proposed the use of the repeated nested logit that combines both the participation and site choice decisions.

Hausman *et al.* (1995) modified the approach of Bockstael *et al*.(1987) and instead proposed that the inclusive value index from the site choice model be rescaled by the price coefficient. The resulting ratio, which is referred to as the price index, is used as the basis for predicting changes in recreational demand and seasonal welfare measures.

Parsons & Kealy (1995) and Feather *et al*. (1995) proposed similar approaches in which the inclusive value index from the RUM is split into price and quality indices. The seasonal trip prediction models are constructed and used to predict recreational demand due to changes in both the price, and quality indices. The two approaches use the same measure for the price index but use different measures for the quality index[21].

To appraise these approaches, Parsons *et al.* (1999a) used a common data set and found little difference in the average benefit estimates calculated using the approach of Bockstael *et al.* (1987) and those of Morey *et al.* (1993) and Hausman *et al.* (1995) However, it was found that the approaches of Parsons and Kealy (1995) and Feather *et al.* (1995) produced seasonal welfare estimates that were different from each other as well as from the approaches of Morey *et al.* (1993) and Hausman *et al.* (1995). The results also indicated some lack of consistency between the site choice model and the seasonal trip prediction model as noted by Parsons *et al.* (1999a, p. 153).

> "Parsons and Kealy's seasonal value of $3.41 for the advisory scenario is about twice the per-choice occasion values at $1.77. This is surprising insofar as the average person in our sample takes over 12 trips. If the site and trip models were consistent, one would expect seasonal values at least as large as 12*$1.77."

The approach of Hausman *et al.* was questioned by Herriges *et al.* (1999) on the basis that it is sensitive to both the utility specifications and participation models employed.

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[21] Based upon the approach of Parsons and Kealy (1995), the quality index is denoted by $Q(p, x) = \sum_i Pr(i)\gamma x_i$, while in the case of the approach of Feather *et al.* (1995), the quality index is calculated as $Q(p, x) = \sum_i Pr(i) x_i$, where x_i is a vector of non-price attributes at site i, γ is a vector of the estimated coefficients from the RUM, and $Pr(i)$ is the probability of site i being chosen.

On the other hand, in comparing the approaches of Morey *et al.* (1993) and Bockstael *et al.* (1987), Feather *et al.* (1995 p.217) state:

"Although the repeated discrete choice approach has been shown to be utility theoretic [...], it is an implausible explanation of behaviour because it assumes participation decisions are made independently over several disjoint time periods. The approach advocated by Bockstael *et al.* [...] is a more plausible and intuitively appealing, [...]" since it doesn't assume trips occur on an independent basis but rather that individuals plan trips over some time period.

Parallel to the RUM, alternative methods capable of modelling site choice and participation decisions concurrently have been developed. The Kuhn-Tucker demand models, also referred to as "continuous demand systems," are the most dominant. These models were first introduced in recreational literature by Hanemann (1978) and were further championed by Wales & Woodland (1983) and Lee & Pitt (1986). In contrast to discrete RUMs, the Kuhn-Tucker approaches assume that the recreational decision processes are made at the beginning of a season and therefore, trip choice occasions are considered to be interdependent. These techniques are best suited for estimating seasonal recreational demand and welfare measures. Kuhn-Tucker demand models have been applied in some studies (e.g. Phaneuf *et al.*, 2000; Von Haefen & Phaneuf, 2005; Whitehead *et al.*, 2010). However, their application is limited due to intensive computational requirements. Additionally, their ability to fit recreational studies with a large number of recreational sites is still being tested (Phaneuf & Smith, 2004).

Consequently, to date travel cost RUMs are increasingly being applied in a number of recreational studies (e.g. Egan *et al.*, 2009; Kaoru, 1995; Morey *et al.*, 2002; Murdock, 2006; Train, 1998). In offering support, Phaneuf & Smith (2004, P.32) state "[…] Research in this area is so extensive that it is impossible to do justice to all of it […]."

There are a number of other methodological issues regarding the use of TCM in general and these include how to incorporate the opportunity cost of travel and on-site recreational time. The need to account for the opportunity cost of time was recognized by earlier researchers in the field including De Grazia (1962) and Clawson & Knetsch (1966). It is argued that both travel time and on-site time should be considered as scarce resources which could be put into other alternative uses that might yield some utility to an individual (Parsons, 2003; Phaneuf & Smith, 2003). As such the opportunity cost of leisure time should be accounted for. In expressing his support, Cesario (1976 p.32) states:

> A favoured method for imputing recreational site demand curves is the so-called Hotelling-Clawson-Knetsch (HCK) approach. […] It suffices to point out merely that a fundamental problem with the application of this method has been the difficulty of capturing effectively the value placed on travel time by consumers of recreational services. Failure to explicitly incorporate this aspect of recreational usage into the HCK analysis results in the imputation of the demand curve which is biased downward from its true position. Consequently, the benefits of the site are estimated conservatively.

Despite this need, there is no consensus on the methodological framework for modelling the opportunity cost of time. Two dominant methodological approaches have been advocated in the literature. One approach involves the use of stated preference methods in which individuals are directly asked the amount of money they would be willing to pay to reduce the travel time, for instance using the contingent valuation method (Casely & Vukina, 1995). Alternatively, choice experiments can be used to infer the value of travel time by observing the trade-offs that individuals make between travel time and other monetary attributes. However, the use of stated preference methods may be subject to hypothetical biases and furthermore, incorporating stated choice tasks into travel cost surveys may over-burden respondents (Fleming & Cook, 2008).

The predominant approach applied in many studies is the use of the wage rate or some fraction of it as a proxy for the value of travel time, as advocated by Cesario (1976). The theoretical foundation for the use of this approach has its roots in the revolutionary work of Becker (1965) and De Serpa (1971) in their classical-labour and leisure decisions[22].

To impute the appropriate wage rates, researchers in the field have mainly relied on the labour-supply behavioural models of Heckman (1974). However, the imputation of the appropriate wage rate has remained controversial among researchers. First, imputing the value of the time as a proportion of the wage rate is based on the assumption of a flexible working schedule where there is perfect substitutability between work time and leisure time as implied by Heckman's labour-supply behavioural models[23]. In this case it is assumed that an individual makes trade-offs between work and leisure until the wage rate at the margin is equal to the value of an hour's leisure time (Bockstael *et al.*, 1987b). However, it is argued that this assumption may not be practically applicable in cases where the number of hours of work is constrained (Feather & Shaw, 1999).

Second, it is difficult to determine the appropriate wage rate for individuals who are not in the work force. To circumvent this problem researchers have advocated the use of the hedonic wage model, in which observed wages for those individuals in the work force are regressed against their respective social economic covariates (Smith *et al.*, 1983; Van Soest, 1995). Consequentially, the wage equation obtained is used to predict wages for those individuals with similar social economic characteristics, but currently not in the work force. The major weakness of this

[22] In the traditional classical theory of choice, households were assumed to maximize utility from the consumption of marketed products subject to income constraints only. These authors revolutionized this early economic thought by advocating the need to recognize that households maximize utility subject to both income and time constraints.

[23] Attempts to refine Heckman's earlier work by relaxing the assumption of flexible hours of work to incorporate fixed hours of work have had little success (Moffit, 1983).

approach is that it assumes that both workers and non-workers have identical preferences for work versus leisure time.

Third, Shaw (1992b) argues that the imputed cost of leisure time based upon the wage rate might not truly reflect the value of time for some individuals. For instance, time might be valued very highly by some individuals with low wages and likewise, some individuals with high wages might not value time highly[24].

To date the use of the wage rate or some proportion of it has generally dominated the recreational literature. Most studies advocate the use of 33% of the wage rate (e.g. Coupal *et al.*, 2001; English & Cameron, 1996; Hagerty & Moeltner, 2005; Hellerstein & Mendelsohn, 1993). As a general guideline, there is a consensus among researchers in the field to consider 25% as the lower bound and 100% or full wage as the upper bound (Parsons, 2003). On the other hand, Ward & Beal (2000) contend that in cases where individuals are engaged in recreational activities during holidays, 0% may be appropriate since no income is foregone. Feather & Shaw (1999) and Parsons (2003) have shown that benefit estimates tend to be sensitive to the ad hoc manner in which wage rates are determined. Despite this the choice of which proportion of the wage rate to use remains subjective.

While researchers have made some advances in developing methodologies for incorporating the value of the opportunity cost of travel time, the treatment of on-site time remains problematic and highly subjective. This stems from the fact that time spent at a recreational site is chosen by each individual and therefore, considered to be endogenous. As a result some researchers regard on-site time as a

[24] In addition to the outlined complexities in estimating the value of travel time, Walsh *et al.* (1990) question whether travel costs in general should be regarded as the true price people are willing to pay to access recreational sites. Their bone of contention is that since individuals might derive some consumptive benefits in travelling to and from the recreational site, the true travel costs should be equal to the net of these consumptive benefits. However, it remains a challenge as to how a researcher can adequately measure and attach a monetary value to these consumptive benefits.

proxy for the amount of recreational activity consumed by an individual and hence already accounted for by the other costs of site access, mainly travel expenses[25] (Phaneuf & Smith, 2004).

Other unresolved methodological issues include multicollinearity problems, how to account for unobserved effects and multiple purpose trips. These issues are further reviewed in subsequent chapters.

Hedonic pricing

Hedonic pricing was originally popularized in the study of automobile demand by Court (1939) and later on by Griliches (1971). The basic notion of the hedonic pricing (HP) technique is that the underlying value of a good is a function of its different characteristics. If the good can be marketed, it is possible to decompose the market value of the good into its constituent parts, which can be regarded as the implicit prices for each of its inherent characteristics. The underlying logic of this technique is to regress the per unit price of the marketed good on various attributes of the good. Some of these attributes can be environmental aspects such as proximity of good water quality and recreational amenities. The implicit marginal prices of non-marketed environmental attributes are inferred from the parameter estimates (Palmquist, 2005; Sinden, 1994).

In environmental valuation, housing and land markets are commonly used to infer the value of ecosystem goods and services (Michael *et al.*, 1996; Palmquist, 2005). HP has been used in the valuation of a number of freshwater ecosystem attributes including water quality (e.g. Michael *et al.*, 1996; Michael *et al.*, 2000; Young & Shortle, 1989); water view (e.g. Luttik, 2000); and stream proximity (e.g. Qui *et al.*, 2006).

[25] Alternatively, the opportunity cost of on-site time is computed as a proportion of the wage rate and added to the other costs.

Palmquist (2005) gives a detailed review of the HP method and various modelling approaches that have been proposed over time, including work by Rosen (1974), Brown & Mendelsohn (1984) and Pendleton & Mendelsohn (2000).

Like the TCM, the main advantage of HP is the use of observations on actual choices made by individuals thus avoiding potential problems associated with hypothetical questions such as strategic responses (Adamowicz *et al.*, 1994). Despite this strength, the use of the HP may be limited in some cases. For this technique to be applied, one major prerequisite is complementarity between the non-marketed environmental resource of interest and some marketed goods. This poses some limitations on the applicability of the technique. For instance, consider three sites, A, B and C on three different lakes with varying water quality. Assume that sites A and B are well developed with various beach properties and other amenities. On the other hand, site C is relatively unexploited with no beach amenities but is occasionally be used for recreational purposes such as canoeing. According to the complementarity requirement, the value of the water at sites A and B can be measured by assessing its contribution to beach property values. However, the value of the water at site B cannot be measured due to the non-existent marketed goods in which the value of the water can be embedded.

Furthermore, the HP method may be unreliable in cases where buyers do not have appropriate variables to measure environmental attributes (En Chee, 2004). Another potential limitation of the HP approach is its assumption that housing markets operate in a competitive equilibrium framework. As noted by Freeman (1979), the market clearing conditions may not be met in real world markets. Just like the TCM, the HP is also prone to multicollinearity and endogeneity problems.

Other less frequently used revealed preferences techniques, generally classified as cost based valuation methods, are outlined in Appendix 3. These include the damage assessment cost method, the production function approach, the avertive expenditure method and the cost of illness approach.

Despite their ability to offer estimates of environmental values which are based on actual market choices, revealed preference methods cannot be used to assess non-use values. To fill this gap, stated preference techniques have been developed and utilized over time. It is also recognized that some of the weaknesses of revealed preference techniques can be overcome through the use of stated preference techniques, including multicollinearity and endogeneity problems. Similarly, stated preference techniques can be enhanced by revealed preference, including the reduction of hypothetical bias[26] (Hensher *et al.*, 2005; Whitehead *et al.*, 2008).

Since the pioneering work of Thurstone in the late 1920s and early 1930s, various forms of stated preference techniques have evolved and been applied to infer the value of non-market goods[27]. The predominant approaches are the contingent valuation method and choice experiments.

The contingent valuation method

The origin of contingent valuation method (CVM) dates back to Ciriacy-Wantrup (1947). He proposed that in order to identify the latent demand for non-marketed goods, "individuals should be asked how much money they are willing to pay for successive additional quantities of a collective extra-market good. If the individual values are aggregated, the result corresponds to the market demand schedule" (P. 1189). The work by Davis (1963), in which both the contingent valuation and the TCM were used to estimate the value of a Maine woods to recreational hunters and wilderness lovers, represents one of the earliest applications of this technique.

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[26] To exploit the benefit of each of technique, a data fusion approach known as the combined revealed preference – stated preference (RP – SP) is advocated (Hensher & Bradley, 1993).

[27] The origin of stated preference methods dates back to Thurstone's work in the 1920s and early 1930s. Using psychophysical judgment concepts, Thurstone conducted a paired comparison experiment involving several crimes in which subjects were asked to rate the seriousness of the offences (Thurstone, 1927a, 1927b, 1927c). Further advances to infer consumer preferences using stated preference methods were made in the early 1930s. In his endeavour to estimate indifference curves, subjects were asked to choose between different combinations of overcoats and hats (Thurstone, 1931)

The use of CVM continued to receive the support of other researchers. For instance, to express his support for the CVM, Schelling (1968, pp. 143-44) states:

> In any case, relying exclusively on market valuation and denying the value of direct enquiry in the determination of government programs would depend on there being for every potential government service, a close substitute available in the market at a comparable price. It would be hard to deduce from first principles that this is bound to be the case[28].

The need to value non-use values of environmental goods, championed by Krutilla (1967), also contributed to the widespread recognition of the CVM. Birol *et al.* (2006) document that more than 5000 CVM studies have been conducted in over 100 countries. Recently, Carson (2011) published a bibliography of over 7500 contingent valuation studies from 130 countries, spanning 50 years. A number of these studies focused on the valuation of water quality and quantity. Studies in the valuation of water quality improvements include the work by Desvouges *et al.* (1987) in which the option price bids for improved recreation from better water quality were estimated. In another study Carson & Mitchell (1993) used the CVM to estimate the benefits of the Clean Water Act for all rivers in the US. Other studies in this area include work by Le Goffe (1995); Brox *et al.* (2003) and Atkins & Burdon (2006).

Parallel to the widespread use of the CVM, especially in the 1980s, was the growing need to scrutinize and validate the reliability of this technique. This was mainly necessitated by federal laws in the US which required parties responsible for natural resource damage to be identified and be made to pay for the clean-up[29].

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[28] This is not focused on CVM only but valuation in general.

[29] One of these laws was the Comprehensive Environmental Response, Compensation and Liability Act of 1980. Its primary purpose was to create a mechanism for identifying sites at which hazardous materials posed a threat to human health or the environment, and also to establish procedures through which parties that were deemed responsible for the contamination could be identified and made to pay for the clean-up (Portney, 1994 p. 6).

The CVM was identified as the means by which environment damages can be assessed. However, the inclusion of non-use values or existence values was highly contested by parties responsible for damages. The Exxon-Valdez oil spill on 24th March 1989 in Prince William Sound in Alaska aroused much controversy regarding the use of the CVM as a valid valuation technique. In response to this extensive oil spill, Congress passed the Oil Pollution Act in 1990. This law directed the National Oceanic and Atmospheric Administration (NOAA), under the auspices of the Department of Commerce, to come up with its own regulations to govern damage assessment. The NOAA sought advice from a panel of experts, chaired by Kenneth Arrow and Robert Solow, on whether or not the CVM was capable of giving reliable non-use values. In January 1993 the panel submitted a report to NOAA which endorsed the CVM as a valid valuation method under a set of guidelines (Portney, 1994).

In spite of its endorsement by the NOAA panel, the use of this technique has been received with mixed feelings. For instance, Diamond & Hausman (1994) have questioned the reliability and validity of contingent valuation estimates on several grounds, including insensitivity to scope or embedding bias.

In line with economic theory, it is expected that the number of respondents willing to pay for a particular good should fall as the price increases. Furthermore, WTP should correspond to the quality or quantity of the good being valued. The two are generally referred to as price and scope tests, respectively. As noted by Carson (2000), while most CVM studies pass the price test, the scope test has been the main source of the controversies surrounding CVM studies. Using empirical evidence from a number of CVM studies, Carson (1997b) refutes the claims of WTP insensitivity to scope as championed by Kahneman & Knetsch (1992). Out of the reviewed studies, 31 studies passed the scope tests, while 4 did not and Carson attributes the insensitivity to scope in these 4 studies to poor survey designs and administration procedures (Carson, 1997b).

Empirical evidence from other studies also offers support to Carson's claim that the CVM estimates may conform to economic theory. Whitehead *et al.* (1998) carried out a study to test if CVM estimates were insensitive to scope of policy using data on water quality improvements in the Albermarle and Pamlico Sounds in North Carolina. The WTP estimates were found to be sensitive to scope of policy and the authors further dismissed the general perception of attributing insensitivity to scope of policy to inexpensive survey methods. In a study by Bateman *et al.* (2006) on the valuation of the benefits of improved water quality, CVM estimates were found to be consistent with both empirical and theoretical expectations.

Furthermore, it is contended that WTP responses may suffer from order effect, whereby "the same good elicits a higher WTP if it is first in the list rather than valued after others" (Kahneman & Knetsch, 1992 p. 58). Some possible causes of order effects include, imperfect information about the decision problem (Halvorsen, 1996) and inexperience with the valuation scenarios (Boyle *et al.*, 1993b). It has been argued that this effect can be circumvented through well designed survey instruments (Carson & Mitchell, 1995; Powe & Bateman, 2003).

Critics have also argued that WTP from CVM studies may be influenced by non-economic motives such as "yea-saying' and "warm glow" or moral satiation tendencies. Mitchell & Carson (1989 pp.240-41) define yea-saying as the "tendency of some respondents to agree with an interviewer's request regardless of their true views." Blamey *et al.* (1999 p. 126) define yea-saying as the "tendency to subordinate outcome-based or "true" economic preferences in favour of expressive motivations when responding to CVM questions". Some environmental valuation studies have acknowledged the effect of yea-saying tendencies on value estimates (Boxall *et al.*, 1996c; Boyle *et al.*, 1993a; McFadden & Leonard, 1993).

Kahneman & Knetsch (1992, p. 64) further contend that WTP for public goods should not be considered as an economic value but rather moral satiation. "We offer the general hypothesis that responses to the CVM question express a willingness to acquire a sense of moral satisfaction (also known as a "warm glow of giving" by a voluntary contribution to the provision of a public good." Some studies, for example Cooper *et al.* (2004), have shown that WTP estimates are not related to individuals' moral obligation to pay for a public good.

Opponents, such as Kahneman & Knetsch (1992) and Diamond & Hausman (1994), have also argued that WTP estimates can be affected by factors such as familiarity with the good in question, hypothetical biases, type of payment vehicle used, the attitudes of the surveyor and the starting bid price. On whether or not respondents can express true preferences for goods that are unfamiliar, Carson (2000, p.9) had this to say:

> To deny that people have meaningful preferences about new commodities, political issues, cultural questions, and the like, without prior personal experience with them would be tantamount to suggesting that only those individuals who had actually visited the Louvre can value the preservation of its art work and that all votes for non-incumbent politicians should be disregarded. These simple specific examples illustrate that specific personal experience is not required for making meaningful economic choices.

Overall, researchers in support of the CVM contend that most biases against CVM can be circumvented through careful design of the surveys (e.g. Carson, 1997a; Carson *et al.*, 2001b; Carson *et al.*, 1996; Hanemann, 1994). Nevertheless, sentiments against the use of CVM led to the development of alternative stated preference techniques known as choice experiments.

Choice experiments

Choice experiments (CE) have gained widespread recognition since their early application by Louviere & Hensher (1982b) and Louviere & Woodworth (1983b) and their application to environmental valuation by Boxall *et al.* (1996). A choice experiment is an attribute-based technique in which respondents are presented with different alternatives defined in terms of product attributes and are asked to state, rank or select their preferred choice. The ranking or rating of alternatives is commonly known as conjoint analysis. The attributes vary from one alternative to another. In environmental valuation, it is recommended that one of the attributes should involve a monetary measure to enable the researcher to derive implicit estimates of monetary value, under a set of well qualified assumptions (Bennett & Blamey, 2001). A number of studies have employed CE in the valuation of water quality improvements (e.g. Hanley *et al.*, 2006; Marsh *et al.*, 2011; Morrison & Bennett, 2004; Viscusi *et al.*, 2008).

CEs are considered to offer a number of benefits over CVM. While CVM values the environmental good as a whole, in CE the good is described in terms of its attributes. This feature of CEs enables researchers to determine the attributes that are most valued by individuals, find out the relative rankings of attributes, obtain the marginal WTP for changes in each of the significant attributes and assess the implied WTP to attain some hypothesized alternative states of an environmental good. Overall, CEs enable researchers to obtain multiple values for an environmental good, unlike CVM which views the good as a whole. This enables policy makers to target improvements in those aspects of an environmental good that are most valued by society (Adamowicz *et al.*, 1998a; Bennett & Blamey, 2001; Hanley *et al.*, 1998).

Furthermore, it is contended that some of the biases against the use of CVM can be minimized through the use of CE. It is argued that since WTP is inferred indirectly through the cost attribute in the choice sets, the “yea-saying” bias is minimized. Additionally, it is also asserted that by varying attributes and attribute levels, the choice experiment estimates are more likely to be stable and sensitive to scope of policy (Bennett & Blamey, 2001; Boxall *et al.*, 1996b; Hanley *et al.*, 1998).

Despite the strengths of CE, detailed experimental designs involving a large number of attributes, attribute levels and alternatives may be over-taxing for respondents. Related to this, pertinent issues being addressed include the effect of choice complexity (e.g. Boxall *et al.*, 2009b; DeShazo & Fermo, 2002; Meyerhoff & Liebe, 2009); whether or not respondents attend to all information in choice cards (e.g. Campbell & Lorimer, 2009; Carlsson *et al.*, 2010; DeShazo & Fermo, 2004; Hess & Hensher, 2010; Ryan *et al.*, 2009; Scarpa *et al.*, 2009): the role of the status quo alternative (e.g. Adamowicz *et al.*, 1998a; Breffle & Rowe, 2002; Hensher & Rose, 2007; Kahneman & Tversky, 1979; Samuelson & Zeckhauser, 1988; Scarpa *et al.*, 2005b) and how to come up with the best experimental designs (e.g. Hess & Rose, 2009; Rose *et al.*, 2008). These issues and their implications for WTP values and how best to circumvent them are some of the topical debates surrounding the use of CE.

In addition to the above valuation methods, benefit transfer can be used to transfer existing information from completed studies in one location to another location. Some authors, for instance Kerr (2011), use the term value transfer since this covers the transfer of both costs and benefits. Benefit transfer is generally applied in cases were primary studies cannot be undertaken due to time and financial constraints. Further discussion of this approach is provided in Appendix 4.

3.5 Non-market valuation studies in New Zealand

In New Zealand, the number of studies using non-market valuation methods to assess the value of environmental resources is steadily increasing. The New Zealand Valuation Database[30] provides a record of most non-market value studies conducted in New Zealand since 1974. Yao and Kaval (2007) provide an overall assessment of the New Zealand non-market value literature up to that date. They found a significant increase in the volume of studies, specifically those requested by government agencies, following the passage of the 1991 Resource Management Act (RMA). These studies were concentrated in three main areas: outdoor recreation, environmental conservation/management, and travel time savings. In spite of this increased activity there is a severe lack of studies in many areas including pest control, water resources and outdoor recreation.

[30] http://www2.lincoln.ac.nz/nonmarketvaluation/

More recently Marsh and Mkwara (2013) provide a review of freshwater non-market studies in New Zealand from 1990. They note a general increase in the number of studies using CE, partly reflecting the world-wide popularity of this technique since its introduction in the 1990s. The TCM and HP were used less commonly, with the latter being applied in only two studies. With the exception of this application no study has utilized travel cost RUMs. Overall there is a lack of data for many freshwater non-market values, this fact being first highlighted by Yao & Kaval (2007). This thesis contributes to the New Zealand non-market valuation literature through the use of travel cost RUMs and CE.

3.6 Chapter summary

The main objectives of this chapter were fourfold. First, to explore the various values or benefits provided by freshwater bodies. Second, to investigate different non-market valuation approaches used to assess the value of goods and services that cannot be sold or bought in markets. Third, to investigate possible strengths and limitations of non-market valuation methods. Fourth, to identify some potential gaps in the literature in the New Zealand valuation context.

Freshwater ecosystems provide both use and non-use values. Use value refers to the benefits that individuals derive from actual use of freshwater resources and are classified into direct, indirect and option values. Direct use values reflect the satisfaction that individuals derive from using freshwater directly and include the use of water for irrigation and recreation. Indirect use values are those that are gained from freshwater through support and protection of other economic activities, including, for example, diluting, storing and detoxifying waste products and pollutants, thus ensuring a healthy environment for human well-being. Option values refer to people's desire to secure a resource for their own future direct or indirect use. Non-use values are independent of people's own use of resources and are classified into bequest and existence values. People may derive satisfaction from the mere knowledge that a resource is preserved for future generations and this is known as bequest value. On the other hand, some people may be satisfied just by the mere knowledge that a resource exists: this aspect of value is called existence value.

While some use values, such as the value of a lake for commercial fishing, can be assessed through market transactions, a number of other values cannot be determined through the markets. For instance, there is the aesthetic appeal that clean water provides to recreational users. Similarly, non-use values cannot be traded through markets. The concept of consumer surplus is generally regarded as the commonly accepted measure of the benefits or losses arising from quantity, price, income and policy changes. To assess the value of non-market goods and services, non-market valuation methods have been developed and used. In environmental valuation the value of natural resources is measured in terms of people's WTP for specific improvements or willingness to accept compensation (WTA) in case of a loss or degradation.

Non-market valuation methods are classified into revealed preference and stated preference techniques**.** Under revealed preference techniques the value of an environmental good for which a market does not exist is inferred indirectly from actual market transactions. The most prominent techniques include the TCM and HP.

The TCM is used for valuing the recreational benefits of natural resources; it assumes that the recreational use value of environmental resources can be inferred indirectly through what people are willing to pay to travel to a recreational site. The HP technique is based on the assumption that the underlying value of a good depends on its different characteristics. For example, it is assumed that the value of a house on the shore of a lake depends in part on water quality in that lake, so the aesthetic and landscape value of the lake can be indirectly inferred through housing prices. As is the case with the TCM, HP is capable of providing value estimates which are inferred from actual market transactions. Both techniques can only be used to assess use values. The prevalence of multicollinearity and endogeneity problems is another limitation of these techniques. Also, specific to travel cost recreational demand models is the ongoing debate on how to appropriately account for the opportunity cost of leisure time.

The main stated preference techniques are CVM and CEs. These methods can be applied to assess the value of both use and non-use values. The CVM relies on a hypothetical market to assess the value of non-marketed environmental services. Respondents are asked to state the maximum price they would be WTP either to obtain more of the services if desirable or WTA compensation if undesirable. In a CE, respondents are presented with different alternatives defined in terms of environmental attributes and are asked to select their preferred choice. The attributes are varied across alternatives. One of the attributes should involve a monetary measure if the researcher wishes to estimate the money value of attributes.

A number of weaknesses regarding the use of the CVM are cited, including hypothetical, embedding, payment vehicle, starting bid and yea-saying biases. Recently, there has been a general paradigm shift towards the use of CEs, which are generally perceived to overcome some of the limitations of the CVM. This is achieved through careful experimental design and also by the fact that WTP or WTA are inferred indirectly from the trade-offs between the monetary and non-monetary attributes. One major limitation of CEs is the general concern that respondents are presented with a lot of information to process. This has raised concerns as to whether or not respondents fully attend to all information provided to them in choice tasks and the possible implications for resultant WTP values. Presently, choice task complexity is one of the topical issues being addressed and methodologies to circumvent it are being tested.

Since their inception, travel cost RUMs have been the most attractive in valuing recreational use of natural resources involving multiple sites. Similarly, in recent decades CEs have gained widespread popularity over the CVM. In New Zealand there are a growing number of studies using CEs. However, a large number of freshwater non-market values remain unexplored. The use of TCM and HP is sparse. The application of travel cost RUMs to trout angling in the Rotorua Lakes is a novel approach in the New Zealand non-market valuation context.

CHAPTER FOUR

WATER QUALITY VALUATION USING TRAVEL COST RANDOM UTILITY MODELS

4.0 Introduction

The Rotorua Lakes are regarded as having "unique cultural, historical, social and economic value locally, regionally, nationally and internationally[31]". A key element of the recreational value of these lakes is the trout fishery which provides benefits to local residents, visitors, tourists and the local, regional and national economy. Eleven lakes offer a wide range of fishing opportunities. Many of the lakes have a world-class reputation and are within an hour's drive from Rotorua. Rainbow trout are most common in Rotorua's lakes, but there are also brown trout, tiger trout (Lake Rotoma only), and brook trout[32].

Despite their importance, falling water quality in some of the lakes is a major threat to the preservation of these values. Currently, initiatives to restore and preserve the lakes are underway. The main purpose of this chapter is to contribute to the ongoing research efforts in the Rotorua Lakes by assessing the monetary value to trout anglers of water quality improvements. The benefits derived may be used in cost-benefit analysis of the various pollution control mitigation measures, in addition to the other uses outlined in Chapter One.

Travel cost RUMs are employed to assess factors that influence anglers' fishing site choice decisions and welfare due to changes in water quality. Travel cost RUMs are increasingly being applied to assess the recreational value of multiple sites as discussed in Chapter Three (section 3.4). These models are popular because they use real data based upon observable individual behavioural patterns.

[31] http://www.hrc.co.nz/human-rights-and-the-treaty-of-waitangi/crown-tangata-whenua-engagement/te-arawa-rotorua-lakes-restoration-programme

[32] http://eastern.fishandgame.org.nz/

Additionally, these models enable the estimation of alternative patterns of substitution across recreational sites induced by policy changes (Phaneuf & Smith, 2004). These models, while widely applied elsewhere, are novel to the New Zealand non-market valuation recreational context.

This chapter is structured as follows. The next section explores some of the site attributes commonly applied in recreational demand models, followed by an outline of the fishing choice and lake characteristics data used in this application. The methodologies for assessing the determinants of angler's choice of lake for fishing are spelt out, followed by empirical results and discussion. Finally, welfare gains or losses emanating from different proposed water quality changes and possible lake closures are outlined.

4.1 Review of relevant site attributes in recreational random utility models

One of the basic premises of the random utility theory is that when presented with a number of alternatives, individuals will choose the one that gives them the highest level of satisfaction. The utility that an individual derives from the chosen alternative is a function of its attributes, individual characteristics and other unknown factors. This section explores some of the site attributes commonly employed in recreational fishing and other related studies by researchers in the field. In his review of past recreational fishing studies, Hunt (2010) classifies fishing site attributes into six distinct categories, namely cost of site access, fishing quality, environmental quality, facility development, regulations, and encounter levels. These attributes and other intervening factors are explored in the remainder of this section.

Cost of site access, generally referred to as travel cost, has two components: direct cost and the opportunity cost of time. Direct costs are the sum of all expenditures on market goods incurred on a recreational activity. They may include fuel expenses and other expenditures incurred while undertaking a recreational activity, including food and accommodation. Although there is a general consensus to include fuel expenses, the inclusion of other expenses depends on whether an analyst is modelling day or overnight recreational trips. For example, food expenses may be considered as incidental in most daytime recreational studies, while both food and lodging expenses may be pertinent if respondents stay overnight at the recreational site (Parsons, 2003). But then it becomes difficult to ascertain what quality of overnight stay is directly linked to the visit as opposed to visitors' preference for lodging, safety, etc.

Despite the need to incorporate the cost of time in recreational demand models, a consensus on the methodological framework for modelling the opportunity cost of time remains elusive (see Chapter Three). The predominant approach applied in the literature is the use of the wage rate (or some fraction of it) as a proxy for the value of travel time as originally advocated by Cesario (1976). However, the choice of which proportion of the wage rate to use remains subjective[33]. As a general guideline, there is a consensus among researchers in the field to consider 25% percent as the lower bound and 100% or full wage as the upper bound (Parsons, 2003). Other proposed approaches are outlined in Chapter Three.

On the other hand, the treatment of on-site time remains controversial (Parsons, 2003). The time spent on a recreational site is considered to be both a utility and a cost. Spending more time fishing on a site should enhance the value of fishing experience, but this time also has an opportunity cost. Due to the dual effect of on-site time, some studies assume that the time spent on site has net zero opportunity cost [34] (Phaneuf & Smith, 2004).

The presence of desirable fish species, their richness and abundance are major determinants of an angler's choice of fishing site, and are often referred to as 'fishing quality'. A number of proxies have been employed to measure fishing

[33] Consequently, some researchers do not include the value of time as a component of travel cost in their recreational demand models (e.g Fleming & Cook, 2008).

[34] Alternatively, the opportunity cost of on-site time is computed as a proportion of the wage rate and added to other costs.

quality. For instance, in studies by Morey *et al*. (1993), Parsons & Hauber (1998) and Hauber & Parsons (2000), fishing quality was approximated by the number of fish species in a water body. Other measures that have been used by researchers as proxies for fishing quality include whether water bodies have been artificially stocked (e.g. Montgomery & Needleman, 1997), the number of fish per square metre (e.g. Johnstone & Markandya, 2006), and the size of the water body (e.g. Egan *et al.*, 2009; Feather, 1994; Lupi & Feather, 1998; Parsons & Kealy, 1994). All these measures are generally considered to positively impact on anglers' choice of fishing site.

A number of other studies have employed anglers' reported catch rates as a measure of fishing quality (e.g. Adamowicz, 1994; Bockstael *et al.*, 1989; Kaoru, 1995; Parsons *et al.*, 2000; Schuhmann & Schwabe, 2004; Whitehead & Haab, 1999). However, Jakus *et al*. (1998) and Lupi & Feather (1998) caution against the use of aggregated catch rates pertaining to different fish species, arguing that such measures may not be useful in predicting individuals' fishing site choice[35]. Additionally, some studies have utilized fish size and expected size of fish as a measure of fishing quality (e.g. Adamowicz, 1994; Train, 1998; Watson *et al.*, 1994).

Furthermore, it has been demonstrated that environmental quality may impact upon individuals' choices of site for recreation either through aesthetics or landscape quality. Proxies for aesthetic quality in recreational fishing studies have included perceptual ratings (e.g. Peters *et al.*, 1995; Train, 1998) and the amount of forested land (e.g. Jones & Lupi, 1999; Tay *et al.*, 1996). These studies have found a positive link between aesthetics and landscape characteristics and recreational fishing site choice.

[35] In a study by Jakus *et al.* (1998), catch rates were found to be significant in the site choice models but were found to be insignificant in predicting site choice: this was attributed to the use of aggregated catch rates pertaining to different species.

Other studies have also indicated that water quality may influence an angler's choice of fishing site through its effects on aesthetics or health. Various indicators of water quality have been employed in previous studies including the use of perceptual ratings (e.g. Peters *et al.*, 1995; Watson *et al.*, 1994), fish consumption advisories (e.g. Jones & Lupi, 1999; Montgomery & Needleman, 1997), Environmental Protection Agency Standards (e.g. Hauber & Parsons, 2000), areas of concern and impacts (e.g. Hausman *et al.*, 1995; Jones & Lupi, 1999). Furthermore, since the 1990s researchers in the field of recreation have recognized the link between recreational demand and direct measures of water quality. Several recreational water-based studies including fishing have used physical, chemical or biological indicators of water quality including Secchi depth, turbidity, biological oxygen demand, dissolved oxygen, ammonia, phosphorous, nitrates, suspended solids, lead, copper, acidity, toxins, oil, and fecal coliform bacteria (e.g. Egan *et al.*, 2009; Johnstone & Markandya, 2006; Kaoru, 1995; Lupi & Feather, 1998). In general, results obtained from these studies indicate that individuals prefer recreational sites with better water quality.

Additionally, it has been acknowledged and validated empirically that facility development is an important determinant of site choice in recreational studies. Measures employed in the literature include the presence and number of boat ramps (e.g. Jakus & Shaw, 2003; Kaoru, 1995; Murdock, 2006; Parsons *et al.*, 1999b) and the availability of campground facilities (e.g. Adamowicz, 1994; Morey *et al.*, 2002; Peters *et al.*, 1995; Train, 1998). These studies generally reported a positive link between these measures and recreational fishing site choice[36].

Recreational site regulations are considered to be another site attribute that may impact upon recreational site choice and participation decisions, mainly through awareness and constraints. For instance, managers may influence recreational site choice through the provision of information that makes individuals aware of all the

[36] A number of other measures may be used as proxies for facility development at a recreational site such as availability of parking lots, toilets, number of access points to the site by road etc.

available recreational sites and whether a particular site is safe for recreation. Managers may also influence recreational site choice through the imposition of constraints such as access fees and other restrictions that may be considered appropriate. Particularly relevant to recreational fishing, regulatory instruments including catch and harvest limits may be used to ensure that fish stocks are not depleted. Hunt (2010) notes the sparse use of regulatory measures in revealed preference methods and attributes this to the general lack of variability in fishing regulations over the fishing sites. However, travel cost studies in which this attribute was incorporated, for instance, Scrogin *et al.* (2004) indicate that regulations may influence anglers' choice of fishing site in either direction.

Encounter levels with other recreational users are also considered to influence anglers' site choices and participation decisions. It is argued that anglers may experience a disutility at the site when encounter levels with other recreational users exceed a certain threshold (Martinson & Shelby, 1992). However, as noted by Hunt (2010), this attribute is usually not included in revealed preference choice models due to lack of data and also the high likelihood of this attribute being correlated with other important variables that are omitted from the model.

In addition to the site attributes described above, researchers have acknowledged the impact of other factors on fishing site choice and participation including past recreational experiences, place attachment and individual social demographic factors. For instance, past fishing experiences are considered to be the major source of the heterogeneity of preferences among anglers with regard to fish species and fishing sites (Perdue, 1993). Siemer & Brown (1994) argue that preferences over species and fishing sites may be a consequence of time and money that anglers invest to develop "appropriate skills to catch particular species of fish and also to learn the ins and outs of the fishing sites". It is also stated that place attachment may impact upon an individual's choice of site for recreation. Recreational social psychology literature asserts that the more individuals visit a particular recreational site, the more they attach emotional and symbolic meaning to the site and as a consequence they tend to visit the same recreational sites habitually (Bricker &

Kerstetter, 2000; Moore & Graefe, 1994; Williams *et al.*, 1992). Some studies of recreational fishing have demonstrated that place attachment is a major determinant of anglers' choices of fishing site (e.g. Adamowicz, 1994; Hailu *et al.*, 2005; Swait *et al.*, 2004)[37].

The relationship between an individual's likelihood of participating in recreational activities and demographic factors has been acknowledged since the 1930s (Manning, 1999). Evidence from previous studies indicates that demographic factors such as gender, age, residence, occupation, number of children, boat ownership and fishing experience may influence an individual's likelihood to participation in recreational activities. For instance males are more likely to participate in fishing than females (Montgomery & Needleman, 1997; Morey *et al.*, 2002) and the likelihood of being engaged in recreational fishing tends to increase with age (Lin *et al.*, 1996; Morey *et al.*, 2002; Morey *et al.*, 1993). Also, fishing participation tends to be higher in anglers who are unemployed (Hausman *et al.*, 1995; Montgomery & Needleman, 1997), have children (Montgomery & Needleman, 1997; Shaw & Ozog, 1999), fish with family members (Kaoru, 1995), own boats (Lin *et al.*, 1996; Shaw & Ozog, 1999) and are more experienced in fishing (Morey *et al.*, 2002; Morey *et al.*, 1993; Shaw & Ozog, 1999). Personal characteristics are generally considered to be a major source of preference heterogeneity over the choice of recreational sites that exist among individuals (Hunt, 2010)[38]. The next section outlines the fishing choice data and the lake characteristics employed in this application.

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[37] On the other hand, Hunt (2010) notes that only a few studies in recreational fishing have addressed the subject and attributes this to the extensive data requirements, which may require the collection of all past recreational trips to the site, possibly spanning several years.

[38] Another approach undertaken by researchers is to assume that the source of preference heterogeneity is unknown but can be accounted for in model estimation by allowing the parameter estimates pertaining to attributes to be random following a particular distribution as specified by the researcher. Alternatively, researchers may assume that heterogeneity among recreational users can be explained jointly by observable individual characteristics and sources which are assumed to be unknown to the researcher (Hunt, 2010).

4.2 The Rotorua Lakes fishing choice and lake attribute data

The fishing trip choice data used in this study was obtained from the 2007/2008 National Angling Survey carried out jointly by the National Institute of Water and Atmospheric Research Ltd (NIWA) and Fish and Game New Zealand (FGNZ). The main objectives of this angling survey were as follows.

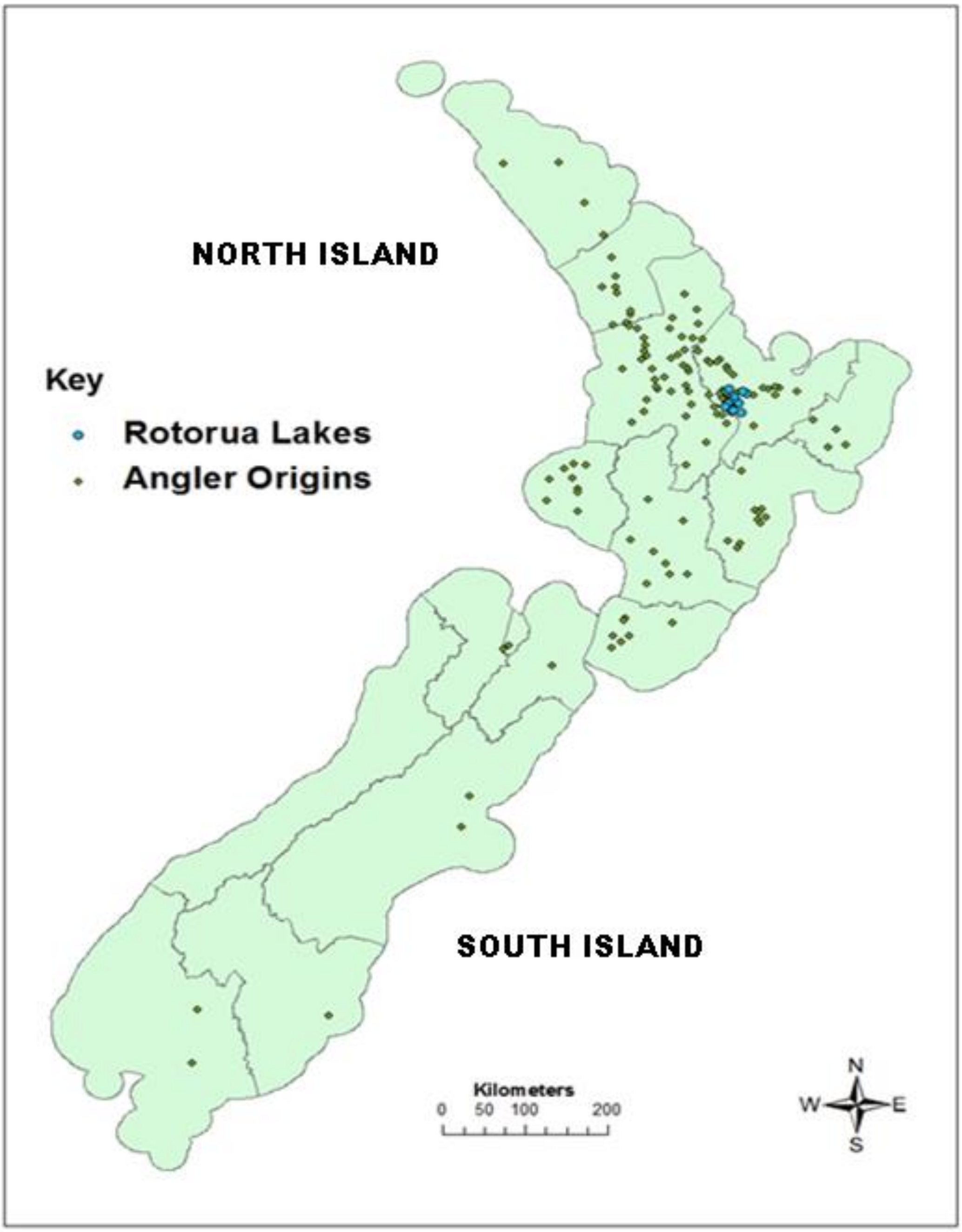
- To obtain consistent estimates of annual usage during the 2007/08 fishing season, by New Zealand resident anglers, for all lake and river fisheries managed by FGNZ.
- To develop and implement a simple email survey to collect corresponding usage data for overseas anglers visiting New Zealand, and to assess the utility of the resulting data.
- To develop a robust method for linking angling usage data to the River Environment Classification (REC) scheme (Unwin, 2009 p.5).

This was a telephone survey of a random sample of anglers drawn from records of fishing licence sales for the 2007/08 angling season, which spanned October, 2007 to September, 2008. The survey was stratified by FGNZ Region, licence type and fishing season. Appendix 5 (Figure A5.1) shows all the twelve FGNZ regions. Rotorua Lakes are within the Eastern Fish and Game Region. Licence types were divided into three strata, namely adult and family whole season licences, junior whole season licences and part-season licences. The licence dates of issue were used to partition sales into two monthly intervals from October-November, 2007 to August-September, 2008. This gave rise to six two-monthly interval strata for the whole fishing season. A random sample of 17,739 anglers was drawn from a population of 97,215 fishing licence holders. Out of this total, 84,875 were New Zealand resident anglers and 12,340 were overseas anglers. In line with the angling survey objectives, the main focus was on the number of days an angler spent fishing on a particular water body. Consequently, anglers were asked if they had fished during the specified two month period. Only anglers who said they had fished were asked to report the places they had fished and number of days spent on each water body (Unwin, 2009).

It should be highlighted that the survey did not collect some of the information that may be necessary for modelling recreational site choice, since such information was not in line with the National Angler study objectives. Notably, no information was collected on whether fishing trips undertaken were day or overnight trips and on whether fishing trips were single or multi-purpose. Furthermore, no information is available on whether or not anglers fished in more than one water body on an individual fishing trip. Also missing from the angling survey is information on the amount of time spent fishing on a particular lake. As noted by Phaneuf & Smith (2003), all this information has implications for the estimation of resources given up in order to access the recreational site. Despite the missing information, this survey has potential advantages due to its ability to provide comprehensive choice data at a national scale spanning the whole year. The ability of the data set to account for angler usage of lakes at different time periods in the fishing season makes this survey the most appropriate for this application, since it aligns well with varying water and fishing quality attributes across the fishing season.

The following assumptions were made to allow use of National Angling data to apply the TCM RUM: first, the main focus in this application is on single day fishing trips and individual level choice data. The simplifying assumption is made that each day of fishing reported represents a single-purpose day trip to a single fishing destination. Figure 4.1 below shows the distribution of anglers, in terms of their home origins across New Zealand, who had fished the Rotorua Lakes during the 2007/08 fishing season. This distribution constitutes all adult New Zealand fishing licence holders only.

Figure 4.1: Distribution of anglers to the Rotorua Lakes across New Zealand during the 2007/08 fishing season



Source: Unwin (2009)

The distribution indicates that the majority of anglers who fished in the Rotorua Lakes during the 2007/08 fishing season came from North Island[39]. Of these, the

[39] Based on FGNZ regulations, anglers are free to buy their fishing licences from any of the twelve Fish and Game Regional Councils and fish in any of these regions (Unwin, 2009).

majority came from regions which are closest to the lakes. Given this distribution of anglers in terms of their home origins, to assume that each day of fishing represents a single day trip would be unrealistic for anglers who lived further away from the Rotorua Lakes. Consequently, only anglers who lived within 241 km of the lakes are assumed to live close enough to be able to make a 'reasonable' fishing day trip[40]. However, it should be noted that even within the sample of anglers fulfilling this criterion, a proportion of anglers are likely to have stayed overnight, while others will have made multi-purpose trips or fished in more than one water body.

Given the requirement for individual level fishing choice data, only adult individual licence holders are considered[41]. A sample of 414 anglers fulfilled these two criteria and is employed in this application. The distribution of this sample of anglers in terms of their home origins within the FGNZ Regions and FGNZ licence issuing regional councils is shown in Appendix 5 (Table A5.1). From this table, it can be seen that out of the total sample of 414 anglers used in this application, 243 (59%) of anglers came from the Auckland/Waikato Fish and Game Region, 167 (40%) of anglers came from the Eastern Region and about 1% came from other FGNZ Regions[42]. All adult individual licence holders from the Eastern Region fell within the 241 km distance from the lakes criteria for a day trip. Only parts of the Auckland/Waikato Fish and Game Region fulfilled this criterion. The sample of

[40] In determining which recreational sites to include in choice sets, some researchers have used 150 miles as a benchmark for the maximum distance for a day trip (McConnell & Strand, 1994; Parsons & Kealy, 1992)

[41] Family licence holders are not included in the sample because no information was collected on the number of individuals in the family unit fishing together and how costs were shared. Furthermore, no information was collected on whether the fishing choice decisions were made jointly or not and travel cost RUMs are best suited for individual level data (Parsons, 2003).

[42] The table also indicates that the majority of anglers coming from home regions within Auckland/Waikato Fish and Game Region bought their fishing licences from the Eastern Region. The predominant movement of anglers from Auckland/Waikato Fish and Game Region to fish in the Eastern Region is also reported by Unwin (2009).

anglers used in this application and their home regions are depicted in Appendix 5 (Table A5.2).

It is evident that the procedure used to select the sample of anglers for this application is not a random sampling procedure. Hence, the distribution of fishing days across the lakes is not representative of the distribution of angler lake usage reported by the National Angling Survey as shown in Table 4.1 below. The lakes presented in this table represent the choice set used in this study[43].

[43] As already outlined in Chapter Two, Lake Rotokakahi is not open to the public for recreation and this reduces the choice set to eleven lakes.

Table 4.1: Estimated angler days for the 2007/08 National Angling Survey versus the number of fishing days for this study sample

Lake Name	National Angling Survey estimated angler-days ± 1 standard error	As a percentage	Number of fishing days for this study sample	As a percentage
Rotoiti	48070 ± 3710	33.9	673	29.4
Tarawera	34220 ± 3440	24.1	548	23.9
Rotorua	32000± 3200	22.5	583	25.4
Rotoma	11110 ± 2040	7.8	233	10.2
Okaitana	6290 ± 1070	4.4	95	4.1
Rerewhakaaitu	3830 ± 800	2.7	99	4.3
Rotoehu	3720 ± 1210	2.6	33	1.4
Okareka	2040 ± 530	1.4	19	0.8
Tikitapu	370 ± 140	0.3	3	0.1
Okaro	260 ± 170	0.2	5	0.2
Rotomahana	70 ± 50	0.0	1	0.0
Total	141980	100	2292	100

Source: Unwin (2009)

The distribution of fishing days across lakes estimated from the National Angling Survey is regarded as a benchmark for the true population distribution since that survey was designed following random sampling procedures. From Table 4.1 above, the sample employed in this application either over-states or under-states the true distribution of fishing days across the lakes for some lakes (e.g. Rerewhakaaitu). To account for under-sampling and over-sampling, in our choice analyses the choice variable is re-weighted to correspond to angling distributions reported by the National Angling Survey using choice based weighting procedures outlined by Hensher *et al.* (2005).

Altogether, the sample of 414 anglers used in this study reported a total of 2,292 fishing days in the Rotorua Lakes for the 2007/08 fishing season. Each fishing day is assumed to be a single choice occasion. In line with the home origins of anglers in the study sample, the appropriate study population consists of all anglers who bought fishing licences during the 2007/08 fishing season whose home origins are within the Eastern Region and Auckland/Waikato Fish and Game Councils. Thus the study population of interest is equal to 21,883 anglers (Unwin, 2009). This figure excludes junior licence holders since the analysis is based on adult licence holders only. The remainder of this section outlines some of the lake attributes employed in this application.

Drawing on the literature and local expert opinion and data availability, the following lake attributes are used in this application: cost of site access, measures of environmental quality, fishing quality and facility development. The cost of lake access is a key variable in all travel cost recreational demand models. In this application the cost of lake access consists of fuel expenses and the opportunity cost of travel time. As already stated above, no information was collected on the expenses incurred during fishing trips or regarding on-site time. Consequently, these aspects of costs are not included in the calculation of the cost of lake access. As highlighted by Parsons (2003), most of the expenses incurred for day trips may be regarded as incidental and are often not included in the cost of recreational site access. The opportunity cost of on-site time is considered to be endogenous and is not included in the cost of site access (Phaneuf & Smith, 2004).

As a measure of environmental quality, objective measures of water quality are used. In particular, water clarity is used because it is generally considered to align well with people's perceptions of water quality. In addition to water quality, the amount of forested land is another environmental attribute which is employed in this application as a measure of aesthetics and landscape quality.

Related to water quality, this chapter also explores the impact of health warnings on recreational fishing choice and participation. Historical health warning data between 2001 and 2007 is used. Some of the Rotorua Lakes, including Rotorua, Rotoiti, Tarawera, Rotoehu and Okaro have had health warnings put in place due to algal blooms during this period. Data on water quality and health warnings were obtained from the Regional Council (Environment Bay of Plenty).

Data on catch rates were available for only five of the lakes in the choice set, so the yearly average weight of fish caught in each lake is used as a proxy for fishing quality. Data on weight of fish were obtained from the Eastern Region Fish and Game Council. The Fish and Game council monitors fishing quality through yearly summer and winter creel surveys for Lakes Rotorua, Rotoiti, Tarawera and Okataina. During these surveys, the quality of fish caught by anglers is assessed using indicators such as weight and length. Monitoring the growth of fish in these lakes and other Rotorua Lakes is also accomplished through a data watch tagging programme in which most of the trout that are liberated into the lakes are tagged with a plastic tag. Once these tagged fish are caught, anglers return the tag along with the fish details.

This study also uses proxies for facility development, which include the number of boat ramps and number of key access points to the lakes. In addition, the study also explores the impact of the level of urban development around the lakes on fishing site choice decisions. An outline of how they enter the utility function is given in the subsequent section.

4.3 Study questions and hypotheses

The main objective of the fishing site choice model described below is to assess the extent to which water quality impacts upon anglers' choice of lake for fishing. In addition to water quality, the extent to which other lake attributes influence anglers' choice of lake for fishing is investigated. In particular, the following questions are addressed.

1. Does water quality influence anglers' choice of lake for fishing?
2. Which other site attributes influence anglers' choice of lake for fishing?
3. What value do anglers place on water quality improvements?
4. What is the recreational trout angling value of the Rotorua Lakes?

In the next section the methodologies used to address these study questions are outlined.

4.4 Random utility fishing site choice model

In this section, the random utility site choice model for lake fishing recreation is developed. To specify the model, it is assumed that on each fishing trip, an angler is faced with a choice of J possible lakes to visit. Each angler is assumed to choose the lake that gives them the highest level of expected utility. Furthermore, it is assumed that while each angler knows the utility that is derived from the chosen lake, this utility remains unknown from the researcher's perspective. Hence the utility that an individual would derive from a chosen alternative consists of the deterministic and stochastic components from the researcher's perspective (Ben-Akiva & Lerman, 1985). Total utility that an angler would derive from choosing lake j is specified as;

$$U_{nj} = V_{nj} + \varepsilon_{nj} \tag{1}$$

V_{nj} is referred to as the representative utility and represents the systematic part of utility that can be identified by the researcher. ε_{nj} is the stochastic component of utility that captures all the unobserved factors that may have an effect on the angler's utility but are not accounted for in V_{nj}[44]. In this application, it is assumed that the utility an angler derives from participating in recreational fishing at lake j is a linear combination of travel costs and lake attributes[45]. The conditional indirect utility function for angler n from fishing at lake j can be specified as follows:

$$U_{nj} = V_{nj}\left(C_{nj},\ Q_j\right) + \varepsilon_{nj} \tag{2}$$

C_{nj} is the implicit price of accessing lake j for angler n, which in this study, includes fuel expenses and the opportunity cost of travel time. The implicit price of lake access varies over individuals and lakes. Q_j denotes a vector of lake attributes including water quality and fishing quality.

A more detailed specification is given in equation (3), showing how each of the variables described above enters the conditional indirect utility function in a fishing choice model.

$$\mathrm{U_{nj}} = \delta C_{nj} + \beta SD_j + \varphi FW_j + \gamma LZ_j + \omega FDV_j + \tau URBN_j + \sigma FR_j + \vartheta DPTH_j + \rho HRN_j + \varepsilon_{nj} \tag{3}$$

[44] The random error term ε_{nj} accounts for unobserved individual characteristics and/or attributes of sites. Some of the site attributes may be known to the researcher but are not included in the utility specification.

[45] Under the random utility modelling framework, each choice occasion is assumed to be a separate process i.e. other consumption decisions can only affect the choice decisions indirectly through income which is available to an individual when the recreational choice decision is being made.

Where C_{nj} is the implicit price of lake access. The vector of lake attributes, $Q_j = [SD_j,\ FW_j,\ LZ_j,\ FDV_j,\ URBN_j,\ FR_j\ ,\ DPTH_j,\ HRN_j\ ]$ where: SD_j is Secchi Depth, FW_j is weight of fish, LZ_j is lake size, FDV_j is a proxy for facility development, $URBN_j$ is the amount of urban development around the lakes, FR_j is the amount of forested land, $DPTH_j$ is lake depth and HRN_j is health warning, while $\delta, \beta, \varphi, \gamma, \omega, \tau, \sigma, \vartheta, \rho$ are parameters to be estimated.

Specifically, the following null and alternative hypotheses are tested in line with study questions 1 and 2 outlined in section 4.3.

$$H_o : \delta = 0;\ \beta = 0;\ \varphi = 0;\ \gamma = 0; \omega = 0;\ \tau = 0; \sigma = 0; \vartheta = 0;\ \rho = 0 \qquad (4)$$

$$H_A: \delta \neq 0;\ \beta \neq 0;\ \varphi \neq 0;\ \ \gamma \neq 0; \omega \neq 0;\ \tau \neq 0; \sigma \neq 0; \vartheta \neq 0;\ \rho \neq 0 \qquad (5)$$

More detailed description of the variables entering the utility function and how they are measured is presented below.

The cost of lake access (C_{nj}) is computed following the standard procedure proposed by Cesario (1976) using the following formulation:

$$C_{nj} = \frac{(Distance_{nj})(Cost\ of\ fuel\ per\ km)}{(Group\ Size_i)} + (\%Wage)\left(\frac{Income_n}{Total\ annual\ working\ Hrs}\right) \qquad (6)$$

where $Distance_j$ is the round-trip road distance from an angler's home to the Rotorua Lake j. The zip code was used to locate the angler's home and distances between the zip code area centroids and fishing site centroids were measured using GIS software[46]. The cost of fuel was estimated at NZ$0.19 per kilometre for all anglers. This was calculated based on the average prices of fuel in the last quarter of 2007 and quarters one to three of 2008, obtained from the Ministry of Economic Development website (MED, 2010).

[46] Both road and straight line distances were computed, but road distances were more preferred.

Group size is the number of people with whom travel expenses were shared. In this study, no information was collected on this aspect and therefore it is assumed that travel expenses were not shared ($Group\ Size_i = 1$). Data obtained after thesis submission for the Rangitata River recreational fishing (Kerr & Greer, 2004) and instream water values for Rakaia and Waimakariri Rivers studies (Kerr *et al.*, 2004) showed average angler group size of at least 2. If this holds for the sample of anglers used in this thesis, then resource values estimated here are likely to be inflated.

$\%Wage$ is the percentage of wage rate applied to value the opportunity cost of recreational time: it is usually between ¼ and ½ and in this application 25% of the average wage rate is applied[47]. $Income_n$ is the total annual income of an angler. Information on income was not collected in the National Angling Survey and as a proxy estimated median income for each region the angler came from is used.[48] The estimated median income is from the 2006 census data by Statistics New Zealand. The estimated median income is divided by the average total working hours per year. The study assumes total working hours of 2000 hours per year following the conventional standards. $Time_j$ is the estimated round-trip travel time in hours to lake j corresponding to the estimated road distances from the angler's residential location to Rotorua Lake j[49] The coefficient of the cost of lake access, δ is expected to be negative. In general anglers would prefer lakes which are closer to their homes, since they would incur less travel costs both in money and time.

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[47] The choice of this percentage wage rate is arbitrary, but is considered to be reasonable since it falls within the generally accepted range.

[48] Freshwater sport fishing tend to be dominated by wealthier sectors of society due to high expenditures on items such as travel, boat running, accommodation, charters and guides, food and fishing equipment (Cowx, 2002). Consequently, the cost of lake access is likely to be underestimated and WTP estimates are likely to be under-valued.

[49] Travel time was estimated assuming a travel speed of 60km/hour within the city centre and 80km/hour outside the city centre. This was done to account for the fact that anglers traveling the same distance could face different travel times depending on where they live.

With regards to the other lake attributes, Secchi disc depth (SD_j) is a measure of water clarity and is measured in metres. Generally, the higher the value of SD, the better the water visibility and hence quality. Therefore, the coefficient of SD, β is expected to be positive.

FW_j is the weight of fish in kilograms. Generally, the coefficient of this variable, $\varphi,$ is expected to be positive, since bigger fish are preferred to smaller ones.

It is also expected that the size of the lake (LZ_j) measured in square kilometres can influence the choice of which lake to fish from. Generally, bigger lakes may be expected to contain a large number of fish and fish species, and also to be preferred by anglers with bigger boats. However, it is difficult to predict the expected sign of the utility weight pertaining to this variable beforehand, since other intervening factors, such as how well the lake is stocked with trout, might play an important role.

It is also anticipated that the angler's utility derived from fishing at lake j can be affected by facility development, including the number of access points and boat ramps. The coefficients for these variables are expected to be positive, since lakes with more of these facilities would be more convenient to anglers.

Furthermore, it is expected that the amount of land around the lakes devoted to urban development, $URBN_j$, measured as a percentage, can impact upon angler's utility. However, the sign of the coefficient for this variable cannot be determined *a priori*, since some anglers may enjoy the convenience of urban surroundings while others might prefer more natural surroundings.

The amount of forested land (FR_j), measured as a percentage, is included in the utility specification as a measure of the aesthetic beauty of the natural surroundings of the lakes. A positive link between the amount of forested land and fishing site choice is anticipated and hence σ is expected to be positive.

It is also anticipated that the angler's utility derived from fishing at lake j can be affected by lake depth $(DPTH_j)$, measured in metres (m). Although lakes that are greater in depth may be more challenging to fish from than shallower ones, the sign of the coefficient for this attribute cannot be determined *a priori*. This emanates from the fact that anglers tend to seek different adventures (Hunt, 2010), consequently, the level of challenge the lake presents could be considered as part of the adventure by some anglers.

In addition, health warnings due to algal blooms may affect the aesthetic quality of the lakes and might impact negatively on anglers' choice of fishing destination and participation. Therefore, ρ is expected to be negative.

Estimation

Estimation of the parameters in equation (3) requires the decomposition of utility into its deterministic and stochastic parts. Following the standard discrete choice approach, the utility that angler n derives from fishing at lake j on any choice occasion is specified as follows:

$$\mathrm{U_{nj}} = \delta C_{nj} + \beta SD_j + \varphi FW_j + \gamma LZ_j + \omega FDV_j + \tau URBN_j + \sigma FR_j + \vartheta DPTH_j + \rho HRN_j + \varepsilon_{nj} \qquad (7)$$

In estimation, the unobserved effects are accounted for through the inclusion of alternative specific constants (ASCs) in the representative utility, V_{nj} (Train, 2002). Although the inclusion of ASCs is plausible both from the econometric and behavioural perspective, in recreational discrete choice models, these constants pose two main challenges. First, the unobserved site characteristics may be correlated with the site attributes included in the model and this may lead to biased parameter estimates and therefore, biased welfare estimates. Second, including a full set of alternative specific constants implies that parameters pertaining to site attributes which vary only across sites and not across individuals or over time cannot be estimated because of identification problems[50] (Murdock, 2006).

To overcome these limitations, the standard procedure has been to either exclude the alternative specific constants or partially account for unobserved factors by including common constants in a subset of alternatives. Under the latter approach, the common practice is to include a common constant for sites that are either different from the rest or sites considered to be close substitutes (e.g. Jakus *et al.*, 1997; Parsons *et al.*, 1999b; Parsons & Needelman, 1992). However, partially accounting for unobserved factors may lead to loss of information which may lead to loss of efficiency in the estimation of parameters. The intuition behind this is summarized by Murdock (2006, p.4) as follows.

> Including alternative specific constants leaves these parameters to be estimated from variation in the observed site attributes for all sites excluding those with alternative specific constants. Including group specific constants leaves parameters to be estimated from variation within groups and not across groups.

As a consequence, Train *et al.* (2000) argue that alternative specific constants in recreational RUMs should be used thoughtfully. Berry (1994) developed a modelling framework for handling unobserved product characteristics in the analysis of discrete choice models of product demand for differentiated goods. This modelling framework was further applied by Berry, Levinsohn and Pakes (1995) (also known as the BLP approach) in the estimation of a discrete choice model of automobile demand. Building on the work by Berry (1994) and the BLP approach, Murdock (2006) proposed procedures to address unobserved characteristics in recreational demand which involve the use of a two-stage process. In the first stage, a discrete choice model with a full set of alternative specific constants and variables

[50] The identification problem will arise because the dummy variables for the alternative specific constants will "...capture all variation across alternatives, which leave no variation for simultaneous estimation of parameters on any variables that only vary across alternatives" (Murdock, 2006, P.3).

which vary across sites and individuals is estimated. The second stage involves the use of the ordinary least squares method (OLS) in which the alternative specific constants from the first stage are regressed against the site attributes that vary only across sites and not across individuals. However, Murdock's approach can only be applied in recreational studies with a very large number of recreational sites and is therefore, not suited to this study[51].

This application uses the standard procedure in which ASCs are either excluded or partially accounted for through the use of common constants in a subset of alternatives. The model utilized in this chapter is specified as follows and the variables are as defined in equation (3).

$$V_{nj} = \delta C_{nj} + \beta SD_j + \varphi FW_j + \gamma LZ_j + \omega FDV_j + \tau URBN_j + \sigma FR_j + \vartheta DPTH_j + \rho HRN_j \qquad (8)$$

In order to estimate the parameters in the representative utility in equation (8) some assumptions have to be made regarding the distribution of the random error term (ε_{nj}). Following McFadden (1974), it is assumed that the random components of utility, ε_{nj} are independently and identically distributed (IID) type I extreme values, giving rise to the multinomial logit model[52]. This model has been found to be the most attractive because the choice probabilities take a closed form. Following McFadden (1974), the probability (P_{nj}) that angler n chooses to fish at lake j on a given day can be expressed as:

$$P_{nj} = \frac{e^{V_{nj}}}{\sum_{j=1}^{J} e^{V_{nj}}} \qquad (9)$$

---

[51] Using Murdock's (2006) two-stage process in recreational studies with few sites results in the second stage regression having too few observations to estimate reliable parameter estimates.

[52] In deriving this model, the scale parameter is normalized to be equal to one. Furthermore, since angler characteristics are not included, the model used in this application is referred to as the conditional logit model.

The closed-form property of the logit probabilities makes the estimation of the parameters in the representative utility relatively simple without requiring the use of maximum-likelihood procedures. Instead, the likelihood function derived from the choice probability is used to find the value of the parameter estimates in the representative utility. Following Train (2003), assuming angler n is observed to have chosen lake j on a fishing occasion, the likelihood function that angler n chooses the lake that he or she was actually observed to choose can be expressed as:

$$L_n(\mu) = \prod_{j=1}^{J} (P_{nj})^{y_{nj}} \tag{10}$$

Where $y_{nj} = 1$ if angler n chooses lake j and zero otherwise. Since $y_{nj} = 0$ for all lakes not chosen by an angler, $L_n(\mu) = P_{nj}$ which is the probability of the lake actually chosen by an individual on a single choice occasion and hence considered to be the contribution that each angler makes to the likelihood function. Assuming further that the fishing site choices made by different anglers are independent of each other, the probability of all anglers in the sample choosing the lake that they were observed to have actually chosen is equal to the product of each angler's likelihood contribution (Train, 2003) as specified below:

$$L_n(\mu) = \prod_{n=1}^{N} \prod_{j=1}^{J} (P_{nj})^{y_{nj}} \tag{11}$$

Expressing the above likelihood function in logarithmic form, the log-likelihood function is:

$$\ln(L_n(\mu)) = \sum_{n=1}^{N} \sum_{j=1}^{J} y_{nj}\, lnP_{nj} \tag{12}$$

The estimators (μ) include the cost of site access parameter, δ; water clarity parameter, β, the weight of fish parameter, φ, the size of lake parameter, γ, the facility development parameter, ω, the urban development parameter, τ, the amount of forested land parameter, σ, the lake depth parameter, ϑ and ρ the health warning parameter. These parameters are estimated by maximum likelihood using the standard routines implemented in Nlogit 4.0.

4.5 Estimation results and discussion

This section addresses two main study questions spelled out in the previous chapter. First, it provides an empirical investigation of how water quality impacts on anglers' choices of lakes for fishing. Second, the effects of other lake characteristics on anglers' choices of which lakes to visit are explored. The fishing choice data used in this application is an unbalanced panel data set with a large proportion of anglers reporting visiting the lakes only once over the fishing season[53]. The dependent variable is choice which is equal to 1 if the lake is chosen by an angler and 0 otherwise.

Table 4.2 presents the summary statistics for the lake attributes used in estimation of equation (8).

[53] However, this does not mean that such anglers visited the Rotorua lakes only once during the year, but it may simply imply that they were not included in the other sub-samples, since re-sampling was done at two-monthly intervals.

Table 4.2: Summary statistics for the Rotorua Lakes attributes

Variable	Mean	Std.Dev.	Minimum	Maximum
Weight of fish (kg) (2006/2007 fishing Season)	1.66	0.35	1.2	2.3
Weight of fish (kg) (2007/2008 fishing season)	1.54	0.23	1.2	2
Secchi depth (metres)	6.39	3.36	2.3	13.3
Lake size (square km)	18.71	23.31	0.31	80.6
Number of boat ramps	2.27	2.00	1	7
Number of access points	2.36	2.06	0	7
Depth (metres)	29.33	19.68	7	60
Urban development (% of lake catchment area)	1.41	2.27	0	8.1
Amount of forested land (% of lake catchment area)	56.82	26.53	6	94

Data on water quality were obtained from Environment Bay of Plenty
Data on the weight of fish were obtained from Eastern Region Fish and Game Council
Data on catchment area, lake size, depth and forest cover were obtained from Allan *et al.* (2007) and Burns *et al.*(2005)
Data on boat ramps, key lake access points and toilets were obtained from Environment Bay of Plenty Lakes guide for recreational users

In general there is a considerable variability across lakes in terms of water clarity, with the lowest average SD readings of 2.3 metres at Lake Okaro and a maximum of 13.3 metres for Lake Rotoma. A detailed description of water quality in the Rotorua Lakes is presented in Chapter Two. Lake attributes which display the highest variability include lake size, depth and the proportion of forested land in the lake catchment area. Lake Rotorua is the largest of all the Rotorua lakes, while Okaro is the smallest. The deepest lakes are Rotoiti and Rotomahana with a depth of 60 metres. Lake Tikitapu has the highest forest cover, covering 94% of its catchment area.

The estimated results for the conditional logit models are presented in Table 4.3. The choice variable is regressed against the cost of lake access and lake attributes presented in Table 4.2 and health warning. The latter enters the utility specification as a dummy variable which takes the value of 1 if health warnings due to algal blooms were issued between 2001 and 2007 fishing seasons and zero otherwise. All other lake attributes entered the utility specifications assuming a linear form except for the size of lake variable, in which the log-linear specification was used to account for diminishing marginal utility to size. Boat ramps and number of access points to the lakes were found to be highly collinear, and so could not be included in the same utility specification. Instead, the number of lake access points is used as a proxy for facility development.

Table 4.3: Estimated results from the conditional logit model

	Model 1 (2007/08 weight of fish)			Model 2 (2006/07 weight of fish)		
Variable	*Coefficient*	*Std Error*	*|t-value|*	*Coefficient*	*Std Error*	*|t-value|*
TRAVEL COST	-0.072***	0.007	10.79	-0.072***	0.007	10.99
WATER CLARITY	0.190***	0.017	11.44	0.185***	0.019	9.57
WEIGHT OF FISH	1.376***	0.291	4.70	0.630***	0.212	2.98
LOG OF LAKE SIZE	3.407***	0.279	12.20	3.651***	0.371	9.84
FACILITY DEVELOPMENT	0.356***	0.025	14.44	0.326***	0.024	13.81
URBAN	-0.352***	0.036	9.75	-0.389***	0.049	7.90
FOREST	0.015***	0.002	7.50	0.015***	0.002	6.68
LAKE DEPTH	-0.056***	0.007	8.52	-0.050***	0.007	7.35
HEALTH WARNING	-0.606***	0.141	4.29	-0.640***	0.146	4.37
Summary Statistics						
Log-Likelihood	-3843.65			-3848.60		
McFadden R-Squared	0.274			0.273		

Note: The dependent variable is choice which is equal to 1 if the lake is chosen by an angler and 0 otherwise.

Two conditional logit models were estimated. In Model 1 the 2007/08 yearly average weight of fish for each lake was used. On the other hand, Model 2 employed the 2006/07 yearly average weight of fish to account for the fact that anglers' current fishing decisions may be impacted more by previous fishing quality.

In terms of the model fit as measured by the log-likelihood, Model 1, in which the 2007/08 yearly average weight of fish data was used, performed slightly better than Model 2 by about 5 log-likelihood points. The explanatory power as indicated by the McFadden R-squared is the same between the two models and indicates an overall good model fit to the data. Furthermore, the utility weights are consistent between the two models except for the yearly average weight of fish variable. Appendix 6 (Table A6.1) shows the summary statistics for the yearly average weight of fish variable between the two time periods for individual lakes in the choice set. The yearly average weight of fish is generally consistent between the two periods, except for Lakes Rotorua and Rerewhakaaitu which reported higher average weight of fish during the 2007/08 fishing season by about 0.5 kilograms.

The COST variable is negative and highly significant, indicating that lakes that were closer to anglers' residences were generally preferred. The WATER CLARITY attribute is positive and highly significant as expected, indicating that in general anglers favoured lakes with better water quality. The yearly average WEIGHT OF FISH attribute is also positive and highly significant as expected, indicating that generally anglers preferred lakes with bigger fish.

The size of lake variable (LOG OF LAKE SIZE) is positive and highly significant indicating that generally bigger lakes were preferred by anglers. The FACILITY DEVELOPMENT variable is positive, as expected, and significant at the 1% level, signifying that generally anglers preferred lakes with more recreational facilities such as number of access roads and boat ramps. Additionally, results show that in general anglers preferred lakes surrounded by more FOREST cover. On the other hand, the presence of URBAN development around the lakes had a negative effect on fishing site choice probability. Generally, lakes that are greater in depth (LAKE DEPTH) are less preferred by anglers and HEALTH WARNING impacted negatively on anglers' choices of lakes for fishing.

4.6 Policy simulations and welfare measures

In this section the last two study questions outlined in section 4.3 are addressed. First, this section attempts to answer the question: what value do anglers place on water quality improvements? The second question addressed is: what is the trout angling value of each of the Rotorua Lakes? In order to assess the value that anglers place on water quality improvements, the welfare gain each angler is expected to receive from fishing at each lake under a hypothetical improved condition is compared to the corresponding welfare in the baseline conditions.

The procedures for estimating the welfare gain under hypothetical water quality conditions are outlined as follows: Assuming that angler n has the property right to the initial endowment i.e. the right to remain in the pre-policy water quality level, the change in welfare as a result of a change in water quality at a site (compensating variation, CV) can be defined as the amount of money an angler is willing to pay or to accept that would leave the angler as well off as before a change (Hicks, 1939a; Kaldor, 1939).

To obtain the compensating variation, suppose all the lake attributes are denoted by Q_j such that $q^{w^0} \in Q_j$ and $q^s \in Q_j$, where q^{w^0} is the baseline water quality at lake j and q^s denotes other lake attributes at lake j excluding water quality[54]. Following Small & Rosen (1981) and Hanemann (1982) the Hicksian welfare measure (CV) in discrete choice models for a change in water quality at lake j from q^{w^0} to q^{w^1} can be calculated from unconditional indirect utility functions using the following formulation;

[54] In this formulation, the time subscripts for attributes that vary across the fishing season are suppressed.

$$V_j\left(Y - C_j, Q_j^{w^0}\right) + \varepsilon_j = V_j\left(Y - C_j - CV, Q_j^{w^1}\right) + \varepsilon_j \quad (13)$$

Where $Q_j^{w^0}$ is a vector of lake attributes with water quality at the baseline (q^{w^0}) and $Q_j^{w^1}$ is a vector of lake attributes with water quality at a changed level (q^{w^1}) following a policy change. C_j is the vector of prices, in this case fuel expenses and opportunity cost of time, Y is income with constant marginal utility over alternatives (sites), CV is the compensating variation that equates utility after the hypothesized change in water quality to utility before the hypothesized water quality changes and ε_j is the stochastic component of utility.

The presence of a stochastic component of utility in the formulation of per choice occasion welfare estimate from site j entails that the compensating variation is not deterministic on the part of the researcher. As a result the researcher can only come up with the expectation of the compensating variation conditional on the distribution of the stochastic component of utility ε_i. Hanemann (1982) has shown that when the unconditional indirect utility is assumed to be linear for income and the stochastic component of utility is assumed to follow type I extreme value distribution, the expected per trip welfare measure (CV) can be calculated using the log-sum formula below[55].

$$CV = \frac{ln\left[\sum_{j=1}^{J} e^{V_j\left(Q_j^{w^0}\right)}\right] - ln\left[\sum_{j=1}^{J} e^{V_j\left(Q_j^{w^1}\right)}\right]}{\alpha_m} \quad (14)$$

In equation 14, V_j represents the deterministic component of utility evaluated based on the estimated coefficients of the indirect utility specification in equation (8) and α_m is the marginal utility of income, which is equal to the negative of the cost of lake access coefficient. The expressions $ln\left(\sum_{j=1}^{J} e^{V_j\left(Q_j^{w^0}\right)}\right)$ and

[55] Since the conditional indirect utility functions are assumed to be linear in income, income drops out of the log-sum formula and therefore is not included in the calculations of the expected compensating variation.

$ln\left(\sum_{j=1}^{J} e^{V_j\left(Q_j^{w^1}\right)}\right)$ are referred to as the inclusive value indices pertaining to the baseline water quality and post-policy water quality, respectively. According to Loomis (1995 p.60) the inclusive value index "represents the net utility (benefits of site visit directly related to site quality, minus the travel costs) from the site being available on any choice occasion[...]".

In line with the third study question pertaining to the assessment of the value that anglers place on water quality improvements, the main hypotheses to be tested in this section are:

$$H_o: ln\left(\sum_{j=1}^{J} e^{V_j\left(Q_j^{w^1}\right)}\right) - ln\left(\sum_{j=1}^{J} e^{V_j\left(Q_j^{w^0}\right)}\right) = 0 \qquad (15a)$$

Against

$$H_A: ln\left(\sum_{j=1}^{J} e^{V_j\left(Q_j^{w^1}\right)}\right) - ln\left(\sum_{j=1}^{J} e^{V_j\left(Q_j^{w^0}\right)}\right) > 0 \qquad (15b)$$

Similarly, the expected per trip welfare loss due to hypothesized lake closure is assessed using the following expression:

$$CV_{j-1} = \frac{ln\left[\sum_{j=1}^{J} e^{V_j\left(Q_j^{w^1}\right)}\right] - ln\left[\sum_{j=1}^{J-1} e^{V_j\left(Q_j^{w^1}\right)}\right]}{\alpha_m} \qquad (16)$$

where CV_{j-1} is the per trip welfare loss per angler due to hypothesized lake closure. This welfare loss will be used as a proxy for the recreational angling value for each of the lakes in the choice set. Lake closure may occur for a number of reasons including an effective water quality below the recommended recreational water quality guidelines. Some of the Rotorua Lakes, or just parts of the lakes, are closed for some periods during the fishing season due to excessive algal blooms (see Chapter Two). Assessing the recreational loss due to lake closure might therefore have important policy implications in this respect.

As highlighted in Chapter Three (section 3.4) the travel cost RUMs are of limited use for estimating seasonal welfare estimates that account for changes in recreational participation levels induced by changes at one or more sites. To overcome this limitation, researchers have suggested linking the travel cost RUMs to participation models to account for both the substitution effect from the site choice model and changes in the number of trips through the participation model (Parsons *et al.*, 1999a). Social demographic data to enable the estimation of participation models is not available and hence the welfare estimates derived are to be considered conservative and a lower bound on the real values. The remainder of the section explores the impact of hypothetical changes in water clarity on the probability of site choice and anglers' welfare. This is followed by an assessment of anglers' welfare loss due to lake closure.

This study hypothesizes a 1 and 3 metre increase in water clarity for all the eutrophic lakes (Rotorua, Rotoehu, Okaro)[56] and mesotrophic lakes (Rotoiti, Okareka, Rotomahana, Rerewhakaaitu), concurrently and also individually. Of particular interest are the changes in the probability of fishing site choice, depicting the redistribution of anglers across the lakes following hypothesized changes in water clarity. This is demonstrated in Table 4.4 below assuming an increase in water clarity of 3 metres in each of the lakes with poor and average water quality for Model 1.

[56] Lake Okaro, although included in the eutrophic lake category, is actually supertrophic.

Table 4.4: Percentage changes in the probability of lake visit for a 3 metre rise in water clarity

	Rotoiti	Rotorua	Rerewhakaaitu	Rotoehu	Okareka	Rotomahana	Okaro
Lakes with poor and average water quality							
Rotoiti	13.429	-4.687	-0.809	-0.650	-0.285	-0.088	-0.001
Rotorua	-4.407	11.109	-0.552	-0.397	-0.199	-0.061	-0.001
Rerewhakaaitu	-0.668	-0.484	2.424	-0.061	-0.032	-0.011	0.000
Rotoehu	-0.527	-0.347	-0.060	1.854	-0.021	-0.006	0.000
Okareka	-0.231	-0.171	-0.031	-0.021	0.875	-0.003	0.000
Rotomahana	-0.071	-0.052	-0.011	-0.006	-0.003	0.275	0.000
Okaro	-0.001	-0.001	0.000	0.000	0.000	0.000	0.004
Lakes with good water quality							
Tarawera	-4.769	-3.515	-0.631	-0.432	-0.221	-0.069	-0.001
Rotoma	-1.811	-1.191	-0.215	-0.195	-0.073	-0.023	0.000
Okataina	-0.823	-0.572	-0.099	-0.082	-0.035	-0.011	0.000
Tikitapu	-0.121	-0.090	-0.016	-0.011	-0.006	-0.002	0.000

Note: The dark shading denotes own probability of site visit

For an increase in water clarity in each lake individually, the lakes enjoying the biggest increase in own probability of site visit are Rotoiti and Rotorua with predicted increases in site visits of 13% and 11%, respectively. Lakes Rotomahana and Okaro have the least own probability of site visit of 0.3% and 0.004%, respectively. The model further predicts most anglers redistributing their fishing effort from Lakes Tarawera and Rotoma following the hypothesized improvement in water quality in other lakes. Presently, these two lakes are among those with the best water quality.

The monetary values measured in terms of the compensating surplus (CS) for all concurrent hypothesized changes in water quality improvements from Model 1 are presented in Table 4.5 below. The table presents the compensating surplus per choice, per angler, for the whole sample and population for the entire 2007/08 fishing season. The values in parentheses are the 95% confidence intervals for the mean compensating surplus.

Confidence intervals for the compensating surplus were calculated using a simulation method proposed by Krinsky and Robb (1986). The simulated approach made use of the estimates of the parameter vector, denoted by β and the estimated asymptotic variance-covariance matrix for the coefficients, denoted by VC and obtained from the conditional logit model in Table 4.3 above. Five thousand random draws from a multivariate normal distribution with variance–covariance matrix VC and mean β were used to simulate the sampling distribution of the vector $\hat{\beta}$ estimates. For each draw of $\hat{\beta}$, inclusive values for the baseline and post-policy water quality were calculated. The difference between the two inclusive values gave an approximate sampling distribution of the compensating surplus, based on the Slutsky theorem on the consistency of continuous functions of maximum likelihood estimates. A 95% confidence interval for the mean was obtained by ranking a vector of the calculated compensating surplus values and dropping the top and bottom 2.5 % of the simulated values. All simulations were done in Excel.

Table 4.5: Welfare estimates due to changes in water quality in 2008 New Zealand Dollars

Lakes	Population (N=21883)	Sample (n = 414)	Per angler/year	Per choice
1 metre increase in water clarity in all lakes with poor and average water quality concurrently				
Rotorua, Rotoiti, Okaro, Rotoehu, Rotomahana, Okareka & Rerewhakaaitu	2,283,723.58	43,205.30	104.36 [86.75 122.14]	1.71
3 metre increase in water clarity all lakes with poor and average water quality concurrently				
Rotorua, Rotoiti, Okaro, Rotoehu, Rotomahana, Okareka & Rerewhakaaitu	7,286,699.54	137,855.58	332.98 [274.26 393.08]	5.47
1 metre increase in water clarity in each lake individually				
Rotoiti	1,268,168.02	23,992.21	57.95 [47.23 69.25]	0.95
Rotorua	821,926.75	15,549.86	37.56 [30.28 45.06]	0.62
Rerewhakaaitu	125,186.89	2,368.39	5.72 [4.40 7.20]	0.09
Rotoehu	91,140.07	1,724.26	4.16 [3.16 5.36]	0.07
Okareka	46,227.54	874.57	2.11 [1.49 2.88]	0.03
Rotomahana	14,220.32	269.03	0.65 [0.46 0.89]	0.01
Okaro	203.93	3.86	0.01 [0.00 0.02]	0.0002
3 metre increase in water clarity in Lakes Rotoiti and Rotorua individually				
Rotoiti	4,286,320.64	81,092.02	195.87 [156.83 237.94]	3.22
Rotorua	2,856,606.82	54,042.80	130.54 [102.91 159.74]	2.14

In considering use of these estimates, policy decision makers should be aware of potential bias from the single traveller, median income and day trip assumptions used in this thesis. If the travelling party for the sample of anglers used in this thesis is greater than one, then resource values estimated here are likely to be inflated. Freshwater sport fishing tend to be dominated by wealthier sectors of society due to high expenditures involved (Cowx, 2002). Using regional median income as a proxy for anglers' income is likely to underestimate the cost of lake access and WTP estimates are likely to be under-valued. On the other hand, if overnight trips were undertaken by some anglers, then resource values estimated are likely to be undervalued.

Considering a 1 metre increase in water clarity in all the lakes with poor and average water quality concurrently, the model predicts welfare gains of about $104 per angler per year, with a corresponding population welfare gain of about $2.3 million. In line with the home origins of anglers in the study sample, the appropriate study population consists of all anglers who bought fishing licences during the 2007/08 fishing season and whose home origins are within the Eastern Region and Auckland/Waikato Fish and Game Councils. The study population of interest is equal to 21,883 anglers (Unwin, 2009). For a 3 metre increase in water clarity, welfare gains of about $333 per angler per year and population welfare gains amounting to about $7.3 million are predicted.

For a 1 metre increase in water clarity in each of the lakes with poor and average water quality individually, the highest welfare gains are predicted for Lake Rotoiti of about $58 per angler per year. This is followed by Lake Rotorua with predicted welfare gains of about $38 per angler per year. The lowest welfare estimates are predicted for Lakes Okaro and Rotomahana with welfare gains of less than $1 per year. When aggregated over the target population, the total welfare gains range from a minimum of about $204 for Lake Okaro to about $1.3 million for Lake Rotoiti.

Additionally, a 3 metre increase in water clarity in Lake Rotoiti will lead to welfare gains of $196 per angler per year with a corresponding population estimate of about $4.3 million. For Lake Rotorua, welfare gains are predicted at $131 per angler per year, while corresponding population estimates are projected to be around $2.9 million. Corresponding welfare gains from Model 2 are presented in Appendix 6 (Table A6.2). The predicted welfare gains are consistent with those obtained from Model 1 above.

Table 4.6: Welfare loss due to lake closure in 2008 New Zealand Dollars

Lakes	Population	Sample	Per angler/year	Per choice
Rotoiti	7,672,488.44	145,154.24	350.61	5.76
			[324.85 377.00]	
Tarawera	5,088,027.92	96,259.36	232.51	3.82
			[213.16 252.56]	
Rotorua	4,535,940.93	85,814.54	207.28	3.40
			[189.91 225.65]	
Rotoma	1,662,859.58	31,459.30	75.99	1.25
			[66.15 86.96]	
Okataina	801,543.71	15,164.24	36.63	0.60
			[29.82 44.67]	
Rerewhakaaitu	612,801.23	11,593.46	28.00	0.46
			[22.68 34.22]	
Rotoehu	444,272.80	8,405.11	20.30	0.33
			[15.68 26.10]	
Okareka	224,373.99	4,244.89	10.25	0.17
			[7.11 14.32]	
Tikitapu	118,448.40	2,240.90	5.41	0.09
			[3.34 8.25]	
Rotomahana	69,710.80	1,318.84	3.19	0.05
			[1.89 5.07]	
Okaro	1,010.29	19.11	0.05	0.001
			[0.01 0.12]	

Table 4.6 above presents the welfare loss estimates in case of lake closure for each of the eleven lakes in the choice set. Welfare losses are computed even for the lakes with good water quality, recognizing the fact that lake closure may occur for a number of other reasons in addition to poor water quality. The highest welfare loss of about \$351 per angler per year is predicted for Lake Rotoiti. When aggregated across the study population, a total welfare loss of about \$7.7 million per year is predicted. This is followed by Lakes Tarawera and Rotorua with a welfare loss of about \$233 and \$207 per angler per year with corresponding population welfare losses of about \$5.1 and \$4.5 million, respectively

The highest welfare losses for these lakes stem from a large number of anglers fishing in these lakes. Lake Rotoiti has average water quality and is generally considered to offer relatively large trout. On the other hand, Lake Tarawera offers good water quality and relatively big trout, in addition to scenic beauty. Lake Rotorua is the biggest in the region and easily accessible compared to other lakes. These and many other factors can explain anglers' preferences for these lakes.

The lowest welfare losses are predicted for Lakes Tikitapu, Rotomahana and Okaro. According to Eastern Region Fish and Game (2011), Lake Tikitapu is a popular lake for many recreational activities and as such anglers often have to compete with water skiers, swimmers and canoeists. Consequently, this lake attracts fewer anglers despite its scenic beauty and good water quality. On the other hand, Lake Okaro is the smallest out of all the Rotorua Lakes and has the poorest water quality. For Lake Rotomahana, factors other than the size of lake and water quality could be the main contributors.

4.7 Chapter summary

The main purpose of this chapter was to assess anglers' preferences for better water quality in the Rotorua Lakes. In addition to water quality, a number of other lake attributes that can impact upon anglers' fishing site choice decisions have been explored, followed by a description of the data and the travel cost site choice models employed in the estimation of results. The chapter further spelt out the methodologies for assessing the welfare gains due to water quality changes and welfare losses arising from possible lake closure.

Past recreational studies have shown that a number of attributes can influence anglers' choice of fishing sites, including cost of site access, fishing quality, environmental quality, facility development, regulations, and encounter levels. In terms of cost of site access, most researchers account for travel cost and opportunity cost of travel time to recreational sites. A number of proxies are used for fishing quality including catch rates, size of fish, species abundance, the number of fish per square metre and presence of stocked water bodies. Water quality and amount of forested land are commonly used as a measure of environmental quality. Proxies for facility developments at recreational sites include boat ramps, ease of site access and number of camping facilities. On the other hand, it is acknowledged that encounter levels beyond a certain threshold can impact upon recreational users negatively, but accounting for this variable in estimation is usually limited by data availability. In addition to the above attributes, fishing site choice decisions may be influenced by a number of other intervening factors including past recreational experiences, place attachment and individual social demographic factors.

In terms of data requirements, this application employed the fishing choice data for the 2007/08 fishing season, obtained from the National Angling Survey conducted jointly NIWA and FGNZ. A number of site choice determinants were identified including the cost of site access, water clarity, weight of fish, size of lake, facility development, urban development, percentage of forest cover, lake depth and health warnings due to algal blooms. The impact of these attributes on anglers' fishing site choice destinations was investigated through the conditional logit modelling framework. The results reveal that anglers generally favour lakes with better water quality, bigger fish, lakes that are relatively big in size, with more facilities and situated in natural settings with forest cover. Lake depth, urban development around the lakes and health warnings are major detractors for many anglers.

The hypothesized water quality improvements in lakes with poor and average water quality illustrate that some lakes would attract most anglers (e.g. Rotoiti and Rotorua) while lakes such as Okaro and Rotomahana would attract the least anglers. The welfare measures associated with such water quality changes are also simulated and show the highest welfare gains to be predicted for Lake Rotoiti, with an estimated welfare gain per angler per year of $58 and $196 for 1 and 3 metre increases in water clarity, respectively. This is followed by Rotorua with a predicted welfare gain of $38 and $131 per angler per year for 1 and 3 metre increases in water clarity. The lowest welfare gains are predicted for Lake Okaro of about $0.01 per angler per year for a 1 metre increase in water clarity.

In addition, the study results reveal that welfare losses due to possible lake closures for the Rotorua Lakes are quite diverse, ranging from as high as $351 per angler per year (Lake Rotoiti) to as low as $0.05 (Lake Okaro).

Analysis of welfare losses due to possible lake closures allows us to estimate the total welfare that anglers obtain from each lake. On this basis the overall level of angler benefits is highest for Lake Rotoiti, followed by Tarawera, Rotorua, Rotoma and Okataina. Lakes Rotomahana and Okaro provide the lowest level of angler benefits.

CHAPTER FIVE

THE EFFECT OF WITHIN-SEASON VARIABILITY IN SITE ATTRIBUTES ON WELFARE ESTIMATES

5.1 Introduction

In Chapter Four the conditional logit fishing site choice model for the Rotorua Lakes was developed. A number of site attributes were found to influence anglers' choices of lake for fishing, including water quality. The welfare gains due to water quality changes and welfare losses arising from possible lake closure were assessed.

The analysis is extended further by exploring whether failure to account for within-season variability in recreational site attributes can have significant effects on welfare estimates. Within-season variability in site attributes, such as fishery regulations, catch rates and congestion, is acknowledged to be affecting site choice (Provencher & Bishop, 2004; Swallow, 1994). However knowledge of this subject area remains sparse. This can partly be attributed to insufficient variation in natural conditions that characterizes most datasets of recreational site attributes. The variability in water quality and fish growth in the Rotorua Lakes across the year presents an opportunity to investigate the subject.

Welfare estimates from models accounting for and those ignoring within-season variability in water clarity and weight of fish are compared. Within-season variability is accounted for through the use of bimonthly averages of water clarity and weight of fish. Two main factors that may lead to differences in welfare estimates between models utilizing bimonthly versus annual averages of water clarity and weight of fish are explored. These include differences in attribute and collinearity levels.

The subsequent sections review of within-season variability in recreational site attributes and multicollinearity in revealed preference data analysis. This is followed by an outline of the methodology and estimated results. Finally, the implications on welfare estimates of accounting for within-season variability in water quality and fish growth are investigated.

5.2 Within–season variability in site attributes

In fishery management and recreation studies, within-season variability in site attributes such as fishery regulations, catch rates, congestion and fish growth are well recognized in both practice and principle. The seasonality in fishing quality caused by variability in any one of these fishing quality indicators may cause demand shifts between sub-seasons. This may have implications for predicted trip frequency and welfare estimates (Swallow, 1994).

Fishery regulations such as catch quotas may vary across the fishing season, which may cause anglers to switch days between sub-seasons. For example, anticipation of a better option later on might delay a planned fishing trip, or sudden news of good conditions at some destinations might bring forward a trip planned for later on in the season (Andrews, 1988; Clark, 1980; Swallow, 1994). However, the extent to which welfare estimates might be impacted by an intervention or regulation depends on whether it comes earlier or later in the recreational season. For instance, Woodward *et al.* (2001, p. 1) have demonstrated that "when an intervention […] involves an early closure of the resource so that some of the trips are not realized, then the underlying theoretical foundation does matter and very different welfare estimates can result." On the contrary, based on the standard neoclassical demand theory, willingness to pay for an additional trip is expected to be lower for a later closure (Woodward *et al.*, 2001).

Similarly, variability in catch rates over the recreation season has long been recognized (Carpentar *et al.*, 1994; Hall & Brown, 2008; Lux & Smith, 1960; Van Poorten & Post, 2005). Although in most travel cost recreational studies catch-rates are assumed to be constant over time (Carpentar *et al.*, 1994; Van Poorten & Post, 2005), an exception is provided by Provencher & Bishop (2004). In their study of anglers fishing in the Milwaukee–Racine waters of Lake Michigan in 1996-1997, they found that average catch rates varied between the two years and within each year[57]. Average catch rates in 1997 were 31% higher than in 1996. For the first half of the season the catch rate in 1997 was much higher than the catch rate in 1996, and for the second half of the season the catch rate in 1997 was somewhat lower than in 1996 (Provencher & Bishop, 2004, p.793). A 5-day moving average of the sample catch rate was used to account for intra-seasonal variability in fish catch. They found that anglers' responses to changes in fish catch were not as elastic as predicted by the static models used in the analysis, possibly due to their inability to account for dynamic behaviour.

Furthermore, there is a general lack of studies investigating the impact of within-season variability in congestion levels on recreational participation and site choice. The study by Schuhmann & Schwabe (2004) represents one unique case exploring this aspect. Their study results indicated substantial differences in anglers' per-trip welfare estimates depending on how expected congestion was measured.

Some studies in fishery management have also highlighted the possible variability in fish growth between seasons and water bodies. This variability is attributed to a number of factors including differences in water temperatures and food availability over time (Finstad *et al.*, 2004; Rätza & Lloret, 2002).

[57] Trip information was collected from a sample of 97 anglers at biweekly intervals from May to September of each year, using telephone interviews.

Researchers in recreational fishing studies have acknowledged and in some cases accounted for within-season variability in fishery regulations, catch rates and congestion levels. However, there is a general lack of studies accounting for within-season variability in fish growth and water quality. This can be attributed to insufficient variability in these attributes and/or the general preference for annual data. Extensive data requirements could be another limiting factor.

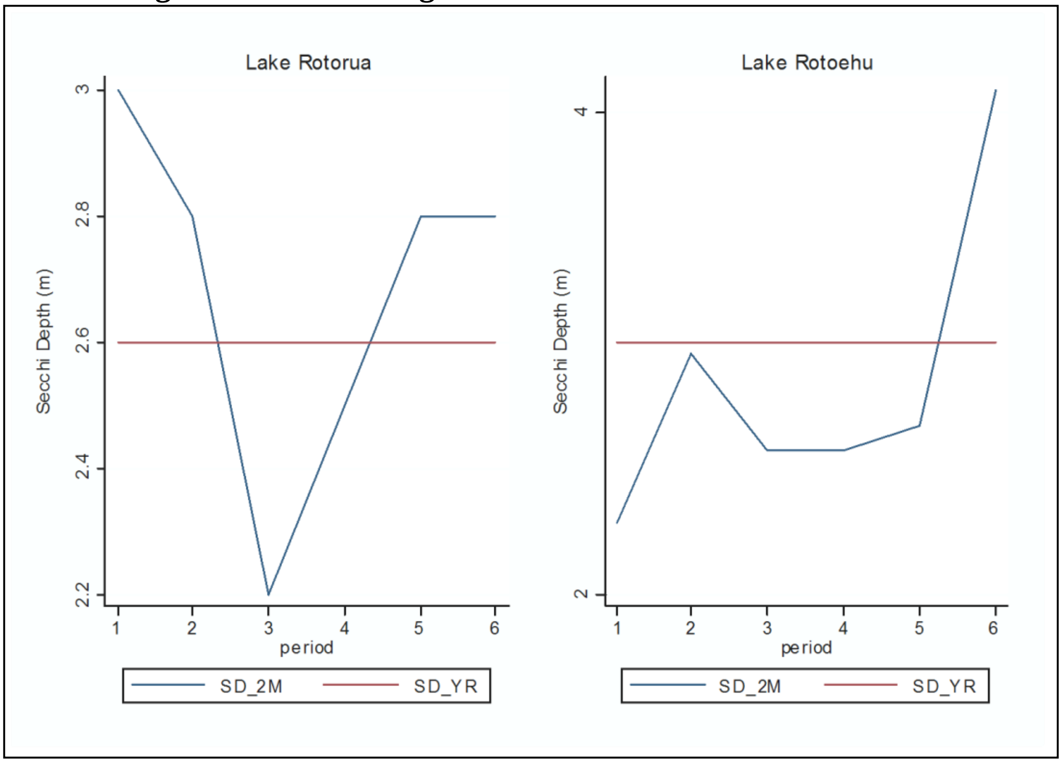
The Rotorua Lakes present an opportunity for this analysis for two reasons. First, the 2007/08 fishing choice data used in this application account for seasonality in angler demand. In addition to region and licence type, the survey was stratified by time, with the 12 month survey period divided into six two-monthly intervals (Unwin, 2009). This was done to account for the variability in angler usage of water bodies across the fishing year. Studies have shown that besides within-season variability in site attributes, angler's demand might cause demand shift across sub-seasons. For instance, some anglers may prefer to fish during good weather (Cameron & James, 1987) or on summer vacation (Andrews & Wilen, 1988). Swallow (1994, p.925) stipulates that "anglers' tastes and preferences may induce seasonality in demand, regardless of any seasonal pattern in fishing quality".

Second, the Rotorua Lakes are appropriate for this exploration because water quality and fish growth tend to vary across the year and between lakes. A detailed outline of water quality and its inter-temporal variability across the year is given in Chapter Two and a succinct summary is provided below.

5.3 Within-season variability in water quality and trout growth in the Rotorua Lakes

Water clarity is used as a measure of water quality because it aligns well with anglers' perceptions of water quality[58]. Figure 5.1 below shows the variability in water clarity for eutrophic lakes. Periods 1, 2, 3, 4, 5 and 6 denote October-November, December-January, February-March, April-May, June-July and August-September for the 2007/08 fishing season. SD_2M and SD_YR refer to bimonthly (within-season) and annual averages of water clarity, respectively.

Figure 5.1: Bimonthly versus annual averages of water clarity for eutrophic lakes during the 2007/08 fishing season



For Lake Rotorua, anglers who went fishing in period 1 experienced water clarity levels of about 3 m, while for anglers who fished in period 3, water clarity was

[58] In general there is a very strong correlation between water quality and clarity, although in some cases clear water may not necessarily be of good quality.

about 2.2 m, representing a difference of about 0.8 m. In Lake Rotoehu, for anglers who fished in period 1, water clarity was about 2.3 m, while for anglers who fished in period 6, water clarity was about 4.1 m, representing a difference of about 1.8 m between these two periods. In general it is anticipated that the effect on anglers' utility of a change in water clarity across sub-seasons will depend on the prevailing baseline conditions. For instance, the effect on utility of a 1 m change is expected to be higher for a lake with average water clarity of 2 m than a lake with clarity of 10 m.

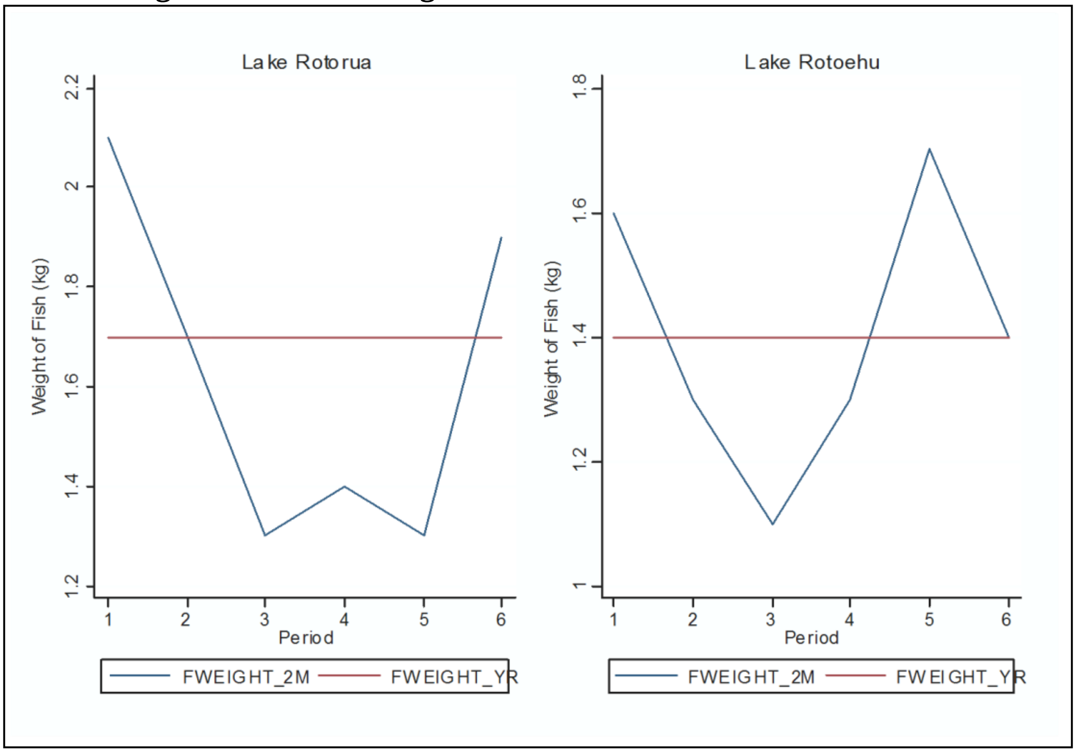
In addition, trout is the main fish species in the Rotorua Lakes, can grow to trophy sizes in cooler months (autumn, winter and spring) as they prey on a large number of migrating smelt that enter nearby tributaries spawn. [59] However, no conclusive evidence exists yet on the exact link between water quality and fish growth. For instance, Pitkethley (2008 p. 173), the Regional Manager of the Eastern Region Fish and Game Council reporting on the subject, wrote:

> We know on other lakes like Rotoehu, declining water quality has certainly decreased fish growth. It appears that trout are not affected by blooms on some lakes, but for other lakes they certainly cannot handle declining water quality. […]. The first thing we discovered is that angling success declines dramatically during these peak algal blooms. Catch rates drop significantly […].

Figure 5.2 below shows the variability in trout growth during the 2007/08 fishing season for Lakes Rotorua and Rotoehu.

[59] http://www.nzfishing.com/fishingwaters/eastern/ERFishingWaters/EROhauChannel.htm

Figure 5.2: Bimonthly versus annual averages of weight of fish for eutrophic lakes during the 2007/08 fishing season



FWEIGHT_2M and FWEIGHT_YR refer to the bimonthly (within-season) and annual averages of weight of fish, respectively. For Lake Rotorua the average weight of fish fluctuated from about 2.1 kg in period 1 to around 1.3 kg between period 3 and 5. Similarly, fluctuations of up to 0.7 kg are observed for Lake Rotoehu between periods 3 and 6. Whether differences in the average size of fish as low as 0.7 kg can be considered substantial enough to lead to possible differences in utility derived by anglers across sub-seasons remains an empirical investigation.

In Table 5.1, the summary statistics for the bimonthly averages of water clarity and weight of fish for all lakes are presented.

Table 5.1: Summary statistics for the bimonthly averages of water quality and weight of fish

	Secchi depth 2007/08 fishing season (m)				Weight of fish 2007/08 fishing season (kg)			
Lakes	**Mean**	**St.dev**	**Min**	**Max**	**Mean**	**St.dev**	**Min**	**Max**
Rotorua	2.7	0.3	2.2	3.0	1.6	0.3	1.3	2.2
Rotoiti	4.9	0.7	3.8	5.9	2.0	0.2	1.4	2.3
Tarawera	9.3	0.7	8.4	10.7	1.7	0.2	1.0	2.0
Okataina	10.9	0.6	9.8	11.5	1.9	0.5	1.4	2.6
Rotoma	13.3	1.5	11.7	15.5	1.5	0.3	0.8	2.3
Okareka	7.9	0.8	5.8	9.1	1.3	0.3	0.7	1.6
Rotoehu	2.8	0.4	2.3	4.1	1.3	0.2	1.1	1.7
Rerewhakaitu	4.8	0.6	3.8	5.5	1.3	0.2	1.0	1.8
Tikitapu	6.8	0.5	5.3	7.4	-	-	-	-
Rotomahana	4.8	0.9	3.9	6.7	-	-	-	-
Okaro	2.4	0.8	1.3	3.6	-	-	-	-

Comprehensive data on water quality were obtained from Environment Bay of Plenty. Weight of fish data were obtained from Fish and Game Eastern Region. Comprehensive data on weight of fish were not available for Lakes Tikitapu, Rotomahana and Okaro and annual averages are used instead.

The mean values were computed from the bimonthly averages of water clarity and weight of fish for six period intervals. The minimum and maximum values correspond to lower and upper bounds of the bimonthly averages for the six period intervals for each lake.

The difference between the minimum and maximum average levels of water clarity range from as low as 0.8 m for Lake Rotorua to as high as 3.8 m for Lake Rotoma. For weight of fish, the differences range from 0.6 kg for Lake Rotoehu to 1.5 kg for Lake Rotoma. The bimonthly averages of water clarity and weight of fish align with the fishing choice data which was also divided into bimonthly partitions for the 2007/08 fishing season. This ensures that anglers' preferences are estimated using water quality and weight of fish attribute levels existing during the period they recorded a fishing trip. Ignoring this seasonal variability might fail to account for differences in utility derived by anglers across sub-seasons. Welfare estimates could be affected in either direction. Whether the effects would be significant is an empirical investigation.

Another potential benefit that can be extracted from the use of disaggregated data on water clarity and weight of fish is the increased variability in the data beyond that which is provided by the cost of lake access. Increased variability is an indicator of how rich the data is and has a direct effect on empirical identification of parameters as illustrated by Cherchi & Dios Ortúzar (2008, pp. 110-111) below.

> It is well-known that the capability of estimating a correct model (i.e. with identifiable parameters and free of confounding effects) depends on the amount of explanation that can be extracted from the data. However how rich (in terms of variability) the data should be to avoid empirical identification problems and to produce correct models is not known […][60].

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[60] Moreover, although the problem of data richness is often related to the number of observations, this is not a necessary condition. In fact, this is basically the difference between revealed and stated preference data, where usually small samples are sufficient in the latter case because information available for each individual is richer (Cherchi & Dios Ortúzar, 2008, p. 111).

It is conjectured that the increased variability obtained from the variation in water clarity and fish growth attributes across time might reduce the multicollinearity problem ubiquitous in travel cost data, which is briefly reviewed in the following section.

5.4 Multicollinearity in travel cost data

The problem of multicollinearity in econometric models has received widespread attention by researchers in various fields since the pioneering work by Frisch (1934). Multicollinearity is defined as the intercorrelation among regressors in a model. The possible causes, detection and consequences of multicollinearity are well documented (Koutsoyiannis, 1977; Maddala, 1992). Some of the effects of multicollinearity commonly cited in the literature include: the possibility of obtaining coefficient estimates with wrong signs and magnitudes; instability of the estimated coefficients; and the likelihood of obtaining insignificant coefficient estimates with large standard errors. Some earlier econometric studies suggested that even moderate or low levels of collinearity can affect the precision of the parameter estimates (Koutsoyiannis, 1977; Maddala, 1992).

A number of possible solutions to address multicollinearity have been suggested. These include excluding the variables responsible for multicollinearity; tolerating multicollinearity if coefficient estimates are not seriously affected; pooling cross-section and time series data, and increasing the sample size (Koutsoyiannis, 1977; Maddala, 1992). The proposed solutions have their own limitations. For instance, dropping collinear variables from the model can lead to loss of information and may cause mis-specification errors arising from omitted variables (Koutsoyiannis, 1977). Pooling cross-section and time series data is not feasible if such data are unavailable. Increasing the sample size might not guarantee well behaved data and can also be expensive, both in money and time costs (Jagpal, 1982).

The problem of multicollinearity associated with use of the TCM has been recognized since its earliest applications by Clawson (1959) and Clawson & Knetsch (1966). The initial concern was the collinearity between travel cost and travel time when used as separate regressors in recreational demand models. This issue is widely addressed, especially in transportation and environmental valuation literature (e.g. Allen *et al.*, 1981; Brown & Nawas, 1973; Gum & Martin, 1975; Loomis *et al.*, 2001; Wetzstein & McNeely, 1980).

To address collinearity between travel costs and travel time, Cesario (1976) and Cesario & Knetsch (1976) proposed combining the two variables into a single regressor. Other studies have suggested the use of less aggregated data (Brown & Nawas, 1973; Gum & Martin, 1975). However, there is little evidence that such an approach can reduce collinearity due to the strong association between travel costs and travel time (Allen *et al.*, 1981).

Currently, the generally acceptable methodology to reduce multicollinearity is through the joint estimation of revealed and stated preference data, commonly denoted as the RP-SP. The strategic design of attribute levels in the stated preference survey can reduce some of the collinearity inherent in revealed preference quality characteristics (Adamowicz *et al.*, 1994)[61]. In environmental valuation, the joint estimation of RP-SP data was introduced by Cameron (1992). This approach has been widely applied in environmental valuation literature (Adamowicz *et al.*, 1994; Bhat, 2003; Kling, 1997; Von Haefen & Phaneuf, 2008) and also in transportation studies (Ben-Akiva & Morikawa, 1990; Brownstone *et al.*, 2000; Hensher & Bradley, 1993).

[61] In addition to reducing multicollinearity problems, combining RP-SP data has been found to be important for extending the market beyond existing consumers and products and reducing the hypothetical bias in SP data (Hensher *et al.*, 2005; Whitehead *et al.*, 2008).

Nevertheless, the use of revealed preference (RP) data alone still remains more common than RP-SP[62]. One major reason for the popularity of the RP approach is the less extensive data requirement as opposed to the combined RP-SP. However, it is acknowledged that multicollinearity remains one of the major methodological issues of concern in RP studies (Arnot *et al.*, 2006; Brownstone *et al.*, 2000; Hensher, 2001; Morrison, 2001; Small *et al.*, 2005; Whitehead *et al.*, 2008).

The approach used in this chapter mirrors that of Brown & Nawas (1973) and Gum & Martin (1975) except that disaggregation is done across time and involves non-monetary site quality characteristics. The extent to which increased variability from the use of less aggregated data can reduce multicollinearity is tested. It is expected that collinearity levels from models accounting for within-season variability in water clarity and weight of fish will be lower. Consequently, more precise parameter estimates can be obtained. Therefore, differences in welfare estimates between models accounting for, and those ignoring, within-season variability may result from two sources:

(i) Differences in utility weights caused by differences in attribute levels between the annual and bimonthly averages of water clarity and weight of fish.

(ii) Differences in the precision of the estimated parameters due to differences in collinearity levels.

In addition, within-season variability in water clarity and weight could provide the increased variability required for the identification of unobserved effects. In revealed preference data, unobserved effects can be identified if site attributes vary

[62] This is evidenced by the large number of RP recreational studies (Egan *et al.*, 2009; Johnstone & Markandya, 2006; Morey *et al.*, 2002; Murdock, 2006; Thiene & Scarpa, 2008). If parameter estimates are not seriously affected, collinearity is either tolerated or in some cases researchers simply acknowledge the problem (Englin *et al.*, 1996; Johnstone & Markandya, 2006).

either across individuals and sites (for instance, the cost of lake access) or across sites and time (Murdock, 2006).

In the next section the methodology for assessing whether models that account for within-season variability in water clarity and weight of fish and those that do not is provided in detail.

5.5 Methods

The main question addressed in this chapter is whether models that account for and those that ignore within-season variability in water clarity and weight of fish give significantly different welfare estimates. To achieve this objective, conditional logit models using bimonthly and annual averages of water clarity and weight of fish are estimated. To test whether the annual and bimonthly attributes induce different collinearity levels, the determinants of the asymptotic variance covariance matrix (AVC) computed from the negative of the Hessian of the log-likelihood function are compared.

Welfare estimates from the alternative use of the two data types are also compared. The compensating surplus is calculated using a simulation method proposed by Krinsky & Robb (1986). Five thousand random draws from a multivariate normal distribution with variance-covariance matrix VC and mean β are used to simulate the sampling distribution of the vector $\hat{\beta}$ estimates. For each draw of $\hat{\beta}$, inclusive values for the baseline and post-policy water quality are calculated. The difference between the two inclusive values gives an approximate sampling distribution of the compensating surplus, based on the Slutsky theorem on the consistency of continuous functions of maximum likelihood estimates. A 95% confidence interval for the mean is obtained by ranking a vector of the calculated compensating surplus values and dropping the top and bottom 2.5% of the simulated values.

The convolutions test by Poe, Welsh, & Champ (1997) is used to assess whether there are significant differences across empirical distributions of the compensating surplus. The simulations are done in R console using the mded software package recently developed by Aizaki (2012) to measure the difference between two empirical distributions of the willingness to pay. The model specification is provided in the next section.

Model specification

The conditional logit model is applied in this analysis. A more detailed outline of the model is provided in Chapter Four. It is assumed that the utility angler n obtains from a fishing trip is a function of observed variables related to lake j and unobserved factors.

$$U_{nj} = \beta x_{nj} + \varepsilon_{nj} \tag{1}$$

where β is a vector of parameters and x is a vector of observed factors influencing the angler's choice of fishing destinations including the cost of site access and lake attributes. ε_{nj} is a vector of unobserved determinants only known to the angler.

Following McFadden (1974), the probability (P_{nj}) that angler n chooses to fish at lake j on a given day can be expressed as:

$$P_{nj} = \frac{e^{V_{nj}}}{\sum_{j=1}^{J} e^{V_{nj}}} \tag{2}$$

where $V_{nj} = \beta_n' x_{nj}$ and is the representative part of utility assuming linearity in parameters.

The probability of angler n choosing the lake that she or he was actually observed to have chosen can be expressed as:

$$L(\beta) = \prod_{j=1}^{J} (P_{nj})^{y_{nj}} \tag{3}$$

where $y_{nj} = 1$ if angler n chooses lake j and zero otherwise. Since $y_{nj} = 0$ for all lakes not chosen by an angler, $L(\beta) = P_{nj}$ is the probability of the lake actually being chosen by an individual on a single choice occasion and hence considered to be the contribution that each angler makes to the likelihood function.

Assuming that anglers' choices are independent of each other, the probability of each angler in the sample choosing the lake that they were observed to have actually chosen is equal to:

$$L(\beta) = \prod_{n=1}^{N} \prod_{j=1}^{J} (P_{nj})^{y_{nj}} \tag{4}$$

Expressing the above likelihood function in logarithmic form, the log-likelihood function is:

$$LL(\beta) = \sum_{n=1}^{N} \sum_{j=1}^{J} y_{nj} \ln P_{nj} \tag{5}$$

In estimation, the objective is to determine the parameter estimate (β) that maximizes the log-likelihood function conditioned on the data X (cost of lake access, lake attributes) and the observed choices, y. Parameters are estimated by maximum likelihood using the standard routines implemented in Nlogit 4.0.

McFadden (1974) has shown the formal derivation of the gradient and the Hessian of the log-likelihood function with respect to parameters. Assuming generic parameters, the first derivative for the multinomial logit model is given by:

$$\frac{\partial LL(\beta|X,y)}{\partial\beta} = \sum_{n=1}^{N}\sum_{j=1}^{J}\left(y_{nj} - P_{nj}(X|y)\right)X_{nj} \tag{6}$$

Denoting the vector of parameter estimates at which the log-likelihood function is maximized as β_t, the gradient at this point can be expressed as:

$$g_t = \left(\frac{\partial LL(\beta|X,y)}{\partial\beta}\right)_{\beta_t} \tag{7}$$

The Hessian is the matrix of the second derivative of the log-likelihood function at β_t as shown in equation (8).

$$H_t = \left(\frac{\partial^2 LL(\beta|X,y)}{\partial\beta\partial\beta'}\right)_{\beta_t} \tag{8}$$

A value of the Hessian close to 0 is an indication that the model is not identified. The asymptotic variance covariance matrix (AVC) is derived from the Fisher Information matrix (I_t) which is the negative of the expected value of the Hessian matrix.

$$I_t(\beta/X,y) = -E_y\left(\frac{\partial^2 LL(\beta|X,y)}{\partial\beta\partial\beta'}\right)_{\beta_t} = -E_y H_t \tag{9}$$

The AVC matrix (Ω_t)is the inverse of the Fisher Information matrix as shown below.

$$\Omega_t(\beta/X,y) = \left(-E_y\left(\frac{\partial^2 LL(\beta|X,y)}{\partial\beta\partial\beta'}\right)_{\beta_t}\right)^{-1} = I_t^{-1} \tag{10}$$

The effects of collinearity are reflected in relatively high standard errors of the parameters, β_k. The standard errors are the roots of the diagonal elements (variances) of the asymptotic variance-covariance (AVC) matrix. In general the reliability of the parameter estimates is dependent on the size of the standard errors. A model with parameter estimates having smaller standard errors is said to be more efficient. A number of efficiency measures have been proposed including the determinant of the AVC matrix commonly known as the D-error.

$$D - error = det\left(\Omega_t(\beta/X, y)\right)^{1/K} \qquad (11)$$

where K is a scaling factor for the efficiency measure and is equal to the number of estimated parameters. Generally, the lower the D-error, the lower the collinearity and therefore, the more efficient the estimated parameters. The estimated results from the conditional logit model are presented in the subsequent section.

5.6 Estimated results

The description of variables used in estimation is presented in Table 5.2 followed by the estimated results.

Table 5.2: Description of regressors used in estimation

COST	Cost of lake access. It includes travel cost and opportunity cost of travel time
SD	Average water clarity in metres
SD_YR	Annual average of water clarity in metres
SD_2M	Bimonthly average of water clarity in metres
FWEIGHT	Average weight of fish in kilograms
FWEIGHT_YR	Annual average of weight of fish in kilograms
FWEIGHT_2M	Bimonthly average of weight of fish in kilograms
LKSIZE	Log of lake size in km^2
FDV	Facility developments around the lakes. Number of boat ramps and key lake access points were found to be highly collinear. Therefore, the number of key lake access points is used as a proxy for facility development
URBAN	Percentage of land around the lake devoted to urban development
FOREST	Percentage of land around the lake with forest cover
DEPTH	Lake depth measured in metres
HWARNING	Dummy variable indicating whether a health warning due to algal blooms was issued to a lake between 2001 and 2007. It takes a value of 1 if yes and 0 otherwise.

Estimated results from the conditional logit models are given in Table 5.3 below.

Table 5.3: Estimated results from models utilizing annual versus bimonthly averages of water clarity and weight of fish

	Model 1 *SD_YR & FWEIGHT_YR*		**Model 2** SD_2M & FWEIGHT_2M		**Model 3** SD_2M & FWEIGHT_YR		**Model 4** SD_YR & FWEIGHT_2M	
Variable	*Coefficient*	|*t-value*|	*Coefficient*	|*t-value*|	*Coefficient*	|*t-value*|	*Coefficient*	|*t-value*|
COST	-0.072*** (0.007)	10.79	-0.074*** (0.006)	11.50	-0.072*** (0.007)	10.97	-0.074*** (0.007)	11.35
SD	0.190*** (0.017)	11.44	0.134*** (0.009)	13.43	0.149*** (0.012)	4.44	0.154*** (0.011)	3.65
FWEIGHT	1.376*** (0.291)	4.70	0.282*** (0.079)	3.59	1.069*** (0.241)	12.94	0.283*** (0.078)	13.94
LKSIZE	3.407*** (0.279)	12.20	3.327*** (0.265)	12.54	3.305*** (0.264)	12.5	3.361*** (0.265)	12.67
FDV	0.356*** (0.025)	14.44	0.326*** (0.022)	14.76	0.341*** (0.022)	10.51	0.329*** (0.023)	9.99
URBAN	-0.352*** (0.036)	9.75	-0.350*** (0.034)	10.18	-0.346*** (0.033)	15.31	-0.343*** (0.034)	14.44
FOREST	0.015*** (0.002)	7.50	0.017*** (0.002)	6.60	0.012*** (0.002)	6.9	0.013*** (0.002)	7.26
DEPTH	-0.056*** (0.007)	8.52	-0.038*** (0.004)	10.10	-0.050*** (0.005)	9.65	-0.039*** (0.004)	10.34
HWARNING	-0.606*** (0.141)	4.29	-0.598*** (0.158)	3.78	-0.581*** (0.148)	3.94	-0.632*** (0.159)	3.97
Log-Likelihood	-3843.65		-3842.95		-3842.45		-3846.88	
McFadden R^2	0.275		0.275		0.275		0.274	
D-error	**6.6E-34**		**3.3E-35**		**3.15E-34**		**4.08E-35**	

***, **, * denotes significance at 1%, 5% and 10% level respectively. The figures in ( ) are the standard errors.

In Model 1 within-season variability is not accounted for through the use of the annual averages of water clarity (SD_YR) and weight of fish (FWEIGHT_YR). Model 2 accounts for within-season variability by utilizing the bimonthly averages of water clarity (SD_2M) and weight of fish (FWEIGHT_2M). Two additional models which partially account for within-season variability are estimated. Model 3 only accounts for within-season variability in water clarity while model 4 accounts for within-season variability in weight of fish. SD and FWEIGHT denote water clarity and weight of fish, respectively. All other regressors are common to all models. They include the cost of lake access (COST), lake size (LKSIZE), urban development (URBAN), facility development (FDV), amount of forested land (FOREST), lake depth (DEPTH) and health warning (HWARNING).

There is no difference in model performance between all models as indicated by the log-likelihood and McFadden R-Squared. All the attributes are highly significant with expected signs. The parameter for the cost of lake access is negative and highly significant in all models indicating that lakes that were closer to angler's homes were more preferred. In addition anglers generally preferred lakes with better water clarity and bigger fish, and preferred bigger lakes, with more facilities and forested land. Lakes with more urban development, which are greater in depth and with health warnings, were generally less preferred.

The determinant of the asymptotic variance covariance matrix, the D-error, is used to assess whether the models accounting for and those ignoring within-season variability in water clarity and weight of fish induced similar collinearity levels. Model 1, in which the annual averages of water clarity and weight of fish were used, has the highest D-error of all models. The D-error from Model 1 is 20 times higher than Model 2, in which the bimonthly averages of water clarity and weight of fish were used. The use of the bimonthly average weight of fish reduces the size of the standard error by up to 73%.

The D-error from Model 1 is also compared to that of Model 3. In both models the annual averages of weight of fish were used, except that in Model 3 the bimonthly averages of water clarity were used. The D-error from Model 1 (6.6E-34) is about 2 times higher than that of Model 3 (3.15E-34). This implies that accounting for within-season variability in water clarity alone reduced the D-error by about a half.

The D-error from Model 1 was further compared to Model 4. Annual averages of water clarity are used in both models except that in Model 4 the bimonthly average of weight of fish was employed. The D-error from Model 1 is about 16 times higher than that of Model 4. The use of the bimonthly averages of weight of fish reduced the D-error substantially. These findings suggest that the higher D-error from Model 1 is largely induced by the annual average weight of fish, rather than water clarity.

Comparison of the D-errors from Models 2 and 4 further confirms this assertion. The bimonthly averages of weight of fish are used in both Model 2 and Model 4. The only difference is the use of the bimonthly averages of water clarity in Model 2 while the annual average is used in Model 4. The D-errors obtained from Model 2 (3.3E-35) and Model 4 (4.08E-35) are of similar magnitude. These results indicate that when the bimonthly average weight of fish is used, the D-errors from models utilizing the bimonthly and annual averages of water clarity are of similar magnitude. However, substantial differences in the D-error arise when the annual average weight of fish is used.

Overall the annual average weight of fish induced higher collinearity levels with other regressors in the model compared to its counterpart. On the other hand, when higher collinearity levels induced by the annual average weight of fish are controlled for, the collinearity level between models utilizing the annual and bimonthly averages of water clarity are generally similar.

Furthermore, the extent to which increased variability from the use of bimonthly averages of water clarity and weight of fish can enable the identification of unobserved effects was tested. The estimated results are presented in Appendix 7 (Table A7.2). The variability provided by the cost of lake access and bimonthly averages of water clarity and weight of fish was not sufficient to identify a full set of alternative specific constants. Only three alternative specific constants (ASC) could be included in Model 5, in which annual averages of water clarity and weight of fish are employed. However, the annual weight of fish, facility development and amount of forested land variables turned up with negative signs due to possible collinearity with unobserved effects. On the other hand, up to five ASCs could be estimated in Model 6 in which bimonthly averages were used. The bimonthly average weight of fish attribute is positive and significant as expected.

5.7 Comparisons of welfare estimates

The overall objective is to assess whether models accounting for and those ignoring within-season variability in water clarity and weight of fish give similar welfare estimates. In line with this objective, the marginal willingness to pay (MWTP) and changes in consumer surplus are compared. The MWTP from Models 1, 2, 3 and 4 are presented in Table 5.4 below.

Table 5.4: Comparison of MWTP from models using annual versus bimonthly averages of water clarity and weight of fish

	Model 1 SD_YR & FWEIGHT_YR	**Model 2** SD_2M & FWEIGHT_2M	**p-value** Model 1 vs 2	**Model 3** SD_2M & FWEIGHT_YR	**p-value** Model 2 vs 3	**Model 4** SD_YR & FWEIGHT_2M	**p-value** Model 2 vs 4
Variable	***MWTP1***	***MWTP2***	***MWTP1 > MWTP2***	***MWTP3***	***MWTP3 > MWTP2***	***MWTP4***	***MWTP4 > MWTP2***
Weight of fish	19.34 [10.99 28.44]	3.81 [1.74 5.97]	**0.0001**	14.93 [8.06 22.55]	**0.001**	3.84 [1.79 5.97]	0.491
Water clarity	2.66 [2.14 3.29]	1.81 [1.49 2.18]	**0.004**	2.08 [1.69 2.56]	0.162	2.09 [1.75 2.51]	0.137
Log of lake size	47.89 [37.85 59.84]	44.99 [36.08 55.53]	0.350	46.13 [36.58 57.67]	0.440	45.69 [36.77 56.35]	0.460
Facility development	4.95 [3.84 6.24]	4.73 [3.73 5.88]	0.400	4.82 [3.79 6.04]	0.457	4.66 [3.67 5.80]	0.541
Urban	5.00 [4.07 6.12]	4.40 [3.59 5.37]	0.193	4.76 [3.89 5.83]	0.295	4.47 [3.64 5.46]	0.460
Forest	0.20 [0.15 0.27]	0.16 [0.11 0.22]	0.138	0.17 [0.12 0.23]	0.367	0.18 [0.12 0.24]	0.321
Lake depth	0.81 [0.60 1.06]	0.51 [0.39 0.64]	**0.009**	0.69 [0.52 0.90]	**0.050**	0.53 [0.41 0.66]	0.414
Health warning	8.48 [4.68 12.41]	8.05 [3.96 12.15]	0.441	8.06 [4.15 12.14]	0.499	8.54 [4.42 12.71]	0.433
D-error	6.6E-34	3.3E-35		3.15E-34		4.08E-35	

Note: The numbers in [ ] are the 95% confidence intervals computed using Krinsky and Robb's (1986) procedures.

The figures in square brackets are the 95% confidence intervals for the MWTP computed using the procedures of Krinsky and Robb (1986). First, the MWTP from models using annual (Model 1) versus bimonthly (Model 2) averages of water clarity and weight of fish are compared. The MWTP for an additional kilogram of fish is 5 times higher in Model 1 than in Model 2. Similarly, the MWTP for water clarity is about 1.5 times that of Model 2. The other notable difference is the MWTP to avoid deeper lakes which is about 1.6 times higher in Model 1. The convolutions test (p-value) confirms that the distributions of the MWTP for weight of fish, water clarity and lake depth between the two models are significantly different from each other.

The MWTPs from Model 2 are compared to Model 3. Model 2 is used as a benchmark because it accounts for within-season variability in both water clarity and weight of fish attributes. The bimonthly water clarity attribute is employed in both models. The only difference is the use of the annual weight of fish attribute in Model 3. This was done to control for differences in attribute levels in the water clarity attributes. The MWTP for an additional kilogram of fish is still significantly higher for the annual weight of fish attribute compared to its counterpart in Model 2. The higher MWTP for annual weight can be attributed to the combined effect of the differences in attribute and collinearity levels. Since the two are confounded, it is difficult to isolate the two effects.

Additionally, the MWTPs from Model 2 are also compared to Model 4. The bimonthly average weight of fish is used in both models to control for differences in the weight of fish attribute levels. The only difference is in the attribute levels of water clarity through the use of annual averages in Model 4. Comparison of the willingness to pay values shows that the MWTPs for water clarity from Model 2 and 4 are not statistically different from each other. This demonstrates that the differences in attribute levels between the annual and bimonthly averages of water clarity do not lead to significant differences in MWTP if the difference in collinearity levels is controlled for. Furthermore, it can be deduced that the difference in the MWTP for the annual water clarity in Model 1 and its bimonthly counterpart in Model 2 was largely induced by higher collinearity levels between the annual averages of water clarity and weight of fish attributes.

Similarly, the statistically significant differences in the MWTP to avoid deeper lakes between Models 1 and 2 and also Models 2 and 3 can be attributed to higher collinearity levels between the annual weight of fish and lake depth attributes. When collinearity from the annual weight of fish attribute is controlled for in Model 4 this effect disappears.

In conclusion, welfare estimates between models that used annual (Model 1) and bimonthly (Model 2) averages of water clarity and weight of fish are compared. Table 5.5 presents predicted welfare estimates for a 1 metre rise in water clarity for all the lakes with poor and average water quality, concurrently and also individually.

Table 5.5: Welfare estimates from models using annual versus bimonthly averages of water clarity and weight of fish

	Model 1		Model 2		Model 1 vs 2
Lakes	**CS per angler**	**95% C.I**	**CS per angler**	**95% C.I**	**p-value**
Welfare estimates for a 1 metre rise in water clarity in all lakes with poor and average water quality concurrently					
All lakes listed below	104.36	[86.75 122.14]	71.42	[61.43 81.48]	**0.001**
Welfare estimates for a 1 m rise in water clarity in each of the lakes with poor and average water quality individually					
Rotoiti	57.95	[47.23 69.25]	40.51	[34.13 47.26]	**0.004**
Rotorua	37.56	[30.28 45.06]	23.73	[19.87 27.62]	**0.0004**
Rerewhakaaitu	5.72	[4.40 7.20]	3.90	[3.12 4.80]	**0.011**
Rotoehu	4.16	[3.16 5.36]	2.67	[2.12 3.32]	**0.007**
Okareka	2.11	[1.49 2.88]	1.49	[1.02 2.07]	**0.076**
Rotomahana	0.65	[0.46 0.89]	0.96	[0.14 0.80]	0.982
Okaro	0.01	[0.00 0.02]	0.01	[0.01 0.02]	0.592

Note: the figures in [ ] are the 95% confidence intervals computed using Krinsky and Robb (1986) procedures

Predicted welfare estimates are about 1.5 times higher in model 1, which used annual averages of water clarity and weight of fish attributes. The differences in welfare estimates are statistically significant except for smaller and less utilized lakes (Rotomahana and Okaro) as indicated by the significance of the convolutions test (p-value) generally falling far below 1%. This difference in welfare estimates is consistent with the differences in the MWTP for annual and bimonthly water clarity attributes in Models 1 and 2 (Table 5.4). As highlighted in the preceding discussion, this difference in welfare estimates is due to higher collinearity levels between the annual water clarity and weight of fish attributes.

5.8 Chapter summary

The main objective in this chapter was to assess the implications on welfare estimates of accounting for within-season variability in recreational site attributes that vary across the recreational season. Specifically, the chapter addressed the welfare effect of accounting for within-season variability in water quality and fish growth attributes. This was accomplished through the use of annual versus bimonthly averages of water clarity and weight of fish attributes. The bimonthly averages were used to account for within-season variability in water quality and fish growth.

Two possible sources of differences in welfare estimates between models using annual versus bimonthly averages of water and weight of fish were investigated.

(i) Differences in attribute levels
(ii) Differences collinearity levels

With regard to the first objective, the Rotorua Lakes are generally characterized by their variability of water quality across the year. On average, water quality tends to be poorer in summer and early autumn. This decline in water quality is often accompanied by algal blooms in lakes with poorer water quality, such as Lakes Okaro, Rotoehu and Rotorua. In addition, trout growth, (trout being the main fish species in the Rotorua Lakes), is said to improve during the autumn and winter seasons. To account for this variability, bimonthly averages of water clarity and weight of fish were used in the analysis and were contrasted to models utilizing annual averages. The bimonthly averages also aligned well with the fishing choice data, which were partitioned into bimonthly intervals from October - November (2007) up to August - September (2008).

Differences in collinearity levels were assessed by comparing the D-errors derived from the variance covariance matrix of the estimated parameters from the conditional logit models. The models that used the annual averages of weight of fish (Model 1 and 3) gave the highest D-errors that were about 16 to 20 times higher compared to Model 2 in which the bimonthly averages were used. These findings demonstrate significant differences in collinearity levels between the annual and bimonthly average weight of fish attributes.

On the contrary, there were no substantial differences in the collinearity level between annual and bimonthly water clarity attributes when the bimonthly average weight of fish was used in both models. The difference became substantial when higher collinearity between the annual weight of fish and water clarity attributes was not controlled for. It should also be highlighted that collinearity levels for all attributes are all within the acceptable range as demonstrated by generally very low standard errors. In terms of the bivariate correlation, the yearly average weight of fish attribute displayed relatively moderate to high levels of collinearity with the yearly average of water quality, lake size and lake depth attributes with scores of 0.29, 0.54 and 0.68, respectively. All of these scores are within the acceptable level of collinearity as shown in Appendix 7 (Table A7.1).

The effect on welfare estimates of failure to account for within-season variability was further investigated by comparing the MWTPs obtained from models using annual and bimonthly averages of water clarity and weight of fish. The MWTP for annual weight attribute was 5 times higher compared to its bimonthly counterpart. This difference was attributed to the combined effect of the differences in attribute and collinearity levels. However, it was difficult to isolate the two effects since the two were confounded.

In the case of water clarity it was possible to disentangle the separate effects of differences in attribute and collinearity levels on the MWTP. In the first scenario, only differences in attribute and collinearity levels between the annual and bimonthly water clarity were allowed, by using the bimonthly weight of fish attribute in both models. The MWTPs for the annual and bimonthly water clarity were not statistically different. These findings imply that differences in the attribute levels in the bimonthly and annual water clarity attributes did not have any significant effects on welfare estimates. Furthermore, the collinearity level between the bimonthly and annual water clarity attributes were found to be of similar magnitude when the bimonthly weight of fish was used in both models (Models 2 and 4). The second scenario allowed for differences in attribute and collinearity levels in both the water clarity and weight of fish attributes. The MWTP for the annual water clarity was found to be significantly higher than that of bimonthly water clarity attribute. The higher MWTP for the annual water clarity attribute was largely induced by higher collinearity levels with the annual weight of fish attribute.

Similarly, the higher collinearity level between the annual weight of fish and lake depth had a direct effect on the MWTP. It was found that the MWTP to avoid deeper lakes was higher in the model using annual averages of weight of fish compared to the model in which the bimonthly weight of fish attribute was used. When collinearity from the annual weight of fish attribute was controlled for this effect disappeared.

In addition, welfare estimates for a 1 metre increase in water clarity between models using annual and bimonthly averages of water clarity and weight of fish were also compared. Welfare estimates for water clarity were found to be 1.5 times higher in models using annual averages of water clarity and weight of fish attributes. The differences in welfare estimates were statistically significant except for smaller and less utilized lakes. This difference was mainly attributed to differences in collinearity levels induced by the annual weight of fish attribute.

Overall findings from this chapter have demonstrated that accounting for within-season variability in recreational site attributes can have a significant effect on welfare estimates. The major potential gain from accounting for within-season variability might be reduced collinearity through the use of less aggregated data. The results further illustrate that even the relatively low to moderate levels of collinearity typically tolerated in revealed preference studies can have an effect on welfare estimates. In the absence of a counterfactual, these effects remain latent and unexplored. These findings are pertinent in travel cost studies where collinearity among regressors is ubiquitous.

CHAPTER SIX

TESTING THE STABILITY OF WELFARE ESTIMATES OVER TIME

6.0 Introduction

This chapter extends the preceding work by carrying out an assessment of whether welfare estimates remain stable over time. While the spatial transferability of values has received considerable scrutiny in various fields, with divergent results, the transferability of values over time has received relatively little attention. Assessing the stability of values over time is vital because stated preference studies only provide a snapshot of values at a particular point in time. However, policy analysts are often required to extrapolate these values to future scenarios.

Studies conducted to explore the stability of values over time report mixed results. A number of factors are said to contribute to changes in values over an extended period of time, including changes in preferences, choice sets, economic and other social contextual factors. More recently, empirical evidence from the field of transportation seems to suggest that scale heterogeneity across sampled individuals can strongly affect differences in mean estimates of the value of travel time saving across studies. It has also been noted that studies that assumed scale homogeneity might have erroneously concluded that mean WTP estimates for travel time saving were transferable between studies. This conclusion may have been caused by failure to account for scale heterogeneity in the sampled population (Hensher *et al.*, 2011).

It appears that all environmental non-market valuation studies testing the stability of values over time have used models that assume scale homogeneity across respondents. This chapter explores the extent to which scale heterogeneity across individuals can contribute to differences in welfare estimates across data sets. The work presented here represents one of the first environmental non-market valuation studies to investigate this issue. The availability of two independent fishing choice data sets for the Rotorua Lakes, collected six years apart, allowed this investigation to be carried out.

In the next section a review of literature on the subject is provided. This is followed by an outline of methodology and a description of the data. A discussion of the estimated results and comparison of welfare estimates is carried out in the remainder of the chapter.

6.1 Temporal stability of environmental values

Much of the growing interest in the stability of values emerged following the introduction of the contingent valuation method (CVM). Temporal stability of values is usually considered to be an indicator of the reliability of a valuation instrument because the values can be reproduced in follow-up experiments (Bliem *et al.*, 2012; Carson *et al.*, 2001a; Loomis, 1989; Mitchell & Carson, 1989). Stability of values is also important because stated preference studies only provide a snapshot of values at a particular point in time. On the other hand, policy analysts are often required to extrapolate these values to some future time period (Liebe *et al.*, 2012; Loomis, 1989). Benefit transfer applications, which are often undertaken with a considerable time lag, represent one such scenario.

Interest in the stability of values over time has spanned many fields, including environmental, transportation and health economics. The stability of values is predominantly assessed using a reliability test referred to as a test-retest of the valuation instrument. It involves the repeated administration of the survey to the same subjects or to different samples from the same population over two or more distinct time periods. The time interval may range from a few weeks to several years. A test-retest with a very short time interval is generally not considered to be a true test of reliability because of the high likelihood of carry-over or recall effects (Liebe *et al.*, 2012; Teisl *et al.*, 1995). Some approaches suggest reducing the recall effects by conducting the second test after a sufficiently long time lag, using a different sample, or using an alternative form of valuation question. On the other hand, if the time interval for a test-retest is long there is a high likelihood that respondents' values may actually change. Either way "a reliable […] instrument is the one that reflects the constancy of values when preferences and choice sets do not change, and reflects changes in values when preferences or choice sets have changed" (Teisl *et al.*, 1995, p. 614).

To gain further insight on the subject, a review of some of the studies conducted with an emphasis on environmental applications is provided in the remainder of this section. A study by Loomis (1989) is one of the earliest applications to test the stability of environmental values over time. The reliability of the CVM was assessed by a test-retest of two target populations, concerning WTP for water quality in Mono Lake in California. In the first survey a sample of California households was used. The retest sample consisted of visitors to Mono Lake contacted on the site. The initial survey was conducted in 1986 and was followed by a retest in 1987, allowing a nine-month interval between the surveys. The estimated WTP values for various water quality levels showed evidence of preference stability between the two periods.

Reiling *et al.* (1990) assessed the stability of estimates of WTP for the control of black flies along a section of the Penobscot River in Maine using household data. Two split samples were used to control for carry-over effects, in which respondents could repeat the responses given in the previous survey. The contingent valuation survey was administered to one half of the sample during the peak black fly season in August and September 1987. The other half of the sample answered the same survey after the black fly season in late October and November 1987. The authors reported similar mean WTP between the two periods. They also noted that there were only six published studies testing the reliability of contingent valuation values, in contrast to a large number of validity studies.

Stevens *et al.* (1994) investigated the temporal stability of existence values for bald eagles in New England over a three year duration, from 1989 to 1992 using the same sample of respondents. The study results showed evidence of stability of WTP values over time.

The study by Cameron (1997) assessed respondents' WTP to improve water quality in the Hawkesbury-Nepean river to a safe level for recreation and watering stock. The same CVM questionnaire was presented to the same group of respondents at yearly intervals from 1993 to 1995. The findings indicated no significant differences in mean WTP over time.

The CVM studies reviewed so far had a relatively short test-retest period of less than 3 years. Using a longer time span, Whitehead & Hoban (1999) used two samples drawn from the same population to test the stability of WTP for an improvement in water pollution and air quality in Gaston County over a five year period. The first survey was administered in 1990 followed by a retest in 1995. It was found that respondents in a retest group had less favourable attitudes towards the environment. After accounting for the change in attitudes, they found that the 1990 and 1995 values were not significantly different from each other.

Similarly, Brouwer & Bateman (2005) compared WTP for flood control and wetland conservation in the Norfolk Broads in the UK across a five-year period (1991 and 1996), and found that WTP estimates changed significantly over time[63]. They also noted that the stability of values over time was mostly reported in CVM studies with a relatively shorter test-retest period, ranging from 2 weeks to 2 years.

More recently, Bliem & Getzner (2012) investigated the stability of WTP bids for river restoration in the Danube National Park in Austria from two identical surveys employed one year apart. The contingent valuation web-based surveys were conducted in November 2007 and December 2008 using two samples with similar socioeconomic characteristics. The study results indicated temporal stability of preferences for river restoration between the two periods.

[63] The CVM survey was applied to the same sample population.

In contrast, choice experiment applications testing the stability of values in environmental non-market valuation are sparse. The study by Bliem *et al.* (2012) is one of the first choice experiment study to test the stability of values studies in environmental valuation. They assessed the stability of people's preferences for river restoration in Austria using two identical web-based choice experiment surveys that were administered to two independent samples with a one-year lag. The first survey was carried out in 2007 and the second one in 2008. The authors did not find any significant difference in WTP estimates between the two surveys.

Another test-retest choice experiment was carried out by Liebe *et al.* (2012) on landscape externalities of onshore wind power in Central Germany. The survey was presented to the same respondents with a one-year lag. Findings from the study indicate that preferences were fairly stable between the two periods.

Studies investigating the stability of values in the recreational demand literature using revealed preference methods are also limited. Two of these studies are reported here. Bhattacharjee *et al.* (2009) used Kuhn-Tucker demand models to test the stability of households' recreational demand at Iowa lakes. The test-retest surveys were carried out in 2002 and 2003 using the same sample of households. They found that the null hypothesis of stability of recreational demand over time could not be rejected.

Parsons & Stefanova (2009) used trip data sets for Delaware residents to beaches in the Mid-Atlantic region collected in 1997 and 2005 to test the stability of recreational preferences over time. Two different samples were used and their study results showed evidence of qualitative stability in consumer preferences over time.

Overall, as noted by Brouwer & Bateman (2005), the stability of values over time is mostly reported in studies with a relatively short test-retest period, ranging from 2 weeks to 2 years. In contrast, the stability of environmental values in studies with a test-retest period of five or more years shows mixed results. A number of factors can contribute to changes in preferences over an extended period of time, including changes in preferences, choice sets, economic and other social contextual factors (Habib *et al.*, 2013; Teisl *et al.*, 1995).

Additionally, recent empirical evidence from the field of transportation seems to suggest that scale heterogeneity might contribute to differences in mean estimates of WTP across studies. Hensher *et al.* (2011, 2012) compared the value of travel time saving (VTTS) from seven data sets; five Australian and two New Zealand toll road studies conducted between 1999 and 2008. The choice experiment studies were very similar in content and design. Their main objective was to investigate whether there was "greater synergy in the WTP evidence within model form across comparable data sets compared to cross model forms within data sets" (Hensher *et al.*, 2011, p. 1). They found that scale heterogeneity in scaled multinomial logit (S-MNL) and generalized mixed multinomial logit (G-MNL) models appeared to "inordinately contribute more to differences in mean estimates of VTTS across studies" than preference heterogeneity in mixed multinomial logit (MMNL) models (Hensher *et al.*, 2011, 2012).

Precisely, Hensher *et al.* (2011, p.10) reported:

> Empirical evidence seems to suggest that scale heterogeneity appears to exert a greater influence on producing differences in mean estimates of VTTS across studies than does preference heterogeneity (as accounted for in MMNL while ignoring scale heterogeneity). If as it appears, this is the empirical situation, then previous studies that have ignored scale heterogeneity have in effect increased the chance of transferability of VTTS when in fact this is misleading as a consequence of failing to recognise scale heterogeneity in the sampled population.

To the best of my knowledge studies testing the stability of values over time in environmental economics have used models that assume scale homogeneity across respondents. The main question addressed in this chapter is whether welfare estimates remain stable over time. The extent to which scale heterogeneity can contribute to differences in welfare estimates across data sets is also explored. The work in this chapter is the first to explore the stability of values over time by using models that account for scale heterogeneity and those that do not. The availability of two independent fishing choice data sets for the Rotorua Lakes, collected six years apart, permit this investigation to be carried out. The methodology used is provided in the subsequent section.

6.2 Methods

Swait & Louviere (1993) were the first to recognize that parameter estimates in MNL models from different data sets may differ in magnitude due to scale factor differences. Recently, it has been argued that much of the taste heterogeneity typically assumed in MMNL models choice applications can be better described as scale heterogeneity[64] (Louviere, 2001; Louviere & Eagle, 2006; Louviere & Meyer, 2007; Louviere *et al.*, 1999). Typically, the scale and utility weights are confounded and cannot be separately identified unless specific reparameterisations, and hence assumptions, are implemented. This problem is circumvented in logit model estimation by normalising the scale or standard deviation of the idiosyncratic error to a constant. More recently, models that allow for scale heterogeneity to be accounted for at individual level have been developed. Fiebig *et al.* (2009) proposed the estimation of the Generalized Multinomial Logit Model (G-MNL) accounting for both scale and preference heterogeneity using a specific set of assumptions and attendant reparameterisation. The G-MNL is a mixed logit specification that allows for heterogeneity both in error scale and attribute preferences. Greene & Hensher (2010) specify the G-MNL model building on the G-MNL model by Fiebig *et al.* (2009) and mixed logit models by Train (2003). Assuming individual i chooses alternative j in choice situation t, Greene & Hensher (2010, pp. 414-417) specify the G-MNL model as follows starting with the mixed logit model.

[64] In fact they argue that normal mixing distributions used in MMNL models may be seriously mis-specified.

$$Prob(choice_{it} = \ j|x_{itj}, z_i, v_i) = \frac{exp(v_{itj})}{\sum_{j=1}^{J_{it}} exp(v_{itj})} \tag{1}$$

where $v_{itj} = \ \beta_i' x_{itj}$

$\beta_i = \ \beta + \Delta z_i + \Gamma v_i$

x_{itj} = the K attributes of alternative j in choice situation t faced by individual i

z_i= a set of M characteristics of individual i that influence the mean of the taste parameters; and

v_i = a vector of K random variables with zero means and known (usually unit) variances and zero covariances.

The mixed logit formulation above captures both observed heterogeneity, Δz_i and unobserved heterogeneity in preferences, Γv_i. The basic MNL model is derived by assuming $\Delta = 0$ and $\Gamma = 0$.

The G-MNL is obtained by accommodating scale heterogeneity across individuals in the mixed logit model above through random specific constants. The model in equation (1) is modified as follows:

$$\beta_i = \ \sigma_i[\beta + \Delta z_i] + \ [\gamma + \sigma_i(1-\gamma)]\Gamma v_i \tag{2}$$

where σ_i is the individual specific standard deviation of the idiosyncratic error term

$\sigma_i = exp(\overline{\sigma} + \delta' h_i + \ \tau w_i)$

h_i = is a set of M characteristics of individual i and may overlap with z_i,

δ = parameters in the observed heterogeneity in the scale term

w_i = the unobserved heterogeneity which is assumed to be standard normally distributed

$\overline{\sigma}$ = the mean parameter in the variance

τ = the coefficient of the unobserved scale heterogeneity

γ = a weighting parameter that indicates how variance in residual preference heterogeneity varies with scale, with $0 \leq \gamma \leq 1$.

“The weighting parameter, γ, is central to the generalized model. It controls the relative importance of the overall scaling of the utility function, σ_i, versus the scaling of the individual preference weights contained in the diagonal elements of Γ” (Greene & Hensher, 2010, p. 415). If $\gamma = 0$, the G-MNL model reverts to the scaled mixed logit model.

$$\beta_i = \sigma_i[\beta + \Delta z_i + \Gamma v_i] \quad (3)$$

The Scaled MNL model[65] is derived by assuming $\Delta = 0 \; and \; \Gamma = 0$

$$\beta_i = \sigma_i \beta \quad (4)$$

The G-MNL model or any other model forms in equations (3) and (4) above are estimated by maximum simulated likelihood. Fiebig *et al.* (2009) and Greene & Hensher (2010) give a detailed discussion of the complications that arise in model estimation. They note that $\overline{\sigma}$ is not separately identified from τ. To identify the model σ_i is normalized so that $E[\sigma_i^2] = 1$. This is achieved by letting $\overline{\sigma} = -\tau^2/2$ instead of zero. Furthermore, to ensure non-negative values of τ, “the model is fit in terms of λ, where $\tau = exp(\lambda)$ and λ is unrestricted” (Hensher *et al.*, 2011, p. 6).

Greene & Hensher (2010, p. 417) specify the simulated log likelihood function as follows:

$$logL = \sum_{i=1}^{N} log \left\{ \frac{1}{R} \sum_{r=1}^{R} \prod_{t=1}^{T_i} \prod_{j=1}^{J_{it}} P\left(j, x_{it,}\beta_{ir}\right)^{y_{itj}} \right\} \quad (5)$$

where r=1,…,R are the draws required for simulation

[65] In the basic MNL model, the standard deviation of the idiosyncratic error term is assumed to be homogenous across the sampled individuals, $\sigma_i = \sigma$; therefore, $\beta_i = \sigma\beta$. It is standard practice to normalize σ to 1, since it is not possible to identify both β and σ.

$$\beta_{ir} = \sigma_{ir}\big[[\beta + \Delta z_i] + \ [\gamma + \sigma_{ir}(1-\gamma)]\Gamma v_{ir}\ \big] \tag{6}$$

$$\sigma_{ir} = exp(-\tau^2/2 + \delta' h_i + \ \tau w_{ir})$$

where v_{ir} and w_{ir} are the simulated draws on v_i and w_i, respectively

y_{itj} equals 1 if individual i chooses alternative j in choice situation t and zero otherwise

The Scaled MNL model is derived by assuming $\Delta = 0\ and\ \Gamma = 0$ and accordingly equation (6) reduces to:

$$\beta_{ir} = \ \sigma_{ir}\beta \tag{7}$$

The probability of individual i choosing alternative j in choice situation t is given by:

$$P(j, x_{it}, \beta_{ir}) = \frac{exp\big(x'_{itj}\beta_{ir}\big)}{\sum_{j=1}^{J_{it}} exp\big(x'_{itj}\beta_{ir}\big)} \tag{8}$$

The G-MNL model also offers a convenient way of reparameterising the model to estimate the taste parameters in WTP space. WTP space models are said to be behaviourally appealing alternative ways of directly obtaining an estimate of WTP over preference space models, where WTP is obtained indirectly as the ratio of the non-monetary attributes to the cost parameter[66]. Recent application of WTP space models include studies by Train & Weeks (2005), Sonnier *et al.*, (2007), Scarpa *et*

[66] Estimating models in preference space poses some challenges in panel mixed logit models if taste heterogeneity is assumed for both the cost and non-monetary attributes. This includes obtaining counter-intuitive distributions of WTP values. This can, for example, include the use of the normal and log-normal distribution for the non-monetary and cost attributes, respectively. It is further demonstrated that for most distributions, values of the cost coefficient close to zero may cause the ratio to be very large, causing the WTP distributions to have an excessively long upper tail. The resultant mean and variance may be much higher than otherwise expected (Scarpa *et al.*, 2008b; Train & Weeks, 2005).

al. (2008b) and Hole & Kolstad (2012). A specific discussion of the advantages that this reparameterisation offers in testing hypotheses on WTP distributions in the estimation stage is provided by Thiene & Scarpa (2009b).

Empirical evidence has shown that the G-MNL model is superior to the S-MNL model since it accommodates both scale and preference heterogeneity (Fiebig *et al.*, 2009; Greene & Hensher, 2010). However, the S-MNL model always provides a model fit at least as good as the MNL model, as the latter is a special case of the former.

In this application the S-MNL model is used. The G-MNL is best suited for panel data sets with repeated choice observations. The fishing choice data used in this application is an unbalanced panel data set with a large proportion of anglers reporting visiting the lakes only once over the fishing season. However, this does not mean that such anglers visited the Rotorua lakes only once during the year, but it may simply imply that they were not included in the other sub-samples, since re-sampling was done at two-monthly intervals. The WTP obtained from the S-MNL is compared to that of the MNL models. Model specifications concerning the MNL models are discussed in the previous chapters. A detailed description of the data is presented in the following section.

6.3 Data sources

Two data sets from the New Zealand National Angling Survey that was conducted during the 2001/02 and 2007/08 fishing seasons are used in this chapter. These surveys were carried out jointly by NIWA and FGNZ. The main objectives of the surveys were to obtain consistent estimates of angler usage for all New Zealand lake and river fisheries managed by FGNZ.

Both were telephone sample surveys, based on random samples of anglers drawn from records of fishing licence sales for the angling season, which spans from 1 October to 30 September of each year. Licence holders were asked to identify lakes and rivers they had fished over the previous two months, and the number of days spent on each water. The 2001/02 angling survey was limited to New Zealand residents only, while the 2007/08 survey also included overseas anglers[67]. The surveys were stratified by FGNZ region, time (with the 12 month survey period divided into six two-monthly intervals), and licence type. Licence strata included adult whole-season and family licences, young adult and junior whole season licences and part-season licences (Unwin, 2009; Unwin & Image, 2003).

For the 2001/02 the survey population was limited to the subset of licence holders who were able to be communicated with by telephone. A total of 19,098 licence holders were contacted, of whom 10,847 (56.8%) had fished in at least one of the recognised lake and river fisheries during the two-month survey period of interest (Unwin & Image, 2003).

The 2007/08 survey consisted of a random sample of 17,739 anglers drawn from a population of 97,215 fishing licence holders. Out of this total, 84,875 were New Zealand resident anglers and 12,340 were overseas anglers (Unwin, 2009).

As highlighted in Chapter Four, these surveys did not collect all the information that may be necessary for modelling recreational site choice because such information was not in line with their study objectives. No information was collected on whether the fishing trips undertaken were day trips or involved an overnight stay, or whether fishing trips were single or multi-purpose. Furthermore, no information is available on whether or not anglers fished in more than one water body during a reported day of fishing. Also missing from the angling survey is information on the amount of time spent fishing on a particular lake[68]. As noted by Phaneuf & Smith (2003), all this information might have implications on how to measure the resources given up in order to access the recreational site.

[67] Overseas anglers were contacted by email

[68] Information on social economic demographic factors was not collected in the 2001/02 National Angling Survey. In the 2007/08 National Angling Survey, only data on age and gender is available for a limited number of anglers.

The angling surveys have been adapted to suit this study in the following ways: The main focus in this application is on single day fishing trips and individual level choice data. To meet these criteria only adult individual licence holders who lived within 240 km of the lakes are included in the sample. This distance measure is considered to be a reasonable benchmark for day trips (McConnell & Strand, 1994; Parsons & Kealy, 1992).

A sample of 524 and 414 anglers fulfilled these criteria for the 2001/02 and 2007/08 fishing seasons, respectively. The total number of fishing days for these samples compared to the total angling days reported in the National Angling Surveys are presented in Table 6.1 below. In total 2,200 and 2,292 fishing days were reported for the 2001/02 and 2007/08 samples, respectively.

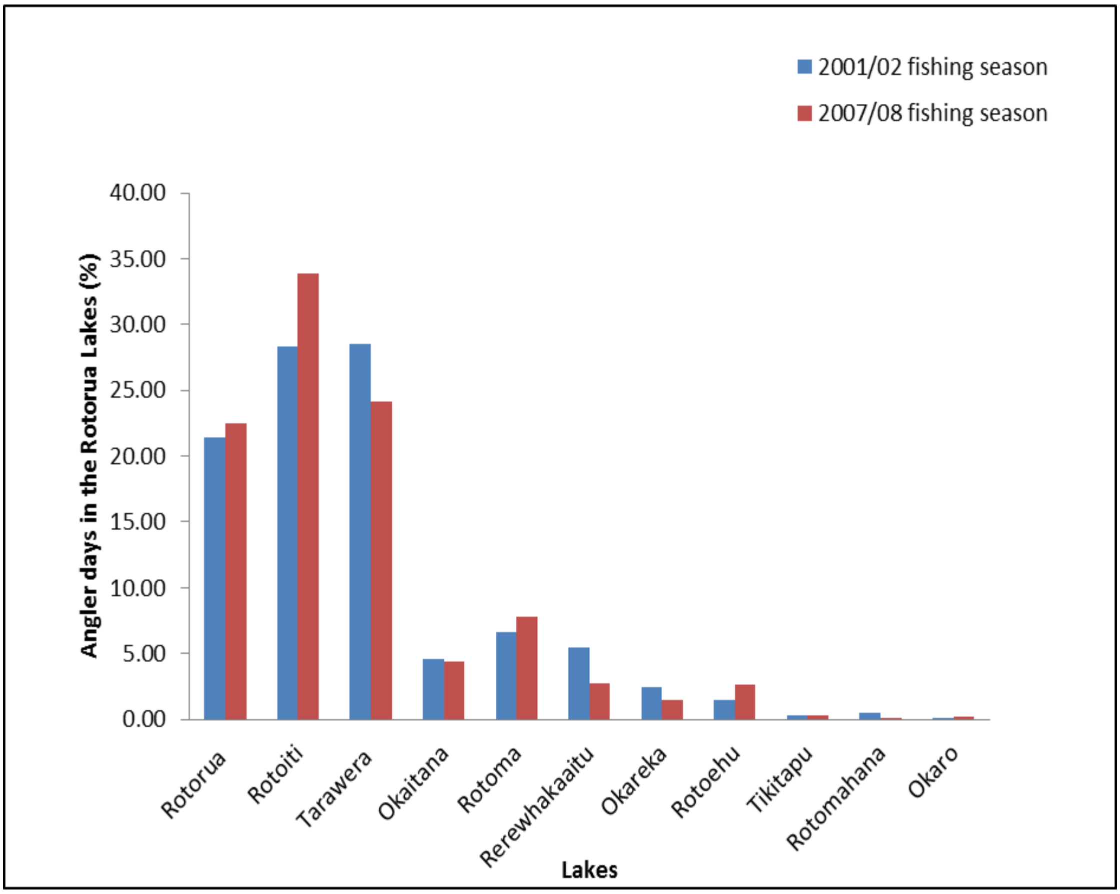
Table 6.1: Estimated angler days for the 2001/02 and 2007/08 National Angling Surveys versus samples utilised in this study

Lake Name	2001/02 National Angling Survey estimated angler-days ± 1 standard error	Number of fishing days for the 2001/02 sample	2007/08 National Angling Survey estimated angler-days ± 1 standard error	Number of fishing days for the 2007/08 sample
Rotoiti	43080 ± 3120	668	48070 ± 3710	673
Tarawera	43480 ± 2940	863	34220 ± 3440	548
Rotorua	32640 ± 2580	748	32000± 3200	583
Rotoma	10130 ± 1260	76	11110 ± 2040	233
Okaitana	7050 ± 890	192	6290 ± 1070	95
Rerewhakaaitu	8380 ± 1320	169	3830 ± 800	99
Rotoehu	2190 ± 770	52	3720 ± 1210	33
Okareka	3750 ± 1240	82	2040 ± 530	19
Tikitapu	470 ± 190	7	370 ± 140	3
Okaro	200 ± 120	4	260 ± 170	5
Rotomahana	820 ± 380	7	70 ± 50	1
Total		**2200**		**2292**

Source: Unwin & Image (2003) and Unwin (2009)

In Figure 6.1 below the estimated angler days on the Rotorua Lakes for the 2001/02 and 2007/08 National Angling Surveys are further compared.

Figure 6.1: Angler days at each lake as a percentage of the total angling days at the Rotorua Lakes



The distributions of angling days for the 2001/02 and 2007/08 National Angling Surveys are broadly similar. In both surveys the most angling days were reported for lakes Rotoiti, Tarawera and Rotorua.

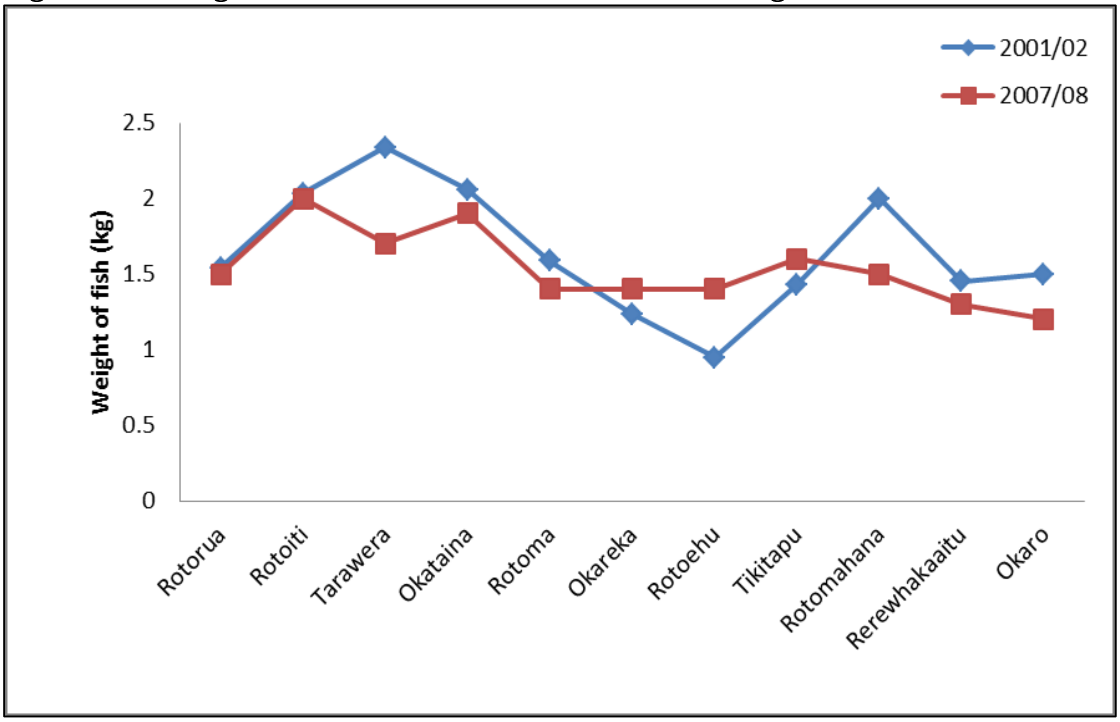
The estimated angler days reported in the National Angling Survey are used as a benchmark for the true population distribution since the surveys were designed following random sampling procedures. From Table 6.1 above, there is a clear indication that the samples employed in this application either over-state or under-state the true distribution of fishing days across the lakes. To account for under-sampling and over-sampling, choice-based sampling techniques were used, following procedures outlined by Hensher *et al.* (2005).

Lake attribute data

The lake attribute data used in this chapter are the same as those described in Chapters 2 and 3 with a few exceptions. They include travel cost, weight of fish, water clarity, size of lake, urban development, facility development, amount of forested land and lake depth. The lake size and depth attributes are invariant across time. Similarly, urban development, facility development and amount of forested land attribute levels were generally constant between the two study periods. On the other hand, there were some slight changes in weight of fish and water clarity for some lakes between the two periods as further elaborated below.

Travel cost includes the cost of fuel expenses only, unlike in Chapters 2 and 3 where the opportunity cost of travel time was accounted for. This is because information on income was not collected in both surveys. In the previous chapters the opportunity cost of travel time for the 2007/08 fishing choice data was calculated based on estimated median income from the 2006 census data. The median income data from the 2001 census would be the most appropriate for the 2001/02 fishing choice data. However, in both surveys the address fields were very broad, so determining consistent area units from which to derive the income data was not possible. To avoid any possible biases this might cause, the opportunity cost of travel is not included and hence the welfare estimates derived are to be considered conservative lower bounds on the real values. The cost of fuel was estimated at NZ$0.12 and NZ$0.19 per kilometre for the 2001/02 and 2007/08 fishing choice data, respectively. The 2001/02 fishing trip costs were recalculated in 2008 New Zealand dollars using the all price consumer index. The procedures for calculating travel costs are outlined in Chapter Four (section 4.4). The weight of fish and water clarity for the two study periods are compared in Figure 6.2.

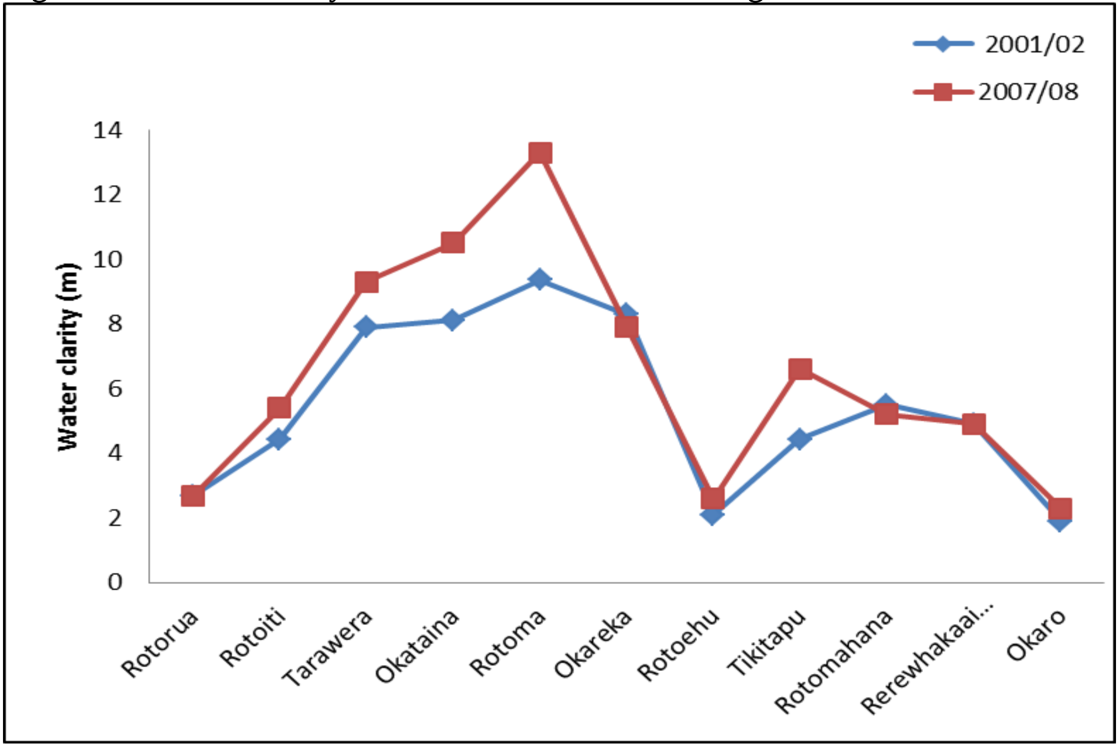
Figure 6.2: Weight of fish for 2001/02 and 2007/08 fishing seasons



Source of data: Eastern Region Fish and Game Region

The annual average weights of fish were generally similar for most lakes, except for Lakes Tarawera, Rotoehu and Rotomahana. For Lake Tarawera the annual average weight of fish was 2.4 kg in the 2001/02 fishing season compared to 1.6 kg during the 2007/08 fishing season. Lake Rotoehu registered an improvement in the average weight of fish from about 1 kg during the 2001/02 fishing season to 1.4 kg in the 2007/08 fishing season. There was a decline in the average weight of fish for Lake Rotomahana from 2 kg in the 2001/02 fishing season to 1.5 kg during the 2007/08 fishing season. The average weight of fish for the 2001/02 fishing season was also compared to that of the previous fishing season (2000/01 fishing season) and was found to be consistent across lakes. Similarly, the average weight of fish for the 2006/07 and 2007/08 fishing seasons was also consistent. In Figure 6.3 the annual average water clarity during the two survey periods are compared.

Figure 6.3: Water clarity for 2001/02 and 207/08 fishing seasons



Source of data:(Scholes, 2009)

There was a slight improvement in water clarity for Lakes Rotoiti, Tarawera, Okataina, Rotoma and Tikitapu. For the other lakes water clarity remained stable during the two periods. Improvements in water clarity occurred in lakes which already had good water quality. In general, it is anticipated that an improvement in water clarity in lakes with poorer water quality would be more valued. To account for variability in these attributes between the two study periods in the estimation, year-specific averages of weight of fish and water clarity are used.

The summary statistics for the lake attributes are presented in Table 6.2 below. A more detailed description of how these variables are measured and entered in the utility function was presented in Chapter Four (section 4.4).

Table 6.2: Summary statistics of the Rotorua Lakes attributes

Variable	Mean	Std.Dev.	Minimum	Maximum
Weight of fish (kg) (2001/2002 fishing season)	1.65	0.41	0.95	2.43
Weight of fish (kg) (2007/2008 fishing season)	1.54	0.23	1.2	2.0
Water clarity (metres) (2001/2002 fishing season)	5.42	2.65	1.90	9.36
Water clarity (metres) (2007/2008 fishing season)	6.39	3.36	2.3	13.3
Lake size (square km)	18.71	23.31	0.31	80.6
Number of boat ramps[69]	2.27	2.00	1	7
Number of access points	2.36	2.06	0	7
Depth (metres)	29.33	19.68	7	60
Urban development (% of lake catchment area)	1.41	2.27	0	8.1
Amount of forested land (% of lake catchment area)	56.82	26.53	6	94

The results are outlined in the remainder of the sections in this chapter.

6.4 Estimated results

The estimated models for the 2001/02 and 2007/08 fishing choice data are presented in Table 6.3. Parameters are estimated by maximum likelihood estimation procedures in Nlogit 4.0.

[69] Boat ramps and number of access points are highly collinear and therefore, boat ramps are used as a proxy for recreational facility development around the lakes.

Table 6.3: Estimated results for the 2001/02 and 2007/08 fishing seasons

	2001/02 fishing season				**2007/08 fishing season**			
	Model 1		**Model 2**		**Model 3**		**Model 4**	
	MNL		**S-MNL**		**MNL**		**S-MNL**	
Variable	*Coefficient*	|*t-value*|	*Coefficient*	|*t-value*|	*Coefficient*	|*t-value*|	*Coefficient*	|*t-value*|
Travel cost	-0.166***	22.08	-0.167***	22.12	-0.090***	13.56	-0.121***	9.75
Weight of fish	0.301***	3.02	0.296***	2.97	0.310***	3.88	0.303***	3.0
Water clarity	0.169***	10.82	0.174***	11.16	0.168***	13.69	0.282***	6.11
Lake size	1.978***	16.22	2.033***	16.94	2.946***	14.13	4.533***	5.76
Urban development	-0.282***	13.05	-0.288***	13.31	-0.343***	10.39	-0.509***	4.93
Facility development[70]	0.348***	19.57	0.349***	19.27	0.289***	12.96	0.443***	5.87
Amount of forested land	0.001	0.21	0.001	0.57	0.014***	7.28	0.025***	4.32
Lake depth	-0.035***	9.65	-0.036***	9.87	-0.042***	10.19	-0.070***	4.36
Scale parameter (τ)			0.020	0.51			0.633***	6.19
Summary statistics								
Log-Likelihood	-4641.482		-4638.241		-3830.147		-3824.373	
Mc Fadden R-Squared	0.265		0.271		0.273		0.282	
Number of respondents	524		524		414		414	

***, **, * implies significant at 1%, 5% and 10% level, respectively.

[70] Facility development is the average of boat ramps and number of access points to the lakes because the two attributes were highly correlated and could not enter the utility specification separately.

Models 1 and 2 consist of the estimated results for the 2001/02 fishing season from the MNL and S-MNL models, respectively. The estimated results for the 2007/08 fishing season are presented in Models 3 and 4. In terms of model performance, the S-MNL models perform slightly better than the MNL models in both data sets as indicated by both the log-likelihood and McFadden R-squared.

For the 2001/02 model, the average annual weight of fish variable has a negative sign, probably due to collinearity. Therefore, the previous year bimonthly average weight of fish is used in both data sets instead. On the other hand, water clarity levels corresponding to the current fishing year are used[71]. In all the models the travel cost coefficient is negative and highly significant, implying that anglers preferred lakes that were closer to their homes. Urban development and lake depth are negative and highly significant in both models. These findings suggest that in general lakes with more urban development and deeper ones were less preferred by anglers. Furthermore, the results show that lakes with bigger fish, better water clarity, larger size and more recreational facilities were generally preferred, as indicated by positive and highly significant coefficients for these attributes. On the other hand, the coefficient for the amount of forested land is positive but significant only in Models 3 and 4 (2007/08 fishing season).

The scale parameter (τ) for the S-MNL model is only significant in the 2007/08 fishing choice data, implying greater scale heterogeneity in the 2007/08 sample than the 2001/02 sample. Comparison of parameter estimates obtained from different samples is impossible without accounting for scale factor differences (Hensher, 2012; Swait & Louviere, 1993). Parameter estimates for the S-MNL models (Model 2 and 4) can be compared. Since the concern in this chapter is to test the null hypothesis of equality of welfare estimates, the equality of utility weights is of less concern. In the remainder of the chapter the equality of welfare estimates is tested.

[71] Water clarity in the previous and current recreational fishing years was similar.

Comparison of welfare estimates

The marginal WTP values measured by the ratio of the non-monetary attributes to the travel cost coefficient are presented in Table 6.4 below.

Table 6.4: Comparison of marginal WTP values

	MNL 2001/02 sample			MNL 2007/08 sample			S-MNL 2001/02 sample			S-MNL: 2007/08 sample		
Variable	*MWTP*	*95% Confidence Interval*		*MWTP*	*95% Confidence Interval*		*MWTP*	*95% Confidence Interval*		*MWTP*	*95% Confidence Interval*	
Weight of fish	1.81	[0.64	2.99]	3.43	[1.68	5.18]	1.78	[0.60	2.96]	2.53	[0.94	4.13]
Water clarity	**1.02**	**[0.81**	**1.22]**	**1.86**	**[1.56**	**2.16]**	**1.05**	**[0.84**	**1.25]**	**2.33**	**[1.82**	**2.84]**
Lake size	**11.89**	**[10.03**	**13.74]**	**32.63**	**[26.06**	**39.20]**	**12.24**	**[10.39**	**14.09]**	**37.68**	**[27.82**	**47.54]**
Urban development	1.69	[1.42	1.97]	3.80	[2.95	4.66]	**1.73**	**[1.46**	**2.00]**	**4.23**	**[2.88**	**5.58]**
Facility development	2.09	[1.84	2.35]	3.21	[2.58	3.84]	**2.10**	**[1.84**	**2.36]**	**3.68**	**[2.70**	**4.66]**
Amount of forested land	-	-	-	0.15	[0.11	0.20]	-	-	-	0.21	[0.13	0.29]
Lake depth	0.21	[0.16	0.25]	0.46	[0.35	0.57]	**0.21**	**[0.17**	**0.26]**	**0.58**	**[0.37**	**0.79]**

Figures in [ ] are the 95% confidence intervals

Figures in bold imply significant differences in the mean WTP estimates

The marginal WTP and confidence intervals were estimated by simulating approximate distributions of WTP estimates using the Krinsky–Robb procedure with 5000 draws (Krinsky & Robb, 1986).

In general, anglers in the 2007/08 sample were willing to pay more to access lakes with bigger fish, better water clarity, bigger lakes, with more recreational facilities and with more forest cover in the catchment area[72]. In the case of MNL models, the mean WTP for an additional kilogram of fish is 90 percent higher in the 2007/08 sample. The mean WTP for clarity is 82 percent higher, 174 percent higher for bigger lakes and 53 percent higher for more recreational facilities in the 2007/08 sample. The mean WTP to avoid lakes with more urban development and deeper lakes is also higher in the 2007/08 sample. The mean WTP values are 125 and 119 percent higher, respectively. However, the null hypotheses of equality of mean WTPs across models within the same data set and across data sets need testing.

The mean WTP obtained from MNL and S-MNL models for the 2001/02 sample are not significantly different from each other based on the non-overlapping confidence interval criteria. Similarly, the mean WTP values from the MNL and S-MNL models for the 2007/08 sample are of the same magnitude. These results seem to be supportive of the findings by Greene & Hensher (2010), who reported that accounting for scale heterogeneity without preference heterogeneity in a single study appeared to have little effect on behavioural outputs such as direct elasticities and WTP.

Comparisons of MNL model estimates across the two data sets indicates similar mean WTP for all attributes, except for water clarity and lake size for the 2001/02 and 2007/08 samples. The higher mean WTP estimate for water clarity in the 2007/08 data set could possibly be attributed to the increased need for better water quality over the years since its marked decline in the early 2000s. One possible factor that could explain the higher WTP for bigger lakes in the 2007/08 sample is

[72] The coefficient for the amount of forested land in the 2001/02 data set was not significant.

ease of boat launching. It is conjectured that with the increase in the number of anglers using these lakes over time, boat launches in bigger lakes would be relatively more convenient than in smaller lakes. A number of other unknown factors could potentially explain the higher preference for bigger lakes in the 2007/08 sample.

On the contrary, the mean WTP estimates from the S-MNL models for the 2001/02 and 2007/08 samples are significantly different from each other except for the weight of fish attribute. It appears that accounting for scale heterogeneity significantly contributes to identification of differences in mean WTP across the two data sets. Hensher, Rose, & Li (2011, 2012) reported similar findings. They compared the value of travel time saving (VTTS) from seven choice experiment data sets conducted between 1999 and 2008 and found that accounting for scale heterogeneity inordinately contributes to differences in mean estimates of VTTS across studies. Assumptions about scale homogeneity seem therefore to be crucial in testing for equality of mean WTP estimates, and hence for preference stability.

6.5 Chapter summary

The main question addressed in this chapter was whether welfare estimates remain stable over time. The extent to which scale heterogeneity across individuals can contribute to differences in welfare estimates across data sets was also explored. To achieve this objective, welfare estimates obtained from the multinomial logit models (MNL) and scaled-multinomial logit models (S-MNL) for the 2001/02 and 2007/08 fishing choice data sets were compared.

In all the models anglers in the 2001/02 and 2007/08 samples generally preferred lakes that were closer to their home regions, with bigger fish, better water clarity, larger size and more recreational facilities. The findings also suggest that, in general, lakes with more urban development and greater depth were less preferred. The amount of forested land was only positive and significant in the 2007/08 sample. In terms of model performance, the S-MNL models performed slightly better than the MNL models in both data sets.

To assess whether the estimated mean WTP for the lake attributes remained stable between the two periods, results from the MNL and S-MNL models were compared, both within the same data set and across data sets. The within same data set comparison showed that the mean WTP estimates from the MNL and S-MNL models were not significantly different from each other in both the 2001/02 and 2007/08 data sets. These results seem to support findings by Greene & Hensher (2010), who reported that accounting for scale heterogeneity without preference heterogeneity in a single study appeared to have little effect on behavioural outputs such as direct elasticities and WTP.

On the other hand, comparison of estimated mean WTP from the MNL models across the 2001/02 and 2007/08 data sets showed evidence of relative stability for all attributes except for water clarity and lake size attributes. However, results from the S-MNL model do not support the stability of estimated mean WTP for any attributes, except for the weight of fish. It appears that scale heterogeneity across individuals, as accounted for in the S-MNL model, contributed significantly to differences in MWTP across the two samples. Similar findings are reported by Hensher *et al.* (2011).

To the best of my knowledge studies testing the stability of values over time in environmental economics have used models that assume scale homogeneity across respondents. Findings from this chapter have demonstrated that ignoring scale heterogeneity across the sampled population may result in the erroneous conclusion that mean WTP estimates are stable over time, when in fact they are not. This calls for a re-examination of previous empirical evidence which has not allowed for scale variability, and suggests the need to systematically account for it in future applications.

CHAPTER SEVEN

Acknowledgements

The work presented in Chapter Seven was jointly produced by my supervisors and their contributions are acknowledged as outlined hereunder. The chapter replicates what was published in the following journal.

Marsh, D., Mkwara, L., & Scarpa, R. (2011). Do Respondents' Perceptions of the Status Quo Matter in Non-Market Valuation with Choice Experiments? An Application to New Zealand Freshwater Streams. *Sustainability, 3,* 1593-1615. doi:10.3390/su3091593

Contributions made by different authors

1. **Lena Mkwara**

   - Planned the initial paper and identified and described the way in which it contributes to the non-market valuation literature.
   - Data analysis and write up.
   - Wrote the first draft of the paper and incorporated comments/suggestions from supervisors into subsequent drafts.

2. **Dr Dan Marsh**

   - Design of the choice experiment, collection of data and data preparation.
   - Supervision of data analysis.
   - Reviewing the paper.
   - Finalizing the paper and sending it for publication.

3. **Professor Riccardo Scarpa**

   - Proposed the aspects of the methodology used in this paper (e.g. use of error component model, coding to assess status quo bias)
   - Proposed comparison of the status quo bias between the status quo provided respondents and status quo perceived respondents
   - Supervision of data analysis
   - Suggested various relevant journal articles.
   - Reviewing the paper
   - Final review of the paper and assistance with redrafting after receiving comments from referees

CHAPTER SEVEN

DO RESPONDENTS' PERCEPTIONS OF THE STATUS QUO MATTER IN NON-MARKET VALUATION WITH CHOICE EXPERIMENTS? AN APPLICATION TO NEW ZEALAND FRESHWATER STREAMS

7.0 Introduction

Even "clean and green" New Zealand has its share of environmental problems. This is especially true in areas exposed to intensive agricultural production such as the Waikato region which accounts for around 30% of New Zealand's dairy production. Policy makers are torn between supporting the country's leading export industry and ensuring sustainably high environmental quality for the 400,000 people who live in the region. Water pollution from agricultural activities is considered to be one of the most important environmental issues facing New Zealand and is the most frequently mentioned environmental concern for the region's residents (Gravitas Research and Strategy Ltd, 2007). These concerns are well founded since levels of nitrogen and phosphorus in many streams, rivers and lakes have increased over the last two decades leading to a progressive decline in water quality and increased incidence of algal blooms in freshwater bodies (Ministry for the Environment, 2008).

Technical and regulatory mechanisms to reduce this non-point source pollution from agriculture are now the focus of an intensive research effort. Policy makers are showing increasing interest in non-market valuation and the use of market based tools to try and attain environmental improvement. It was in this context that a research program was started in 2008, to assess the potential trade-offs between cost, water quality improvements and job losses, using choice experiments. It is intended that the findings will inform the policy process by allowing decision makers to consider both the costs and the benefits of different levels of water quality improvement for long term sustainability of the freshwater system in the catchment.

In this paper we describe a choice experiment on a community's willingness to pay for water quality improvements in streams. We investigate the preferences of residents of the Karapiro catchment which stretches over 155,000 hectares of the Waikato region from Lake Arapuni to the Karapiro dam. Land use is predominantly for dairy (34%), pastoral (13%) and forestry (48%) production. The amount of nitrogen and phosphorus reaching waterways in the catchment has generally been increasing and is expected to continue to rise because of intensification and conversion of land from forestry to dairy. Even with widespread adoption of "best management practices" it is expected that the streams and rivers in the catchment will support more algae, water clarity will fall and the water system's ecological health may decline (Environment Waikato, 2009). Levels of *E. coli* may also increase. These changes may endanger the overall environmental sustainability of the current agricultural system.

Discrete choice experiments have gained widespread recognition since their early application by Louviere & Hensher (1982a) and Louviere & Woodworth (1983a) and their earliest application to environmental valuation by Boxall *et al.* (1996a). Choice analysis is an attribute-based technique in which respondents are presented with different alternatives defined in terms of environmental attributes and cost. They are then asked to select their preferred one. The trade-offs that they reveal during this exercise between the cost of the proposed options and their environmental attributes are used to derive implicit estimate of monetary value, under a set of well qualified assumptions.

In environmental valuation studies using choice experiments, researchers often need to provide respondents with descriptions of status quo conditions. Such descriptions are typically derived from environmental baseline studies and may differ from those perceived by respondents. Such discrepancy may lead to problem in benefit estimation because ignoring differences in utility baselines may affect the magnitude of utility changes and hence bias the implied estimates of benefits from the proposed environmental policies. We investigate this issue, taking the case of respondent perception of the quality of local streams.

In order to study the preferences of respondents with respect to departures from the current environmental conditions, the so-called status quo (SQ), analysts often place this as an alternative in all choice sets. However, recent studies have shown that description of the status quo, or its mere presence in the choice context is not neutral to the choice outcome (Adamowicz *et al.*, 1998a; Boxall *et al.*, 2009a; Brazell *et al.*, 2006; Breffle & Rowe, 2002; Dhar & Simonson, 2003; Scarpa *et al.*, 2005a). Later in this paper we review the literature on current research results involving status quo in choice experiments, but we will focus on one area of relatively poor investigation, namely that of identifying the specific effect that respondent's perception of status quo conditions has on implied welfare estimates. In particular, respondents may or may not have a clear perception of how the status quo conditions they experience relate to the attributes and levels considered in the choice exercise. In short, some respondents may not be able to map into the descriptors of environmental status used by the researcher. In this case, it is necessary for the purpose of the choice exercise to provide respondents with a description of the SQ conditions using the specific metric selected for the experimental design. So, one can distinguish two types of respondents. A first type, whose perceptions of the SQ can be mapped into the choice experiment, and a second group, to whom a mapping needs to be supplied during the course of the interview on the basis of some previous, possibly technical, knowledge. Our contribution to the literature is that of investigating whether the effects of such an asymmetry of treatment systematically results in different welfare estimates from an endogenous split sample design.

We proceed by first reviewing the different formats for the SQ alternative in choice experiments. Hess & Rose (2009) categorized the SQ alternatives into three formats as follows:

> "Firstly, […] the presence of a status quo alternative which is represented as a null alternative with the attributes and attribute levels of the alternative not shown as part of the experiment. A second form of these experiments involves respondents being shown alternatives with attribute levels based on their own experiences but not the exact levels as described. A final form of these experiments involves the inclusion of one or more alternatives in the choice task being described with exact levels representing each respondent's recent experiences." (p. 299).

An example of the use of the first format is provided in the study by Campbell *et al.* (2008) on rural environmental landscape improvements in the Republic of Ireland, in which the SQ alternative was labeled "No Action" without specifying the attribute levels. In this case it is quite obvious that the respondent is left to her own devices as to what conjecture to make about the SQ. Furthermore, the analyst does not collect any information on such conjecture. In this study we are particularly interested in the second and third formats above. The attributes described to respondents might either represent some average population measure of the good being valued and as such be described quantitatively to respondents (as in the second case above) or might be tailored to suit each individual's specific experiences (as in the third case above and Rose & Scarpa (2008). The use of the second approach is the most prevalent in the existing literature on environmental valuation, to which our study contributes. Typically, this approach involves the use of the SQ alternative described in terms of the average population measures of the prevailing environmental quality (e.g.,Kragt & Bennett, 2009; Morrison & Bennett, 2004).

Such average population measures are obtained through a consultative process involving the recording of expert assessments and public opinions, usually through focus groups. Additionally, other information obtained from a literature search may also be incorporated (Adamowicz *et al.*, 1998b). In as much as the latter approach is the most commonly used in environmental valuation the following issues are worth addressing. First, what if the predicted average levels of environmental quality deviate from the attribute levels perceived by respondents? Second, in the face of a discrepancy between the perceived attribute levels and predicted average attribute levels for the SQ alternative, how will respondents perceive the choice tasks presented to them? Third, what are the implications for the implied welfare measures of using SQ scenarios that directly account for individual specific perceived knowledge of environmental quality?

Exploratory and pioneering work on the differences between perceived and objective attribute measures was published as early as 1997 (Adamowicz *et al.*, 1997b). The first and second questions above were more recently addressed by Barton *et al*. (2009) and Kataria *et al.*(2009a). The former analysed respondents' understanding of water quality in different lakes compared to objective measures. The latter asked respondents whether they believed in the description provided for the status quo and whether they found the overall scenarios presented to them credible. They found that not accounting for respondents' beliefs in the proposed scenarios could lead to biased welfare estimates.

To date, we are aware of only one other study by Glenk (2011) in environmental valuation that has attempted to address the third question presented above. It is against this backdrop that this study endeavours to contribute to the environmental valuation literature by assessing the implications on welfare estimates of using a SQ alternative based upon each respondent's specific perceptions of water quality *vs* the use of a fixed SQ based upon average measures of water quality for the overall population.

We use choice experiment data on streams in the Karapiro catchment to investigate whether respondents' perceptions agree with our chosen description of the SQ alternative (an average measure of stream quality in the catchment), which we provided to them. Instead of simply asking respondents whether or not they believed in the described SQ scenario—as was the case in a study by Kataria *et al.* (2009a)—respondents in our study were asked to state their perceived water quality attribute levels at the SQ. Only those respondents who were unable to give their own assessment were given the average assessment of the current condition of streams in the catchment. Such treatment is labeled henceforth as *SQ provided*. Respondents who were able to assess current water quality used their own SQ in the choice experiments, or *SQ perceived*. We investigate the nature of the SQ effect emanating from the use of these two alternative formats for the SQ alternative and the implications for the implied welfare estimates.

The remainder of the paper is organized as follows. The next section briefly reviews the nature of status quo effects in choice experiments. Section 3 covers methods and the empirical model used in this study. An outline of the survey and experimental design are presented in Section 4. Results and discussions are presented in Section 5, and finally, conclusions and implications of the study are presented in Section 6.

7.1 Status quo effects in choice experiments

Initially the use of SQ alternatives in choice experiments was supported mainly on the basis of making choice tasks more realistic. It was shown that individuals making decisions tend to refer to past experiences. Therefore, relating experimentally designed alternatives to a previously experienced reference point makes stated choice tasks more realistic to respondents and informative to analysts[73] (Ortúzar & Willumsen, 2001; Starmer, 2000). This is consistent with psychological and behavioral theories, for example, prospect theory by Kahneman & Tversky (1979) and case-based decision theory by Gilboa *et al.* (2002). In later studies the inclusion of the SQ alternatives in choice experiments was justified on other grounds, including avoidance of forced choices (Adamowicz & Boxall, 2001; Dhar & Simonson, 2003); improvement in model fit, ensuring unbiased estimates (Adamowicz *et al.*, 1998a); and increase in design efficiency (Hensher *et al.*, 2007).

More recently, studies have shown that the status quo description and even its mere presence in the choice context is not neutral to the choice outcome. In particular, it

[73] For a choice modelling process to be consistent with economic theory, it is important that choice experiments are framed with a standard reference alternative so that options are evaluated against some constant base. Inclusion of some constant alternative within choice sets allows the resulting data to be combined for estimating MNL parameters.

has been found that respondents presented with both SQ and experimentally designed alternatives have a bias towards sticking with the SQ alternatives, generally referred to as the status quo bias effect, even though Scarpa *et al*. (2005a) discuss how SQ effect can be due to either a predilection for the SQ or a reluctance to stick with it, depending on the definition of the attributes of alternatives. This asymmetry in preferences between the SQ alternative and non-experienced alternatives is consistent with reference-dependent utility theories (Bateman *et al.*, 1997; Kahneman *et al.*, 1991; Kahneman & Tversky, 1979; Samuelson & Zeckhauser, 1988). The main explanations that have been put forward for this SQ effect include loss aversion (Kahneman & Tversky, 1979), cognitive misperceptions and regret avoidance (Samuelson & Zeckhauser, 1988), protesting (Adamowicz *et al.*, 2011; Meyerhoff & Liebe, 2009) and choice task complexity (Boxall *et al.*, 2009a). It has also been argued that respondents tend to avoid the cognitive burden associated with evaluating choice task alternatives that have not been experienced (Breffle & Rowe, 2002; Dhar & Simonson, 2003) and that respondents presented with unattractive alternatives are likely to choose the SQ (Brazell *et al.*, 2006).

Similarly, methodologies for accounting for the SQ effect on utility have been developed. The common approach has been to include the alternative specific constant (ASC) to capture the SQ effect on the systematic component of utility. The conditional logit model is usually applied to measure such effects. On the other hand, the SQ effect on the stochastic component of utility which represents the correlation of the error structure between alternatives, is commonly modeled through the nested logit framework; see for example (Lehtonen *et al.*, 2003; Li *et al.*, 2004).

Currently, studies have demonstrated that such specifications are limited in that they fail to simultaneously account for the SQ effect on the systematic component of utility and the variance differences in utilities between experienced SQ and conjectured utility from experimentally designed alternatives. To overcome such limitations, Scarpa *et al.* (2005a) proposed the use of error components (MXL-EC) in which the additional variance of utility of alternatives different from the SQ can be identified. Since their application, numerous other studies have found the MXL-EC to be better suited in capturing the SQ effects than the conditional logit and nested logit frameworks, and even MXL models without error components (Campbell *et al.*, 2008; Ferrini & Scarpa, 2007; Hess & Rose, 2009; Hu *et al.*, 2009; Scarpa *et al.*, 2008a; Scarpa *et al.*, 2007a). Within the MXL-EC framework, the SQ effect on the systematic component of utility can be measured by the ASC, while the effect on the stochastic component of utility can be captured by introducing a common error component shared by the utilities associated with alternatives different from the SQ, which takes account of the correlation patterns and increased error variance due to the conjectural nature of the experimentally designed alternatives.

It has already been argued that when the SQ alternative is included in the utility specification, the utility from experimentally designed alternatives tends to be more correlated amongst these, than with the SQ alternative. This correlation pattern can be attributed to the fact that the utility associated with the SQ alternative is experienced by the respondents while that of experimentally designed alternatives is not and can only be conjectured, giving rise to higher variance. Additionally, the attribute levels pertaining to the SQ alternative are fixed while those of experimentally designed alternatives are variable across choice occasions. This implies that respondents face a higher cognitive burden in evaluating experimentally designed alternatives than the SQ alternative and therefore, extra errors in addition to the usual Gumbel Type I error are expected to be made. These extra errors would induce a common correlation structure across the experimentally designed alternatives and can be captured within the MXL-EC framework through the introduction of a dummy variable (Campbell *et al.*, 2008; Ferrini & Scarpa, 2007; Scarpa *et al.*, 2005a; Scarpa *et al.*, 2007a). For this reason we adopt this modeling approach in our estimation.

7.2 Methods

We employ a mixed logit specification that combines both the random parameter and error component interpretation, following the approach detailed in Scarpa *et al.* (2005c). Train (2003) has shown how the mixed logit model can give rise to two different interpretations, the random coefficient and the error component interpretations. The random coefficient interpretation accounts for taste variations over the sampled individuals and has been widely applied in many studies (e.g.,Banzhaf *et al.*, 2001; Revelt & Train, 1998; Train, 1998). On the other hand, the error component interpretation refers to the decomposition of the error term and accounts for different correlations patterns among utilities for different alternatives (Ben-Akiva *et al.*, 2001; Brownstone & Train, 1999; Herriges & Phaneuf, 2002; Train, 2003).

In the case of this study, the choice tasks consisted of two experimentally designed alternatives and the SQ alternative. We therefore define the following utility structure:

$$\begin{cases} U(a) = \tilde{\beta} x_a + \varepsilon + \mu_a \\ U(b) = \tilde{\beta} x_b + \varepsilon + \mu_b \\ U(sq) = \tilde{\beta} x_{sq} + \mu_{sq} \end{cases} \quad (1)$$

Where $\tilde{\beta}$ denotes the random preference parameters for different water quality attributes used in this study; β_{sq} is a fixed SQ specific constant which in our case takes a value of 1 for the SQ and 0 for the other alternatives; x is a vector of attributes describing the alternatives as well as selected respondents' characteristics; μ_a, μ_b and μ_{sq} depict the unobserved component of utility and are assumed to be i.i.d. Gumbel-distributed. Instead, the error component ε is distributed $N(0,\sigma^2)$. The σ^2 adds to the Gumbel variance of μ_a and μ_b.

Assuming a balanced panel of discrete choices, with T choices made by each individual *n*, the joint probability of a sequence of *T* choices $\langle y_1, y_2, y_3, ...y_T, \rangle$ made by an individual is given by:

$$P(y_1, y_2, ..., y_T) = \int_\beta \int_\varepsilon \prod_{t=1}^{T} \frac{\exp(\tilde{\beta} x_{ti} + \varepsilon_i)}{\sum_{j=a,b,sq} \exp(\tilde{\beta} x_{tj} + \varepsilon_j)} \varphi(\varepsilon \mid \sigma^2) f(\beta \mid \theta) d\varepsilon \, d\beta \quad (2)$$

Where ε_j is equal to zero when $j=sq$

Since the integral in Equation (2) has no closed-form, it is approximated in the log-likelihood function by numerical simulation, in our case by using quasi-random Halton draws (Hensher *et al.*, 2005; Train, 1998). We first illustrate the methods for the estimation of the random utility model and then the specific tests used to evaluate the difference between simulated distributions from models with different SQ data.

7.3 Model estimation

The model in Equation (2) for the *SQ provided* and *SQ perceived* treatments was estimated in NLOGIT 4.0 by maximum simulated likelihood using 350 Halton draws (Hensher & Greene, 2003; Train, 2003). The random parameters were assumed to be independent and normally distributed, except for the cost attribute which was assumed to follow a triangular distribution constrained to have the scale parameter equal to the median. Such distribution was used for the cost parameter so as to ensure non-negative willingness to pay values (Hensher *et al.*, 2005). Attributes with parameters which were repeatedly found to show insignificant standard deviation estimates were eventually specified as non-random. The final estimates are presented in Table 7.3.

7.4 Testing differences in the implied WTP distributions

We focus on the marginal WTP for the stream water quality attributes. Rather than estimating the individual-specific WTP conditioned on the observed individual choices, we derived estimates of the population mean WTP for each of the non-monetary attributes for the model estimates based on both the *SQ described* and the *SQ perceived* samples. Population moments were simulated in R-Console using 50,000 random draws to obtain WTP distributions for each non-monetary attribute in the two sub-samples, following the approach of Thiene & Scarpa (2009a). Non-parametric procedures using the Kolmogorov-Smirnov test were used to test for equality in the WTP distributions between the two treatments. The Kolmogorov-Smirnov test statistic does not make any assumptions about the underlying distribution of the data and therefore it is appropriate for the simulated WTP distributions for which no closed form exists. The WTP distributions were found to be highly skewed. Therefore, instead of testing for the differences in the mean WTP between the two treatments, we opted for the differences in median WTP. The differences in the median WTP are graphically described using box plots as outlined by Chambers *et al.* (1983).

7.5 Survey and experimental design

The sample households for the survey were residents of the Karapiro catchment from Lake Arapuni to the Karapiro dam including contributing tributaries. Four focus groups were held to derive an understanding of people's views on water quality in the catchment and to identify attributes for inclusion in the choice experiment. These sessions were also used to test early versions of the questionnaire and to discuss the appropriate range of values for the payment variable. Best practice procedures for running the focus groups were developed drawing on Krueger (1994) and on more specific New Zealand experience from Bell & Yap (2004) and Kerr & Swaffield (2007).

Focus group discussions highlighted the increasing number of fences on farms restricting livestock access to streams and creeks, and hence livestock pollution.

This was recognized as an improvement and many participants thought that stream water quality was improving, especially when streams were protected by fenced areas of bush, which create a natural filter. Focus group participants from different areas had different perceptions of the quality of their local streams. For example, while some streams experienced by participants at the Karapiro focus group were perceived as with poor water quality, participants further upstream at the Waotu group reported high quality streams with trout, the water from which was used as a supply of domestic drinking water.

Questionnaire development and improvement took place over an extended period. Testing started using focus group participants and was followed by a pilot survey using two groups of six participants and a pre-test of 21 questionnaires. The water attributes identified by focus groups participants were supplemented by literature review and discussions with experts in the field. The attributes eventually selected for the final study were:

- Suitability for swimming (percentage)
- Ecology (percentage of excellent readings)
- Native, fish and eels (presence of)
- Trout (presence of)
- Water Clarity (Can you usually see the bottom?)

Suitability for swimming and ecological quality were defined by reference to criteria already defined by the Waikato Regional Council (WRC) whereby water is assessed as being suitable for swimming (or not) and ecological health is assessed as being excellent, satisfactory or not satisfactory. The suitability for swimming attribute aligns with the proposed national policy statement for freshwater management that is designed to ensure that appropriate freshwater resources reach or exceed a swimmable standard. This attribute is also intended as a catch all that enables respondents to state their preference for water that is safe for all forms of contact recreation (swimming, paddling, fishing, eeling *etc*.).

The ecology attribute aligns with data collected by WRC on the ecological health of waterways in the catchment. Based on 100 monitoring sites across the region, WRC reports that ecological health readings for undeveloped catchments range from 23% to 100% excellent, but for developed catchments the percentage of excellent readings is much worse, between 0 and 25%. The Karapiro catchment falls under the lower Waikato catchment zone where 68% of ecological health readings are reported to be unsatisfactory with only 2% excellent. Ecological health and presence/absence of native fish and eels vary together and so are both included in a single ecological health attribute, for example poor water quality results in only small eels being found in most catchment streams while high water quality leads to large eels, bullies and smelt being found.

The ecology of rivers and streams in the catchment has been adversely affected by clearance of forests and riverside vegetation, habitat loss and creation of barriers to fish passage (including dams). Aquatic plants and animals have also been affected by reduced water quality, changes to flow regimes, habitat loss (due to drainage and changes in land use) and introduced species that compete with or eat native fish (Environment Waikato, 2010).

Native fish populations in the Waikato Region are documented in Joy (2005). These species are highly affected by the Waikato dams which prevent fish migration. The population of eels depends on recruitment (which has been falling steadily in recent years) and the number of elvers transported over the hydro dams. Shortfin eels (Anguilla australis) are very tolerant of poor water quality and may even increase with rising levels of N and P. In poor conditions these eels would mainly be 30 to 40 cms in length. If water quality increases (and sufficient numbers are moved over the hydro dams), then the population of longfin eels (Anguilla dieffenbachia) should increase. This species is far less tolerant of poor water quality and can grow to 2 meters in length. Native bullies and smelt should be migratory but landlocked populations exist in Lake Taupo. Numbers of these species may be expected to increase with better water quality. Respondents were asked for their assessment of the condition of streams in the catchment based on the attributes and levels used for the choice cards. Respondents who indicated that they had ‘no idea’ of the quality of the streams in the catchment were presented with the status quo defined as ‘our assessment of the current overall condition of streams in the catchment’ (see Table 7.1).

During the survey, respondents who felt able to make their own assessment of stream quality in terms of the metrics used in the choice experiment scenario descriptions used their perceived quality assessment as the status quo. In this case attribute levels were entered onto a transparent overlay and placed on top of each page of choice cards to make it easy for respondents to compare their perceived status quo with the alternative levels offered in each choice card.

Attributes, attribute levels and labels used in the survey are defined in Table 7.1. Choice cards were based on an orthogonal design of 72 choice sets, with each respondent completing six choice tasks.

The initial sample for this study was drawn by intersecting the Land Information New Zealand (LINZ) property title database with the catchment boundary layer in ArcGIS. In this way a list of all 7627 properties in the catchment was produced including physical location, territorial authority and other variables. The population was broken down into three geographical strata to reflect the markedly different socioeconomic characteristics of these areas; namely Tokoroa, Putaruru/Tirau and the remaining rural areas. Address lists were drawn up for each stratum and a pseudo-random number generator was used to draw up lists of addresses to be visited by each enumerator. Field work proved to be very time consuming with each enumerator only able to complete three to six surveys each day. Field work was carried out both during the day and at weekends to try to avoid bias towards people staying at home. In the later stages of the survey a quota system was used to try and reduce bias towards people over 60.

Comparison of socioeconomic and attitudinal characteristics for our sample, with data for the Waikato Region as a whole (Table 7.2) enables some conclusions to be drawn. Men appear to be over represented at 62%. This may be due to the fact that more males than females were at home during the time of the survey or in cases where a couple was at home then the male was more likely to participate. Differences between the sub-samples are also observed particularly in levels of education and income; for example 49% of the respondents in the perceived category achieved at least a diploma or a certificate compared to only 23% in the provided group. Similarly, 65% of respondents in the perceived category earn at least $50,000 compared to 39% in the provided category. Given random sampling, the differences in representation are mainly attributed to differences in propensity to take part in the survey, for example refusal rates were higher in lower socio-economic status urban areas and lower in rural areas.

Table 7.1: Attribute levels and labels

Attribute	Current Situation	Improvement Levels		
Suitability for Swimming *(% of readings rated as satisfactory for swimming)*				
	30%	50%	70%	90%
Variables		SWIM50	SWIM70	SWIM90
Ecology *(% of readings rated as excellent)*				
	<40%	40-70%	>70%	
	Only small eels	Small eels, bullies and smelt	Large eels, bullies and smelt	
Variables		ECOM	ECOH	
Trout	*No* Trout	Trout *are* found (TROUT)		
Water Clarity	Usually you *cannot* see the bottom	Usually you *can* see the bottom (CLARITY)		
Cost to Household	*$ per year for the next 10 years* (COST)			
	$0	$50, $100, $200		

Labels	
ASC	fixed SQ specific constant which is equal to 1 for the SQ and 0 for the other alternatives
σ_ε	error component capturing the extra variance associated with the experimentally designed alternatives.
Per	denotes attributes pertaining to the *SQ – perceived models*
Pro	denotes attributes pertaining to the *SQ – provided models*

Table 7.2: Socio-demographic data for the sample and region

	Provided	Perceived	Sample	Region
Gender (%)				
Males	60	62	62	49
Females	40	38	38	51
Age (%)				
Under 30	11	16	14	18
30-44	21	20	20	30
45-59	27	29	29	28
60+	40	34	37	25
Education (%)				
Any post secondary qual.	44	49	47	
Vocational/trades	19	21	16	
Diploma or certificate (>1 year)	19	37	24	
Bachelors degree	3	8	5	
Higher degree	1	4	2	
Income (%)				
<$30,000	44	14	30	53
$30 to $50,000	18	21	19	21
$50 to $70,000	10	19	16	9
$70 to $100, 000	12	20	13	4
>$100,000	10	15	11	3
Not revealed by respondent	7	11	11	11
Work on or own a farm (%)			25	
Location (%)				
Town	63	52	57	
Settlement	19	10	13	
Rural	4	16	11	
Farm	14	22	19	
Sample Size	*73*	*103*	*178*	

7.6 Results and discussion

Respondents in the *SQ perceived* subsample generally registered higher incomes and better education levels than their counterparts in the *SQ provided* subsample. So, we proceeded by comparing the two sub-samples before and after controlling for outliers in income and qualification. In Table 7.3 we report the models for these comparisons. Models 1 and 3 include all respondents and pertain to the subsamples *SQ provided* and SQ perceived, respectively. Models 2 and 4 are based on subsamples in which respondents with income levels of over NZ$50,000 and those with any tertiary qualification in education were excluded. We excluded these to try and ensure that differences in the estimated results can be attributed to differences in the SQ treatment alone, rather than to the effect of outliers in socio-economic covariates in one of the two sub-samples.

Table 7.3: Estimation results

	Model 1 SQ-Provided All Respondents		*Model 2* SQ-Provided High Income & Qualification excluded		*Model 3* SQ-Perceived All Respondents		*Model 4* SQ-Perceived High Income & Qualification excluded	
	Coefficient	*|t-value|*	*Coefficient*	*|t-value|*	*Coefficient*	*|t-value|*	*Coefficient*	*|t-value|*
Variable								
ASC	-2.293^{f}	5.04	-2.143^{f}	3.79	0.792^{f}	2.19	0.550^{f}	1.45
SWIM50	0.344^{r}	1.34	0.504^{f}	1.74	0.601^{f}	3.18	0.792^{f}	3.04
SWIM70	1.130^{f}	4.45	1.020^{f}	3.28	0.954^{f}	4.65	1.103^{f}	3.99
SWIM90	1.641^{r}	5.07	1.510^{f}	4.25	1.281^{r}	5.17	1.765^{r}	4.71
ECOM	0.301^{f}	1.47	0.131^{r}	0.53	0.829^{f}	4.83	0.954^{f}	3.98
ECOH	0.602^{r}	2.27	0.687^{r}	2.21	1.187^{r}	5.59	1.438^{r}	4.77
TROUT	0.711^{f}	3.84	0.636^{f}	2.91	1.014^{r}	5.12	0.834^{r}	3.18
CLARITY	0.507^{f}	2.65	0.532^{f}	2.35	0.820^{r}	5.14	0.835^{f}	4.06
COST	-0.035^{r}	5.04	-0.041^{r}	6.75	-0.017^{r}	8.59	-0.023^{r}	6.04
Error Component σ_{ε}	2.692	6.91	2.487	5.93	3.341	7.22	2.181	5.86
Summary Statistics								
Log L		-513.6		-342.7		-742.2		-387.3
AIC		1.202		1.206		1.223		1.213
BIC		1.273		1.296		1.282		1.301
R^2 (McFadden)		0.466		0.469		0.453		0.466
N (Observations)		876		588		1236		660

Note: ^{f} and ^{r} denote whether the attributes were estimated as fixed or random variables.

7.7 Models from *SQ* provided sample

Models 1 and 2 refer to respondents who lacked information on the SQ conditions and were informed that the SQ is currently assessed as having poor suitability for swimming and poor ecological health. These models show estimates of utility weights with the expected signs for all attributes. The alternative specific constant (ASC) is negative and highly significant at the 1% level in both models, implying preference for a change from the status quo. In a study by Scarpa *et al.* (2005) on customer preference for water service provision, a negative ASC was attributed to dissatisfaction with the current provision of the good being valued. While this might be one of the possible explanations for the negative ASC in the *SQ provided* models, this inclination towards change might be further attributed to lack of familiarity with the SQ by this group of respondents. Since they were less familiar with the SQ, the perceived loss of leaving it might have been lower than if they were more familiar with it. This explanation is also consistent with the loss aversion hypothesis by Kahneman & Tversky (1979) and it also minimizes regret Loomes & Sugden (1982).

In terms of the preferences for water quality attributes, the results reveal that respondents have very strong preferences for water quality that is (a) highly suitable for swimming (SWIM70, SWIM90); and (b) where TROUT is found. Both models indicate lower preferences for the ecology attributes with ECOH being significant at 5% level while ECOM is not statistically significant. The COST attribute is negative and highly significant in both models, in accordance with expectations.

The error variance in both models is highly significant indicating that the inclusion of the SQ alternative had a significant effect on the stochastic component of the utility structure of the experimentally designed alternatives. The total variance associated with the unobserved component of utility pertaining to experimentally designed alternatives for Model 1 is given by $2.692^2 + \pi^2/6 \approx 8.89$; where $\pi^2/6 \approx 1.645$ is the Gumbel error variance. For Model 2, the total variance for experimentally designed utilities is equal to $2.487^2 + \pi^2/6 \approx 7.83$, which is slightly lower than that of Model 1. The total variance of indirect utilities associated with experimentally designed alternatives is much larger than what Gumbel error accommodates for both models. This is in line with the findings of the proponents of this approach (Scarpa *et al.*, 2005c; Scarpa *et al.*, 2007a).

7.8 SQ perceived models

Models 3 and 4 refer to respondents who felt able to make their own assessment of the status quo and to describe them using the required metric. On average these respondents considered the condition of streams to be better than the assessment we provided to those who 'had no idea' of these conditions. Comparison of Models 3 and 4 shows that all water quality attributes are highly significant at the 1% level demonstrating that respondents had very strong preferences for all the water quality attributes. The only difference is observed for CLARITY which is heterogeneous across respondents in Model 3 but fixed in Model 4.

The ASC is positive and significant at the 5% level in Model 3, but positive and insignificant in Model 4. The positive ASC reveals that respondents in this category are inclined to remain with the status quo. Since the SQ alternative in this model was dependent upon each individual's specific experiences the bias towards the status quo might be taken as a confirmation of the loss aversion hypothesis by Kahneman & Tversky (1979). It should also be noted that since these respondents provided their own status quo, this will in some cases have been perceived to be better than the alternative options provided. However, other explanations cannot be ruled out, such as avoidance of cognitive burden associated with the evaluation of the experimentally designed alternatives as championed by Samuelson & Zeckhauser (1988) and others.

The total variance associated with the unobserved component of utility pertaining to experimentally designed alternatives in Model 3 is approximately equal to $3.341^2+\pi^2/6 \approx 12.81$, which is almost twice as high as the variance in Model 4 given by $2.181^2+\pi^2/6 \approx 6.40$. These results demonstrate that the inclusion of the SQ alternative had a significant effect on the stochastic component of the utility structure of the experimentally designed alternatives, consistent with findings from the *SQ provided* models. In addition, these results demonstrate that respondents with higher income and qualification levels in the *SQ perceived* treatment seem to have had relatively high valuation errors as indicated by the higher variance in Model 3 compared to that in Model 4, where such respondents were removed.

Comparison is made between the respondent's willingness to pay (WTP) for water quality improvements in the two treatments. The simulated population mean and median WTP values for the different attributes are presented in Table 7.4 below, as derived from the estimated random parameter models.

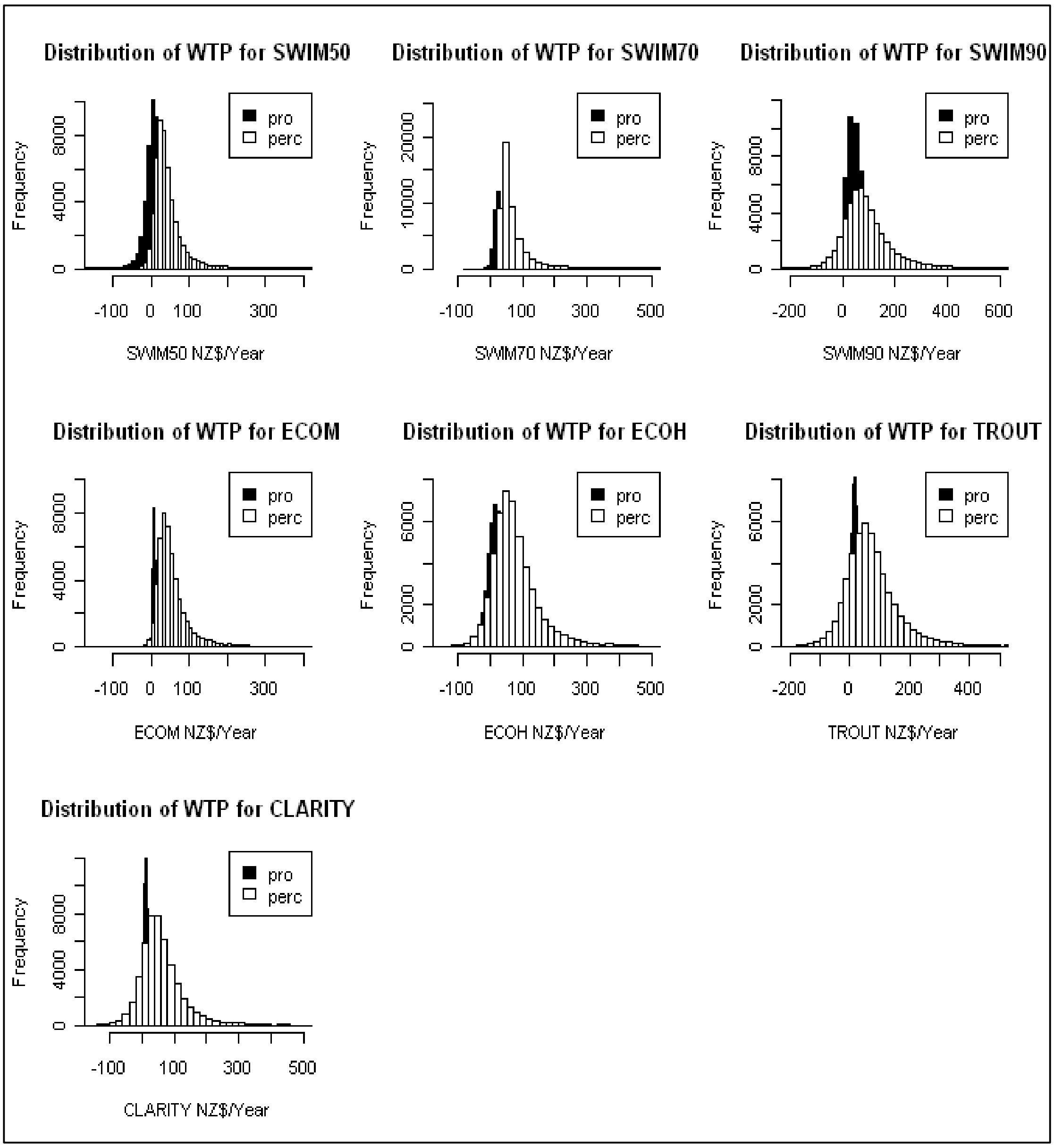
Table 7.4: Mean and median marginal WTP estimates in NZ$/Year.

	Model 1		*Model 3*		*d-stat*	*Model 2*		*Model 4*		*d-stat*
	SQ-Provided All Respondents		SQ-Perceived All Respondents			SQ-Provided High Income & Qualification Excluded		SQ-Perceived		
Attribute	*Mean*	*Median*	*Mean*	*Median*		*Mean*	*Median*	*Mean*	*Median*	
SWIM50	13.4	9.56	48.4	34.82	0.455	17.63	12.64	48.28	34.7	0.524
SWIM70	42.59	30.72	77.65	55.86	0.505	32.01	22.99	67.21	48.34	0.447
SWIM90	67.19	48.05	109.05	78.67	0.249	51.97	37.24	92.89	66.765	0.281
ECOM	11.74	8.47	64.41	46.33	0.780	4.92	3.52	63.98	46.15	0.941
ECOH	30.29	21.71	91.01	65.61	0.408	23.83	17.07	83.85	60.28	0.529
TROUT	27.69	19.95	85.46	61.79	0.475	19.91	14.26	51.39	36.93	0.398
CLARITY	19.75	14.15	69.3	49.99	0.526	16.52	11.84	45.99	33.16	0.745

All *d*-statistics have significance at *p*-value < 0.001

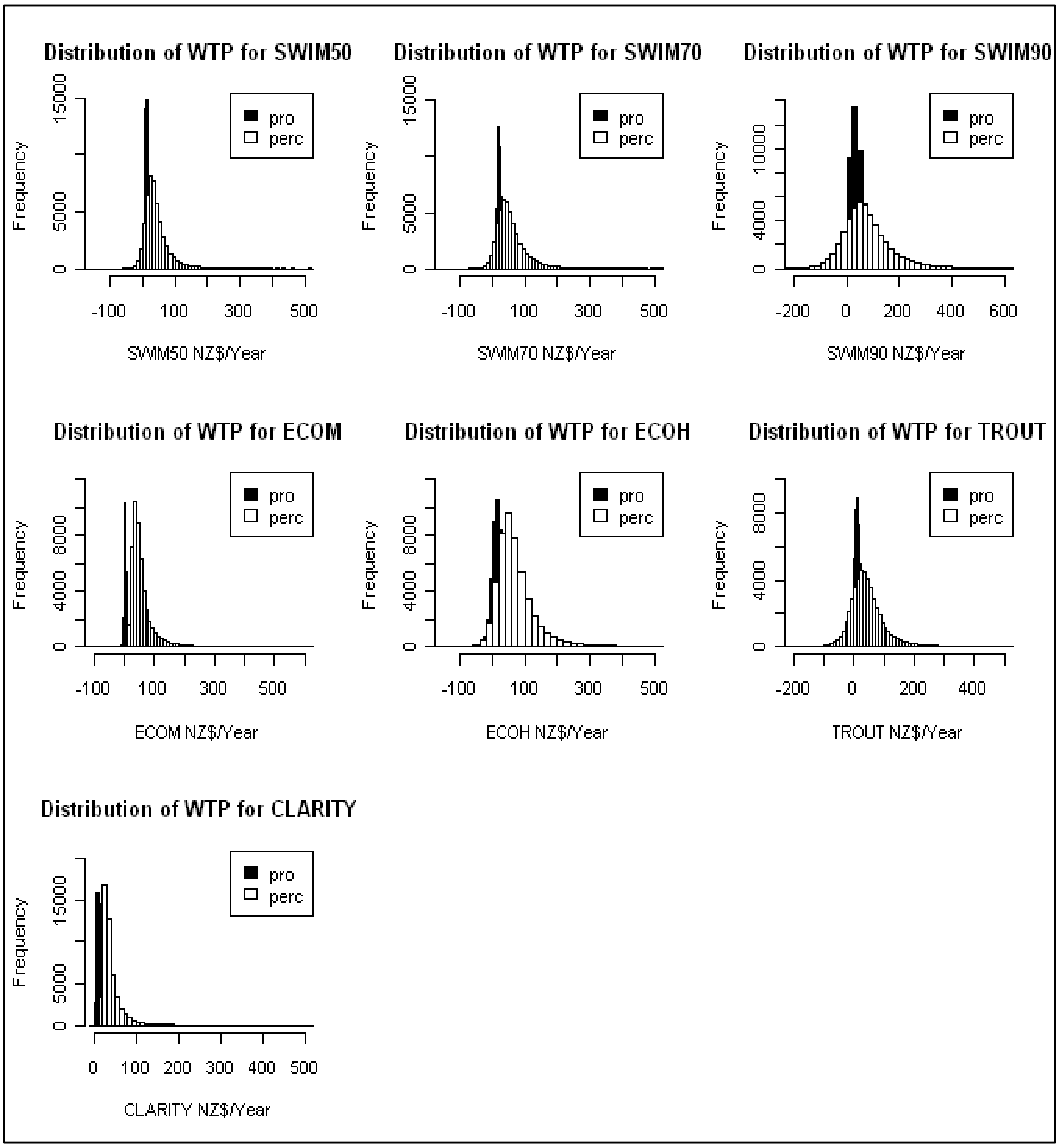
Comparing the mean and median WTP in Models 1 and 3 there is a clear indication that respondents in the *SQ perceived* model are willing to pay more for water quality improvements than those in the *SQ provided* model for all attributes. A similar trend is observed in Models 2 and 4 in which respondents with high income and qualification levels were excluded from the analysis. The median WTP values are less than the mean WTP values in both treatments for all attributes, indicating that the distributions are highly skewed upwards. In general the differences in WTP values between the two treatments appear to be quite substantial. A graphical comparison of the distributions of WTP values across the two SQ treatments based on models estimated on all respondents (Model 1 and 3) is presented in Figure 7.1.

Figure 7.1: Histogram showing distribution of marginal WTP for models 1 and 3



The distributions are highly skewed with long and fat tails towards the upper end of the scale. Further, analysis of the histograms highlights that although the distributions of the WTP for all attributes overlap, the WTP for most respondents in the *SQ provided* model is relatively lower than their counterpart. The Kolmogorov-Smirnov test (*d*-statistic) in Table 7.4 reveals that there are significant differences in WTP distributions for all attributes in the two treatments. Likewise, the simulated distributions of WTP for Model 2 and 4 are compared and presented in Figure 7.2 below:

Figure 7.2: Histograms showing distribution of marginal WTP for models 2 and 4



Once more, the distributions are highly skewed with relatively fat tails towards the upper end of the scale, with the simulated population distribution of WTP from the *SQ provided* model being relatively lower than that from the *SQ perceived* model. The Kolmogorov-Smirnov test (*d*-statistic) again reveals that there are significant differences in the distributions of WTP values from the two subsamples (Table 7.4).

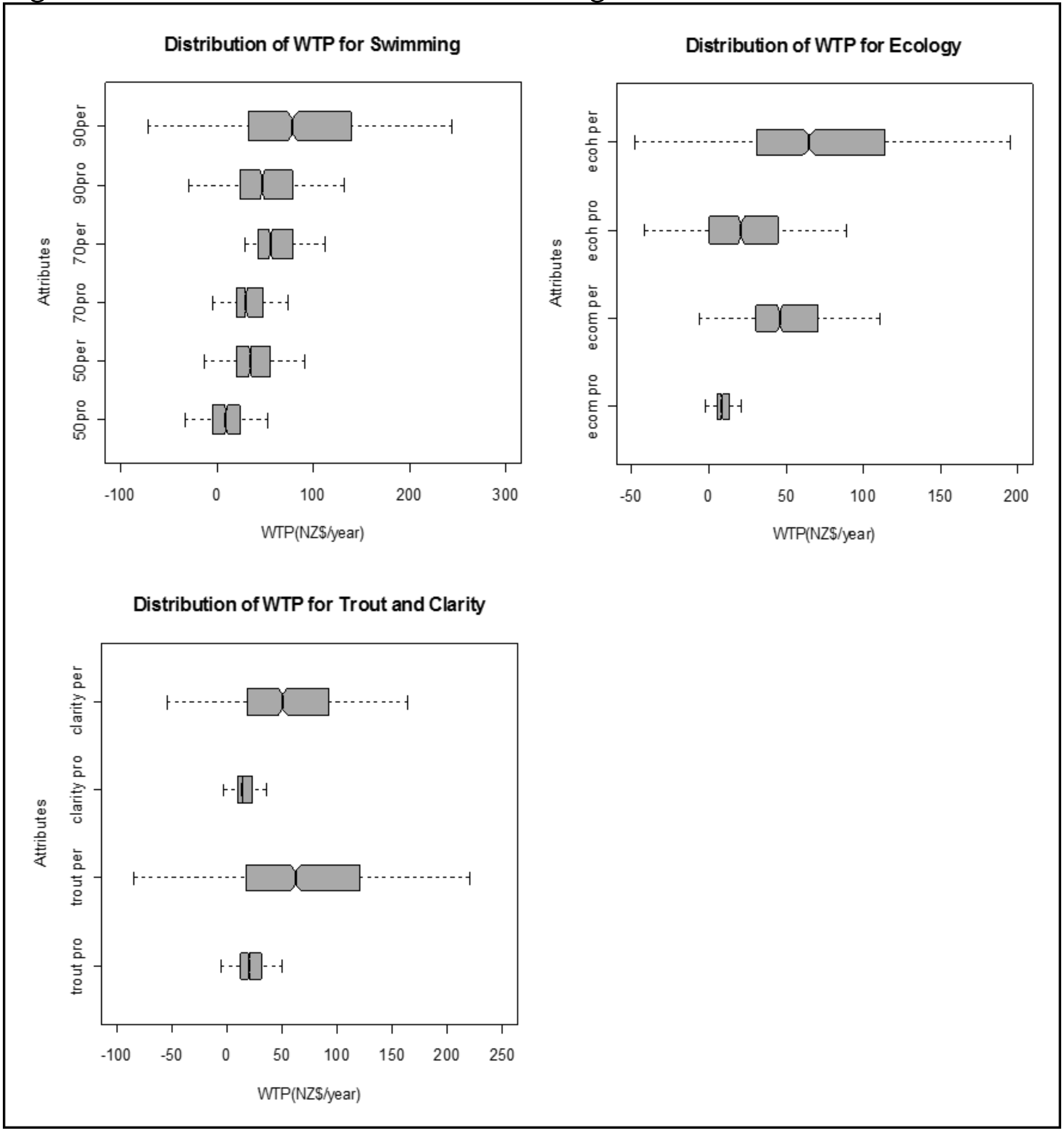
Our results suggest that the distributions of WTP values between the two treatments are significantly different. Poe *et al.* (1994) states that:

"Differences in estimated WTP distributions do not necessarily imply that the means derived from these distributions are different. For instance, it is possible that two significantly different distributions can cross and have identical means."

To graphically explore the differences in the simulated measures of central tendency between the two treatments, the quartiles of the distributions of WTP are compared using box plots see Tukey (1977) and reported in Figures 7.3 and 7.4. The box plots display the upper and the lower limits of the cumulative distributions, and the inter-quartile range showing the first quartile, the median and the third quartile. Given that the distributions of WTP are highly skewed, the median is used as a basis of comparison as opposed to the mean, since the latter can be influenced by extreme values.

Figure 7.3 shows the box plots for Models 1 and 3 with all respondents included in the analysis.

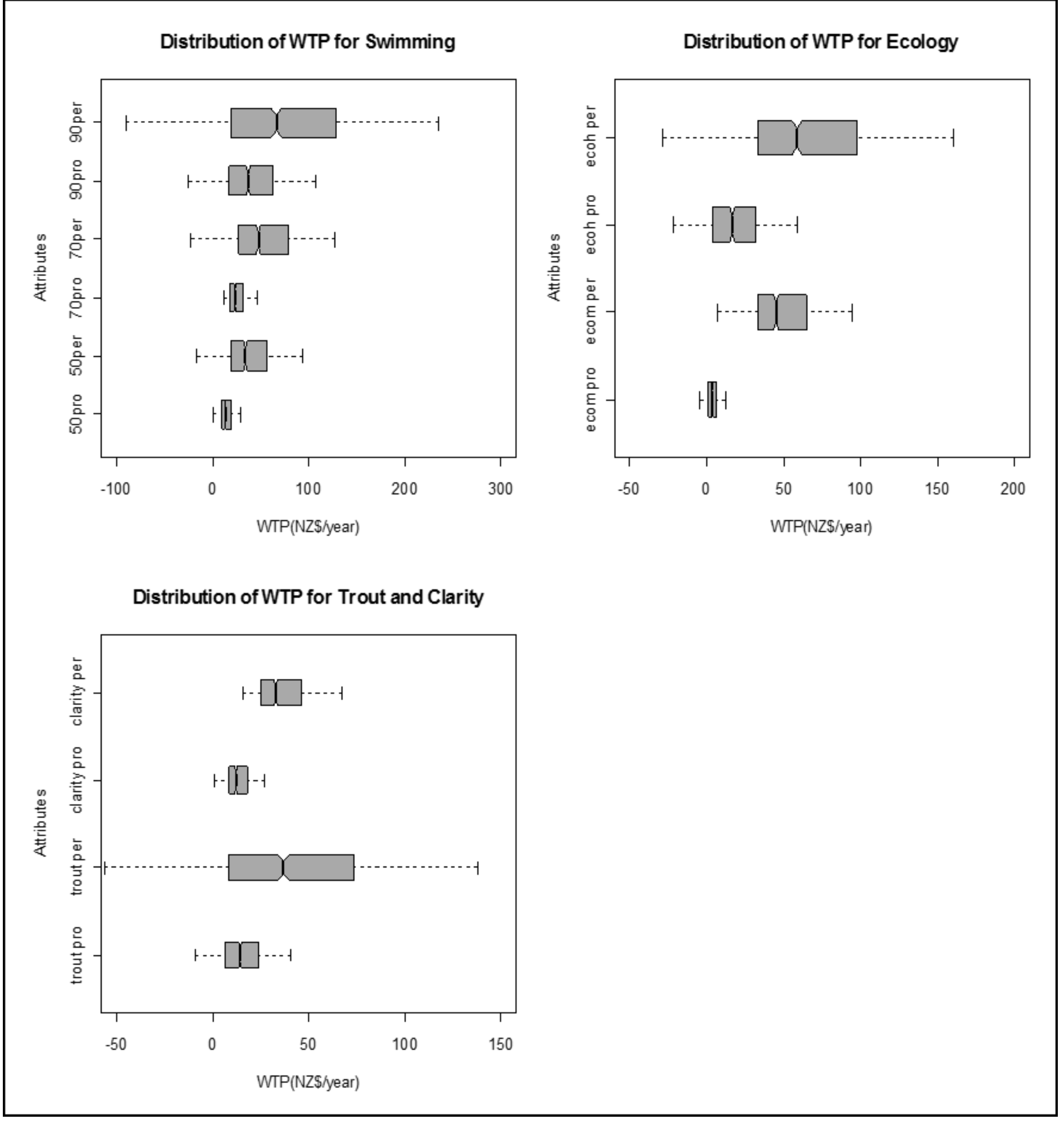
Figure 7.3: Box Plots for distributions of marginal WTP for Models 1 and 3



The quartile distributions are consistent with the previous results, with respondents in the *SQ perceived* model generally showing higher WTP for all attributes than those in the *SQ provided* model. Specifically, the notches in the box plots signify the 95% confidence interval for the median. According to Chambers *et al.* (1983), if the notches do not overlap, the null hypothesis of equal medians is rejected.

A similar comparison between the median WTP values for Models 2 and 4 in which respondents with high income and qualification levels were excluded from the analysis is presented in Figure 7.4 below:

Figure 7.4: Box plots for distributions of marginal WTP for Models 2 and 4



Inspection of the box plots demonstrates that the notches do not overlap for all stream water quality attributes and therefore, the hypothesis of equal medians is rejected. This test is a further confirmation that respondents in the *SQ perceived* models display stronger preferences, as implied by higher WTP values, than those in the *SQ provided* models. The results further highlight that there is more variance in the WTP values in the *SQ perceived* models, especially for SWIM90 (90 % of readings satisfactory for swimming), ECOH (excellent ecological health) and presence of trout, than in the *SQ provided* models.

7.9 Chapter summary

The broader purpose of this research was to assess a community's preferences for stream water quality improvements. A specific focus in this paper was placed on the effect of accounting for perceived versus described status quo levels. The study revealed that about 58% of respondents had their own perceived baseline condition of water quality and that they could map it into the framework of attributes and levels proposed in the survey. On the other hand 41% of respondents were provided a SQ description by researchers because these respondents either had little or no prior knowledge of the prevailing conditions of water quality in streams or they had this knowledge but could not map it into the proposed framework. We believe that such a dichotomy is common in many nonmarket valuation studies, and hence its consequences for policy prescription via value estimation are worth exploring.

The results of our investigation show marked differences in the marginal value that these two groups of respondents place on water quality improvements and this has implications for their willingness to pay values. The respondents who were provided with status quo descriptions expressed strong preference for water that is suitable for swimming, has good clarity and where trout can be found. Yet, this group displayed a reluctance to stay with the SQ scenario. We argued that this might be the case because of their comparative ignorance of baseline water quality conditions. The second group of respondents, who adopted their own perceived SQ scenario, expressed significantly *stronger* preference for improvements across all the attributes subject of this study, but this tendency was attenuated by a general reluctance to embrace policy options implying changes from the SQ, about which they had quite good knowledge. For this group estimates of marginal willingness to pay values are higher across the entire distribution than for respondents to whom the SQ information was *provided*.

Economic theory suggests that marginal WTP should be proportional to the expected improvement and this in turn depends on individual perceptions in one group and the provided description in the other. In our individual perception data we observe that on average perceived quality of the SQ conditions was higher than the one that was provided. This might be the cause for the observed reluctance to abandon the SQ, as manifested by a positive and significant alternative specific constant for the SQ alternative. In principle for this group the expected improvement would be perceived as smaller, and so would the associated marginal WTP when compared to that held by the *SQ provided* group. However, this holds only for quality changes within evaluations by the same respondent. Unfortunately this cannot be tested here because of the lack of a counterfactual.

The present study demonstrates the effects of using a coding specification of the status quo directly built on respondents' perceptions. Our results are supportive of the findings by Kataria *et al.* (2009a) which showed that failure to take account of respondents' beliefs leads to biased welfare estimates and earlier similar findings by Adamowicz *et al.* (1998b).

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CHAPTER EIGHT

CONCLUSIONS

The overall objective of this thesis was to assess peoples' preferences and willingness to pay for water quality improvements. Two methods of investigation were used, namely (a) a travel cost random utility model of trout angling in the Rotorua Lakes and (b) a choice experiment of resident's preferences and willingness to pay for stream water quality improvements in the Karapiro catchment. Specifically, seven main questions were addressed with the main conclusions being summarized below.

(Q1). Does water quality influence anglers' choice of lake for fishing? If so, what value do they place on water quality improvements?

In Chapter Four the travel cost RUM was introduced to New Zealand non-market valuation literature. It was applied to explore how changes in water quality would impact upon anglers' choices of fishing destinations in the Rotorua Lakes. A sample of 414 anglers drawn from the 2007/08 National Angling Survey conducted jointly by NIWA and FGNZ was used (Unwin, 2009). It was shown that trout anglers fishing in the Rotorua Lakes generally preferred lakes with better water quality, that were larger in size, with bigger fish, more facilities and more forest cover. Lakes which were greater in depth, with more urban development around the catchment and which had past health warnings due to algal blooms were generally less preferred.

It was further revealed that for a 1 metre increase in water clarity in each of the lakes with poor or average water quality, the welfare gains would be $58 for Lake Rotoiti; $38 for Lake Rotorua; $6 for Lake Rerewhakaaitu; $4 for Lake Rotoehu; $2 for Lake Okareka and less than $1 for Lakes Okaro and Rotomahana per angler per year. The aggregate annual benefit would be $1.3 million for Lake Rotoiti and $0.8 million for Lake Rotorua for a 1 metre increase in water clarity. The aggregated annual benefits for all of these lakes would be $2.3 million[74]. These estimates should assist with funding decisions in a context where territorial authorities and central government must decide the sums of money which are to be spent on water quality improvements over the coming years.

The travel cost RUM was further used to assess the overall benefit that trout anglers obtain from each lake. The annual level of these benefits totals $21.7 million (Rotoiti $7.7m; Tarawera $5.1m; Rotorua $5.0m; Rotoma $1.7m; Okataina $0.8m; Rerewhakaaitu $0.6 m; Rotoehu $0.4m; Okareka $0.2m; Tikitapu $0.1m; Rotomahana $0.07m and Okaro $0.001m).

As highlighted in Chapter Four (Section 4.6) any use of these estimates for policy decision should be made in full awareness of potential biases from the single traveller, regional median income and day trip assumptions used in this thesis. If the travelling party for the sample of anglers used in this thesis is greater than one, then resource values estimated here are likely to be inflated. Using regional median income as a proxy for anglers' income is likely to underestimate the cost of lake access and WTP estimates are likely to be under-valued[75]. On the other hand, if overnight trips were undertaken by some anglers, then resource values estimated are likely to be undervalued.

These findings have illustrated the importance of the travel cost RUM in providing information that can be useful for policy decisions involving recreational-based natural resources in New Zealand. The travel cost RUM uses real data based on observable individual behaviour and therefore closely mimics the measurement of

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[74] These findings were based on the population of 21,883 anglers who bought fishing licences during the 2007/08 fishing season and whose home origins were within the Eastern Region and Auckland/Waikato Fish and Game Councils, which was the study population of interest in this application.

[75] Freshwater sport fishing tend to be dominated by wealthier sectors of society due to high expenditures involved (Cowx, 2002).

economic values based on market prices. The use of this technique can complement environmental decision making by enabling the impact of alternative policies to be assessed while accounting for substitution effects across sites. Recreational sites of policy interest or those that are of most value to society can be identified and prioritised.

The 2007/08 Rotorua Lakes fishing choice data were further used to address three related research questions highlighted below.

(Q2) Does accounting for within-season variability in recreational site attributes that are variable across the season matter? (Q3) Can the use of less aggregated data reduce multicollinearity in revealed preference data? (Q4) Can levels of collinearity typically considered tolerable have an effect on welfare estimates?

Accounting for within-season variability in site attributes in valuation studies using revealed preference data is uncommon, although seasonal variability in some attributes is acknowledged (Andrews, 1988; Clark, 1980; Provencher & Bishop, 2004; Swallow, 1994). This can partly be attributed to insufficient variation in natural conditions that characterize most datasets of recreational site attributes. Alternatively, researchers might consider such variability to be too small to have any substantial effects on recreational site choice decisions and implied welfare estimates. This issue was explored in Chapter Five and was considered relevant to the Rotorua Lakes because water quality and fish growth tend to vary across the year and between lakes. The 2007/08 National Angling Survey from which the study sample was drawn accounted for seasonality in angler demand across the year. In addition to region and licence type, the survey was stratified by time, with the 12 month survey period divided into six two-monthly intervals (Unwin, 2009). The bimonthly averages of water clarity and weight of fish were computed to correspond to the two monthly partitions in the angling data and also to account for inter-temporal variability in these attributes across the year. This ensured that anglers' preferences were estimated using the water quality and weight of fish attribute levels relating to the period of each fishing trip. Specifically, the effect on welfare estimates of using bimonthly versus annual averages of water clarity and weight of fish was investigated.

It was conjectured that differences in welfare estimates could result from differences in attribute levels and collinearity levels or the combined effect of both. This was the first study to investigate whether collinearity typically tolerated in revealed preference data could have a significant effect on welfare estimates. Differences in collinearity levels between models using bimonthly and annual averages of water clarity and weight of fish were assessed by comparing the D-errors derived from the variance covariance matrix of the estimated parameters from the conditional logit models. Models that used annual weight of fish averages had D-errors 16 to 20 times higher than models based on bimonthly averages. The use of the bimonthly average weight of fish reduces the size of the standard error by up to 73%.

These findings demonstrated that use of less aggregated data for the weight of fish attribute led to a substantial reduction in collinearity levels. The MWTP for fish weight attribute was five times higher using annual, rather than bimonthly, data. This difference was attributed to the combined effect of the differences in attribute and collinearity levels. It was difficult to isolate the two effects since the two were confounded.

On the other hand, the collinearity levels for annual and bimonthly water clarity data were of similar magnitude when the bimonthly weight of fish was used in both models. In this case the MWTP estimates for water clarity, using bimonthly versus annual data, were not statistically different. However, the MWTP based on annual data was found to be significantly higher than the bimonthly estimate when the annual average weight of fish was used in the model. This result was largely induced by collinearity with the annual weight of fish attribute.

The overall findings from Chapter Five demonstrate that accounting for within-season variability in recreational site attributes can have a significant effect on welfare estimates. These findings are very pertinent to travel cost studies where collinearity among regressors is ubiquitous. Use of less aggregated data to better account for within-season variability has the potential to enable major gains from reduced collinearity, although, in the absence of a counterfactual, these effects remain latent and unexplored and may be data specific. These results further illustrate that even the relatively low to moderate levels of collinearity typically tolerated in revealed preference studies can have an effect on welfare estimates.

In Chapter Six the analysis was extended further to address research questions (5) and (6).

(Q5) Do WTP estimates remain constant over time? (Q6) Can scale heterogeneity across individuals significantly contribute to differences in WTP across data sets?

To address these questions, welfare estimates obtained from the MNL and S-MNL models using the 2001/02 and 2007/08 fishing choice data sets were compared[76]. The estimated MWTP from MNL models demonstrated evidence of relative stability for all attributes except for water clarity and lake size attributes. In comparison, results from the S-MNL model did not support the stability of estimated MWTP for all attributes except for the weight of fish attribute. Scale heterogeneity across individuals in the S-MNL model seemed to have contributed significantly to differences in MWTP across the two samples. To the best of the author’s knowledge, all studies testing the stability of values over time in environmental economics have used models that assume scale homogeneity across respondents. Therefore, this analysis was one of the first environmental non-market valuation studies to demonstrate that scale heterogeneity across individuals can lead to significant differences in MWTP across studies.

[76] The 2001/02 data set consisted of a sample of 524 anglers drawn from the 2001/02 National Angling Survey by NIWA and FGNZ.

Findings from this chapter demonstrate that ignoring scale heterogeneity across the sampled population may result in researchers erroneously concluding that MWTP estimates are stable over time. Similar findings have been reported in the field of transportation by Hensher *et al.* (2011). This calls for a re-examination of previous empirical evidence that did not allow for scale variability, and suggests the need to systematically account for it in future applications.

Finally, in Chapter Seven, a choice experiment was conducted to assess the benefits of cleaner streams for Karapiro catchment residents. The main objective was to provide answers to the research question stated below.

(Q7) Do respondents' perceptions of the status quo matter in non-market valuation with choice experiments?

In choice experiments researchers often provide descriptions of status quo conditions which may differ from those perceived by respondents. An investigation was carried out to assess whether ignoring this difference in utility baselines can affect the magnitude of estimated utility changes and hence benefit estimates of proposed environmental policies. This was achieved by comparing WTP between respondents using their own perceived quality of streams and those provided with descriptions of the status quo conditions.

The study revealed that 58% of respondents had their own perceived baseline condition of water quality and that they could map it into the framework of attributes and levels proposed in the survey. On the other hand, 41% of respondents were given a baseline description because they had little or no prior knowledge of the prevailing water quality in streams or they had this knowledge but could not map it into the proposed framework.

The results indicated marked differences in the marginal value that these two groups of respondents placed on water quality improvements and this had implications for their WTP values. The respondents who were provided with status quo descriptions expressed strong preference for water that was suitable for swimming, had good clarity and where trout could be found, yet this group displayed a reluctance to stay with the SQ scenario. This may be because of their comparative ignorance of baseline water quality conditions. The second group of respondents, who adopted their own perceived SQ scenario, expressed significantly *stronger* preference for improvements across all the attributes subject of this study but this tendency was attenuated by a general reluctance to embrace policy options implying changes from the SQ, about which they had quite good knowledge. For this group, estimates of marginal willingness to pay values were higher across the entire distribution than for the respondents to whom the SQ information was *provided*. The individual perception data indicated that, on average, perceived quality of the SQ conditions was higher than the one that was provided. This might be the cause for the observed reluctance to abandon the SQ.

This study demonstrates the effects of using a coding specification of the status quo directly built on respondents' perceptions. The results are supportive of the findings of Kataria *et al.* (2009b) which showed that failure to take account of respondents' beliefs leads to biased welfare estimates, and of earlier similar findings by Adamowicz *et al*. (1997a) in the context of integrating revealed preference data, in which the status quo was based on respondents' subjective perceptions, and stated preferences, where it was objectively described to them. More recently, findings by Glenk (2011) also showed that failure to account for asymmetric preference formation can result in biased estimates of WTP.

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Appendix 1: Land use in the Rotorua Lakes' catchment area

Figure A1.1: Land use in the eutrophic and supertrophic lakes' catchments

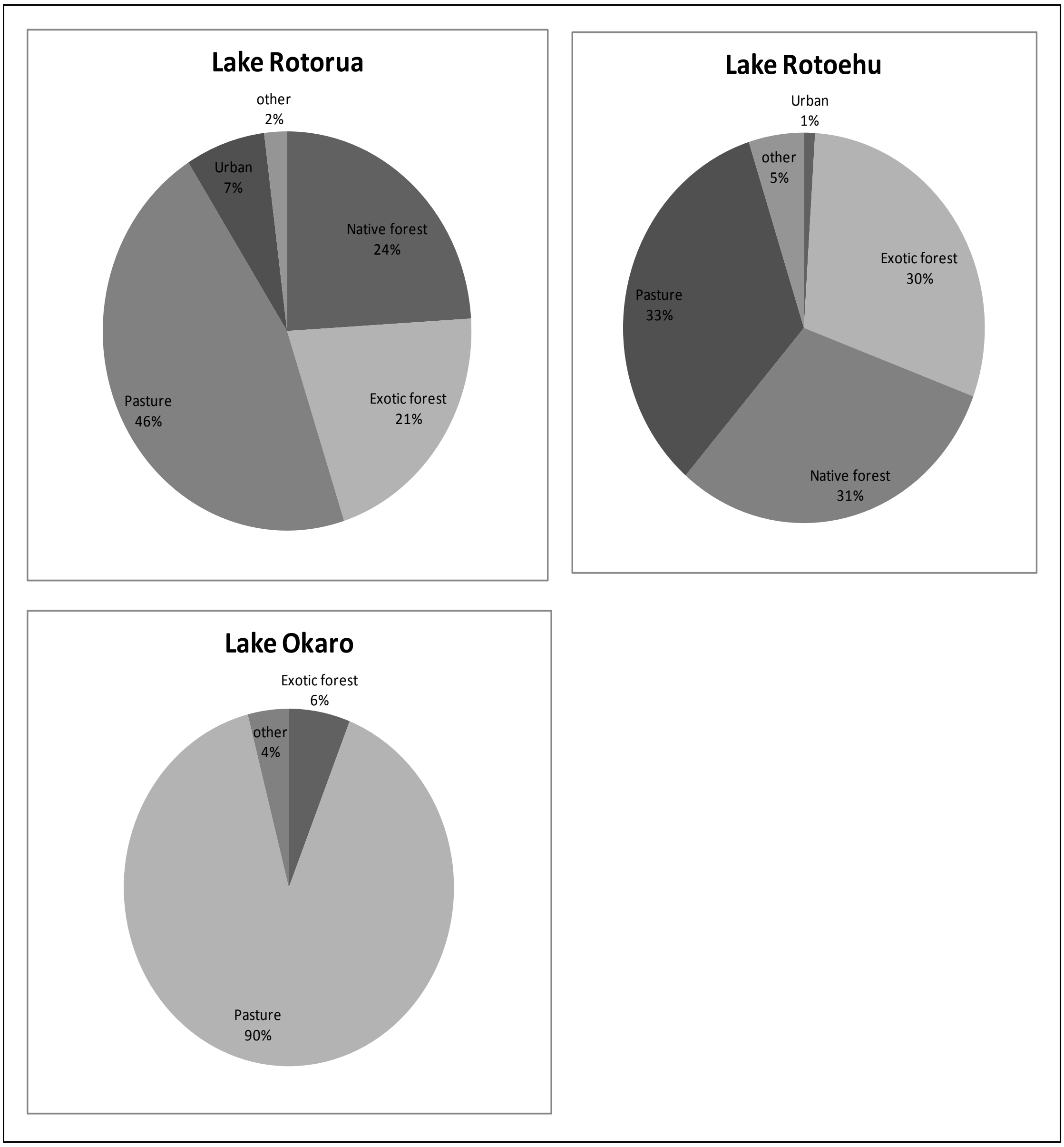


Figure A1.2: Land use in the mesotrophic lakes' catchments

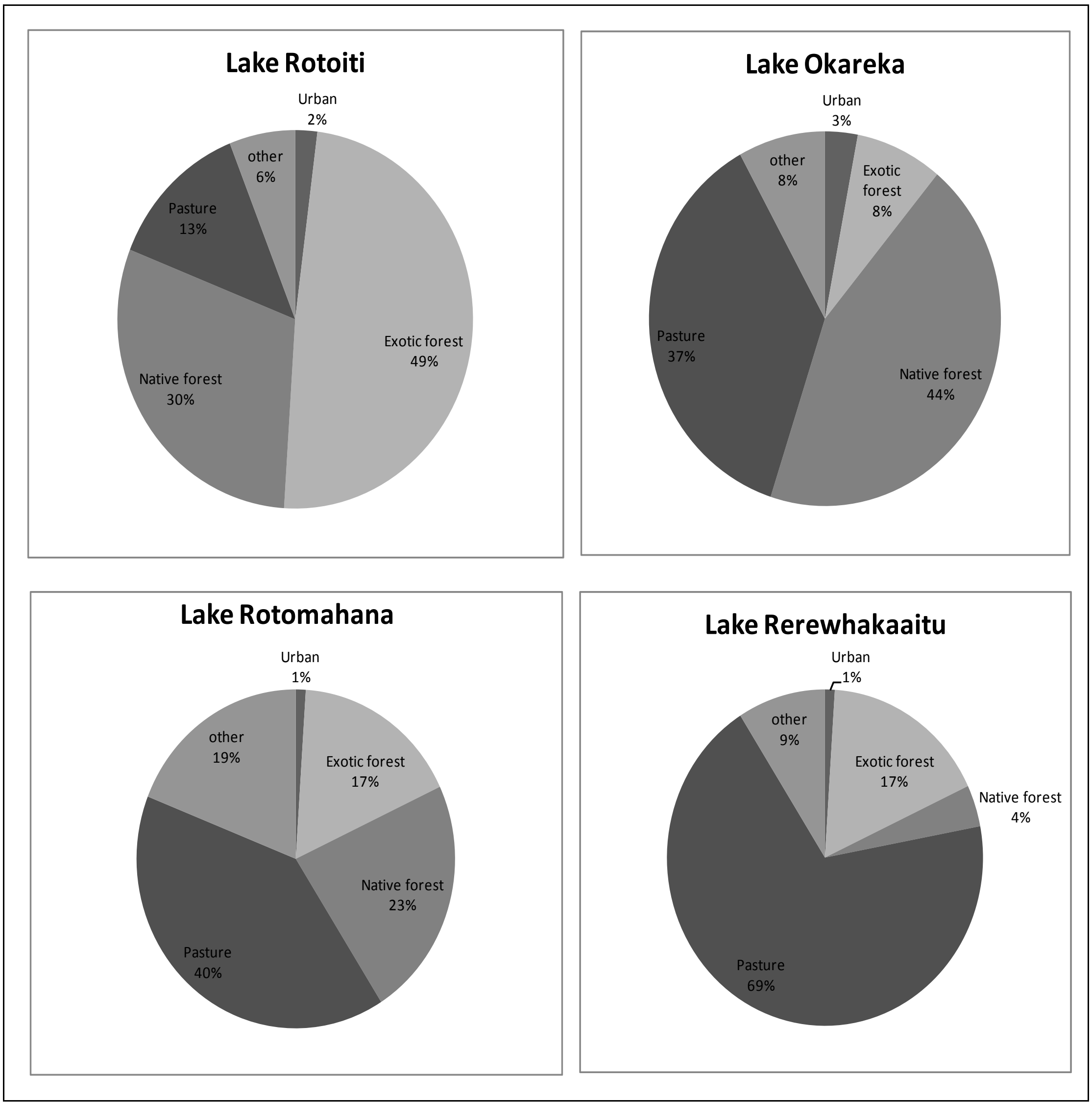
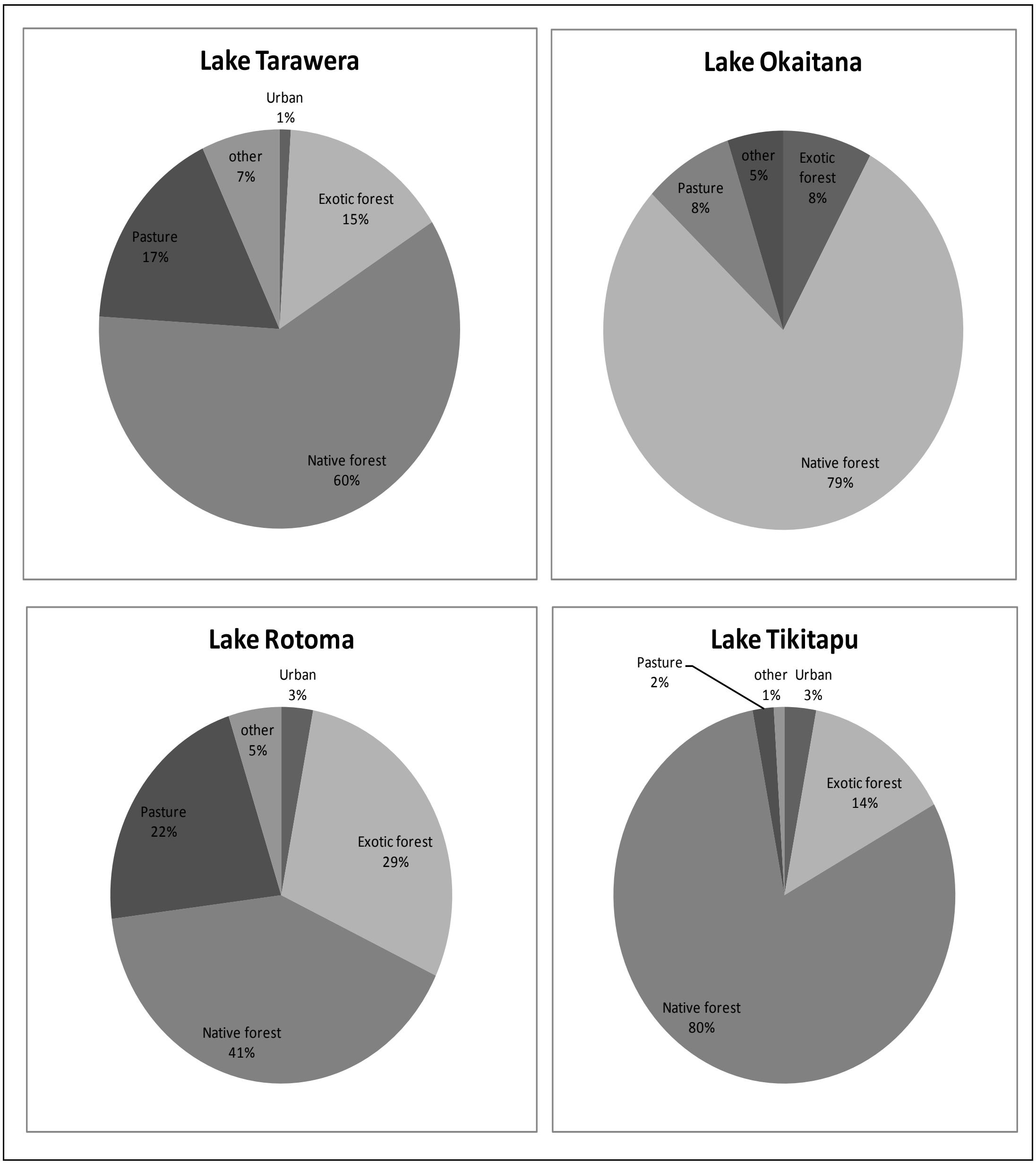


Figure A1. 3: Land use in the oligotrophic lakes' catchments



Appendix 2: Freshwater non-market values and ecosystem services

Table A2.1: Freshwater non-market values and ecosystem services

Final goods of freshwater habitat	Lakes & Rivers	Wetlands	Examples and relationships
Provisioning			
Food	☒	☒	Wetlands provide grasses for grazing, silage and hay. Lakes and rivers are a significant source of commercial fisheries.
Biomass: fibre and energy materials, including peat	☐	☒	Wetlands produce reeds and osiers under saturated conditions. Peatlands provide energy and soil improvement products.
Water for use	☒	☐	Freshwater bodies are a source of water supply for household use, agricultural and other industrial processes.
Navigation services	☒	☐	Lakes and rivers with sufficient depth provide waterways for navigation.
Health products	☒	☒	Mineral spas, medicinal plants, medical leeches.
Regulating			
Carbon regulation	☐	☒	Wetlands are vital for carbon storage in organic soils, thereby helping in maintaining a balanced chemical composition in the atmosphere
Water flow and flood regulation	☒	☒	River flow is influenced by landscape location, connection with other water bodies and discharge excessive water flows. Flood reduction relies on available water storage; Wetlands temporarily store excessive water flows, which moderate flood impacts on downstream environments.
Water quality regulation	☒	☒	Freshwater systems can dilute, store and detoxify waste products and pollutants. Wetlands perform a vital function of water purification by removing nitrogen and phosphorus from agricultural runoff, preventing eutrophication of rivers and lakes.
Human health regulation	☒	☒	Freshwater ecosystems with good water quality and aesthetic appeal can enhance the well-being of individuals through physical recreation. Poor water quality can be a source of water borne diseases.

Table A2.1: Freshwater non-market values and ecosystem services (continued)

Final goods of freshwater habitat	Lakes & Rivers	Wetlands	Examples and relationships
Biodiversity	☒	☒	All freshwater habitats with open water; species depend on conditions such as temperature, oxygen level, depth and velocity of water and area with suitable conditions.
Nutrient recycling	☒	☒	Recycling of soil and water natural and artificial nutrients occurs in wetlands, supporting enhanced water quality.
Cultural services			
Science and education	☒	☒	Lakes and wetlands sequences contain archives and human (pre)history and artefacts that may be lost if disturbed. Freshwater ecosystems are important outdoor laboratories.
Recreation and tourism	☒	☒	Recreational fisheries and tourism depend on landscape appeal and iconic species. Good water quality and visual appearance required for natural swimming and boating.
Cultural and historic information	☒	☒	Water is important in defining specific landscape character and features strongly in art and local culture. Freshwaters are a recurrent feature at the heart of many historically important places.
Spiritual and historic	☒	☒	Freshwater bodies and their features can be places of significant spiritual value.

Adapted from Morris & Camino (2011)

☒ means the goods and services are provided by the specific freshwater body

☐ means the goods and services are not provided

Appendix 3: Other revealed preference approaches

Cost based valuation methods

Other less prominent techniques that can be used to value water quality improvements include the damage cost method, cost of illness approach, production function approach, demand functions, and the avertive expenditures method.

Damage assessment costs method or replacement cost method

This approach assumes that damages caused to environmental resources including water can be valued through various costs imposed upon different parties in society. Integrating all such costs would represent the cost of replacing the damaged environmental asset or providing compensation to the affected parties. An example of such studies was carried out by (Morey *et al.*, 2002) to assess the value of damages caused by mining wastes on anglers who fished in cold-water trout rivers and streams in Southwestern Montana in the United States of America.

The production function approach

The production function is usually applied in the valuation of goods that do not have a market value if such goods can be used as inputs, together with some other marketed inputs, in the production process of marketed output goods. Since the amount of a marketed output produced is assumed to be dependent upon the factors of production, the demand for a non-marketed good can be derived from the demand for the marketed output produced (Birol *et al.*, 2006; Young, 2005). This approach is commonly used in the valuation of water for agricultural irrigation purposes and can be applied in other production processes as well. For instance, (Bell, 1997) used a recreational fisheries production function to derive the value of wetlands in supporting recreational fishing in the South-eastern United States.

Avertive or defensive expenditure method

This approach generally refers to the precautionary measures that individuals may take to protect themselves from harmful effects of environmental damages such as water pollution. For example, in response to possible harmful effects from contaminated household drinking water, individuals may opt for defensive behaviours such as buying bottled water or boiling water. The value of good water quality can be inferred from the amount of money spent on these defensive behaviours. This approach can also apply to other uses of water, for instance in reaction to water contaminants in a lake, some individuals may opt to construct their own swimming pools. The amount of money spent on such activities can be taken as the value that such individuals place on good water quality in lakes or rivers. However, as noted by (Birol *et al.*, 2006), this approach tends to be limited by the fact that averting behaviours may present multiple benefits to individuals. For instance, in addition to the health benefits derived from using privately owned swimming pools, people may also attain some social status benefits which may be unaccounted for during the valuation process.

Cost of illness approach

This technique assumes that the cost of environmental degradation, including polluted water, can be measured by proxies such as the amount of money spent on medical treatment of water-borne related infections and the amount of production lost due to such illnesses. Thus the total cost of illnesses due to polluted waters can be regarded as the value of good water quality to society. Dwight *et al.* (2005) used this technique to measure the health burden from illnesses due to exposure to polluted recreational marine waters in Orange County, in the U.S.A.

Appendix 4: Benefit transfer

Benefit transfer is the term used to describe the transfer of information from completed studies in one location to another location. Benefit transfer is generally applied in cases were primary studies cannot be undertaken due to time and financial constraints (Boyle & Bergstrom, 1992). Some authors, for instance, Kerr (2011) use the term value transfer since this covers the transfer of both costs and benefits.

Application of benefit transfer requires a number of criteria to be met for reasonably accurate transfer of values. These have been well documented in the literature for many years. One of the requirements is that primary studies should be based on adequate data, sound economic methods and correct empirical techniques (Freeman, 1984). Some empirical studies have shown that errors in benefit transfer tend to decrease as the number of primary studies used in meta-analysis function transfer increases (for example, Rosenberger & Loomis, 2000).

In addition to the need for high quality primary studies, other widely accepted criteria for valid benefit transfer reported by Desvousges *et al.* (1992) are: the basic commodities to be valued must be essentially equivalent; the baseline and extent of change should be similar; and the affected populations should be similar.

These benefit transfer criteria are also highlighted in the US EPA (2000) guidelines for economic analyses. In New Zealand, Sharp & Kerr (2005) emphasize that careful consideration should be given to matching environments, users and proposed changes as closely as possible. A useful summary is provided by Boyle *et al.* (2009, p. 2) who states “Key issues in establishing the credibility of any benefit transfer include the definition of value, the quality of the original studies, and the need to address differences in environmental quality and consumer characteristics between the original study and new policy applications”.

Generally, two broad pathways for benefit transfer have been developed: value transfer and function transfer. Value transfer involves either the direct transfer of single measures of benefit estimates such as the mean WTP values to a policy site or the use of a range of value estimates from one study or a combination of studies (also called meta-analysis) (Rosenberger & Loomis, 2003). The value transfer approach is advantageous over other methods in that it is easy to apply. However, the approach has been criticized because of its failure to take into account differences in demographic and environmental quality factors at the policy site (Desvouges *et al.*, 1992).

Consequently, the use of function transfer is widely advocated in non-market valuation literature. Function transfer involves the use of statistical models such as WTP functions developed for a particular study site for use at a new but similar site. The statistical model is adjusted to take into account differences in demographic and environmental quality factors at the policy site[77] (Dumas *et al.*, 2005; Houtven *et al.*, 2007).

However, effective implementation of benefit transfer is still under intense scrutiny. In general the use of benefit transfer remains a contested issue due to its potential for large transfer errors which may limit the usefulness of results (Johnston & Rosenberger, 2010).

[77] Function transfer can be categorized into benefit function transfer, and meta-regression analysis. The benefit function transfer expresses the benefit estimate from a single study site as a function of demographic and environmental quality factors pertaining to the policy site. In contrast under meta-regression analysis the value of the benefit estimate at a policy site is found by regressing benefit estimates obtained from several similar studies against the site-specific and demographic factors prevailing at the policy site (Rosenberger & Loomis, 2003)

Appendix 5: Fish and Game Regions

Figure A5.1: Fish and Game Regions

Source: (FGNZ, 2011)

Table A5.1: Angler FGNZ home region and FGNZ licence regional council

	Angler FGNZ Home Regions				
FGNZ Licence Council	Auckland/Waikato	Eastern	Taupo	Taranaki	**Total**
Auckland/Waikato	107	7	-	-	114
Eastern	126	156	3	-	285
Other	10	4	-	1	15
Total	**243**	**167**	**3**	**1**	**414**

Source: Unwin (2009)

Table A5.2: Study sample and location of home region within FGNZ licencing district

Home Region	FGNZ Home District	Number of anglers	Number of anglers as a %
Auckland (Papakura, Pukekohe, Tuakau & Manukau City)	Auckland/Waikato	112	46
Coromandel	Auckland/Waikato	9	4
King Country	Auckland/Waikato	3	1
South Waikato	Auckland/Waikato	15	6
Waikato	Auckland/Waikato	43	18
Western BOP	Auckland/Waikato	61	25
Total		**243**	**59**
Bay of Plenty	Eastern	55	33
Rotorua	Eastern	104	62
Gisborne	Eastern	8	5
Total		**167**	**40**
Other (Wanganui & Taupo)	Taranaki & Taupo	4	1
Grand Total		**414**	**100**

Source: Unwin (2009)

Appendix 6: Summary statistics and welfare estimates from model using the 2006/07 weight of fish data

Table A6. 1: Summary statistics for weight of fish data for the 2006/07 and 2007/08 fishing seasons

	2006/07 Fishing Season				2007/08 Fishing Season			
Lakes	Mean	St.dev	Min	Max	Mean	St.dev	Min	Max
Rotorua	2.08	0.36	1.4	2.5	1.62	0.27	1.3	2.2
Rotoiti	2.19	0.22	1.9	2.7	2.03	0.23	1.4	2.3
Tarawera	1.56	0.13	1.4	2.0	1.66	0.22	1.0	2.0
Okataina	1.88	0.21	1.7	2.4	1.86	0.47	1.4	2.6
Rotoma	1.27	0.29	0.6	1.8	1.47	0.34	0.8	2.3
Okareka	1.68	0.49	1.2	2.9	1.34	0.27	0.7	1.6
Rotoehu	1.28	0.20	1.0	1.6	1.32	0.17	1.1	1.7
Rerewhakaitu	1.83	0.73	1.0	2.7	1.28	0.16	1.0	1.8

Table A6.2: Welfare estimates from the model using the 2006/07 weight of fish data in 2008 NZ$

	Population	Sample	Per angler	Per choice
		1 metre increase in water clarity		
Rotorua, Rotoiti, Okaro, Rotoehu, Rotomahana, Okareka & Rerewhakaaitu	2,193,922.53	41,506.37	100.26 [83.82 115.23]	1.65
		3 metres increase in water clarity		
Rotorua, Rotoiti, Okaro, Rotoehu, Rotomahana, Okareka & Rerewhakaaitu	6,995,017.70	132,337.31	319.66 [263.45 370.67]	5.25
		1 metre increase in water clarity		
Rotoiti	1,207,860.99	22,851.27	55.20 [43.42 68.29]	0.91
Rotorua	786,421.10	14878.14	35.94 [27.91 3.37]	0.59
Rerewhakaaitu	126,735.52	2,397.68	5.79 [3.94 8.79]	0.10
Rotoehu	86,635.55	1.639.04	3.96 [2.33 7.18]	0.07
Okareka	44,244.49	837.05	2.02 [1.43 2.74]	0.03
Rotomahana	19,772.73	374.08	0.90 [0.70 1.17]	0.01
Okaro	161.58	3.06	0.01 [0.00 0.02]	0.0001
		3 metres increase in water clarity		
Rotoiti	4,070,047.33	77,000.39	185.99 [146.42 230.41]	3.05
Rotorua	2,725,580.05	51,564.69	124.55 [93.87 153.47]	2.05

The figures in brackets are the 95% confidence intervals for mean welfare estimates.

Appendix 7: Bivariate correlations and estimated results from models partially accounting for unobserved effects

Table A7.1: Bivariate correlations between regressors

	SD_YR	SD_2M	$FWEIGHT\_YR$	$FWEIGHT\_2M$	$COST$	$LSIZE$	$URBAN$	FDV	$FOREST$	$DEPTH$
SD_YR	1.000									
SD_2M	0.976	1.000								
$FWEIGHT\_YR$	0.294	0.264	1.000							
$FWEIGHT\_2M$	0.183	0.187	0.687	1.000						
$COST$	0.013	0.013	-0.029	-0.026	1.000					
$LSIZE$	0.157	0.134	0.543	0.380	-0.029	1.000				
$URBAN$	-0.230	-0.212	-0.215	-0.021	-0.070	0.191	1.000			
FDV	-0.134	-0.131	0.104	0.028	-0.029	0.573	0.421	1.000		
$FOREST$	0.090	0.096	0.191	0.173	-0.028	0.113	0.067	0.004	1.000	
$DEPTH$	0.405	0.373	0.675	0.471	-0.002	0.429	-0.252	0.261	0.179	1.000

Table A7.2: Estimated results from conditional logit models partially accounting for unobserved effects

	Model 5 (SD_YR & FWEIGHT_YR)			Model 6 (SD_2M & FWEIGHT_2M)		
Variable	*Coefficient*	*Std Error*	*|t-value|*	*Coefficient*	*Std Error*	*|t-value|*
ASC_Rotorua	3.062***	1.010	3.03	-0.888	0.898	0.99
ASC_Rotoiti	8.901***	1.433	6.21	5.306***	0.935	5.68
ASC_Tarawera	2.741***	0.600	4.57	2.544***	0.658	3.86
ASC_Okataina	-	-	-	0.655	0.585	1.12
ASC_Rotoma	-	-	-	2.570***	0.414	6.21
COST	-0.071***	0.007	10.68	-0.0731***	0.007	10.99
SD	0.432***	0.057	7.59	0.088***	0.029	3.05
FWEIGHT	-2.610***	0.638	4.09	0.284***	0.083	3.41
LKSIZE	2.488***	0.311	8.01	2.411***	0.322	7.49
FDV	-0.298***	0.092	3.24	-0.090	0.082	1.10
URBAN	-0.319***	0.090	3.55	-0.017	0.098	0.18
FOREST	-0.0193***	0.005	3.79	-0.020***	0.007	2.87
DEPTH	-0.086***	0.015	5.72	-0.077***	0.013	6.11
			Summary Statistics			
Log-Likelihood	-3823.90			-3813.73		
McFadden R-Squared	0.278			0.280		

***, **, * denotes significance at 1%, 5% and 10% level respectively.

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Article

Do Respondents' Perceptions of the *Status Quo* Matter in Non-Market Valuation with Choice Experiments? An Application to New Zealand Freshwater Streams

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Abstract: Many issues relating to the sustainability of environmental resource use are informed by environmental valuation studies with stated preference surveys. Within these, researchers often provide descriptions of *status quo* conditions which may differ from those perceived by respondents. Ignoring this difference in utility baselines may affect the magnitude of estimated utility changes and hence bias benefit estimates of proposed environmental policies. We investigate this issue using data from a choice experiment on a community's willingness to pay for water quality improvements in streams. More than 60% of respondents perceived streams' water quality at the *status quo* to be better than the description we provided in our scenario. Results show that respondents who could provide details of their perception of the *status quo* displayed stronger preferences for water quality improvements—and hence higher marginal willingness to pay—than their counterparts. However, respondents who referred to their own *status quo* description displayed a higher inclination to prefer the *status quo*, while other respondents tended to prefer the proposed improvements. We argue this might be linked to the amount of knowledge each group displayed about the *status quo*: a kind of reluctance to leave what one believes he/she knows well.

Keywords: choice experiments; fixed *status quo*; people's perceived *status quo*; *status quo* effect; willingness to pay

1. Introduction

Even "clean and green" New Zealand has its share of environmental problems. This is especially true in areas exposed to intensive agricultural production such as the Waikato region which accounts for around 30% of New Zealand's dairy production. Policy makers are torn between supporting the country's leading export industry and ensuring sustainably high environmental quality for the 400,000 people who live in the region. Water pollution from agricultural activities is considered to be one of the most important environmental issues facing New Zealand and is the most frequently mentioned environmental concern for the region's residents [1]. These concerns are well founded since levels of nitrogen and phosphorus in many streams, rivers and lakes have increased over the last two decades leading to a progressive decline in water quality and increased incidence of algal blooms in freshwater bodies [2].

Technical and regulatory mechanisms to reduce this non-point source pollution from agriculture are now the focus of an intensive research effort. Policy makers are showing increasing interest in non-market valuation and the use of market based tools to try and attain environmental improvement. It was in this context that a research program was started in 2008, to assess the potential tradeoffs between cost, water quality improvements and job losses, using choice experiments. It is intended that the findings will inform the policy process by allowing decision makers to consider both the costs and the benefits of different levels of water quality improvement for long term sustainability of the freshwater system in the catchment.

In this paper we describe a choice experiment on a community's willingness to pay for water quality improvements in streams. We investigate the preferences of residents of the Karapiro catchment which stretches over 155,000 hectares of the Waikato region from Lake Arapuni to the Karapiro dam. Land use is predominantly for dairy (34%), pastoral (13%) and forestry (48%) production. The amount of nitrogen and phosphorus reaching waterways in the catchment has generally been increasing and is expected to continue to rise because of intensification and conversion of land from forestry to dairy. Even with widespread adoption of "best management practices" [3] it is expected that the streams and rivers in the catchment will support more algae, water clarity will fall and the water system's ecological health may decline. Levels of *E. coli* may also increase. These changes may endanger the overall environmental sustainability of the current agricultural system.

Discrete choice experiments have gained widespread recognition since their early application by Louviere and Hensher [4] and Louviere and Woodworth [5] and their earliest application to environmental valuation by Boxall *et al.* [6]. Choice analysis is an attribute-based technique in which respondents are presented with different alternatives defined in terms of environmental attributes and cost. They are then asked to select their preferred one. The tradeoffs that they reveal during this exercise between the cost of the proposed options and their environmental attributes are used to derive implicit estimate of monetary value, under a set of well qualified assumptions.

In environmental valuation studies using choice experiments, researchers often need to provide respondents with descriptions of *status quo* conditions. Such descriptions are typically derived from environmental baseline studies and may differ from those perceived by respondents. Such discrepancy may lead to problem in benefit estimation because ignoring differences in utility baselines may affect the magnitude of utility changes and hence bias the implied estimates of benefits from the proposed environmental policies. We investigate this issue, taking the case of respondent perception of the quality of local streams.

In order to study the preferences of respondents with respect to departures from the current environmental conditions, the so-called *status quo* (SQ), analysts often place this as an alternative in all choice sets. However, recent studies have shown that description of the *status quo*, or its mere presence in the choice context is not neutral to the choice outcome [7-12]. Later in this paper we review the literature on current research results involving *status quo* in choice experiments, but we will focus on one area of relatively poor investigation, namely that of identifying the specific effect that respondent's perception of *status quo* conditions has on implied welfare estimates. In particular, respondents may or may not have a clear perception of how the *status quo* conditions they experience relate to the attributes and levels considered in the choice exercise. In short, some respondents may not be able to map into the descriptors of environmental status used by the researcher. In this case, it is necessary for the purpose of the choice exercise to provide respondents with a description of the SQ conditions using the specific metric selected for the experimental design. So, one can distinguish two types of respondents. A first type, whose perceptions of the SQ can be mapped into the choice experiment, and a second group, to whom a mapping needs to be supplied during the course of the interview on the basis of some previous, possibly technical, knowledge. Our contribution to the literature is that of investigating whether the effects of such an asymmetry of treatment systematically results in different welfare estimates from an endogenous split sample design.

We proceed by first reviewing the different formats for the SQ alternative in choice experiments. Hess and Rose [13,14] categorized the SQ alternatives into three formats as follows:

> "Firstly, … the presence of a *status quo* alternative which is represented as a null alternative with the attributes and attribute levels of the alternative not shown as part of the experiment. A second form of these experiments involves respondents being shown alternatives with attribute levels based on their own experiences but not the exact levels as described. A final form of these experiments involves the inclusion of one or more alternatives in the choice task being described with exact levels representing each respondent's recent experiences." (p. 299).

An example of the use of the first format is provided in the study by Campbell *et al.* [15] on rural environmental landscape improvements in the Republic of Ireland, in which the SQ alternative was labeled "No Action" without specifying the attribute levels. In this case it is quite obvious that the respondent is left to her own devices as to what conjecture to make about the SQ. Furthermore, the analyst does not collect any information on such conjecture. In this study we are particularly interested in the second and third formats above. The attributes described to respondents might either represent some average population measure of the good being valued—and as such be described quantitatively to respondents (as in the second case above)—or might be tailored to suit each individual's specific experiences (as in the third case above and Rose *et al.* [16]). The use of the second approach is the most prevalent in the existing literature on environmental valuation, to which our study contributes. Typically, this approach involves the use of the SQ alternative described in terms of the average population measures of the prevailing environmental quality (e.g., [17,18]).

Such average population measures are obtained through a consultative process involving the recording of expert assessments and public opinions, usually through focus groups. Additionally, other information obtained from a literature search may also be incorporated [19]. In as much as the latter approach is the most commonly used in environmental valuation the following issues are worth addressing. First, what if the predicted average levels of environmental quality deviate from the attribute levels perceived by respondents? Second, in the face of a discrepancy between the perceived attribute levels and predicted average attribute levels for the SQ alternative, how will respondents perceive the choice tasks presented to them? Third, what are the implications for the implied welfare measures of using SQ scenarios that directly account for individual specific perceived knowledge of environmental quality?

Exploratory and pioneering work on the differences between perceived and objective attribute measures was published as early as 1997 [20]. The first and second questions above were more recently addressed by Barton *et al*. [21] and Kataria *et al.* [22]. The former analyzed respondents' understanding of water quality in different lakes compared to objective measures. The latter, asked respondents whether they believed in the description provided for the *status quo* and whether they found the overall scenarios presented to them credible. They found that not accounting for respondents' beliefs in the proposed scenarios could lead to biased welfare estimates.

To date, we are aware of only one other study [23] in environmental valuation that has attempted to address the third question presented above. It is against this backdrop that this study endeavors to contribute to the environmental valuation literature by assessing the implications on welfare estimates of using a SQ alternative based upon each respondent's specific perceptions of water quality *vs* the use of a fixed SQ based upon average measures of water quality for the overall population.

We use choice experiment data on streams in the Karapiro Catchment to investigate whether respondents' perceptions agree with our chosen description of the SQ alternative (an average measure of stream quality in the catchment), which we provided to them. Instead of simply asking respondents whether or not they believed in the described SQ scenario—as was the case in a study by Kataria *et al.* [22]—respondents in our study were asked to state their perceived water quality attribute levels at the SQ. Only those respondents who were unable to give their own assessment were given "the average assessment of the current condition of streams in the catchment". Such treatment is labeled henceforth as *SQ provided*. Respondents who were able to assess current water quality used their own SQ in the choice experiments, or *SQ perceived*. We investigate the nature of the SQ effect emanating from the use of these two alternative formats for the SQ alternative and the implications for the implied welfare estimates.

The remainder of the paper is organized as follows. The next section briefly reviews the nature of *status quo* effects in choice experiments. Section 3 covers methods and the empirical model used in this study. An outline of the survey and experimental design are presented in Section 4. Results and discussions are presented in Section 5, and finally, conclusions and implications of the study are presented in Section 6.

2. *Status Quo* Effects in Choice Experiments

Initially the use of SQ alternatives in choice experiments was supported mainly on the basis of making choice tasks more realistic. It was shown that individuals making decisions tend to refer to past experiences. Therefore, relating experimentally designed alternatives to a previously experienced reference point makes stated choice tasks more realistic to respondents and informative to analysts [24,25]. This is consistent with psychological and behavioral theories, for example, prospect theory by Kahneman and Tversky [26] and case-based decision theory by Gilboa *et al.* [27]. In later studies the inclusion of the SQ alternatives in choice experiments was justified on other grounds, including avoidance of forced choices [11,28], improvement in model fit, ensuring unbiased estimates [7] and increase in design efficiency [29].

More recently, studies have shown that the *status quo* description and even its mere presence in the choice context is not neutral to the choice outcome. In particular, it has been found that respondents presented with both SQ and experimentally designed alternatives have a bias towards sticking with the SQ alternatives, generally referred to as the *status quo* bias effect, even though Scarpa *et al.* [12] discuss how SQ effect can be due to either a predilection for the SQ or a reluctance to stick with it, depending on the definition of the attributes of alternatives. This asymmetry in preferences between the SQ alternative and non-experienced alternative is consistent with reference-dependent utility theories [26,30-32]. The main explanations that have been put forward for this SQ effect include loss aversion [33] cognitive misperceptions and regret avoidance [31], protesting [34,35] and choice task complexity [36]. It has also been argued that respondents tend to avoid the cognitive burden associated with evaluating choice task alternatives that have not been experienced [10,11] and that respondents presented with unattractive alternatives are likely to choose the SQ [8].

Similarly, methodologies for accounting for the SQ effect on utility have been developed. The common approach has been to include the alternative specific constant (ASC) to capture the SQ effect on the systematic component of utility. The conditional logit model is usually applied to measure such effects. On the other hand, the SQ effect on the stochastic component of utility which represents the correlation of the error structure between alternatives, is commonly modeled through the nested logit framework; see for example [37,38].

Currently, studies have demonstrated that such specifications are limited in that they fail to simultaneously account for the SQ effect on the systematic component of utility and the variance differences in utilities between experienced SQ and conjectured utility from experimentally designed alternatives. To overcome such limitations, Scarpa *et al.* [12,39] proposed the use of error components (MXL-EC) in which the additional variance of utility of alternatives different from the SQ can be identified. Since their application, numerous other studies have found the MXL-EC to be better suited in capturing the SQ effects than the conditional logit and nested logit frameworks, and even MXL models without error components [13,15,39-43]. Within the MXL-EC framework, the SQ effect on the systematic component of utility can be measured by the ASC, while the effect on the stochastic component of utility can be captured by introducing a common error component shared by the utilities associated with alternatives different from the SQ, which takes account of the correlation patterns and increased error variance due to the conjectural nature of the experimentally designed alternatives.

It has already been argued that when the SQ alternative is included in the utility specification, the utility from experimentally designed alternatives tends to be more correlated amongst these, than with the SQ alternative. This correlation pattern can be attributed to the fact that the utility associated with the SQ alternative is experienced by the respondents while that of experimentally designed alternatives is not and can only be conjectured, giving rise to higher variance. Additionally, the attribute levels pertaining to the SQ alternative are fixed while those of experimentally designed alternatives are variable across choice occasions. This implies that respondents face a higher cognitive burden in evaluating experimentally designed alternatives than the SQ alternative and therefore, extra errors in addition to the usual Gumbel Type I error are expected to be made. These extra errors would induce a common correlation structure across the experimentally designed alternatives and can be captured within the MXL-EC framework through the introduction of a dummy variable [12,15,39,40,42]. For this reason we adopt this modeling approach in our estimation.

3. Methods

We employ a mixed logit specification that combines both the random parameter and error component interpretation, following the approach detailed in Scarpa *et al.* [44]. Train [45] has shown how the mixed logit model can give rise to two different interpretations, the random coefficient and the error component interpretations. The random coefficient interpretation accounts for taste variations over the sampled individuals and has been widely applied in many studies, e.g., [46-48]. On the other hand, the error component interpretation refers to the decomposition of the error term and accounts for different correlations patterns among utilities for different alternatives [45,49-51].

In the case of this study, the choice tasks consisted of two experimentally designed alternatives and the SQ alternative. We therefore define the following utility structure:

$$\begin{cases} U(a) = \tilde{\beta}x_a + \varepsilon + \mu_a \\ U(b) = \tilde{\beta}x_b + \varepsilon + \mu_b \\ U(sq) = \tilde{\beta}x_{sq} + \mu_{sq} \end{cases} \tag{1}$$

where $\tilde{\beta}$ denotes the random preference parameters for different water quality attributes used in this study; β_{sq} is a fixed SQ specific constant which in our case takes a value of 1 for the SQ and 0 for the other alternatives; x is a vector of attributes describing the alternatives as well as selected respondents' characteristics; μ_a, μ_b and μ_{sq} depict the unobserved component of utility and are assumed to be i.i.d. Gumbel-distributed. Instead, the error component ε is distributed $N(0,\sigma^2)$. The σ^2 adds to the Gumbel variance of μ_a and μ_b.

Assuming a balanced panel of discrete choices, with T choices made by each individual *n*, the joint probability of a sequence of *T* choices $\langle y_1, y_2, y_3, \ldots y_T, \rangle$ made by an individual is given by:

$$P(y_1, y_2, \ldots, y_T) = \int_\beta \int_\varepsilon \prod_{t=1}^{T} \frac{\exp(\tilde{\beta}x_{ti} + \varepsilon_i)}{\sum_{j=a,b,sq} \exp(\tilde{\beta}x_{tj} + \varepsilon_j)} \varphi(\varepsilon \mid \sigma^2) f(\beta \mid \theta) d\varepsilon\, d\beta \tag{2}$$

where ε_j is equal to zero when $j = sq$.

Since the integral in Equation (4) has no closed-form, it is approximated in the log-likelihood function by numerical simulation, in our case by using quasi-random Halton draws [48,52]. We first illustrate the methods for the estimation of the random utility model and then the specific tests used to evaluate the difference between simulated distributions from models with different SQ data.

3.1. Model Estimation

The model in Equation (4) for the *SQ provided* and *SQ perceived* treatments was estimated in NLOGIT 4.0 by maximum simulated likelihood using 350 Halton draws [45,53]. The random parameters were assumed to be independent and normally distributed, except for the cost attribute which was assumed to follow a triangular distribution constrained to have the scale parameter equal to the median. Such distribution was used for the cost parameter so as to ensure non-negative willingness to pay values [52]. Attributes with parameters which were repeatedly found to show insignificant standard deviation estimates were eventually specified as non-random. The final estimates are presented in Table 3.

3.2. Testing Differences in the Implied WTP Distributions

We focus on the marginal WTP for the stream water quality attributes. Rather than estimating the individual-specific WTP conditioned on the observed individual choices, we derived estimates of the population mean WTP for each of the non-monetary attributes for the model estimates based on both the *SQ described* and the *SQ perceived* samples. Population moments were simulated in R-Console using 50,000 random draws to obtain WTP distributions for each non-monetary attribute in the two sub-samples, following the approach of Thiene and Scarpa [54]. Non-parametric procedures using the Kolmogorov-Smirnov test were used to test for equality in the WTP distributions between the two treatments. (The Kolmogorov-Smirnov test statistic does not make any assumptions about the underlying distribution of the data and therefore it is appropriate for the simulated WTP distributions for which no closed form exists.) The WTP distributions were found to be highly skewed. Therefore, instead of testing for the differences in the mean WTP between the two treatments, we opted for the differences in median WTP. The differences in the median WTP are graphically described using box plots as outlined by Chambers *et al.* [55].

4. Survey and Experimental Design

The sample households for the survey were residents of the Karapiro catchment from Lake Arapuni to the Karapiro dam including contributing tributaries. Four focus groups were held to derive an understanding of people's views on water quality in the catchment and to identify attributes for inclusion in the choice experiment. These sessions were also used to test early versions of the questionnaire and to discuss the appropriate range of values for the payment variable. Best practice procedures for running the focus groups were developed drawing on Krueger [56] and on more specific New Zealand experience from Bell [57] and Kerr and Swaffield [58].

Focus group discussions highlighted the increasing number of fences on farms restricting livestock access to streams and creeks, and hence livestock pollution. This was recognized as an improvement and many participants thought that stream water quality was improving, especially when streams were protected by fenced areas of bush, which create a natural filter. Focus group participants from different areas had different perceptions of the quality of their local streams. For example, while some streams experienced by participants at the Karapiro focus group were perceived as with poor water quality, participants further upstream at the Waotu group reported high quality streams with trout, the water from which was used as a supply of domestic drinking water.

Questionnaire development and improvement took place over an extended period. Testing started using focus group participants and was followed by a pilot survey using two groups of six participants and a pre-test of 21 questionnaires. The water attributes identified by focus groups participants were supplemented by literature review and discussions with experts in the field. The attributes eventually selected for the final study were:

- Suitability for swimming (percentage of *E. coli* readings that are satisfactory for swimming)
- Ecology (percentage of excellent readings)
- Native, fish and eels (presence of)
- Trout (presence of)
- Water Clarity (Can you usually see the bottom?)

Suitability for swimming and ecological quality were defined by reference to criteria already defined by the Waikato Regional Council whereby water is assessed as being suitable for swimming (or not) and ecological health is assessed as being excellent, satisfactory or not satisfactory. The suitability for swimming attribute aligns with the proposed national policy statement for freshwater management that is designed to ensure that appropriate Freshwater Resources reach or exceed a swimmable standard. This attribute is also intended as a "catch all" that enables respondents to state their preference for water that is safe for all forms of contact recreation (swimming, paddling, fishing, eeling *etc*.).

The ecology attribute aligns with data collected by Waikato Regional Council (WRC) on the ecological health of waterways in the catchment. Based on 100 monitoring sites across the region, WRC reports that ecological health readings for undeveloped catchments range from 23% to 100% excellent, but for developed catchments the percentage of excellent readings is much worse, between 0 and 25%. The Karapiro catchment falls under the lower Waikato catchment zone where 68% of ecological health readings are reported to be unsatisfactory with only 2% excellent. Ecological health and "presence/absence of native fish and eels" vary together and so are both included in a single ecological health attribute, for example poor water quality results in "only small eels being found in most catchment streams" while high water quality leads to "large eels, bullies and smelt being found".

The ecology of rivers and streams in the catchment has been adversely affected by clearance of forests and riverside vegetation, habitat loss and creation of barriers to fish passage (including dams). Aquatic plants and animals have also been affected by reduced water quality, changes to flow regimes, habitat loss (due to drainage and changes in land use) and introduced species that compete with or eat native fish [59].

Native fish populations in the Waikato Region are documented in Joy [60]. These species are highly affected by the Waikato dams which prevent fish migration. The population of eels depends on recruitment (which has been falling steadily in recent years) and the number of elvers transported over the hydro dams. Shortfin eels (Anguilla australis) are very tolerant of poor water quality and may even increase with rising levels of N and P. In poor conditions these eels would mainly be 30 to 40 cms in length. If water quality increases (and sufficient numbers are moved over the hydro dams), then the population of longfin eels (Anguilla dieffenbachia) should increase. This species is far less tolerant of poor water quality and can grow to 2 meters in length. Native bullies and smelt should be migratory but landlocked populations exist in Lake Taupo. Numbers of these species may be expected to increase with better water quality. Respondents were asked for their assessment of the condition of streams in the catchment based on the attributes and levels used for the choice cards. Respondents who indicated that they had 'no idea' of the quality of the streams in the catchment were presented with the *status quo* defined as 'our assessment of the current overall condition of streams in the catchment' (see Table 1).

During the survey, respondents who felt able to make their own assessment of stream quality in terms of the metrics used in the choice experiment scenario descriptions used their perceived quality assessment as the *status quo*. In this case attribute levels were entered onto a transparent overlay and placed on top of each page of choice cards to make it easy for respondents to compare their perceived *status quo* with the alternative levels offered in each choice card.

Attributes, attribute levels and labels used in the survey are defined in Table 1. Choice cards were based on an orthogonal design of 72 choice sets, with each respondent completing six choice tasks.

The initial sample for this study was drawn by intersecting the Land Information New Zealand (LINZ) property title database with the catchment boundary layer in ArcGIS. In this way a list of all 7627 properties in the catchment was produced including physical location, territorial authority and other variables. The population was broken down into three geographical strata to reflect the markedly different socioeconomic characteristics of these areas; namely Tokoroa, Putaruru/Tirau and the remaining rural areas. Address lists were drawn up for each stratum and a pseudo-random number generator was used to draw up lists of addresses to be visited by each enumerator. Field work proved to be very time consuming with each enumerator only able to complete three to six surveys each day. Field work was carried out both during the day and at weekends to try to avoid bias towards people staying at home. In the later stages of the survey a quota system was used to try and reduce bias towards people over 60.

Comparison of socioeconomic and attitudinal characteristics for our sample, with data for the Waikato Region as a whole (Table 2) enables some conclusions to be drawn. Men appear to be under represented at 62%. This may be due to the fact that more males than females were at home during the time of the survey or in cases where a couple was at home then the male was more likely to participate. Differences between the sub-samples are also observed particularly in levels of education and income; for example 49% of the respondents in the perceived category achieved at least a diploma or a certificate compared to only 23% in the provided group. Similarly, 65% of respondents in the perceived category earn at least $50,000 compared to 39% in the provided category. Given random sampling, the differences in representation are mainly attributed to differences in propensity to take part in the survey, for example refusal rates were higher in lower socio-economic status urban areas and lower in rural areas.

Table 1. Attribute levels and labels.

Attribute	**Current Situation**	**Improvement Levels**			**Labels**	
Suitability for Swimming *(% of readings rated as satisfactory for swimming)*					ASC	fixed SQ specific constant which is equal to 1 for the SQ and 0 for the other alternatives
	30%	50%	70%	90%		
Variables		SWIM50	SWIM70	SWIM90		
Ecology *(% of readings rated as excellent)*					σ_{ε}	error component capturing the extra variance associated with the experimentally designed alternatives.
	<40% Only small eels	40–70% Small eels, bullies and smelt	>70% Large eels, bullies and smelt			
Variables		ECOM	ECOH		Per	denotes attributes pertaining to the *SQ—perceived models*
Trout	*No* Trout	Trout *are* found (TROUT)			Pro	denotes attributes pertaining to the *SQ—provided models*
Water Clarity	Usually you *cannot* see the bottom	Usually you *can* see the bottom (CLARITY)				
Cost to Household *$ per year for the next 10 years* (COST)						
	$0	$50, $100, $200				

Table 2. Socio-demographic data for the sample and region.

	Provided	Perceived	Sample	Region
Gender (%)				
Males	60	62	62	49
Females	40	38	38	51
Age (%)				
Under 30	11	16	14	18
30–44	21	20	20	30
45–59	27	29	29	28
60+	40	34	37	25
Education (%)				
Any post secondary qual.	44	49	47	
Vocational/trades	19	21	16	
Diploma or certificate (>1 year)	19	37	24	
Bachelors degree	3	8	5	
Higher degree	1	4	2	
Income (%)				
<$30,000	44	14	30	53
$30 to $50,000	18	21	19	21
$50 to $70,000	10	19	16	9
$70 to $100, 000	12	20	13	4
>$100,000	10	15	11	3
Not revealed by respondent	7	11	11	11
Work on or own a farm (%)			25	
Location (%)				
Town	63	52	57	
Settlement	19	10	13	
Rural	4	16	11	
Farm	14	22	19	
Sample Size	*73*	*103*	*178*	

5. Results and Discussion

Respondents in the *SQ perceived* subsample generally registered higher incomes and better education levels than their counterparts in the *SQ provided* subsample. So, we proceeded by comparing the two sub-samples before and after controlling for outliers in income and qualification. In Table 3 we report the models for these comparisons. Models 1 and 3 include all respondents and pertain to the subsamples *SQ provided* and SQ perceived, respectively. Models 2 and 4 are based on subsamples in which respondents with income levels of over NZ$50,000 and those with any tertiary qualification in education were excluded. We excluded these to try and ensure that differences in the estimated results can be attributed to differences in the SQ treatment alone, rather than to the effect of outliers in socio-economic covariates in one of the two sub-samples.

Table 3. Estimation results.

	Model 1 **SQ-Provided All Respondents**		***Model 2*** **SQ-Provided High Income & Qualification excluded**		***Model 3*** **SQ-Perceived All Respondents**		***Model 4*** **SQ-Perceived High Income & Qualification excluded**	
	Coefficient	*|t-value|*	*Coefficient*	*|t-value|*	*Coefficient*	*|t-value|*	*Coefficient*	*|t-value|*
Variable								
ASC	-2.293^{f}	5.04	-2.143^{f}	3.79	0.792^{f}	2.19	0.550^{f}	1.45
SWIM50	0.344^{r}	1.34	0.504^{f}	1.74	0.601^{f}	3.18	0.792^{f}	3.04
SWIM70	1.130^{f}	4.45	1.020^{f}	3.28	0.954^{f}	4.65	1.103^{f}	3.99
SWIM90	1.641^{r}	5.07	1.510^{f}	4.25	1.281^{r}	5.17	1.765^{r}	4.71
ECOM	0.301^{f}	1.47	0.131^{r}	0.53	0.829^{f}	4.83	0.954^{f}	3.98
ECOH	0.602^{r}	2.27	0.687^{r}	2.21	1.187^{r}	5.59	1.438^{r}	4.77
TROUT	0.711^{f}	3.84	0.636^{f}	2.91	1.014^{r}	5.12	0.834^{r}	3.18
CLARITY	0.507^{f}	2.65	0.532^{f}	2.35	0.820^{r}	5.14	0.835^{f}	4.06
COST	-0.035^{r}	5.04	-0.041^{r}	6.75	-0.017^{r}	8.59	-0.023^{r}	6.04
Error Component σ_{ε}	2.692	6.91	2.487	5.93	3.341	7.22	2.181	5.86
Summary Statistics								
Log L		−513.6		−342.7		−742.2		−387.3
AIC		1.202		1.206		1.223		1.213
BIC		1.273		1.296		1.282		1.301
R^2 (McFadden)		0.466		0.469		0.453		0.466
N (Observations)		876		588		1236		660

Note: ^{f} and ^{r} denote whether the attributes were estimated as fixed or random variables.

5.1. Models from SQ Provided Sample

Models 1 and 2 refer to respondents who lacked information on the SQ conditions and were informed that the SQ is currently assessed as having poor suitability for swimming and poor ecological health. These models show estimates of utility weights with the expected signs for all attributes. The alternative specific constant (ASC) is negative and highly significant at the 1% level in both models implying, preference for a change from the *status quo*. In a study by Scarpa *et al.* [12] on customer preference for water service provision, a negative ASC was attributed to dissatisfaction with the current provision of the good being valued. While this might be one of the possible explanations for the negative ASC in the *SQ provided* models, this inclination towards change might be further attributed to lack of familiarity with the SQ by this group of respondents. Since they were less familiar with the SQ, the perceived loss of leaving it might have been lower than if they were more familiar with it. This explanation is also consistent with the loss aversion hypothesis by Kahneman and Tversky [26] and it also minimizes regret [61].

In terms of the preferences for water quality attributes, the results reveal that respondents have very strong preferences for water quality that is (a) highly suitable for swimming (SWIM70, SWIM90); and (b) where TROUT is found. Both models indicate lower preferences for the ecology attributes with ECOH being significant at 5% level while ECOM is not statistically significant. The COST attribute is negative and highly significant in both models, in accordance with expectations.

The error variance in both models is highly significant indicating that the inclusion of the SQ alternative had a significant effect on the stochastic component of the utility structure of the experimentally designed alternatives. The total variance associated with the unobserved component of utility pertaining to experimentally designed alternatives for Model 1 is given by $2.692^2 + \pi^2/6 \approx 8.89$; where $\pi^2/6 \approx 1.645$ is the Gumbel error variance. For Model 2, the total variance for experimentally designed utilities is equal to $2.487^2 + \pi^2/6 \approx 7.83$, which is slightly lower than that of Model 1. The total variance of indirect utilities associated with experimentally designed alternatives is much larger than what Gumbel error accommodates for both models. This is in line with the findings of the proponents of this approach [40,44].

5.2. SQ Perceived Models

Models 3 and 4 refer to respondents who felt able to make their own assessment of the *status quo* and to describe them using the required metric. On average these respondents considered the condition of streams to be better than the assessment we provided to those who 'had no idea' of these conditions. Comparison of Models 3 and 4 shows that all water quality attributes are highly significant at the 1% level demonstrating that respondents had very strong preferences for all the water quality attributes. The only difference is observed for CLARITY which is heterogeneous across respondents in Model 3 but fixed in Model 4.

The ASC is positive and significant at the 5% level in Model 3, but positive and insignificant in Model 4. The positive ASC reveals that respondents in this category are inclined to remain with the *status quo*. Since the SQ alternative in this model was dependent upon each individual specific experiences the bias towards the *status quo* might be taken as a confirmation of the loss aversion hypothesis by Kahneman and Tversky [26]. It should also be noted that since these respondents provided their own *status quo*, this will in some cases have been perceived to be better than the alternative options provided. However, other explanations cannot be ruled out, such as avoidance of cognitive burden associated with the evaluation of the experimentally designed alternatives as championed by Samuelson and Zeckhauser [31] and others.

The total variance associated with the unobserved component of utility pertaining to experimentally designed alternatives in Model 3 is approximately equal to $3.341^2 + \pi^2/6 \approx$ original 12.81, which is almost twice as high as the variance in the Model 4 given by $2.181^2 + \pi_2/6 \approx 6.40$. These results demonstrate that the inclusion of the SQ alternative had a significant effect on the stochastic component of the utility structure of the experimentally designed alternatives, consistent with findings from the *SQ provided* models. In addition, these results demonstrate that respondents with higher income and qualification levels in the *SQ perceived* treatment seem to have had relatively high valuation errors as indicated by the higher variance in Model 3 compared to that in Model 4, where such respondents were removed.

Further comparison is made between the respondent's willingness to pay (WTP) for water quality improvements in the two treatments. The simulated population mean and median WTP values for the different attributes are presented in Table 4 below, as derived from the estimated random parameter models.

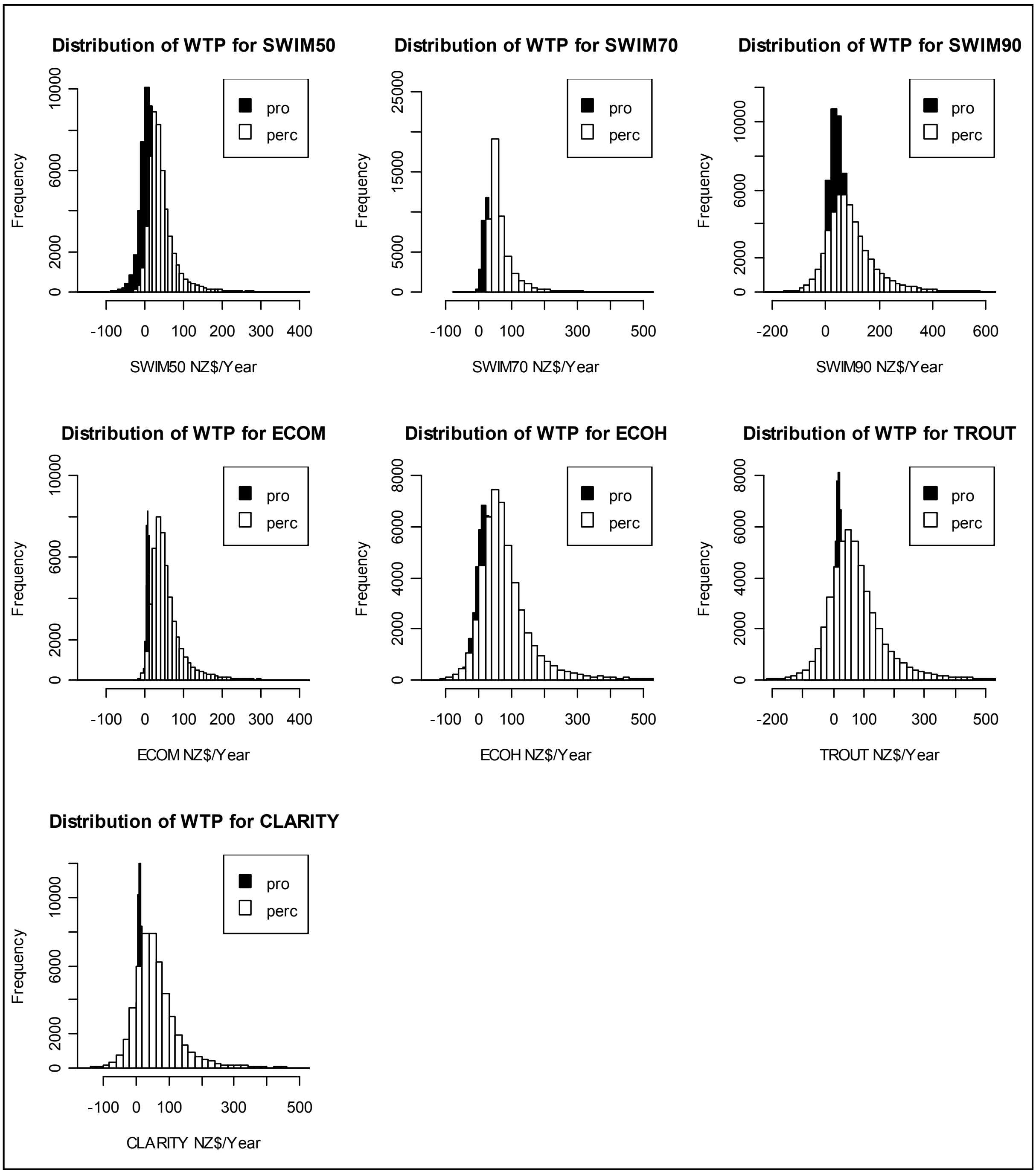
Table 4. Mean and median marginal willingness to pay (WTP) estimates in NZ$/Year.

	Model 1		***Model 3***		***d-stat'***	***Model 2***		***Model 4***		***d-stat'***
	SQ-Provided		**SQ-Perceived**			**SQ-Provided**		**SQ-Perceived**		
	All Respondents		**All Respondents**			**High Income & Qualification Excluded**				
Attribute	*Mean*	*Median*	*Mean*	*Median*		*Mean*	*Median*	*Mean*	*Median*	
SWIM50	13.4	9.56	48.4	34.82	0.455	17.63	12.64	48.28	34.7	0.524
SWIM70	42.59	30.72	77.65	55.86	0.505	32.01	22.99	67.21	48.34	0.447
SWIM90	67.19	48.05	109.05	78.67	0.249	51.97	37.24	92.89	66.765	0.281
ECOM	11.74	8.47	64.41	46.33	0.780	4.92	3.52	63.98	46.15	0.941
ECOH	30.29	21.71	91.01	65.61	0.408	23.83	17.07	83.85	60.28	0.529
TROUT	27.69	19.95	85.46	61.79	0.475	19.91	14.26	51.39	36.93	0.398
CLARITY	19.75	14.15	69.3	49.99	0.526	16.52	11.84	45.99	33.16	0.745

All *d*-statistics have significance at *p*-value < 0.001.

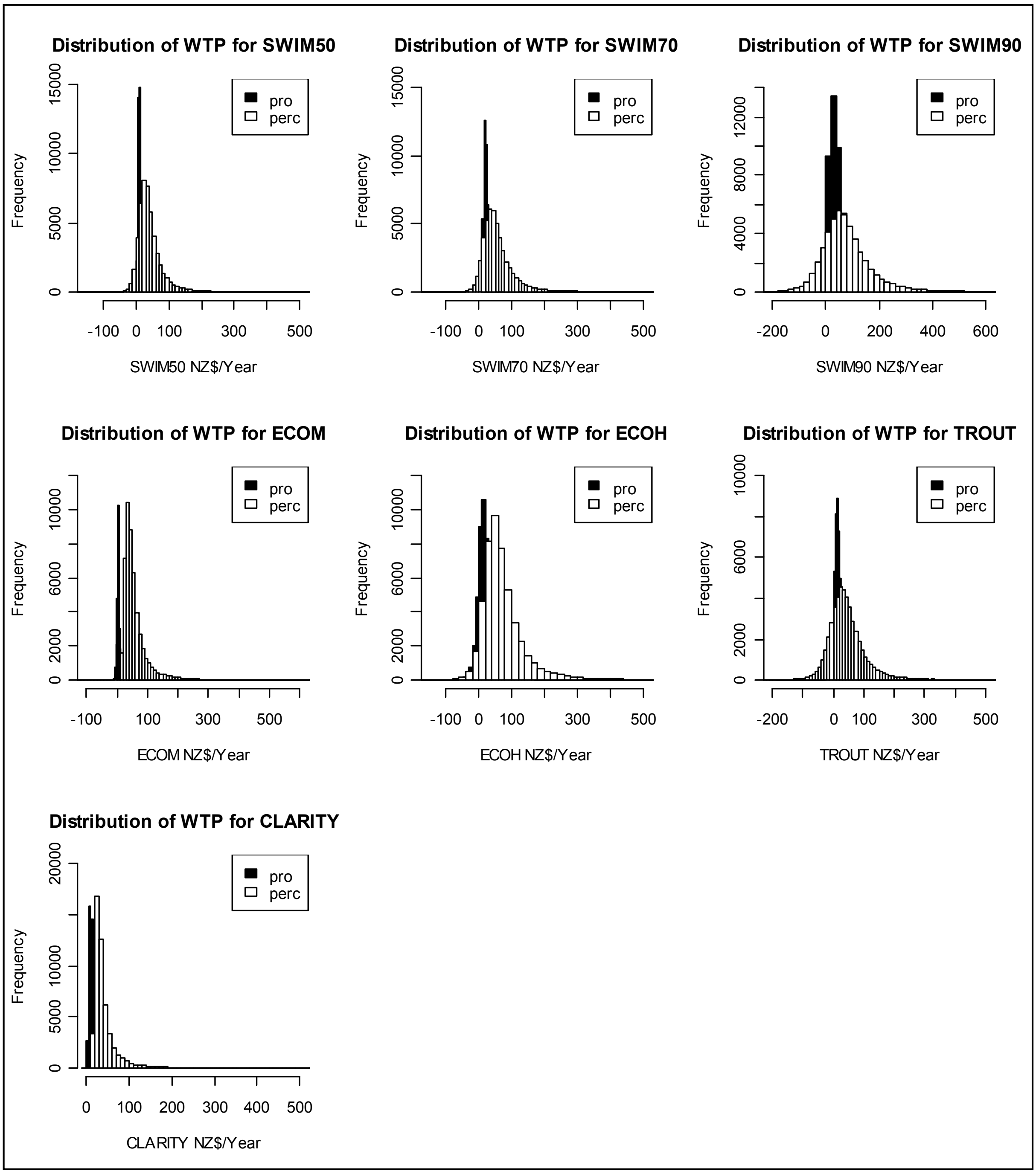
Comparing the mean and median WTP in Models 1 and 3 there is a clear indication that respondents in the *SQ perceived* model are more willing to pay for water quality improvements than those in the *SQ provided* model for all attributes. A similar trend is observed in Models 2 and 4 in which respondents with high income and qualification levels were excluded from the analysis. The median WTP values are less than the mean WTP values in both treatments for all attributes indicating that the distributions are highly skewed upwards. In general the differences in WTP values between the two treatments appear to be quite substantial. A graphical comparison of the distributions of WTP values across the two SQ treatments based on models estimated on all respondents (Model 1 and 3) are presented in Figure 1.

Figure 1. Histograms showing distribution of marginal WTP for models 1 and 3.



The distributions are highly skewed with long and fat tails towards the upper end of the scale. Further, analysis of the histograms highlights that although the distributions of the WTP for all attributes overlap, the WTP for most respondents in the *SQ provided* model is relatively lower than their counterpart. The Kolmogorov-Smirnov test (*d*-statistic) in Table 4 reveals that there are significant differences in WTP distributions for all attributes in the two treatments. Likewise, the simulated distributions of WTP for Model 2 and 4 are compared and presented in Figure 2 below:

Figure 2. Histograms showing distribution of marginal WTP for models 2 and 4.



Once more, the distributions are highly skewed with relatively fat tails towards the upper end of the scale, with the simulated population distribution of WTP from the *SQ provided* model being relatively lower than that from the *SQ perceived* model. The Kolmogorov-Smirnov test (*d*-statistic) again reveals that there are significant differences in the distributions of WTP values from the two subsamples (Table 4).

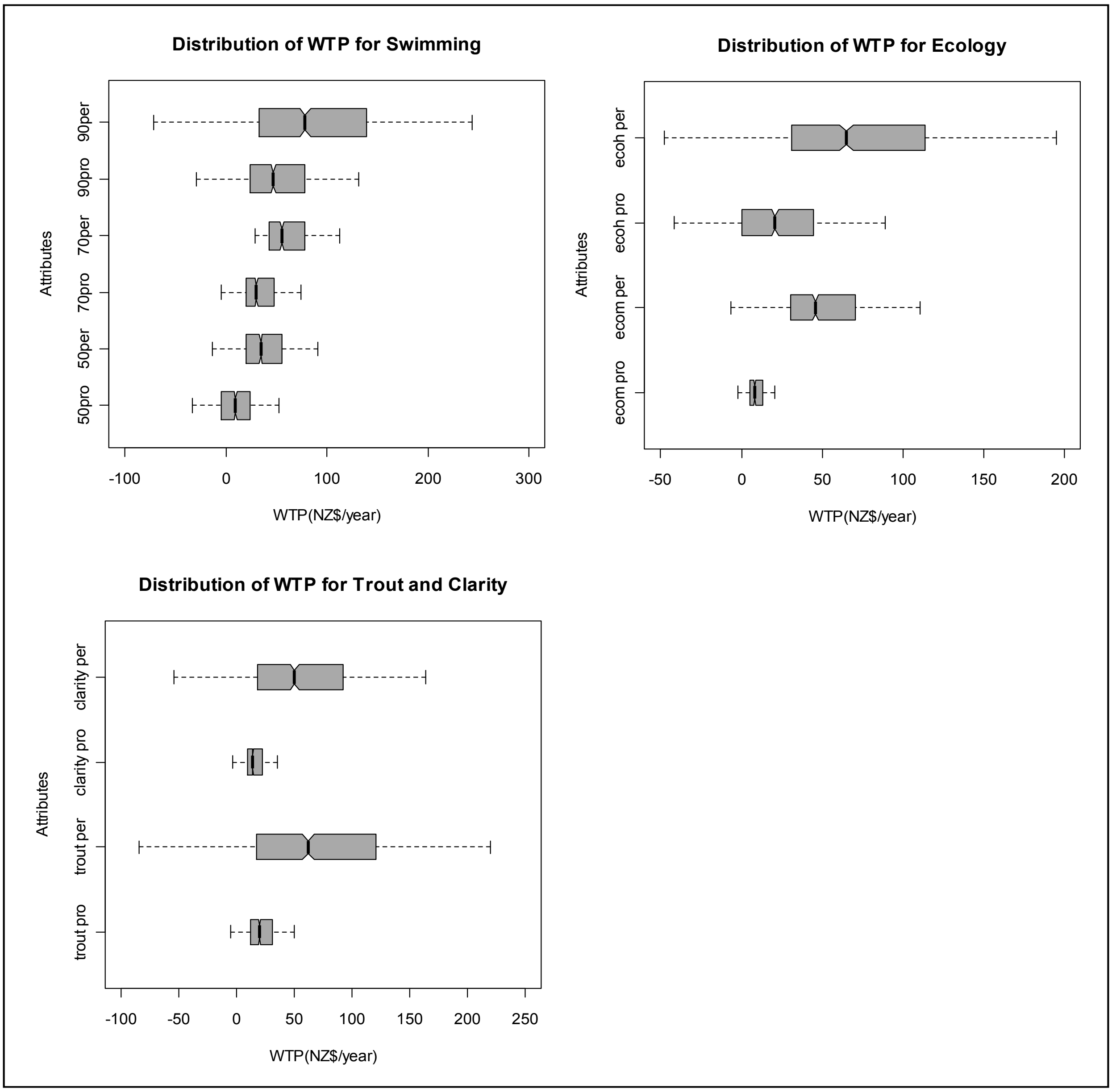
Our results suggest that the distributions of WTP values between the two treatments are significantly different. Poe *et al.* [62] states that:

"Differences in estimated WTP distributions do not necessarily imply that the means derived from these distributions are different. For instance, it is possible that two significantly different distributions can cross and have identical means."

To graphically explore the differences in the simulated measures of central tendency between the two treatments, the quartiles of the distributions of WTP are compared using box plots see [63] and reported in Figures 3 and 4. The box plots display the upper and the lower limits of the cumulative distributions, and the inter-quartile range showing the first quartile, the median and the third quartile. Given that the distributions of WTP are highly skewed, the median is used as a basis of comparison as opposed to the mean, since the latter can be influenced by extreme values.

Figure 3 shows the box plots for Models 1 and 3 with all respondents included in the analysis.

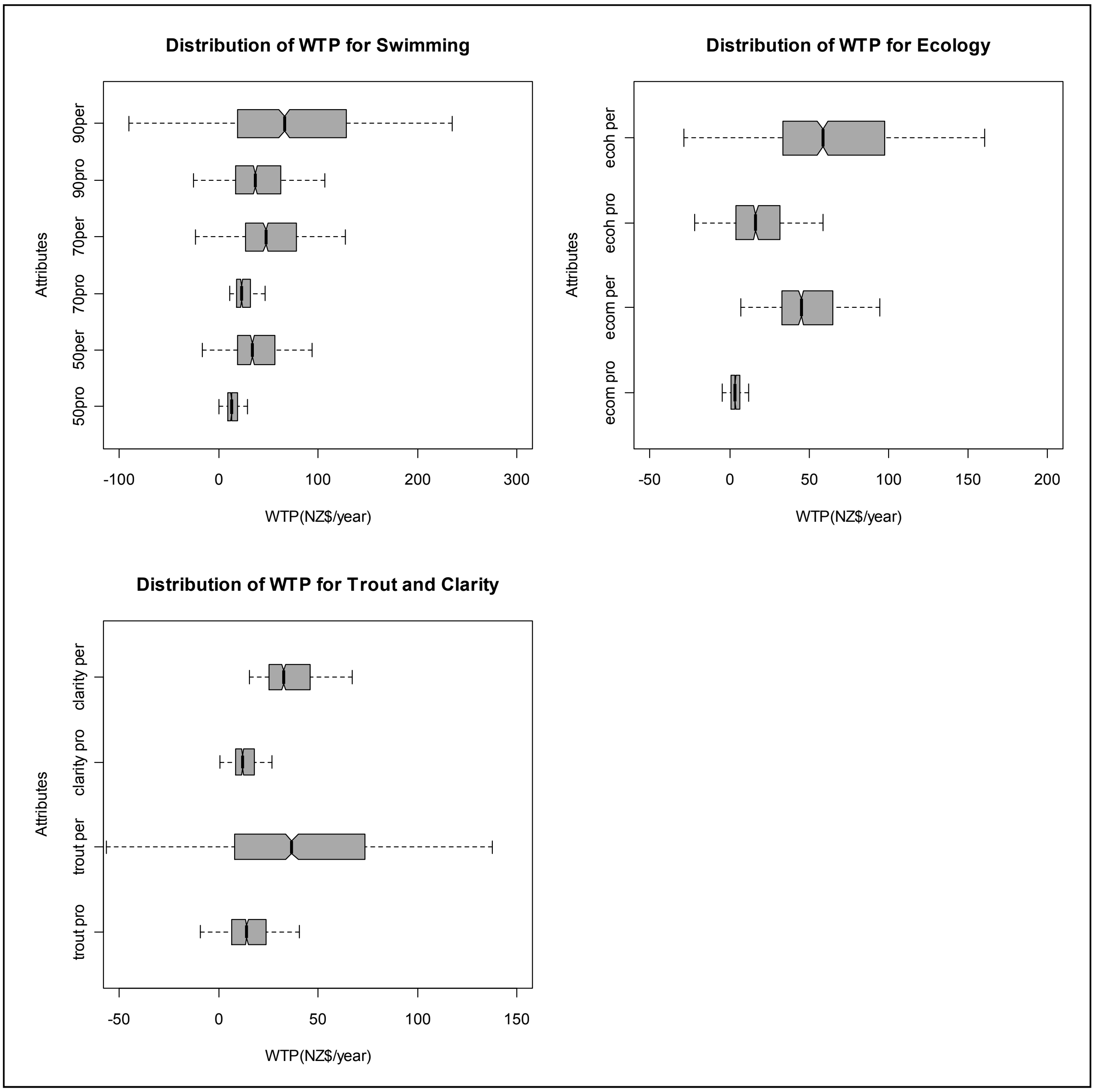
Figure 3. Box plots for distributions of marginal WTP for models 1 and 3.



The quartile distributions are consistent with the previous results, with respondents in the *SQ perceived* model generally showing higher WTP for all attributes than those in the *SQ provided* model. Specifically, the notches in the box plots signify the 95% confidence interval for the median. According to Chambers *et al.* [55], if the notches do not overlap, the null hypothesis of equal medians is rejected.

A similar comparison between the median WTP values for Models 2 and 4 in which respondents with high income and qualification levels were excluded from the analysis is presented in Figure 4 below:

Figure 4. Box plots for distributions of marginal WTP for models 2 and 4.



Inspection of the box plots demonstrate that the notches do not overlap for all stream water quality attributes and therefore, the hypothesis of equal medians is rejected. This test is a further confirmation that respondents in the *SQ perceived* models display stronger preferences as implied by higher WTP values than those in the *SQ provided* models. The results further highlight that there is more variance in the WTP values in the *SQ perceived* models especially for SWIM90 (90 % of readings satisfactory for swimming), ECOH (excellent ecological health) and presence of trout, than in the *SQ provided* models.

6. Conclusions and Implications of the Study

The broader purpose of this research was to assess a community's preferences for stream water quality improvements. A specific focus in this paper was placed on the effect of accounting for perceived *vs* described *status quo* levels. The study revealed that about 58% of respondents had their own perceived baseline condition of water quality and that they could map it into the framework of attributes and levels proposed in the survey. On the other hand 41% of respondents were provided a SQ description by researchers because these respondents either had little or no prior knowledge of the prevailing conditions of water quality in streams or they had this knowledge but could not map it into the proposed framework. We believe that such a dichotomy is common in many nonmarket valuation studies, and hence its consequences for policy prescription via value estimation are worth exploring.

The results of our investigation show marked differences in the marginal value that these two groups of respondents place on water quality improvements and this has implications for their willingness to pay values. The respondents who were provided with *status quo* descriptions expressed strong preference for water that is suitable for swimming, has good clarity and where trout can be found. Yet, this group displayed a reluctance to stay with the SQ scenario. We argued that this might be the case because of their comparative ignorance of baseline water quality conditions. The second group of respondents, who adopted their own perceived SQ scenario, expressed significantly *stronger* preference for improvements across all the attributes subject of this study, but this tendency was attenuated by a general reluctance to embrace policy options implying changes from the SQ, about which they had quite good knowledge. For this group, estimates of marginal willingness to pay values are higher across the entire distribution than for respondents to whom the SQ information was *provided*.

Economic theory suggests that marginal WTP should be proportional to the expected improvement and this in turn depends on individual perceptions in one group and the provided description in the other. In our individual perception data we observe that on average perceived quality of the SQ conditions was higher than the one that was provided. This might be the cause for the observed reluctance to abandon the SQ, as manifested by a positive and significant alternative specific constant for the SQ alternative. In principle for this group the expected improvement would be perceived as smaller, and so would the associated marginal WTP when compared to that held by the *SQ provided* group. However, this holds only for quality changes within evaluations by the same respondent. Unfortunately this cannot be tested here because of the lack of a counterfactual.

The present study demonstrates the effects of using a coding specification of the *status quo* directly built on respondents' perceptions. Our results are supportive of the findings by Kataria *et al.* [22] which showed that failure to take account of respondents' beliefs leads to biased welfare estimates and earlier similar findings by Adamowicz *et al.* [20] in the context of integrating revealed preference data, in which the *status quo* was based on respondent's subjective perceptions, and stated preferences, where it was objectively described to them.

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